SEVENTH FRAMEWORK PROGRAMME

Monitoring and Evaluation of Spatially Managed Areas

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Review Document on the Management of Marine Areas with particular regard on Concepts, Objectives, Frameworks and Tools to Implement, Monitor, and Evaluate Spatially Managed Areas

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EXECUTIVE SUMMARY

The main objectives if this document were to review the existing information on spatial management of marine areas, identifying the relevant policy objectives, to identify parameters linked to the success or failure of the various Spatially Managed marine Areas (SMAs) regimes, to report on methods and tools used in monitoring and evaluation of the state of SMAs, and to identify gaps and weaknesses in the existing frameworks in relation to the implementation, monitoring, evaluation and management of SMAs. The document is naturally divided in two sections: **Section 1** reviews the concepts, objectives, drivers, policy and management framework, and extraneous factors related to the design, implementation and evaluation of SMAs**; Section 2** reviews the tools and methods to monitor and evaluate seabed habitats and marine populations.

Chapter 1.1 summarizes existing knowledge about the process of marine spatial management ranging from the conceptual and theoretical to the operational and practical. At the conceptual level marine spatial management and adaptive management are reviewed. At the practical level, legislation, operational objectives and performance measures in operation in Europe are summarized. The collated information forms the basis for the generic framework for monitoring and evaluation of spatially managed areas to be developed in WP2.

The general concept and the key principles of an ecosystem based management (EBM) are described in detail. Although EBM is fully accepted as an approach that recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species, or ecosystem services in isolation its implementation is still scarce. Marine spatial planning (MSP) and Ocean Zoning (OZ) are tools that can support EBM. MSP is an integrated planning framework to manage human activities in space and time to deliver on defined planning objectives and OZ is a set of regulatory measures used to implement marine spatial plans. While the concept of MSP is still open for interpretation a number of successful implementations have been reviewed here. In theory the basic components of monitoring, evaluation and adaptation are necessary to ensure that marine management measures are both effective and efficient. Implementation of adaptive management still faces a number of challenges such as policy constraints, difficulties in monitoring populations and their responses to interventions, lags in system responses, limits in the ability to depict all possible states of nature and institutional fragmentation.

Marine spatial management is driven by high level goals that management aims to achieve through the implementation of measures. High-level goals and objectives need to be translated into more operational objectives before specific targets, limits and measures can be elaborated. A hierarchical approach was used to structure the review of high level goals and objectives. The first level describes international goals and objectives derived e.g. from United Nations Convention on the Law of the Sea (UNCLOS), Convention on Biological Diversity (CBD) or the United Nations Agenda 21. At the next level EU goals and objectives linked to the Marine Strategy Framework Directive (MSFD), Water Framework Directive (WFD), Habitats Directive (HD), Birds Directive (BD), Common Fisheries Policy (CFP), and Strategic Environmental Assessment Directive (SEA) are summarized. As an example for the national level UK, Greek and Italian goals and objectives are described in detail. Finally an outline is provided for how generic performance measures will be described within the MESMA framework and how they will be linked to the operational objectives from legislation.

Marine protected areas (MPAs) are a well-established tool to manage impacts on marine values (**Chapter 1.2**). They have been applied as a cornerstone in conservation of marine biodiversity, management of fish populations, development of coastal tourism etc. As a result of this broad range of potential applications, each MPA or network of MPAs must have its own specific and clearly defined goals and objectives. Monitoring systems should be tailored toward these goals and objectives as well as the biology of focal species and habitats. Solid monitoring frameworks are the foundation of adaptive management, as they provide the necessary information to evaluate MPA performance and the effectiveness of management actions.

The design and management of MPAs are to a very large extent dependent on various drivers that ultimately determine the goals and objectives of each MPA. For instance, in the case of biodiversity/species management the optimal size of a MPA is highly dependent on the objectives, the life history, mobility and dispersal patterns of focal species and, ultimately, political and societal compromises. In some cases a network of ecologically coherent MPAs may prove more effective or politically feasible. In all cases proper, informed selection of MPA sites is crucial factor. Effects of MPAs include increases in biomass and size of species as well as spillover and recruitment effects. Spillover effects (export of juvenile and adult individuals from MPAs) are measurable and have been documented for fish species in marine reserves in e.g. the Mediterranean. Recruitment effects (export of eggs and larvae) are difficult to measure and are usually based on predictions based on modelling, e.g. sandeel in the North Sea.

Most coastal nations have environmental legislation in place to establish protected areas in their marine territory. In Europe many of these have been small and largely coastal. Within the EU, however, establishment of truly marine protected areas (incl. offshore MPAs) is currently being driven forward by the Habitats Directive, which prescribes the designation of Special Areas of Conservation. Measures implemented through the European Common Fisheries Policy aimed at reducing access and fishing effort, as well as restrictions on catch and gear to protect e.g. juvenile fish species are effectively applications of the same principles which underpin MPAs from a fisheries management perspective. The EU Marine Strategy Framework Directive will be an important instrument to ensure integration and coherence across sectors in MPA developments within efforts to achieve good environmental status in EU waters.

The effectiveness of a MPA is directly linked to how human activities are being managed, how different interests and priorities are being balanced against each other, and how conflicts are being addressed through various governance approaches. Ensuring that MPAs can be managed in an effective and equitable manner often requires the strengths of top-down and bottom-up appraoches to be combined through the use of different incentives, i.e. economic, interpretative, knowledge, legal and participative, which makes MPA governance frameworks more resillient to the perturbing effects of driving forces.

There are similarities and differences between MPAs and other types of MSP, and the place and importance of MPAs vary in a broader scale MSP system, depending on the primary objective of the latter. However MPAs will continue to be a key component in MSP policies and practices underpinned by an ecosystem approach. MPAs can also offer valuable lessons for implementing MSP to address both ecological and socio-economic needs, as well as serving as testing grounds for the application of innovative scientific and governance tools and approaches in implementing MSP.

There is little doubt that society places value in the preservation and maintenance of the marine environment. However the nature and distribution of these values is a complex matter. The marine environment is a source of valuable resources all of which command prices on markets. As well as providing exploitable resources the marine environment is important to the functioning of the economy. It is a highway for international trade, a playground for tourists and it assimilates pollution from the production cycle. The oceans also provide life support systems upon which humanity relies: the sea controls the earth’s climate and it plays an important role in nutrient cycling and helps maintain the atmosphere. Society also places value on the very presence of marine biodiversity, this may be a part of long held cultural traditions or the more modern ‘culture’ of conservation. Increasingly environmental legislation is used to protect the marine environment; protected status reflects the value society places in the environment. Some people question the effectiveness of environmental legislation suggesting that we under-value the environment. The media often focuses on this concern highlighting pollution and the “destructive” effects of human exploitation. This concern for the marine environment is itself just another expression of value.

It is clear that the pattern of values in the marine environment is complex and that these values may conflict with each other. The active management of marine resources (including the designation of MPA’s) will cause a shift in the distribution of these values, for example by placing restrictions on fishing activity to enhance conservation values. There is a generally presumption that good environmental management decisions involve producing a net gain i.e. the benefits of the decision outweigh the costs. The purpose of **Chapter 1.3** is to explore methodologies for the assessment of values in the marine environment and how these methodologies may be applied in the context of MPA’s.

There appears to be consensus on the fact that global climate change is underway and that this change is having, and will have, more and more of an influence on the structure and functioning of marine and terrestrial ecosystems, human activities and public health, with important repercussions also for the social structure of human populations and for the economy. The increase in air and sea temperatures in different parts of the world has been well documented. The estimate, even with some discrepancies, has put the increase in temperature of coastal waters at 1°C over the last 30 years. The effects of climate change have to be incorporated in the spatial management of marine areas (**Chapter 1.4**).

The increase in temperature and the greater frequency of temperature anomalies may involve a series of interrelated problems such as the rise in sea level, the change in the chemical composition of sea water and of the atmosphere, the changes in rainfall with consequent modification of the hydrological cycles, the increase in the risk of coastal erosion and saltwater intrusion, the invasion of alien species, and the outbreaks of indigenous species. Governments have developed strategies to taking into account climate change in several aspects of interest for human life as well SMAs (legal activities, specific plans and programs, research, monitoring, adaptation measures, international cooperation, public awareness).

Decisions on how best to adapt to climate change must be based on solid scientific and economic analysis. It is therefore important to increase the understanding of climate change and the impacts it will have. There is a need to create, in Europe and elsewhere, a mechanism where information on climate change risks, impacts and best practices would be exchanged between governments, agencies, and organisations working on adaptation policies.

Chapter 1.5 reviews the existing information in order to assess geohazard risk and to incorporate it in the integrated tools of the spatial management of marine areas. The term geohazard is used to define the geological state that represents or has the potential to develop further into a situation leading to damage or uncontrolled risk. Geohazards can be relatively small features, but they can also be of huge dimensions and affect local and regional socio-economy to a large extent. Some examples of geohazards are: submarine landslides, naturally occurring gas hydrates, mud flows, mud volcanoes, earthquakes, volcanoes and the associated tsunamis. This document focuses on the tsunamigenic potential and its impact to the Europe's coastal area. The natural forces for the tsunamigenesis are earthquakes, volcanic activities and submarine landslides As the tsunami waves strike the coastline they often cause widespread flooding across low-lying coastal areas and in many cases cause heavily destruction of property and occasionally loss of life. The sensitive areas along the coastal areas of Europe under inundation risk were highlighted and it was suggested to incorporate the geohazard risks in the SMA plans, using the principles of the Integrated Coastal Zone Management (ICZM).

Habitat mapping (**Chapter 2.1**) is essential for the management of marine areas. In the context of marine habitat classification and mapping, 'habitat' can be defined as "*a recognizable space which can be distinguished by its abiotic characteristics and associated biological assemblage, operating at particular spatial and temporal scales*". The relation between scale in habitat mapping and management needs was investigated because the way in which a variable is sampled will affect the scale at which it can be meaningfully displayed or classified, and thus it is important to match how habitats are sampled with the overall scale of the project. National and international mapping programs, mapping technologies, and methods to predict the distribution of biotopes were reviewed.

The marine environment is confronted with an increased development of anthropogenic activities. In order to reveal pressures or impact of these activities on the ecosystem, the quality status must be determined. Environmental indicators represent a powerful tool to evaluate trends in the state of the environment, help the identification of priority policy needs and the formulation of policy measures, monitor the progress made by policy measures in achieving environmental goals, and make the general public aware of environmental issues. In **Chapter 2.2** a critical review of commonly used environmental indicators is provided including both abiotic and biotic indicators with regard to the applicability of six important benthic ecosystem components (macrophytes, meiofauna, epibenthos, macrofauna, hyperbenthos, and fish). To satisfy future needs it is recommended that extra research effort should be put forward to (1) establish a clear format for a good indicator in order to reduce the bewildering array of available indicators, by identifying those indicator approaches, components and formulations that are most widely successful;

(2) establish minimum criteria for indicator validation processes, which can demonstrate the accuracy and reliability of the indicator; (3) compare and intercalibrate methods to achieve uniform assessment scales across geographical boundaries and habitats; (4) integrate indicators across different ecosystem components.

In **Chapter 2.3.1** the sampling techniques most used in today's marine research are introduced. The methods are divided according to their use of sampling space, whether samples are physically obtained as opposed to non-destructive monitoring and if a recapturing is required or not. All methods are described in detail and information about the assumptions, ways of modelling, statistical approaches, study design and limitations is provided. We distinguished the following main general methods:

1. *Plot sampling* is a comprehensive method used mainly for measurements of population size in terms of abundance. Furthermore, plot samples can be used for estimating other relevant parameters of a population, such as biomass or length structure. The key idea of plot sampling is to estimate population abundance by "scaling up" the counts of animals from the covered (surveyed) area to the study area.
2. *Distance sampling* comprises a set of methods for estimating density and/or abundance of biological populations by properly accounting for detectability. Abundance estimation is often confounded by detection probability, which is the probability of correctly recording the presence of an individual within the sampled area. Failure to properly account for detection probability often leads to biased abundance estimators and possibly false estimates of population status and trends.
3. *Occupancy Estimation and Modelling* by simultaneously accounting for detectability. Estimation of density or abundance is often costly and requires substantial effort or may be unfeasible for various reasons (e.g., in the case of rare or very cryptic species). Alternatively, species occupancy, defined as the proportion of area, patches, or sampling units occupied (or alternatively as the probability of presence in a sampling unit) may be seen as a low-cost surrogate of abundance. Occupancy would be the state variable of choice in studies of distribution and range, alien invasions, metapopulation studies, community studies, and large-scale monitoring.
4. *Mark-Recapture* is a common technique used to estimate the size of populations, to study movements and migration of individuals and to provide information on birth, death and growth rates of species. MR methods are based on capturing and marking individuals from a population and then resample the same population to count the number of marked and unmarked individuals. The size of the entire population can be estimated from the proportions of marked and unmarked individuals.
5. *Removal and Catch-effort* methods allow for the estimation of the density and abundance of animal populations**.** The most commonly used methods include: (i) Removal method (ii) Change-in-ratio method (iii) Catch-effort method. All of them require at least two samplings (surveys) and are based on the concept that changes in detection rate, following removal of a number of animals, provide valuable data that allows for estimation of the population size.

Compared to many other animal groups, the monitoring of marine mammal abundance (**Chapter 2.3.2**) is particularly difficult. They are generally only visible when they come to the surface to breathe, or when they haul out on ice or land. Some species are distributed over large areas and migrate long distances. Marine mammal population sizes can be estimated by a variety of techniques. For absolute abundance estimates, the main methods are distance sampling, mark-recapture, migration counts and colony counts. Other approaches, providing mostly relative abundance estimates, are acoustic methods such as towed or stationary hydrophones.

Many different aspects of seabird biology are monitored and hence, many different survey and monitoring techniques are used (**Chapter 2.3.3**). At sea, many seabirds disperse widely and are thus hard to follow. Densities and distributions are normally mapped by ship-based or aerial surveys. Such mapping exercises are normally limited to single seas or areas under national jurisdiction, i.e. rarely will cover the entire ranges of seabirds at sea. Moreover, as monitoring requires repeated surveys, not all mapping at sea should be seen as monitoring, although the methods used are fit for these purposes as well.

There are many different methods for monitoring turtle populations (**Chapter 2.3.4**), but universally, nest counts are the most standard technique for assessing sea turtle population. Although nest counts only provide a good assessment of the number of adult females, nest counts are the primary response variable for assessing changes in sea turtle population size. Nest counts also provide additional information such as number of eggs per nest, number of hatchlings and number of nests per females during a breeding season. Additional methods for monitoring turtle populations include visual surveys from boats and aircraft, mark recapture techniques such as PIT tagging on nesting beaches, and satellite tracking of individuals, which provide important data on internesting intervals and remigration intervals.

Abundance, biomass and demographic structure of fish populations can be monitored and assessed with indirect and direct methods (**Chapter 2.3.5**). Indirect methods are based on commercial fisheries data and include (a) Virtual Population Analysis (VPA) and other similar methods, and (b) methods based on catch and effort data. VPA is based on catch composition data that are raised to the total catch composition of the fishery, then used to estimate the relative abundance of different age classes or cohorts and to determine the current mortality rate in the stock. The methods based on catch and effort data make use of the weight of commercial catch and the relevant fishing effort raised to the total fishery, collected through interviews at landing points or by observers on board of fishing vessels. Such data are then used to construct biomass dynamic models, which are centered on the need to find the maximum sustainable yield (MSY), i.e. the highest catch that can be sustainably harvested in the long term.

Direct methods are based on research surveys and aim at avoiding the biases deriving from the analysis of commercial catches. Four main methods are used: (i) bottom otter trawl surveys (for all demersal fish and invertebrate species), (ii) beam trawl surveys (for flatfish), (iii) ichthyoplankton surveys (to estimate stock biomass from eggs and larvae), (iv) underwater visual surveys (for shallow-water fish on soft and hard bottoms). The first three methods make use of a vessel and specific sampling gear to collect fish or ichthyoplankton samples, while the fourth is based on direct underwater visual observations made by scuba divers or by video equipment.

The spatial distribution of fish populations can be monitored with two main methods: (a) spatial analysis of geo-referenced data, and (b) acoustic/radio telemetry. Geo-referenced raw catch and effort data from commercial fisheries, fisheries surveys or visual surveys may be analyzed to provide insight on fish spatial distribution and essential fish habitats. Such data can be related simply to depth strata or, more effectively, to environmental and spatial variables that are expected to affect directly or indirectly fish distribution, and can be interpolated over wide areas, even at the regional scale. Telemetry methods are based on electronic tags inserted in single fish or invertebrate specimens, which can be tracked by means of manual or automated receivers. Ultrasonic/acoustic telemetry uses sound waves in the 20-300 kHz frequency range; radio telemetry uses radio waves in the 30-170 MHz range.

Approaches, sampling strategies, methods, tools, sample treatment and analytical techniques to monitor invertebrate populations, including macro-zoobenthos in soft substrata, epibenthos, hyperbenthos, meiofauna and zooplankton were reviewed in **Chapter 2.3.6**.

A wide variety of sampling gears and methods has been developed for macro-zoobenthos sampling in soft substrata such as grabs, corers, dredges, benthic landers or diver-operated samplers. Such sampling target on bottom-dwelling animals retained on a mesh screen of 1 mm or 0.5 mm size and aim to define the biocommunity structure and its spatial/temporal variation in the study area. Epibenthos are animals living on or immediately above the seafloor. Epibenthos is routinely sampled by vessel-towed gears (such as trawls, dredges, and sledges) but also by diver operated sampling, camera sledges, ROVs and acoustic methods. Hyperbenthos is a term applied to the association of small sized bottom-dependent animals (mainly Crustaceans) that have good swimming ability and perform, with varying amplitude, intensity and regularity, seasonal or daily vertical migrations above the seabed. Hyperbenthos is only correctly sampled using sledges because of its swimming capacity.

Plot Sampling or Distance Sampling with SCUBA or free diving are commonly applied techniques for abundance estimations of benthic megafauna. Other non-destructive methods based on SCUBA or free diving include Line Intercept Transects, Point Intercept Transects, Chain Transects, Fixed-Time Swims or SCUBA surveys, Nearest-Neighbor and Point-to-Nearest-Individual surveys. In deeper waters megabenthic fauna can be monitored with visual techniques using submersibles, remotely operated vehicles (ROVs) or drop cameras.

During the last years, meiofauna (i.e., organisms displaying a size intermediate between the smaller micro-benthos, such as bacteria and diatoms, and the larger macro-benthos) has been largely used as collective indicator of alteration of the functioning of marine ecosystems. Due to their high sensitivity to environmental perturbations, to the high number of individuals, to the lack of larval dispersion and to the short life cycle, meiofauna are becoming a common target to evaluate the disturbance and re-colonization of marine environments.

Methods to monitor zooplankton (sampling design, sampling systems, and analytical techniques) as well as recent approaches and examples for shellfish monitoring and stock assessment are also provided in Chapter 2.3.6.

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CHAPTER 1:
REVIEW OF THE CONCEPTS, OBJECTIVES, POLICY FRAMEWORK, AND CRITICAL ISSUES RELATED TO THE IMPLEMENTATION OF SPATIALLY MANAGED MARINE AREAS

1.1 REVIEW OF CONCEPTS, APPROACHES, DEVELOPMENTS, AND LEGISLATION RELATED TO THE SPATIAL MANAGEMENT OF MARINE AREAS

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1.1.0 GENERAL REMARKS

This review comprises four sections covering the concepts of and approaches to marine spatial management (1.1.1), adaptive management in theory and practice (1.1.2), legislation and operational objectives in practice (1.1.3) and performance measures for operational objectives in practice (1.1.4). The review forms the basis for the generic framework for monitoring and evaluation of spatially managed areas to be developed in WP2.

1.1.1 CONCEPTS OF AND APPROACHES TO MARINE SPATIAL MANAGEMENT

1.1.1.1The general concept of an ecosystem based management

Current governance of marine systems is implemented by a sector by sector approach, which has lead to fragmentation and spatial/temporal mismatches in governance (Crowder and Norse 2008). Conventional sector by sector management is not equipped to handle all the human activities that affect the ocean as it does not take into account the interaction among activities and the cumulative impacts of these activities amongst other factors (Halpern et al. 2008). In contrast, an ecosystem-based management is an environmental management approach that recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species, or ecosystem services in isolation. The goal of ecosystem-based management is to maintain an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need (McLeod et al, 2005).

Also Foley et al. (2010) stated that the future of the ocean depends on successful, immediate implementation of comprehensive governance framework that moves away from a sector-by sector management approach to one that balances the increasing number, diversity and intensity of human activities with the ocean's ability to provide ecosystem services; incorporate appropriate ecological, economic, social and cultural perspectives; and supports management that is coordinated at the scale of ecosystems as well as political jurisdictions.

The key principles for marine ecosystem-based management include those stated by Norse (2005) and Crowder et al. (2006):

- Plan based on desired ecological objectives for different areas
- Protect and recover biodiversity and ecosystem functions by offering a range of protections within ecologically meaningful boundaries
- Use best available science
- Learn and manage adaptively based on new information
- Educate the public
- Acknowledge existing obligations and rights
- Make planning open, inclusive and accountable
- Separate incompatible uses
- Ensure that there is one managing entity in each zone or group of zones

The successful development and implementation of ecosystem based management requires the use of best available science. Marine ecologists and oceanographers have been actively developing and using new tools such as geospatial analysis, remote sensing, molecular techniques, telemetry, modelling, and quantitative analysis to understand the spatial and temporal dynamics of marine ecosystems and their component organisms in relationship to environmental variation. These new tools have broadened the understanding of the linkages between marine habitats and population dynamics, and between spatial and temporal dynamics and the function of marine food webs (Crowder and Norse, 2008). Furthermore, as expressed by the way forward to support ecosystem based management is by data integration across traditional disciplines at appropriate scales: i.e. building upon existing information collected for a variety of specific purposes and providing a framework to combine these data (Carollo et al. 2009). In addition to making use of the best science available, managers have to continuously learn and manage adaptively whenever new information is available.

According to Day (2008), monitoring, evaluation, reporting and adaptive management are widely considered as fundamental components for effective marine management. The author further comments that any successful marine spatial management system must be adaptive, as well as able to include any changes. These may include new information becoming available or circumstances changing. However, an in-depth review of the application of the ecosystem approach, carried out by the Convention on Biological Diversity, revealed that several barriers are preventing the actual implementation of ecosystem based management (Douvere and Ehler, 2008). The is mainly due to the fact that the ecosystem approach is still more of a concept which is widely discussed at scientific fora, but with very limited examples of actual practice.

1.1.1.2 Tools to support the implementation of an ecosystem based management

Marine spatial planning (MSP)

MSP has emerged as a means of resolving inter-sectoral and cross-border conflicts over maritime space and has been therefore defined as a way of improving decision making and delivering an ecosystem-based approach to managing human activities in the marine environment (Ehler and Douvere 2008). The recent EC Communication on maritime spatial planning (COM (2008)791) lays out some guiding principles stating that the sustainable management of marine regions depends on the condition of the respective ecosystem. Therefore the ecosystem approach is an overarching principle and MSP is a tool to for its implementation. Although MSP is an essential tool for an ecosystem-scale sea use management approach (Ehler, 2008; Halpern et al., 2008), it has not yet been clearly defined and the therefore the concepts of MSP are still open to very diverse interpretations Gilliand and Laffoley (2008).

According to Ehler and Douvere (2007) MSP is a public process of analyzing and allocating the spatial and temporal distribution of human activities in marine areas to achieve ecological, economic, and social objectives that usually have been specified through a political process. A recent study by Foley et al (2010) defines ecosystem based MSP as an integrated planning framework that informs the spatial distribution of activities in and on the ocean in order to support current and future uses of ocean ecosystems and maintain the delivery of valuable ecosystem services for future generations in a way that meets ecological, economic and social objectives.

Initially the concept of MSP was stimulated by international and national interests in developing marine protected areas (MPAs). More recent attention has been placed on planning and managing multiple uses of marine space, particularly in areas where use conflicts are already well known and specified. Ecosystem-based MSP aims to bridge the gap between science and practice and help fill the current need of both governments and non-governmental organizations for more practical tools that move forward the implementation of ecosystem-based management in marine places (Douvere, 2008). Moreover, the MSP decision-making process should be based on ecological principles such as native species diversity, habitat diversity and heterogeneity, key species, and connectivity with clearly defined targets for those ecological attributes (Foley et al. 2010). In addition the context and uncertainty must be taken into account in the MSP planning process. Recently a guide of ten practical steps on how to make MSP operational was launched, based on international examples of MSP at different stages of development from all around the globe (Ehler and Douvere, 2009a).

An increasing number of countries are using marine spatial management to achieve sustainable use and biodiversity conservation in ocean and coastal areas. While some countries have specific legislation for MSP (e.g., the UK has theMarine and Coastal Access Bill), most do not and certain countries use existing authorities — including environmental legislation (like Belgium) or biodiversity legislation (like Australia) — or existing land-use planning legislation that is extended to the sea (The Netherlands and Germany) as a basis of authority for MSP (Ehler and Douvere, 2009b). The following are some examples of countries using the MSP process (see www.unesco-ioc-marinesp.be/msp_around_the_world).

One of the best-known MSP examples is Australia's Great Barrier Reef Marine Park being among the largest MPAs in the world. The perception that the Great Barrier Reef was degrading was a fundamental driver in the process of establishing the marine park in the late 1960s and early 1970s. Main threats included oil drilling and limestone mining, pollution from shipping, land-based sources of pollution, and increased fishing and tourism activity. Spatial planning and zoning, considered as the cornerstones of the management strategy for the protection of the Great Barrier Reef, were established; plans of management that aimed to complement zoning, have been prepared for intensively used, or particularly vulnerable groups of islands and reefs, and for the protection of vulnerable species or ecological communities. A permit system is also used to implement zoning plans.

In 1997, Canada became the first country in the world to adopt comprehensive legislation for integrated ocean management. Five Large Ocean Management Areas (LOMAs) have been identified to address large-scale ocean space issues and provide the context for future integrated management and spatial planning. MSP is furthest developed so far for one of the LOMA regions the Eastern Scotian Shelf, where a strategic plan for integrated ocean management has been published in 2007. As part of the plan, human uses have been identified and mapped and objectives have been set for the future integrated management of ocean space on the Eastern Scotian Shelf. Although MSP is not explicitly addressed in the particular plan, many of its objectives would require MSP for implementation.

In Europe most ecosystem based, sea use management and MSP initiatives are driven by international and European legislation (Douvere and Ehler, 2008). Various legal and policy documents have been introduced by the European Commission to implement MSP, one of which is the Green Paper on Maritime Policy which identifies MSP as a key tool for the management of a growing and increasingly competing maritime economy, while at the same time safeguarding maritime biodiversity (Com(2006) 275 final). There are several other important key drivers aiding the introduction and the implementation ecosystem based management and MSP in European Union Member States (Douvere, 2008). These include the Birds Directive (1979) which provides a framework for the identification and classification of special protection areas and the Habitats Directive (1992) that requires European Union Member States to select, designate and protect sites that support certain natural habitats or species of plants or animals.

Belgium was among the first countries to implement an operational, multiple-use MSP system that covers its territorial sea and exclusive economic zone. The main drivers for spatial planning in Belgium came from the demand for offshore wind energy and international requirements for the protection and conservation of ecologically and biologically valuable areas. MSP in Belgium aims at achieving both economic and ecological objectives, including the development of offshore wind farms, the delimitation of marine protected areas, a policy plan for sustainable sand and gravel extraction, the mapping of marine habitats, protection of wrecks valuable for biodiversity, and the management of land-based activities affecting the marine environment. Together, these objectives provided the basis for a Master Plan, which has been implemented incrementally since 2003.

MSP experiences revealed that the stakeholder participation and involvement is important for the success of MSP as it reduces conflicts Ehler (2008). The stakeholders that are involved in the process must reflect the existing complexity in reality (Pomeroy and Douvere, 2008). Moreover, stakeholders should be involved in developing the overall MSP framework and process, rather than only when consultations on an actual plan reaches them, as well as the planning production process itself (Gilliand and Laffoley 2008).

Currently most MSP initiatives are confined within national boundaries and only take into account local habitats and ecosystems (Maes 2008). But various uses such as shipping, fisheries and pipelines may have impacts across boundaries and therefore MSP has to be implemented on a regional and international level (Douvere and Ehler, 2008). Ultimately, national MSPs initiatives should further develop into cross-border and regional marine spatial plans to fully implement a sustainable marine ecosystem based management.

Ocean Zoning

Marine space has been zoned for individual human uses for decades; fisheries have been opened or closed in particular areas/zones, marine transportation has been managed within designated lanes or zones especially in intensively-used areas, rights to explore or exploit energy or mineral resources have been leased on an area basis, marine protected areas have been designated in many places in the world. However, these zones and others have usually been planned on a single-sector basis.

Ocean zoning (OZ) is a set of regulatory measures used to implement marine spatial plans and it is considered as one tool for MSP (Ehler and Douvere, 2007). Zoning partitions a region into zones that are designed to allow or prohibit certain activities, with the intent to maintain the provision of an overall set of ecosystem services provided by the overall zoned area; such a zoning process needs to pay particular attention to the consequences of allowing multiple conflicting activities to occur within the same location (Halpern et al., 2008a). Zoning in which environmental protection is harmonized with uses of the sea is likely the most effective approach to mitigate and possibly reverse extensive and increasing human impacts on marine and coastal ecosystems (Agardy, 2009). Comprehensive ocean zoning may not only reduce conflicts through the creation of use-priority areas but also act as a catalyst for users within zones to coordinate their activities, especially with the creation of dominant-use zones (Sanchirico et al., 2010).

Zoning in general was developed for use on land (Douvere et al., 2008). Whereas terrestrial zoning is usually small-scale, within the remit of municipal planning authorities, OZ often must recognize the wide linkages across marine and coastal ecosystems. It must also systematically address uses of, and impacts on, the marine environment at a regional scale. Lack of property rights in the ocean can hinder efforts to apply the land-based principles of zoning to the marine environment. However, zoning of communal or common property such as marine space and resources is possible by amending legislation toward use rights. Experience in the largest zoned area of ocean, the Great Barrier Reef (GBR) Marine Park off Queensland, Australia showed that a simple zones classification was crucial for public acceptance, while the zones spectrum has evolved and changed considerably over the years (Day, 2002). China has also adopted ocean zoning to ensure that areas where water quality is still suitable for aquaculture are not given over to other competing uses. In Belgium, zoning presents an example of how comprehensive ocean zoning could proceed in other marine areas with a high degree of use and heavy congestion of users. In the context of the research initiative a comprehensive method was developed in the frame of the GAUFRE project for investigating alternative spatial sea use scenarios, and on the project's website structural maps feature that can be used as guiding records for future actual zonation or reallocation of existing activities. In fact, tools for ocean zoning are just starting to emerge, and a popular one is "Marxan with Zones" (Watts et al., 2009), which helps create alternative zoning configurations that maximize the achievement of social, economic, and ecological objectives while minimizing the total social, economic, and ecological cost. Another emerging analytical method useful for ocean zoning focuses on mapping the cumulative impact of different suites of human activities on the ocean; with this type of analysis, the overall impact of all human activities on ocean condition can be assessed and particular activities can be included or excluded from consideration to determine what suite of activities can best meet objectives for a given zone (Halpern et al., 2008b).

1.1.2 Adaptive Management in Theory and Practice

An increasing number of scientists and resource managers recognise that successful marine management approaches, including marine spatial planning (MSP), cannot occur without effective monitoring, evaluation and adaptation (Day 2008). Thus, adaptive management is recommended as a tool to acknowledge uncertainty, and to follow a plan by which decisions are modified as we learn by doing (Parma et al., 1998).

The basic components monitoring, evaluation and adaptation are necessary to ensure that marine management measures are both effective and efficient. There are a number of fundamental principles for marine monitoring, evaluation and adaptive management, but the challenge is how to develop realistic and measurable objectives and indicators against which effectiveness can be practically measured. This can be quite complicated as the focus of marine planning and management strategies are not only 'single species' but also 'habitats' and whole 'ecosystems'. In addition to this, it may also be relevant to set multiple objectives taking into account interactions between ecological, economical and societal factors. This makes the development of measurable objectives (referred to as operational objectives) and the associated criteria and indicators even more complicated. Because both natural systems and management approaches are never static, continuous and iterative monitoring, evaluation, reporting and adaptive management are fundamental components for effective marine management.

Monitoring, evaluation and adaptation

Monitoring is a fundamental management tool to document environmental impacts and assess the effectiveness of management actions. Monitoring management performance is also becoming a more important task in order to know if implemented management measures are efficient, effective and equitable. The fundamental principles for monitoring include identifying the objectives; monitoring options, scale and identification of costs and benefits. Answering the six key questions shown in Table 1.1.1 will provide a comprehensive evaluation of any marine managed area or management/planning process.

Table 1.1.1: Framework for assessing management effectiveness (taken from Hockings et al 2000)

Evaluation element	Context	Planning	Inputs	Processes	Outputs	Outcomes
Key question	Where are we now?	Where do we want to be?	What do we need?	How do we go about it?	What were the results?	What did we achieve?
Focus of evaluation	Current status	Desired outcome	Resources required	Efficiency	Effectiveness	Effectiveness and appropriateness against objective(s)

In recent years, governments have placed growing emphasis on outcome-based (rather than activity-based) performance reporting, which includes measures of performance in achieving objectives or targets. The first, and most fundamental, requirement for measuring performance in a marine managed area is setting clear, operational objectives and specifying management strategies to achieve those. Effectiveness is then measured through the processes of evaluation and monitoring against those operational objectives. The most important reasons for evaluating management performance and effectiveness in marine ecosystems is to demonstrate the extent to which the objectives of marine planning and management have been achieved as well as providing evidence-based feedback about what's working and what's not and learn more about interactions between ecological components and management efforts. Jones (2008) provides one example of an evaluative management framework comprising seven key steps:

- Step 1: Identify management objectives
- Step 2: Define key desired outcomes
- Step 3: Identify performance indicators
- Step 4: Undertake monitoring
- Step 5: Periodically assess results
- Step 6: Report findings and recommendations
- Step 7: Adjust management as necessary

Pomeroy et al. (2005) developed a step-by-step guide to managers and practitioners for evaluating effectiveness of MPA management based on a set of biophysical, socioeconomic and governance goals and 42 corresponding indicators that can be selected and adapted to fulfill different MPAs' evaluation needs. The guide aids in selecting relevant indicators, developing a step-by-step process and checklist for planning and implementation of evaluations, collecting and analyzing data for selected indicators and in using and communicating results to inform and adapt planning processes. These four steps are illustrated in Fig. 1.1.1.

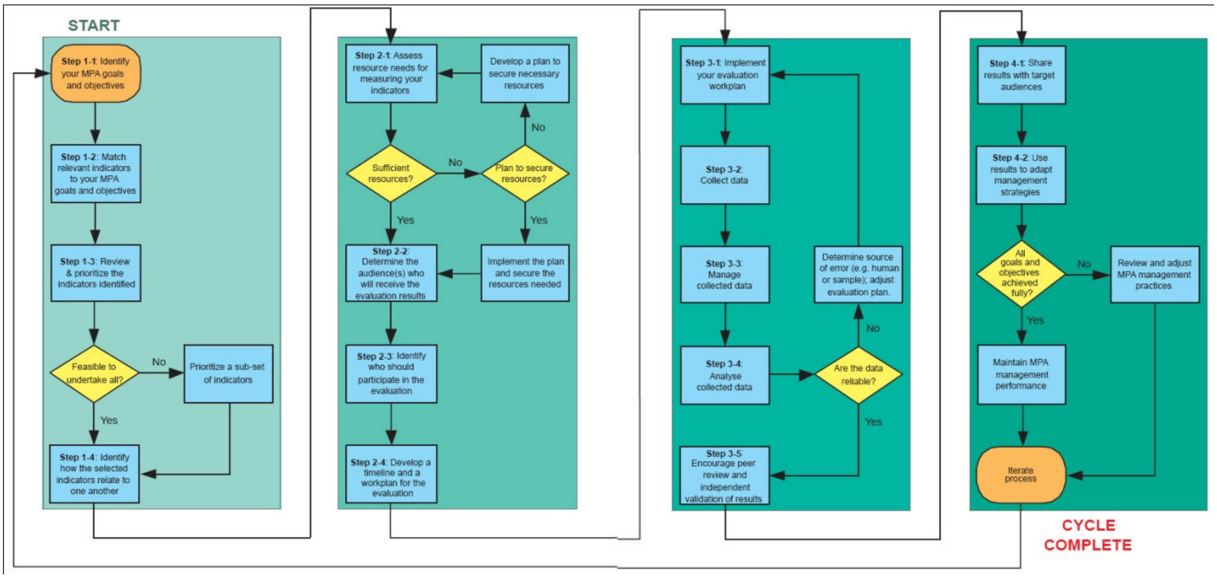

Figure 1.1.1: Step by step guide for the performance assessment of MPAs (taken from Pomeroy et al. 2004)

As there has been a history of adapting methods and concepts (such as e.g. zoning) from MPA experiences to a wider marine spatial planning and ecosystem based management context, such evaluations may provide a good basis for development of a framework for the monitoring and evaluation of spatially managed areas in general.

Pomeroy et al. (2005) emphasised the importance of the feedback processes in evaluation of MPA management effectiveness (far left in Fig. 1.1.1). Evidence-based feedback (Day 2008) is crucial to the achievement of the objectives of any adaptive management framework (Walters and Hilborn 1976; Pomeroy et al. 2005; Grafton and Kompas 2005; Ekebom et al. 2008; Blæsbjerg et al. 2009; Foley et al 2010), as it provides important output from a past process that must inform and influence current or future processes. Considering that most evaluations of spatially managed areas in the marine environment focus on biophysical aspects and that these evaluations rarely involve management staff (Ehler and Douvere 2009), it is important that evaluations are formulated and communicated in a way that ensures that feedback re-enters the adaptive management cycle as effectively as possible to improve the likelihood of successful management in future.

Pomeroy et al. (2005) suggest that, in relation to MPAs, results of evaluations can be used to highlight the progress of MPA management and in setting of priorities. Feedback from evaluations may also facilitate in the ongoing identification of research needed to inform management in relation to knowledge gaps (Ehler and Douvere 2009). In addition, stakeholders and coastal communities may utilise evaluation results to establish whether or not their interests have been taken into account and addressed in management (Pomeroy et al. 2005). To ensure that feedback is understood outside e.g. the scientific community, however, all of the above examples require effective communication strategies (Pomeroy et al. 2005; Pomeroy and Douvere 2008; Ehler and Douvere 2009).

A well described example for a successful spatial management approach including its monitoring and evaluation is the in Great Barrier Reef Marine Park (GBRMP). Over the last 30 years, the GBRMP has successfully established a multiple-use spatial management approach that allows both high levels of environmental protection and a wide range of human activities. A comprehensive ocean zoning system is here developed to ensure that the wide range of marine activities is ecologically sustainable. The multiple-use zoning system governs all human activities, providing high levels of protection for specific areas, while allowing a variety of other uses elsewhere. The main experiences from this project were the importance of starting with a modest monitoring programme for a few key performance indicators and expand programs by experience. Thus priority should be given to the achievement of key objectives, conservation values at risk and important/complex management issues. Scientific knowledge played an important part of the rezoning of the GBRMP, but the successful outcome relied significantly upon three other critical, inter-dependent aspects i) effective leadership, ii) the high level of public participation, and iii) the consequent socio-political support. Encouraging stakeholder participation was found to be important and users already out in the marine area could e.g. be helpful in a monitoring program. Experiences from this project also showed the importance of innovative monitoring approaches that may be more affordable or acceptable and that there is often a need to measure indicators both within a particular marine managed area and outside the area to determine relative changes.

Adaptive management

In spite of the efforts in ecological management, mismanagement of biological resources has been common and well publicized (Parfit,1995; Chadwick, 1996; Gallangher and Carpenter, 1997; Mitchel,1997). Although economical and political factors are major contributors to these failures (Holmes, 1994) the limits of our ability to predict the response of ecosystems to human interventions have certainly played an important role, as can be seen in cases in which management disasters can be traced to seriously flawed assessments and forecasts (Walters and Maguire, 1996). To get the most out of human interventions, a plan is needed that 1) recognizes the uncertainty, 2) contemplates monitoring system responses to interventions, and 3) anticipates that future management interventions will be modified as we gather more information and learn about the behavior of the system.

Adaptive management has been proposed as a method for ecological intervention in the fact of uncertainty (Ludwig et al., 1993). Although originating already in the 1970's it is now a buzzword, commonly confused with an ad hoc trial and error approach to management under uncertainty as in 'action first, science later' (Parma et al, 1998). In order to clarify what adaptive management really is and what it can do for biological resource management we need to i) know about the different forms of uncertainty that plague natural systems (uncertainty of natural processes; model uncertainty; observation uncertainty), ii) indicate consequences of not acting adaptively, iii) point out major problem areas in for instance population control for which adaptive management could be used, and iv) discuss the intrinsic limitations of adaptive management that have prevented its implementation.

Parma et al. (1998) define adaptive management as managing according to a plan by which decisions are made and modified as a function of what is known and learned about the system, including information about the effect of previous management actions. Key components include i) locally appropriate policies, ii) management systems that implement those policies, iii) monitoring to determine the system responses and evaluation of the monitoring results, and iv) consequent adjustment of management if the evaluation shows it is necessary. Any successful marine spatial management system must integrate all these aspects. Furthermore, it must be adaptable and be able to incorporate changes such as new information becoming available or as circumstances change. Irrespective of whether a change in marine management results from new data, 'in-the-field' experience, or as a result of external circumstances, marine spatial management practices must be periodically reviewed and updated where appropriate. Adaptive management enables managers to be flexible and to expect, and deal with, the unexpected (Day 2008). Adaptive management knows two modules and it needs to be distinguished from the precautionary approach (Table 1.1.2).

Table 1.1.2: Definitions and some advantages (+) and drawbacks (-) of Adaptive Management (AM) and the Precautionary Approach (Parma et al., 1998, (FAO, 1995).

A M		Precautionary Approach
Passive AM	Active AM	
Definition: >When policies are adapted in response to new information, but learning is not incorporated as a management goal< (Parma et al., 1998)	Definition: >As in experiments, in manage-ment the information value of alternative candidate actions is considered in evaluating choices<(Parma et al., 1998)	Definition: >We should not proceed with ecological intervention unless we are reasonably sure that this will not cause a significant long-term loss of productivity or a significant long-term impact on the environment< (FAO, 1995)
• possible high costs; we don’t know how much we loose due to the component ‘don’t know enough detail’	• probable great costs because the environment is manipulated as a whole	• experimental management must, in the first instances, be conservative
• misses the opportunity to learn fast	• Short-term negative effects perceivable	
• ignore future learning	->Benefits expected from learning must overcompensate the short-term costs of implementing experiments.	
Whenever a strategy is actively of passively adaptive, the system responses are monitored, effects of past actions are evaluated, and management is able to respond in an effective and timely manner to what is learned.		

Experimental management, such as adaptive management, has its limitations, which may explain why it has not been more commonly implemented. The major constraints are constraints in the policies that can be considered, difficulties in monitoring populations and their responses to interventions, lags in system responses, and limits in our ability to depict all possible states of nature (Parma et al., 1998). Adaptive management is the best mind-set for ecological intervention. We cannot control or manage populations or ecosystems; rather we control the level of human interaction with an intervention in natural systems. Adaptive management also forces us to evaluate the effects of past actions as part of the management plan (Mangel et al., 1996), and implies that management is able to respond effectively. Finally, adaptive management is based on the recognition that our actions in the future will change as new information is obtained (Platt, 1964).

Future challenges

Most efforts to date have concentrated on the bio-physical aspects/condition in a few selected areas and there are few comprehensive assessments of management effectiveness which include social or economic aspects. There are also few monitoring studies of performance mainly because high costs and institutional barriers. Even though cumulative impacts are increasingly becoming recognised as important, most legislation for marine management worldwide has been based upon single issue statutes. Lack of sufficient knowledge about marine ecosystems often prevents the setting of meaningful objectives in an outcome-oriented language. There is also a need to put in place a system of monitoring for the unexpected. The focus needs to exceed proving for the problems we already know about and detect and avert tomorrow’s disasters. AM is a promising tool to attain this goal Parma et al., 1998). Establishment of robust systems for evaluating management effectiveness of marine spatial management plans often require major institutional reorientation at the policy level (Day, 2008)

1.1.3 LEGISLATION AND OPERATIONAL OBJECTIVES IN PRACTICE

1.1.3.1 Operational objectives: A general introduction

In general, planning or management are driven either implicitly or explicitly by goals. Goals are high-level outcomes that a planning process or management aims to achieve through the implementations of measures. High level goals are a crucial part of any management and are based on societal or cultural values. In contrast, criteria, attributes and indicators that are associated to a high level objective are developed in the scientific domain (Moilanen et al., 2009 and citations therein). However, high-level goals and objectives need to be translated into more operational objectives before specific targets, limits and measures can be elaborated.

Experience from the implementation of an ecosystem approach to fisheries has shown the importance of translating high level policy strategies and goals into operational objectives (Garcia et al., 2003). The authors also identified the need to clarify operational objectives in relation to management structure, process, and measures, rationalizing and bringing transparency in the implementation path between high-level policies and on-the-ground fishing operations.

In recent years the formulation of operational objectives and operational deliveries has been proposed in the wider context of an ecosystem based approach to marine management. For instance (Rogers et al., 2007) described a hierarchical framework that incorporates the marine objectives and delivery statements of ecological, social and economic sectors. This framework outlines the relationship between high-level goals and principles, a high-level and operational delivery. Another practical example of operational descriptions is given by (Pomeroy et al., 2005) where a GOIS (Goal – Objective – Indicator - Success Criteria) framework was used to assess the management performance of marine protected areas.

We define operational objectives as those for which operational, quantifiable, targets can be set such that management measures can be targeted and performance can be evaluated (**s**pecific-**m**easureable-**a**chievable-**r**ealistic-**t**ime limited objectives).

In the following this review looks at the operational objectives of the main marine policies that have been implemented worldwide over the last number of years. It initially deals with international objectives which have been the result of comprehensive international collaborations such as the Convention on Biological diversity. It then goes on to look at European Union directives such as the Marine Strategy Framework Directive. Finally, it looks at some national and local objectives from key areas around Europe. This review of marine objectives is not intended to be exhaustive, rather it covers those policies which are most influential in Europe and therefore most relevant to future work within MESMA.

This section of the literature review has two purposes; 1. To provide background information on the main directives and their operational objectives that affect the MESMA regions. 2. To be a starting off point for case study leaders when defining goals and operational objectives and to be used at stage 5 to assess findings of the case studies against the operational objectives. Information from this section will be used to develop a working manual for the framework and so is an important review of directives which may affect spatially managed areas not just for the case studies but for the framework in general.

1.1.3.2 High level goals and international objectives

A number of overarching international conventions, treaties and laws recognise the need to consider human pressures in the marine environment through an integrated, ecosystem approach to management of maritime activities. Some of these mainly address the coastal zone, as has been the common practice until recent years, where focus has expanded to include the open sea and the high seas. In the following, central examples of such agreements are presented.

United Nations Convention on the Law of the Sea (UNCLOS)

The United Nations Convention on the Law of the Sea (UNCLOS; UN 1983) is a comprehensive regime of law and order for the world's oceans and seas, governing all uses of the oceans and their resources. It recognises that all problems of ocean space are closely interrelated and need to be addressed as a whole. Consisting of 320 articles and nine annexes, the Law of the Sea governs all aspects of ocean space, such as delimitation, environmental control, marine scientific research, economic and commercial activities, transfer of technology and the settlement of disputes relating to ocean matters(www.un.org/Depts/los/convention_agreements/convention_overview_convention.htm).

UNCLOS governs the way maritime transport is conducted and is key to the designation of shipping lanes, rights of passage, etc. UNCLOS is also the mechanism through which the designation of exclusive economic zones is coordinated. Coastal States have sovereign rights in a 200-nautical mile exclusive economic zone (EEZ) with respect to natural resources and certain economic activities, and exercise jurisdiction over marine science research and environmental protection. All other States have freedom of navigation and in the EEZ, as well as freedom to lay submarine cables and pipelines. These maritime zones, however, do not coincide with ecosystem boundaries (Ehler and Douvere 2007). UNCLOS includes a number of other provisions concerning the marine environment in relation to e.g.:

- pollution of the marine environment
- alien species
- global and regional cooperation
- marine scientific research
- highly migratory species

Convention on Biological Diversity (CBD)

The Convention on Biological Diversity (CBD) (1992) is the most comprehensive and significant international instrument addressing the threats to marine and coastal biodiversity, and protecting, understanding and using marine resources sustainably. The Convention requires all member nations to establish a system of protected areas and to develop guidelines for the selection, establishment and management of protected areas. The Convention recognises that protected areas are not the only mechanism for conserving biodiversity but that they are an important element of the overall approach. The strategic plan of the Convention laid down the target of achieving a significant reduction in the current rate of biodiversity loss by 2010. This target was confirmed in the plan of implementation adopted at the World Summit on Sustainable Development (WSSD 2002) in Johannesburg in 2002. CBD has three overarching objectives:

- the conservation of biological diversity
- the sustainable use of its components
- the fair and equitable sharing of benefits arising from the use of genetic resources.

These objectives are to be met through the implementation of a number of measures including:

- the development of national strategies
- the integration of biodiversity considerations into sectoral and cross-sectoral plans
- the establishment of monitoring programmes
- extensive measures for *in-situ* and *ex-situ* conservation (e.g.establishing protected areas, controlling alien organisms, restoring degraded ecosystems)

CBD prescribes the ecosystem approach in achieving these objectives and states that implementation must be consistent with the The United Nations Convention on the Law of the Sea (UNCLOS).

The Jakarta Mandate on Marine and Coastal Biodiversity (1995) focuses on the relationships between conservation, the use of biological diversity and fishing activities and formed a valuable part of the implementation of the Convention on Biological Diversity. The five thematic issues identified within the Jakarta Mandate include:

- integrated marine and coastal area management
- marine and coastal protected areas
- sustainable use of marine and coastal living resources
- mariculture
- alien species

In summary the Convention on Biological Diversity has a high level objective of achieving a significant reduction in the current rate of biodiversity loss by 2010 which will be implemented by four operational directives; 1. the development of national strategies, 2. the integration of biodiversity considerations into sectoral and cross-sectoral plans, 3. the establishment of monitoring programmes and 4. extensive measures for *in-situ* and *ex-situ* conservation.

United Nations Agenda 21

The Agenda 21 programme was introduced in 1992 during the United Nations Conference on Environment and Development and full implementation was affirmed during the World Summit on Sustainable Development in 2002, during which States also committed themselves to promote the sustainable development of marine ecosystems through the application of the ecosystem approach by 2010, and promoted integrated, multisectoral, coastal and ocean management at the national level. Chapter 17 of Agenda 21 (UN 1992) deals with the protection of oceans and prescribes "new approaches to marine and coastal area management and development, at the national, subregional, regional and global levels, approaches that are integrated in content and are precautionary and anticipatory in ambit...".

Among the programme areas are integrated management and sustainable development of coastal areas; marine environmental protection; sustainable use and conservation of marine living resources under national jurisdiction and strengthening international, including regional, cooperation and coordination. Agenda 21 sets a number of concrete objectives, some of which relate directly to SMA's and MSP:

- integrated policy and decision-making process, including all involved sectors, to promote compatibility and a balance of uses
- identify existing and projected uses of coastal areas and their interactions
- apply preventive and precautionary approaches in project planning and implementation, including prior assessment and systematic observation of the impacts of major projects

Agenda 21 prescribes a number of activities that aim to support the above objectives:
Establishing coordinating mechanisms for integrated management and sustainable development of coastal and marine areas and their resources, at both the local and national levels to support, among other things *conservation and restoration of altered critical habitats* and *integration of sectoral programmes on sustainable development for settlements, agriculture, tourism, fishing, ports and industries affecting the coastal area;*
International cooperation and coordination...within a subregional, interregional, regional or global framework...to support and supplement national efforts of coastal States to promote integrated management and sustainable development of coastal and marine areas.

FAO Code of Conduct for Fisheries

The FAO Code of Conduct for Fisheries (1995) provides principles and standards applicable to the conservation, management and development of all fisheries. It also covers the capture, processing and trade of fish and fishery products, fishing operations, aquaculture, fisheries research and the integration of fisheries into coastal area management. Objectives of the Code include aim to e.g.:

- establish principles, in accordance with the relevant rules of international law, for responsible fishing and fisheries activities, taking into account all their relevant biological, technological, economic, social, environmental and commercial aspects;
- establish principles and criteria for the elaboration and implementation of national policies for responsible conservation of fisheries resources and fisheries management and development;
- serve as an instrument of reference to help States to establish or to improve the legal and institutional framework required for the exercise of responsible fisheries and in the formulation and implementation of appropriate measures;
- provide guidance which may be used where appropriate in the formulation and implementation of international agreements and other legal instruments, both binding and voluntary;
- facilitate and promote technical, financial and other cooperation in conservation of fisheries resources and fisheries management and development;
- promote the contribution of fisheries to food security and food quality, giving priority to the nutritional needs of local communities;

- promote protection of living aquatic resources and their environments and coastal areas;

While the Code (FAO 1995) does not explicitly call for the establishment of MPAs, the FAO Technical Guidelines on fisheries management (FAO, 1997) state that *"marine protected areas can have a critical role to play in sustainable fishing... [and] can also play an important role in preserving critical habitats or sensitive life stages of species."*

1.1.3.3 Regional objectives

1.1.3.4 EU goals and objectives

Marine Strategy Framework Directive (MSFD)

The Marine Strategy Framework Directive (MSFD) was published in June 2008 with the aim of *'establishing a framework for community action in the field of marine environmental policy*'. The words 'strategy', 'framework' and 'policy' together with the concise 22 page length of the English version point to the fact that the MSFD is a high level document and requires further development and specification before it can be applied to specific case studies. This process of development and specification we term 'operationalisation' because it is what allows the MSFD to be applied operationally to specific situations.

The ultimate aim of the MSFD is expressed in its third paragraph :
'The marine environment is a precious heritage that must be protected, preserved and, where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive. In that respect, this Directive should, inter alia, promote the integration of environmental considerations into all relevant policy areas and deliver the environmental pillar of the future maritime policy for the European Union.'

Important aspects are that seas should be *clean, healthy and productive* and that the MSFD should integrate all other policy in the marine environment to achieve that. In terms of operationalisation the MSFD centres on 'Good Environmental Status' (GES) and that 'member states shall take the necessary measures to achieve or maintain good environmental status in the marine environment by the year 2020 at the latest.' (Article 1). Annex 1 of the MSFD provides 12 qualitative descriptors of Good Environmental Status which can be considered as the bases for the directives high level objectives:

Qualitative Descriptors of Good Environmental Status	
1)	Biological diversity
2)	Non-indigenous species
3)	Commercial fish
4)	Foodwebs
5)	Eutrophication
6)	Sea floor integrity
7)	Hydrography
8)	Contaminants
9)	Contaminants in food
10)	Marine litter
11)	Energy including noise

Member States should then determine for their marine waters a set of ***characteristics*** for GES. For those purposes, it is appropriate to make provision for the development of ***criteria*** and ***methodological standards*** to ensure consistency and to allow for comparison between marine regions or subregions of the extent to which good environmental status is being achieved.

The words highlighted in bold (*characteristics, criteria* and *methodological standards*) are the next step in the operationalisation of the MSFD. The descriptors of Good Environmental Status are explicitly defined as being qualitative, to be operational quantification is required.

Task Groups (TGs) have been organised by ICES and JRC to provide recommendations to DG ENV to assist with the development of these criteria and methodological standards {ref draft task groups report}. One Task Group has been created for each descriptor, with the exception of descriptor 7 Hydrography. The Task Groups in their draft report have identified '*key attributes*' of each *descriptor*. It is not quite clear whether these *key attributes* are equivalent to the *characteristics* defined in the MSFD itself.

1) Biological diversity
    a. Habitat diversity
    b. Species diversity
    c. Diversity within species
2) Non-indigenous species
    a. Number of NIS
    b. Abundance and distribution of NIS
    c. Impact on native communities
    d. Impact on habitats
    e. Impact on ecosystem functioning
3) Commercial fish
    a. Exploited sustainably consistent with high long-term yield
    b. Full reproductive capacity
    c. Healthy age and size distribution
4) Foodwebs
    a. Energy flows
    b. Structure of food webs (size and abundance)
5) Eutrophication
    a. Light availability
    b. Organic decomposition
    c. Algal community dominance
6) Sea floor integrity
    a. Substrate
    b. Oxygen concentration
    c. Bio-engineers
    d. Species composition
    e. Life history traits
    f. Size composition of the biotic community
    g. Trophodynamics
    h. contaminants
7) Hydrography : no Task Group
8) Contaminants
    a. Concentration of contaminants
    b. Levels of pollution effects
    c. Change over time in concentrations and pollution effects
9) Contaminants in food
    Key attributes not yet identified
10) Marine litter
    a. Amount and composition washed ashore
    b. Amount and composition in water column
    c. Amount and composition ingested by marine animals
    d. Amount and composition of microplastic particles
11) Energy including noise
    a. Loud, low and mid-frequency impulsive sounds

b. Low frequency, continuous sound
c. High frequency impulsive noise

The MSFD identifies (paragraph 27) that the next step should be '*the establishment of environmental targets and monitoring programmes …*' where an *environmental target* is defined (article 3.7) as 'a qualitative or quantitative statement on the desired condition of the different components of, and pressures and impacts on, marine waters …'

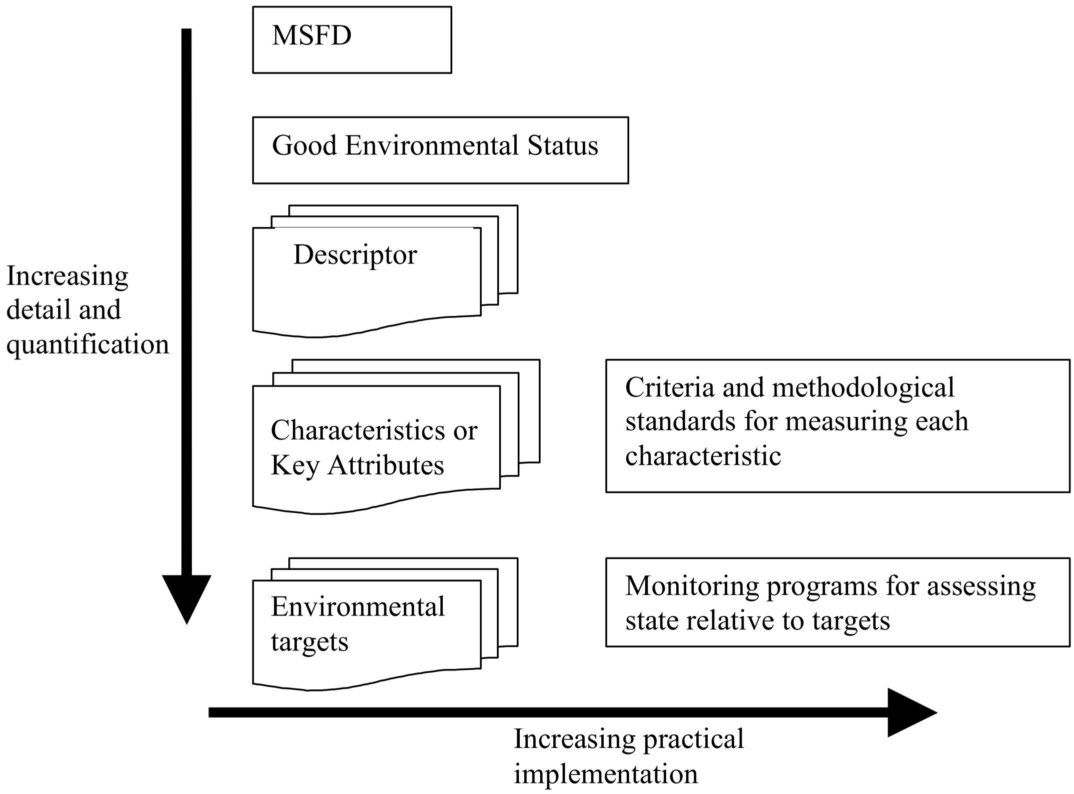

Figure 1.1.2: The process of operationalisation of the MSFD.

Water Framework Directive (WFD)

The EU Water Framework Directive (WFD) entered into force in December 2000. The official title of the WFD is "Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy". The WFD is a legislative framework that rationalises and updates existing water legislation by setting common EU wide objectives for water (inland surface waters, transitional waters, coastal waters and groundwater) and introduces an integrated and coordinated approach to water management in Europe. The high level goal of the WFD is long-term sustainable water management based on a high level of protection of the aquatic environment as expressed in its 19th paragraph 'This directive aims at maintaining and improving the aquatic environment in the Community. This purpose is primarily concerned with the quality of the waters concerned. Control of quantity is an ancillary element in securing good water quality and therefore measures on quantity, serving the objective of ensuring good quality, should also be established.'

With a view to the achievement of good quality status, the WFD contains provisions for the coordinated elaboration of River Basin Management Plans (RBMPs). A RBMP consists of a general description of the river basin district, a summary of all significant pressures and anthropogenic impacts, mapping of the protected areas, maps of the monitoring networks for the water bodies, including protected areas (inter alia Natura 2000 sites), a list of environmental objectives for the water use, a summary of all measures and programs of measures adopted, a list of the competent authorities, and a summary of public information and consultations measures. As part of a RBMP, a monitoring network has to be established (Article 8) to provide a coherent and comprehensive overview of ecological

and chemical status within each river basin. WFD Annex V specifies the information that is required from monitoring, it should permit the classification of all surface water bodies into one of five status classes: high, good, moderate, poor and bad. Three types of monitoring for surface waters are described: 1. Surveillance monitoring, 2. Operational monitoring and 3. Investigative monitoring. At present RBMPs have to be reported to the Commission by the Member States.
See http://ec.europa.eu/environment/water/participation/map_mc/map.htm for current reports.

To meet the high level goal the main aim of the WFD is to protect and restore clean water across Europe and ensure its long-term, sustainable use. This main aim is described in Article 1(a): The WFD "*prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystem*"
For making operational the programmes of measures specified in River Basin Management Plans, the environmental objectives to be achieved by the Member States are defined in Article 4. The main environmental objectives are to achieve and maintain 'good ecological status' for all surface waters by 2015. This is to be accomplished by implementing the measures necessary to

- prevent deterioration of the status of waters,
- protect, enhance and restore all bodies of surface waters and ground waters.
- promote sustainable water use (through effective pricing of water services),
- progressively reduce discharges of priority substances and cease or phase discharges of priority hazardous substances for surface waters,
- ensure progressive reduction of pollution of groundwater,
- mitigate the effects of floods and droughts,
- ensure sufficient supply of water,
- protect the marine environment.

Ecological status is 'an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V'. The ecological status classification is based on biological and physico-chemical monitoring results, the lower of the values stipulates for the relevant quality. As regards chemical status the scope of WFD is extended to cover all territorial waters. In terms of quality elements for transitional and coastal waters, the biological quality elements that should be taken into account include phytoplankton, macroalgae, angiosperms, benthic invertebrate fauna and fish fauna. Physico-chemical elements are the general conditions (transparency, oxygenation conditions or nutrients) and pollutants. Normative definitions for the high, good, and moderate status for different water categories and quality elements are given in Annex V, 1.2, Table 1.2.3 (transitional waters) and 1.2.4 (coastal waters). These normative definitions provide a general description how the critical biological components such as taxa composition, diversity, abundance, etc. change as response to environmental degradation and pressures, and thus provide generic description what is a good ecological status. The descriptors can be translated into specific quantitative metrics (e.g. various diversity indices or biomass metrics, or metrics describing numbers of sensitive vs. non-sensitive species in the marine environment) and practical quality targets need to be set by the Member States.

As an example of implementation of high level objectives into operational objectives see the ecological assessment of German transitional and coastal waters (Voss et al. 2010, in German).

In summary the goal of the WFD is long-term sustainable water management based on a high level of protection of the aquatic environment of the EU. To reach good water quality by 2015 important steps are the assessment of the ecological status and setting of practical management targets for the environmental objectives of the directive. The WFD defines which elements must be taken into account and provides a description what is a good ecological status, Member States translate these descriptors into specific quantitative metrics and practical quality targets need to be set.

Habitats Directive (HD)

The Habitats Directive was established in 1992 by the European Community, officially, Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (European Council 1992). The directive came into legal force in June 1994. It is through this directive that member states can fulfill their obligations as part of the Convention on the Conservation of European Wildlife and Natural Habitats (European Council 1979) ratified in 1982, otherwise known as the Bern Convention.

The high level objectives of the Habitats Directive are to '...ensure biodiversity through the conservation of natural habitats and wild fauna and flora and to maintain and restore natural habitats and species of wild fauna and flora of community interest.'

The objectives are ensured by the selection, designation and protection of a network of sites throughout Europe known as Special Areas of Conservation (SACs). Potential SACs were selected depending upon a list of Annex I habitats, which contained 189 habitats of conservation importance, and Annex II species of which there were 788 species. These annexes have been added to and as of 2006 there are 218 Annex I habitats and 887 Annex II species listed as of Community importance. Member States were asked to draw up national lists of Sites of Community Importance (SCIs) and the sites were then designated SACs within 6 years. These SACs are combined with Special Protection Areas (SPAs), sites designated to protect wild birds under the EU Birds Directive (European Council 1979), to form the Natura 2000 Network of important ecological sites across Europe. Member states are committed to providing an update, in the form of a report, on their protected sites every 6 years.

The criteria for the assessment of important sites were listed in Annex III of the directive.
Stage 1: The criteria for selecting sites eligible for identification as sites of community importance and designation as special areas of conservation are:
For habitats (Annex I):

1. *The degree of representativity of the natural habitat type on the site.* This is the degree to which a habitat corresponds to its described type.

2. Area of the site covered by the natural habitat type in relation to the total area covered by that natural habitat type within national territory. Examples with the largest surface area are most desirable.

3. *Degree of conservation of the structure and functions of the natural habitat type concerned and restoration possibilities.* Structure relates to biotic and abiotic features such as species composition and ground morphology whilst function relates to how these features interact over time. Those examples that are closest to typical structure and function are most desirable.

4. *Global assessment of the value of the site for conservation of the natural habitat type concerned.* This provides an integrated assessment of the other selection criteria and is a judgement of the overall value of the site for the conservation of the relevant Annex I habitat.

For species (Annex II):

1. Size and density of the population of the species present on the site in relation to the populations present within national territory. The biggest and most dense populations are most desirable for protection.

2. Degree of conservation of the features of the habitat which are important for the species concerned and restoration possibilities. Areas that have the best surrounding habitats in which the species can survive are regarded as the highest quality sites.

3. *Degree of isolation of the population present on the site in relation to the natural range of the species.* Only isolated populations that have a high density recorded over a number of years were considered.

4. Global assessment of the value of the site for conservation of the species concerned. An evaluation of the overall value of the site for the species concerned.

These criteria were used to draw in national lists of sites of Community importance. Stage 2 involved an assessment of the Community importance of the sites on the national lists. Community importance sites were assessed on their contribution to maintaining or re-establishing a natural habitat (Annex I) or species (Annex II).

The criteria for this are:
- Relative value of the site at national level
- Geographical situation of the site in relation to migration routes of species in Annex II and whether it belongs to a continuous ecosystem situated on both sides of one or more internal Community frontiers
- Total area of the site
- Number of natural habitat types in Annex I and species in Annex II present on the site
- Global ecological value of the site for the biogeographical regions.
- Sites chosen after this stage were designated SACs.

In summary the Habitats Directive high level goal is to achieve favourable conservation status for a list of Annex I habitats and Annex II species that are of European Community interest. This is to be achieved by the establishment of SACs throughout Europe, each of which is established to target certain habitat(s) or species.

Birds Directive (BD)

The Birds Directive [EU 2009] states (Article 2) that member states should 'maintain the population of the species referred to in Article 1 [natural wild birds] at **a level which corresponds in particular to ecological, scientific and cultural requirements**, while taking account of economic and recreational requirements, or to adapt the population of these species to that level'

and (article 3.1) :
'preserve, maintain or re-establish a sufficient diversity and area of habitats for all the species …'

and (article 3.2) :
'The preservation, maintenance and re-establishment of biotopes and habitats shall include primarily the following measures:
(a) creation of protected areas;
(b) upkeep and management in accordance with the ecologicalneeds of habitats inside and outside the protected zones;
(c) re-establishment of destroyed biotopes;
(d) creation of biotopes.'

Common Fisheries Policy (CFP)

The main objectives of the Common Fishery Policy CFP, stated in 2002, are:
CFP-O1 - Overcapacity reduction
CFP-O2 - Long term management plans
CFP-O3 - Ecosystem based approach to fishery
CFP-O4 - Stakeholders involvement in management processes
CFP-O5- Selective uses of public funds for activities coherent with CFP
CFP-O6- Bilateral fisheries agreements
CFP-O7 - Integration of CFP in a broader maritime context

However, the objectives agreed in 2002 to achieve sustainable fisheries have not been met and the Commission has started a CFP reform process (Green Paper, 2009) which states that the Common Fisheries Policy shall ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions. No priority is set for these objectives and while direct references are made to adopting a precautionary and an ecosystem approach, it is not clear how this relates to economic and social conditions. There are no clear indicators and yardsticks that could provide more concrete guidance or to help measure policy achievements.'

This followed from a Commission working document (EU 2008) 'Reflections on further reform of the Common Fisheries Policy' which noted related to objectives that the CFP has too many objectives mixing long-term and short-term concerns and social, economic and environmental factors with no clear order of priority. These objectives partly conflict with each other and generate a bias in the decision-making process.'

In identifying reform options the working document (EU 2008) identifies that future CFP objective have to be focused and prioritised in order to enable accountability and clear guidance. The long term ecological sustainability of fisheries must be the first priority because the past development of the CFP has demonstrated that healthy fish stocks and healthy marine ecosystems are a sine qua non for an economically and socially healthy fisheries sector. The objectives must be sufficiently specific to enable accountability and monitoring of performance.

The Green paper (EU 2009) consists largely of an identification of current failures and a series of questions to prompt debate and stakeholder input to the reform process. With respect to objectives it clearly outlines the problem of having multiple, potentially conflicting objectives:
'The economic and social viability of fisheries can only result from restoring the productivity of fish stocks. There is, therefore, no conflict between ecological, economic and social objectives in the long term. However, these objectives can and do clash in the short term, especially when fishing opportunities have to be temporarily reduced in order to rebuild overexploited fish stocks.'

The commission released a working document on the results of the CFP consultation in April 2010 (EU 2010) that included the following section on prioritisation of objectives :
"The EP [European Parliament] and others are against a priori prioritisation of objectives. Most catching industry (and some regional authorities) insist on a balance between the three pillars - with job creation as an objective in its own right, and trade unions additionally emphasize social aspects. MS [Member State] opinions range from ecological sustainability at the core to equal weight on the three pillars of sustainability. Some MS focus on fisheries for food supply and food security (as a new objective), others on fisheries as a source of employment in coastal communities. Environmental NGO see ecological sustainability as the core of the policy, linking the CFP with the wider maritime policy and the Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC). Traders, retailers and some processing industries prioritize ecological sustainability.

There is broad consensus that maximum sustainable yield (MSY) must be among the targets (as stated in COM 2006) on the World Summit on Sustainable Development declaration. The EP and catching industry generally look for a flexible timeframe for implementation particularly in mixed fishery, considering MSY as a direction rather than a specific target, and they consider that social and economic aspects should also be included in 'sustainable'. Environmental NGO on the other hand worry that MSY may not be precautionary in all cases, claiming that wider impacts of fisheries on the ecosystem are part of 'sustainable' in MSY.

Many contributions insist on minimization or elimination of discards as an important aim for ecological sustainability, although some contributions maintain that discards are inherent to mixed fisheries.

In summary the adoption of CFP reform is envisaged for early 2011 (EU 2010), thus while the existing CFP has seven main objectives, our dealing with these in the MESMA generic framework should be sufficiently flexible to cope with future changes, particularly given that the identification and prioritisation of objectives has been a key part of reform documents to date (EU,2008,2009,2010).

Strategic Environmental Assessment Directive (SEA)

The European Directive 2001/42/EC (EC 2001) on the assessment of the effects of certain plans and programmes on the environment ('Strategic Environmental Assessment', hereinafter the 'SEA Directive') was published in July 2001. The SEA Directive requires certain public plans and programmes (P&P) to undergo an environmental assessment. This is to ensure that environmental consequences of certain P&P are identified and assessed during their preparation and before their adoption.

The high level goal of the SEA Directive is described in its Article 1, which lays down two key aspects: 1. to provide for a high level of protection of the environment and 2. to contribute to the integration of environmental considerations into the preparation and adoption of certain plans and programmes with a view to promoting sustainable development. Article1:
'The objective of this Directive is to provide for a high level of protection of the environment and to contribute to the integration of environmental considerations into the preparation and adoption of plans and programmes with a view to promoting sustainable development, by ensuring that, in accordance with this Directive, an environmental assessment is carried out of certain plans and programmes which are likely to have significant effects on the environment.'

The high level goals of the SEA Directive are ensured by the established basic framework for the assessment of environmental effects of certain P&P. The P&P covered by the Directive are subject to an environmental assessment during their preparation, and before their adoption. This includes the drawing up of an environmental report in which the likely significant effects on the environment and the reasonable alternatives are identified, and the carrying out of consultations (with the public, the environmental authorities, and with other Member States of the European Union (MS) in the case of transboundary impacts). The environmental report and the results of the consultations are taken into account before adoption. Once a P&P is adopted, the environmental authorities and the public are informed and relevant information is made available to them. In order to identify unforeseen adverse effects at an early stage, significant environmental effects of the P&P are to be monitored.
The responsibility of developing a detailed procedure within the basic framework of the SEA Directive is left to the Member States. The Directive is to be either integrated into existing procedures, or new procedures must be developed to comply with the Directive. The SEA Directive has formal and explicit links with the Habitat and EIA Directives, for example the environmental information to be collected and assessed under the SEA Directive also covers the considerations required by the HD.

The criteria for selecting relevant P&P are given in Annex II:
1. The characteristics of plans and programmes, having regard, in particular, to

   – the degree to which the plan or programme sets a framework for projects and other activities, either with regard to the location, nature, size and operating conditions or by allocating resources,
   – the degree to which the plan or programme influences other plans and programmes including those in a hierarchy,
   – the relevance of the plan or programme for the integration of environmental considerations in particular with a view to promoting sustainable development,
   – environmental problems relevant to the plan or programme,
   – the relevance of the plan or programme for the implementation of Community legislation on the environment (e.g. P&P linked to waste-management or water protection).

2. Characteristics of the effects and of the area likely to be affected, having regard, in particular, to
   – the probability, duration, frequency and reversibility of the effects,
   – the cumulative nature of the effects,
   – the transboundary nature of the effects,
   – the risks to human health or the environment (e.g. due to accidents),
   – the magnitude and spatial extent of the effects (geographical area and size of the population likely to be affected),
   – the value and vulnerability of the area likely to be affected due to:
   – special natural characteristics or cultural heritage,
   – exceeded environmental quality standards or limit values,
   – intensive land-use,
   – the effects on areas or landscapes which have a recognised national, Community or international protection status.

The SEA Directive itself does not create new obligations on environmental protection. Objectives can be derived from environmental protection objectives established in international (e.g. Biodiversity Action Plan), Community (e.g. HD) or Member State (e.g. national legislation) law, policy, or other plans and programmes, or from a review of baseline information and environmental problems. The environmental protection objectives should cover at least the 12 environmental factors and issues listed in Annex I, paragraph (f):

> 'the likely significant effects on the environment, including on issues such as biodiversity, population, human health, fauna, flora, soil, water, air, climatic factors, material assets, cultural heritage including architectural and archaeological heritage, landscape and the interrelationship between the above factors.' 'These effects should include secondary, cumulative, synergistic, short, medium and long-term permanent and temporary, positive and negative effects.'

It is useful to express the objectives in the form of targets. Indicators and monitoring procedures (new or already existing monitorings) have to be developed to measure the environmental effects of the plan or programme to show whether they are as predicted, or to help identify adverse effects.

By 2009, all MS have transposed and implemented the SEA Directive in their national planning system. However, very few MS report that they have established monitoring methods or drawn up national guidance on how to establish monitoring indicators (EC 2009). An informative guide on the application of the SEA Directive for those P&P which geographically cover parts of the UK is published by ODPM et al. (2005).

In summary the purpose of the SEA Directive is to ensure that environmental consequences of certain P&P are identified and assessed during their preparation and before their adoption. Therefore, the SEA Directive provides a basic framework and systematic means of identifying, describing, evaluating and reporting on the environmental effects of P&P. To achieve the goal of sustainable development, MS have to develop procedures themselves to determine (amongst other things) environmental protection objectives, indicators and monitoring measures.

1.1.3.5 National and local objectives

As an example of national and local objectives we reviewed in the following sections three examples.

Greek National and local objectives

Greek environmental policy objectives are mainly the result of international obligations and relevant legislation according to the 1971 Ramsar Convention on Wetlands of International Importance, the Birds Directive 79/409/EEC, and the Habitats Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora. Greece, as all other EU member states has the obligation not only to incorporate into national law and implement the European Directives, but also to actively participate in all procedures for further legal and institutional strengthening of environmental protection. However, according to Roumeliotou (2004) a main flaw of the transposition of Article 6 of the Directive into the Greek legal order relates to the issue of environmental impact assessment of plans not connected with or necessary to the management of Natura 2000 sites. More specifically, although article 6.3 of the Directive asks explicitly for the assessment of "plans" and not only of projects, the transposing provision restricts its field of application only to the assessment of projects or activities, according to existing EIA national legislation. This is aggravated by the fact that Greece has not fulfilled its obligation to integrate the protection of nature into sectoral policies and legislation, e.g. on land-use planning. Hence, the introduction of "strategic impact assessment" obligation into the Greek legal order due to the adoption of the relevant EU Directive 2001/42/EC may bridge this gap.

A remarkable area of the Greek territory has been included in the European Ecological Network Natura 2000; Greece includes at its National List 163 Special Protection Areas (SPAs) and 239 Sites of Community Importance (SCIs) respectively according to Directives 79/409/EEC and 92/43/EEC. In fact, under the Habitats Directive, which was incorporated into Greece's national legislation by the Joint Ministerial Decision (JMD) 33318/3028/1998 (Gov. Gaz. 1289B/28-12-1998), Greece has one bio-geographical region (mediterranean), and according to the Article 17 National Summary for Greece, 80 % of the 'Marine-Mediterranean' habitat types listed in Annex I to Habitats Directive in Greece have an 'unfavorable-inadequate' status, while the remaining 20 % have an 'unknown' status (COM, 2008).

Following JMD 33318/3028/1998, Greece has enacted specific statutory measures for only two of the sites included in its national list of SCIs, i.e the National Marine Park of Alonissos – Northern Sporades, regarding the protection of the Annex II priority species Mediterranean monk seal (*Monachus monachus*) and the National Marine Park of Zakynthos, regarding the protection of the Annex II priority species loggerhead turtle sea (*Caretta caretta*).

The National Marine Park of Zakynthos (NMPZ) was established for the protection of the loggerhead turtles- one of Europe's most endangered marine species. In particular, the Bay of Laganas in Zakynthos Isle has proved to be the single most important nursery for this species in the whole Mediterranean. In 1984 a presidential decree was issued introducing regulations on tourists and residential development, as well as land use behind nesting beaches. Strict regulations are in force to regarding boats, mooring and fishing and water sports within designated zones. The Park consists of seven terrestrial zones and two maritime zones. Founded in 1992, the National Marine Park of Alonnisos and the Northern Sporades is home to striped, bottlenose and common dolphins, and the Mediterranean monk seal. The monk seal is considered as the most endangered mammal in Europe, with less than 400 surviving worldwide, 30 of which are found living and breeding around the islands north of Alonnisos. In both cases, there are Presidential Decrees that include management guidelines, rules and restrictions that satisfy the provisions of article 6(1) of the Habitats Directive. The Boards of the Management Bodies of the parks are established following specific JMDs, while sites that are also designated as RAMSAR Wetlands (GR2310001 Acheloos Delta, GR2310002 Mesolongi Lagoons and GR2320001 Kalogria Lagoon) are under the supervision of official Management Bodies, and enjoy some degree of legal protection and management measures. According to Roumeliotou (2004) an awkward situation has developed by the fact that the JMDs setting up the Management Bodies of the areas are in force, while the instruments containing the legal designation, zoning and overall regulation of each area are still in the draft stage at best, while the relevant temporary JMDs have expired. Moreover, the Management Bodies have not been able to work effectively at least due to lack of approved internal regulations and the ensuing inability to absorb the limited funds that were made available to them by the Greek state (basically EU funds).

Finally, under the CFP, the type of measures that can be established to limit fishing mortality and the environmental impact of fishing activities include the establishment of "zones and/or periods in which fishing activities are prohibited or restricted including for the protection of spawning and nursery habitats" and "specific measures to reduce the impact of fishing activities on marine eco-systems and non-target species. In Greece beside the CFP, there is a variety of measures applied through specific presidential orders or ministerial decisions and aiming at the sustainable management of fisheries resources, which include closed areas and seasons, and minimum depths and distances from shore for fishing. Hence, in Greek waters there are fisheries reserves (mainly enclosed bays) which constitute areas where fishing is fully or partially forbidden, while there are also temporal restrictions prohibiting fishing during a certain period of the year (eg. there is a four-month trawl ban during summertime aiming to protect spawners of demersal fish stocks).

UK National and local objectives

On 20 April 2009, the UK Government, Welsh Assembly Government, Northern Ireland Executive and Scottish Government published their joint High Level Objectives for the UK marine area - Our seas – a shared resource: High Level Marine Objectives. This followed a consultation carried out in 2008.The High Level Marine Objectives take forward the UK vision for the marine environment of *"clean, healthy, safe, productive and biologically diverse oceans and seas."* They set out the outcomes that all UK Administrations are seeking to achieve in the UK marine area, as well as providing a vision of what success will look like. They will steer the development of policies to achieve sustainable development in the marine area and, more widely, will help inform and educate the public, business and voluntary sectors. They will be used to underpin the development the joint Marine Policy Statement, which is provided for in the Marine and Coastal Access Act. UK Administrations will be working together to develop the Marine Policy Statement, which is expected to be completed in 2011 (see UK national objectives through recent publications and the website of the Department for Environment, Food and Rural Affairs www.defra.gov.uk).

Within the High Level Marine Objectives [ref] document UK objectives are expressed in terms of five principles of sustainability

- Infrastructure is in place to support and promote safe, profitable and efficient marine businesses.
- The marine environment and its resources are used to maximise sustainable activity, prosperity and opportunities for all, now and in the future.
- Marine businesses are taking long-term strategic decisions and managing risks effectively. They are competitive and operating efficiently.

- Marine businesses are acting in a way which respects environmental limits and is socially responsible. This is rewarded in the marketplace.
- People appreciate the diversity of the marine environment, its seascapes, its natural and cultural heritage and its resources and act responsibly.
- The use of the marine environment is benefiting society as a whole, contributing to resilient and cohesive communities that can adapt to coastal erosion and flood risk, as well as contributing to physical and mental wellbeing.
- The coast, seas, oceans and their resources are safe to use.
- The marine environment plays an important role in mitigating climate change.
- There is equitable access for those who want to use and enjoy the coast, seas and their wide range of resources and assets and recognition that for some island and peripheral communities the sea plays a significant role in their community.
- Use of the marine environment will recognise, and integrate with, defence priorities, including the strengthening of international peace and stability and the defence of the United Kingdom and its interests.

Living within environmental limits

- Biodiversity is protected, conserved and, where appropriate, recovered, and loss has been halted.
- Healthy marine and coastal habitats occur across their natural range and are able to support strong, biodiverse biological communities and the functioning of healthy, resilient and adaptable marine ecosystems.
- Our oceans support viable populations of representative, rare, vulnerable, and valued species.

Promoting good governance

- All those who have a stake in the marine environment have an input into associated decision-making.
- Marine, land and water management mechanisms are responsive and work effectively together for example through integrated coastal zone management and river basin management plans.
- Marine management in the UK takes account of different management systems that are in place because of administrative, political or international boundaries.
- Marine businesses are subject to clear, timely, proportionate and, where appropriate, plan-led regulation.
- The use of the marine environment is spatially planned where appropriate and based on an ecosystems approach which takes account of climate change and recognises the protection and management needs of marine cultural heritage according to its significance.

Using sound science responsibly

- Our understanding of the marine environment continues to develop through new scientific and socio-economic research and data collection.
- Sound evidence and monitoring underpins effective marine management and policy development.
- The precautionary principle is applied consistently in accordance with the UK Government and Devolved Administrations' sustainable development policy.

The UK Marine Monitoring and Assessment Strategy (UKMMAS) sets the framework within which we collaborate in assembling the evidence necessary to monitor progress towards our vision. Indicators and contributory objectives are being developed within the various UKMMAS groups and will address some aspects of the high level objectives outlined above.

Straights of Sicily National and local objectives

The Strait of Sicily connects the Eastern and the Western Mediterranean basins and is shared between three countries: Italy, Malta and Tunisia, of which only the first two are in the EU. Natural resources and man-made structures inevitably present cross-border issues.

The Straits of Sicily host a number of different coastal and shallow-water priority habitats that deserve protection according to the Habitat Directive, like coastal dunes, coastal lagoons and Posidonia beds. The seafloor presents a wide array of different benthic communities, some of which - coralligenous habitats, “maerl” beds and Posidonia beds. (Garofalo et al 2004).

The Habitats Directive (Council Directive 92/43/EEC) is one of the most important and directly applicable regulations of the EU. It has been transposed into Maltese law primarily by the Flora, Fauna and Natural Habitats Protection Regulation, 2003 (LN 311 of 2006). The latter provides the legal framework for designating terrestrial and marine areas of national and international importance in the Maltese Islands.

Two Italian Acts regulate the conservation of the natural environment: no. 979/1982 (Act on the defence of sea) and no. 394/1991 (Act on protected areas): following these laws, 25 MPAs have been created in Italy. Italian MPAs range in size from 20 to more than 50,000 hectares in total surface area. They include one or more no-take/no-access zones formally defined as ‘A zones’ according to Italian law, surrounded by buffer zones (defined as ‘B and C zones’, where restrictions to human uses, including fishing, become progressively more lax) (Villa et al 2002; Guidetti et al 2008). All Italian MPAs are under the control of the national Ministry of the Environment is which then delegates responsibility for management.

Two MPAs occur in the Strait of Sicily: Isole Egadi and Isole Pelagie. The main operational objectives of these two MPAs are: Conserve biodiversity, Protect habitats and endangered species,, Restock fishery resources, Ensure economic and social welfare growth of small scale fishers and other stockholders

The operational objectives of these Sicilian MPAs meet some qualitative objectives of the MSFD and HD and in particular: biodiversity, seabed integrity, commercial stocks, species at risk and rare habitat.

Under the Common Fisheries Policy, the Italian Decree 24 (March 2009) which concerned the “Adozione dei Piani di Adeguamento dello sforzo di pesca” of Italian MiPAAF, brought about a reduction of Sicilian fishing fleet during 2008-2013. This reduction amounts to 25%, 3% and 10% for trawlers, purse-seiners and artisanal fisheries respectively targeted at demersal fishing. The reduction acts trough a one-off vessel scrapping scheme, funded by the reg. (CE) 1198/2006. A long term management plan (LTMP) for 2008-2013 has been agreed for Italian distant trawlers operating in the Strait of Sicily (GSA 12, 13, 14, 15 and 16).

Italy and Malta are subjected to the COUNCIL REGULATION (EC) No 1967/2006 of 21 December 2006 concerning management measures for the sustainable exploitation of fishery resources in the Mediterranean Sea, amending Regulation (EEC) No 2847/93 and repealing Regulation (EC) No 1626/94. This regulation protects sensitive coastal habitats such as the seagrass beds, the “maerl” bootms and the coralligenous habitas (art. 4). An important spatial management measure, aimed to protect both sensitive areas and essential fish habitat in coastal bottoms, is due to the art. 13 stated that the use of towed gears is prohibited within 3 nautical miles of the coast or within the 50 m isobath where that depth is reached at a shorter distance from the coast. In coastal bottoms of the Sicilian southern coasts (GSA 16), due to the morpho-batimetry of the area, the 3 miles distance is in force. Together with the protection of seagrass beds, this measure is furthermore effective to protect the essential fish habitats of commercial species, such as red mullets or pandoras, having their nurseries in shallow coastal bottoms. Unfortunately the “maerl” bottoms which are in offshore bottoms on Adventure and Malta Banks are not protected by the art. 13 and needs more precise conservation measures. Furthermore, the art. 13 is not effective in protection of nurseries of very important target species of trawling in the Strait of Sicily such as the deep water rose shrimp (Parapenaeus longirostris) and the hake (Merluccius merluccius). The nurseries of these species were identified, being in the eastern side of the Adventure and the Malta banks. These areas cover about 1000 km2,and and show persistent patterns of highest concentration of recruits. Therefore a year around trawling ban was proposed inside the LTMP in order to improve the fishing pattern of demersal fisheries in the Strait of Sicily.

Another spatial management measure is the prohibition to use towed gears in bottom deeper than 1000 m, in order to protect Mediterranean deep water species which are supposed very sensitive to trawling impacts.

Moreover, the Maltese Islands are surrounded by a 25 nautical miles (nm) fisheries management zone, where fishing effort and capacity are being managed by limiting vessel sizes, as well as total vessel engine powers (EC 813/04; EC 1967/06). Trawling is allowed within this designated conservation area, however only by vessels not exceeding an overall length of 24m and only within designated areas (Camilleri 2007). Such vessels fishing in the management zone hold a special fishing permit in accordance with Article 7 of Regulation (EC) No 1627/94, and are included in a list containing their external marking and vessel's Community fleet register number (CFR) to be provided to the Commission annually by the Member States concerned. Moreover, the overall capacity of the trawlers allowed to fish in the 25nm zone cannot exceed 4 800 kW, and the total fishing effort of all vessels is not allowed to exceed an overall engine power and tonnage of 83 000 kW and 4 035 GT respectively. The fishing capacity of any single vessel with a license to operate at less than 200m depth cannot exceed 185 kW.

With regards to the conservation of commercially important fish stocks, the Maltese Fisheries Conservation and Management Act (2001) provides for the establishment of marine closed areas for the preservation of fish stocks through Article 38(2). Moreover, the more recent regulation EC 1967 / 2006 states that EU member states are required to submit information relevant to the establishment of fishing protected areas, and to the possible management measures to be applied therein, where the protection of nursery / spawning grounds or of the marine ecosystem from harmful effects of fishing is needed. In order to fulfil this obligation, an investigation into the spatial distribution of both nursery and spawning areas of commercially important demersal species was recently carried out by MRRA. A site suitable for the establishment of a Marine Protected Area was located just outside the Maltese FMZ in international waters. However the study also revealed a number of critical habitats within the FMZ, which would need to be incorporated in a spatial management plan.

1.1.3.6 Summary of operational objectives

Within most of these directives a high-level objective is set out which is then broken down into several operational objectives. As discussed in the introduction, high-level objectives are those that are set based upon societal or cultural values whereas operational objectives are those which can be used to develop more concrete targets, limits and measures which will work to achieve the high-level goals.

There is also a legislative scale upon which more local and detailed policies are drafted which aim to ensure that the requirements of a higher level directive are met. This is seen clearly in the Straits of Sicily where the Flora, Fauna and Natural Habitats Protection Regulation, 2003 (LN 311 of 2006) was established to provide a legal framework for which The Habitats Directive (Council Directive 92/43/EEC) can be implemented in the local area. It should therefore be remembered in creation of the MESMA framework (WP2) that even at a more local scale, operational objectives from a larger perhaps international directive will also be applicable.

1.1.4 PERFORMANCE MEASURES FOR OPERATIONAL OBJECTIVES IN PRACTICE

Performance measures are those attributes and criteria used in practice to measure the above defined operational objectives and to determine whether or not they have been achieved. Indicators and benchmarks can be considered to be a subset of performance measures where there is a more precise definition of what is to be measured and definition of the quantitative values that constitute good/bad or improving/deteriorating.

The aim in this review and in the MESMA generic framework is not to define a precise set of indicators and benchmarks, that would be too ambitious a task (for example indicators for the MSFD have yet to be defined) and lead to an insufficiently generic framework. Instead we will summarise a description of the attributes of an operational objective which have to be measured. For instance the recommendations of the ICES/JRC task groups on the MSFD descriptors are structured by recommendations on the attributes and criteria for the respective descriptor.

There are two recent texts that describe frameworks for the measurement of the achievement of operational objectives in the marine environment. One is Pomeroy et al. (2004) ‘How is your MPA doing? A Guidebook of Natural and Social Indicators for Evaluating Marine Protected Area Management Effectiveness’, and the other is IOC(2006) ‘A handbook for measuring the progress and outcomes of integrated ocean management’. In both texts indicators are grouped into 3 classes: biophysical, socio-economic and governance in Pomeroy et al. (2004), and ecological, socioeconomic and governance in IOC (2006). The objectives and indicators for governance in each text are shown in tables 1.1.3 and 1.1.4. For the MESMA generic framework we will develop similar tables to these, learning from both. The MESMA framework will, however be different, in that the operational objectives will be taken directly from the relevant legislation as summarized in 1.1.3. The MESMA tables of performance measures and operational objectives linked to legislation will allow case study participants and future marine scientists to link their monitoring activities directly to the relevant legislation.

Table 1.1.3. Governance indicators and their relation to operational objectives, as described in IOC(2006).

Figure 3-1 Matrix of relevance of ICOM governance indicators to goals and objectives
(note: for simplification of the matrix, the parameters of the indicators have not been included; see Table 3-1)

Goal	Objective	Coordination mechanism G1	Legislation G2	Environmental assessment G3	Conflict resolution mechanism G4	Integrated management plans G5	Active management G6	Monitoring and evaluation G7	Human, technical, and financial resources G8	Inputs from scientific research G9	Stakeholder participation G10	NGO and CBO activity G11	Education and training G12	Technology G13	Economic instruments G14	Sustainable development strategy G15
Ensuring adequate institutional, policy and legal arrangements	Ensuring the coordination and coherence of administrative actors and policies	■	■	■	■	■	■									■
	Supporting integrated managements through adequate legislation and regulations		■	■												
	Assessing the environmental impacts of policies, plans, programmes and projects	■	■		■											
	Resolving conflicts over coastal space and resources					■	■									
Ensuring adequate management process and implementation	Managing the coastline through integrated plans							■		■						
	Implementing and enforcing ICOM plans and actions								■							
	Routinely monitoring, evaluating and adjusting of ICOM efforts							■		■						
	Supporting ICOM through sustained administrative structures									■	■	■				
Enhancing information, knowledge, awareness and participation	Ensuring the management decisions are better informed by science															
	Ensuring sustained support from engaged stakeholders									■	■	■				
	Ensuring NGO and CBO involvement										■	■	■			
	Ensuring adequate levels or higher education and professional preparation for ICOM								■				■			
Mainstreaming ICOM into sustainable development: economic instrument mainstreaming	Enabling and supporting ICOM through technology, including environmentally-friendly technology												■	■		
	Incorporating economic instruments into coastal management policies														■	
	Mainstreaming coastal and ocean management into sustainable development															■

Table 1.1.4. Governance indicators and their relation to operational objectives, as described in Pomeroy (2004).

Figure 4 Governance goals, objectives, indicators

Governance goals (n=5) and objectives (n=21) commonly associated with MPA use

GOAL 1	**Effective management structures and strategies maintained**
1A	*Management planning implemented and process effective*
1B	*Rules for resource use and access clearly defined and socially acceptable*
1C	*Decision-making and management bodies present, effective, and accountable*
1D	*Human and financial resources sufficient and used efficiently and effectively*
1E	*Local and/or informal governance system recognised and strategically incorporated into management planning*
1F	*Periodic monitoring, evaluation, and effective adaptation of management plan ensured*
GOAL 2	**Effective legal structures and strategies for management maintained**
2A	*Existence of adequate legislation ensured*
2B	*Compatibility between legal (formal) and local (informal) arrangements maximized or ensured*
2C	*National and/or local legislation effectively incorporates rights and obligations set out in international legal instruments*
2D	*Compatibility between international, national, state, and local rights and obligations maximized or ensured*
2E	*Enforceability of arrangements ensured*
GOAL 3	**Effective stakeholder participation and representation ensured**
3A	*Representativeness, equity, and efficacy of collaborative management systems ensured*
3B	*Resource user capacity effectively built to participate in co-management*
3C	*Community organizing and participation strengthened and enhanced*
GOAL 4	**Management plan compliance by resource users enhanced**
4A	*Surveillance and monitoring of coastal areas improved*
4B	*Willingness and acceptance of people increased to behave in ways that allow for sustainable management*
4C	*Local ability and capacity built to use resources sustainably*
4D	*User participation in surveillance, monitoring, and enforcement increased*
4E	*Application of law and regulations adequately maintained or improved*
4F	*Access to and transparency and simplicity of management plan ensured and compliance fostered*
GOAL 5	**Resource use conflicts managed and reduced**
5A	*User conflicts managed and/or reduced: 1) within and between user groups, and/or 2) between user groups and the local community or between the community and people outside it*

Summary table

How the governance indicators relate to the common goals and objectives

	G1	G2	G3	G4	G5	G6	G7	G8	G9	G10	G11	G12	G13	G14	G15	G16
GOAL 1																
1A			●													
1B				●												
1C		●														
1D						●										
1E																
1F																
GOAL 2																
2A					●											
2B																
2C					●											
2D																
2E					●								●	●		
GOAL 3																
3A												●				
3B											●					
3C								●								
GOAL 4																
4A						●							●	●	●	
4B									●				●			●
4C										●						
4D													●			
4E				●	●											
4F																
GOAL 5																
5A	●															

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1.2 MPAs, MARINE RESERVES AND MARINE SPATIAL MANAGEMENT

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1.2.1 GENERAL INTRODUCTION

Marine protected areas (MPAs) are a well-established management tool to manage impacts on marine values, incl. biodiversity, cultural heritage, scientific reference sites, commercial fish populations and habitats. However, applied approaches and accompanying needs in relation to e.g. monitoring and evaluation are extremely variable, depending on the specific objectives that drive the MPA processes and the management focus.

MPAs cover a highly diverse range of marine spatial designations involving different types of management strategies with a wide range of management objectives. MPAs may be established to protect sensitive species and habitats from certain pressures and may exclude some or all human uses to achieve this goal. MPAs have often been cited as a means to achieve marine ecosystem based management (EBM) or ecosystem based fisheries management (EBFM). Halpern et al. (2010) conclude that MPAs can never be a substitute for EBM, but will in almost all cases be a necessary component. MPAs may also be established for scientific purposes, i.e. to maintain scientific reference areas that can be used to monitor the surrounding environment. Coastal tourism (e.g. diving, snorkeling) may be a driver of MPA establishment in some regions such as the Mediterranean, while in other European regions tourism may have a less prominent role. In all cases the design, function and management of MPAs are to a very large extent dependent on various drivers and contexts that ultimately determine the goals and objectives of each MPA.

The current global movement towards integrated marine spatial planning (MSP) and management has put a spotlight on the interlinkages between MPAs and MSP, with stakeholders often assuming that MSP and MPAs are interchangeable concepts. MPAs, however, are not synonymous with MSP (e.g. Symes 2005; Blæsbjerg et al. 2009). MSP has a broader remit and provides an overall spatial framework for ecosystem based management of maritime activities, whereas MPAs are one of the management tools within that planning framework, i.e. providing a means for emphasised protection of features and processes in a given ecosystem that merit site specific management measures.

In the following, central issues will be described relating to technical aspects of MPA designation and management as well as the policies that are drive MPA processes in Europe and the governance mechanisms that regulate them. Furthermore, the role of MPAs in relation to wider MSP will be reviewed and discussed.

1.2.2 DEFINITIONS

Marine protected areas (MPAs) and MPA networks have come to mean many different things to many different people. Marine reserve, sanctuary, marine park, conservation area and box, etc. are but a few examples of terms that have become more or less synonymous with MPAs. As a result, the term MPA has in effect become generalized and used to describe anything from completely closed, no-take marine reserves/sanctuaries to areas with a multitude of uses, including technical measures for fisheries management.

The MESMA project follows the IUCN (1988) definition of a marine protected area, i.e. 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 or all of the enclosed environment.

This definition in principle has the scope to include almost any area-based marine management measure. Nonetheless, this has a number of advantages, i.e. reduced sector dependence in a time when integration is an overarching aim in marine policies as well as making room for e.g. voluntary agreements as the basis for protected area designation and management.

A new definition for the general term “protected area” was developed by IUCN in 2008 (Dudley 2008): A clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values. IUCN expects this definition to supersede the above definition of marine protected area (MPA News 2008). In connection with the development of the new definition, IUCN (2008) also published guidelines to assist in the application and interpretation of the term and IUCN’s protected area categories. A number of MPAs individually designated within the same marine region does not automatically constitute a MPA network. The term network (as opposed to e.g. a set of MPAs) implies that there are ecological connections between designated MPAs that serve to enhance the effectiveness of the individual MPAs, i.e. by optimising the use of known ecological characteristics of habitats and populations within, such as known dispersal corridors, sources and sinks (e.g. Crowderet al 2000; Planes et al 2008, 2009; Christensen et al 2009; Gaines et al 2010; McCook et al 2010). Optimally designed networks enhance cumulative interactions that result in networks that are more effective than the sum of the individual MPAs (Gaines et al 2010).

1.2.3 Policy drivers for MPAs

The majority of Europe’s coastal states have national strategies and legislation that underpin the establishment of marine protected areas in their waters. These can be driven by national nature conservation needs, a need to develop sustainable coastal industries such as tourism/diving or by overarching obligations in relation to international treaties and conventions. Among EU Member States, however, much of this development is currently driven by the national implementation of EU directives such as the Marine Strategy Framework Directive and especially the Habitats and Birds directives.

1.2.3.1 EU policies

1.2.3.1.1 Natura 2000: the Habitats and Birds Directives

Two major drivers of establishment of protected areas in the waters of EU Member States are the Birds Directive (79/409/EEC) and (especially) the Habitats Directive (92/43/EEC). Together these Directives serve as a legally binding basis for the establishment of a set of terrestrial and marine protected areas, collectively known as Natura 2000.

The Birds Directive

*Council Directive 79/409/EEC on the Conservation of wild birds*is more commonly known as the Birds Directive (BD). It constitutes the legal framework needed for the EU as a whole to meet its obligations for bird species under the Bern and Bonn Conventions, i.e. providing a framework for the conservation and management of, and human interactions with, wild birds in Europe. It is the responsibility of each Member State to implement national legal mechanisms to achieve a number of broad objectives of the Directive.

The overall objective of the BD is to achieve a*favourable conservation status*(see below) for all of the bird species included in the annexes of the Directive. Among other provisions, the Directive prescribes the identification and classification of Special Protection Areas (SPAs) for listed rare or vulnerable species, as well as for all regularly occurring migratory species. As a result, Member States have designated SPAs, which include marine areas that serve as habitat for sea birds during all or parts of the year. Many of these sites have previously been under some form of protection through national nature conservation (or hunting) legislation or the Ramsar Convention.

One of the challenges that Member States currently face is the evaluation of different threats to birds in marine sites from e.g. drowning in gillnets (see e.g. the EMPAS project (ICES 2008)), overfishing of prey species (fish & shellfish) or collisions with wind turbines and other permanent structures.

The Habitats Directive

The Habitats Directive (HD), formally known as *Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora*, was adopted by the EC in 1992 to meet the obligations of the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention). The HD is accompanied by annexes which list 189 habitats (Annex I) and 788 species (Annex II) to be protected in the European Community. Of these, only a small fraction are *marine* species and habitats. The following are the 8 habitat types included in the HD: Sandbanks which are slightly covered by sea water all the time; Posidonia beds; Estuaries; Mudflats and sandflats not covered by seawater at low tide; Coastal lagoons; Large shallow inlets and bays; Reefs; Submarine structures made by leaking gases. Examples of marine species are e.g. harbour porpoise and other cetaceans, several seal species, sea lamprey, twait shad and allis shad. There are no commercially important fish species listed in the Directive. Annex IV of the HD lists a number of animal and plant species that require protection in all areas where they are found (i.e. beyond SAC boundaries). For example, harbour porpoises are listed in both Annexes II and IV.

Article 3 of the HD requires the establishment of a European network of important high-quality conservation sites that will make a significant contribution to conserving habitats and species. These sites are called Special Areas of Conservation (SACs) and in marine areas these are, pending effective management, in many aspects synonymous with the traditional concept of marine protected areas (MPAs).

Favourable conservation status

Both the HD and the BD have the overall aim to maintain or restore natural habitats and species of wild fauna and flora of community interest at a *favourable conservation status*.

For habitats this term implies that the natural range of a habitat is stable or increasing, the species structure and functions which are necessary for its long term maintenance exist and are likely to continue to exist for the foreseeable future, and the conservation status of its typical species is favourable. For habitats, the main elements for which Member States must monitor and report conservation status are: range, area, structure and function and future prospects.

In order for a species to have a favourable conservation status, it must be maintaining itself and its natural range on a long-term basis as a viable component of its natural habitats and there must be and continue to be a sufficiently large habitat to maintain its populations on a long-term basis (ICES 2010). For species, the elements to monitor and report are: range, population, habitat (extent and condition) and future prospects.

Special Areas of Conservation: site selection

When Member States designate SACs (and SPAs), they initially propose Sites of Community Importance (pSCIs) to be included in the Natura 2000 network. These proposed sites and underlying documentation are then delivered to the EC. The EC then evaluate the proposals in light of their importance on an EU scale. Both of these assessments are carried out based on criteria described in the HD Annex III. Selected sites are included in a List of Sites of Community Importance, and from this stage Member States are required to comply with the provisions of *national* legislation to designate all of these sites as SACs within a period of six years.

The HD distinguishes two cases in which the conservation status of habitats and species is expected to be assessed. According to the HD Member States have to monitor and report every six years to the EC on the conservation status of habitats and species regardless of whether they are situated inside a Natura 2000 site or outside. If the overall conservation status of a species or habitat in a given biogeographical region is not favourable, the conservation measures must be improved (ICES 2010).

Regulation of plans and projects within Natura 2000 sites

In connection with any plan or project not directly connected with or necessary for the management of the Natura 2000 site, Member States must evaluate whether disturbance might occur, inside or outside of a Natura 2000 site, that could be significant in relation to the objectives. If such plans or projects, either individually or in combination with other plans or projects (i.e. cumulative effects), are deemed likely to have a significant effect, the plan or project shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives (i.e. a proper impact assessment). A plan or project can then only go forward after having confirmed that it will not adversely affect the integrity of the site concerned and, if appropriate, after going through a public hearing process.

Fisheries and Natura 2000

Among those activities that are likely to be most affected by wide-scale, pan-European implementation of the Natura 2000 Directives and associated protected areas is the fishing sector. As commercial fishing is one of the most spatially widespread of those maritime activities that have impacts on marine species and habitats, fishing areas and gears employed will inevitably need to be regulated, including spatial and/or temporal restrictions.

It is the responsibility of Member States to designate protected areas in the marine area under national jurisdiction and to establish conservation measures in order to protect vulnerable and endangered habitats and species. Member States must also provide monitoring and control measures and a specific control and inspection program to implement those measures.

EU Member States are free to take measures for conservation and management of stocks in waters within their 12 nautical miles, under the conditions mentioned above, if they apply solely to their own fishing vessels. If the proposed measures are liable to affect the vessels of other Member States, these proposed measures can only be adopted after the Commission, other Member States and the Regional Advisory Councils (RAC) concerned have been consulted on a draft of the measures.

Beyond nearshore waters (12 nautical miles), however, any Natura 2000 related management measures proposed that might lead to a limitation of fishing opportunities must either be implemented through the Common Fisheries Policy or via bilateral agreements between those Member States that have fisheries in a given area.

1.2.3.1.2 MPAs and the Marine Strategy Framework Directive

The EU Marine Strategy Framework Directive (MSFD) (see Ch. 1.1) includes provisions to achieve*good environmental status*(GES) for 11 different descriptors that aim to cover most elements and impacts relating to the marine environment (e.g. seafloor integrity, energy/noise, food webs, commercial fish populations, litter, hydrodynamics, biodiversity). These descriptors are accompanied by a number of indicators to monitor and evaluate the state of a given descriptor in a region.

The MSFD specifically suggests both MPAs and MSP as possible tools to aid in the achievement of GES. In comparison to e.g. the Habitats Directive, which aims to protect a limited list of specific habitats and species (none of which are commercially important as such), the MSFD addresses all marine waters as a whole. As a result, the MSFD in effect may require EU Member States to protect additional sites if all prevailing habitats, species and biotopes are not currently represented in existing MPAs. The inclusion of the descriptor relating to commercial fish populations will likely be a driver of a more integrated approach to MPA design and implementation, i.e. one where commercial fish species and their habitats may be included in the overarching objectives of the protected areas.

1.2.3.1.3 MPAs and the EU Common Fisheries Policy

It could be argued that measures aimed at reducing access and fishing effort, as well as restrictions on catch and gear to protect e.g. juvenile fish species are effectively applications of the same principles which underpin MPAs from a fisheries management perspective (PROTECT 2006). The European Common Fisheries Policy (CFP) has on numerous occasions, and with varying degrees of success, implemented spatial measures and gear restrictions in European seas. These often fall under the term “closures” or “boxes”, such as e.g. the North Sea cod closure, the Plaice Box, the Shetland Box, etc. (PROTECT 2006). Such decisions are informed by advice from the International Council for the Exploration of the Sea (ICES) and the ECs Scientific, Technical and Economic Committee on Fisheries (STECF).

In a review carried out by the PROTECT project (PROTECT 2006), it became apparent that the majority of the closed areas established as fisheries management measures in e.g. the North Sea had been established without formulating overall objectives or establishing reference conditions through baseline monitoring efforts. In addition, most of the sites had inappropriate monitoring programs as well as a number of derogations that made measurement of effectiveness impossible. In one case (the North Sea cod closure), the closed area was set up without taking account of displacement of fishing effort. As a result, the closed area which was in effect for a few months, may actually have done more harm than good due to fishing vessels moving to previously unfished areas during the short time the closure was in place (Rijnsdorp et al. 2001). In a separate review by STECF (SEC(2007)) of fisheries technical measures, it was found that most of the specific closures had been established without any clear objectives, i.e. making it very difficult to evaluate the effectiveness, regardless of the amount of evidence that might be available. One of the main recommendations by the group was: *...when a closed area is established, explicit consideration* (should) *be given to its objectives and ways of measuring whether or not those objectives have been met.*

The Common Fisheries Policy includes measures to protect sensitive species and habitats from the impacts of fishing. As a result, there are a number of examples where the CFP has been the key instrument in the protection of deep sea habitats and species (such as corals) in e.g. Irish waters (EC 2007) as well as in the waters off the Azores, Madeira and the Canary Islands (EC 2004) from damage from fishing activity.

1.2.3.2 Co-evolving nature of MPA and MSP policies/practices – which is driving which?

The development of marine spatial planning (MSP) systems has been a relatively new phenomenon, with most MSP-focused policy processes being initiated during in the last decade, although earlier examples of MSP can be dated back to the late 1970s with the launching of the first zoning plan for the Great Barrier Reef Marine Park (Day 2002; Douvere 2008). Some elements of MSP, *e.g.* a cross-sectoral management approach and linkages between terrestrial and marine environments, have been applied to coastal management through Integrated Coastal Zone Management (ICZM) initiatives from the 1970s (Douvere 2008). Most early examples of MSP are focused on establishing MPAs or MPA networks (Douvere 2008). The concept of MPA networks was first mentioned by McManus (1994) and was further explored by Roberts (1997a), who investigated patterns of connectivity in Caribbean coral reefs and the implications for designing networks of interdependent marine reserves. However, developing systems of MSP is increasingly seen as an integrated and holistic planning and management approach and a practical strategy to implement the ecosystem approach in the management of ocean resources (Rosenberg and McLeod 2005; Crowder et al. 2006; Douvere 2008; Maes 2008). The ecosystem approach to nature conservation first emerged from the ecological studies of terrestrial national parks in the United States in the earlier 1930s (Grumbine 1994). The centrality of the ecosystem approach in guiding the planning and management of the oceans became clear when studies in a number of fields over the past two decades address the need for reforming and integrating existing management efforts, in order to sustain ecosystem functions services. These include the recognition of the need for ecologically coherent MPA networks (Roberts 1997a; Roberts et al. 2003; Agardy 2005; IUCN-WCPA 2008), the impacts of fishing on marine ecosystems and the failures of conventional fisheries management regimes in addressing such broader ecosystem concerns (Bostford et al. 1997; Pauly 1998; Pauly 2005), the need for looking into the social dimensions and governance issues in managing the 'peopled seascape' (Kelleher et al. 1995, 1996; Mascia 2003; Jones 2006, 2009), and the impacts of climate changes on marine ecosystems and its implications for ocean management (Hayward 1997; Hughes et al. 2003; Cheung et al. 2008, 2009).

At the international level, there has been a clear trend of evolution in the development of policies and laws in support of MSP (see Maes 2008). The provisions of the 1982 Law of the Sea Convention (UNCLOS) do not refer to MSP, but do not prevent individual states from undertaking such initiatives in their maritime areas. Since the early 1990s international policies and agreements began introducing the ecosystem approach to the conservation of oceans and other ecosystems, which was endorsed by the 1992 Rio Declaration, the 1992 Convention on Biological Diversity (CDB) and the 2002 World Summit on Sustainable Development (WSSD). The most important driver for an ecosystem-based MSP at the international level comes from the CBD (Maes 2008). Decisions adopted by the Conference of Parties (COP) to the CBD support and advocate the main principles underlining an ecosystem-based MSP, such as integrated marine and coastal area management under the jurisdictions of coastal states, the application of environmental impact assessments for individual projects and strategic environmental assessments for the planning of different activities, and the necessity to establish ecologically representative networks of MPAs.

In Europe, one of the most important drivers for MSP is the European legislation on conservation, as part of EU's obligation to implement the 1992 CBD (Douvere 2008). These legislations include the Birds and Habitats Directives, which require EU member states to designate SPAs and SACs that together form a network of protected areas across the Europe, known as Natura 2000 (see section 1.2.3.1.1). Similarly, the development of the EU Integrated Maritime Policy (IMP) started with the process that later led to the establishment of the 'environmental pillar' of IMP –the Marine Strategy Framework Directive (Koivurova 2009). However, the growing interest in offshore renewable energy and the increasing competition for space in recent years in many parts of European's seas seem to be carving out a greater role for MSP not only as a means to address the cumulative environmental impacts from different activities, but also to facilitate and provide a stable regulatory environment for the planning of future activities in existing and new economic sectors. This is reflected in the 2006 Green Paper 'Towards a Future Maritime Policy for the Union: A European Vision for the Oceans and Seas', which envisages that a system of ecosystem-based MSP will play a key role in managing conflicting uses, designating MPAs in addition to the Natura 2000 sties, facilitating the development of offshore renewable energy, and reducing uncertainty in making investment decisions in various marine sectors. It is also stated in the 2006 Green Paper that a system of MSP should be created under the jurisdiction of the Member States, and that *'it should build on the ecosystem-based approach laid down in the Thematic Strategy for the Marine Environment, but should also deal with licensing, promoting or placing restrictions on maritime activities'.* The IMP was adopted by the EU in 2007, and includes the following five action areas:

- Maximising the sustainable use of the oceans and seas
- Building a knowledge and innovation base for the maritime policy
- Delivering the highest quality of life in coastal regions
- Promoting Europe's leadership in international maritime affairs
- Raising the visibility of maritime Europe

Actions in all areas will be guided by the principles of subsidiarity and competitiveness, the ecosystem approach, and stakeholder participation. The MSFD is regarded as being nested within the IMP (Juda 2010), particularly within the first action area on the sustainable use of the oceans and seas. Both MSP and ICZM are stated in the IMP as contributing to *'meeting the commitments deriving from the Thematic Strategy for the Protection of the Marine Environment and provide operators with improved predictability for their planning of future investments'.*

The IMP can be seen as the most comprehensive policy ever adopted by the EU, as it encompasses all possible policy areas that have a stake in the marine environment, and is the first-ever social experiment in integrated ocean policy developed by a supranational organization (Juda 2010). However the IMP also raises major challenges in terms of its implementation. The fundamental underpinning principle of the IMP is an integrated approach to the management of ocean use, which includes both vertical (*i.e.* hierarchical) and horizontal (*i.e.* sectoral) integration. In terms of vertical integration, the EU differs from most other federal states in that it only has exclusive jurisdiction over fisheries, therefore an overwhelming challenge will be to coordinate the actions of member states excising sovereignty within their maritime areas (Juda 2010). In terms of horizontal integration, how the IMP can balance between different sectoral interests and between use and protection remains unclear.

In relation to horizontal integration, one institutional challenge of the EU is the sectoral fragmentation of relevant Directorate-Generals (DGs) in the European Commission. The Commission is divided into departments known as Directorates-General, each covering a specific policy area. Central in relation to MPAs and MSP are DG Environment (DG ENV) and DG Maritime Affairs and Fisheries (DG MARE). DG ENV is responsible for implementation of environmental EC legislation and nature/biodiversity management, incl. marine biodiversity. Central policies of DG ENV include the Habitats & Birds Directives, the Water Framework Directive and most recently the Marine Strategy Framework Directive. DG MARE is responsible for the EC Maritime Policy and the Common Fisheries Policy.

The objectives of these policies are highly interlinked. For instance, Natura 2000 MPAs will likely require some degree of restriction of fisheries activities in order to achieve their objectives. However, the authorization of any restriction of fishing opportunities in European waters is the mandate of the Common Fisheries Policy, which falls under DG MARE. Even though the CFP contains provisions for the protection of sensitive species and habitats from negative effects of fishing, the implementation of appropriate management of Natura 2000 sites has so far proven to be a relatively complex and in some cases turbulent process. The Marine Strategy Framework Directive may provide even more complex examples, as the Directive directly addresses commercial fish populations, sea floor integrity, biodiversity and food webs, all of which are affected by commercial fishing.

Institutional integration (or lack thereof) is crucial on every level of marine spatial management (incl. MPAs) and the complexity herein is multiplied in wider MSP as the number of stakeholders and sectors increases. Case studies within MESMA will provide a venue to explore this complex issue.

MSP-related policies and practices in different European countries also show the multiplicity of MSP goals and objectives (see table below), which include:

- Socio-economic objectives
  - Promoting rational use of ocean space
  - Reducing uncertainty in making investment decisions
  - Minimizing user conflicts
  - Enhancing transparency in the planning and management of sea use
- Ecological objectives
  - Addressing cumulative ecological impacts of different activities on marine ecosystems
  - Promoting sustainable resource management
  - Conserving biodiversity
  - Maintaining or restoring ecosystem functions and services
- Legal and administrative objectives
  - Promoting integrated sea use management
  - Creating new legislative frameworks and government bodies with cross-sectoral regulatory authority in licensing and restricting different uses of marine resources, e.g. the creation of a user permit system in the Netherlands and a Marine Management Organization in the UK to oversee license applications and other related issues

Outside Europe, MSP initiatives have been launched in a few countries. Countries like Australia, Canada and China have already established integrated national ocean policies and legislations, and developed spatial plans in some biogeographical or administrative marine regions (Foster et al. 2005; Li 2006). There are also MSP policies and practices that are focused on establishing systems of zoning and planning for MPA networks, such as the Great Barrier Reef Marine Park (GBRMP) Zoning Plan and the California Marine Life Protection Act (Day 2002; Douvere 2008). The concept of ecologically coherent networks of MPAs with reference to ecological connectivity between different marine habitats was first explored in Roberts (1997a). The implementation of the GBRMP Zoning Plan in 2004 produced the world's largest network of no-take marine reserves, covering 115,395 km^2 (33.4% of the GBRMP), which represents a milestone in the application of ecological theories in designing large-scale MPA networks (Fernandes et al. 2004; McCook et al. 2010).

There are some similarities and differences between MPAs and MSP systems that are created mainly for integrated ocean management. As more and more multiple-use MPAs are being designated, MPAs, like other types of MSP, can also serve multiple ecological, socio-economic, educational and cultural purposes (Jones 2001; Halpern et al. 2010a). Both MPAs and MSP are recognised as practical 'tools' for implementing the ecosystem approach in managing the marine environment, and both represent area-based tools that are recognised as being insufficient in achieving the all-encompassing ecosystem management goals if applied on their own (Douvere 2008; Halpern et al. 2010a). Such view is based on the fact that many stressors to marine ecosystems, such as land-based pollution and climate change, are not spatial in nature (Halpern et al. 2010a). Douvere (2008) recommends that a range of other management measures, including input, output and process control measures, should be used in combination with MSP to manage human activities in the marine environment. A key difference between MPAs and other types of MSP is that well designed MPAs and MPA networks tend to have clearer conservation objectives and guiding principles regarding the relationship between conservation and use, for example, the objective for each zone in the GBRMP has 'conservation' or 'protection' specified as an overriding aspect (Day 2002). There are also fewer economic sectors involved in MPAs in general, and MPA zoning restrictions mainly target various forms of fishing and recreational use, although some may require upstream impacts, such as pollution and coastal development, to be addressed. Implementing MSP in a broader scale and heavily-exploited marine ecosystems may involve more economic sectors and competing interests, and to effectively address the trade-offs between different uses and between use and protection will certainly be more challenging.

There are also different perspectives regarding the place and importance of MPAs in broader scale ecosystem management and MSP. Some believe that MPAs and biodiversity conservation should be at the heart of ecosystem-based management, and clear mandates should be developed so that all agencies, regardless of their primary responsibility, would be required to manage sectoral activities for the purpose of sustaining biodiversity (Palumbi et al. 2010). Implementing such an approach would be politically difficult, but it would ensure much-needed coordination and consistency between different agencies (Palumbi et al. 2010). Others argue that MPAs can only contribute to ecosystem-based management if the dominant impact on a marine ecosystem comes from fishing, although this is the case in many areas of the oceans (Halpern et al. 2010). There are also concerns that current systems of MPAs are too small and ineffectively enforced to be beneficial to management at an ecosystem scale (Halpern et al. 2010a; Kenchington et al 2010). It would be safe to conclude that the importance of MPAs to a broader-scale MSP system depends on the primary objective of the latter, which can vary from being the key driver of a MSP imitative aiming at reversing the trend of biodiversity decline, for instance, to being merely a potential type of 'use' of ocean space in a MSP system designed to maximize rational resource exploitation.

MPAs, however, are key components of any MSP system if its underlying principle is to restore or maintain ecosystem functions and services. MPAs are essential if an ecosystem approach were to be adopted in managing our oceans (Roberts 1997b, Pauly et al 1998, Pauly et al 2002; Palumbi 2003, 2010; Jones 2007), as fishing is still a key contributor to the deterioration of marine ecosystems worldwide (Pauly et al 2005). Marine ecosystems exhibit complex system behaviours and often managers cannot safely assume they will recover if stressors are reduced (Crowder and Norse 2008). MPAs, with no-take marine reserves at its core, provide a precautionary buffer against uncertainty (Murray et al 1999, Guénette et al 1998, Lauck et al 1998, Stefansson and Rosenburg 2005; Jones 2007). Pitcher (2005) recognises that reducing uncertainty in ecosystem simulation techniques and making decisions robust against climate change will be a challenge, therefore setting aside a significant portion of ocean space as highly protected MPAs can enhance ecosystem resilience and the potential for recovery against unforeseeable changes.

In practice, experiences in designating and managing MPAs and MPA networks over the past few decades can offer valuable lessons for MSP. Spatial planning in large-scale multiple-use MPAs demonstrates how and to what degree different interests (*e.g.* conservation, fisheries, marine tourism, indigenous use etc) can be accommodated and balanced against each other. There are a large number of studies on the potential of MPAs in delivering multiple benefits, particularly for biodiversity conservation and sustainable exploitation of fisheries resources. For example, the experience with zoning in the GBRMP shows that systematic planning and management efforts can address objectives of conservation and sustainable use in a large marine ecosystem (Kenchington 2010). Rezoning and the increase in the area of no-take zones in the GBRMP have generated multiple ecological and socio-economic benefits. These benefits include improvements in overall ecosystem health and resilience, as reflected in the increased density of targeted reef fish species, a marked reduction in outbreaks of crown-of -thorns starfish (COTS) and consequently higher abundance of corals in the no-take areas (Russ et al. 2008; Sweatman 2008; McCook et al. 2010). Increases in the marine reserve network in 2004 affect fishers, but preliminary economic analysis also shows considerable net benefits, in terms of enhanced environmental and tourism values (McCook et al. 2010). It is important to note, however, that the success of the GBRMP zoning and management system builds upon a specific governance and political context, with widespread concerns for and national and global significance of an iconic marine ecosystem contributing significant support for its continuously evolving management institutions (Kenchington 2010). This experience cannot be transposed to other contexts without the development of substantial political and public support (Kenchington 2010).

Designing MPA networks has benefited from the development of biographic classification schemes and a growing number of mathematical site selection algorithms and software programs, which provide the possibility of effectively protecting a representative sample of all marine habitats, ecosystems and eco-regions, and fulfilling other ecological criteria, such as connectivity (Possingham et al. 2000; Day & Roff 2000; Leslie et al. 2003; Lourie and Vincet 2004). The science behind the development of MPA networks is equally applicable to an ecosystem-based MSP, as the key ecological principles will need to be fulfilled if marine ecosystems were to be restored or managed to a healthy and productive status, and to support continued human uses of marine resources (Roberts et al. 2003; NOAA 2007; Crowder and Norse 2008).

MPAs are also testing grounds for developing supportive governance and institutional frameworks for MSP. The debates on the merits of 'top-down' versus 'bottom-up' approaches in governing MPAs continue to evolve, but the reality is that these approaches will need to be combined because MSP, like most MPAs, has statutory obligations to fulfil, but also needs to provide scope for public and stakeholder participation. There are examples of MPAs in different contexts that have successfully applied a combination of different incentives to steer MPA management and address related conflicts, as well as lessons to be learned to prevent governance arrangements from being ineffective and/or inequitable (see section 1.2.8).

To summarize, international and EU policies for managing the marine environment have established that an ecosystem approach is fundamental to MSP. Such legal and policy provisions serve as a key driver for developing systems of MSP, however it should also be recognised that MSP is increasingly being seen as a means to facilitate new developments in the oceans and to address multiple and often competing priorities, including both use and protection. There are similarities and differences between MPAs and other types of MSP, and the place and importance of MPAs vary in a broader scale MSP system, depending on the primary objective of the latter. However MPAs will continue to be a key component in MSP policies and practices underpinned by an ecosystem approach. MPAs, including those present in MESMA case studies, can also offer valuable lessons for implementing MSP to address both ecological and socio-economic needs, as well as serving as testing grounds for the application of innovative scientific and governance tools and approaches in implementing MSP.

Country (EU)	National Policies and legislations	Policy goals for MSP
Belgium	'Master Plan' for MSP in Belgian territorial sea and EEZ (2003)	The core issues of the Belgian MSP policy framework include the development of offshore wind farms, the delimitation of marine protected areas, a policy plan for sustainable sand and gravel extraction, enhanced financial resources for the prevention of oil pollution, the mapping of marine habitats, protection of wrecks valuable for biodiversity, and the management of land-based activities affecting the marine environment.
Germany	Federal Spatial Planning Act (extended to the German EEZ in 2004)	To establish sustainable development of ocean space, in which social and economic demands for space are consistent with the ecological functions of space
	Mecklenburg-Vorpommern (Baltic Sea) Spatial Planning Programme	To ensure conflict management between the demands of new technologies, tourism and nature protection and traditional sectors like shipping, fishing and defense at an early stage
The Netherlands	National Spatial Planning Policy Document, the Netherlands (2005)	To enhance the economic importance of the North Sea and maintain and develop the international ecological and landscape features by developing and harmonizing sustainable economic activities in the North Sea, taking into account the ecological landscape features of the North Sea
United Kingdom	A Sea Change: Marine Bill White Paper (2007)	To create a strategic marine planning system that will clarify marine objectives and priorities for the future, and direct decision-makers and users towards more efficient, sustainable use and protection of our marine resources

Source: Douvere (2008).

1.2.4 MPA DESIGN AND FUNCTION

In the rapidly growing field of MPA science, ecological criteria and characteristics relevant to the design of MPAs have been developed (e.g. Botsford et al. 2003; Hastings and Botsford 2003; Halpern 2003; Roberts et al. 2003a, b; Gubbay 2004; Gaines et al. 2010). In the following the main aspects are described in relation to size and other MPA design parameters.

1.2.4.1 MPA size

The optimal size of a MPA is highly dependent on the objectives, the life history, mobility and dispersal patterns of focal species and, ultimately, political and societal compromises. In relation to mitigation of direct impacts on sessile species and habitats or species with limited movements, it is intuitive that MPA effectiveness will be positively correlated with MPA size. Optimal size becomes a more difficult parameter to estimate when dealing with MPAs established to e.g. protect mobile species or enhance populations of mobile fish species in areas adjacent to MPAs.

MPAs established to protect temperate fish species in northern European waters have traditionally been relatively large in size and established to protect one or several commercial fish species or their habitats (PROTECT 2006). Examples of such MPAs include the North Sea Plaice Box, which was established to reduce bycatch of juvenile flatfish species, and the Norway Pout Box, which was established to reduce levels of fishing mortality on juvenile gadoids such as haddock and whiting in the Norway pout fishery. MPAs established to protect biodiversity in northern European waters have traditionally been nearshore and relatively small. However, recent years have seen the designation of large Natura 2000 sites in offshore waters of e.g. the North Sea and Baltic Sea (e.g. ICES 2008) as well as a number of large MPAs to protect cold-water corals (e.g. PROTECT 2009; Anon 2006; Fosså & Skjoldal 2009).

The FP6 project PROTECT focused on several representative case study MPAs and scenarios in northern seas. One of the case study fish species was the sandeel in the North Sea, where it is an important mid-trophic level species in the ecosystem and the target of commercial fisheries. Modelling studies of sandeel life history traits coupled with knowledge of ocean currents led to the conclusion that proper, informed selection of MPA sites is a more important factor than size when designating MPAs for sandeel in the North Sea (PROTECT 2006, 2009; Christensen et al. 2009).

Mediterranean MPAs are often smaller in scale, and to a higher degree address local ecosystem concerns rather than differentiating between biodiversity and fisheries, which to a degree makes them comparable with MPAs in tropical regions (Sørensen & Thomsen 2009). Nonetheless the Mediterranean-focused FP6 EMPAFISH project mainly studied fish and fish assemblages within and around MPAs and found that reserve effects increase along with reserve size (Claudet et al. 2008), i.e. increasing the size of the no-take zone resulted in increased density of commercially harvested fishes within the reserve compared to outside. In contrast, increasing the size of an intermediary, multiple-use buffer zone reduced the effectiveness of the reserve.

Gaines et al (2010) synthesized 33 scientific publications representing 57 case studies that explicitly examined how much area should be protected from fishing to maximize long-term fishery yield and/or profit. Approximately half of the studies covered showed that yields/profits had been improved with the use of marine reserves. Of these studies, they examined the frequency distribution of the percentage of fishing grounds recommended to be included in MPAs in the studies. They found that peak benefits are projected to occur when MPAs constitute as much as half of the total habitat, while the majority of cases with fisheries benefits recommended inclusion of 10-40% of fishing grounds. Gaines et al. (2010) argue that these moderately large proportions support the conclusion that MPA networks can provide simultaneous conservation and fisheries benefits for some species and locations.

Concerns over feasibility and utility have limited the establishment of MPAs to manage the pelagic environment and the exploitation of its resources. However, Game et al (2009) argue that advances in conservation, oceanography and fisheries science confirm MPAs as defensible and feasible instruments for pelagic conservation. The concept of *mobile MPAs* has been proposed as a tool to protect highly dynamic populations of e.g. marine mammals (Porter 2001). Furthermore, real time closures, i.e. closing and opening areas when and where needed to protect e.g. spawning or nursery areas, have become increasingly common as fisheries management measures in Europe and beyond.

1.2.4.2 MPA networks

While some MPAs may need to be very large to encompass all focal species and habitats and/or populations of fish to provide substantial benefits, such large sites may not be feasible to establish. Furthermore many species have life cycles that require various habitats dispersed over large distances (e.g. spawning and nursery areas). In such cases, ecologically coherent networks of smaller MPAs may be a more effective management measure than single, large MPAs (e.g. Gaines et al. 2010). Recent studies on the functioning of southern European MPAs highlight that an efficient strategy for designing multiple use MPAs should combine a network of no-take areas, areas where fishing activity is regulated, and a multi-zoning scheme allowing recreational fishing and diving (Mangi and Austen 2008; Pérez-Ruzafa et al. 2008).

According to OSPAR (2006) the following points can be identified as contributing to coherence in a network of MPAs:
A network's constituent parts should firstly be identified on the basis of criteria which aim to support the purpose of the network.

The development of an ecologically coherent network of MPAs should take account of the relationships and interactions between marine species and their environment both in the establishment of its purpose and in the criteria by which the constituent elements are identified.

A functioning, ecologically coherent network of MPAs should interact with, and support, the wider environment as well as other MPAs although this is dependent on appropriate management to support good ecosystem health and function within and outside the MPAs.

Other factors that play an important role in effective MPA networks are concepts such as representativity, connectivity and replication. Representativity implies that all focal species and/or habitats, depending on the objectives of the network, are represented in the MPAs that make up the network. Connectivity entails ensuring that "blue" transport corridors such as certain substrates or ocean currents provide ecological connections between sites, i.e. via passive dispersal or active migration. Replication is an insurance against failure of a given MPA within the network, ensuring that focal species/habitats are represented in more than one MPA in the network.

1.2.5 EFFECTS OF MPAS

Many studies on the ecological effects of MPAs highlight that they provide multiple benefits such as protection of habitat and biodiversity, recovery of depleted stocks and opportunities for scientific investigation, education and recreation (Agardy 1994; Ramos-Esplà and McNeill 1994; Harmelin et al. 1995; Allison et al. 1998; Bohnsack 1999, 1996; Zabala 1999). However, most literature reporting MPA effects concentrates on the effects of MPAs on fish populations and fisheries in adjacent waters. Provided that MPAs are effectively managed, the expected potential effects inside the reserves include (1) reduction of fishing mortality; (2) increase of density of target species; (3) increase in the size of target species; (4) increase in biomass of target species; and (5) increased production of eggs and larvae of target species. On the other hand, the expected effects outside the reserves are (6) net export of adult (post-settlement) individuals - the spillover effect; and (7) net export of eggs and larvae – the recruitment effect. The spillover effect results in fisheries enhancement outside the reserves. The recruitment effect enhances the supply of recruits to fished areas (Russ 2002). In addition, there could be community-wide effects, both as changes in the community structure and the biomass sizes of individuals in the assemblage (Russ 2002).

1.2.5.1 Fishing mortality of target species

Whether or not MPAs are able to effectively reduce fishing mortality of target species depends on the biology of the focal species and the overarching fisheries management that is in place for the focal species. If, for instance, the fish species is highly mobile and managed through a quota system, then it is likely that MPAs will have a limited effect unless closures are very large, as fish may be caught sooner or later when they traverse MPA boundaries. In such situations a general reduction in fisheries mortality may be better achieved through overall effort reductions (e.g. Degnbol et al 2006). However, if fish have strong associations with certain benthic habitats (e.g. reef associated species) or if populations of mobile species congregate in localised spawning grounds where catchability of the species during spawning can be very high then an overall reduction of fishing mortality is feasible through establishment of permanent or temporary closures. Fishing mortality may also be reduced through MPA establishement if focal fish species in specific areas are vulnerable to bycatch in other fisheries during parts of their life cycles, e.g. in nursery habitats. However, mortality may in some cases also be reduced through implementation of technical reglations in fishing gears (e.g. increased mesh size).

1.2.5.2 Size of target species

Reports of significantly larger mean sizes of target fishes inside MPAs compared with fished localities provide evidence of gradients of decreasing target species size with distance from MPAs in the Philippines (Abesamis and Russ 2005), Kenya (McClanahan and Mangi 2000), and Spain, where mean size of fish near the boundary were intermediate between larger sizes within the MPA and smaller ones in fishing grounds further away (Stobart et al. 2009). Interestingly, biomass size-spectra showed slopes significantly steeper in the trawl ban area of the Gulf of Castellammare than in control unprotected gulfs (Sweeting et al. 2009). This unpredictable result was attributed to the exclusion of trawlers, which lacked catch-size selectivity and to the continued, more size selective fishing by artisanal gears within the MPA. In the Øresund, a sound between Sweden and Denmark, a trawl ban has been in place since 1932 as part of maritime safety regulations (Øresund is very busy and narrow shipping lane), in effect serving as a *de facto* MPA. While fish populations have been dwindling for decades in the adjacent Kattegat and Western Baltic Sea, populations of cod, haddock, plaice and other commercial species have been consistently stable and with a large proportion of big, old individuals (Svedäng 2010) despite a substantial commercial gillnet fishery and intensive angling. In addition, despite being flanked by two major cities, biological communities that have become very rare in trawled areas are still present in the Sound.

1.2.5.3 Biomass of target species

Overall, the main biological effect that follows MPA establishment is the increase of biomass. This occurs within the no-take area or where the highest level of protection is enforced (e.g. gear restriction areas, seasonal closures and the like). Biomass increase is a consequence of reduced fishing mortality (Jennings 2000) and has been documented is several MPAs (Badalamenti et al. 2008, Murawski et al. 2000, 2004, 2005, Pipitone et al. 2000; Svedäng 2010) although the size of the no-take area may play a significant role in shaping the size of this effect (Claudet et al. 2008).

1.2.5.4 Production of eggs and larvae

It can be expected that the production of eggs and larvae of species of moderate vagility inside the MPAs islarger than in exploited areas. Such effect would be due to the higher density and size of target species, and by the fact that fecundity is exponentially related with female size (Plan Development Team 1990). However, increases in production of propagules is very difficult to measure and has rarely been unequivocally tested by field experiments (reviewed by Russ 2002).

1.2.5.5 Egg and larvae export (recruitment effect)

Because MPAs can increase the reproductive output of target populations per unit area, increased recruitment from egg and larval export is anticipated to produce even larger benefits for fisheries than spillover of adults (Russ 2002, Moffitt et al. 2009). However, such a recruitment effect is hard to detect due to the high spatial and temporal variability of larval survival and settlement, as well as the large area over which it can occur (Botsford et al. 2009). Although empirical support for yield enhancement at the stock or fisheries management scale is sparse, evidence of enhancement at the local scale is growing (Goñi et al. in press). At present, and despite their intuitiveness, empirical evidence of egg and larval export effects on fisheries is still lacking and most research is currently focused on unraveling larval dispersal and population connectivity patterns (Goñi et al. in press) including modelling of larval drift (Christensen et al. 2009).

1.2.5.5 Adult biomass export (spillover effect)

The spillover effect is defined as a biomass net export from reserves to surrounding areas, hence supporting adjacent fisheries. The emigration of adult (post-settlement) fishes from MPAs is based on density dependent mechanisms (Sanchez-Lizaso et al. 2000; Halpern 2010b). However, due to the lack of data from before the MPA establishment and of control sites, well documented spillover effects are rare. Direct evidence of net emigration of exploitable individuals from MPAs to fished areas has been demonstrated in only a handful of cases for lobsters and reef fish, or surmised from spatio-temporal patterns of fishery recaptures (Goñi et al. 2006, 2008, 2010). Importantly, the degree of connectivity among sitable habitat patches modulates the intensity of spillover across reserve boundaries (McClanahan and Mangi 2000; Forcada et al. 2009). Better evidence of spillover come from catch per unit effort (CPUE) patterns, that is the increase in CPUE over time in the fisheries near MPAs (Yamasaki and Kuwahara 1989; Alcala et al. 2005; Murawski et al. 2005; Stobart et al. 2009). Declining gradients of CPUE with distance from the MPA boundaries has also been reported. However, whether spillover causing these CPUE patterns translates into greater catches in fisheries adjoining MPAs is less well documented (Goñi et al. in press). Russ (2002) and Halpern et al. (2010b) reviewed some literature and concluded that spillover for species that respond positively to protection has a delay after MPA implementation of up to 8 years and is limited to a scale ranging from hundreds of meters to a few kilometers from the reserve boundaries, depending on the species.

1.2.5.6 Change in the community structure

Whether or not spillover can supply a more diverse catch has been poorly assessed. Stobart et al. (2009) observed that as a consequence of spillover processes, mean taxonomic distinctness of the multi-species fish catch near the Columbretes MPA was lower than within the MPA but higher than farther fished areas. Unexpectedly, species richness and diversity indexes were lowest inside the MPA and similar among close and far fished areas, indicating the current lack of understanding of the responses of these indexes to fishing in temperate areas (Goñi et al. in press). Furthermore, in no part of the Kattegat Sea area mentioned above could a similar abundance or size distribution of cod be found as in the Öresund, where trawling has been banned for decades. The pattern was similar for haddock, whiting, plaice and lemon sole (Svedäng 2010).

1.2.5.7 Trophic cascades

Stock re-building in MPAs includes an increase in the abundance of predators, which are mostly targeted by fisheries. The rise in predator biomass may in turn affect the food web through a trophic cascade. According to Pinnegar et al. (2000), trophic cascades are predatory interactions involving three trophic levels, whereby primary carnivores, by suppressing herbivores, increase plant abundance, although cascade-type effects have been reported to extend through four or more trophic levels in some instances.

Species and habitats are interlinked across ecosystems in complex relationships, and changes made to one ecosystem component will likely impact others. For instance, depletion of predator fishes (e.g. cod) often causes rapid population increase of their prey species (e.g. sea urchins), disturbing food webs and ecosystem function (e.g. Norderhaug& Christie2009). In the specific example, sea urchins graze on kelp forests, leading to loss of important shelter for juvenile cod. Such ecological changes are very difficult to prevent or reverse through small-scale, localised MPAs. Networks of well managed MPAs, together with an effective ecosystem based fisheries policy, are more likely to promote resilience and recovery in such systems.

1.2.6 Monitoring of MPAs (biological and socioeconomic)

1.2.6.1 Introduction

MPAs can help to achieve many societal goals such as ecosystem protection and biodiversity conservation, enhancement and sustainable exploitation of fisheries, and expanded knowledge and understanding of marine ecosystems. Each MPA or network of MPAs will have its own specific goals and objectives that ideally would be clearly stated in its management plan. Monitoring systems should be tailored toward these goals and objectives. Establishing good monitoring is the cornerstone of adaptive management, and therefore MPA success, as it will be the source of the necessary information to evaluate MPA performance and the effectiveness of management actions. Evaluation consists of reviewing the results of actions taken and assessing whether these actions are producing the desired outcomes. Such objective evaluation is needed to assess future needs and adapt current practices so as to improve the effectiveness of management efforts and optimize related human and financial resource allocation (Pomeroy et al. 2005). Monitoring is an important tool for improving the principles of MPA design and for teaching us what does and does not work in marine reserves (Sladek Nowlis and Friedlander 2004). Without monitoring one can’t tell whether management is working (on an MPA scale, but also local, national, or eco-regional scale) and we are unable to demonstrate the benefits of MPAs.

1.2.6.2 MPA effectiveness

Hundreds of peer-reviewed scientific articles and dozens of literature reviews have been published the last decades on the effectiveness of MPAs. Scientific documentation shows that MPAs can increase within their boundaries the density, abundance, biomass, individual size, proportion of larger/older individuals, and fecundity of exploited populations (e.g. Roberts 1995; Gell & Roberts 2001; Côté et al. 2001; Roberts et al. 2001; Halpern 2003; García-Charton et al. 2008). Reserves may also result in a more desirable population structure (characterised by age, gender or individual size) that may improve breeding success and raise mean recruitment into the harvested population (Bohnsack 1998; Jennings 2000). Increased abundance and increased average individual size within reserves may lead to positive spillovers in harvested areas as adult individuals migrate from reserves to adjacent locations and eggs or larvae are transferred by currents (Roberts et al. 2001; Gell & Roberts 2003). Furthermore, MPAs can protect biodiversity, habitats and ecological processes, preserve endangered species, enhance nonconsumptive activities and opportunities, and expand our knowledge and understanding of marine species (Sobel and Dahlgren 2004; Boudouresque et al. 2005; García-Charton et al. 2008).

However, there are many examples of MPAs, perhaps the majority, failing to accomplish their stated targets, mainly because of wrong design and set up or unrealistic expectations (Kelleher et al. 1995; Alder 1996; McClanahan 1999; Jameson et al. 2002). The increased 'marine-based' touristic and recreational activities (such as diving) may have negative effects and can cause increased damages to some habitats or species (García-Charton et al. 2008). The effectiveness of MPAs for sustaining fish stock yields has been challenged by many authors arguing that conventional fisheries management approaches are more effective and MPAs might lead to reduced fish stock production due to density-dependent factors (Shipp 2003; Steele and Beet 2003; Norse et al. 2003; Kaiser 2005).
The efficacy of a particular MPA to achieve specific management objectives may depend on a variety of factors, such as the specific MPA objectives, design characteristics of the MPA, adequacy of management actions, and public compliance. MPA management should include defined objectives and goals from the outset, site selection, zoning, planning and implementing a surveillance and enforcement system, as well as monitoring actions (Kelleher 1999).

1.2.6.3 Optimal monitoring setups

1.2.6.3.1 Monitoring frameworks – effectiveness indicators

Performance evaluation of management impact and value is regarded as a top priority in order to assess and adapt management needs where marine protected areas are actively being used around the world. Evaluation of protected area management effectiveness is seen to play a critical role in providing for and demonstrating long-term positive impacts on biodiversity and the human communities that depend on these resources (Hockings et al. 2000).

One of the factors that restricts effective decision-making in adaptively managing MPAs is a lack of information about the status and nature of conditions (including threats) operating within or around MPAs. Obtaining such information requires a periodic and comprehensive assessment of the natural and social processes occurring within and outside the boundaries of MPAs. As such, there is an increasing interest in the development and use of an adequately comprehensive set of indicators that measure the socio-economic, biophysical, and institutional (governance) outcomes from the management process associated with MPA use (Watson et al. 2003).

In 2000, IUCN's World Commission on Protected Areas (WCPA) Marine and the World Wide Fund for Nature (WWF) launched a collaborative initiative to improve the management of MPAs focused on working with managers, planners, and other decision-makers to develop a set of indicators for assessing the effectiveness of MPA use. This initiative is aimed at both enhancing the potential and capability for adaptive management of MPAs as well as improving our understanding of how effective MPAs are being used at present around the world. The initiative builds on the IUCN Management Effectiveness Framework, developed through the WCPA Management Effectiveness Theme (Watson et al. 2003).

Since the development of WCPA framework in 2000, technical experience increased rapidly resulting in a range of assessment systems based upon the framework. There are now three basic approaches;

1. In-depth, evidence based assessments aimed at building monitoring systems and long-term understanding of management in an individual protected area, such as the Enhancing our Heritage system being developed for World Heritage sites (Hockings et al. 2004).

2. System-wide peer–based assessment developed specifically for use on a system-wide scale such as the WWF RAPPAM system (Ervin 2003) and the systems developed in Finland (Gilligan et al. 2005) and Catalonia, Spain (Mallarach and Varga 2004).
3. Scorecard expert–based assessments

An overview is given of several known Management Effectiveness Evaluation tools (partly taken from the Convention on Biological Diversity website (http://www.cbd.int):

The **MPA IUCN/NOAA/WWF guidebook** provides a step-by-step process for planning and evaluating the management effectiveness of MPAs. The guidebook is designed to provide step-by-step guidance to managers and other practitioners in: (a) selecting the relevant biophysical, socioeconomic, and governance indicators for the evaluation of a particular MPA, (b) developing a process for planning for and implementing this evaluation, and (c) using the results generated to inform and adaptively manage the MPA (Pomeroy et al. 2004, 2005).

The **WWF Rapid Assessment and Prioritization of Protected Area Management (RAPPAM)** methodology provides a country-wide assessment of the effectiveness of protected area management, threats, vulnerabilities and degradation (Ervin 2003).

The **World Heritage Areas Enhancing Our Heritage** method is an evaluation methodology developed for detailed site level assessment. The Workbook provides guidelines and assessment tools for each element of the WCPA Framework. These tools have been designed to allow specific needs and circumstances of the site to be taken into account and to provide a means for integration of existing monitoring data into the evaluation system. While designed specifically to meet the needs of natural World Heritage sites, the methodology is applicable to any protected area (Hockings et al. 2004).

A marine version of the **World Bank/WWF Alliance Tracking Tool** was prepared by the World Bank for use in Marine Protected Areas (Staub and Hatziolos 2003).

A **DPSIR General Conceptual Framework** to analyze the socioeconomic issues, environmental changes and policy responses of MPAs, was developed by Ojeda-Martinez et al. (2009). The DPSIR framework is an extended version of the Pressure-State-Response (PSR) approach, that is based on the idea that anthropogenic activities impact the environment and that adverse environmental impacts drive humans to control the pressures. Under DPSIR, environmental problems and solutions are simplified into variables that stress the cause–effect relationships between human activities that exert pressures on the environment, the condition of the environment and society's response to the condition (Mangi et al. 2007). The DPSIR scheme of indicators is a flexible framework that can be adapted to the necessities of specific programs to identify the different actors and processes affecting the MPA and surrounding areas. Its structure can be used to select indicators as is being done in the implementation of e.g. European Water Framework Directive (La Jeunesse et al. 2003; Mysiak et al. 2005; Borja et al. 2007; Ojeda-Martinez et al. 2009).

1.2.6.3.2 Monitoring of biological indicators

Biological indicators include population density or abundance of targeted, rare, key or especially vulnerable species, abundance of species classified into trophic guilds (e.g., piscivores, herbivores), biodiversity indices of benthic communities, measures of habitat quality and ecosystem integrity (e.g., status of essential habitats such as sea grasses or coral reefs, structural complexity and abundance of key shelter-providing species), size distribution and vital rates (growth, mortality, fecundity) (Sladek Nowlis and Friedlander 2004; Pomerey et al. 2005; García-Charton et al. 2008).

Sampling to monitor the effect of MPAs on biological indicators is made difficult by intrinsic natural spatial and temporal variability (Guidetti 2002). Ideally, sufficient spatial and temporal replication before and after the establishment of MPAs is necessary to detect and measure the effects of protection on biological indicators (Guidetti 2002; Sladek Nowlis and Friedlander 2004). The so-called 'beyond BACI' (Before/After and Control/Impact) designs (Underwood 1992, 1993) represent ideal approaches to assess the effects of MPAs (Guidetti 2002; Frederiksen et al. 2008). Such designs are based on the contrast between multiple protected areas with multiple control locations, before and after the establishment of protection. The use of several protected locations (replicates) and several unprotected locations as controls would allow preventing misinterpretation due to natural variability among locations. However, such spatial replication is often impossible, e.g. when there is a single MPA. In such cases, an asymmetrical design (Underwood 1992) with one protected and a set of unprotected locations (controls) may still be possible. Similarly, to account for the temporal variability in the biological indicators examined, temporal replication should be incorporated in experimental design, i.e. by sampling several times before and several times after the establishment of a MPA (Guidetti 2002).

In the described design 'controls' are assumed not to be influenced directly or indirectly by protection (e.g., due to spillover effects), and thus should not be areas adjacent to an MPA. However, if controls are far away they may be influenced by different ecological processes and events and thus be less valuable for comparison. A useful way to overcome this difficulty is to examine groups of locations at varying distances from MPAs and thus also examine the dilution of the MPA effect (Sladek Nowlis and Friedlander 2004). When the biological indicator varies across strata (e.g., habitat types) stratified random sampling should be preferred and comparison should be made for similar strata among protected and unprotected areas.

1.2.6.4 Monitoring implementation (in reality)

In the early 1990s, 383 out of 1,306 MPAs were assessed for management effectiveness. Roughly one-third were judged to have met their management objectives, one-third partially met their objectives, and the remaining had inadequate information, suggesting that perhaps one-third failed to meet their objectives (Kelleher, 1995). More recently, in Southeast Asia, of about 332 MPAs whose management effectiveness can be determined, only 14% are effectively managed, 48% have partially effective management and 38% have inadequate management (Burke et al. 2002; Watson et al. 2003).

The mixed success rate of current MPA performance demonstrates an important and immediate need to build capacity for MPA teams to evaluate the effectiveness of their management strategies and actions so that they may be able to adaptively manage their efforts and improve the impact and scope of their protective efforts through time (Watson et al. 2003).

1.2.6.4.1 IUCN MPA MEI Guidebook

The cornerstone of the guidebook methodology is the selection and measurement of indicators of MPA management effectiveness. The guidebook outlines a four-part process: (1) select the appropriate indicators, (2) plan and prepare for the evaluation, (3) collect and analyze data for the selected indicators, and (4) communicate and use evaluation results to adapt the MPA's management.

The MPA MEI guidebook lists 42 MPA-specific indicators that MPA managers can choose to use for evaluating their site, of which ten are biophysical (Table 1.2.1), sixteen socioeconomic, and sixteen governance indicators. Each set of indicators is associated with general goals and objectives that may be part of a MPA (Pomeroy et al. 2005).

Table 1.2.1: IUCN MPA MEI Guidebook MPA biophysical goals and indicators (from Pomeroy et al. 2005)

Five common MPA biophysical goals and the 10 associated indicators used to evaluate progress being made against them

Goals (no. of associated objectives)	Indicators
1. Marine resources sustained or protected (6)	B1—Focal species abundance
2. Biological diversity protected (7)	B2—Focal species population structure
3. Individual species protected (4)	B3—Habitat distribution and complexity
4. Habitat protected (4)	B4—Composition and structure of the community
5. Degraded areas restored (5)	B5—Recruitment success within the community
	B6—Food web integrity
	B7—Type, level, and return on fishing effort
	B8—Water quality
	B9—Area showing signs of recovery
	B10—Area under no or reduced human impact

The biophysical indicators address various factors relating to the natural environment:

- Six focus on the biotic context (indicators B1, B2, B3, B4, B5, and B6), including two at the species level (B1 and B2), one on habitat (B3), and three on community ecology (B4, B5, and B6).
- One measures the 'goods' generated (B7).
- One is an abiotic measure (B8).
- Two are 'aerial' indicators of observed change (B9 and B10).
- (Pomeroy et al. 2005).

1.2.6.4.2 DPSIR General Conceptual Framework

A general conceptual framework using the DPSIR methodology to analyze the socioeconomic issues, environmental changes and policy responses of MPAs, was developed by Ojeda-Martinez et al. (2009). This framework was developed through a participation process which involved an expert panel but must be used by managers and evaluated by the stakeholders implicated in the MPA. From this conceptual framework a set of variables for each DPSIR component were defined.
Ojeda-Martinez et al. (2009) identified /developed the following indicators for impact (Table 1.2.2):

Table 1.2.2: Indicators for Impacts (Ojeda-Martinez et al. 2009).

Impacts	Fishing	Surface affected by a gear	Total surface of a determinate kind of habitat affected by a gear.
	Fishing	Surface affected	Total surface of a determinate kind of habitat
	Fishing	Changes in density	Temporal and spatial changes of the quantity of key species that are in the MPA boundaries
	Fishing & tourism	Changes in covertures	Changes produced in the state of the key elements during the time a pressure is affecting them.
	Fishing	Changes in community structure	Temporal and spatial changes in the community structure.
	Fishing	Species size variation	Temporal and spatial variation of the size of the different key elements selected.
	Fishing	Relative abundance	Temporal and spatial variations on the relative abundance of the individuals for each key species.
	Fishing & tourism	Changes in abundance	Temporal and spatial variations of the quantity of each key specie that can be found in the MPA
	Fishing & tourism	Changes in diversity	Temporal and spatial variations on the species composition structure in the MPA boundaries.
	Fishing & tourism	Changes in richness	Temporal and spatial variations on the number of the key species.
	Fishing	Changes in dominance	Temporal and spatial variations on the abundance of the dominant species.
	Fishing	Changes in sediment	Changes in sediment composition and/or quality.
	Fishing	Species substitution	Temporal and spatial substitution of the species
	Fishing	Families substitution	Temporal and spatial substitution of the families
	Fishing	Changes in recruitment	Temporal and spatial variations on changes in the recruitment rate
	Fishing	Breaking index	Temporal and spatial variations of breaking index of key species.
	Fishing	Rugosity	Temporal changes in the rugosity of key elements
	Fishing	Changes in habitat heterogeneity	Temporal and spatial habitat changes
	Fishing	Changes in trophic levels	Temporal and spatial changes in trophic levels
	Fishing	Opportunistic species	Appearance of opportunistic species.
	Fishing	Sensitive species	Changes in sensitive species
	Tourism	Species size	Variation of the targeted species size
	Tourism	Species weight	Variation of the targeted species weight
	Tourism	Mortality rate	Changes in mortality rate
	Tourism	Captures	Temporal changes in captures
	Tourism	Recruitment rate	Evolution in the recruitment rate
	Tourism	Extracted biomass	Evolution of the extracted biomass
	Tourism	Extracted biomass by specie	Evolution of the extracted biomass by specie
	Tourism	Fragile species	Decrease of fragile species
	Tourism	Protected species	Disappear rate of protected species
	Tourism	Sediment	Changes in the sediment composition and/or quality
	Tourism	Opportunistic species	Opportunistic species evolution
	Tourism	Filter species	Evolution of filter species
	Tourism	Anchoring	Evolution of the surface damaged by anchoring
	Tourism	Diving activities	Evolution in the surface affected by the diving activities.
	Tourism	Whale watching	Temporal and spatial variations in whale watching
	Tourism	Sea mammals	Number of impacts with sea mammals
	Tourism	Trampling	Evolution in the surface affected by the influx of visitants.
	Tourism	Water quality	Changes in water quality

1.2.6.4.3 Monitoring of biological indicators

It is extremely unlikely that the ideal 'beyond BACI' sampling design will be applied. Due to the lack of communication between researchers and decision makers, it is rare that MPAs are monitored before protection measures are in place (Guidetti 2002; Sladek Nowlis and Friedlander 2004). The problem of statistical replication is severe and the lack of temporal and spatial replication in many of the studies on MPA effectiveness complicates interpretation of the results (Halpern 2003). Even in MPA networks, reserves are often not implemented at the same time and thus it is not feasible to have spatial replication of protection. Studies lacking replication cannot be logically interpreted and it is not possible to discriminate between protection-effect and natural variability, especially in simple inside/outside studies (Guidetti 2002; Halpern 2003). Socio-political processes cause biases in the selection of fully protected zones that can seriously confound MPA effectiveness studies (often leading to underestimation of the magnitude of resource recovery within reserves), when data prior to protection are lacking (Edgar et al. 2004).

A critical evaluation of the experimental designs employed by published studies between 1990 and 2001 of MPA effectiveness brought to light many problems with replication and lack of control sites (Willis et al. 2003): insufficient sample replication (e.g., only one site sampled inside and outside a reserve, or no control sites sampled at all); lack of temporal replication (most studies consisted of surveys conducted at only one time); lack of replication at the reserve level; non-random placement of reserves, i.e. often reserves are sited to include special or unique features, which makes it difficult to find similar control sites; spatial confounding (e.g., all control sites located only at one end of a reserve). Wilis et al. (2003) found no study that avoided these spatial and temporal replication problems and only a handful of studies that at least attempted to fulfill good design criteria. Halpern (2003) conducted a meta-analysis of 89 independent studies on marine reserve effects, nearly all of which compared inside *vs* outside a reserve at a single point in time. Only 17 studies included 'before-after' measurements, only 9 studies included both 'before-after' and 'control-impact' data, and only seven studies included multiple measurements through time. Furthermore, in many of these studies sample sizes were not large enough to draw statistically significant conclusions and many studies simply had not reported the statistical significance of their results (Halpern 2003). In a review of 30 papers on MPA effects in the Mediterranean Sea, Fraschetti et al. (2002) found only ten papers that included more than one control site and only one paper with 'before' data (but without control outside the protected area).

1.2.6.4.4 Gap between guidelines and practice – practical obstacles

MPAs are often challenged in their ability to achieve their objectives, evaluate their effectiveness and provide for informed management decision-making. This is due to small management staff size, insufficient financial, logistical, and technical support, lack of scientific information, lack of experience to perform monitoring and evaluation, unfamiliarity with certain indicators and measurement methods, insufficient institutional, decision-making, and political support, and reluctance of local populations to respond to interviews or participate in group evaluation activities as a result of distrust, unfamiliarity, or time requirements (Kelleher et al. 1995; Pomeroy et al. 2005).

The most commonly identified obstacles for proper monitoring of biophysical, socioeconomic and governance indicators are the need for additional financial resources, time, and technical capacity and infrastructure. In an attempt to evaluate the management effectiveness of 18 MPAs around the world, Pomeroy et al. (2005) found that socioeconomic and biophysical indicators (especially the latter) were the most expensive and time consuming to measure, due to the high cost of materials, time required for field trips and survey preparation, and the cost and time required to mobilize necessary technical support. On the contrary, governance indicators were four to five times less than what was required with biophysical and socioeconomic indicators. Financial support is lacking from many MPAs and monitoring has to compete with shorter-term priorities. This results to monitoring designs that are not defined according to scientific standards but are greatly constrained by the availability of financing and technical support.

Consequently, in most cases a singular 'inside-outside' survey is conducted after the establishment of MPAs, aiming to compare sites within the reserve with sites just outside. The most common biological indicator is the population density of selected species, mainly fish of commercial importance or key benthic species. Such investigations are generally confounded by intrinsic natural variability between sites inside and outside MPAs and by the absence of information before the establishment of protection.

1.2.7 MPA EFFECTIVENESS AND GOVERNANCE

1.2.7.1 Socioeconomic evaluations

MPAs of various types are a form of resource management that regulates human activities in particular locations (Sale et al., 2005). International conventions aimed at the protection of marine habitats, like the Barcelona Convention signed in 1982 , oblige the signatory parties not only to take all appropriate measures to protect those marine areas which are important for the safeguarding of the natural resources but also to safeguard the cultural heritage.

Indeed, according to Mangel et al. (1996) all conservation problems have scientific, socialand economic aspects the relative mix of which will vary, but it is essential to recognise all three components. Mangel et al. have listed a series of principles to be evaluated before the establishment of a marine protected area and Pomeroy et al (2004) have dedicated a section to socioeconomic indicators in their book "How is your MPA doing?" Despite the numerous suggestions, principles and recommendations, a) assessment of the possible sociological effects of resource use almost neverprecedes either proposed use and proposed restriction or expansion of ongoing resource use; b) regulation of the use of living resources almost never takes into account the sociological influences that directly or indirectly affect resource use; c) the full range of knowledge and skills from the social sciences is almost never brought to bear on conservation problems. To date, only in very few cases have socioeconomic and cultural aspects been considered before MPA establishment (Badalamenti et al. 2000; Charles and Wilson, 2009; Vella et al. 2009) and in some cases an assessment has been carried out afterwards (Charles and Wilson, 2009; Himes, 2003, 2007; Mangi and Austen, 2008).

In most developing nations, communities in the vicinity of a reserve expect it to contribute to the social and economic well-being of local people (Ezebilo and Mattsson, 2010). However, the more remote the reserve, the more rural are the human settlements around it and, often, the less know-how the local community has to exploit the economic potential of the reserve (Badalamenti et al. 2000). Newcomers may then take advantage of the opportunities available, severing the reserve-local community link. While the socio-economic benefits of an MPA can be perceived as short term by residents in non-isolated areas, they are considered to be long or very long term by communities around isolated MPAs. In any case, it is now accepted that the economic benefits to be drawn from the presence of an MPA are predominantly from tourism, not from fisheries.

1.2.7.2 What have we learnt?

Studies carried out in Mediterranean MPAs up to now have dealt mainly with their ecological aspects rather than the social, political and economic implications (Richez 1991). Despite the fact that MPAs have existed in one form or another for some time in the Mediterranean (Boudouresque 1994), few data are available (Ramos 1991) and few studies have been carried out to assess the socioeconomic aspects connected to protection (Badalamenti *et al.* 1998). Moreover, most of these studies contain limited and qualitative analyses or are too short term.

It is important to take into account the human component of MPAs and those areas directly or indirectly influenced by them (Fiske 1992). Keeping local communities informed and encouraging them to participate throughout all the stages of planning, establishing and managing MPAs contributes substantially to the likelihood of long-term success of the initiative. In Ireland, choice experiment surveys showed that commercial fishermen are not the only stakeholders with an interest in deep sea coral reefs. The resources protected by MPAs often have a potentially larger non-use value than use (market) value. This non-use value requires determination to help balance the costs and benefits of MPA implementation. The surveyed part of the Irish public was in favour of the protection of cold-water corals. The preferences and political endorsement lay with selective exclusion of trawling rather than the designation of a no-take zone, which endorses the applicability of the findings of the Norwegian redfish modelling. The stance of the Irish electorate in terms of the area of reefs protected was, however, more precautionary and all-inclusive, for which they were prepared to pay personally. The findings of this study also provide advocates of MPA designation for the corals stated endorsement of their actions. It is also feasible for policy makers to estimate the level of support for various combinations of the management options or use the results as controls within a modelling context. Reference to environmental valuation techniques and particularly, choice experiments is a critical part of pre-project evaluation (PROTECT 2009).

MPAs can lead to a recovery in the productive potential of fishery resources. Increases in the number and biomass of many species which occur in MPAs (Buxton & Smale 1989; Cole *et al*. 1990; Polunin & Roberts 1993) and fishery resources will in many cases spill over into surrounding areas (Rowley 1994; Russ & Alcala 1996; Watson *et al*. 1996). MPAs will often have significant impacts on the local and regional economy, typically as a result of expenditure derived from tourism and especially from diving-related activities. This in turn may generate multiplier effects, so that initial expenditure in the tourist industry creates further rounds of spending that raise incomes in other sectors. There are still recurring worries relating to the use of MPAs for tourism ends (Wilkinson *et al*. 1994). Diving activities in particular are responsible for damaging benthic communities (Harriot *et al*. 1997) and the danger exists that over-exploitation of the economic resource which tourism represents could jeopardize its future viability (Davis & Harriot 1996).

All these findings indicate that the effectiveness of a MPA is directly linked to how human activities are being managed, how different interests and priorities are being balanced against each other, and how conflicts are being addressed in a specific context. They also show that engagement with marine resource users, stakeholders and the public in governing a MPA can enhance the long-term effectiveness of a MPA. Such issues are central themes in the study of MPA governance, which will be further explored in section 1.2.8.

1.2.7.3 Drivers of designation: Evolving, overlapping nature of MPA objectives within wider MSP frameworks.

In the past decades fisheries stakeholders and nature conservation proponents have been the main drivers of the general debate over MPAs. Meanwhile, a number of other sectors have made an ever-growing number of claims in marine areas for e.g. aggregate extraction, pipelines and wind farms. The designation of wind farm sites and the construction of turbines at sea in particular, driven by the current demand for clean energy, have had a highly significant effect on the political landscape surrounding sea use.

Increasingly, conservationists and fisheries stakeholders can be found on the same side of the fence in public debates (e.g. http://www.spiegel.de/international/germany/0,1518,683040,00.html) over impacts of e.g. construction plans or importance of specific marine sites, i.e. revealing that marine habitats and species can no longer be considered in isolation in fisheries management and nature conservation regimes. In order to optimize space allocation and to add sufficient weight to claims, these sectors will likely need to identify marine areas that represent essential sites for both sensitive species and habitats as well as for living resource management. Overlaps in geography and/or within sectoral objectives are likely to facilitate the discovery of mutual opportunities and benefits that are important to consider in relation to MPA development within wider MSP frameworks.

One example of mutual opportunity may be found in the designation of protected areas or MPA networks to conserve *essential fish habitats*, which in the US Magnuson-Stevens Act of 1996 are defined as the waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity. There is evidence (e.g. Armstrong & Falk-Petersen 2008; Wieland et al. 2009; Hunter & Sayer 2009) that habitat quality and complexity play an important role in the population dynamics of fish species and in overall species diversity.

Gaines et al (2010) have formulated guidelines for designing MPA networks with the aim to reduce or eliminate tradeoffs in achievement of fisheries and conservation goals. On the basis of biology and resource exploitation patterns the guidelines address issues pertaining to scale, spacing, location and configuration of MPA networks to enhance biological conservation and reduce fishery costs and/or increase fisheries yields and profits. Ultimately, the assessment of these principles is currently limited by the absence of data from MPA networks that have been implemented in practice.

1.2.7.4 National Example of drivers: Diving Parks in Greece

Until recently, scuba diving in Greece had been highly restricted by numerous spatial regulations, mainly on the grounds of protecting marine antiquities. In 2005, a new law (L. 3409: *Recreational Scuba Diving and Other Regulations*) withdrew most of the existing restrictive regulations for SCUBA diving and, at the same time, introduced a new concept, that of “Recreational Scuba Diving Parks” (DP), therein vaguely determined as coastal areas strictly reserved for scuba and free diving touristic activities, managed and exploited by either public, private or joint agencies.

This initiative was originally aimed at promoting scuba diving tourism in Greece, as DPs are listed under the category of “Productive Activities” in the framework of National Spatial Planning of Tourism Development. Although little information is given about the very nature of this institution or the criteria that will determine its *modus operandi*, DPs are rather envisaged as small coastal marine areas (about 1 square nm, and within depths between 0-50m) closed to all fishing activities, guarded and managed by trust companies and only accessible to divers for some certain admission charge and under the surveillance of the managing company. Thus, due to the regulation of human activities and exclusion of all fishing and other extractive activities, DPs are clearly a type of small-scale MPAs.

Given the originality of this approach, several public discussions followed the passage of this legislation. Concern was raised about the whole process of creating DPs (basic principles for site selection, operational framework, stakeholder participation, management and monitoring) and the criteria that should be met before mass diving tourism is to be attracted in specific Greek coastal areas. Taking into account international paradigms, as well as the numerous scientific studies on the impacts of diving tourism in Marine Protected Areas (e.g. Sala *et al.*, 1996; Garrabou *et al*., 1998; Coma *et al*., 1999; Zabala, 1999), there seem to be several issues that are yet to be addressed within a national scale, before the new legislation comes into effect. Some of these issues involve:

- integrating local communities and adopting suitable preventive and compensative measures to timely resolve possible conflicts, especially with coastal fisheries
- formulating a national planning scheme for designating and siting the DPs, based on sound ecological and socio-economic criteria
- establishing a monitoring framework to assess the impact (positive or negative) of DPs to habitats and populations of key-species
- providing the appropriate legal and scientific framework to ensure the environmental sustainability of DPs

Should the above issues be adequately considered, this initiative could fully comply with the rationale of Integrated Coastal Zone Management (UNEP/MAP/PAP, 2001), and DPs might indeed prove as an effective and valuable tool for sustainable tourism development, apropos enhancing marine conservation.

Till this moment however, no further legislative clarifications have been given, no relevant pilot studies have been implemented and no DPs have yet been designated in Greece. There are several on-going projects by private or public agencies to create DPs and a few applications have already been submitted. It is expected that the first DPs will begin to function during the next couple of years.

1.2.8 GOVERNANCE ISSUES: WHAT DOES 'COMBINING TOP-DOWN AND BOTTOM-UP APPROACHES' MEAN?

The following is a summary of key findings from an UNEP project on MPA governance, in which a framework was developed to answer the above question through analyzing 20 case studies of MPAs in different contexts. For more information on the project and case studies, please visit www.mpag.info.

How best to govern protected areas has long been a debate in both academic and policy communities. Such debates are taking place in the much wider context of debates about how we should go about managing people and the social, economic, political and bureaucratic systems of which they are a part. These debates are not confined to recent times, Plato's philosophies (360BC) considering the role of the state in 'steering' human affairs, the word 'governance' being derived from his use of the Greek verb 'to steer'. Since Plato, many other influential thinkers have put forward various observations, ideals and theories concerning the relative importance of the roles of different approaches to governance:

- state control – government and bureaucracy
- market forces – capitalism and economies
- public interests – people and civil society

There is a growing recognition in governance debates that there is a need to move beyond ideological arguments as to which approach is 'right' and, instead, develop governance models, frameworks and approaches that combine the 'steering' role of states, markets and people. Such integrated, pragmatic perspectives enable us to move on from ideological debates about whether we should rely on the strong hand of state power, the invisible hand of market forces or the hands of the people, and to consider how the three approaches can be effectively combined. These three perspectives on environmental governance are represented in the more specific context of protected area governance, where they are discussed in terms such as the following:

(1) Top-down: the need for state control through laws and other regulations to ensure that biodiversity and natural resources are actually 'protected' against degradation and destruction;

(2) Bottom-up: the need to adopt community-based approaches to protected area governance that decentralise decision-making processes and empower local people by involving them in deliberations and decisions (*e.g.* community-based management – CBM).

(3) Market incentives: the need for economic initiatives to support alternative, compatible livelihoods, *etc.*.; the need to attach an economic value to biodiversity in terms of natural capital and ecosystem services as a means of providing for balanced decisions, through environmental cost-benefit analyses, that might otherwise favour exploitation; the need to attach property rights to environmental resources is also often emphasized as a means of improving governance by using market incentives to promote economic rationalism;

Collaborative management or co-management is a common concept or narrative that is employed in natural resource governance, including protected areas, to explore the challenges of combining these three approaches, whereby local communities and the state work on a partnership basis to sustainably exploit natural resources and/or conserve protected areas, potentially involving all three of the approaches listed above. Co-management does, however, arguably simply serve as a new framing device as to the relative emphasis that should be placed on the three general approaches outlined above, in the same manner as for the concepts of sustainable development and the ecosystem approach.

Protected areas are an important focus for debates concerning how these different approaches can be combined to promote effective governance. It is clear that debates concerning the relative merits in governance of the steering role of the state, the ‘invisible hand’ role of markets and the democratic role of people are occurring with a focus on Marine Protected areas (MPAs), as are related debates as to whether wider sustainable resource exploitation approaches through the assignation of property rights to local people to promote community-based management might be more effective than narrower site-based approaches such as MPAs. It is widely accepted that the co-management of MPAs is the way forward, but there are many different interpretations of this concept and it is applied in many different ways amongst MPAs in different contexts. One way of considering the challenges of co-managing MPAs is to consider the question:

What does ‘combining top-down and bottom-up approaches’ actually mean?

Answers to this question are often context-specific, depending significantly on the challenge and the attributes of the local context in which the challenge has emerged, as well as the national and international contextual attributes, particularly related to strategic statutory biodiversity conservation obligations. This question, therefore, needs to be addressed on an open and realistic basis, rather than on the basis of theoretical and ideological ideals by which a particular governance approach might be considered to be ‘right’ or ‘best’.

One way of assessing the effectiveness of MPA governance in an open and realistic manner is to look at the different sources of ‘steering’, deriving from state hierarchies, market forces and/or participation of communities, in terms of **incentives, which are defined as:**

Protected area governance approaches that are instrumentally designed to encourage people to choose to behave in a manner that provides for certain policy outcomes, particularly biodiversity conservation objectives, to be fulfilled through collective actions.

These were divided into five categories that can be related to the three modes of governance discussed above:

Economic incentives – using economic and property rights approaches to promote the fulfilment of MPA objectives: **Market control**

Interpretative incentives – promoting awareness of the conservation features of the MPA, the related objectives for conserving them, the policies for achieving these objectives and support for related measures: **supporting all three approaches**

Knowledge incentives – Respecting and promoting the use of different sources of knowledge (local/traditional and expert/scientific) to better inform MPA decisions: **supporting all three approaches**

Legal incentives – use of relevant laws, regulations, *etc.* as a source of 'state steer' to promote compliance with decisions and thereby the achievement of MPA obligations: **State control**

Participative incentives – providing for users, communities and other interest groups to participate in and influence MPA decision-making that may potentially affect them in order to promote their 'ownership' of the MPA and thereby their potential to cooperate in the implementation of decisions: **people control**.

Exploring MPA governance through investigating how these incentives are being combined to steer relevant processes enables us to overcome the divide between top-down and bottom-up approaches, and to explore how the strengths from both approaches can be combined in a strategic manner to address the challenges in governing MPAs in an effective, fair and equitable way. Through the lens of the incentives, a variety of different governance approaches have been recognized in addressing MPA-related conflicts and supporting the achievement of MPA objectives. These include:

Approach I

MPAs managed primarily by the government under a clear legal framework (government-led)
MPA governance under this category is characterised by having a well established legal framework, with clearly defined MPA objectives, restrictions on different uses, jurisdictions and responsibilities of different government institutions, and rights and obligations of the public. Legal incentives are the key drivers in most MPA-related processes, including user and public participation, which is provided for and guided by legal provisions as a means to promote transparency, equity and compliance in MPA management and enforcement. It is important to note that the MPAs categorised as government-led also employ the other four categories of incentives and that having a strong government lead certainly does not preclude opportunities for user participation, though legal incentives are the most important source of steer (Figure 1.2.1).

Examples of MPAs adopting this governance approach include the Great Barrier Reef Marine Park (Australia), the European Natura 2000 system, Darwin Mounds candidate Special Area of Conservation (UK), North-East Kent European Marine Site (UK), Wash and North Norfolk Coast European Marine Site (UK), California Marine Life Protection Act (US) and US National Marine Sanctuary System (US).

Approach II

MPAs managed by the government with significant devolution and/or influences from private organisations (devolved governance). MPA governance under this category is characterised by a sharing of authority and responsibilities between central/federal governments and lower levels of government, or between government institutions and NGOs/private entities. MPAs are managed in accordance with formal regulations and/or through partnerships and negotiations between different parties. A variety of governance incentives are employed in MPAs that adopt this approach, depending on the context and main focuses of MPA-related efforts, but economic incentives were most frequently cited as being currently used whilst legal incentives were most frequently cited as being needed to improve governance (Figure 1.2.2).

Examples of MPAs adopting this governance approach include Sanya Coral Reef National Marine Nature Reserve (China), Seaflower Marine Protected Area (Columbia), Galápagos Marine Reserve (Ecuador), Karimunjawa Marine National Park (Indonesia), Wakatobi National Park (Indonesia), Tubbataha Reefs Natural Park (the Philippines), and Ha Long Bay World Natural Heritage Area (Vietnam).

Approach III

MPAs managed primarily by local communities under collective management arrangements (community-led). MPA governance under this category is characterised by local communities taking a lead in the conservation and sustainable management of marine resources, which is essential for the long-term social and economic well-being of communities. Community institutions (*e.g.* local fishing cooperatives) are often granted a significant level of autonomy to collectively decide the rules governing MPA management. External organisations, such as government departments and conservation NGOs, may have an important role in enabling and reinforcing such community initiatives, and ensuring that such community efforts are consistent with existing legal and policy frameworks, including fisheries and biodiversity conservation objectives/obligations, that govern the management of marine resources at a national or other wider scale. Again, all categories of incentives are employed but economic incentives were most frequently cited as being used to promote community stewardship of MPAs whilst legal incentives were most frequently cited as being needed (Figure 1.2.3). MPAs adopting this governance approach are Isla Natividad (Mexico) and Os Miñarzos Marine Reserve of Fishing Interest (Spain).

Approach IV

MPAs managed primarily by the private sector and/or NGOs granted with property/management rights (private-led). MPA governance under this category is characterised by non-governmental and/or private organisations taking the main responsibility for MPA management and enforcement. Such organisations are often granted with permanent property rights or temporal management rights to a particular area of sea, where they carry out conservation and resource management work. Such organisations work independently of their own volition, but often collaborate with public institutions to enhance the effectiveness of their conservation efforts. Incentives employed to steer MPA management vary between MPAs that belong to this category depending on the context as well as the core values of the leading organisation, but economic incentives were most frequently cited as being used to promote effective governance whilst legal incentives were most frequently cited as being needed (Figure 1.2.4). MPAs adopting this governance approach are Chumbe Island Coral Park (Tanzania) and Great South Bay Marine Conservation Area (United States).

Approach V

No clearly recognisable effective governance framework in place. The development of MPA governance in this category is hindered by a lack of political will, leadership and capacity from all levels to develop effective governance structure and arrangements that would support the achievement of any MPA objective, often in the face of strong driving forces counter to conservation. Few incentives are successfully applied to address conflicts and steer MPA processes in this category and interpretative, knowledge and legal incentives are most frequently cited as being used, whilst legal and economic incentives are most frequently cited as being needed to improve governance (Figure 1.2.5). MPAs adopting this governance approach are Baleia Franca Environmental Protected Area (Brazil), Pirajubaé Marine Extractive Reserve (Brazil), and Cres-Lošinj Special Marine Reserve (Croatia).

Overall, all five categories of incentives have been almost evenly applied to steer MPA governance in the case study MPAs, based on the sum of the frequency with which individual incentives within each category are cited as being used (Figure 1.2.6), though there are differences in this respect between the case study governance approach groups. In general, across all 20 case studies, economic and legal incentives were most frequently cited as being used, though the differences are minor. There are, however, larger differences in the frequency with which incentives within each category are cited as being needed. It is particularly notable that legal incentives were cited as being needed to improve governance more often (f=38) than the other four categories of incentives combined (f=27) (Figure 1.2.6). This illustrates the importance of legal incentives for improving and reinforcing governance frameworks, based on this sample of 20 case studies analysed through the MPAG framework. This is partly because successful implementation of all governance approaches requires a sound legal basis, such as legal provisions to ensure public rights to participate in governing processes, to protect community property rights to natural resources against corporate development, and most importantly, to prevent over-exploitation by incoming and local users that may lead to catastrophic declines in marine resources vital to the livelihoods of coastal communities.

Notwithstanding the differences in context and the governance approach adopted amongst the case studies, some key factors can be identified as being essential to developing good MPA governance in most cases, these include:

Provision of sustainable economic development opportunities within or adjacent to MPAs

Fair sharing of economic benefits and costs from MPAs

Public communication, education and awareness-raising on the importance/vulnerability of marine ecosystems and the benefits of MPAs

Use of all available information and knowledge to guide/inform MPA decision-making
Political will and capacity for passing and enforcing laws and regulations that provide for effective MPA management
Provision of opportunities for different user and public groups to participate in MPA decision-making processes
Leadership from individuals and organisations within the government, NGO and private sectors, academic institutions, and/or local communities
Strong sense of stewardship of the MPA among communities and users

This study also shows that the same type of governance incentives can be used to empower different people (the most important being the state and local users) in MPA management, depending on how and by whom they are being used. It is clear that MPA governance should be considered in terms of how incentives can be combined, rather than whether any particular category of incentives is 'correct', and that many incentives can be employed to support both top-down and bottom-up approaches. Accepting that all five categories of incentives potentially have a role to play in any given MPA context, the emphasis becomes one of combining the use of as great a **diversity** of incentives as feasible in order to develop a governance framework that is more **resilient** to the perturbing effects of driving forces (global fish markets, corporate tourism, incoming users, *etc.*). As such, **discussions concerning theresilience of governance frameworks resonate with discussions concerning the resilience of ecosystems**.

In a similar manner, this **study concludes** that it is the **combination and inter-connection of different incentives from different categories that makes governance** frameworks more resilient, with legal incentives constituting strong links that reinforce the governance framework against potential perturbing driving forces, and incentives from the other four categories constituting weaker links, without which the framework is inherently unstable. Simple governance frameworks, consisting mainly of incentives from any one category, including strong legal or participative incentives, will not be resilient to the potentially negative impacts of driving forces on marine biodiversity and resources.

Resilience in MPA governance frameworks is woven by complex webs involving incentives from all five categories. Recognition of this addresses the questions 'what does combining top-down and bottom-up approaches mean?' and will also allow us to move on from debates about which category of incentives is 'best' towards more practical debates about how incentives can be combined and inter-linked in order to develop resilient governance frameworks.

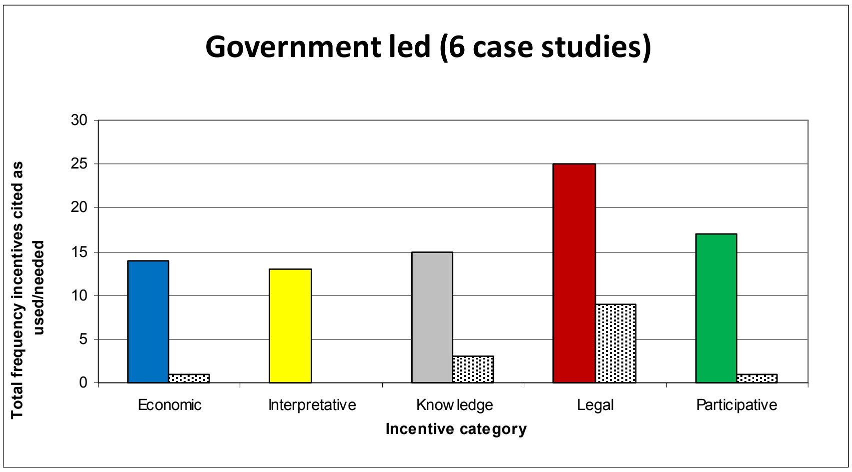

Figure 1.2.1 Incentives used/needed – government-led case studies

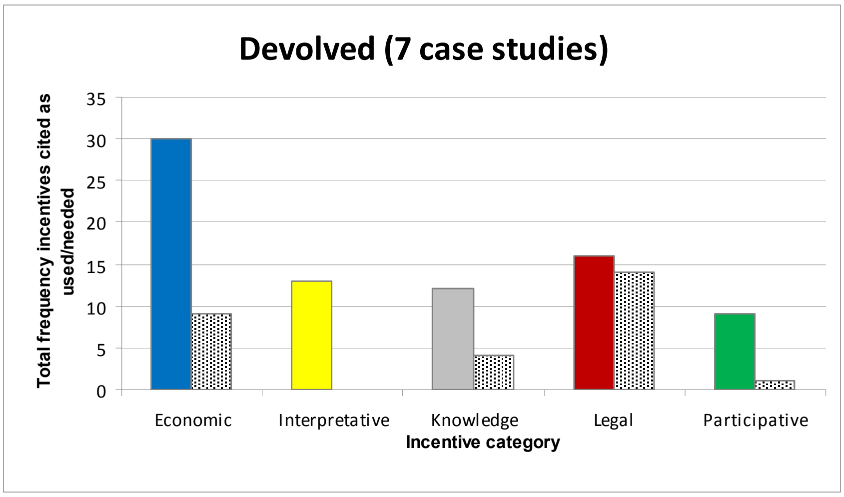

Figure 1.2.2 Incentives used/needed – devolved case studies

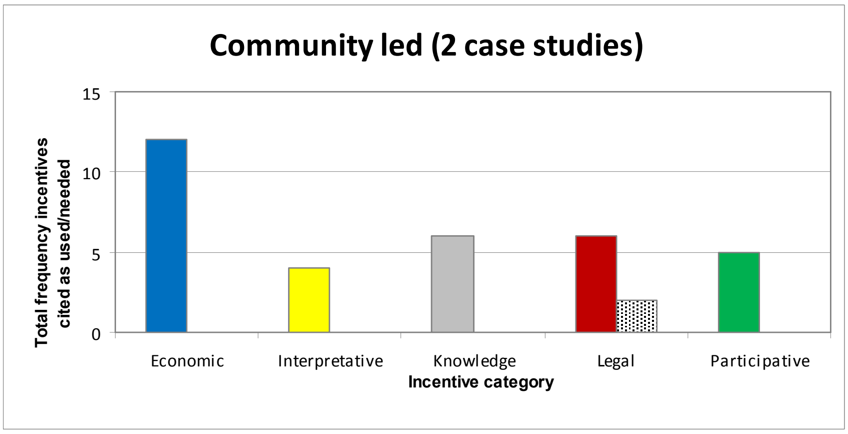

Figure 1.2.3 Incentives used/needed – community-led case studies

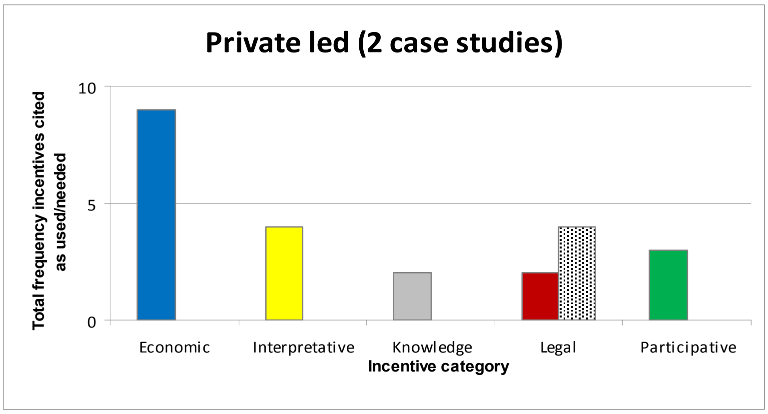

Figure 1.2.4 Incentives used/needed – private-led case studies

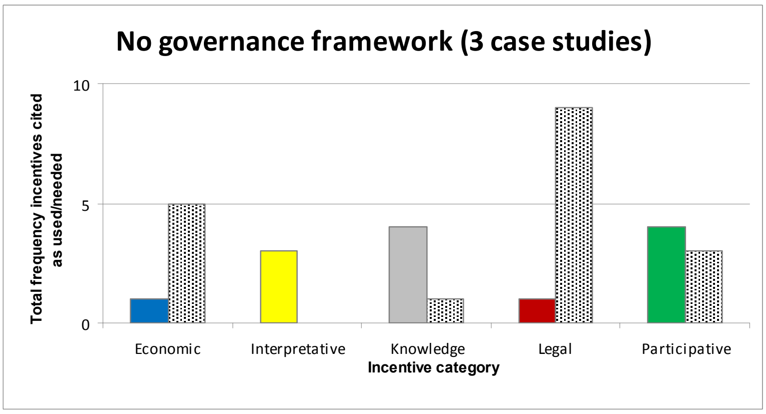

Figure 1.2.5 Incentives used/needed – no effective governance framework case studies

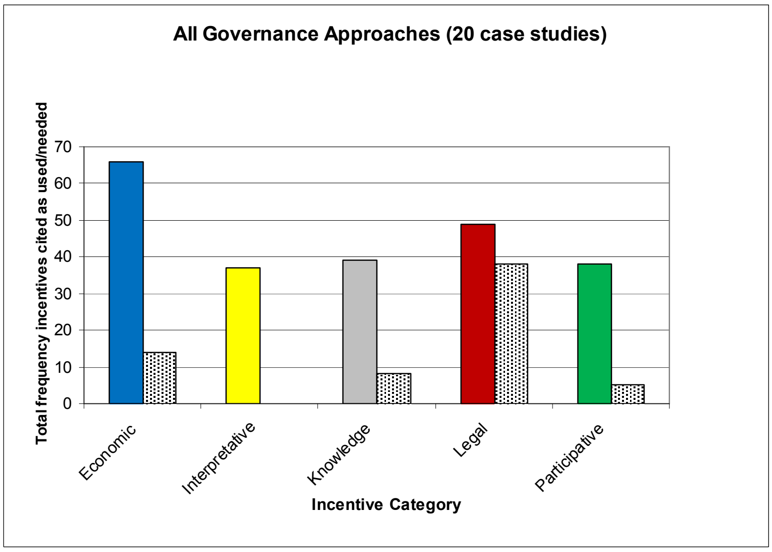

Figure 1.2.6 Incentives used/needed – all case studies

Key for Figures 1-6: coloured bars represent incentives *used*, whilst grey-hashed bars represent incentives *needed,* as identified by the project participants.

Greek MPAs: national policy and current status

The main effective legislative framework for the establishment of MPAs in Greece is Law 1650/1986 for the 'Protection of the Environment'. National Parks are a special category of this law, and some of the established

National Parks include important marine coastal areas. When the Park exclusively or mainly occupies marine areas it is classified as a National Marine Park. There are currently two established National Marine Parks in Greece, the National Marine Park of Zakynthos (NMPZ) in the Ionian Sea and the National Marine Park of Alonnisos - Northern Sporades (NMPANS) in the Aegean Sea. National Parks are managed by Management Bodies that have the responsibility and authority for the design and implementation of management plans, establishment of operational rules, monitoring and surveillance of the park, promotion of scientific research, and public awareness.

Among the 239 Natura-2000 sites in Greece, 114 include marine areas of a total surface of 6,344 km^2, covering 5.5% of the territorial waters. In most of them the marine part extends offshore to a depth of 50 m. To have a protection status, these marine sites need to be announced protected areas according to the national legislation (Law 1650/1986) and a Management Body should be established. This has not yet been promoted and thus the majority of the Natura-2000 sites is not included in legally designated protected areas. Furthermore, for many of the marine Natura-2000 sites, habitat mapping and basic knowledge of their environmental status is lacking. Most of the protected areas still require management plans and management responsibilities often rest with many authorities at central and local levels, with consequent overlapping, coordination problems and weak enforcement.

NMPZ and NMPANS were established rather on the narrow scope of protecting endangered marine species than on an integrated ecosystem based approach. The main aim of NMPZ was to conserve the nesting beaches of the endangered loggerhead sea turtle *Caretta caretta*,as the bay of Laganas in NMPZ hosts the most important known nesting aggregation of the species in the Mediterranean. NMPANS was originally established to protect the indigenous populations of the critically endangered Mediterranean Monk seal *Monachus monachus*. Management plants and monitoring frameworks in the two national parks still focus on these two species and there is no regular monitoring of other key species and habitats, biodiversity, ecological status, and overall effectiveness of MPAs.

The establishment of marine protected areas in Greece has mostly been a top-down process, often leading to serious conflicts with local communities and stakeholders. The role of environmental NGOs (mainly Archelon in NMPZ and MOM in NMPANS) has been central and crucial. In both national marine parks, fishing-conservation and tourism-conservation are the major conflicts that in the last decade instigated strong reactions involving even violent acts (e.g., attacks on NGO volunteers and the personnel of the parks, arson of a forest area in the terrestrial part of NMPZ). Local stakeholders, land-owners, fishermen and tourism entrepreneurs perceived the establishment of both National Marine Parks as a serious threat to local economic interests. These groups, often supported by local authorities such as municipalities, prefectures, port and police agencies, mounted resistance to the implementation of laws that restricted their use of local resources. The lack of appropriate compensation strategies for affected stakeholders was also a reason for such strong reactions.

Although Greece has now moved to a more integrated and participatory management approach, pre-existing conflicts and lack of trust between the Management Bodies and local communities remain among the main challenges of Greek MPAs. Currently, the Management Boards of the two National Marine Parks are composed of representatives from national, regional and local authorities, local stakeholders and NGOs and great effort is made to gain trust and support from local communities.

MPAs in Denmark

The most common way in which nature has traditionally been protected through spatial management measures in terrestrial and marine areas of Denmark has been through the establishment of nature reserves. There are approx. 100 of such reserves in Danish waters and many of these are rooted in legislation pertaining to nature protection or (especially) hunting (water fowl). In relation to the former, Denmark has unique nature protection legsilation in that it applies to the entire exclusive economic zone, albeit it is rarely used. In marine areas, most reserves are designated to protect populations of migratings birds and as a result most of these sites are shallow coastal areas where migratory birds rest, breed or forage on blue mussels, eelgrass and other benthic fauna and flora. Other areas include seal reserves. However, of all Danish reserves in marine areas, only two have been formally designated to protect what may be considered *marine* values (seafloor, benthic species, etc.). One such reserve serves as a reference area to monitor effects of coastal blue mussel fisheries, in the other area bottom trawling has been forbidden in a very small area surrounding submarine structures created by leaking gases (“bubbling reefs”).

It was not until 1992 that Denmark, through the adoption of the Habitats Directive, became obligated to protect marine habitats and species in all territorial waters. The first sites were formally designated in 1998 and new sites have been proposed in 2003 and 2008. These Special Areas of Conservation (SACs) currently cover 16.142 km^2 (or 15,3 %) of Danish waters. Special Protection Areas (SPAs) designated under the Birds Directive cover 12.112 km^2 (or 11,4 %) of Danish waters. However, there is a large overlap between the two, and only 6.610 km^2 or approx. 40% of SACs are single-status SACs under the Habitats Directive.

The legally binding Natura 2000 directives and their translation into national law remain the only drivers of protected area establishment in Danish seas. However, this may change due to the provisions of the ECs Marine Strategy Framework Directive (MSFD), which mentions MPAs as one tool through which Member States may achieve good environmental status on their regional waters. As there are many habitats in Danish seas that are not included in the Natura 2000 directives and thus are not protected in SACs, the MSFD may perhaps provide the legal basis for additional protected area designations.

There are no formal transnational MPAs designated between Denmark and her neighbours apart from the Trilateral Cooperation on the Protection of the Wadden Sea, an agreement signed in 1978 by the Netherlands, Germany and Denmark to coordinate activities and measures for a comprehensive protection of the Wadden Sea. A number of Natura 2000 sites in Danish waters are situated adjacent to Natura 2000 sites in German waters, but there is no formal cooperation of management planning in place.

Several of the Danish Natura 2000 sites also have status as HELCOM Baltic Sea Protected Areas (BSPA). However, BSPAs are not legally binding and they have therefore received less attention since the introduction of Natura 2000. It is the responsibility of the Danish Ministry of Environment, Agency for Spatial and Environmental Planning to designate Natura 2000 sites and to manage these sites. The provisions of the Natura 2000 directives, however, have been written into sectoral legislation. For instance, according to the Danish fisheries legislation it is the responsibility of the Ministry of Agriculture and Fisheries to ensure that the fisheries sector does not prevent the achievement of favourable conservation status for the species and habitats including those within Natura 2000 sites.

Land-based tourism (as opposed to e.g. diving) may be considered a priority in relation to protected area establishment in the coastal zone. In addition to existing (mainly bird) reserves in coastal areas, where Danes and foreign visitors come to enjoy nature and to watch birds, national parks have in recent years been established (including in coastal areas) to enhance public awareness and create nature-based recreational opportunities. None of the proposed national parks with significant marine components have succeeded in gaining local support and have therefore been rejected. Beyond the coastal zone, tourism is not a significant driver of protected area establishment in Denmark.

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1.3 SOCIO ECONOMIC VALUATION OF SPATIALLY MANAGED AREAS

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1.3.1. INTRODUCTION

There is little doubt that society places value in the preservation and maintenance of the environment and the marine environment is no exception.This may seem like a banal and obvious comment.However the nature and distribution of these values is a complex matter.The marine environment is a source of valuable resources fish, shellfish, marine algae, and mineral resources etc all of which command prices on markets.As well as providing exploitable resources the marine environment is important to the functioning of the economy. It is a highway for international trade, a playground for tourists and it assimilates pollution from the production cycle.The oceans also provide life support systems upon which humanity relies:- the sea controls the earth's climate and it plays an important role in nutrient cycling and helps maintain the atmosphere.Society also places value on the very presence of marine biodiversity, this may be a part of long held cultural traditions or the more modern 'culture' of conservation.Increasingly environmental legislation is used to protect the marine environment; protected status reflects the value society places in the environment.Some people question the effectiveness of environmental legislation suggesting that we under-value the environment.The media often focuses on this concern highlighting pollution and the "destructive" effects of human exploitation.This concern for the marine environment is itself just another expression of value.

It is clear that the pattern of values in the marine environment is complex and that these values may conflict with each other.The active management of marine resources (including the designation of MPA's) <u>will</u> cause a shift in the distribution of these values for example restrictions placed on fishing activity to enhance conservation values. Unless there is a pure *Pareto efficient* move (i.e. values are enhanced or remain the same) the management of marine resources will enhance some values and diminish others.Put simply there will be winners and losers.There is a generally presumption that good environmental management decisions involve producing a net gain i.e. the benefits of the decision outweigh the costs.

The purpose of the following sections is to explore methodologies for the assessment of values in the marine environment and how these methodologies may be applied in the context of MPA's.Before examining specific methodologies it is necessary to consider the nature of environmental values and how they are categorised.

1.3.1.1 Values, value expressions and utility

In economics the notion of value is closely linked the concept of utility.Utility was first properly described by Jeremy Benthan as "*that property in any object whereby it tends to produce benefit advantage, pleasure, good or happiness*" (Bentham 1789).For Bentham good government and good legislation was about maximizing net utility and, generally speaking, this remains the implicit objective of government policy in democratic society.Building on Bentham's ideas William Jevons based his *Theory of Political Economy* on a calculus of pleasure and pain the objective being to maximise utility by purchasing pleasure at the lowest cost of pain (Jevons 1871).This laid the foundation of what we now routinely call cost benefit analysis (CBA) formalised by Pigou (1932) and specifically Kaldor (1939) and Hicks (1939).CBA examines values (generally expressed in monetary terms) to determine the changing distribution of utility produced by a proposed course of action.

What is the relevance of this discussion in the context of Spatially managed areas? The discipline of economics suggests that the values that society places on the marine environment, and the resources it produces, is the sum of the utility experienced by individual people (see Nunes and van den Berg 2001).Importantly this utility is entirely *anthropocentric* in nature. This means that all values are ultimately rooted in individual experiences.This view of utility eschews the notion of value systems outside the human condition.The possibility of *ecocentric* values (or intrinsic values) implicit in nature, suggested by some deep green ecologists (e.g. Naes 1988), is rejected by the economic model.The extension of ethics to the environment (see Leopold 1945), or the "*obligations to nature*" sometimes implied in the sustainable development debate (Dobson 1996) are equally disregarded. Even so called 'scientific values' or 'conservation values' are understood simply as expressions of individual human utility.Derous et al (2007(a), 2007(b)) examine the concept of biological valuation in more detail, identifying what they describe as "valuation criteria".These criteria (largely based on MPA selection criteria) are intended to assess biological value "without reference to anthropocentric use".This criteria include: (i) rarity, (ii) aggregation, (iii) fitness consequences; (iv) naturalness; (v) proportional importance.This approach has the potential to deliver a decision tool for transparent and consistent MPA site selection.However it is debatable whether this represents "intrinsic value" as claimed by the authors (Derous et al (2007(a)), or whether it simply represents the values of conservation biologists. From the perspective of economics all value changes generated by an spatially managed area, and its management, are human in origin and no distinction is made between the values of one stakeholder group over another.

1.3.1.2 Sustainable development

This anthropocentric interpretation of utility and value described above has important implications for the concept of sustainability. From this worldview decisions are made through the lens of the current generation and its values.The 'anticipated' values of future generations are accounted for only in-so-far as they affect the utility the current generation. It is possible to argue that this is at odds with the post-Bruntland concept of suitability i.e. maintaining a perpetual stock of environmental capital to bequeath to future generations.

A Rawlsian approach to environmental justice suggests that environmental decision making should place equal weight on values of all generations (Rawls 1971). While it may be possible make a strong philosophical argument why such an approach may be 'just' (unable to put ourselves in the shoes of future generations) it is of limited real practical use for decision making.When CBA includes future values they are expressed in present value terms.This is achieved by the process of discounting (levelising) which, attempts to reflect individuals' preference for consumption now over consumption in the future, by deflating future values.Consequently in any CBA future values beyond the life of one generation are generally trivialised in the decision making process.

1.3.1.3 The search for values

The introduction to this section suggested that society holds a multiplicity of values in the various utilities provided by the environment.It is fair to say that for many decades mainstream economics concerned itself only with values expressed on conventional markets where goods and services are traded, i.e. the cash economy.In terms of the environment this typically meant the value of resources (fisheries etc).However the idea that the environment may generate values additional to those expressed on money markets has its roots in welfare economics.Pigou (1932) discussed the use of environmental taxes to maximise social utility.Ciracy Wanturp (1947) explored the possibility of valuing public good aspects of national parks in the USA. Interest in "environmental values" and their assessment grew in the 1970's and 1980's and a new branch of economics we now call *environmental economics* emerged into the mainstream.This was a response to a number of factors including inter alia:

Increasing use of CBA to assess public expenditure project

The 'polluter pays' concept and the extension of environmental liability legislation to include non market aspects of the environment.

A renewed enthusiasm for market driven decision making;

A drive for market based instruments to resolve environmental problems (e.g. taxes and permits) in an attempt to internalise the external (social) costs of pollution.

1.3.1.4 A typology of values

Environmental economics uses various typologies to categorize the range of values present in the environment.Catching the mood of the times, David Pearce did much to popularise Total Economic Value (TEV) as an appropriate metric to measure the effectiveness of public policy in the environment (Pearce 1989).TEV consists of the sum of all market values (e.g. fish catch) and non market values (e.g aesthetic value) in a particular environment. Alternatively we can think about use-values where activity takes place in the marine environment (e.g. recreation) and non-use values where values are experienced passively (e.g. existence value). Use values may in turn be direct (e.g.

recreational diving) or indirect (e.g. the economic benefits of using the sea to assimilation pollution).Figure 1.3.1 shows the relationship between these sets of values.

The evolution of this typology of values has itself been an interesting process.The notion of non-market use value in the environment has a long history that can be traced back to the writing of John Muir who argued in favour of creating forest parks in the US to protect non-market public good values (Muir 1909).At the same time Giford Pinchot argued that publicly owned forests in the US provided a direct market resource in the form of timber (in Turner 1997).Without using the phrase, writers like Leopold (1949) and Carson (1968) described the importance of maintaining a flow of *ecosystem services* (water, soil, assimilation of waste etc) to the formal market economy.The first attempts at valuing non-market aspects of the environment were again linked to public rights to hunt in national parks in the US (Ciracy Wartup 1947, Davies 1964).The possibility of Existence value – i.e. that value could exist in absence of the physical presence of the observer – was first argued by Krutilla (1967).Bishop (1982) extended the concept to include values placed on retaining an option to experience the environment or consume its resources at a later date.

		Examples	
Use value (UV)	**Direct Use Value (DUV)**	*Aesthetic Value*: tourism and recreation benefits e.g. visits to the beach, diving, swimming etc	Market and non- market
		Marine resources with commercial value e.g. fisheries, sea weed collection, sand and gravel extraction, energy extraction	Market
		Communication e.g. shipping, pipelines cables	Market
	Indirect Use Value (IUV)	Marine ecosystems and ecological function e.g. climate regulation, assimilation of pollution, nutrient cycling	Market
Non-use value (NUV)	**Bequest Value (BV) (Option Value)**	Legacy and deferred benefits: Heritage of marine living resources for future generations.Retaining option to consume/experience marine resources at a later date.	Non-market
	Existence Value (EV)	Existence Benefits. e.g. knowledge that marine living resources exist.	Non-market

Use value (UV) + **Non-use value (NUV)** = **Total Economic Value (TEV)**

Adapted from, van der Bergh et al (2002)

Figure 1.3.1. A typology of potential economic values provided by the management of marine areas

By the 1980's the basic typology of values described in figure 1.3.1 had been established. This typology is particularly useful when considering the application of valuation techniques.We will discuss individual valuation techniques in the next section; however it is important to note that different valuation techniques have varying abilities to capture the different values expressed in Figure 1.3.1. It is equally important to note that the distribution of the values identified in Figure 1.3.1 will vary significantly from one environment to another.We would expect the Great Barrier Reef to exhibit high levels of existence value, option value and use value associated with recreational visitors it may have relatively lower values associated with fishery resources.The central southern North Sea is an important fishery (i.e. direct use market value) with but one could reasonably expect to see low levels of aesthetic and existence value associated with this area.

1.3.2. VALUATION TECHNIQUES

Figure 1.3.2 sets out the main valuation techniques that have been used to place monetary values on ecosystems.For the time being we are concerned with monetary valuation techniques – non-monetary approaches to assessing values will be described later in section 3. Table 1.3.1 describes some of the generic issues surrounding these valuation techniques and their use.The following sections will look at each technique in turn in the contexts of marine ecosystems and their valuation.It is not the intention here to give a detailed description of each methodology many other references and manuals exist describe current best practice.

Table 1.3.1 divides valuation techniques into two broad categories
(1) Revealed preference
(2) Stated Preference

Revealed preference techniques are valuation tools which examine existing behaviour.For example the Travel Cost Method (TCM) is based on the principle that recreational users of the environment will travel further (incurring increased expenditure) to visit sites that they value most highly.The cost of travel is therefore a market expression of the utility experienced by that individual visitor.Thus the value preferences of the individual are *revealed* by examining behaviour – in this case expenditure on travel.

Stated preference techniques directly ask the opinion of stakeholders.For example, Contingent Valuation simply asks respondents what they are willing to pay (WTP) for a given change in environmental quality.

At this point it is important to reiterate that many of the valuation techniques will only capture some of the values described in figure 1.3.1. In the following sections we will spend more time with the most commonly applied techniques.

Table 1.3.1. Main Valuation techniques (Adapted from World Bank (2004))

Methodology	Approach	Applications	Values	Data requirements	Limitations
Revealed Preference					
Production Function	Trace impact of change in ecosystem services on produced goods	Any Impact that affects produced goods	Direct use (market)	Change in service; impact on production; net value of produced goods	Data on change in service and consequent impact on production often lacking
Cost of illness Human capital	Trace impact of change in ecosystem services on morbidity and mortality	Any impact that affects health (e.g. water pollution)	Indirect use (non-market) Unclear	Change in service: impact on health (dose response functions); cost of illness or value of life	Dose-response functions linking environment to health often lacking; underestimates as omits preferences for health; value of life not easily estimated
Replacement Cost (and variants e.g. relocation cost)	Use cost of replacing the lost good or service	Any loss of good or service	Unclear	Extent of loss of goods or services, cost of replacing them	Tends to over-estimate actual value. Replacement costs and value are not directly interchangeable. Must be used with caution.
Travel Cost Method (TCM)	Derive demand curve from data on actual travel costs	Recreation	Direct use (market/non market)	Survey to collect monetary and time, costs and distance of travel to destination	Really limited to recreational benefits; hard to use for multi purpose travel
Hedonic pricing	Extract effect of environmental factors on price of goods that include those factors	Air quality, scenic beauty, cultural benefits	Direct use (market)	Prices and Characteristics of goods. Generally housing or labour markets	Requires a large volume of data, statistically complex, needs an existing (surrogate) market. Only measures values of a small segment of the population (e.g. householders employees). Rarely used in practice.
Stated Preference Methods					
Contingent Valuation (CV)	Asks respondents directly their WTP for an environmental good or service	Wide application. In principle any good or service	Potentially all	Survey that elicits a WTP for specified good.	Many potential sources of bias. Guidance exists on how to minimise bias. Some serious unresolved issues e.g part whole bias. Difficult to apply in situations with poor respondent knowledge
Choice modelling	Asks respondents to choose their preferred option from a set of alternatives with particular attributes	Wide application. In principle any good or service	Potentially all	Survey of respondents	Similar to CV analysis of data generated is complex
Other methods					
Benefits transfer	Uses results obtained in one context in a different situation	Any where comparisons are available	Potentially all	Valuation studies at another, similar site	Data may not be directly comparable. Also validity of source data must be checked. Use with caution. However increasingly used for ecosystem valuations.

1.3.2.1 Production Function

The production function refers to the impact that a change in the ecosystem has on the net value of produced goods.The final valuation in the process is the market value of the change in production.For example is the clearance of tropical forest high in a watershed may impact on rice crops further downstream, see figure 1.3.2.

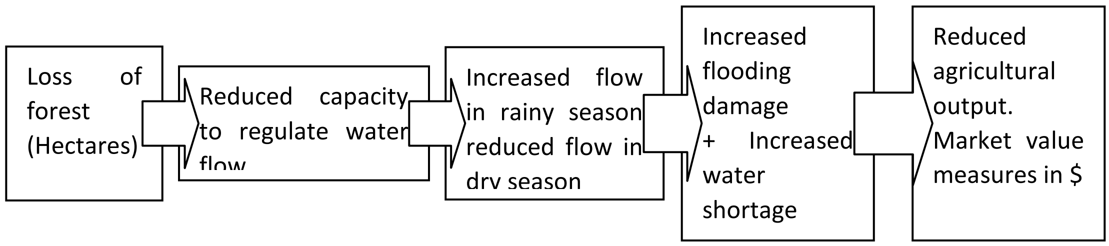

Figure 1.3.2. Production function **example**

The example illustrated in figure 1.3.2 is easy to understand. However undertaking such a valuation requires a detailed (and quantifiable) understanding of the links in the chain (i.e. the dose response relationship).The final values are revealed market values.In the above case this might be tonnes of rice.However the primary difficulty is quantifying the dose response relationship i.e. how many tonnes of rice production are lost as a result of the destruction of a hectare of forest.

In the context of MPA designation the potential effects of fisheries is one of the key issues.This effect may be positive or negative.An MPA may act as a refuge for fish or shellfish helping seed surrounding waters and potentially increasing/maintaining fish catches.Alternatively if fishers lose access to an area this may result in a loss of income.Clearly the dose response relationship is critical and it is likely to vary from region to region and with the type of fishery.There are a number of factors that must be quantified before the economic impact on fishers can be established:

- Biomass increase
- Biomass export
- Egg and larvae export
- Size

Biomass increase

Overall, the main biological effect that follows the establishment of a Marine Protected Area (MPA) is the increase ofbiomass and numbers of individuals. This occurswithin the no-take area or where the highest level of protection is enforced (e.g. gear restriction areas, seasonal closures and the like). Biomass increase is a consequence of reduced fishing mortality (Jennings, 2001) and has been documented is several MPAs (Badalamenti et al., 2008; Murawski et al.,2000, 2004, 2005; Pipitone et al. 2000) although the size of the no take area may play a significant role in shaping the size of this effect (Claudet et al. 2009).

Biomass export

In the buffer zone, nearby fisheries should benefit of biomass increase through biomass and egg and larval export (i.e. spillover effect), the first being a result of density dependent mechanisms (Sanchez-Lizaso et al., 2000) and the second of passive diffusion (Planes et al., 2000). However, due to the lack of data from before the MPA establishment and of control sites well documented spillover effects are rare. Direct evidence of net emigration of exploitable individuals from MPAs to fished areas has been demonstrated in only a handful of cases for lobsters and reef fish, or surmised from spatio-temporal patterns of fishery recaptures (Goñi et al., 2006, 2008, 2010).

Better evidence of spillover come from CPUE patterns, that isthe increase in CPUE over time in the fisheries near MPAs, (Alcala et al., 2005; Murawski et al., 2005; Stobart et al., 2009; Yamasaki and Kuwahara 1989). Here, a declining gradients of CPUE with distance from the MPA boundaries has been reported. However, whether spillover causing these CPUE patterns translates into greater catches in fisheries adjoining MPAs is less well documented (Goñi et al, in press).

Egg and larvae export

Because MPAs can increase the reproductive output of target populations, increased recruitment from egg and larval export is anticipated to produce even larger benefits for fisheries than spillover of adults (Jennings, 2001; Russ, 2002; Moffitt et al*.* 2009). However, such a recruitment effect is hard to detect due to the high spatial and temporal variability of larval survival and settlement, as well as the large area over which it can occur (Botsford et al. 2009).Although empirical support for yield enhancement at the stock or fisheries management scale is nonexistent, evidence of enhancement at the local scale is growing (Goñi et al. in press). At present, and despite their intuitiveness, empirical evidence of egg and larval export effects on fisheries remains non existent and most research is currently focused on unraveling larval dispersal and population connectivity patterns (Goñi et al., in press).

Size

This has been evidenced from gradients of decreasing target species size with distance from MPAs in the Philippines (Abesamis and Russ, 2005), Kenia (McClanahan and Mangi, 2000), and Spain, where mean size of fish near the boundary were intermediate between larger sizes within the MPA and smaller ones in farther fishing grounds (Stobart et al. 2009). Interestingly, biomass size spectra showed slopes significantly steeper in the trawler ban area of the Gulf of Castellammare than in control unprotected gulfs (Sweeting et al., 2009). This unpredictable result was attributed to the exclusion of trawlers, which lacked catch-size selectivity and to the continued, more size selective fishing by artisanal gears within the MPA.

Theoretical considerations relevant to socio-economic impacts

A number of fishery models have been used to investigate the effects of closed areas (or no-take zones) on yields from the fishery.In general these have demonstrated that the relative rates of reproduction and rate of dispersal of the stock, and the size and geometry of the closed areas, are key elements to understanding the impacts of closed areas on yields. Of great importance to understanding the economic efficiency of closing an area to fishing is the extent to which the stock is over-fished, or alternatively is being fished at optimal levels of fishing effort.The examination of these features requires a calibrated model so that before and after scenarios for the introduction of closed areas can be properly compared.As one example Figure 1.3.3 shows the effect on differing rates of dispersion of a fish stock for a given size and shape of closed area.

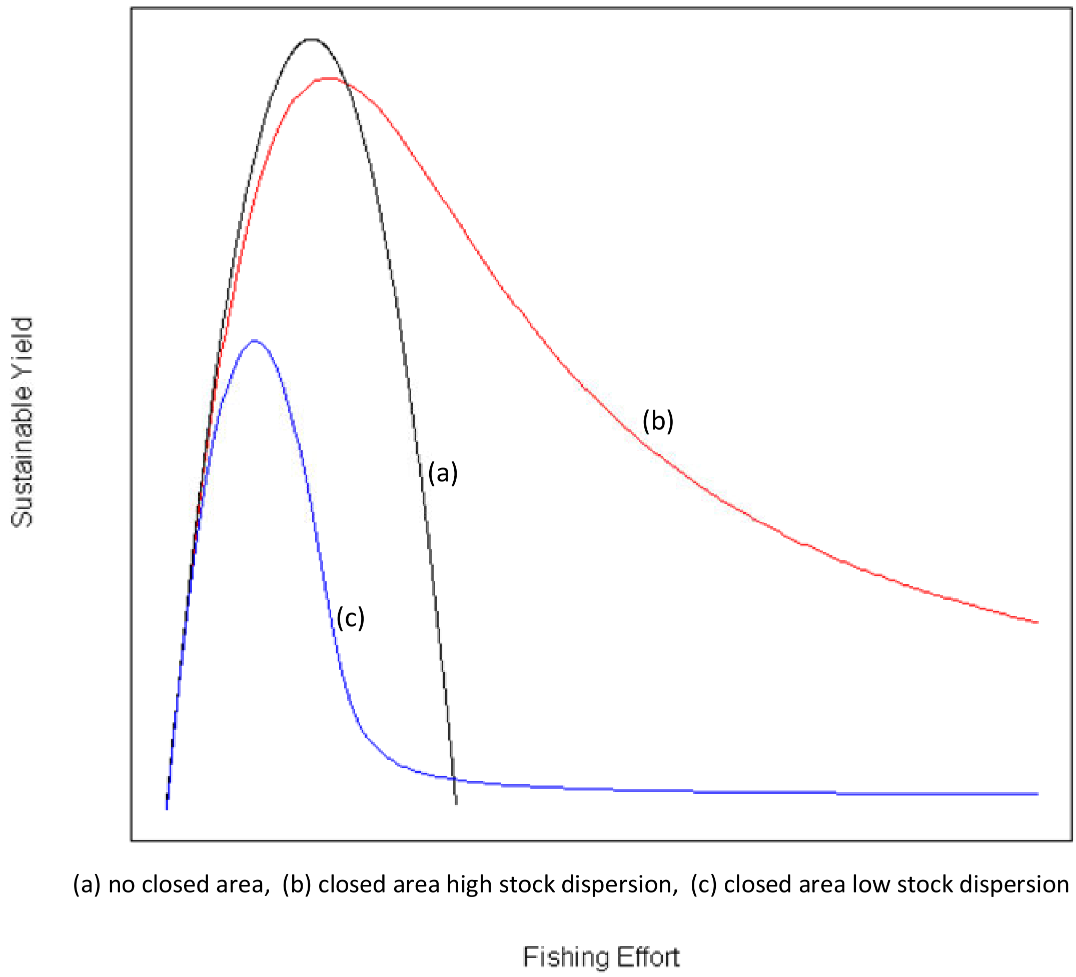

Figure 1.3.3. Two exaggerated scenarios of closed area effects on the yield effort curve of a fishery (after Side and Jowitt, 2005)

In case (b), with high dispersion rates, it is clear that in one case the losses to the fishing fleet from the introduction of a closed area, even at optimal fishing levels are small but the benefits in affording protection to the stock at higher levels of fishing effort are great.But a converse case (c) where the rates of dispersion are less (in relation to the reproductive rate of the stock) results shows that there are significant costs to the fishers as the stock benefits are restricted to the closed area.

1.3.2.2 Cost of Illness / Human Capital

This is effectively another dose response method but this time explicitly linked to human health or mortality.This method is widely used in the public sector in the field of road safety improvements allowing the benefits (saved lives) of public works to be compared to the costs.The measured benefits are (in the case of mortality) the statistical value of lives or (in the case of illness) treatment costs and/or loss of incomes.

In the context of the marine environment there are several examples of health effects that could potentially be valued. The method relies on establishing a clear and quantifiable link between a change in environmental quality and the health effect

Shellfish poisoning can cause severe illness and occasionally death.Paralytic, Amnesic and Diarrhetic shellfish poisoning (PSP, ASP, DSP) is associated with periodic blooms of harmful dynoflagelate plankton.In turn harmful algae blooms have been linked to nutrient enrichment of the sea.It would in theory be possible to estimate the non-market cost of a shellfish poisoning incident. However, accurately establishing the benefits of intervention to improve water quality is more problematic.The main difficulty would be determining a dose response relationship between any change in enriching input to the sea and level of illness.

Similarly sewage pollution can lead to gastrointestinal disorders associated with bathing or shellfish consumption.In extreme cases sewage pollution may be associated with cholera. Some 10,000 people are believed to have perished in Latin America between 1991 and 1995 from Cholera initially linked to the consumption of sewage contaminated seafood.Infectious hepatitis is another major concern.This disease can result in liver failure and in the worst cases can even be fatal.A World Health Organisation (WHO) study estimates that sewage contaminated sea food causes 2.5million cases of infection hepatitis annually, killing 25,000 people, with a further 25,000 being left with long term disability.It has been suggested that in the UK 25% of infectious hepatitis cases come from eating polluted shellfish (Shuval 2005).Again if the sewage/illness dose response relationship can be resolved then it would be possible to place a value on reducing sewage emissions.

1.3.2.3 Replacement Cost

Replacement cost and variants such as relocation cost (sometimes called shadow project) are based on the concept that the cost of replacement of a damaged environment is somehow a measure of the value of that environment.For example if a flood defence scheme means that an area of wetland is destroyed then the cost of recreating that wetland at another location is taken as a measure of the value of the habitat destroyed.There are several practical and theoretical problems associated with the method.

It is true that on a conventional market there is a close relationship between the cost of production and value.It is irrational for the equilibrium price of a product to be less than the cost of production.Theory also tells us that in competitive markets profits are minimised to the extent that price can only be marginally above cost.However, when dealing with environmental goods there is no market.Consequently market logic cannot be expected to govern the relationships between cost, price and value.There is absolutely no reason why the replacement cost of an environmental good should be correlated to value.It is also unclear how replacement cost related to the values sets described in Figure 1.3.1.

At a more practical level the method assumes that we can recreate or replace existing environments in another location.It is not at all clear that this is possible for any but the simplest of ecosystems.Perhaps an artificial reef could recreate some lost inshore reef habitat for the benefit of fish or shellfish.However it is fanciful to think that a more extensive and complex habitat (e.g. cold water coral) could be recreated.

In practice the method may have very limited applications in the case of artificial reefs or flood defence where environmental remediation is actually required to limit the net impacts of a development.In many marine situations it will simply not be possible to produce a value.It is also doubtful whether the method actually values the environment or the services it provides.

1.3.2.4 Travel Cost Method (TCM)

The TCM is based on the assumption that people will be willing to incur travel costs in order to experience environmental quality. First proposed by Hotelling (1947) the method was initially applied to valuation of amenity services from US lakes and rivers. The method is almost exclusively associated with the valuation of recreational sites.The amount that consumers are WTP will be reflected in the distance that they are willing to travel. In theory the number of people willing to visit a site should be inversely proportional to the travel distance. What the TCM is in effect trying to do is determine the demand curve for a particular facility. The demand curve will be downward sloping, demand decreasing as distance (cost) increases, (Dixon et al 1998).

To undertake a TCM analysis it is necessary to conduct a survey and discover the origin of visitors to the site. This can take the form of a survey of visitors at the site or of residents at potential origins. The latter is preferable as it is less likely to be affected by temporal factors, it is however likely to prove more costly. If the survey is conducted at the site, samples must be taken over an extended period of time.

It is likely that there will be distinguishable differences between the preferences of different socio-economic groups within each origin zone. It is necessary to take this into account when aggregating WTP figures for each zone, thus data concerning the socio-economic profile of each zone will be required.

Visitor rates are a function of several factors:

Vi = f (Ci, Ti, Ai, Si, Yi) where;

Vi = visitation rate with zero entrance fee;
Ci = round trip travel costs between zone i and site;
Ti = total time for the round trip;
Ai = taste;
Si = attractiveness of alternative sites
Yi = average income in zone i.

To take account of the multi-colinearity between travel cost and time spent in travel the model may be simplified to:
Vi = f(TCi, STCi, Yi)

Where:

TCi = monetary, time and travel cost of round trip from zone i.
STCi = monetary, time and travel cost of round trip to alternative site from zone i.

Simple regression analysis can be used to determine the relationship between Vi and TC. A consumer only travels to a site if he is receiving a surplus in terms of utility, i.e. there is a net benefit from the visit. Having discovered the relationship between TCi and Vi it is possible to inflate TC to a point where demand is choked off. The increase in cost and decrease in demand can be plotted, producing a downward sloping demand curve. If this is done for each zone, the results can be summed horizontally to produce a demand curve for the whole population from this the consumer surplus can be derived.

While the TCM is used to value recreational sites the method is not without its drawbacks. The cost of travel is assumed to be a combination of the "resource cost" of travelling (i.e. cost of fuel etc.), and the "cost of time" spent travelling. The valuation of journey time is very problematic, as the disutility of travel time will vary from individual to individual. Some individuals may even derive positive utility from time spent travelling. The resource cost of travelling is also difficult to estimate and should include factors such as tolls, and additional costs due to congestion and vehicle depreciation. It is also unclear how local visitors, who travel on foot, or cycle to the site, should be included.

The TCM also assumes that perfect knowledge regarding products exists, if costs of a visit were higher than expected, or enjoyment from the visit less than expected, then results will be flawed. There may be significant proportions of zone inhabitants who are unfamiliar with a site and would visit if they had knowledge of the facility. It is also difficult to take account of people who have a number of reasons for making the journey or of people who are on holiday and are living locally.
The TCM can only deal with very specific locations; it cannot value environmental impacts over a wide area. The TCM is also unable to value the day to day environmental quality of individuals, at work or at home. The TCM completely fails to take into account either existence or option values (or ecosystem services). The TCM is only really applicable to the valuation of recreational sites and even then it will not give a complete valuation, simply a value of the WTP for utility associated with physical participation. It must be concluded that, as a means of valuing the wider benefits of the environment pollution control measures, the TCM has little to offer.

In terms of the MESMA case studies TCM is only of relevance where there are clear recreational values.It will not be applicable across the suite of MESMA case studies.

1.3.2.5 Hedonic Pricing

The HVM of valuing environmental quality relies on the examination of surrogate markets. The method is underpinned by the theory that the value of any good is the sum of the value of bundles of attributes held by that good.

The most commonly studied market is the housing market. The value of a house is determined by a number of attributes held by that house, size, age, physical condition etc. One of these attributes will be the environmental quality of the surrounding area. If one can determine the variation in property value attributable to a fixed reduction/ increase in environmental quality, either at a single location or between areas, then we have a monetary measure of that environmental quality change. The change in value is a market expression of the value of that environmental factor which has changed, this difference is termed the "rent differential".

While it is obvious that environmental quality is an important determinant of rent differentials between locations, it is hard to assess exactly which attributes cause what change in value. There are many determinants of house value including size, age, condition, proximity to areas of economic activity, accessibility, planning restrictions, socio-economic profile of district, regional economic factors, national economic consideration etc. It is very difficult to assign values to each attribute.

The HVM also assumes perfect market conditions, where prices reflect all factors and information is perfect. In reality housing market are far from perfect, the very fact that it is possible to speculate making money from buying and selling land, is a priori evidence that the property market is imperfect. Property markets also tend to be sticky not reacting immediately to factors affecting price, high transaction costs increase the effect. An alternative hedonic approach is to examine wage differentials, between areas of differing environmental quality. Dixon et al (1988) noted that: "The method rests on the theory that in a perfectly competitive equilibrium, the demand for labour equals the value of the marginal product of the workers and the supply of the labour varies not only with wages but also with working and living conditions".

This suggests that, in order to attract labour, companies may have to compensate workers with additional wages in areas of poor environmental quality. This being the case the wage differential between areas of differing environmental quality, ceteris paribus, is a measure of WTP for that differential in environmental quality. Clark and Khan (1989), claim that hedonic approaches, the hedonic wage approach in particular, have the advantage of dealing with real data sets "based on actual utility maximising behaviour of individuals". All hedonic approaches however suffer from complexity, it is difficult to assess how much each of the determinant variables contribute to the commodity price. The methods are also subject to the imperfections of existing surrogate markets. The other significant failure of hedonic approaches to valuing environmental quality is their inherent failure to take any existence or option value into account. The HVM can only measure what is already internalised in existing markets which means option and existence values will not be included.

It is possible to imagine hedonic valuation being used to assess the uplift in property values associated with coastal locations.However no account is taken of other environmental values or ecosystem services.It is hard to imagine how hedonic valuation approaches could be applied in the context of the MESMA case studies.

1.3.2.6 Contingent Valuation (CV)

Of all the environmental valuation methods described in this section Contingent Valuation (CV) is arguably the most important.It is certainly the most widely applied environmental valuation technique it also highly controversial with several unresolved methodological and theoretical problems.The basic principle behind CV is simple.A questionnaire survey asks a sample of the population what their Willingness to Pay (WTP) is for the provision of an environmental good or service.The survey sample results are averaged and then aggregated across the population to give a total WTP value for the environmental good in question.A fuller description of the methodology is presented in Appendix 1.

Contingent valuation is a controversial technique but it is also the method which, in principle, can be applied to the widest variety of situations valuing aesthetic, option and existence values. The highest profile use of CV followed the Exxon Valdez oil spill. On March the 24th 1989 the oil tanker Exxon Valdez ran aground in Prince William Sound of the coast of Alaska spilling over 11,000,000 gallons of crude oil. The spill resulted in considerable damage to the coastal environment.The State of Alaska commissioned a CVM study with the aim of using the results as the basis for a claim in respect of damage to non-use environmental values. The study resulted in an estimated damage claim of $2.8 billion. Damages were eventually settled out of court at $1.5 billion (Hanneman 1994).This case led directly to the standardisation of rules for implementing CV in the case of oil spills and wider institutional acceptance of the methodology in the US (NOOA 1993).This process of standardisation and development of best practice guidance has continued (e.g. Bateman et al 2002).

Of all the valuation techniques CV is, in principle, able to measure all the values identified in figure 1.3.1. For this reason more than any other it is the most widely used technique.It is always possible to conduct a CV study and get a *prima facia* valid result.We will discuss problems with the technique below but the ability to always produce a result is a strong motivator behind its use.

The technique has a number of limitations and opinion reminds sharply divided about the validity of CV results.The main difficulties in relation to the marine environment are detailed below:

- Lack of knowledge in the respondent population.It is possible to ask respondents to comment on aspects of the environment for which they have little or no knowledge and still obtain a result.However one must raise questions over the validity of this or at the very least the stability of this value which is likely to change (Reiling et al 1990).Providing in order to overcome a lack of knowledge can bias results.Arguably CV is more reliable the more familiar the respondent is with the environmental good in question.Respondents may have a clear understanding of recreational values of bathing water or beaches.However deep sea benthic ecosystems may be harder for respondents to conceptualise and value.
- Part whole bias.Using CV the sum of the values of individually valued goods tends to be higher than the value of these goods if valued together (Boyle et al 1994).If we ask WTP for preservation of one species of fish and then 100 species of fish the second answer may only be marginally higher than the first.This appears to go against market logic. In conventional markets the value of a basket of goods is the sum of the value of the individual purchases.There are several possible explanations of part whole bias. One explanation is linked to the fact that CV creates a hypothetical market which internalizes a single environmental good.The respondent considers what utility he receives from this good and then relocates their budget to accommodate this new expenditure.However all other environmental goods remain free.The consequence is that whatever good the CV is attempting to internalise the respondent is making a broadly similar calculation (i.e. “what can I afford to spend on this new good”) and this results in similar WTP values.Put simply the respondent allocates what they can afford in the knowledge that the rest of the environment is still free.
- Boundaries and equity.CV estimates an average WTP for a sample and then aggregated the result across the whole population.Deciding what the whole population should be is not always easy.The size of the population will have a major influence on the result.Inter- and intra- regional income distribution will affect results.*Ceteris paribus* we would expect wealthy people to have higher WTPs.Consequently their preferences receive higher weight in any analysis.This may be an issue where there are large disparities in incomes across the respondent groups (e.g. regional scale studies, tourists and locals). This issue is at odds with the concept of environmental justice.However it is worth noting that markets are not ‘fair’, and it is the ability to pay that determines the distribution of resources in real markets.
- There is a final and more philosophical concern.A systems approach to conceptualising the economy and the global environment suggests that the economy is a subset of the global system.The economy uses ecosystem services.The size of the economy (GDP) is largely determined by the rate at which resources are consumed and goods and services produced (see figure 1.3.3).

WTP values (and revealed preference values) are expressed in the money economy (A).The size of the money economy and consequently WTP values is determined by the vigour of the production process (i.e. the state of the economy).Non-market values are experienced outside the money economy in the wider ecosystem.By using a money measure like (WTP) which is bounded by incomes we are limiting the values we place on aspects of the *Global ecosystem* (A) by the size of the subset B (i.e. the economy). There is no *a priori* reason to believe that the utility derived from non-market benefits could not be greater than the value of the economy.However WTPs are constrained by the size of the economy.

The extent to which these methodological and philosophical arguments rule environmental valuation in or out is likely to vary from case to case.If boundaries are clear, the environmental good is well understood, and the benefit has to be paid for (e.g. reduction in sewage related pollution) then it is easier to justify the use of CV.If the environment is not well understood, boundaries are unclear, and we are valuing damage to the environment (e.g species extinction) it is much harder to ignore the problems outlined above.

1.3.2.7 Choice modelling

The term choice modelling (CM) encompasses a range of methods, including:

- Choice experiments – usually choosing between two alternatives
- Contingent ranking – rank a series of alternatives
- Contingent rating – score alternatives on a scale
- Paired comparison – score pairs of alternatives on similar scale.

CM relies on the identification of attributes or characteristics that people value in the environment. Choices are then constructed with different levels of these attributes associated with different monetary values.Respondents are then presented with these choices.One choice may be selected or alternatives ranked.The selected choice, or ranking, indicates preferences for specific attributes.If there are many variable attributes then it is not possible to ask respondents to rank or rate them all.Generally the more familiar respondents are with the subject in question the more alternatives it is possible to offer.Imagine we have 3 attributes each with 3 potential states eg:

Fish stocks	1. Degraded	2. No change	3. Enhanced
Sea birds	1. Degraded	2. No change	3. Enhanced
Seals	1. Degraded	2. No change	3. Enhanced
Cost	1. No cost	2. Current cost	3. Increased cost

In the above case there are 3x3x3x3 = 81 different combinations available to offer respondents.It is not possible to consider al the possible combinations in a questionnaire.Statistical design theory (*fractional factorial design*) is used to produce a smaller set of alternatives. Individual questionnaires may then present smaller subsets of choices to respondents. WTPs from CM experiments is statistically demanding though procedures are well documented (e.g. Bateman et al 2002). The presentation of alternative scenarios in choice modelling means that it is potentially subject to the irrational behaviour discussed in consumer choice literature (Huber et al 1982, Loomes et al 1989).

CM methods have the advantage the respondents are not being asked values but rather presented with options.This can make it easier to elicit a response.Further more the analysis of the results can allocate values to different attributes without valuing the independently.CM methods are subject to most of the same problems as CV in terms of good familiarity, part whole bias etc.

1.3.2.8 Benefits transfer

Benefits transfer (BT) is not itself a direct valuation technique.BT refers to the use of valuation evidence from completed studies at other locations to derive a value for a new location. Environmental valuation studies particularly those involving lengthy questionnaires can be both expensive and time consuming.Using data from existing studies of similar habitats is cost effective when compared to the cost of commissioning a new study.In order to facilitate this process several databases collating environmental valuation data. The best known of these is the Environmental Valuation Reference Inventory (EVRI) (www.evri.ca).The EVRI is a searchable database of environmental valuation studies.The valuation studies in question use the methods outlined above and are therefore subject to the limitations of these individual methodologies.The simplest form of benefits transfer involves identifying a unit value for the good in question (e.g. $/hectare mangrove) and using this at a new locations.While the attractions of BT are easy to see there are many drawbacks to the approach including:

- The environments in question may not be identical
- Differences in socio-economic characteristics of the populations between the sites may be significant.
- Different cultural attitudes to the environment.
- Difference in prevailing market conditions and the economy
- Different in the purpose of the valuation and the proposed chance in utility

- Quality of the original survey

The direct transfer of values from one location to another may be viable in closely similar situations (Piper and Martin 2001).However some research indicates that very large errors can occur if this condition is not met (Brouwer et al 1999).Other sciences use meta-analysis to collate data from different studies. This is a statistical process which identifies a common measure of effect size and should result in more powerful results than individual results from a single study under a single set of assumptions and conditions.Meta-analysis is common in the medical sciences where experiments or tests may be repeated many times by different research groups.However, meta-analytical techniques rely on high standardisation of experimental techniques which does not exist in valuation studies.

1.3.2.9 Ecosystem Valuation

It is possible to regard the environment as a provider of services upon which the economy and human wellbeing relies. This worldview builds upon the early systems thinking of Boulding (1968), Shumacher (1973) and others.

In 1997 a group of economists lead by Bob Costanza published a controversial paper which claimed to have placed a value on the world's total ecosystem services (Costanza 1997).There are 4 stages to this approach(i) identify ecosystem services (17 in total), (ii) identify global biomes the services they provide and their extent (16 biomes were identified), (iii) valuation of the services provided by each biome, (iv) aggregation.The Costanza team estimated that the value of global ecosystem services was US$33trillion yr^{-1}.The team reviewed over 100 valuation studies to produce a value per hectare for each biome.This is therefore essentially a benefits transfer study on large scale.All the issues and problems of BT remain intact.This work was both high profile and controversial in equal measure.While Costanza's results have been widely quoted by scientists and environmentalists they are also regarded by many economists as being fatally flawed (IUCN 2004). Despite methodological concerns various other studies have followed the Costanza approach. Pimentel et al (1997) and Patterson (1999) valued biodiversity in the US and New Zealand using this approach while Williams et al (2003) estimated the value for the Scottish environment.

The basic concept of maintaining and managing ecosystem services (if not their valuation) is now firmly established as a key driver of international and UN environmental policy (IUCN 2004). The Millennium Ecosystem Assessment (MEA) in particular has been critical in advancing institutional acceptance of ecosystem approaches (MEA 2003).The National Environmental Assessment (NEA) project in the UK is one example of the application of the MEA approach on a national level. (http://lwec.org.uk/activities/nea).In simple terms the ecosystem valuation approach now emerging involves four stages:

1. Description of core ecosystem processes (e.g. nutrient cycling)
2. Identification of beneficial ecosystem processes (e.g. biomass production)
3. Identification of specific benefits (fisheries)
4. Valuation of benefits (value of catch)

While the identification of beneficial services is possible the main difficulty in following the process to its conclusion arises in the monetary assessment of benefits as the valuation difficulties described above (and in 2.10 below) still apply. In addition to concerns over the veracity of the valuation process there is a wider debate about reliance upon ecosystem valuation for policy making. Redford and Adams (2009) suggest ecosystem service approaches are a useful means of explaining the dependence of the human economy upon nature.However that there is a risk that over emphasis on the valuation of ecosystem services will lead to the undervaluation, or disregard, of aspects of biodiversity which provide no direct economic services (Redford and Adams 2009). Mark Sagoff, a long standing critic of environmental valuation, suggests that ecosystem valuation confuses 'prices' with 'values', and that market price (or WTP)"*does not correlate with value, benefit or utility*" (Sagoff 2008).Even Adam Smith noted that "*the things which have the greatest value in use have frequently little of no value in exchange"* (Smith 1776).Sagoff argues that the real 'value' of ecosystem services is their use value which in many is not fully reflected in market prices.

1.3.2.10 Environmental Valuation of Marine resources

The discussion above has described the principal valuation techniques together with their strengths and weaknesses. There is a modest but growing body of work which has applied these techniques in the marine environment.Ledoux and Turner (2002) undertook a review of valuation studies on ocean and coastal resources, the results of this survey are summarised in the table below.The majority of the studies identified by Ledoux and Turner are contingent valuation studies of beaches with the majority of work being done in the US, see table 1.3.2.

Table 1.3.2: Summary of valuation studies (adapted from Ledoux and Turner 2002)

	Total[1]	S	V	C	V	MV	Other
Beaches	30	2	1	8	2	0	0
Storm Protection	1	1	0	3	0	0	1
Water Quality	8	4	6	0	0	0	0
Fish	6	3	0	0	0	6	2
Habitats	6	4	6	0	0	0	0
Multiple function	6	6	1	0	0	5	0
Total Economic Value	2	1	1	0	0	1	1

1. Rows do not all add up as some studies used two valuation techniques.
US – United States, CV- contingent valuation, TC - travel cost,
HV- Hedonic valuation, MV - Market Value

In a more recent review Pendelton et al (2007), focusing on the US, identified a total of 91 studies covering marine assets, see figure 1.3.4.

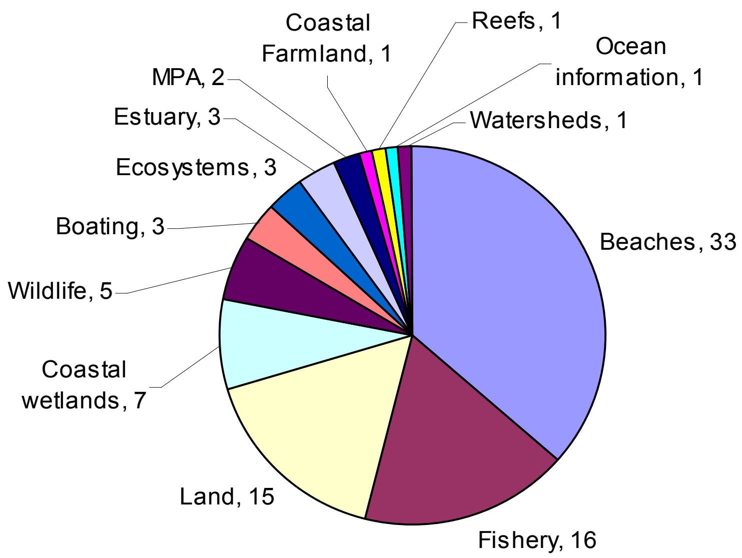

Figure 1.3.4: US Marine Valuation Studies 1977-2005 (after Pendelton 2007)

Again this survey reveals a strong emphasis on the assessment of recreational values of beaches with travel cost hedonic, valuation and contingent valuation accounting for most of the studies identified. A recent European Union review involved an international call for environmental valuation studies, this generated 291 responses only 4 of which referred to marine and coastal valuation studies (Markandya et al 2008).

1.3.2.11 Marine Ecosystem Services

Following in the footsteps of Costanza ecosystem valuation approaches have been applied in the marine environment.In the UK Firn and McGlashan (2001) estimated the value of the ecosystem services from the coastal zone.Beaumont et al (2008) attempted to value the goods and services provided by marine biodiversity in the UK this study attempted to capture market and non market values and ascribe these to goods and services delivered by biodiversity.However the study suffered from a lack of available or reliable data.The tabulation of services and corresponding values may help illustrate the range of services provided by the marine environment and their relative importance thus helping “raise awareness of the importance of marine biodiversity” (Beaumont et al 2008).However the authors question the practice of aggregating these results into one value (as done by Constanza and others).In a statement, echoing many other studies, the authors note that the monetary values presented in the study cannot be aggregate “*as different methods have been used to calculate values and hence they are not directly comparable*” (Beaumont et al 2008). There is an irony here as arguably the only purpose of reducing values to a common monetary metric is exactly to facilitate direct comparison.Equally if environmental values cannot be compared with each other how is it possible to justify comparing any individual value to other money measures (e.g. the cost of undertaking management measures)?

Hussain et al (2010) attempt to assess the value of ecosystem services from a proposed network of UK marine MPAs. Interestingly this work uses several of the values derived by Beaumont et al (2008) despite concerns raised in this earlier work.Furthermore it is interesting to note that the criteria for selecting the potential network of MPAs is the protection of OSPAR (Oslo and Paris Convention) habitats and species and not ecosystem services.It seems inconsistent to evaluate management regimes for this MPA network on the basis on ecosystem services which were not the rationale for the initial designation.In a recent development in the UK the Crown Estate (who own and manage the sea bed in UK waters) have commissioned ecosystem valuation work (Saunders et al 2010) which it hopes to incorporate into a GIS based decision support tool known as MaRS.

1.3.2.12 Some observations on monetary valuation.

There is a clear rational which underpins the desire to place monetary values on environmental change.Monetary values should allow us to compare environmental costs and benefits (improved or degraded environments) with market costs and benefits (management expenditure, incomes lost profits etc).Great advances have been made in the development of valuation techniques.However all of the environmental valuation techniques (particularly non-market) have methodological limitations and some serious theoretical issues remain unresolved.Valuation techniques appear to be most robust where:

- The environment is well understood as are the services it provides
- Boundaries are clear (effects and beneficiary groups) and
- Relationships are clear (dose response etc)
- Where change is incremental
- Where environmental improvements must be paid for
- There is an existing market

It is also worth noting that many of these techniques involve careful design and extensive questionnaire survey followed by detailed statistical analysis.Consequently valuations may be both time consuming and expensive.

It is worth considering for a moment the policy environment within which valuation studies are supposed to support decision making.Environmental economists are quick to justify valuation studies by suggesting that the results are an aid to policy makers.Even where results are clearly unreliable it is often suggested that they are a ‘useful guide’ or some how “indicative” of relative values.However we are not aware of any research examining how environmental valuation studies actually influence policy decisions.This raises important issues about the nature of policy decisions and the ability of the policy making landscape to take account of monetary values.

Monetary valuation processes are about reducing attributes of a goods or services to a single metric.This allows us to compare costs and benefits and make decisions about the best future course of actions.Such an approach is possible where an individual has absolute control over resources e.g. personal investment decisions; making choices on markets; or where an authority is able to dictate outcomes.However, in a situation where individual winners and losers hold rights in the environment (fishing, navigation, access etc) or where decisions have to be reached on the basis of consensus, reducing values to a single monetary metric (even if it is possible) may be of limited actual use to decision makers.Valuation processes which reveal the pattern of values across stakeholder groups identify potential sources of conflict and potential areas for consensus may be more useful.O’Niel (1993) suggests that “cost benefit analysis provides policy without debate”.If policy makers are required to develop management strategies which require collaboration with stakeholders then a single monetary value might not help this process.

1.3.3 NON MONETARY ASSESSMENTS OF VALUE.

1.3.3.1 Background

The circumstances where monetary valuation techniques appear to be most robust were summarised in section 2.11. They can be seen to be a relatively rare set of conditions. A more general situation could be described by the opposites:

- The environment and the services it provides are poorly understood
- Boundaries are unclear
- Relationships are unclear
- Change is rapid
- Environmental improvements are not paid for
- There are no established markets

These conditions are especially applicable to the marine environment. As measures to enclose the marine commons gather pace there is a great need for understanding about who the stakeholders are and how, and to what extent, their diverse values can be a accommodated within a coherent policy of planning and regulation. The EU Marine Strategy Framework Directive requires the achievement of 'Good Environmental Standards' (GES) by 2020. It calls for an eco-system led approach at a time when new human activities, such as marine renewables, in the EEZs are set to expand rapidly and the prospect of conflict with traditional activities has seldom been greater. Both new and traditional activities are in potentially greater conflict with the marine eco-system. Among the many 'unknowns' are the values placed on the marine commons by coastal communities and the public at large who may not be regular users of the marine environment but currently have freedom of potential access which is to be curtailed. In the absence of knowledge about the environment, boundaries and relationships it is hard to see how monetary methods of valuation can be applied.

Alternatives to monetary valuation have been sought. Non-monetary methods do not lead to a single metric or permit individual control over resources. The results of non-monetary assessments guide policy makers with a framework for debate and negotiation between interested groups and individuals allowing weight to be given as necessary to potential areas of conflict and consensus. Methodologies of non monetary assessment include Environmental Impact Assessment (EIA), Opinion Polling (OP) and Multi-Criteria Analysis (MCA). Opinion polling has recently been employed in Oregon, USA to sound out public knowledge about marine renewable energy, their source of knowledge and feelings about it [Conway et al. 2009].

1.3.3.2 Multi-Criteria Analysis (MCA)

Various MCA techniques are widely used in government to inform policy development. However, in the UK they are largely seen only as a complement to CBA and part of a conventional top down normative approach [DCLG 2009]. MCA is described:

"Multi-cultural analysis establishes preferences between options by reference to an explicit set of objectives that the decision making body has identified, and for which it has established measurable criteria to assess the extent to which objectives have been achieved."

The UK government MCA manual lists the steps as:

- Establish the decision context. What are the aims of the MCA, and who are the decision makers and key players?
- Identify the options
- Identify the objectives and criteria that reflect the value associated with the consequences of each option
- Describe the expected performance of each option against the criteria. Score the options (i.e. assess the value associated with the consequence of each option)
- Assign weights for each of the criteria to reflect their relative importance to the decision
- Combine the weights and scores for each of the options to derive overall value
- Conduct a sensitivity analysis of the results to changes in scores or weights

A development of MCA is MCDA (multi-attribute decision analysis) which addresses complex problems characterised by a mix of monetary and non-monetary objectives.

1.3.3.3 AGORA - a participatory conflict management algorithm

Plans to extend planning and management to the marine commons prompted further development of MCA techniques which are truly 'bottom-up' and not normative. They were first developed around the complexities of the land/sea frontier and Coastal Zone Management (CZM) then extended to other marine issues including the capture of value expressions about the creation of Marine Protected Areas (MPAs). Davos et al. (2007) reported results of an analysis of the conflicts that the zoning of MPAs might generate, in this case in the Galapagos and San Andreas archipelagos. They preface discussion with acknowledgement that *"…..formulating a cooperation strategy requires an array of information in addition to such other attributes as intuition, experience, familiarity with established institutional structures, and political savvy."* They point out that a large number of stakeholders with diverse interests faced with a common problem must think strategically and act cooperatively. The elements of the information needed to assist stakeholders are:

- The issues - <u>criteria</u>
- The comparative significance - <u>priority</u>
- The similarities among the priorities of several stakeholders that might point to potential cooperation allies - <u>potential coalitions or cooperation strategies</u>
- The extent to which members of a cohort group of stakeholders agree on their priorities - <u>solidarity</u>

In the Galapagos/San Andreas project the method used to gather, analyse and manage the data from the study area is known as AGORA (Assessment of Group Options with Reasonable Accord) - a participatory conflict management algorithm. AGORA uses Multi-Criteria Evaluation Methods, Core Theory and Game Theory (Santorineau et al. 2008).Rather than waiting for 'top down' normative decisions, a group of stakeholders agree to participate in a 'bottom up' process. Issues around a collective problem are addressed through answers to a questionnaire designed to an agenda set by the stakeholders in participation.

"The participants are asked to first rank the criteria in order of significance and then to indicate for each pair of consecutively ranked criteria how much more significant is the top ranked criterion over that ranked below it. The potential coalitions are identified by a k-means Cluster analysis (Euclidean distance) and the solidarity of cohort groups of participants with ANOVA."(Davos et al. 2007)

The focus is to help stakeholders develop cooperation strategies by analysing their priorities - no effort is made to identify a single statistical representation of all priorities; the objective is to expose potential conflict and consensus. The VALCOAST project set out to ascertain stakeholder willingness to participate and cooperate in coastal management (Davos et al. 2002). It used AGORA and argued that a greater emphasis should be placed on policy process as opposed to policies focused on achieving outcomes based on value laden agendas imposed from the top down (EU 1997). Another marine research project to use AGORA was Project Fisher in the ESRC funded 'Science and Society' programme. It set out to identify the relationship of fishers with science (Kerr et al. 2006).

An extension to the AGORA methodology has come about by integrating elements of the participatory conflict management algorithm with GIS (Geographic Information Systems) and CA (Cellular Automata) to create Spatial-AGORA. This has been used in the urban coastal region of Perama in Athens, Greece to focus on the management of spatial conflicts related to interest, use and values created by land use change planning options. The objective was to facilitate cooperative decisions supportive of sustainable development. The participatory conflict management rules applied use elements from Core Theory (Tesler 1994) and the MAXMIN Theory of Justice (Rawls 1971).

1.3.4. Conclusions and Implications for MESMA

Socio-economic values, responses and assessment are critically important aspects in the designation and management of Spatially Managed Areas. The marine ecosystem is endlessly complex and beyond determinism. Almost as complex are the human relationships and interactions which any management regime must struggle to at least part understand if it is to be effective. The greatest threats to the success of Spatially Managed Areas are most likely to be found in the inter-human conflicts and disagreements about ecosystem values. In these chapters we have discussed the meaning of value in the marine environment and introduced the various valuation tools and techniques which have been developed - the two main headings are monetary and non monetary.

The idea of placing monetary values on the environment is not new or unusual. The valuation process is relatively straightforward where market goods, such as fisheries, are being assessed. However, most environmental values are non market and commentators recognise the difficulty of reducing such complexities to a monetary unit. Policy makers and academics have addressed the problems for well over fifty years and a wide range of valuation tools have been developed to address a variety of circumstances. However there remain sharply contrasting views and opinions. This debate has been brought into sharper focus still by the ever more prevalent worldview which considers the environment as a provider of ecosystem services which can be valued. It is argued that a total economic value can be calculated for all ecosystem services and used to determine the cost or gain of any incremental change in environmental quality. The fact that ecosystems provide services to society is self evident but their monetary valuation remains highly controversial. Ecosystem valuation relies on the same diverse and variously based monetary techniques discussed above.The weight of current research indicates there is insufficient information to support a complete ecosystem valuation of the marine environment. However there are currently several large studies attempting complete valuations of marine ecosystem services while yet more commentators reject the whole idea.

Non monetary valuation techniques attempt to understand the cause, distribution and strength of various socio-economic values held by stakeholders. The drive here is to win the widest possible consensus of stakeholders in the design of a management policy thereby minimising conflict and reducing the need for difficult and costly enforcement in the future. Many resource management failures in the marine environment are the result of value conflicts between stakeholders. The problems of the Common Fisheries Policy are often and unfairly blamed on the science when the intensity of the value conflicts actually makes any meaningful dialogue fraught with difficulty. Consequently, reductive or scientific explanations of monetary values and subsequent cost benefit analysis are unlikely to lead to effective resource management. Non monetary valuation presents difficulties of comparability and presentation of results and requires time, effort and persistence in communicating with stakeholders. However, techniques of non monetary valuation have advanced to a point where results are reliable and meaningful guides to policy development. Ultimately any resource manager, and in this case the MESMA framework, must ask the questions:

- What are the management objectives?
- What are the issues, which groups or individuals are they important to and why?
- How are the objectives obstructed by the issues and how may conflict be avoided or at least reduced?

Socio-economic analysis only has purpose if it helps to address one or more for these questions. ‘Top down’ policy solutions can be imposed where knowledge is great and enforcement is good. In these circumstances a reductive monetary valuation of ecosystem services and cost benefit analysis may give adequate support to the development of policy. However, if we have a more complex policy environment, such as the marine environment with vast unknowns about how the ecosystem actually works, a weak regulatory and enforcement regime, long held traditional rights of open access on which individuals and communities depend and resources of potentially great monetary value, then good governance based on consensus, stakeholder agreement and a ‘bottom up’ approach is essential. Less reductive techniques which avoid the temptation to try and distil diverse issues into a single financial unit of measure are indicated.The socio-economic values held by stakeholders, and the reasons for them, are the measures which should alleviate conflicts and drive policy.

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APPENDIX 1: FIVE STAGES OF A CONTINGENT VALUATION

Stage one: hypothetical market.

Contingent valuation attempts to create a hypothetical market, it tries to find out what the respondent would be willing to pay if they really did have to pay for the good.

• Establish the scenario reason for payment.

The scenario should be as realistic and plausible as possible. For example if trying to estimate the environmental damage costs of an oil spill one could ask the question:
"what would you be prepared to pay towards the provision of a standby tug which would prevent a repeat of the oil spill."

• Identify bid vehicle.

How payments would be made in any hypothetical scenario must be indicated. This can be a sensitive issue. For example if the scenario asks "what would you be prepared to pay in additional tax ....." this may provoke a response to the fact that there is an additional tax rather than a careful consideration of the environmental good in question.

• Identify the information to be given to the respondents: framing the good.

Additional information may be required about the particular good in question. For example if you are examining the environmental costs of a development project you will clearly need to describe the project and the impacts it will have on the environment. However in general it is better to avoid introducing too much additional information as this may inadvertently bias the response.

Stage two: obtaining bids.

• Design questionnaire.

A questionnaire must be designed which asks the payment question. However the questionnaire will need to ask for any other information requires to complete the assessment. You will need to gather information on any variables that might affect the results given (eg age, income, education etc)

• Willingness to pay / Willingness to accept.

A decision must be made about whether the questionnaire will ask whether the respondent is willing to Accept (WTA) for loss of a good or Willing to Pay (WTP) for an improvement in the environment. However in general questions will use the WTP format. This is because of WTP/WTA divergence see below.

• Dichotomous Choice or Open Ended questions.

There are several ways to frame the WTP question in the questionnaire. However the two principle methods are to as an open-ended (OE) question and record the result. Or alternatively to offer a randomly selected value (from within a reasonable range) and record whether the responded would or would not be willing to pay this figure This is called a dichotomous choice (DC) question.

• Identify sample.

The sample for the survey must be identified. The aim is to get a representative sample of the whole population. If a DC question is asked the sample will have to be large (several hundreds).

• Conducts survey (face to face, telephone, postal).

The survey can be conducted in various ways. Telephone and postal are cheaper however they get very low responses rates and face to face interview are very much the preferred option. It is essential that any survey is given a trial with a small number of respondents. This will help identify any flaws in the questionnaire. Also in the case of DC questions a pre survey will reveal the likely range of answers from which offers should be made.

Stage three: estimating average WTP

• Estimate mean and median WTP.

Once the survey is completed the average WTP must be estimated for the sample. This is considered in more detail below.

Stage four: identify the effect of variables.

• Regression analysis on variables.

Regression analysis should be undertaken to identify the key variables which are driving the answers given. It is essential that this is done to check the sample is taken is representative of the population as a whole. For example if the regression analysis indicates that age is a key determinant of WTP then the age structure of the sample must reflect the age structure of the population as a whole.

Stage five: aggregating data.

• Aggregate up data to whole population, (if required adjust for any inconsistencies in sample).

The sample average is multiplied by the relevant population to get a total value. Clearly the size of the population will have a significant influence on the result. It is not always easy to decide what the whole population is. As noted above it is also important that the sample is representative of the whole particularly in respect of those variables which influence the WTP.

One particular problem is dealing with apparently anomalous results in the data set. It is common to find that a small number of answers are far in excess of the others. This may be because the questionnaire has not been well understood or perhaps the respondent is making some sort of protest, is not engaging with the process or is acting strategically and trying to influence the result of the study. Such outliners are usually noted but excluded from the calculation.

Estimation of Mean and Median WTP from CV data

To help us understand how mean and median are estimated it is worth considering fig A1 and A2.

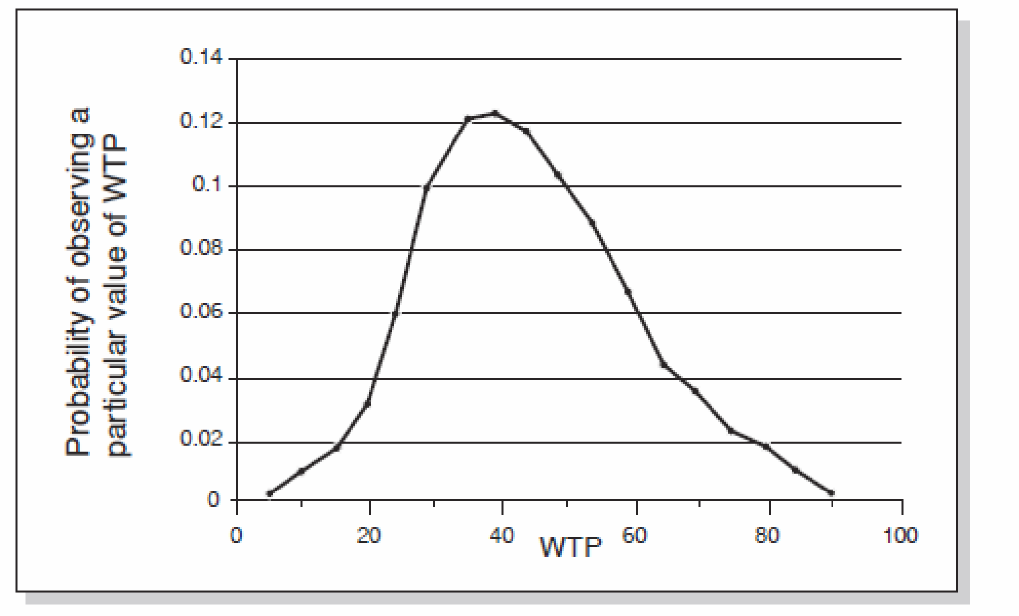

Figure A1: Probability density Function PDF

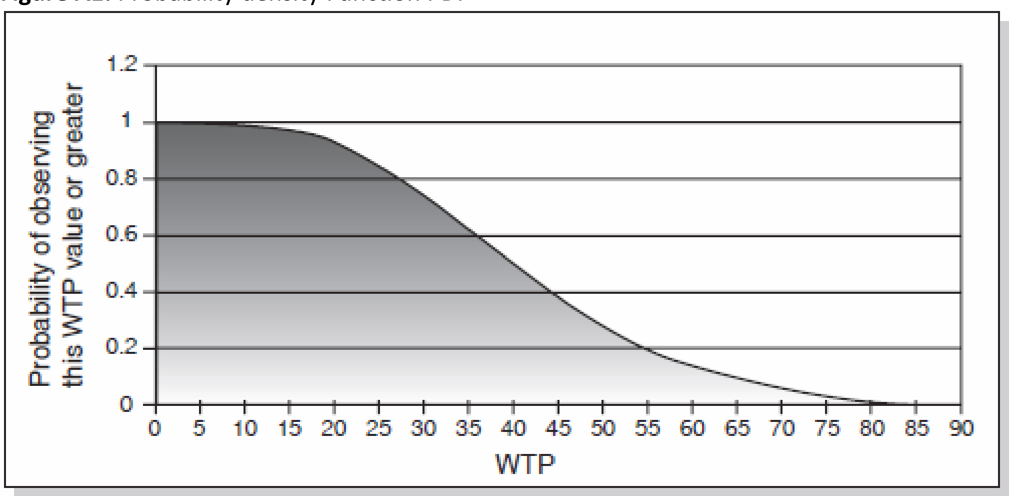

Figure A2: Probability of Survivor Function

The probability density function (PDF) in Figure A1 maps out the probability of getting any one WTP answer. This can be estimated from the range of answers given in a OE WTP survey. The following rule can estimate the mean under the PDF:

mean = (each observed WTP x probability of its occurrence)

The probability survivor function (PSF) in Figure A2 presents the data in a different way showing the probability of observing a WTP greater than a particular value. With the PSF the area under the curve represents the mean WTP. The median can also be calculated as the WTP where there is a 50% chance of getting a higher or lower value. This translates as a probability of 0.5 on the PSF y-axis, the value can then be read from the x-axis.

Open-ended CVM Data.

To make the calculation we can construct a survivor function. This has been done below in Figure A3 for a hypothetical data set. The function is represented by a series of steps. The height of the step is the probability of observing a WTP greater than the one identified on the x-axis. The median is the WTP value, which has a 0.5 probability and can be read from the graph. The mean is the area under the graph, which can be easily calculated (the sum of the areas under each step). With continuous WTP data from an open-ended (OE) CV the mean can also be simply calculated by summing all WTP values and dividing by the sample number. The survivor function becomes more important when one considers data from dichotomous choice questionnaires.

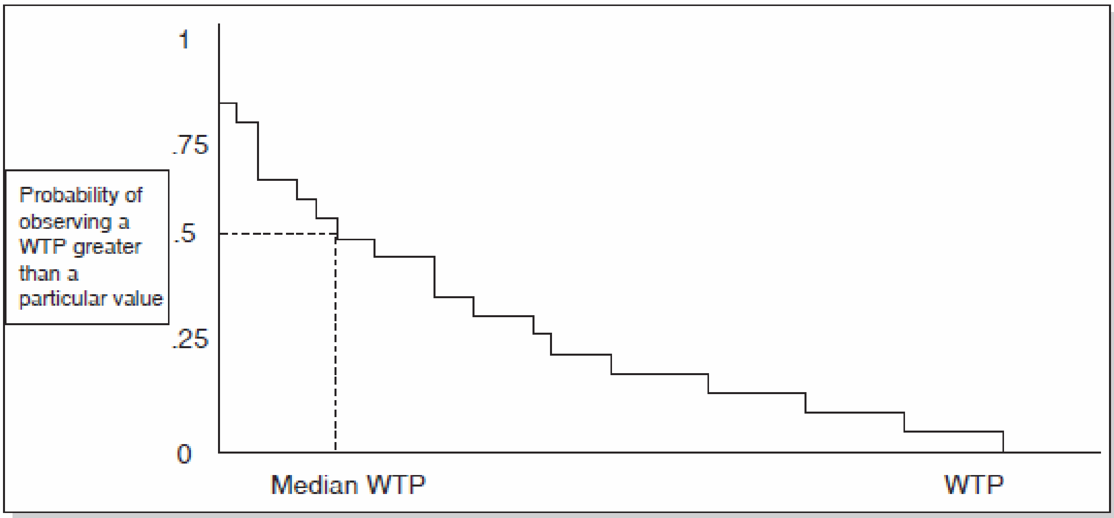

Figure A3: Survivor function for continuous WTP data.

Dichotomous Choice Data.

While mean can be calculated simply for OE data it is not so easy for DC. We cannot use a simple arithmetic calculation of mean because we do not know what the individual WTPs are. However we can construct a survivor function using DC data and estimate the area under the function and in this way arrive at the mean. This process is illustrated in Table A1 and the corresponding Figure A4 below. Remember in DC questionnaire respondents are offered a value (a bid) and asked if they would be WTP this or more.

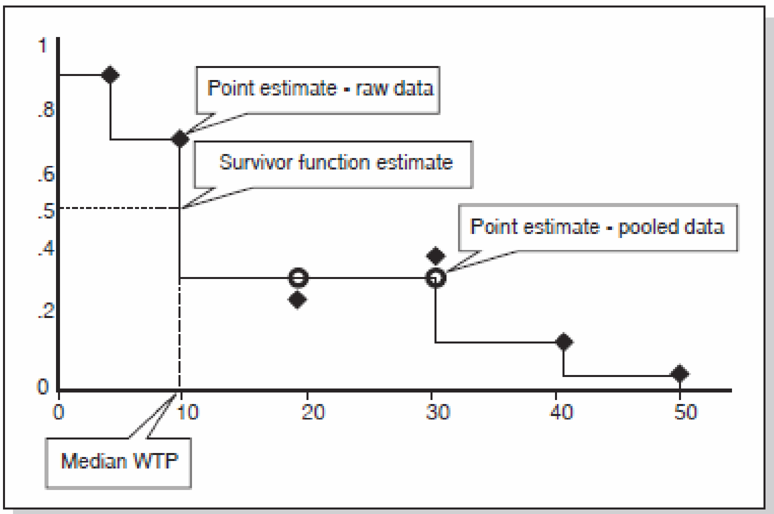

Figure A4: Estimation of survivor data with DC

Economic theory tells us that the function should slope from left to right. Fewer people WTP each successively higher bid. The estimated survivor function values for £20 and £30 bids do not correspond to this theory. We can correct this potential error by pooling the data over the two bids to produce an average result, which is then used in the graph: (25+30)/(0.31+0.35) = 0.34

Table A.1.

(a) Bid offer	(b) Number of respondents to bid	(c) Number answering 'yes' to bid	(d) Point estimate of survivor function (b/c) (raw data)	(e) Point estimate of survivor function pooled data	(f) Area under each bid/ section
0	-	-	1	-	-
5	80	70	0.88	-	4.38
10	98	72	0.74	-	3.68
20	74	25	0.31	0.34	3.4
30	85	30	0.35	0.34	3.4
40	86	15	0.17	-	1.7
50	87	6	0.07	-	.7
				Estimated mean value	17.26

In the Figure A4 a stepped line joins the point estimates. If we could map out the function for all values we would expect to see a smooth curve. However we only have discrete data points so these must be connected some how. We could use regression analysis (linear or log) to estimate the curve. One conservative approach (used here) is allocating the lower survivor probability to values lying between two points. This maps out the stepped line on graph A4.

As before the median WTP can be read from the graph. The mean (which, in the case of DC data, cannot be calculated simply as the sum of values divided by the number of observations) can be estimated as the area under the schedule in Figure A4. Of course the more bids we have data for the more accurate our graph will be. However, any increase in the number of bids offered requires a corresponding increase in sample size and hence survey cost.

1.4 INCORPORATING CLIMATE CHANGE IN THE SPATIAL MANAGEMENT OF MARINE AREAS

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1.4.0 INTRODUCTION

There appears to be consensus on the fact that global climate change is underway and that this change is having (IPCC 2007; Rosenzweig et al., 2008), and will have, more and more of an influence on the structure and functioning of marine and terrestrial ecosystems, human activities and public health, with important repercussions also for the social structure of human populations and for the economy. The increase in air and sea temperatures in different part of the world has been well documented by respected scientists. The estimate, even with some discrepancies, has put the increase in temperature of coastal waters at 1°C over the last 30 years.
The increase in temperature and the greater frequency of temperature anomalies may involve a series of interrelated problems:

- the rise in sea level
- the change in the chemical composition of sea waterand of the atmosphere
- the changes in rainfallwith consequent modification of the hydrological cycles
- the increase in the risk of coastal erosion and saltwater intrusion.
- the Invasion of alien species
- the outbreaks of indigenous species

All the main marine ecological systems and the human activities therein developed such as fishing and aquaculture, public health, conservation and the tourism industry, are destined to be influenced by climate change.

For the fishing industry, significant changes may occur in the spatial distribution of economically important species and in the circulation of currents with a cascade effect on zooplankton production and thus fishery production. Again, as regards fishery production, worrying changes are occurring in important physiological aspects of fish such as reproductive biology.

As regards the impact on marine and coastal ecosystems, one of the greatest dangers is the rise in sea level and risk of flooding. Other threats include a change in the range extensions of sensitive species, loss of biodiversity, catastrophic mass mortality, arrival of exotic species, changes in community structure and in the functioning of ecosystems and blooms of harmful species, with repercussions for the tourism activity.

Lastly, climate change is predicted to have an impact on our cultural and archaeological heritage, and on our leisure time. In addition, effects on energy supplies are feared, due to the increased demand for electricity during hot periods and the reduced productive capacity of hydroelectric power stations.

1.4.1 STRATEGIES TO TAKING INTO ACCOUNT CLIMATE CHANGE

Governments have developed strategies to taking into account climate change in several aspects of interest for human life as well SMAs . We identified the followings

- legal activities,
- specific plans and programs,
- research,
- monitoring,

- adaptation measures,
- international cooperation
- public awareness

1.4.1.1 Legal activities

Actions to fight climate change have been in place since the last two decades. Several countries approved/adopted plans for the containment of carbon dioxide emissions by 2000 at 1990 levels, aiming at achieving the country's Kyoto target and regularly updated the national plans to reduce greenhouse gas emissions. Early in the year 2000th following the ratification of the Kyoto Protocol (for those who ratified the protocol) countries developed an overall national strategy to meet the Kyoto target.

In Copenhagen the UN has set itself the goal of reducing CO2 emissions by 50% - compared to 1990 levels - by 2050. It must also prevent the global temperature increases more 'of 2 ° C above preindustrial levels. According to the official delegation of the European Parliament, who attended the conference, the weak final agreement on climate change reached in Copenhagen, is a huge disappointment. The progress of the negotiations and the final result showed the need for urgent reform in the method of work of the UN.

European Union (EU) Member States are also subject to the EU climate policy, implementing the EU Common and Coordinated Policies and Measures (CCPMs) relevant for climate change.

Only a few countries have established the national registry under Article 7 of the Kyoto Protocol while some areoperating theirregistry under Article 19 of the Directive 2003/87/CE, establishing the EU Emission Trading Scheme, according to the Regulation No. 2216/2004 of the European Commission, which require national registries to be compliant with the United Nations (UN) Data Exchange Standards specified for the Kyoto Protocol.

1.4.1.2 Plans and programs

In its efforts to meet the commitments under the UNFCCC and the Kyoto Protocol, several countries have implemented a number of sectoral and cross-sectoral policies and measures that have had or are expected to have a direct or indirect effect on the reduction of greenhouse gas emissions.

The most relevant cross-sectoral initiative is represented by the White Certificates system, aimed at promoting energy efficiency and delivering emission reductions in all the energy end-use sectors. The system is designed to achieve a primary energy saving target of 2.9 Mtoe per year by 2009. As regards additional measures still under discussion, there is a realistic chance that the White Certificate system will soon be extended firstly to 2012 and lately to 2020.

The European Union Emissions Trading Scheme (EUETS) and the flexible mechanisms of the Kyoto Protocol are also assessed and are expected to deliver reductions for respectively 13.25 and 20.75 MtCO2 per year by 2010.

In the energy supply sector, strong reductions are expected from implemented and planned policies and measures in the renewable energy sources field, where reductions for 6.87, 19.01 and 26.60 MtCO2 per year will be delivered respectively by 2010, 2015 and 2020. The major policy mechanism through which the Government supports the development of new renewable capacity is the Green Certificates system that introduced the obligation on electricity producers to feed the grid with a minimum share of electricity produced from renewable energy sources.

EU Marine Strategy Framework Directive

Adopted in June 2008, this environmental pillar of the EU's Integrated Maritime Policy aims to achieve healthy marine waters by 2020. It applies an integrated approach to ecosystems and strives to contain the collective pressure of human activities within sustainable levels. It also establishes a clear regulatory framework for adaptation to climate change and allows for the regular update of environmental targets to take into account the variations caused by climate change. The Directive calls for the development of a marine strategy by each Member State. By 2012, they must provide a comprehensive assessment of the state of the environment, identifying the main pressures on their respective marine regions, and defining targets and monitoring indicators. By 2015, they will have to develop coherent and coordinated programmes of measures. To reach the 2020 target, they will have to achieve efficient

communication and close cooperation, notably through regional sea conventions. The goal of the Marine Strategy Framework Directive is in line with the objectives of the 2000 Water Framework Directive 2000 which requires surface freshwater and ground water bodies - such as lakes, streams, rivers, estuaries, and coastal waters - to be ecologically sound by 2015 and that the first review of the River Basin Management Plans should take place in 2020.
Other EU initiatives (*http://ec.europa.eu/environment/water/marine/pdf/leaflet091117.pdf*)

The Arctic:

The Arctic is of crucial importance in regulating the world's climate. It is severely affected by climate change and, over the past 50 years, Arctic temperatures have increased by twice the global average. Sea ice cover in summer has fallen to an all-time low, so that a larger portion of the sun's heat is absorbed by the sea. A huge decrease in permafrost cover is also foreseen by 2050. The melting of this permanently frozen layer of soil or bedrock releases further greenhouse gases into the atmosphere, as well as persistent organic pollutants and mercury that accumulate in the thick fatty tissue of Arctic animals and may be consumed by humans.
In November 2008, the EU outlined its key objectives:

- to protect and preserve the Arctic in unison with its population
- to promote the sustainable use of resources - to contribute to enhanced multilateral governance of the Arctic.

The EU's main policy goal is to support adaptation to climate change and tackle its negative impacts. This implies close cooperation between the eight countries that have land within the Arctic Circle and a number of other key players in the region such as OSPAR, the Barents Euro-Arctic Council, the European Environment Agency and the Arctic Council. Through the Northern Dimension, a policy shared with Iceland, Norway and the Russian Federation, the EU is helping to fund environmental and nuclear clean-up projects in the Arctic region

Integrated Coastal Zone Management

European Commission adopted a Recommendation on integrated coastal zone management (ICZM) in 2002 and has been promoting intensively this cross-cutting instrument of the EU's Integrated Maritime Policy. ICZM integrates all policies, sectors and interests into the planning and management of human activities to achieve sustainable coastal development. The 2009 Commission White Paper on adapting to climate change provides for European guidelines on adaptation in coastal and marine areas. The OURCOAST initiative, launched in 2009, will build up a database of coastal planning and management practices, with a key focus on adaptation to risks and climate change. In addition, the Commission is planning a further proposal to strengthen the Recommendation in 2011, to further support comprehensive and effective climate strategies for coastal zones.

1.4.1.3 The role of Regional Sea Conventions

***The Barcelona Convention** for the Protection of the Marine Environment and the Coastal Region of the Mediterranean*
The European Community and the countries surrounding the Mediterranean are parties to this Convention implemented through the Mediterranean Action Plan. This formulates policies and strategies to protect biodiversity and the marine and coastal environment. The IPCC has singled out the Mediterranean as a 'hot-spot' for climate change. In 2008, the parties to the Barcelona Convention signed a Protocol on Integrated Coastal Zone Management in the Mediterranean, identifying adaptation to climate change as a priority. The Marrakesh Declaration, adopted by the Barcelona Convention in November 2009, highlights the need for urgent action to counter the serious impacts of climate change on ecosystems and resources.

***The Helsinki Convention** on the Protection of the Marine Environment of the Baltic Sea*
The European Community and all the states bordering the Baltic Sea are parties to this 1992 Convention. It strives to achieve a harmonious balance of all biological components in a healthy Baltic Sea environment, thus supporting a wide range of sustainable economic and social activities. The Thematic Assessment on Climate Change in the Baltic Sea Area (2007) projects that the average sea surface temperature could increase by 2ºC to 4ºC and the length of the ice season be reduced by two to three months, hindering further the achievement of the key goals of the HELCOM Baltic Sea Action Plan (2007).

***The OSPAR Convention** for the Protection of the Marine Environment of the North-East Atlantic*

The OSPAR Commission, comprising representatives of 15 countries and the European Community, aims to conserve marine ecosystems and safeguard human health in the North-East Atlantic by preventing pollution. Its key objective is to protect the marine environment from the adverse effects of human activities and contribute to the sustainable use of the seas. The OSPAR Quality Status Report reflects the overarching impact of climate change and the further obstacles it creates to achieving environmental objectives in the marine region. Together with the European Communities, OSPAR has adopted environmental measures to ensure the safe storage of carbon dioxide in geological formations under the seabed.

*The **Bucharest Convention** on the Protection of the Black Sea against Pollution*
The Bucharest Convention of 1992 initiated environmental cooperation in the Black Sea and its Strategic Action Plan for Environmental Protection and Sustainable Management of the Black Sea is a pillar of regional cooperation. While the European Community is not yet party to this Convention, an amendment allowing it to participate was proposed in April 2009. Biannual scientific conferences assess the impacts of climate change on the Black Sea ecosystem and on the sustainable development of coastal areas. They bear witness to the current warming up of the Black Sea waters as well as to a slow but constant rise in sea levels,

1.4.2 RESEARCH

Research programs cover mainly the sciences of climate change and to a lesser extent climate vulnerability, impacts and adaptation and several international and European projects have been funded or are still running. Do date, several projects have been funded focusing on the effects of climate change.
In Europe the creation of a new research infrastructure, the Euro-Mediterranean Centre for Climate Change, is ongoing which is a network of Public and Private Research Centres focused on research on climate change and impacts of climate change over the Mediterranean area.

Activities in the field of impact studies are mainly concentrated on the effects of sea level changes due to climate change in coastal areas and ecosystem modelingto estimate the impact of climate change plays a major role.

Several countries have established a monitoring system for GHG . Several international bodies are also involved in research on climate change. These includethe Group on Earth Observations (GEO), which will establish the Global Earth observation system of systems (GEOSS) and through several institutions it plays a large role in the Global Monitoring for Environment and Security (GMES) initiative.

1.4.3 MONITORING

Several countries have specific plans to monitor greenhouse gas emissions (CTE). Plans are aimed atregularly monitor progress in implementation of policies and measures, based mainly on indicators and sector-level emissions, and to identify additional measures to meet the Kyoto target on the basis of cost-effective analysis and taking into account progress in achieving the Kyoto target.

1.4.4 ADAPTATION MEASURES

As climate change impacts have become more and more visible, adaptation to climate change impacts is gradually gaining importance on the political agenda.

The focus of climate change adaptation is on mainstreaming actions into the sector policies rather than developing a stand alone adaptation strategy. In such respect relevant ministries, local government and specific authorities work in parallel in their respective areas.

Adaptation is developed in particular in the fields of coastal protection, agriculture, and desertification but also of biodiversity safeguard, protected areas, marine environment and environmental impact assessment. Studies are generally restricted to combat desertification by identifying the most vulnerable areas and by defining strategies for prevention, mitigation and adaptation and to the agricultural sector, focusing on management of seeding and harvesting methods and on allocation of water resources. Prevention plans of the health effects of heat-waves are also common (heat/health watch/warning system (HHWWS). They include development of a forecasting model; identification of intervention plans for each city; identification of the network of organizations /services to be involved; and evaluation of the effectiveness of the system in preventing excess mortality.

1.4.4.1 EU Adaptation measures

In April 2009 the European Commission presented a policy paper known as a White Paper which presents the framework for adaptation measures and policies to reduce the European Union's vulnerability to the impacts of climate change.

Decisions on how best to adapt to climate change must be based on solid scientific and economic analysis. It is therefore important to increase the understanding of climate change and the impacts it will have. The White Paper outlines the need to create a Clearing House Mechanism by 2011 where information on climate change risks, impacts and best practices would be exchanged between governments, agencies, and organisations working on adaptation policies.

Adapting to climate change will be integrated into all EU policies and will feature prominently in the Union's external policies to assist those countries most affected.

1.4.4.2 International cooperation

Wealthy countries promote international cooperation to offer economic resources and advanced technologies to the other countries.

1.4.4.3 Public awareness

Governmental agencies, regional and local administrations, NGOs and the mass media are active in formal education and training, including on environment and climate issues.

1.4.5 Review Of The Main Impacts Driven By Climate Change

1.4.5.1 Risks from rises in sea level

Recent indications of the rise in sea level, together with predictions of its acceleration due to global warming, indicate the considerable importance of planning human activities in the future. A sea-level rise of around 0.18 meters (the minimum forecast by IPCC for the next 100 years) (IPCC, 2007) would have a negative impact on the coast

Impact of sea-level rise on salt-marshes and vegetated mudflats

Elevation relative to mean sea level is a critical variable for the establishment and maintenance of biotic coastal communities and a threat to biodiversity (Kirwan et al. 2008); hence, for the risk of storm-surge flooding with long-term sea-level rise. In tidal areas, this elevation determines the duration and frequency at which coastal habitats are submerged by tides; this being one of the factors controlling the productivity of macrophyte communities (Morris et al. 2002). The sea-level rise is causing already the retreat of the coastal forests (Williams et al. 2003, DeSantis et al. 2007), the loss of saltmarshes (Denslow and Battaglia 2002, Chust et al. 2009), coral reef bleaching (Jokiel , Brown 2004), and loss of goods and services (Michael 2007).

Impact of sea-level rise on coastal vegetated dunes

Coastal vegetated dunes, specially the grey dunes, hold high specialised floristic diversity and are included as priority habitat in the Natura 2000 network (Habitats Directive 1992). Although the threat of dune habitat is less than for the supralittoral sandy beaches, its historically loss, degradation and squeezing by urbanization along the coast makes them more vulnerable (Chust et al*.* 2010). As the sandy beach preserves the dune vegetation from erosion, the sandy shores retreat and, especially, the overall loss of a beach makes coastal vegetated dunes more prone disappear.

Coastal squeeze

The so-called ‘‘coastal squeeze’’ effect (Schleupner 2008) is the situation where the coastal margin is squeezed between the fixed landward boundary (artificial or natural) and the rising sea level. Wetlands and marshes estuaries are squeezed by artificial urbanization, and croplands and pastures that lie within the original upper intertidal zone (Chust et al. 2009a). These croplands are protected by walls and drained to be used for agriculture purposes since they lie below the present maximum astronomic high tide level. These human activities are, hence, vulnerable to the rise in sea level expected by 2100, especially when extreme events, such as high tides and river floods, occur simultaneously. Likewise, if agriculture activity continues to decline, as throughout the 20th Century (Cearreta et al. 2004), these areas may be abandoned and susceptible to be recolonized by marsh communities (Garbutt et al. 2006, Marquiegui and Aguirrezabalaga 2009). On the basis of these latter socio-economic and climate change scenarios, saltmarshes and wetlands might expand, as predicted for other areas (CCSP 2009).

Impact of air and sea warming on coastal biodiversity

The projected rise in air temperature and sea temperatures might also interact negatively with sea-level rise having implications in terms of loss of coastline biodiversity. On the one hand, the warming of marine waters might yield the migration of intertidal species with narrow-range niche to higher latitudes. On the other hand, the eventual loss of small saltmarshes lead to estuarine habitat patches along the coast to fragment; as such, to reduce the potential connectivity between local populations (Fahrig 2003). Thus, similarly to the contraction of alpine species distribution, estuarine species populations of narrow-ranged niches, with limited dispersal potential (Defeo et al. 2009), and with fragmented habitat by historical urbanisation (Chust et al. 2009), may be at local extinction risk under climate change.

Impact of rainfall changes on hydrological cycles

The regime of flood events will also change in northern boreal countries due to the expected decrease in mean precipitation (-0.7 ± 0.3% per decade) and the increase of extreme daily rainfall (10%) for the 21st century (Moncho 2009). Thus, lower river flow and higher intensity floods are expected, as well as changes in water properties of the estuarine waters. Since the major concentrations of suspended matter take place during intense rainfall after dry periods (Uriarte et al. 2004), the new expected precipitation regime will induce the intensification of these conditions. As a consequence, the estuarine ecosystems will be subjected to more contrasted periods from lower river flow waters to higher hydrodynamic, turbid waters.

Socioeconomic repercussions

From the socio-economic point of view, there will be an impact on the industries of fishing, aquaculture and tourism, as well as on public health.

Tourism

The mortality of emblematic benthic species recorded in the Ligurian Sea (Cerrano et al., 2000; Rodolfo-Metalpa et al., 2000, 2005, 2006) has created some problems for diving tourism. Sandy beaches may suffer shoreline retreats as a consequence of a sea level rise. Likewise, the number of beaches in which the supralittoral zone is expected to disappear is high. Within this scenario of beach retreated, either tourism may reduce or alternatively the pressure of recreational seashore activities will overwhelm sandy beaches.

Adaptation strategies for spatial management of coastal area to face beach erosion (induced by sea-level rise) should include measures to promote coastal resilience such as protection, regeneration of dune plants and stabilisation, the maintenance of sediment supply and the provision of buffer zones, providing setback zones which would allow the beach to migrate landward as the sea rises (Defeo et al., 2009).

1.4.5.2 Fishery and aquaculture

Global capture fisheries production in 2006 was about 92 million tonnes, with an estimated first-sale value of US$91.2 billion, comprising about 82 million tonnes from marine waters (FAO 2009). This production has been relatively stable in the past decade with the exception of marked fluctuations. How these trends will continue is, amongst other things, a question of how climate change will affect the fisheries industry.

Climate change is predicted to have a range of impacts on marine capture fisheries, with implications for fisheries-dependent economies, coastal communities and fisherfolk (Daw et al. 2009). The effects of climate change can be direct or indirect, resulting from processes in the ecological system or by political, economic and social systems. An indirect ecological impact on capture fisheries can be reduced production and decline in yields for calciferous marine resources and ecologically related species, potentially caused by increased CO_2 and ocean acidification with effects on calciferous animals e.g. molluscs, crustaceans, corals, echinoderms and some plankton. Potential adaptation measures for reduced fisheries productivity and yields are access to higher value markets or implementation of integrated and adaptive management. An example for direct socio-economic impact is the disturbance of coastal infrastructure and fishing operations, which is potentially caused by sea level rise and increased frequency of storms. Costs of adaptation lead to reduced profitability, risk of storm damage increases costs of insurance and/or rebuilding. Small-scale fisheries are particularly exposed to this impact. Furthermore, climate change mitigation measures to reduce greenhouse gas emissions may impact fisheries by increasing the cost of fossil fuel use.

Cultured food fish supplies currently account for nearly 50% of that consumed globally (FAO, 2009) and are targeted to increase to 60% by 2020 (FAO, 2008).
The contribution of cultured components to the global fish supply has increased significantly over the last ten year period to reach 47% in 2006 . Within that, fresh water production reached 30%.

Considering that the capture fisheries component of fish supply has almost reached a saturation level of approximately 100 million tonnes per year, and that nearly 25%of this is channelled to reduction processing industries and is therefore unavailable for direct human consumption, (Jackson, 2006; Hassan et al., 2007), it is unlikely that there will be any further increases to the human food basket from capture fisheries, perhaps with the exception of potential developments in the inland fisheries sector in the tropics which appears to be gaining some momentum.

This means that growing food fish needs for human consumption, as a consequence of population increase and increasing per capita consumption among certain sectors (driven by the health benefits of consuming fish) will have to be provided primarily through aquaculture. For more current knowledge on climate change implications for fisheries and aquaculture see the comprehensive overview by Cochrane et al. (2009).

1.4.5.2.1 Impacts of climate change on fisheries

According to the recent FAO conference on climate change for fisheries and aquaculture (FAO, 2008): "Fisheries can be affected by physical and ecological impacts driven by climate changes. Formarine waters these include: water warming, which is expected to be more intense in surface waters; higher frequency and intensity climate processes, such as El Niño-Southern Oscillation (ENSO); changes in ocean salinity; increase in acidity with likely negative consequences to calcium-bearing organisms; possible change in ocean upwelling patterns; average sea level rising with large coastal land losses. Models predict a slight decrease in primary production and shifts to smaller phytoplankton which are likely to lead to changes in food webs in general". Changes are in some cases perceived as positive because new resources are available for fisheries (i.e. the northward migration of species or the establishment of stable populations of valuable species. But see UNEP-MAP-RAC/SPA, 2008). In other cases, increased temperature may have a negative impact, affecting fish physiology. This would have also significant impacts on aquaculture.

In general, climate change impacts on fisheries are expected to be rather negative than positive (Grafton 2010). To evaluate the impacts of climate change on capture fisheries, it is important not only to know the multiple pathways of impacts (e.g. global warming leads to acidification, which has effects on production ecology, fishing operations, communities, livelihoods, the wider society and economy by impacts on species composition and distribution, coral bleaching, calcification, diseases, fisheries production, yield, etc.), but also to understand the underlying relevant ecosystem (abiotic and biotic) processes. However, the specific effects of climate change on particular marine ecosystems and populations are difficult to predict and quantify. For a comprehensive review about observed effects of climate variability and change on ecosystem and fish production processes see Barange and Perry (2009), which includes regional scenarios. While climate change has direct effects on the performance of individual organisms in physiology, morphology and behaviour at various stages in their life history, climate impacts also occur at the population level via changes in transport processes that influence dispersal and recruitment. Community-level effects are mediated by interacting species, such as predators or competitors, and include climate driven changes in both the abundance and the strength of interactions among these species. The combination of these impacts results in emergent ecological responses, which include alterations in species distributions, biodiversity, productivity and microevolutionary processes (Harley et al. 2006). In the following we describe some aspects on changes in capture fisheries-dependent processes.

Changes in physiological processes

Physiological, spawning and recruitment processes and responses are used by individuals and populations to adjust to climate change. These processes evolved to match prevailing physical (such as temperature, salinity, currents) and biological (such as food) conditions to maximise the chances for an individual to become a reproducing adult. Temperature has significant influence on specific characteristics of spawning, such as its timing, the size of eggs and consequent size of larvae at hatch. In general, the spawning period of marine organisms appears to be correlated to a particular temperature range. For cold water species such as Atlantic cod and Atlantic salmon, warmer conditions lead to earlier (younger) age of sexual maturity (Brander 1994; Jonsson and Jonsson 2004). Since thermal tolerance of marine organisms is non-linear, optimum conditions are at midrange and poorer growth at temperatures which are too high or too low. For example, specific growth rates and fecundity of Altantic cod decline at higher latitudes (Pörtner et al. 2001). “Bioclimate envelope” is a term to define the interacting effects and limits of temperature, salinity, oxygen, etc. on the performance and survival of species (e.g. Pearson and Dawson 2003). Such bioclimate envelopes are used for “statistical bioclimatic envelope models” and are among the most popular approaches to simulate species-climate change impacts (Heikkinen et al. 2006). They can be used to model changes in species’ abundance and distributions patterns.

Changes in spatial distribution

Climate change is influencing the distribution of many marine species in terms of physical and biological impacts. In general, warmer-water species are being displaced towards the poles (Parmesan and Yohe 2003) and are experiencing changes in habitat size and productivity. While the range of colder-water species gets contracted. Observations of these distributional changes have been recorded in, among others, the North Sea and the North Atlantic for copepods, demersal invertebrates, intertidal organisms and fish species (e.g. Brander et al. 2003; Perry et al. 2005; Dulvy et al. 2008). Most evident are shifts in distribution near the northern or southern boundaries of the geographic range of the species, where warming or cooling theoretically drives species to higher or lower latitudes, respectively (Rose 2005). In addition to latitudinal shifts, shifts in other water depth are likely. A shift to deeper water in response to climate change is shown e.g. for the North Sea demersal fish assemblage (Dulvy et al. 2008). Although, the most rapid changes in fish communities occur with pelagic species, and include vertical movements to counteract surface warming. Apart from many distribution changes of marine species in response to climate variability or change, comparative studies highlight a substantial proportion of species that do not appear to change distribution, at least not within the last fifty years (ICES 2007; 2008).

In general, the timing of many animal migrations has followed decadal trends in ocean temperature, being later in cool decades and up to one to two months earlier in warm years (e.g. Southward et al. 2005). Beaugrand et al. (2003) showed that fluctuations in plankton abundance have resulted in long-term changes in cod-recruitment in the North Sea through three bottom-up control processes (changes in mean size of prey, seasonal timing and abundance). A possible scenario is that that climate change leads to extinctions, at least model projections result in numerous local extinctions in the sub-polar regions, the tropics and semi-enclosed seas (Cheung et al. 2009).

Changes in food web structure

Climate change is likely to affect ecosystems and their species both directly and indirectly through food web processes. Whether direct or indirect processes predominate is likely to depend on whether they are structured from ‘top down’ or from ‘bottom up’. The observed and expected changes in the spatial distribution of fish populations mentioned above may have effects on the entire food web by ‘top down’ effects. But in turn changes in the lower trophic levels may also affect commercially important fish species by ‘bottom up’ effects. After Barange and Perry

(2009) it is most common to observe synchronized changes in several trophic levels, without a clear cause-effect relationship. For example in the North Sea, changes to planktonic and benthic community composition and productivity have been observed since 1955 (Clark and Frid 2001), and may have reduced the survival of young cod since the mid-1980s (Beaugrand et al. 2003). Numerous processes have been proposed to explain how climate change influences different trophic levels (e.g. Ottersen et al. 2010), but often the precise mechanisms are unknown (Drinkwater et al. 2010). Capture fisheries depend on the productivity of the natural ecosystems on which they are based and therefore changes in primary production and how this production is transferred through the food web will obviously have a major effect on fisheries production. Although global aggregated marine primary production in not expected to change substantially over the next four or five decades, there is a stronger basis for predicting changes in production at regional level. Ecosystem productivity is likely to decline in lower latitudes (i.e. most tropical and subtropical oceans and seas) and increase in high latitudes. Furthermore, climate change is affecting the seasonality of particular biological processes, altering the food webs, which makes consequences for fish production unpredictable (Barange and Perry 2009).

Adaptation and management

Concerning the effects of climate change on fisheries productivity (recruitment, growth, mortality, distribution), and taking into account that three-quarters of worlds marine fish stocks are currently exploited at levels close to or above their productivity capacity (Bruinsma 2003) adaptation options thus centre on altering catch size and effort reductions in the level of fishing. This may benefit fish stocks and lead to sustain yields (Easterling et al. 2007).

In general, adaptation to climate impacts on capture fisheries includes reactive or anticipatory actions by individuals or public institutions. These include abandoning fisheries, developing insurance and warning systems, changing fishing operations. Furthermore, governance of fisheries needs flexibility to account for changes in stock distribution and abundance. Governance will improve the adaptive capacity of fisheries if it aims toward equitable and sustainable fisheries, accepting inherent uncertainty, and based on an ecosystem approach (Daw et al. 2009). The existing knowledge, although in many respects incomplete, provides an adequate basis for improved management of fisheries and of marine ecosystems for adapting to climate change. Due to fisheries changes, such as regional new fisheries are developing, others are 'collapsing' or reduced catches, it is important to link future monitoring and research closely to responsive, flexible and reflexive management systems (Brander 2010). It is important to develop approaches which maintain the resilience of individuals, populations, communities and ecosystems to the combined and interacting effects of climate and fishing (Perry et al. 2010). Adaptive management strategies deal with the unpredictable interactions between people and ecosystems, emphasizing the importance of feedbacks from the environment in shaping policy.This is also stressed within current publications by Badjeck et al. (2010), who explored the wider implications for local livelihoods, fisheries management and climate policies, and Grafton (2010), who published a guide to policy makers and fisheries managers, which is a decision-making framework with strategies and tactics to adapt to the bio-physical, social and economic consequences of climate change.

The impacts of climate change on fisheries are addressed through adaptation and mitigation. Due to limits and costs of adaptation measures, mitigation of emissions to minimise climate change remains a key responsibility by governments. This is underlined by the World Development Report 2010 (World Bank 2010), it shows that reducing overcapacity in fishing fleets and rebuilding fish stocks can both improve resilience to climate change and increase yields, while also reducing greenhouse gas emissions by fishing fleets. The removal of subsidies on fuel for fishing can have a double benefit by reducing emissions and promoting overfishing.

As described above, because of the uncertainties about the nature of future climate changes, fisheries management must be alert to change and responsive. Emphasis should be placed on management and conservation activities that promote resource sustainability and habitat preservation. To reach these goals, marine protected areas (MPAs) are a recognized tool. Further elements (foundations and components) of an ecosystem management approach to fisheries and aquaculture are published on FAO websites (http://www.fao.org/fishery/topic/13261/en). Guidance on management measures for fisheries when formulating management proposals for Natura 2000 sites or similar MPAs were developed for example by the project EMPAS (Environmentally Sound Fisheries Management in Marine Protected Areas). Within the EMPAS project ICES (2009) has elaborated different management options for fisheries in order to minimize conflicts and to meet conservation requirements within the German Natura 2000 sites. The research program BASIN (Wiebe et al. 2009) plans (2010 – 2019) to develop models and management strategies directed at broader ecosystem-based management goals (including protection of vulnerable habitats and preservation of ecosystem structure and function at multi-species and ecosystem levels) by providing detailed information on the dynamics of lower trophic levels and system - wide production levels that could then be used as "drivers" in management – oriented models or in fully integrated models linking the dynamics of the upper trophic levels under exploitation to the lower trophic levels. The programme's goal is to integrate the scientific results into management advice.

1.4.5.2.2 Impacts of climate change on aquaculture

Food fish production, as is the case in all other primary production sectors, is expected to be influenced and or impacted to varying degrees by climate change and the manifestations thereof are expected to occur in varying forms and to varying degrees in different parts of the world

Aquaculture has thus far received only scant attention in the major considerations of climate change induced impacts on food fish production. The most comprehensive study dedicated to aquaculture and climate change was that of Handisyde et al. (2006). In that synthesis, the authors dealt with the influence of predicted climate changes such as temperature, precipitation, sea level rise, extreme events, climate variability and ocean currents on global aquaculture.

Impacts on aquaculture production, aquaculture dependent livelihoods and indirect influences on it through fishmeal and fish oil availability were also dealt with. An extensive modelling exercise was included and a series of sub models developed that covered exposure to extreme climatic events, adaptive capacity and vulnerability.

Unlike other farmed animals, all cultured aquatic animal species for human consumption are poikilothermic. Consequently, any increase and/or decrease of the temperature of the habitats would have a significant influence on general metabolism and hence the rate of growth and therefore total production; reproduction; seasonality and even possibly reproductive efficacy (e.g. relative fecundity, number of spawnings (see Wood and McDonald, 1997); increased susceptibility to diseases and even to toxicants (Ficke et al*.,* 2007).

The lower and upper lethal temperature and the optimal temperature range for fish species differ widely. Therefore, climate change induced temperature variations are bound to have an impact on spatial distribution of species specific aquaculture activities and allow for the culture of species in areas that are currently too cold for them. On the other hand, a region may become too warm to allow the culture of a heat-sensitive species.

Also in temperate regions, increasing sea temperature may result in farmed species range shifts. This may mean that warmer water species which are currently grown in more southern European countries will become possible to farm in northern waters. These species include sea bass, sea bream, pacific oysters, manila clam and scallops. While this may be of benefit to Norwegian mariculture, the northward shift of southerly species ranges may have a devastating effect on southern and central European countries such as Spain and France.

In addition to species range shifts rising water temperatures will increase the growth rate for some species (for example Atlantic salmon, mussels and oysters), but prolonged periods of warmer summer temperatures may cause thermal stress, particularly for cold water species (for example, cod and Atlantic halibut) and intertidal shellfish (oysters), possibly preventing their culture at some sites, causing welfare problems and necessitating temperature control for broodstock of some species (Gubbins, 2006).

Direct impacts of climate change on aquaculture

To date, there has been only one reported direct impact from human induced climatic change on aquaculture. This relates to a smog cloud generated over Southeast Asia during the 2002 El Niño; it cut sunlight and heat to the lower atmosphere and the ocean by 10% and, some authors suggest, contributed to dinoflagellate blooms that impacted aquaculture in coastal areas, from Indonesia to the Republic of Korea, causing millions of US dollars worth of damage to aquaculture (Swing, 2003).

Global warming is a major impact of climate change. Increased temperature brings about associated changes in the hydrology and hydrography of water bodies, exacerbates the occurrence of algal blooms and red tides etc., all factors that could have important impacts on aquaculture.

Climate changes and in particular global warming, could both directly and indirectly impact on mariculture in temperate regions. Species cultured in those regions, predominantly salmonids (e.g. *Salmo salar*) and emerging culture of cod, *Gadus morhua*, have a relatively narrow range of temperature optima. The salmon farming sector has already witnessed an increase in water temperature over the recent past and it is acknowledged that temperatures over 17 °C would be detrimental, when feed intake drops and feed utilization efficacy is reduced. In order to develop possible adaptive measures, research has been initiated on the influence of temperature on feed utilization efficacy and protein and lipid usage for growth as opposed to maintaining bodily functions at elevated temperatures, e.g. 19 °C.

Climate change is predicted to increase global acidification (Hughes et al*.*, 2003; IPCC, 2007). This could impede calcareous shell formation in molluscs, an effect perhaps exacerbated by increased water temperature and thereby to have an impact on mollusc culture. This has received little attention and warrants urgent research. Mollusc culture accounts for nearly 25% of all aquaculture (approximately 15 million tonnes in 2005) and therefore any negative impacts on shell formation could significantly impact on total aquaculture production.

There is practically no information on the potential impact of increased water temperature on the physiology of the most relevant aquaculture bivalves. Nevertheless, if coastal plankton productivity is enhanced by higher temperatures and provided that nutrients are available, there may be a positive effect on the farming of filter feeders. However, increased temperatures associated with eutrophication and harmful algal blooms (Peperzak, 2003) could enhance the occurrence of toxic tides and consequently impact production, and also increase the possibilities of human health risks through the consumption of molluscs cultured in such areas. Clearly more research is needed to provide better forecasts of expected net effects.

Sea level rise over the next decades will increase salinity intrusion further upstream of rivers and consequently impact on fresh water culture practices. Adaptive measures would involve moving aquaculture practices further upstream, developing or shifting to more salinity tolerant strains of these species and/or to farming a saline tolerant species.

Sea level rise and saline water intrusion will also impose ecological and habitat changes,
including mangroves that act as nursery grounds for many euryhaline species. Although in general terms, most aquaculture practices presently rely only to a small extent on naturally available seed supplies (a notable exception being fresh water eel (*Anguilla spp*.), the need for continued monitoring of such changes is paramount to developing adaptive measures.

Moreover, decreased precipitation correlated to climate changes,could alter flushing times in bays and estuaries where mariculture is carried out. This could result in an increase in the accumulation of waste in the seafloor.
Predicted sea level rise and erosion of intertidal zones by increased storm activity could reduce the area available to many forms of mariculture especially in vulnerable areas such as lagoons and estuaries in Portugal and Spain (Ferreira et al*.,* 2008).

Indirect impacts of climate change on aquaculture

Indirect impacts on a phenomenon and or a production sector can be subtle, complex and difficult to unravel and the challenges in developing adaptation measures to combat or overcome them may be formidable.
Because fisheries are a major source of inputs for aquaculture, providing feed in particular and seed to some degree, changes in fisheries caused by global climate change will flow through into aquaculture systems.

Fishmeal and oil supplies

The most obvious and most commonly discussed indirect impact of climate change on aquaculture is related to fishmeal and fish oil supplies and their concurrent usage in aquaculture. Tacon et al. (2006) estimated that in 2003, the aquaculture sector consumed 2.94 million tonnes of fishmeal globally (53.2 % of global fishmeal production), considered to be equivalent to the consumption of 14.95 to 18.69 million tonnes of forage fish/trash fish/low valued fish, primarily small pelagics.

Industrial fishmeal and fish oil production is typically based on a few, fast growing, short lived, productive stocks of small pelagic fish in the subtropical and temperate regions. The major stocks that contribute to the reduction industry are the Peruvian anchovy, capelin, sandeel, and sardines. It has been predicted that the biological productivity of the North Atlantic will decrease by 50% and ocean productivity worldwide by 20% (Schmittner, 2005). Apart from the general loss in productivity and consequently its impact on capture fisheries and hence the raw material available for reduction processes, there are other predicted impacts of climate change on fisheries. It is a possibility that predicted changes in ocean circulation patterns will, result in the occurrence of El Niño type influences being more frequent. The latter, in turn, will influence the stocks of small pelagic fish (e.g. Peruvian anchovy, *Engraulis ringens),* as has occurred in the past. The influence of El Niño on the Peruvian sardine and anchovy landings and consequently on global fishmeal and fish oil supplies and prices are well documented (Pike and Barlow, 2002). Similarly, the changes in the North Atlantic oscillation winter index (Schmittner, 2003), resulting in higher winter temperatures, could influence sandeel (*Ammodytes* spp.) recruitment.

Atkinson et al. (2004) described a decrease in Antarctic krill density (*Euphausia superba*) and a corresponding increase in salps (mainly *Salpa thompsonii*), one of the main grazers of krill. It is supposed that this trend is likely to be exacerbated by climatic changes, sea temperature increases and a decrease in polar ice. The use of krill as a major protein source for replacement of fishmeal in aquaculture feeds has been advocated (Olsen et al., 2006; Suontama et al., 2007) but the current trend appears to indicate that this would not be a possibility (De Silva and Turchini, 2008).

Such changes in productivity of fisheries that cater to the reduction industry will limit the raw material available for reduction and particularly the main fisheries on which fishmeal and fish oil production is based.

In the wake of possible climate changes and consequent negative impacts on wild fish populations that cater to the reduction industries, the way forward is to make a concerted effort to increase and further develop omnivorous and filter feeding finfish aquaculture in the tropics and subtropics (Naylor et al., 1998; 2000). But such an adaptation would require profound changes in consumer and market demands.

Other feed ingredients

Although the emphasis has been on how to reduce fishmeal and fish oil usage in feeds for cultured aquatic organisms, over the last few years new problems are surfacing. For example soybean meal and corn meal are often used in feeds for cultured aquatic organisms and rice bran in tropical semi-intensive aquaculture. An important positive consideration is that in aquaculture feeds the agricultural ingredients used are almost always by-products. For example, soy bean meal used in aquafeeds is a by-product from the extraction of soy oil. Similarly, in semi-intensive aquaculture of carp species, for example in India mustard and peanut oilcakes, by-products after the extraction of oils, are used extensively in feeds (De Silva and Hasan, 2007).

Climate change impacts on terrestrial agriculture are beginning to be quantified and it is generally known that tropical terrestrial agriculture will be negatively impacted, more so than temperate regions (McMichael, 2001). A great majority of the agricultural by-products used in aquafeeds are of tropical origin. Unfortunately studies on price fluctuations of by-products are not readily accessible. There is an urgent need to evaluate the changes in availability, accessibility and price structure for agricultural by-products used in aquafeeds and to develop adaptive strategies to ensure that aquafeed supplies at reasonable prices could be retained well into the foreseeable future, so that aquaculture could remain economically viable.

Trash fish/low valued fish/forage fish supplies

There are other possible indirect impacts of climate change on specific aquaculture practices that are relatively large and, in a socio-economic context of great importance to certain developing countries. Again, these indirect impacts are related to aquafeed supplies and the ingredients thereof; in particular trash fish, low valued fish and forage fish.

It has been estimated that in the Asia-Pacific region the aquaculture sector currently uses 1.603.00 to 2.770.000 tonnes of trash fish or low valued fish as a direct feed source. The low and high predictions for year 2010 are 2.166.280 to 3.862.490 tonnes of trash fish or low valued fish as direct feed inputs (De Silva et al*.,* 2008).

The great bulk of this trash fish or low valued fish is produced by coastal artisanal fisheries in the region that provide thousands of livelihoods to fisher communities. Apart from the general predicted reduction in ocean productivity it has been suggested that the Indian Ocean is the most rapidly warming ocean and consequently climate change would bring about major changes in it and on land, primarily on productivity and changes in current patterns (Gianni et al., 2003). The situation could be further exacerbated by extreme climatic events such as changes in monsoonal rain patterns (Goswami et al., 2006) that influence inshore fish productivity and overall impact on the supplies of trash fish or low valued fish.

Impacts on diseases

There has been much debate about climate change and the associated risks for human health (e.g. Epstein et al., 1998; McMichael, 2003; Epstein, 2005). The potential trends of climatic change on aquatic organisms and in turn on fisheries and aquaculture are less well documented and have primarily concentrated on coral bleaching and associated changes.

It has been pointed out that there is a dearth of knowledge about parasites of aquatic animals other than those deleterious parasites that cause disease in humans. In the wake of the associated effects of climate change on circulation patterns and so forth and using predictions from a General Circulation Model, attempts were made to understand changes in parasite populations in temperate and boreal regions of eastern North America (Marcogliese, 2001). The overall conclusion from the simulations was that climatic change may influence selection of different life-history traits, affecting parasite transmission and potentially, virulence. It is difficult to predict the consequences of such changes on aquaculture *per se*, but the exercise points to the need for the aquaculture sector to be aware of potential and new threats from parasitism.

In temperate regions, change in temperature is likely to result in increased reproductive potential of salmon lice and other parasites. To combat this there may be an increase in the use of chemical therapeutants and their input into the marine environment. However, under conditions of thermal stress, cultured species are likely to be more susceptible to disease. Warmer conditions may also allow the establishment of exotic diseases, whereas diseases that occur under cool conditions, for example, cold water vibriosis, may become much rarer (Gubbins, 2006).

It has been also suggested that the rate of eutrophication and HABs would increase, resulting from oceanic changes brought about by climate change in some oceans and particularly in the North Atlantic and the North Sea (Peperzak, 2003; Edwards et al., 2006), not homogenously but, for example, along the Norwegian coast and elsewhere. HABs will impact marine life and human health through the consumption of affected filter feeding molluscs, commonly referred to as shellfish poisoning. Apart from this impact, the HABs could also bring about harmful effects on cage culture operations of salmon, for example. Accordingly, adaptive measures need to be set in place for regular monitoring and vigilance of aquaculture facilities in areas of potential vulnerability to eutrophication and HABs.

The possibility of climate change enabling both highly competitive species, such as the Pacific oyster (*Crassostrea gigas*) and associated pathogenic species to spread into new areas has been highlighted (Diederich et al., 2005). Related, comparable evidence of the spread of two protozoan parasites (*Perkinsus marinus* and *Haplosporidium nelsoni*) northwards from the Gulf of Mexico to Delaware Bay (Hofmann et al., 2001) has resulted in mass mortalities in the Eastern oyster (*Crassostrea virginica*). It has been suggested that this spread was brought about by higher winter temperatures, when the pathogens otherwise were kept in check by temperatures < 3°C.

Another such example is emerging: an outbreak of *Vibrio parahaemolyticus* has occurred in oysters in Alaska and in all seafood products in southern Chile. In the latter country, the first important outbreak started in early 2004 and has remained since then during summer months (Paris-Mancilla, 2005), apparently related to warmer seawater temperatures during summer. However, other factors, such as increasing nutrients in coastal zones, cannot be ignored (Hernandez et al., 2005).

It is not difficult to predict a general impact of water warming on the spread of diseases such as bacterial diseases in aquaculture because in most cases, incidence and persistence of these are related to fish stress. Increased water temperatures usually stress the fish and facilitate diseases (Snieszko, 1974).
It is clear that the spread of diseases is the most, or one of the most, feared threats to aquaculture. Examples of disease related catastrophes in the aquaculture industry include the spread of the white spot disease in shrimp farming in Ecuador and other Latin America countries (Morales and Morales, 2006) and more recently the case of ISA (*Infectious Salmon Anemia*) which is seriously impacting Chile's Atlantic salmon industry to the point where the industry might shrink in the coming two to five years at least. Given that the spread of pests and diseases is thought to be a major threat under climate change scenarios, the issue must be made a priority for aquaculture considering relevant biosecurity measures as a main adaptation.

Impacts on biodiversity

One of the special issues that received attention from the early stages of the deliberations of the Inter Governmental Panel on Climate Change was the impact on biodiversity (IPCC, 2002).

To date, some introductions of internal parasites associated with such translocations for aquaculture purposes have been reported. But for the devastating impact of one such translocation associated with the introduction of a fungal plague and the consequent dissemination of the native European fresh water crayfish (Edgerton et al., 2004), explicit evidence arising from alien species in aquaculture *per se* on biodiversity is not readily available; but this is no reason for complacency (De Silva et al*.,* 2004).
The impacts on biodiversity from alien species have mostly resulted from competition for food and space with indigenous species (e.g. Moyle and Leidy, 1992; Soto et al., 2006), alteration of habitats (e.g. Collares-Pereira and Cowx, 2004), the transmission of pathogenic organisms, as well as through genetic interactions such as hybridisation and introgression (Dowling and Childs, 1992; Leary et al*.,* 1993; Rhymer and Simberloff, 1996; Araguas et al., 2004) and other indirect genetic effects (Waples, 1991).

The question therefore, is whether the continued, if not increasing, dependence on alien species in future aquaculture developments and the associated seed stock translocations, in the wake of the global climatic change induced phenomena, would impact adversely on disease transmission as well as on biodiversity. The balance of evidence suggests that global climate change will not enhance impacts on biodiversity through aquaculture *per se.* However, in view of the changes in temperature regimes and so forth, particularly in the temperate region, the possibilities of diseases occurring among filter feeding molluscs and fish, for example, could be higher. Furthermore, any new introductions for aquaculture purposes will have to take into consideration such factors in the initial risk assessments undertaken for purposes of decision making.

In global aquaculture developments there are three major species groups that have been translocated across all geographical regions and have come to play a major role in production; these include salmonids in cool temperate waters and tilapias in warm tropical waters. The two species now account for over a million tonnes of production beyond their native range of distribution, closely followed by the white legged shrimp, *Penaeus vannamei,* and so are among the most important alien species in aquaculture.
Climate change could impact the culture of all three species groups; warming in the temperate regions will narrow the distribution range of salmonids aquaculture, whilst the opposite could be true for tilapia and shrimp. In the latter case, extending the distribution well into the subtropics, where currently the culture period is limited to a single growth cycle in the year and the bulk of broodstock is maintained in greenhouse conditions.

Extreme events such as tropical cyclones and storm surges may increase incidence of aquaculture stocks escaping into the wild environment. Increased storminess (higher frequency of strong wind speeds) predicted for certain seasons in some regions such as the North West coasts of Scotland and Ireland and Norway will increase the risk of escapes through equipment failure increasing the impact of escapees on wild populations. Relocation or improvement in equipment may offset these impacts.
Apart from causing genetic changes, escapees from aquaculture are thought to be responsible for increased parasitic infestation of wild stocks, for example, salmon in coastal waters of Canada (Krkošek et al., 2007; Rosenberg, 2008).

Perhaps mass escapes from aquaculture facilities caused by extreme weather events - very different to the small number of escapees at any one time in normal culture practices -, could influence the genetic makeup of native stocks, to their detriment in the long term. Perhaps the design of aquaculture facilities, particularly those located in areas vulnerable to unusual climatic events, needs to consider measures that would minimise mass scale escape.

1.4.5.3 Public health

Increases in temperature, especially in cities, are a well known public health problem, for the elderly and infants in particular. The arrival of harmful exotic species has already been documented (see the case of *Ostreopsis ovata*)but others are likely to arrive (see section on invasive species).

1.4.5.4 Invasive species

Many studies have looked at the impact of climate on the species geographical ranges and it is clear that substantial changes do occur. Latitudinal range expansions of species correlated to changing temperature conditions, local extinction of native species, outbreaks of some alien or native species, effects on species richness, and changes on ecosystem functioning are among the effects of climate change in the marine environment (Harley et al. 2006; Bianchi 2007; Occhipinti-Ambrogi 2007).

Although such changes will probably lead to a new equilibrium state sometime, an important consideration is the action of biological constraints during the transient dynamics of the immediate future. For example, the rate at which northern range limits change will be determined, in part, by dispersal characteristics (Waage et al. 2004). As a consequence of these constraints, there is also likely to be a lag between species loss and the appearance of “replacement species”, which will make ecosystems more susceptible to invasion by non-native species (Manchester and Bullock 2000).
Biological invasions in the marine environment represent a recognized global threat with a strong impact on biodiversity and local economies (Waage et al. 2004; Pimentel et al. 2005; EEA 2006; Gollasch et al. 2006). Alien species can affect ecosystem processes and services, such as the supply of clean water and air, or the functioning of ecosystems to provide resources which support animal and plant communities in food chains, or ecological succession.Obvious examples include alien species which affect the physical environment, such as the structure of waterways and channels, or the turbidity and quality of water (Waage et al. 2004). Such invasions are greatly assisted by climatically driven changes that affect both local dispersal mechanisms and competitive interactions between invasive and native species (Occhipinti-Ambrogi 2007). Marine protected areas (MPAs) have been established mainly to protect biodiversity and enhance fisheries. But can MPAs and other marine spatial planning frameworks effectively confront large scale threats such as the synergistic effects of biological invasions and climate change?

Developing effective prevention strategies in MPAs requires global information on pathways, vectors and potential invaders but most datasets are local or regional. The first global assessment, drawing from over 350 databases and other sources, reports information on 329 marine invasive species, including their distribution, impacts on biodiversity, and introduction pathways. Initial results confirm earlier assessments of the primary importance of shipping and aquaculture as introduction pathways and of the high levels of invasion in the temperate regions of Europe (Molnar et al. 2008). In addition, the role of maritime canals as corridors for the dispersal of marine organisms, and natural canals and Straits for the unintentional and/or shipping-mediated transport is discussed in Gollasch et al. (2006) and ECNC/CoE (2007) respectively.

The European Union for Coastal Conservation (EUCC) has emphasized the importance of marine corridors in the Strait of Dover, Bosporus and the Gibraltar Strait (Council of Europe 2002).In the Mediterranean Sea the main pathway of species introduction remains the Suez Canal (Streftaris et al. 2005) and progressive migration of Indo-Pacific species (the so-called Lessepsian migration) has been enhanced in recent years (Zenetos 2010) attributed mostly to climate change (Raitsos et al. in press).

The role of aquaculture operations in marine invasions has been described by Naylor et al. (2001).Most of these introductions probably occurred accidentally, through oyster farming (with introduced species hitchhiking on shells or equipment).

In recent decades the frequency and geographic extent of harmful algal blooms (HABs) have increased worldwide (Burkholder 1998). The occurrence and magnitude of HABs is predicted to continue to increase with climate change through a variety of mechanisms including shifts in microalgal assemblages towards more flagellates, earlier spring blooms of warm-optimal species of flagellates, and the expansion of the biogeographic range of warmer-water species into higher latitudes (e.g., increased occurrence of tropical species in temperate regions) (Sellner et al. 2003; Dale et al. 2006). One good example of climate change related introduction is exhibited by blooms of benthic dinoflagellates belonging to the tropical genus *Ostreopsis*, which are an increasingly common phenomenon in temperate regions worldwide (Shears and Ross 2009). *Ostreopsis* is one of five genera of epibenthic dinoflagellates that have been found in tropical ciguatera-endemic regions. They have been implicated in ciguatera fish poisoning and are known to produce a variety of potent marine toxins including ostreocin D, an analogue of palytoxin which is one of the most potent marine toxins known (Shears and Ross 2009). These toxins can be sequestered in shellfish (Aligizaki et al. 2008).Available data from 6 years of regulatory monitoring in shellfish farms of the eastern Mediterranean (Greek) coasts has indicated the presence of dense *Ostreopsis*spp. populationsthat are responsible for at least 30% of positive shellfish samples each year (Aligizaki et al. 2010). The presence of *Ostreopsis* has been also linked to clupeotoxism, an illness which can lead to human fatality (Yasumoto 1998). Aerosols from blooms of *Ostreopsis ovata* have also been attributed to respiratory problems in humans (Sansoni 2003). Understandably recent research on *Ostreopsis* species has focused on their toxicity and potential risk to human health (Aligizaki et al. 2008; Ciminiello et al. 2008). In the Mediterranean the increasing harmful blooms caused by the invasive microalga *Ostreopsis ovata*observed along the Italian coasts have led the Italian authorities (Environmental Ministry) to undertake a systematic monitoring programme (ISPRA) on Ostreopsis ovata and Ostreopsis spp (Brescianini et al. 2006). Consequently, in planning MPAs one should consider impacts caused by aquaculture associated alien HABs, and keep shellfish farms as far as possible and also avoid areas with a high risk of HABs due to prevailing favourable environmental conditions.

Until lately Marine Protected Areas (MPAs) were established to protect biodiversity via the complete removal of human exploitation and other activities, and thus were intended to be “islands” of nature and tranquility in a sea of hostile environment (Simberloff 2000; Boudouresque et al. 2005). It was soon recognized that this was a far too simplistic approach and the existence of a few such “islands” would not suffice to conserve biodiversity because they are not isolated from all critical impacts such as marine pollution, climate change and alien invasions (Allison et al. 1998; Simberloff 2000; Boudouresque et al. 2005). Invasive species are a clear example of the insufficiency of MPAs alone for marine conservation. MPAs have not been able to stop or limit the northward expansion of Lessepsian immigrants as evidenced by the presence of 18 lessepsian species (5 out of 27 fish present, 11 zoobenthic/out of 64, 2 macroalgae/out of 25) that are well established, in the Palm islands Nature Reserve, Lebanon (Bitar 2004). The park boundaries of many northwestern Mediterranean marine parks offered no protection to the invasion of the green alga *Caulerpa taxifolia*, once it was present along neighbouring coasts (Meinesz 1999).

International shipping is usually prohibited in MPAs but recreational (Godwin 2001) or small fishing boats (Relini et al. 2000) could equally well be the vectors of alien species introductions. Indeed an assessment of the importance of recreational boating carried out in 100 yachts in Scotland has confirmed that boating must be considered as a high risk vector for non-native species (Ashton et al. 2006). The spreading of *Caulerpa taxifolia* in the western Mediterranean was facilitated by fishing activity, in particular by bottom trawlers and trammel nets (Relini et al. 2000).

Alien invasions cannot be avoided by localized management and strict protection. For example, based on results from the EFFORTS (FP6) Program, the next designation of Marine Protected areas along the French coasts could (and certainly will) lead to deballasting prohibition close or in those areas (Masson 2010). Although preventive measures (such as regulations on ballast-water management, introductions for aquaculture, commercial aquaria and aquarium trade, and scientific research using non-indigenous specimens) or eradication would have some positive outcomes (Simberloff 2000), MPAs will still suffer numerous invasions.

The notion of MPAs has now moved to a notion of marine management aiming at marine conservation within the framework of sustainable development. Thus, human activities are not excluded from MPAs but are controlled by a series of regulations and prohibitions, and no-take zones usually constitute a small percentage of the total MPA area. This notion puts MPAs in a wider framework of marine spatial planning and management, which actually makes the establishment of a network of protected areas less difficult than before. These networks seem to be the only way to confront threats related to climate change, such as alien invasions, as no single reserve can protect against such large scale events.

The establishment of a series of reserves across latitudes or networks of reserves is what has been proposed by many scientists and wildlife managers to effectively protect biodiversity (Allison et al. 1998; Sala et al. 2002; Airamé et al. 2003; Lubchenco et al. 2003; Fernandes et al. 2005; Green et al. 2009). Networks of MPAs are much more resilient to the threats of climate change as they address uncertainty by spatially spreading potential risks and allow species to shift their distribution among reserves in response to large scale changes. An effective network of MPAs needs to span large geographical distances and encompass a substantial area to protect against catastrophes (Lubchenco et al. 2003). It should ensure local integrity, representation and replication of all major habitats particularly at sites that may be more resilient to climate change and biological invasions (e.g., far from alien hotspot areas such as large ports), protect critical habitats (e.g., fish nursery areas, spawning aggregation areas, turtle nesting sites), protect uniqueness (i.e., include biophysically special/unique places), maintain geographic diversity, and ensure connectivity of populations through larval dispersal (Fernandes et al. 2005; Green et al. 2009). Such networks of protected areas of various scales have been applied or designed in several regions, such as the Great Barrier Reef Marine Park (Fernandes et al. 2005), the Gulf of California (Sala et al. 2002), and Kimbe Bay in Papua New Guinea (Green et al. 2009).

Conclusively, the enhanced rate of introduction of alien species due to climate changes should be taken into account in planning any development within MPAs. The establishment of a series of reserves across latitudes or networks of reserves is what has been proposed by many scientists and wildlife managers to effectively protect biodiversity. Stricter, industry-wide control measures could be developed and legal and enforcement structures strengthened to restrict intentional and accidental introductions of harmful species. While avoiding invasive phenomena appears impossible due to natural corridors, other human operations e.g. recreational boats that are vectors of fouling alien species, deballasting near marine reserves, and marine farms can be limited away from MPAs. Policy makers, conservation practitioners, and the aquaculture industry should continue to work together to prevent any future invasions, by improving practices and perhaps limiting new (Molnar et al. 2008). Monitoring of MPAs for the presence and expansion of harmful invasive alien species should be in line with the WFD and MSFD policies.

1.4.6 References

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1.5 INCORPORATING GEOHAZARD RISK ASSESSMENT IN THE SPATIAL MANAGEMENT OF MARINE AREAS

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1.5.1 INTRODUCTION

Herein the existing literature and the existing information are reviewed in order to assess geohazard risk and to incorporate it in the integrated tools of the spatial management of marine areas.

The term geohazard is used to define the geological state that represents or has the potential to develop further into a situation leading to damage or uncontrolled risk. Geohazards are not dangerous per se, but from the perspective of human being certain.

Geohazards can be relatively small features, but they can also be of huge dimensions and affect local and regional socio-economy to a large extent. Some examples of geohazards are: submarine landslides, naturally occurring gas hydrates, mud flows, mud volcanoes, earthquakes, volcanoes and the associated tsunamis.

In the frame of MESMA project the tsunamigenic potential and its impact to the Europe's coastal area is approached. The Japanese word **tsunami** is used within the scientific community “to describe a series of waves that travel across the ocean with exceptionally long wavelengths (up to several hundred kilometres between the wave crests in the open ocean)” (DAWSON, et al., 2004). As the tsunami waves approach the coastline, they are deformed, their speed decreases and their height increases significantly. As the tsunami waves strike the coastline they often cause widespread flooding across low-lying coastal areas and in many cases cause heavily destruction of property and occasionally loss of life.
The natural forces for the tsunamigenesis are:

- earthquakes
- volcanic activities and
- submarine landslides

Sudden displacements of part of the sea floor produce tsunamis. A seismic event or a volcanic eruption event can cause a rupturing of the overlying water column and generating a tsunami. Submarine landslides can also generate, in a similar way, a tsunami.

Tsunamis are an important threat for damaging human constructions and coastal erosion. Tsunami events like this occurred in the Indian Ocean (December 2004) and their destructive effects lead the international scientific community to pay more attention to these phenomena and to build risk scenarios, which can be included in the **Integrated Coastal Zone Management (ICZM).** Tsunami is a natural phenomenon, which cannot be avoided. The study of their destructive effects can be very useful in planning and drawing up risk maps.

In the following at first the mentioned natural phenomena and their tsunamigenic potential for Europe will be discused. Then the way from the tsunami event to coastal inundation will be presented. The sensitive areas along the coastal areas of Europe under inundation risk will be highlighted. Finally it is suggested to **incorporate the geohazard risks** in the **SMA plans**, using the principles of the Integrated Coastal Zone Management (ICZM).

1.5.2 THE TSUNAMIGENIC POTENTIAL IN EUROPE

Europe shows important differences in tsunami source mechanisms within its different areas. The geodynamical regime of Europe differentiates the eastern North Atlantic from the Mediterranean regions.

In the western and northern coastal areas of Europe the most common source of tsunami are the submarine landslides. In the Mediterranean region the majority of tsunami is generated by earthquakes that can also in some cases trigger submarine landslides. Volcanic activity is also an important tsunami generation process in the Mediterranean region.

A relative well known tsunami was generated in the northern North Atlantic region ca. 8000 years ago by one of the world's largest underwater landslides (the Storegga Slide). This tsunami caused flooding along parts of the Norwegian coastline (up to levels 20–35 m above sea level) while along the UK mainland coastline the highest flood levels reached up to ca. + 6 m above sea level(Dawson et al., 1988; Bondevik et al., 1997).
On November 1755 AD a tsunami was generated offshore west of Portugal as a result of an extremely large earthquake (the Great Lisbon earthquake). The maximum tsunami flood levels along parts of the coastline of Portugal were in the order of + 20 m above sea level.

Another relative well-known tsunami has taken place in the eastern Mediterranean ca. 3600 year BP, caused by the eruption of the Santorini (Thera) volcano.

A number of studies has been realized, regarding tsunami phenomena. Most of these studies use mathematical models to simulate how the tsunami approach the coastal area [Synolakis, 1991; Titov and Synolakis, 1998; Pelinovsky et al., 2002; Tinti and Armigliato, 2003; Weiss et al., 2006]. The models are underlying some limitations (mostly absence of morphological data), rely only on the hypothesis of wave behavior and describe their effects on a very theoretical coastline.

Historical tsunami events are testified in different sites by geomorphological and /or sedimentological evidence, like washover funs, boulders, etc and also by archeological evidence [Dominey-Howes, 2002; Kelletat et al., 2005; Mastronuzzi and Sanso, 2000; Mastronuzzi and Sanso, 2004; Mastronuzzi and Sanso, 2006].
The driving forces for tsunamigenesis, as it is mentioned, are:

- submarine landslides,
- earthquakes and
- volcanic activities.

Conclusively we can highlight that by contrast to the general perception that tsunami hazard in Europe is negligible,the study in details of European records of tsunami, together with the identification of tsunami in the geological record, indicate clearly that the hazard is real.

1.5.3 FROM THE TSUNAMI EVENT TO COASTAL INUNDATION

When a tsunami approaches the coastline, its characteristics [height, wave length, period] change in two different zones:

- in the near-shore zone and
- in the overland flooding zone.

In the ***near-shore zone*** the tsunami characteristics are influenced by the bathymetry and the coastal geometry (topography and geomorphology). In fact, when a tsunami travels into shallow waters, the ***shoaling effect*** occurs and its height increases, it also increases when a tsunami is concentrated on headlands. The slope of the sea floor is one of the factors that determines the breaking depth as well as the wave height at breaking point. Wave breaks when the ratio between wave height at breaking point and water depth at the same point, H_b / W_d is approx. 0.71 – 0.78 (Keulegan and Patterson, 1940). Experimental data indicate a value of approx. 0.44 – 0.60 for horizontal bottoms (Massel, 1997) and 0.78 – 1.03 for steeper bottoms (Calvin, 1972).

When the tsunami reaches the coastline it may:

- flood the landscape without breaking, like a rapidly rising sea level,
- arrive as a train of breaking waves,

- become a turbulent water-hammer.

To evaluate the tsunami's inland penetration, the inundation, we need an important parameter the **tsunami height.** Starting from the evaluated tsunami height it is possible to estimate the extent of inland flooding, also called inland penetration and/or inundation.
To calculate the inland penetration of a tsunami impacting the coast perpendicularly the following formula can be used (Hills and Mader, 1997),

X = (HFL)1.33 n-2k

H_{FL} is the wave height at the shoreline,
n is the Manning number,
k is a constant corresponding to 0.06 for many tsunamis (BRYANT, 2001).

Hills and Mader (1997) have provided procedures and empirical formulas that could be used for determining the inland penetration of the overland flow from tsunamis.
In this formula some important parameters are taken into account:
- the type of coastal morphology,
- the density of population along the coastal areas,
- the type of land use,
- the density of the vegetation,
- the type and density of the buildings.

Tanaka et al. (2007) resumed all these conditions with a Manning number of 0.05.

1.5.4 COASTAL AREAS OF EUROPE UNDER INUNDATION RISK

Taking seriously into account that tsunami hazard in Europe is real, coastal areas under inundation risk have to be identified.

The most vulnerable areas of Europe's coasts are the low land coastal areas.
In the map of Fig 1.5.1 we have at a glance the areas under tsunami risk. The German coasts, the Holland and Belgian coasts are the most vulnerable coastal areas. A part of the western French coasts as well as the deltaic areas of the Mediterranean countries are also sensitive areas in a tzunamigenic event.

Figure 1.5.2 gives additional information for European areas under tsunami risk. It is self evident that coastal areas under erosion dynamic in Europe are also more vulnerable in a tsunamigenic phenomenon.

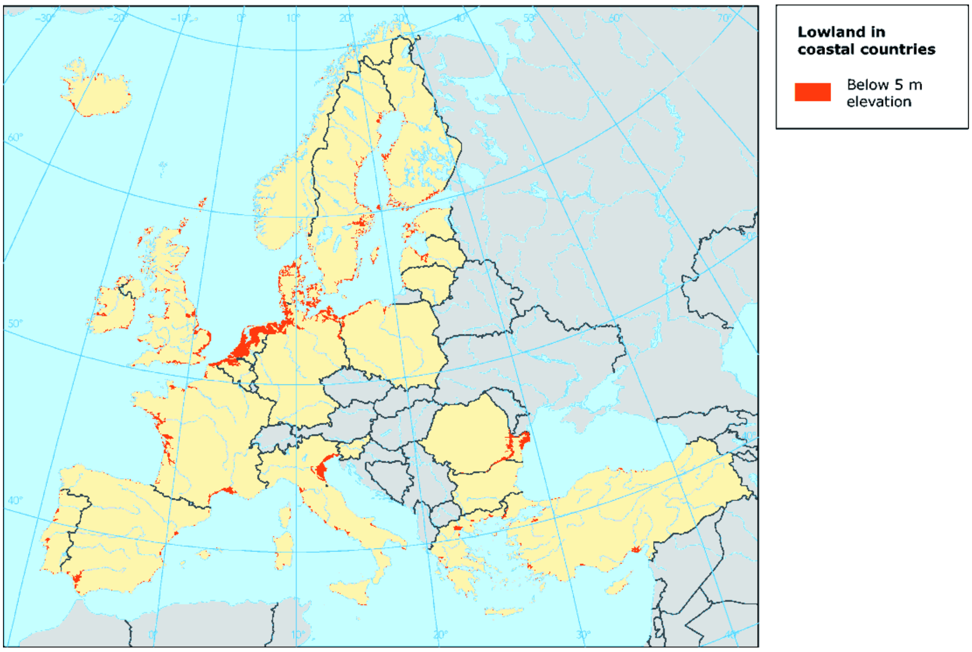

Fig. 1.5.1: Lowland in coastal European countries(source EEA)

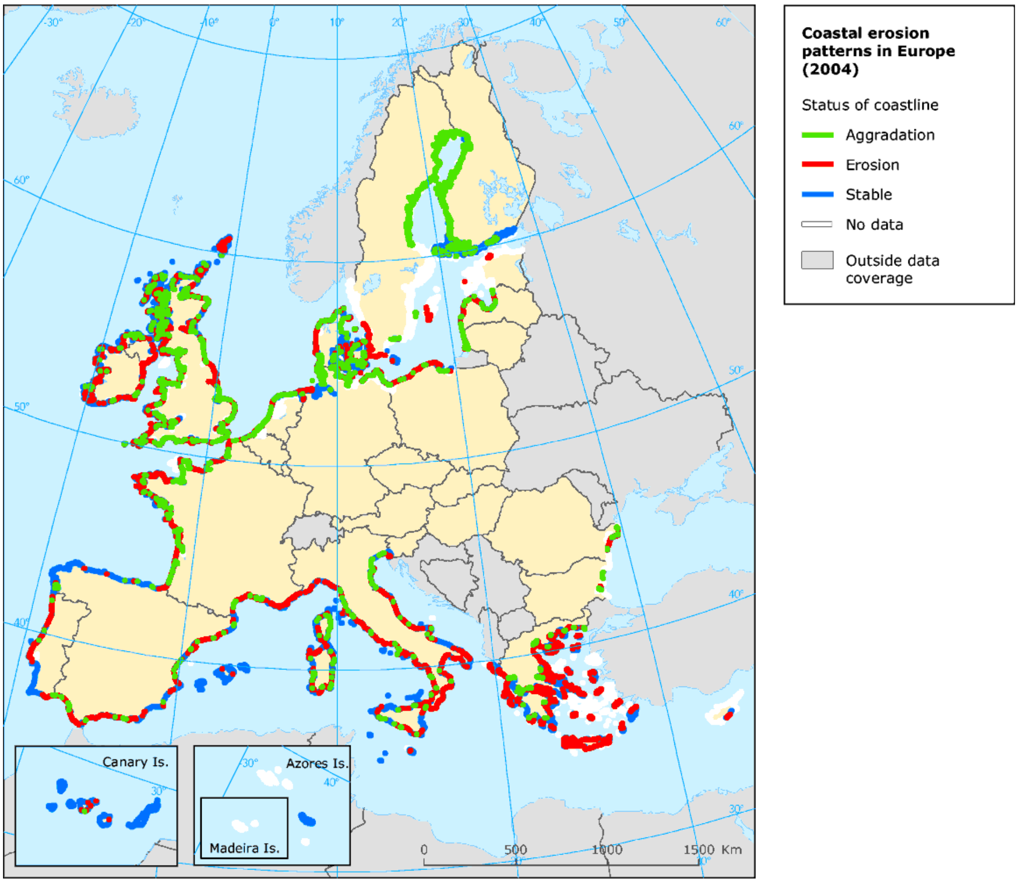

Fig. 1.5.2: Coastal erosion patterns in Europe' s coastal area (source EEA)

1.5.5 INCORPORATION OF GEOHAZARD RISKS IN THE SMA PLANS

The incorporation of geohazard risks, tsunami risks, in the SMA plans can be realized based on the principles of the Integrated Coastal Zone Management (ICZM).

The maps of Fig. 1.5.3 and Fig. 1.5.4, in combination, with the previous regional information about low land coasts and coasts under erosion dynamic help to design a property plan and to emphasize to measures in order to minimize property loss and human's live loss.

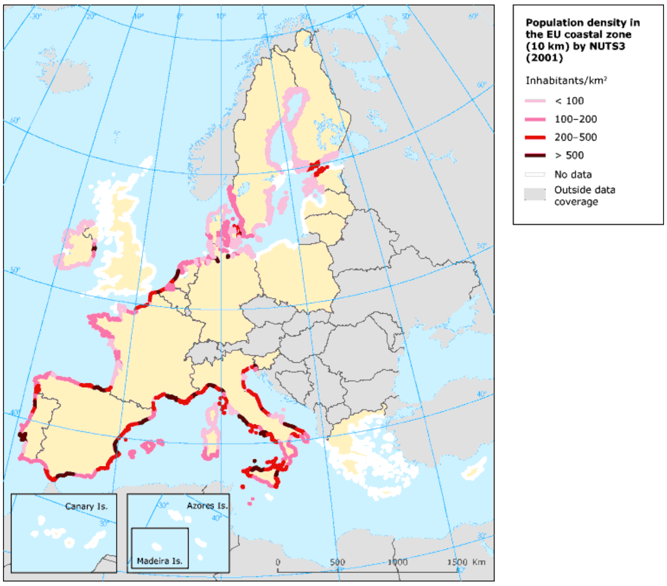

Fig. 1.5.3: Population density in the EU coastal zone (source EEA)

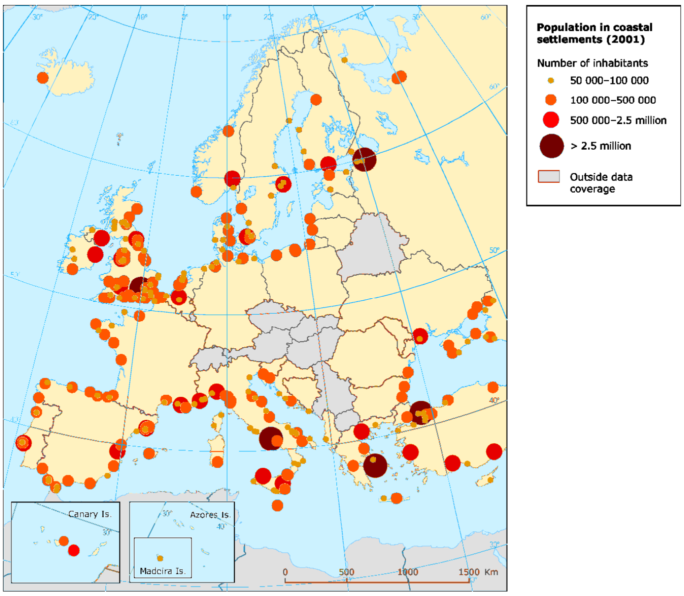

Fig. 1.5.4: Population in settlements along the Europe's coastline (source EEA)

1.5.6 Major tsunami implications which have to be included in a ICZM planning.

a. Property loss

A tsunami event can lead to erosion of substantial land in certain coastal sectors. Beaches can be removed and probably the underlying beach rocks and the steep scarps along the coast will be exposed. Lagoons can be connected with the open sea. The means that the land occupied by users can be no longer available (for tourist resorts the loss of beaches means the loss of business).

b. Buffer zones

Zoning is a strong management tool in ICZM and buffer zones are often used to provide a transition between different use zones or separate potentially conflicting use zones. Along the coast, a buffer zone has to be established to specify the minimum distance for permanent structures to be constructed away from the coastline. The establishment of buffer zones poses problems for the coastal population and leads to various complications. It can also intensify conflicts between different activities, which need to be near the coast. The fishery activities and the tourist "industry" are some of these activities.

c. Coastal ecosystem rehabilitation plans

The ICZM planning has to deal, among others, with issues relating damage to coastal ecosystems and their restoration.

d. Livelihood restoration

e. Establishment of a Tsunami Warning System

The ICZM has also to adopt ***a Tsunami Warning System*** as an approach to mitigate disasters in the coastal areas.

f. Making invisible risks visible: What would be the basis of a tsunami risk education and a tsunami risk communication in coastal communities.

Risk education and risk communication can be defined as the exchange of information among interested parties about the nature, magnitude, significance or control of a risk. Interested parties include government agencies, industry groups, unions, the media, scientists, professional organizations, communities and individual citizens.

The "totally unexpected events" like tsunami events are accounted the invisible risks [YAMASHITA, 2009]. This is the reason why in the conventional planning the probability of a tsunami event is not included.

What would be the basis of a tsunami risk education and a tsunami risk communication in coastal communities?

- Awareness that some risk information is "invisible".
- Awareness of possible miscommunication or manipulation of risk information.
- Culture of questioning and accountability.

1.5.7 Outlook – Suggestions

In the frame of MESMA project it would be very useful to try to include tsunami risks in the SMA plans of each study site.

Detailed maps of the areas as well as maps of the human's settlements and human's installations in combination with estimations of the different tsunami scenarios for each area can lead to the design of concrete planning for the public awareness, the decision makers, the politicians.

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CHAPTER 2: REVIEW OF TOOLS AND METHODS TO MONITOR AND EVALUATE SEABED HABITATS AND POPULATIONS

2.1 HABITAT MAPPING (METHODS AND TECHNOLOGY)

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2.1.1 MARINE HABITAT CLASSIFICATION SYSTEMS AND SCHEMES

The seabed can be characterised and classified at different spatial scales ranging from local environment (habitat) with factors affecting individual organisms, to ecosystems and landscapes (or seascapes) where the substrates, terrain and oceanographic settings influence biological communities or populations.

Habitat definitions

In 2005 the ICES Working Group on Marine Habitat Mapping (WGMHM) reviewed definitions for the terms habitat and marine landscape/seascape (ICES 2006). They advocate the following definition of "habitat" as used in the context of marine habitat classification and mapping:

"A recognizable space which can be distinguished by its abiotic characteristics and associated biological assemblage, operating at particular spatial and temporal scales."

This definition is not intended to alter the classical definition of habitat – which has long been defined as:

"The locality in which a plant or animal naturally lives." (Darwin 1859).

Rather, the working group definition is intended to extend the classical definition to address ambiguities surrounding the term in the context of marine habitat mapping. Within the marine mapping context, three useful definitions that express the inclusion of biotic and abiotic elements to varying degrees are as follows:

"An identifiable and distinct association of physical characteristics and associated biological assemblage used by an organism or community." (Allee et al. 2000).

The European Union Nature Information System (EUNIS) definition of habitat places even more emphasis on biotic communities, but continues to recognize the abiotic elements:

"Plant and animal communities as the characterizing elements of the biotic environment, together with abiotic factors (soil, climate, water availability and quality, and others), operating together at a particular scale." (EUNIS 2002).

Also working with the relationship between biotic elements and the abiotic environment, other definitions of marine habitat place more emphasis on what can be readily mapped, with particular focus on the affiliation between physical elements and macrofaunal assemblages from benthic surveys. Kostylev et al. (2001) and Valentine et al. (2005) define habitat as:

"Spatially recognizable areas where the physical, chemical, and biological environment is distinctly different from surrounding environments." (Kostylev et al. 2001 and Valentine et al. 2005).

These types of variation in habitat definition with degrees of biological or species inclusion reflect the different objectives and applications of the data. The Allee et al. definition is tied to U.S. habitat mapping programs where essential fish habitats are a high priority. The EUNIS system has been extensively applied to near-shore habitat mapping in the United Kingdom and Ireland, where detailed species information has been gathered. The definitions proposed by Kostylev et al. (2001) and Valentine et al. (2005) have been used in studies of the Gulf of Maine, North America, in which several large tracts of multi-beam acoustic mapping have been completed.

In addition to the scientific considerations when using the term "habitat", there are also a variety of legal and administrative definitions of such terms. As one example:

"Natural habitats means terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural" (EU Habitats Directive 92/43/EEC, Anonymous 1992).

The definition of habitat advocated by the ICES WGMHM is built upon the following assumptions:

1) The classical ecological and biological definition of habitat as given above conveys the central intent and meaning of the term.
2) While dependencies exist between individual organisms and their environment, maps of physical environmental features without reference to past or current biotic presence are not habitat maps, but rather more appropriately described as physiotope maps, or maps of marine physiographic conditions. Geological maps of the seafloor are one example of this type of map. Likewise, Greene at al. (2005, 2007) distinguish between 'potential habitat' and 'actual habitat'. While seafloor morphology and texture can be mapped at high resolution using modern digital bathymetric mapping technology (e.g. multibeam echosounder and sidescan sonar), such maps only depict potential habitat unless habitat associations of a species are determined. With this knowledge, actual habitats are mappable and need to be confirmed by ground-truthing.
3) Some authors prefer to more clearly distinguish the physical habitat from the species or group of species which occur within it, referring to habitat strictly as the physical environment of the species or group of species and the combination of habitat and species (generally as groups of species in a community) as a biotope (see below). This distinction has merit, although longstanding convention has established habitat as a synonymous term for biotope (Connor et al. 2004). On top of that, problems arise when biological features (e.g. mussel beds and coral reefs) structure the seabed and the distinction between physical habitat and biotope becomes blurred.
4) From a practical standpoint, marine mapping based largely on physical characteristics of the environment are often good surrogates for habitat maps, however such maps are models based on assumed correlations that may require extensive validation work and biotic surveys before evolving into true habitat maps.
5) Mapped habitats, like all map features, are dependant on the spatial domain and grain employed. At coarse map grain, the thematic habitat descriptions will tend to be generalized. For example, a rocky intertidal habitat at one map scale can be mapped as a series of discrete habitats within tidal zones at finer spatial scales. In this sense, habitats can be hierarchical, though over very large areas, other terms such as "seascape" are more appropriate. Habitats defined solely by physiographic features allow the cartographer to map habitats at any spatial or temporal grain and domain. By including the organism or community, appropriate scales are clarified, and the value of the marine habitat map concept is enhanced.
6) Historic species range, ecoregion boundaries, as well as internal heterogeneity serve to define the habitat domain and limit the extent of physiographic extrapolations. For example, two seemingly identical habitats that occur in different ecoregions should be independently validated for species composition and ecological function.

Biotope

A biotope was originally defined by Dahl (1908) as a complex of factors, which determines physical conditions of existence of a biocoenosis. As such, biotope and biocoenosis were respectively considered as abiotic and biotic parts of an ecosystem (Olenin and Ducrotoy 2006). This original concept was widely used, until in 1990´s a new definition of biotope emerged in the context of classifying marine habitats in the coastal zone. In that sense, a biotope is defined as the combination of an abiotic habitat and its associated community or assemblage of species (MESH Project 2008). This is reflected in the Marine Habitat Classification for Britain and Ireland (Connor et al. 2004), which has habitat-related distinctions at the top two levels (broad habitats and main habitats) and biotope-related distinctions further down the hierarchy (biotope complexes, biotopes and sub-biotopes).

This new definition of biotope differs considerably from the original one as it combines both abiotic (habitat) and biotic (community) factors, whereas the biotope sensu Dahl (1908) was only referring to the physical (abiotic) factors. It also implies a different definition of habitat, which is here taken as the abiotic factors (the physical and environmental conditions such as substrate type, exposure to waves and currents, light, salinity etc.) only. Biotope in this sense is synonymous with habitat as defined in the previous chapter and has been used so as part of the MESH project (MESH Project 2008).

The notion of community also varies depending on the authors. While a community could be defined as a biological entity consistent of interdependent organisms, it is sometimes also used for a statistically defined assemblage of cooccuring species (Olenin and Ducrotoy 2006). Despite the fact that such assemblages are the product of statistical analysis, they are often misinterpreted as biological/ecological entities carrying out a recognised function in the ecosystem. In the working definition of a biotope, a community is identified as a group of organisms occurring in a particular environment, presumably interacting with each other and with the environment, and identifiable by means of ecological survey from other groups (Hiscock and Tyler-Walters 2003).

Biotopes and habitats can be characterized at different spatial scales, ranging from the local environment with factors affecting the vicinity of individual organisms to ecosystems and landscapes where the substrata, terrain, and oceanography influence biological communities or populations. Many biotopes are linked to marine landscapes such as offshore banks, deep-water channels, canyons, submarine slide fields, and abyssal plains, characterized by very different environmental conditions.

Landscape / Marine Landscape / Seascape

In general use and as a legal term in some countries, the term “seascape” is often used to describe a view or picture of the sea, i.e. at its surface. In the context of landscape ecology and marine habitat mapping, the above terms are more specific, and relate to seabed topography. They refer to an area of integrated landforms and biota in which a range of habitat types may occur. In this sense, the terms imply a spatial extent larger than habitats, and smaller than large marine ecosystems and marine ecoregions. Particular marine landscape types may comprise a suite of habitat types which occur together in a recognisable pattern, such as in an estuary or on a seamount. Appropriate spatial scales for habitats – and thus for landscapes are variable.As with the term “habitat”, some authors focus on the physiographic and oceanographic elements of marine landscapes, excluding biotic structure.

There are several approaches to seascape and habitat mapping. Greene et al. (1999) provide a classification scheme for deep seafloor habitats where the issue of scale is dealt with in a hierarchy of classes. The same approach is applied in the EUNIS system (Davies et al. 2004). Both classification systems take into account the biological components of the habitat classes. However, whereas the Greene et al. (1999) classification scheme uses the biological components as modifiers of geological and geomorphological features at an intermediate level (macro and meso habitats), the EUNIS classification emphasises taxonomic composition at the lower levels. The main point of difference is scale and the application of ecological knowledge in the habitat map.Mapping “habitat” normally requires knowledge of the benthic biota and ecosystem details that are not required for landscape approaches. The habitat approach offers most chance of success as prediction normally must involve integration of biota and statistical approach to habitats. Buhl-Mortensen et al. (2009b) make use of broad-scale information about benthic communities to check which geomorphological features display faunal differences.

The concept of marine landscapes is a broad-scale classification of the marine environment based on geophysical features. It was first developed for Canadian waters by Roff and Taylor (2000). The landscape approach is well suited for offshore areas where biological information is often scarce. Roff and Taylor (2000) developed a classification system based on environmental factors such as water temperature, depth/light penetration, substratum type, exposure and slope. They termed the classes ‘seascapes’. Currently, the term ‘marine landscapes’ is commonly used. This level represents an intermediate scale between regional seas and habitats. A marine landscape should have consistent physical and ecological characteristics and provide a practical scale related to the management of human activities such as fishing and hydrocarbon exploration.

Buhl-Mortensen et al (2009b) used multivariate statistical methods to relate taxonomic composition with environmental factors and broad-scale topographic features. This approach can aid the analysis of finer scale variation by identifying sub-sets of data for further analyses.

In the Mediterranean, all classification schemes are biased in favour of shallow habitats, revealing objective gaps of knowledge on deep-sea environment. The classification from Pérès and Picard (1964), largely used in the whole Mediterranean basin, combines physical and biological information to classify ecotypes that represent biological assemblages within habitats. In 2006, the RAC/SPA (Regional Activity Centre for Specially Protected Areas) of Tunis established a new classification scheme suited for describing the various marine habitat types of the Mediterranean region.The classification proposed is hierarchical, phytosociological and uses the following as bases of references: 1) the zonation as defined by Pérès and Picard (1964); 2) the granulometric nature of the seafloor classified according to the model adopted by Dauvin et al. (1994). More recently, Fraschetti et al. (2008) simplified RAC/SPA classification and provided explicit guidelines for classification criteria of Mediterranean habitats, potentially extendable at European scale, calling for more attention to the functional properties of marine habitats.

2.1.2 THE RELATION BETWEEN SCALE IN HABITAT MAPPING AND MANAGEMENT NEEDS

The special relevance for this topic is in relation to what decisions can be made in relation to scale of maps.

Because the way in which a variable is sampled will affect the scale at which it can be meaningfully displayed or classified, it is important to match how habitats are sampled with the overall scale of the project. Unfortunately, data collected at one scale may lose its meaning when displayed at a scale that is inappropriate for either the resolution (spatial density) or extent of the data set. Thus, while data collected at a particular resolution within a given area may be adequate for one purpose, it may not be suitable for other habitat mapping needs. Ultimately, the highest level of a hierarchical classification system that can be applied to an ecosystem will depend on those variables that can be sampled at the smallest scale.

The table below (adapted from Booth et al. 1996 and Greene et al. 1999) schematizes the correspondence between standard mapping scales, resulting display resolutions and management scales.

Scale	1 mm = (m)	1 mm^2= (ha or m^2)	Planning Class	Features that can be displayed at this map scale
1:10^6	1,000	100 ha	Hemisphere	Megahabitats, Biogeograhic regions,species and fisheries range boundaries
1:500,000	500	25 ha	Regional	Megahabitats, Biogeograhic zones, gross shoreline features, resource management jurisdictions
1:250,000	250	6.25 ha	Sub-regional	Megahabitats, Geologic mapping, river mouths, bays, estuaries, habitat features, fishing grounds
1:50,000 to 100,000	50-100	0.25 to 1.00 ha	Local	Mesohabitats, Marine reserve boundaries, small islands and inlets,habitat classes
1:24,000	24	576 m^2	Local, site	Mesohabitats, Fine grain habitat mapping, off-shore rocks, kelp beds, substrate type
1:10,000	10	100 m^2	Site	Mesohabitats, High resolution habitat mapping, seabed texture
1:1,000 to 5,000	1 - 5	1 - 25 m^2	Site	Macro- and Microhabitats, Biotic community and site level mapping

2.1.3 NATIONAL AND INTERNATIONAL MAPPING PROGRAMMES

The habitat mapping project MESH (Mapping European Seabed Habitats) was initiated in 2004 to develop a framework for marine habitat mapping in Europe and to develop the first co-ordinated seabed habitat maps for the north-west Europe region. MESH is being undertaken by a consortium of twelve partners across Belgium, France, Ireland, the Netherlands and the UK, and receives co-funding from the EC INTERREG IIIB programme for north-west Europe. Its duration was extended in late 2006.

The main achievements of the project are:

a. an on-line catalogue of habitat mapping studies, now containing over 1000 metadata entries (www.searchMESH.net/metadata).

b. the collation of available habitat maps from the five countries of north-west Europe covered by MESH, their conversion into a common GIS format and their standardisation in a common habitat classification scheme (the European Environment Agency's EUNIS scheme).
c. development of Data Exchange Formats (DEFs) for habitat mapping and other associated types of data to facilitate rapid exchange of data across countries.
d. development of an interactive guide to marine habitat mapping, including protocols and standards for mapping survey techniques and the interpretation of data, to promote future collection of data which are of high quality and inter-operable (allowing them to be combined with other mapping data). This includes a series of Recommended Operating Guidelines (ROGs) and survey metadata standards. The MESH Guide was released in September 2007 (www.searchMESH.net/mapping-guide).
e. new surveys in over 60 study areas, which have added to the available knowledge on seabed habitats, tested and evaluated survey procedures and facilitated the transnational exchange of expertise. A final survey, in the deep-water canyons in the south-west approaches to the Celtic Sea, was undertaken in summer 2007.
f. development of a series of modelling techniques and studies, from very broad scale to very fine scale, which enable the production of maps that will predict the distribution of habitats. As the coverage of existing habitat maps is still very patchy, and mostly confined to coastal areas, such modelling techniques can fill large gaps in mapping coverage until such times as higher quality habitat surveys can be undertaken. Examples of broad-scale modelled outputs include a marine landscape map for UK waters (www.jncc.gov.uk/UKSeaMap), and similar maps for Netherlands, Belgian and French waters. Modelled distribution maps to EUNIS levels 3 or 4 are now available for Belgian, Netherlands, French and UK waters (www.searchMESH.net/webGIS).
g. case studies examining the use of habitat maps for a variety of management and industry needs.
h. dissemination of the data, reports and maps emanating from the project via the MESH web site including the interactive MESH webGIS application.
i. communication with stakeholders via newsletters, conferences and bespoke stakeholder workshops in each country, to help ensure the work undertaken meets end-user needs. A 'final' MESH conference in Dublin in March 2007 was attended by over 200 delegates from 21 countries (www.searchMESH.net/conference). The conference proceedings were published in October 2007 and include a CD containing the conference presentations.

MAREANO (Marine AREAdatabase for NOrwegian coast and sea areas; www.mareano.no) is a Norwegian mapping programme which covers multiple scales of habitat/biotope mapping in Norwegian waters. MAREANO is a multidisciplinary mapping programme, focusing on offshore areas in the southern Barents Sea. It was initiated to address the lack of knowledge of the seabed and environment, which is required for informed, sustainable management. The mapping programme includes acquisition of multibeam echosounder bathymetry and acoustic-backscatter data together with a comprehensive, integrated biological and geological sampling programme. Mapping outputs from the project consist of bathymetric data, geological maps (morphology, hard and soft seabed, sediment grain size distribution, sedimentary environment, geological genesis), biological maps (including biodiversity and faunal distribution), and benthic biotope maps. Using a variety of sampling tools to ensure that organisms on all types of seabed are represented, MAREANO offers a unique insight into the diversity of benthic species and habitats.

The MeshAtlantic (Mapping Atlantic Area seabed habitats for better marine management) project is funded by the Interreg Atlantic Area Programme 2007-2013. It is already accepted and it is foreseen to start in 2010. Participating countries are France, Ireland, Portugal, Spain and U.K. Mesh-Atlantic main objective is to promote harmonised production and use of marine habitat maps covering the Atlantic Area. To reach this goal, it intends to adapt and enhance previous achievements to the area, hence bringing the region onto equal footing with northern Europe. The key outputs of the project are basically three different sets of maps made homogeneous across the area in the EUNIS nature classification. These maps are primarily those which already exist, but need enhancement and harmonisation, detailed bespoke maps covering a limited set of Natura 2000 sites - with some transnational ones – as well as a broad-scale modelled map resulting from the assemblage of readily available data layers. All of these outputs will be freely accessible on an interactive web mapping system. By building and acquiring this knowledge, Mesh-Atlantic intends to serve the community at large. Based on a multi-faceted communication plan, the project will be carried out in close collaboration with marine environment users, among them fishermen, conservationists and managers who act for sustainable use of marine space and resources.

The Si.Di.Mar project (Sea Protection System) is an italian monitoring programme of the marine coastal environment. It was initiated in the 90s and is, to date, the only italian national database on the marine environment (http://www.sidimar.tutelamare.it/).The information currently archived and visualized through GISinclude:
Marine Environment Data: these are data concerning the evolution of parameters measured at sea during oceanographic cruises or using fixed and mobile stations (water, plankton, sediment, biota - molluscs, benthos)
Maps of *Posidonia oceanica* meadows in the italian coastal areas

- Location, zoning and boundaries of Marine Protected Areas established in the italian seas
- Distribution of alien species in the Mediterranean
- Strandings of cetaceans, sharks and sea turtles in Italy

2.1.4 MAPPING TECHNOLOGIES - POSSIBILITIES AND LIMITATIONS

The approach to technologies can be very broad. For the habitat (ecotope) map of the Waddensea there has been a limited number of variables (depth, emergence time, sediment grain size, silt content, wave action, current speed, organic matter etc). With three to four levels in each variable the end-product contained more than 250 classes.

Methods for the collection and analyses of visual observational data from the seabed can be classified into analyses of frames (still images), georeferencing of occurrences of individual observations of organisms and seabed surface features and analyses of video sequences (Parry et al. 2003, Mortensen and Buhl-Mortensen 2005). Commonly, the observations are analysed from the playback of video records after the cruise. Buhl-Mortensen et al (2009b) used field observations collected in a systematic way with the aid of the Campod Logger event-recording software. This can be regarded as a “quick and coarse” approach to seabed habitat analysis. However, the data lack detailed taxonomical precision and quantification, because they not allow for rewind or pausing of the record in order to study details.

2.1.5. PREDICTION OF HABITATS AND BIOTOPES

Maps of benthic habitats and biotopes can be produced based on visual inspection, mapping of environmental conditions, and interpreting the data collected through multibeam mapping. However, because of the great expense and time needed, it is virtually impossible to produce full-coverage habitat maps from large areas based on samples and observations only. Therefore, relationships between environmental proxies with full spatial coverage and distributions of biological communities can be used to predict the distribution of biotopes.

Bert Brinkman (Imares-Texel) developed a method for "predicting" intertidal mussel beds (Mytilus edulis). These maps were used in spatial management. Norbert Dankers theme also have maps based on similar approaches for (intertidal) Cockles (*Cerastoderma edule* and Seagrass (*Zostera marina*). In the MESH project the Belgians developed this further for the subtidal North Sea. Bert Brinkman just started on a "potential" habitat map for subtidal mussel beds.

In the Mediterranean, Garofalo et al. (2004) proposed a geostatistical approach to develop a large-scale thematic mapping of marine benthic biocenoses in the Strait of Sicily, based on data from scientific trawl surveys. Specifically, presence of indicator species and substrate-type records were analysed togheter with catch data, to assign a biocenosis category, based on the Pérès-Picard (1964) classification, to sampling sites. Predictions at unsampled locations were obtained by applying non-parametric geostatistical models.

Buhl-Mortensen et al. (2009a) describe a procedure for producing maps of predicted biotopes that combines information on the distribution of biological communities with environmental factors and indicators. Detrended correspondence analysis (DCA) was used to relate bottom environment [including multiscale physical descriptors of the seabed derived from multibeam echosounder (MBES) data] and faunal distribution to find the best physical biotope descriptors. Prediction of biotope distribution was performed using a supervised GIS classification with the MBES-derived physical seabed descriptors with the strongest explanatory ability (depth, backscatter, and broad-scale bathymetric position index) identified by the DCA.

Galparsoro et al. (2009) considers the identification of seafloor morphological characteristics, together with wave energy conditions, that determine the presence of European lobster (*Homarus gammarus*); and it predicts suitable habitats over the Basque continental shelf (Bay of Biscay), in summer. This approach demonstrates the applicability of the method in case studies where only presence data are available, together with the inclusion of environmental variables obtained from different sources.

2.1.6 References

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2.2 INDICATORS MEASURING TRENDS IN ECOLOGICAL QUALITY STATUS OF BENTHIC HABITATS

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2.2.1 INTRODUCTION

General introduction

The marine environment is confronted with an increased development of anthropogenic activities such as offshore wind energy, fishing and aquaculture, dredging, mineral extraction, shipping etc. (EEA, 2007). In order to reveal pressures or impact of these activities on the ecosystem, the quality status must be determined.

In Europe, the umbrella legislation for addressing the ecological quality of the water systems is the European Water Framework Directive (2000/60/EG) for lakes, rivers, transitional and coastal waters and the European Marine Strategy Framework Directive (2008/56/EC) for marine waters. The concept of both directives is comparing the current status of an area to an expected status when human alterations were minimal (i.e. the good or reference status), followed by certain interventions to get the area to that desired good status (Van Hoey et al., in prep). From a management point of view, there is an urgent need for simple indicators that can be unequivocally interpreted and which are based on accurate and reliable information on the condition of the biological ecosystem (Borja A, Miles A et al., 2009).

The implementation of the Water Framework Directive (WFD) has provoked a large debate on the use of benthic (and other) indicators to determine the Ecological Quality Status (EQS) of the estuarine and coastal waters in Europe (review by Dauvin and Ruellet, 2007b). The implementation of the Marine Strategy Framework Directive (MSFD) is still in full progress. Several European expert groups are trying to define indicators of good environmental status for the eleven qualitative descriptors as defined in the MSFD. The usefulness of benthic indicators to classify and assess the ecological quality of marine systems is reviewed by Borja et al. (in press).

Indices versus indicators

A distinction can be made between indices and indicators. '**Indices'** are considered as one possible measure of a system's quality status, while **'indicators'** are largely derived from these indices, but they are used to measure the deviation according to a certain reference point. Both 'index' and 'indicator' are used to evaluate and assess ecological integrity in relation to a specific qualitative or quantitative feature of the system. The term 'indicator' is frequently used at the interface between science and policy, but in many cases no differentiation is made between both terms.

According to Dauvin (2007b) an **indicator** is 'any measure that allows the assessment and evaluation of a system state, as well as of any management actions for conservation and preservation that occur in the system'. However, Pinto et al. (2009) almost attributed the same role to the term **indices**, by stating that 'indices are very useful tools in decision-making processes, since they describe the aggregate pressures affecting the system, and can evaluate both the state of the ecosystem and the response of managers'.

It may be clear that although there is a great demand for clear definitions of technical terms in science and policy, the meaning of 'indicator' is still ambiguous (Heink et al., 2010). In this study we will try to consistently use the term 'indicator'.

Classification of indicators

As shown above, environmental indicators reflect trends in the state of the environment, help the identification of priority policy needs and the formulation of policy measures, and monitor the progress made by policy measures in achieving environmental goals. Environmental indicators also represent a powerful tool to make the general public aware off environmental issues.

There are different types of indicators, often corresponding to the level of management system to which the indicator refers. To clarify the inter-relationships between human beings and the environment, the Organisation for Economic Co-operation and Development (OECD), the Food and Agriculture Organization (FAO), the European Environment Agency (EEA), Eurostat and many other institutions have adopted conceptual frameworks for the derivation of indicators. Conceptual frameworks provide a convenient way to organize indicators in relation to the different ecosystem components and ensure that the indicators correspond to different purposes within the system. Commonly used frameworks are known as Pressure-State-Response (PSR), Driving force-State-Response (DSR) or Driving force-Pressures-State-Impact-Response (DPSIR) (EEA, 2003; Indeco, 2005). A classification of indicators based on the DPSIR framework is given by Smeets and Weterings (1999). All conceptual frameworks have the same goal and are devised in a way that they reflect the pressures of human activities, the state of human and natural systems, and the responses of society to changes in those systems (Table 2.2.1).

Table 2.2.1: Classification of different types of indicators, partly based on the DPSIR framework

A) Descriptive indicators

Describe the actual situation with regard to the main environmental issues, such as climate change, acidification, toxic contamination and wastes, in relation to the geographical levels at which these issues manifest themselves. The different types according to the DPSIR model all fit under these ‘descriptive indicators’.

Driving forces indicators (D) express socioeconomic development such as the growth rate of populations in coastal areas or trends in production and consumption which induce pressure on the environment.

Pressure indicators(P) represent the pressure on the environment exerted by the driving forces (e.g. port development, discharge of dredged materials).

State indicators (S) measure the quality and quantity of localized physical and chemical phenomena in the environment and their evolution over time (e.g. concentration of a contaminant in the environment).

Impact indicators (I) describe the immediate impact on the environment(e.g. mortality of marine organisms, ecosystem degradation).

Response indicators (R) measure policy responses to achieve a certain objective (e.g. the number of boats decommissioned if capacity reduction was the objective).

B) Performance indicators

In contrast to descriptive indicators, performance indicators compare actual conditions with a specific set or series of reference conditions and values (Unesco 2003). Performance indicators measure the deviation from those predefined environmental targets.

C) Efficiency indicators

These indicators express the relation between separate elements of a causal chain. They reflect whether or not society is improving the quality of its products and processes in terms of resources, emissions and waste unit output.

D) Total welfare indicators

This indicators try to find an answer on the question: “Are we better on the whole?”.

What determines the choice of the indicator

There is no indicator that consistently outperforms all other classes of indicators. A benthic indicator is unlikely to be universally applicable, since organisms are not equally sensitive to all types of human disturbance (Buhl-Mortensen et al., 2009), to geographical specifications (Dauvin, 2007a) or to different habitat types (Tagliapietra et al., 2009).
The issue of selecting indicators that reflect the properties of habitats and ecosystems, and which can be used in a management context, e.g. in relation to Ecological Quality Objectives (EcoQOs, ICES 2001a) or in relation to regulatory needs (Rees et al. 2008*),* has been extensively discussed in recent decades. Several difficulties pop up when the same indicators are used within the local, national and international management of an area (Daan, 2008). Many indicators only measure broad-scale processes, which are not appropriate on a smaller scale or when specific human activities have to be managed (Dindsdale et al., 2004).

Although there is an urgent need for environmental information to support management decisions, as was shown on a symposium in London (November 2007), progress on the development of good indicators has been rather slow. This can be exemplified by the long time it took to develop an as yet incomplete set of EcoQOs for the North Sea by OSPAR based on scientific advice from ICES (OSPAR, 2008; Skjoldal and Misund, 2008). A major drawback is the fact that the determination of thresholds for biological indicators in management frameworks is mainly based on a “trial and error” approach (Rees et al., 2008).

The choice of an indicator depends on a lot of factors, such as goal, sensitivity to certain pressure types, geographical area, parameter information needed for calculation of the indicator values, available monitoring information. By using a conceptual framework, like the DPSIR framework (see above), the selection process may be facilitated (Marin et al., 2008; Bayer et al., 2008; Samhouri et al., 2009). Several criteria usually come back in the selection of environmental state indicators. They should be: 1) scientifically valid; 2) simple and easy to communicate; 3) able to show spatial and temporal trends; 4) sensitive, i.e. provide an early warning of adverse effects; and 5) cost-effective.

In spite of their diversity, many indicators are based on the same paradigm: disturbances are generating secondary successions during which tolerant species are at first dominant and then progressively replaced by sensitive species (Pearson and Rosenberg, 1978 in Grémare et al., 2009). Based on different papers (Diaz et al., 2004; Quitino et al., 2006; Rees et al., 2006; Borja et al., 2007), it can be summarized that the main differences between the existing indicators can be attributed to:

1) **Purpose**. Some indicators measure performance (e.g. reduction in effluent load in response to a certain regulatory action), while other indicators measure the environmental or ecological quality status of a habitat or ecosystem.
2) **Susceptibility to natural variability.** A core issue in using environmental variables or indicators for management purposes is the ability to discriminate between anthropogenic impacts and natural variability (ICES, 2001; Rees et al., 2008). Natural variability may mask or interact with the effects from anthropogenic impacts. There is a risk of oversimplification when inappropriate indicators are used in the management framework that lack the power to discriminate between anthropogenically induced vs. natural variability (Hauge et al., 2005). However, most indicators are confronted with the problem that they are able to detect disturbances when evaluating the environmental conditions, but this is regardless of whether the conditions were influenced by anthropogenic impacts or by natural processes (Kröncke and Reiss, 2010).
3) **Sensitivity**. Different indicators may lead to contradictory responses for the same impact, due to differences in sensitivity to certain disturbances (see further).
4) **Tolerance of species**. Different methods are used to determine the tolerance or sensitivity of species. Grémare et al. (2009) compared the AZTI Marine Biotic Index and the Benthic Quality Index (see further). Both indicators rely on the Pearson-Rosenberg principle, but differ in the way species sensitivity/tolerance levels are assessed (next to the consideration of species richness, and the procedures used to convert computed indicators of good ecological status). The AMBI assesses sensitivity/tolerance levels on basis of ecological groups and the compilation of expert knowledge, while BQI assumes that species sensitivity/tolerance levels vary according to geographical location.
5) **Survey designand sampling efficiency**. Survey design is of fundamental importance since impact measurements require an evaluation against contemporaneous changes at reference locations. Substratum heterogeneity and versatility of the sampling gear, related to the power to detect changes and the unpredictability of the environmental target itself, all influence the efficiency of sampling.

6) **Scale**. This is another core issue with regard to the selection of environmental variables or indicators for management purposes. For example, for the management of marine aggregate extraction in shallow waters in the Belgian part of the North Sea, detailed maps on the distribution of benthic habitats and communities are produced as part of a regulatory framework (Degrendele et al., in press, De Backer et al., 2010). These are necessary to monitor the changes due to the removal of bottom substrate and eventually the recovery of the habitat and benthic communities after cessation of the extraction works.

On a larger scale, satellite-based monitoring of fishing vessels (VMS) may be used to describe the spatial distribution of fishing activities. However, such data only cover the most recent years. The accumulated impact of bottom trawling on the benthic habitat over several decades is difficult to assess, because of the lack of data before trawling started and the lack of reference areas with a similar bottom type, to directly compare fished and non-fished habitats (Duineveld et al*., 2*007).

All these differences must be taken into account when choosing the best suitable indicators.

2.2.2 OVERVIEW OF EXISTING INDICATORS

The purpose of this review is to give an overview of the existing indicators that can be used to measure trends in the ecological quality status of benthic habitats. During a workshop (WKBEMET), jointly organized by ICES and MARBEF (ICES, 2008), several scientists reviewed the existing benthic indicators. The outcome of that workshop was used as a basis, and has been updated with more recent information.

Information is collected per fauna group and knowledge gaps and problems are displayed. Finally some recommendations for future research are given. We tried to be as complete as possible, based on the knowledge of the contributors to this review and information found in the literature. For some faunal groups, indicators might have been developed, but apparently no information on these indicators was found as yet. However, Borja and Dauer (2008) stated that the scientific world should focus on the evaluation of the suitability and feasibility of the existing indicators, rather than developping new ones.

Comparable with the unclarity between 'indices' and 'indicators', also the terms 'habitat' and 'ecosystem' may be confusing. The UN convention on Biological Diversity (www.biodiv.org/welcome.aspx) describes 'habitat' as the place or type of site where an organism or population naturally occurs (Article 2). From this definition it follows that habitats can range in size from small to large, but this also holds for the term 'ecosystem'. The distinction between 'habitat' and 'ecosystem' is that the latter not only consists of the habitats but also of the inhabiting (and visiting) species.

On the other hand, the Marine Strategy Framework Directive (2008/56/EC) is one of the driving forces in marine environmental research and policy for the following decades. According to the MSFD, the term 'habitat' addresses both the abiotic characteristics and the associated biological community, treating both elements together in the sense of the term 'biotope'. In this study we will follow this 'combined' definition of the term 'habitat'.

The aim of this chapter is to give an overview of the applicability of both abiotic and biotic indicators to reveal the ecological quality of benthic habitats. Several tables and annexes are included on the commonly used indicators per ecosystem component.

2.2.2.1 Abiotic indicators

Several abiotic factors can be used to describe the habitat. Benthic communities show a strong correlation with many of these abiotic variables, but only few are proposed to be used as abiotic indicator of the benthic habitat. Organic matter content (or total organic carbon) has been proposed as an early-warning signal of change in the quality of the marine ecosystem (UNESCO/IOC Ad-Hoc Benthic Indicator Group, 1999, 2005). Dissolved oxygen concentration and the amount of litter are included as descriptors of marine ecosystem integrity in the MSFD.

Total organic carbon

Total organic carbon (TOC) is the amount of carbon bound in an organic compound and is often used as a non-specific indicator of water quality. Organic matter in sediments is an important food source for benthic fauna, but an overabundance of TOC can cause reductions in species richness, abundance, and biomass of the benthos. This is mainly due to the fact that high concentrations of TOC can lead to oxygen depletion and an increase in toxic byproducts (like ammonia and sulphide) associated with the breakdown of organic matter and co-varying with sediment particle size (Hyland et al*.,* 2005).

Pearson and Rosenberg (1978) developed a graphic model to describe a generalized response pattern of benthic communities to organic enrichment. While not a measure of causality, it is anticipated that critical points of TOC (i.e. concentrations at which benthic diversity or biomass changes drastically) may be used as a general 'screening' indicator. This can be used to evaluate the likelihood of a decreased quality of the sediment and the associated bio-effects over broad coastal areas that receive organic wastes and other pollutants related to human activities (Shine, 2005; Magni et al*.*, 2009).

Dissolved oxygen concentrations

In aquatic environments, oxygen that is produced byphytoplankton or oxygen from the atmosphere dissolves in the water. This is used for respiration by all animals, including those from the benthicv habitat. When the supply of oxygento the bottom is cut off or the consumption rate exceeds the resupply of oxygen, the dissolved oxygen (DO) concentration rapidly declines beyond the point that sustainsmost animal life, leading to hypoxia and even anoxia. Hypoxia near the bottom may be induced by two principalfactors, namely water column stratification, which limits the exchange of dissolved oxygen between thebottom water and the oxygen-rich surface water, and the decomposition process of organic matter in bottom waters, which uses high amounts ofdissolved oxygen (Diaz, 2001).

Oxygen deficiency (hypoxia and anoxia) may be the most widespread anthropogenically induced deleterious effect in estuarine and marine environments around the world. Examples of the ecological and economic effects of anoxia on benthos and fisheries in European Seas are given in Table 2.2.2. Over the past 20 years the number of coastal areas with seasonal hypoxia in bottom water is increasing rapidly. The main cause for this is suggested to be an excessive delivery of nutrients to the system. Global warming may accelerate these effects and enlarge the affected areas (Diaz, 2001).

Hypoxia alters both the structure and function of benthic communities, but effects may differ with regional hypoxia history (Rabalais et al*.*, 2010).Diaz and Rosenber (1995) found that a DO concentration of <2 mg l^{-1} was potentially harmful to bottom dwelling fauna. Best et al*.* (2007) proposed DO thresholds for five EQ classes within the European Water Framework Directive. The major effects of reduced DO in bottom water and in the upper sediment layer on the benthos is discussed in Grey et al*.* (2002) and Levin et al*.* (2009). For the MSFD, assessments of oxygen concentration need to be conducted in critical areas and in critical seasons when it is found to be ecologically relevant. Two indicators are proposed with reference to DO concentration: a) extent of area with spatial or temporal hypoxia, and b) ratio of oxygen/hydrogen sulphide concentration.

Table 2.2.2: Comparison of ecological and economic effects in European coastal zones due to anthropogenic induced hypoxia. Data from various sources, compiled by Diaz (2001).

System	Area affected (km^2)	Benthic response	Benthic recovery	Response fisheries
Kattegat, Sweden–Denmark	2 000	mass mortality	Slow	Collapse of Norway lobster, reduction of demersal fish. Hypoxia prevents recruitment of lobsters
Black Sea Northwest Shelf	20 000	mass mortality	Annual	Loss of demersal fisheries, shift to planktonic species
Baltic Sea	100 000	eliminated	None	Loss of demersal fisheries, shift to planktonic species. Hypoxia is bottle-neck for cod recruitment

Comparison of the responses to hypoxia of different fauna groups has shown that the mobile crustacean fauna is less tolerant to hypoxia than other benthic fauna groups. The relationship between oxygen concentration and species richness is generally stronger for hyperbenthos than for infauna. Hyperbenthic amphipods, tanaids and ostracods showed the strongest response to hypoxia, which makes them good candidates as 'early warning' indicators of changes in oxygen conditions. Amongst the infauna organisms, molluscs seemed to be the most susceptible group to low DO. However, most of the infauna is more dependent on other factors, such as sediment composition. The dominating infauna group in numbers of species are the polychaetes. Many of these polychaete species are adapted to sustain periods of low oxygen concentrations in the sediment. Interestingly, species richness of polychaetes correlates equally well with low oxygen and carbon concentrations in the sediment.

Marine litter

Marine litter can be described as any persistent, manufactured or processed solid material that is discarded, disposed or abandoned into the marine and coastal environment. Marine litter consists of plastic, wood, metal, glass, rubber, clothing, paper, etc. Marine litter can be divided in three 'harmful' categories. Firstly, marine litter causes a **social** problem, related to the reduced aesthetic value and public safety. Secondly, marine litter is harmful for economic reasons, e.g. cost to tourism, damage to vessels and fishing gear, loss of fishery operations, and cleaning costs. The third harmful category are the ecological consequences of marine litter, like physical damage, sublethal effects and even mortality in plants and animals through entanglements (ghost-fishing), ingestion of both macro- and microparticles (mainly microplastics), the release of associated chemicals, and the facilitation of the invasion of alien species, which can alter the local benthic community structure (Katsanevakis, 2008; Galgani, 2010).

The effect of litter on the abundance and community structure of soft-bottom epibenthic megafauna was investigated by Katsanevakis (2007). Thompson et al. (2009) made a compilation of up to date knowledge on the environmental (pollution) effects of plastic and the risks to human health. A whole chapter is dedicated on a review of existing survey methods to monitor the abundance of marine litter (Ryan et al., 2009). Because marine litter will persist for years and even centuries, the evaluation of sources alone will not suffice, and long term monitoring in the marine environment will be required. In the OSPAR, HELCOM and Black Sea regions, standards for a beach litter survey have been developed, which can be adjusted, harmonized and applied to other regions.

Two indicators that were proposed by the MSFD as descriptors of good environmental quality, are 'trends in the amount of litter washed ashore or deposited on coastlines', and 'trends in the amount of litter floating at the surface, in the water column and deposited on the sea-floor'. This includes the analysis of the composition, the spatial distribution and the source of marine litter. These indicators need to be developed further, based on the experience in some sub-regions (e.g. North Sea), and may need to be adapted for other regions. The goal of this indicator within the MSFD is that the amount of litter has to be at a level that ensures that the structure and function of the ecosystems are safeguarded and that benthic ecosystems are not adversely affected (Galgani, 2010).

2.2.2.2 Biotic indicators

Depending on the approach, biotic indicators can be divided in different groups (in the ICES WKBEMET report (ICES, 2008) the different indicators were classified as univariate, functional and multimetric indices next to some multivariate and modeling approaches. Mainly the first three groups within the univariate indices (descriptors, indices of diversity and graphical methods) are commonly used for all ecosystem components. The other are specifically developed as macrofauna indicators. Many more indicators have been developed based on macrobenthos data. See the following annexes (Paragraph 2.2.5) for updated lists on macrobenthic indicators.

Please consult the WKBEMET report (ICES, 2008) for the complete references in the table in annex. Section 2.2.5.2. in Annex: Indicators can be based on one parameter (univariate) or on several parameters (multimetric or multivariate). Multimetric indicators are based on a number of different metrics while multivariate approaches rely on statistical analyses to develop predictive models.

Indicators should reflect general ecological principles and in many cases comprise measures (e.g. diversity indicators, number of species) which can easily be calculated for different ecosystem components. However, most indicators that are developed to assess the ecological quality status of the benthic habitat are based on macrofauna data. In principle, most univariate indices and even some multivariate indicators that were developed for one specific benthic component may be applied to other benthic components as well, while for a certain impact one ecosystem component may be better suited than the others to demonstrate changes in the status of an indicator. In this review, an overview of the commonly used indicators is given, with regard to the applicability of six important benthic ecosystem components.

Macrophytes

Marine benthic macrophytes include two fundamentally different groups of plants: seaweeds (macroscopic algae) and seagrasses (vascular plants). Macrophytes form the structural base for some of the most productive ecosystems of the world, including rocky and soft bottom intertidal and subtital zones, coral reefs, lagoons and salt marshes. (Mann, 1973; McRoy and Loud, 1981). Macrophytes, but in particular seagrasses, provide substrate, habitat and shelter for plants and animals, including economically important species (Harmelin-Vivien et al*.,* 1995; Francour, 1997; Pollard, 1984; Edgar 1999 a; b). As the leave canopy diminishes wave energy and currents (Fonseca and Calahan, 1992), macrophytes affect the sediment stability (Fonseca, 1989) and the retention of particles (Bulthuis et al*.,* 1984, Dauby et al*.,* 1994).

The seaweeds are the largest and structurally most complex organisms within the benthic macrophytes, but the three major taxonomic groups (Chlorophyceae, Phaeophyceae and Rodophyceae) all developed similar morphologies as an adaptation to life in a common habitat. Therefore, the ecological classification of seaweeds is mainly based on thallus morphology, longevity and life history (Feldmann, 1951; Chapman and Chapman, 1976; Russell, 1977). Littler and Littler (1980) proposed a classification of the seaweeds into six ecologically meaningful groups, based on a ‘functional form’ model. The experimental tests proved that the functional characteristics, such as photosynthesis, nutrient uptake and grazer susceptibility, are related to the morphology and surface:volume ratios of the seaweeds (Littler 1980, Littler and Arnold 1982, Littler and Littler 1984).

Seaweeds and seagrasses comprise two evolutionary and physiologically different groups (Larkum et al*.* 1989, Hemminga and Duarte 2000, Lobban and Harrison 1994), but have often been examined together because of morphological-functional similarities and the apparent overlap in habitats. Marine benthic macrophytes are photosynthetic, sessile organisms and may act as sensitive indicators of changes in the abiotic and biotic aquatic environment (Orfanidis et al*.*, 2001, 2003). A good example is their response to water eutrophication. Changes in nitrogen and phosphorus concentrations do not necessarily indicate a high or low eutrophication level of a water body, as nutrient concentrations are related to other biological and chemical processes as well (Cloern, 2001). In contrast, a reliable signal of increased eutrophication levels is the replacement of late successional, perennial seaweeds, like *Cystoseira* spp. and *Fucus* spp. by opportunistic species like *Ulva* spp. and *Enteromorpha* spp. (Schramm and Nienhuis, 1996, Schramm, 1999). Some other examples of the response of marine phytobenthic communities to anthropogenic stress are shown in 2.2.5.3.1 in Annex.

Most information on the indicator value of macroalgae and seagrasses is reviewed by Borja et al*.* (in press). For **seagrasses**, several indicators are based on the ecology of coastal species (Orfanidis et al*.*, 2006; Romero et al*.*, 2007; Montefalcone, 2009; Lopez y Royo et al*.*, 2009; Schories et al*.*, 2009), but many others have been implemented for transitional and low-salinity coastal waters (Krause-Jensen et al*.*, 2005; Best et al*.*, 2007; Cabaco et al*.*, 2007; Foden and de Jong, 2007; Selig et al*.*, 2007; García et al*.*, 2009). In contrast, most methods for assessing the quality status of a benthic habitat by means of **macroalgae** have been developed for coastal areas, rather than estuarine habitats (Wilkinson and Rendall, 1985; Sagert et al*.*, 2005; Arévalo et al*.*, 2007; Orfanidis, 2007; Juanes et al*.*, 2008). They all include some measurement of species richness (even as presence/absence), abundance (generally as percentage of cover) or biomass (Gibson et al*.*, 2000; Orfanidis et al*.*, 2001, 2003; Borja A, Franco J and Muxika I, 2004; Wilkinson et al*.*, 2007; Scanlan et al*.*, 2007; Krause-Jensen et al*.*, 2007; Selig et al*.*, 2007; Sfriso et al*.*, 2007).

Several methods make use of the macroalgae classification in ecological or functional groups (Orfanidis et al*.*, 2001) or use the presence of opportunistic and sensitive indicator species to detect disturbances in the studied area. Only few indicators are based on **specific metrics**, such as the algae penetration into the estuary (Wilkinson et al*.*, 2007), the depth range (Selig et al., 2007) or the Rhodophyceae/Chlorophyceae ratio (Sfriso et al., 2007). A general description of the main macrophyte indicators is given in Section 2.2.5.3.2 (Annex).

Meiofauna

The marine meiofauna mainly comprises the nematods, the copepods and small oligochaetes. They satisfy the six criteria proposed by Ward and Jacoby (1992) to be selected as biological indicators of pollution, mainly based on the ecology and life history of all three meiofauna groups. However, the use of meiofauna in environmental assessments also poses some cons, related to practical difficulties like the small size and taxonomic complexity of meiofaunal organisms. Many studies tried to find relationships between meiofauna densities and organic enrichment.

Total **copepod** density is highly correlated with sediment grain size, but like other taxa, large differences in copepod density can be observed due to natural variability (Wormald and Stirling, 1979; Sandulli and Nicola-Giudici, 1989, 1990; Moore and Pearson, 1986; Bodin, 1991). Since different copepod genera display distinct life strategies (epibenthic, endobenthic or mesobenthic) all of them should be taken into account to identify the response of copepods to pollution events. However, copepods are one of the most sensitive taxa to oxygen depletion, and the overall copepod density may be used as an indirect indicator of organic enrichment.

Several copepod genera and species appear to be common in polluted areas (e.g. *Tisbe* spp. and larger epibenthic copepods) or reach their highest densities in organic enriched sediments (Bodin, 1964, Marcotte and Coull 1974, Keller 1986, Moore and Pearson 1986, Sandulli and Nicola-Giudici 1990). At least seven copepod species have been proposed as indicators of organic enrichment (Arlt, 1975; Anger and Scheibel, 1976; Govaere et al*.,* 1980; Coull and Wells, 1981; Heip et al*.,* 1984; Moore, 1987).

Nematodes are generally considered to be less sensitive to organic enrichment than other meiofauna taxa (Heip 1980, Moore and Bett 1989). However, certain nematode groups (e.g. *Pontonema* spp.) are typically found in sediments with high organic contents (Bett and Moore, 1988; Lorenzen et al., 1987) or in sediments affected by sewage sludge (Zullini, 1976; Arthington et al*.,* 1986; Khera and Randhawa, 1985; Keller, 1986). Several nematode species (e.g. *Setosabateria* spp.) have been suggested as indicators of organic enrichment (Vanreusel, 1990; Vincx et al*.,* 1990; Lampadariouet al*.,* 1997) as they appeared to be sensitive to hydrocarbon pollution (Danovaro et al*.* 1995b) or to a Cu-increase (e.g. *Monhystera disjuncta*), the latter mainly related to a shorter generation time (Vranken and Heip, 1986; Warwick, 1988; Bayne et al*.,* 1988).
Meiobenthic **oligochaetes** are often found in high densities in low salinity areas where industrial dumping is taking place, and can therefore be considered potential indicators of pollution (Coull and Wells 1981, Hodda and Nicholas 1985). Some oligochaetes (e.g. *Limnodrilus* spp.) can be used as indicators of hydrocarbon pollution in oligohaline areas. The main problem when using these species as indicators is that in some systems oligochaetes are completely absent. Also gastrotrichs (e.g. the genus *Turbanella*) can be utilized as indicators of organic enrichment in polluted beaches.

Some specific meiofauna indicators have been developed. Parker (1975) and Raffaelli and Mason (1981) proposed the use of the **nematodes to copepods ratio (Ne/Co**) in meiobenthic assemblages to monitor marine pollution, with higher values of the Ne/Co ratio in areas subjected to urban sewage. However the ratio is also influence by sediment grain size, which can obscure the pollution pattern. Several adaptations were proposed to strengthen the power of the Ne/Co ratio, taking into account only similar trophic groups for both copepods and nematods (Warwick, 1981), or groups that show similar modalities of substrate colonization (Shiells and Anderson, 1985). Due to several criticisms (Platt et al*.,* 1984; Lambshead, 1986; Danovaro et al*.,* 1995), it might be concluded that the Ne/Co ratio is probably only valuable at a locale scale once pre-pollution conditions are clearly defined.

The **Index of Trophic Diversity (ITD)** is based on the identification of different feeding groups of nematodes (Heip et al*.,* 1984; Danovaro et al*.,* 1995; Mirto et al*.,* 2002). This can be related to a certain pollution level, where the lowest ITD represents only one trophic guild for the whole nematode community. The **Nematode Maturity index (NMI)** refers to the maturity state of a nematode assemblage that colonizes a given ecosystem (Bongers et al. 1991, Bongers and Ferris 1999). All nematode genera may be classified on a scale from 1 to 5, between “colonizers” (r-strategy, fast generation time, rapid colonisation, and high tolerance to disturbance) and “persistents” (characterized by a longer life cycle, K-reproductive strategy, slow colonisation and low tolerance to disturbance). The NMI is computed as the weighted mean of the c-p values for the nematode genera present in a certain area, where a higher value generally correlates with a higher pollution level. In annex 2.2.5.4 , several examples of meiofaunal response to different kinds of pollution are given.

Epibenthos

As already shown earlier, many indicators (eg univariate measures, like number of species) can be applied to different fauna groups. No indicators that were particularly developed on basis of epibenthic fauna are commonly known.

Macrofauna

During the last two decades at least 64 benthic indicators were developed, mainly based on macrofauna data (Diaz, 2004). Twelve macrofauna indicators have been officially accepted by European Member States to monitor the quality of their coastal waters within the Water Framework Directive (Borja A, Muxika I et al., 2009). This certainly reflects the importance of macrofauna in different ecosystem processes (they play a vital role in nutrient cycling, detrital decomposition and as food source for higher trophic levels), but it also reflects the practicality to sample and analyse macrofauna (easy methodology) and the availability of quantitative benthos data.

Macrofauna species are sensitive indicators of changes in the marine environment caused by natural or anthropogenic disturbances, due to their sessile habit and the difficulty to avoid unfavourable conditions (Pearson and Rosenberg, 1978; Dauer, 1993; Reiss and Kröncke, 2005; Kröncke and Reiss, 2010). The main disadvantage is that macrobenthic species may respond in the same way on different stressors, but this is also true for the other ecosystem components.

As macrobenthic species are relatively long-lived, they integrate water and sediment quality conditions with time, which may indicate both temporal and chronic disturbances. Compared to abiotic indicators, such as water or sediment quality, macrofauna response may be a more sensitive and reliable indicator of adverse effects, reflected in the loss of biodiversity and in the dominance of a few tolerant species. In polluted areas the benthic food web can get so simplified to the point where ecosystem processes are irreversibly changed (Karr and Chu, 1997; Lerberg et al., 2000, Pinto et al. 2009).

Comparable to other faunal groups, the life modes of macrobenthic species can be directly related to disturbances in the environment. R-selected (or opportunistic) species, characterized by high fecundity, fast growth and a short life span, are often found in disturbed environments as they are able to efficiently use newly available resources (e.g. food, space). In contrast, K-selected species, characterized by low fecundity, slow growth and a relatively long life span, indicate rather undisturbed habitats. The ratio between both macrobenthic life modes is used in several indicators, such as the **AZTI's Marine Biotic Index (AMBI)** and the **Benthic Opportunistic-Polychaete Amphipod (BOPA) index** (Borja et al., 2000; Gomez-Gesteira and Dauvin, 2000).

Several authors and papers recently reviewed the development of the different macrofauna indicators and the applicability of these indicators to capture or to express the quality of the benthic habitat (e.g. Borja et al., 2004; Dauvin et al., 2007ab, Borja et al., in press). Most of the existing benthic indicators can be applied to marine and estuarine habitats. It is beyond the scope of this study to go into more detail on these indicators. Several tables that were compiled in the ICES WKBEMET workshop (ICES, 2008), have been updated with more recent information and can be found in annex . Section 2.2.5.5.1 gives an up-to-date list of some commonly used macrobenthic indicators with a general description of these indicators. 2.2.5.5.2 describes the applicability of some commonly used macrobenthic indicators. Section 2.2.5.5.3 gives an updated list of some benthic indicators that were developed for a certain habitat and stressor, and their applicability to other stressors and habitats. Next to a number of macrobenthic indicators, this table also includes some information on macrophyte indicators. The table in 2.2.5.5.4 provides a list of publications which have compared the use of different macrobenthic indicators.

Hyperbenthos

Hyperbenthos species are good candidates as sensitive indicators for eutrophication, as the correlation between hyperbenthos and hypoxia is stronger than for infauna (see section 2.1). Amphipods, tanaids and ostracods are the most sensitive groups, and are best suited to provide an early warning of changes in oxygen conditions. The relation between minimum oxygen concentration and species richness can be used to estimate loss of species, based on historical records of changes in the oxygen near the bottom. However, specific hyperbenthic indicators have as yet not been published or found in the scientific literature.

Fish

Fishes and fish communities are being used as indicators of environmental quality changes in estuaries and coastal areas for several decades (Karr, 1981; Moore et al., 1995; Deegan et al., 1997; Gill et al., 2001; Whitfield and Elliott, 2002; Harrison and Whitfield, 2004; Viana et al., 2009; Brind'Amour et al., 2009). Fish-based indicators and methodologies are used or developed to monitor the biological quality element 'fish' in the ecological status assessment of different water bodies for the Water Framework Directive (WFD, 2000; Goethals et al., 2002; Borja et al., 2004; Breine et al., 2007; Coates et al., 2007; Uriarte and Borja, 2009).

The advantages of using fish as indicator organisms in environmental monitoring programs can be summarized as follows (Whitfield et al., 2002):

1) Fish are found in all aquatic systems, except in highly polluted waters
2) For most species, information on life-history and environmental response to changes in the ecosystem is available
3) Fish are easy to identify (compared to many invertebrates) and a non-destructive sampling is possible (fishes can be returned to the water afterwards)
4) Fish communities usually represent a variety of trophic levels
5) Fish are comparatively long-lived and therefore provide a long-term record of environmental stress
6) They contain many life forms and functional guilds and probably cover all the components of the aquatic ecosystem under anthropogenic pressure
7) Fish can present both sedentary and mobile life styles, which can reflect different stressors within a certain area. Fish may also be classified in functional groups to get a broader assessment of the effects
8) Acute toxicity and stress effects can be evaluated in the laboratory for a selected number of species
9) Fish have a high public awareness value, where the general public is more likely to rely on information about the condition of the fish community than on results based on invertebrate or aquatic plant data
10) Societal costs of environmental degradation, including cost-benefit analyses, are more easily compensated, because of the economic, aesthetic and conservation values of fish.

Despite the above described advantages, the applicability of fishes as indicators of biological integrity also poses some difficulties and problems, due to:

- the selectivity of the used sampling gear related to type and size of the habitat and the targeted fish species
- the mobility of fishes on seasonal and dial time scales, which can lead to sampling biass
- the tolerance of fishes to substances that are chemically harmful to other life forms
- the ability of fishes to avoid habitats that are impacted by anthropogenic pressures.

Many fish-based indicators that are used to assess the ecological status of a marine habitat follow a certain typology with three major components. The first component highlights the choice of the relevant metrics associated with a certain level of organization (e.g. fish population or community). The second relies on the method that is used to combine the metrics (as an aggregated indicator or as a synoptic table). The third refers to the type of analysis (direct or indirect) that is used to establish the link between the metric and a given pressure (Brind'Amour et al., 2009). An example of the response of a fish index to different anthropogenic pressures based on oxygen saturation was given by Uriarte and Borja (2009).

A general description of some commonly used fish-based indicators is provided in 2.2.5.6.1. A list of publications, where the use and applicability of different fish-based indicators are compared, is given in section 2.2.5.6.2.The study by Martinho et al. (2008), in which the seasonal variation of five selected multimetric indicators was evaluated, revealed a high level of mismatch between the selected indicators. Large-scale intercalibration processes are urgently needed to eliminate this kind of discrepancies.

2.2.3 GAPS, PROBLEMS AND RECOMMENDATIONS

In the report of the ICES WKBEMET workshop (ICES, 2008), some recommendations for future research on benthic indicators were proposed. The follow-up and some new recommendations are discussed below.

The need for clear definitions

Tallin et al*.* (2007) evaluated a wide range of concepts with reference to the quality of benthic habitats, that are currently used in marine science and in the development of high-level marine policy. Heink and Kowarik (2010) analysed the different definitions and applications of indicators that are used in ecology and environmental planning. Both studies concluded that the terms 'indicators' and 'quality of the benthic habitat' are too ambiguous and that one standard definition hasn't been found yet. There is an urgent need for clear, unequivocal definitions to eliminate the risk of misinterpretations.

The use of multivariate approaches and existing indicators

Several investigators stated that for the assessment of the quality status of the benthic habitat, it is preferable to combine several indicators to cover the complexity of the ecosystem and to minimize interpretation errors (Dauvin et al*.,* 2007b). The use of univariate indicators or indices should be avoided, because the initial environmental information is too drastically reduced. Integrated (e.g. BEQI, B-IBI), multimetric (e.g. IQI, DKI, BQI) or multivariate (e.g. M-AMBI, BAT) indicators (see annexes) combine different structural and/or functional parameters. Such indicators have an increased power to detect human pressures in a more accurate manner.

A plethora of methodologies, with hundreds of benthic indicators and evaluation methods are presently available (Diaz et al., 2004), and the number of methods still increases (Pinto et al., 2009; Borja A, Bricker SB et al., 2009). It must be emphasized that the scientific world should focus on the evaluation of the suitability of existing indicators rather than the development of new indicators (Diaz et al., 2004; Borja and Dauer, 2008).

Background knowledge of the ecological processes

Benthic indicators should be able to retain a meaningful part of the original information, to discard noise, and to synthesize the inherent multivariate data upon which they are constructed into a single figure that can be interpreted in a simple way (ICES, 2008). However, in many cases a thorough explanation of the ecological processes, which are at the basis of the indicators is lacking. As such, these indicators may be misused and misleading when they are uncritically linked to the outcome of an analysis.

Some recently developed benthic indicators do provide an ecological basis in peer-reviewed papers, e.g. B-BI (Weisberg et al*.,* 1997), AMBI (Borja et al*.,* 2000); M-AMBI (Borja et al*.,* 2004; Muxika et al*.,* 2007); BQI (Rosenberg et al*.,* 2004). It is recommended that new and existing methods that have not yet done so, should provide detailed scientific information on the ecological processes behind the indicators. This will improve the power of expert judgements - as a benchmark of the benthic status assessment - in the interpretation of the quality status of a habitat (Weisberg et al*.,* 2008; Teixeira et al*.,* 2010).

Applicability and validation of different indicators

Most benthic indicators have initially been developed on geographically delimited datasets, and their application is limited to the local areas for which they were designed (e.g. B-BI in Chesapeake Bay). Few indicators have been tested and compared at broader scales. Grémare et al*.* (2009) compared the AMBI and BQI (based on macrofauna data) at a pan-European scale, and found clear discrepancies between both non-related indicators. In other studies, only one single pressure is investigated. Borja A, Rodriguez K et al*,* (2009) tested the applicability of AMBI and ITI to describe the ecological status of different European habitats where aquacultural activities are taking place. One of the most applied benthic indicators is the AMBI, which has been extensively used to evaluate the quality of benthic habitats under different human pressures throughout Europe, North and South America, North Africa and Asia.

Many publications on benthic indicators lack detailed information on the pressure intensity, nor do they provide information on spatial and/or temporal gradients, which hampers the interpretation and comparison of these indicators when used in other habitats (Josefson et al*.,* 2008, 2009; Borja et al*.,* 2009c; Callier et al*.,* 2009). Moreover, the interaction between pressure intensity and oceanographical factors (e.g. current velocity) is often not adequately studied. Some large scale patterns or changes may obscure the response of the indicator, leading to erroneous interpretations of the indicator results (Borja A and Rodriguez K et al*.,* 2009).

For many indicators, the applicability to other habitats is limited due to the lack of coherence in reference conditions for the different countries (Teixeira et al*.,* 2008). For the Water Framework Directive (WFD), large scale intercalibration excersises are ongoing to consolidate and harmonize the appropriate reference conditions between the different European countries for the selected benthic indicators (Borja et al*.,* 2007; Borja A and Miles A et al.,2009). Therefore, large datasets with well known gradients, pressures and reference conditions are needed.

Adequate design of monitoring programs

Most indicators that are used in ecological quality assessments, are likely to respond in the same way to both man-induced and natural disturbances. Not only the ecological processes behind the indicators should be known, but also their response to natural variation in the environment needs to be monitored on different temporal scales (Reiss and Kröncke, 2005; Zettler et al*.,* 2007; Dauvin and Ruellet, 2007a; Muxika et al*.,* 2007; Kröncke and Reiss, 2010).

Adequate sampling designs are needed to detect changes in the benthic status. Most European countries already implemented a monitoring network for the WFD. These monitoring data can be used to establish reference conditions in case these are not yet defined, but they will also help to detect sources of natural variability, e.g. due to salinity or seasonality (Reiss and Kröncke, 2005; Zettler et al*.,* 2007; Fleischer and Zettler, 2009). It should be clear, that the collection of both environmental and pressure data must be incorporated in the monitoring programs.

Many indicators to assess the quality of the benthic habitat are based on subtidal macrobenthic community data and are mainly applied to open coastal systems. In most cases it is not known how these indicators respond when they are applied to other systems or habitats.For some habitat types, like large intertidal flats, mobile sands, sandy beaches and hard bottom substrata, benthic indicators have not been sufficiently tested, or even developed.

Recently, some indicators have been tested in lagoons (Pranovi et al*.,* 2007; Afli et al*.,* 2008; Munari et al*.,* 2009; Prato et al*.,* 2009) and transitional waters (Borja et al*.,* 2006; Chainho et al*.,* 2008; Dauvin et al*.,* 2009; Ranasinghe et al*.,* 2009; Teixeira et al*.,* 2009). In a second phase of the WFD intercalibration, a large scale comparison for transitional waters will be possible, based on macrofauna (and environmental) data on >7,000 sampling locations from several northeast Atlantic estuaries.

Some systems, like oligohaline and freshwater tidal areas, are not sufficiently investigated, which makes it difficult to interpret the response of the indicators to the natural and human pressures (and eventually the interaction between both) in those areas (Elliott and Quintino, 2007; Dauvin, 2007; Dauvin and Ruellet, 2009). Again, adequate monitoring of such systems is urgently needed to allow for an acceptable quality assessment of the benthic habitat based on existing or newly developed indicators.

Compare the response of different biota

Few studies have compared the response of different indicators (mainly based on diversity) to changes in the environment for different fauna groups. It is known that different biota respond differently to gradients in environmental heterogeneity or to pollution (e.g. hyperbenthos density is a better indicator for eutrophication than infauna density). However, depending on the used diversity indices, different responses may be revealed, which makes it difficult to interprete the results. Diversity indicators should be used with caution when comparing different biota groups, e.g. by changing from finer to coarser sediments, a shift from poor epifauna and rich infauna to the opposite is observed.

The monitoring design should take into account an appropriate sampling of all ecosystem components related to the benthic habitat, using the best suited sampling techniques for each of the fauna (and flora) groups.

Special attention must be given to assess the impact of invasive species (Olenin et al., 2007). Alien species are by definition a threat to biodiversity, but most of the existing indicators (e.g. the trends indicator proposed by EEA/SEBI2010, 2007) don't integrate variables on ecosystem functionality, and thus are not able to detect alterations to the functionality of the habitat.

Some final recommendations

Borja A and Ranasinghe SBet al. (2009) identified four areas on which the scientific world should agree to satisfy future management needs:

1. to establish a clear format for a good indicator in order to reduce the bewildering array of available indicators, by identifying those indicator approaches, components and formulations that are most widely successful (Indicator format)
2. to establish minimum criteria for indicator validation processes, which can demonstrate the accuracy and reliability of the indicator (Indicator validation)
3. to compare and intercalibrate methods to achieve uniform assessment scales across geographical boundaries and habitats (Index intercalibration)
4. to integrate indicators across different ecosystem components (Index integration).

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Ysebaert T, Herman PMJ (2004) The assessment of the ecological status of coastal and transitional waters based on benthic macroinvertebrates: classification and intercalibration within the Water Framework Directive. NIOO-CEME report 2004-01

Ysebaert T, Hermana PMJ, Meirec P, Craeymeerschd J, Verbeeke H, Heipa CHR (2003) Large-scale spatial patterns in estuaries: estuarine macrobenthic communities in the Schelde estuary, NW Europe Est Coast Shelf Sci 57 (1-2): 335-355

Zettler ML, Schiedek D, Bobertz B (2007) Benthic biodiversity indices versus salinity gradient in the southern Baltic Sea. Mar Pol Bull 55: 258-270

Zullini A (1976) Nematodes as indicators of river pollution. Nematol. Med. 1: 13-22

2.2.5 ANNEXES

2.2.5.1 Symposia, workshops, meetings, and projects related to benthic indicators

For an overview of meetings prior to 2006, see the report of the ICES WKBEMET workshop (ICES, 2008).

Symposia

22-27 May, 2007 Ecological complexity and sustainability: challenges & opportunities for 21st century's ecology (ECOSUMMIT) organised by Elsevier, Beijing (China)

Chair & organiser of symposium nr 16: A. Borja (Spain) & J.C. Marques (Portugal)

Symposium nr 16: Integrative tools and methods in assessing ecological integrity in estuarine and coastal systems.

Outreach:

Borja, A., S.B. Bricker, D.M. Dauer, N.T. Demetriades, J.G. Ferreira, A.T. Forbes, P. Hutchings, X. Jia, R. Kenchington, J.C. Marques, C. Zhu, 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Mar Pol Bull. 56: 1519-1537.

Borja, A., S.B. Bricker, D.M. Dauer, N.T. Demetriades, J.G. Ferreira, A.T. Forbes, P. Hutchings, X. Jia, R. Kenchington, J.C. Marques, C. Zhu, 2009. Ecological integrity assessment, ecosystem-based approach, and integrative methodologies: are these concepts equivalent? Mar Pol Bull, 58: 457-458.

4-8 November, 2007 Coastal and Estuarine Research Federation (CERF) conference, Providence (USA)

Chair & organiser: A. Borja (Spain) & J. Ananda Ranasinghe (USA)

Session: Assessing Ecological Integrity: Using Multiple Indicators and Ecosystem Components.

Outreach: a special issue of Mar Pol Bull (vol. 59 (1-3)), March 2009

20-23 November, 2007 Environmental Indicators: Utility in Meeting Regulatory Needs, London UK

Chair & Organisers: HL Rees (UK), E Jagtman (The Netherlands) & H Hillewaert (Belgium)

Outreach: A special issue of the ICES J Mar Sci: ICES (2008) Marine environmental indicators: Utility in meeting regulatory needs. Ices J Mar Sci (vol 65 (8), November 2008)

1-5 November, 2009 Coastal and Estuarine Research Federation (CERF) conference, Portland USA

Chair & organiser of session SCI-076: A. Borja (Spain), D. Dauer (USA) & J. A. Ranasinghe (USA)

Session: "Assessing Ecological Integrity Using Multiple Indicators and Ecosystem Components: The sequel"

14-19 September 2010 47th ECSA Symposium: Integrative tools and methods in assessing ecological quality in estuarine and coastal systems worldwide, Figueira da Foz (Portugal)

Organisers: J.C. Marques (Portugal) & Z. Costa (Portugal)

http://www1.ci.uc.pt/imar/ecsa47/

Legislations are under development worldwide to address the ecological quality or integrity of estuarine and coastal systems (e.g. Oceans Act in USA, Australia or Canada; Water Framework Directive or Marine Strategy in Europe; National Water Act in South Africa). In spite of the increasing attention that has been paid to the development of tools for different physical, chemical or biological elements of the ecosystems, very few methodologies integrate all the elements into a unique evaluation of a water body. The development of such integrative tools to assess ecosystem quality is therefore of major importance, both from a scientific and stakeholder points of view, and must take into account the multidisciplinarity of the problems involved, the need for integration of biotic and abiotic factors, methods for intercalibration and validation, and adequate indicators to follow the evolution of the monitored ecosystems.

Outreach: The ECSA meeting will provide an overview of the current situation through examples from each continent.

20-24 September 2010 ICES Annual Science Conference (ASC), Nantes (France)

Chair of the session on benthic indicators: A. Borja (Spain), D. Dauer (USA) and A. Grémare (France)

Session: Benthic indicators: responding to different human pressures and assessing integrative quality status.

http://www.ices.dk/iceswork/asc/2010/index.asp

Outreach: A special issue of Ecol Indic

Meetings

4-9 June, 2006: ASLO summer meeting Global Challenges Facing Oceanography and Limnology, Victoria British Columbia (Canada)

Chair & organiser of the special session TS-D06: A. Borja (Spain) & D. Dauer (USA)

Session: Assessing the environmental quality status in estuarine and coastal systems: comparing methodologies and indicators.

Outreach: A special issue of Ecol Indic (vol 8 (4): 2008)

25-30 January, 2009 Advancing the science of Limnology & Oceanography (ASLO) meeting, Nice (France)

Chair & organiser of session 4: A. Borja (Spain), D. Dauer (USA), C. Simenstad (USA) & M. Elliott (UK)

Session: "Medium and long-term recovery of marine and estuarine systems- a guide to providing useful information in new scenarios to restore ecological integrity"

Outreach: A special issue of Estuaries and Coasts (to be published in 2010)

7-14 April, 2010 ICES Working Group on Ecosystem Effects of Fishing Activities (WGECO) meeting, Copenhagen (Denmark)

Chair: Ellen Kenchington (Canada)

ToR: To assess the development of integrated ecosystem assessments, in particular focusing on how assessments will be used for the MSFD and considering the use of the IOC's best practice recommendations, including a gap analysis in terms of the availability of suitable state and pressure indicators.

ToR: To review methods used to determine "good environmental status" under the WFD, HD and MSFD, including a discussion of reference points and indicators

Outreach: ICES 2010. Report of the Working Group on Ecosytem Effects of Fishing Activities (WGECO), 7-14 April 2010, Copenhagen, Denmark ICES CM 2010/ACOM: 23.73pp.

19-23 April 2010 ICES Benthic Ecology Working Group (BEWG) meeting, Edgewater (USA) and Oostende (Belgium)

Chair: Steven Degraer (Belgium)

ToR: To consider the status of the BEWG viewpoint paper on benthic indicators and evaluate ongoing developments on ecological quality assessments.

Outreach:

Van Hoey, G; Borja, E; Degraer, S; Fleischer, D; Magni, P; Muxika, I; Reis, H; Schröder, A; Zettler, M; Buhl-Mortensen, L; Birchenough, S (in prep) The use of benthic indicators in Europe: from the WFD to the MSFD. Mar Pol Bull.

ICES 2010 Report of the Benthic Ecology working group (in prep)

2009-2011: Intercalibration meetings of the Water Framework Directive methodologies within the North East Atlantic

The EU Water Framework Directive (WFD) commits the EU member states to achieve good ecological status of all European surface waters. The member states have developed biological assessment systems by which the water bodies are classified. In the intercalibration exercise these national methods are harmonized to ensure that good status is consistent with the directive's requirements and comparable among countries.

Outreach:

Borja, A., A.B. Josefson, A. Miles, I. Muxika, F. Olsgard, G. Phillips, J.G. Rodríguez, B. Rygg (2007). An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive. Mar Pol Bull, 55: 42-52.

Borja, A., A. Miles, A. Occhipinti-Ambrogi, T. Berg (2009). Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive. Hydrobiologia, 633(1): 181-196.

European Commission (2008). Member State monitoring system classifications as a result of the intercalibration exercise in relation to EC Directive 2000/60/EC (document number C(2008) 6016) (2008/915/EC). Official Journal of the European Union, L332: 20-44.

Workshops

11-14 february 2008 Workshop on Benthos Related Environment Metrics (WKBEMET) Oostende(Belgium)

Outreach:

ICES 2008 Report of the Workshop on Benthos Related Environment Metrics (WKBEMET), 11-14 February 2008, Oostende, Belgium. ICES CM 2008/MHC:01.53 pp

29 April, 2008 Workshop "The implementation of the Water Framework Directive (CE2000/60) in Italy: State of the art on benthic indicators and European experiences", Ferrera (Italy)

Organised by the Benthic Committtee of the "Societa Italiana de Biologia Marina"

Outreach: a special issue of Mar Ecol (vol 30 (2), May 2009

European projects

INDECO: Development of Indicators of Environmental Performance of the Common Fisheries Policy (2004-2006).

Sixth Framework Programme Coordinated Action, Priority 8.1 - Policy Oriented Research.

http://www.ieep.eu/projectminisites/indeco/index.php

Outreach: http://www.ieep.eu/publications/publications.php

ECASA: Ecosystem Approach for Sustainable Aquaculture (2004-2007)

Sixth Framework Programme, Priority Integrating and Strengthening the European Research Area, Specific Targeted Research or Innovation Project

http://www.ecasa.org.uk/

Outreach:

The ECASA Toolbox is an internet based resource, containing tested tools for marine aquaculture environmental impact assessments. These tools will be a range of indicators, models and procedures, tailored for the different culture methods and species grown across Europe. http://www.ecasatoolbox.org.uk

Borja, A., J.G. Rodríguez, K. Black, A. Bodoy, C. Emblow, T.F. Fernandes, J. Forte, I. Karakassis, I. Muxika, T.D. Nickell, N. Papageorgiou, F. Pranovi, K. Sevastou, P. Tomassetti, D. Angel (2009). Assessing the suitability of a range of benthic indicators in the evaluation of environmental impact of fin and shellfish aquaculture located in sites across Europe. Aquaculture, 293: 231-240.

Nickell, T.D., C. J. Cromey, Á. Borja, K.D. Black (2009). The benthic impacts of a large cod farm – are there indicators for sustainability? Aquaculture, 295(3-4): 226-237.

MEECE: Marine Ecosystem Evolution in a Changing Environment (2008 -2012)

Seventh Framework Programme

http://www.meece.eu/

MEECE is an integrated project that will use predictive models that consider the full range of drivers to explore the responses of the marine ecosystem in a holistic manner, rather than driver by driver as has been done in the past. MEECE will explore the impacts of both climate drivers (acidification, light, circulation and temperature) and anthropogenic drivers (fishing, pollution, invasive species and eutrophication). One of their tasks is to develop indicators of ecosystem status.

IndiSeas is co-funded by MEECE and it aims to evaluate the effects of fisheries on marine ecosystems by using a panel of Ecol Indic of states and trends, and to facilitate effective communication of these effects.

Outreach:

Shin, Y-J., and Shannon, L. J. 2010. Using indicators for evaluating, comparing and communicating the ecological status of exploited marine ecosystems. 1. The IndiSeas project. - ICES J Mar Sci, 67: 686-691.

Shin, Y-J., Bundy, A., Shannon, L. J., Simier, M., Coll, M., Fulton, E. A., Link, J. S., Jouffre, D., Ojaveer, H., Mackinson, S., Heymans, J. J., and Raid, T. 2010. Can simple be useful and reliable? Using Ecol Indic to represent and compare the states of marine ecosystems. - ICES J Mar Sci, 67: 717-731.

WISER: Integrative systems to assess Ecological status and Recovery (2009-2012)
http://www.wiser.eu/
WISER is a European project that will support the implementation of the Water Framework Directive by developing assessment schemes and by modeling the effects of restoration and climate change on theassessment results. WISER will address biological recovery processes after cessation of various anthropogenic pressures. Therefore, large-scale data will help to identify correlations between pressure variables and ecosystem responses.

Outreach:

WISER (2010) Guidelines for indicator development. Delivereble 2.2 http://www.wiser.eu/download/D2.2-2.pdf

Benthic indicator database: http://www.wiser.eu/results/methods-db/

Various other deliverables

2.2.5.2 Overview of indices and indicators, taken from ICES (2008)

In the ICES WKBEMET report (ICES, 2008) the different indicators were classified as univariate, functional and multimetric indices next to some multivariate and modeling approaches. Mainly the first three groups within the univariate indices (descriptors, indices of diversity and graphical methods) are commonly used for all ecosystem components. The others are specifically developed as macrofauna indicators. Many more indicators have been developed based on macrobenthos data. See the following annexes for updated lists on macrobenthic indicators.
Please consult the WKBEMET report (ICES, 2008) for the complete references in this table.

UNIVARIATE INDICES	
Descriptors	
Abundance	
Number of species	
Biomass	
Frequency of occurrence	
Presence/Absence	
….	
Indices of diversity	
Shannon's indices of diversity and Evenness	Shannon and Weaver, 1949
Simpson's indices of dominance, diversity and evenness	Simpson, 1949
Brillouin indices of diversity and evenness	Brillouin, 1956
Pielou's indices of dominance and evenness	Pielou,1966
Margalef's index	Margalef, 1968
Hurlbert Index	Hurlbert, 1971
Hill's diversity numbers and evennes measures	Hill, 1973
Benthic Pollution Index (BPI)	Leppäkoski, 1975
Taxonomic diversity index and Taxonomic distinctness	Warwick and Clarke, 1995
Graphical methods	
K-dominance curves	Lambshead et al, 1983
RFD (Rank Frequency Distribution)	Frontier, 1985
ABC curves	Warwick and Clarke, 1994
Ecological groups	
Annelid Index of Pollution	Bellan, 1980

Biotic Index (BI)	Majeed, 1987; Grall and Glémarec, 1997; Hily, 1984; Hily et al, 1986
Benthic opportunistic polychaetes amphipods index (BOPA) and BO2A	Gomez Gesteira and Dauvin, 2000; Dauvin and Ruellet, 2007
AZTI Marine Biotic Index (AMBI)	Borja et al, 2000; Borja et al, 2003; Broja et al, 2004
Bentix	Simboura and Zenetos, 2002
Indicator Species Index	Rygg, 2002
Coastal Endofaunic Evaluation Index (I2EC)	Grall and Glémarec, 2003
FUNCTIONAL INDICES	
Infauna Trophic Index (ITI)	Word, 1979; 1980; Maurer et al., 1999
Ecological Evaluation Index (EEI)	Orfanidis et al., 2001
MULTIMETRIC INDICES	
Pollution Coefficient	Satsmadjis, 1982; 1985
Biological Quality Index (BQI)	Jeffrey et al., Wilson and Jeffrey, 1987; Wilson and Elkaim, 1991; Wilson and Jeffrey, 1994
Infauna Ratio-to-Reference of Sediment Quality Triad (RTR)	Chapman et al., 1987
Benthic Index of Estuarine Condition	Weisberg et al., 1993; Schimmel et al., 1994; Strobel et al., 1995
Benthic condition Index (BCI)	Engle et al., 1994; Engle and Summers, 1999; Paul et al, 2001
Benthic Index of biotic integrity (B-IBI)	Ranasinghe et al., 1994; Weisberg and Ranasinghe, 1997; Van Dolah et al., 1999; Llanso et al., 2002a; Llanso et al., 2002b
Benthic Habitat Quality (BHQ)	Nilsson and Rosenberg, 2000
Ecofunctional Quality Index (EQI)	Fano et al., 2002
Infaunal Quality Index (IQI)	Prior et al., 2004; Borja et al., 2007; Miles et al., in prep
Dankse Kvalitet Indeks (DKI)	Borja et al., 2007
Norwegian Quality Index (NQI)	Rygg, 2002; 2006; Borja et al., 2007
Marine Biotic Index Tool (MarBIT)	Meyer et al., 2006
Benthic Quality Index (BQI)	Rosenberg et al., 2007
Benthic Ecosystem Quality Index (BEQI)	Van Hoey et al., 2007
Brackish water Benthix Index (BBI)	Perus et al., 2007
DAPHNE	Forni and Occipinti Ambrogi, 2007
MULTIVARIATE AND MODELLING APPROACHES*	
Benthic Response Index	Smith et al., 2001
Principal Response Curves (PRC)	Pardal et al., 2004
M-AMBI	Borja et al., 2004; Muxika et al., 2007
P-BAT	
* Several software packages can be usted for multivariate and modelling approaches e.g. PRIMER, Canoco, STATISTICA…	

2.2.5.3 Macrophytes

2.2.5.3.1Short description of some commonly used macrophytic indicators

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference
BiPo	Biotic Index using Posidonia		Metrics describing the community and its trophic structure.				Lopez y Royo et al, 2008
EEI	Ecological Evaluation Index	Multivariate approach	The EEI is an original biotic index based on the concept of morphological and functional groups [1, 2]. The species found in various samples are divided into different Ecological State Groups (ESG). In the ESG I the thick leathery, the articulate upright calcareous and k-selected species are grouped. In the ESG II the foliose, the filamentous and the coarsely branched upright species are grouped. Most of them are r-selected species				Orfanidis S, Panayotidis P and Stamatis N. (2001) & 2003
PREI	Posidonia oceanica Rapid Easy Index	Univariate index	The PREI is based on five metrics: shoot density, shoot leaf surface area, E/L ratio (epiphytic biomass/leaf biomass), depth of lower limit, and type of this lower limit.				Gobert et al, 2009

2.2.5.3.2 Anthropogenic impacts on marine benthic macrophytic communities

Anthropogenic stress	Benthic macrophytes	Impact	Literature
Eutrophication	Seaweeds	Dominance of opportunistic species, seaweed blooms, decline of diversity	Chryssovergis & Panayiotidis (1995), Lazaridou et al. (1997), Schramm & Nienhuis (1996), Schramm (1999), Lotze et al. (1999), Lotze & Schramm (2000), Orfanidis et al (Smith and Montagne), Panayotidis et al. (2004)
	Seagrasses	Large scale and regional meadows declines, dominance of seaweed epiphytes	Larkum et al. (1989), Hemminga & Duarte (2000), Bortone (2000)
Organic matter, Siltation	Seaweeds	Light reduction and alteration of hard substrate affects community structure	Lobban & Harrison (1994)
	Seagrasses	Decline of meadows through reduction of light and accumulation of organic matter in sediment	Hemminga & Duarte (2000)
Heavy metals	Seaweeds	Inhibition of reproduction and development, changes in community structure	Lobban & Harrison (1994), Coelho et al.(2000), Crowe et al. (2000)
	Seagrasses	Not a direct effect has been observed	Larkum et al. (1989)
Oil spills	Seaweeds	Short term growth reduction in intertidal species	Lobban & Harrison (1994)
	Seagrasses	Not a direct effect has been documented	
Global warming	Seaweeds	Changes in distribution patterns are expected	Breeman (1990), Pakker & Breeman (1994)
	Seagrasses	Changes in distribution patterns are expected	Hemminga & Duarte (2000)
Increase of water salinity	Seaweeds	Further expansion in estuarine ecosystems	Lobban & Harrison (1994)
	Seagrasses	Species displacement, e.g. Cymodocea instead of Ruppia	Kamermans et al. (1999)
Trawling Fishing	Seaweeds	Damages of sublittoral stands	Blader et al. (2000)
	Seagrasses	Fragmentation - decline of meadows	Jerèz & Esplà (1996), Blader et al. (2000)

2.2.5.4 Meiofauna

Information on the calculation of some meiofauna indicators

Trophic diversity

The trophic diversity of meiofaunal taxa is calculated as follows:

$$ITD = \sum_{i=1}^{n} \theta^2$$

where θ is the contribution of each trophic group to total nematode density, and n is the number of trophic groups. ITD ranges from a lower limit determined by the number of trophic groups (1/n, the highest diversity) to 1 (the lowest diversity; i.e., one trophic guild account for all nematode density).

Nematode maturity index

The nematode maturity index is calculated as follows,

$MI=\Sigma\nu(i)f(i)$

where ν (i) is the value of c-p of each taxon and f(i) is the frequency of each taxon in the sample.

C-P values associated to nematodes taxa (from Bongers et al. 1991)

Actinonema	4	Desmoscolex	4	Quadricoma	4
Adoncholaimus	4	Dichromadora	2	Rhahdocoma	4
Aegialalaimidae	4	Diplolaimella	1	Rhabditidae	1
Aegialoalaimus	4	Diplopeltidae	3	Rhabdodemania	4
Amphimonhystera	2	Dolicholaimus	2	Rhabdodemaniidae	4
Anoplostoma	2	Dracognomus	4	Rhips	3
Anoplostomatidae	2	Draconematidae	4	Rhynchonema	3
Anticoma	2	Echinotheristus	2	Richtersia	3
Anticomidae	2	Eleutherolaimus	2	Sabatieria	2
Antomicron	3	Enchelidiidae	4	Selachinematidae	3
Araeloalaimus	3	Enoplidae	5	Sigmophoranema	3
Ascolaimus	2	Enoploides	2	Siphonolaimidae	3
Atrochromadora	4	Enoplolaimus	2	Siphonolaimus	3
Axonolaimidae	2	Enoplus	5	Southerniella	3
Axonolaimus	2	Epsilonema	4	Sphaerolaimidae	3
Bathylaimus	2	Epsilonematidae	4	Sphaerolaimus	3
Bolbolaimus	3	Ethmolaimidae	3	Spilophorella	2
Calomicrolaimus	2	Euchromadora	3	Spirinia	3
Calyptronema	4	Eurystomina	4	Stephanolaimus	4
Camacolaimus	3	Gammanema	3	Stylotheristus	2
Ceramonematidae	3	Gonionchus	4	Symplocostoma	4
Chaetonema	2	Greeffiella	4	Synonchus	5
Choniolaimus	3	Halalaimus	4	Syringolaimus	4
Chromadora	3	Halaphanolaimus	3	Terschellingia	3
Chromadorella	3	Halichoanolaimus	3	Thalassoalaimus	4
Chromadoridae	3	Haliplectidae	3	Theristus	2
Chromadorina	3	Haliplectus	3	Thoracostoma	5
Chromadorita	3	Hypodontolaimus	4	Thoracostomopsidae	2
Chromaspirina	4	Ironidae	4	Trefusia	4
Cobbia	3	Ixonema	4	Trefusiidae	4
Comesomatidae	2	Laimella	2	Trichotheristus	2
Crenopharynx	4	Latronema	3	Tricoma	4
Cyartonema	3	Leptolaimidae	3	Trileptium	2

Cyatholaimidae	3	Leptolaimus	2	Tripyloides	2
Cyatholaimus	3	Leptonemella	4	Tripyloididae	2
Cylicolaimus	5	Leptosomatidae	5	Tubolaimoides	3
Dagda	3	Leptosomatum	5	Tubolaimoididae	3
Daptonema	2	Linhomoeidae	2	Valvaelaimus	2
Dasynomoidos	3	Linhomoens	2	Viscosia	3
Doontolaimus	3	Mesacanthion	3	Wieseria	4
Desmodora	2	Metachromadora	2	Xyala	3
Desmodoridae	3	Metadesmolaimus	2	Xyalidae	2
Desmolaimus	2	Metalinhomoeus	2		
Desmoscolecidae	4	Metaparoncholaimus	4		

Examples of meiofaunal response to different kinds of pollution

Meiofaunal density generally decreases due to **organic enrichment** (Olsson et al. 1973, Anger and Scheibel 1976, Aissa and Vitiello 1984, Ansari et al. 1984, Keller 1984, 1985, Varshney 1985, Vitiello and Aissa 1985, Moore 1987, Sandulli and Nicola-Guidici 1990, Mazzola et al. 1999, 2000, Mirto et al. 2000, 2002). Only in a few cases an increase in meiofauna density was noted (Gowing and Hulings 1976, Nichols 1977, Van Es et al. 1980, Raffaelli 1982, Hennig et al. 1983, Vidakovic 1983, Bouwman et al. 1984, Khera and Randhawa 1985, Arthington et al. 1986, Moore and Pearson 1986, Lorenzen et al. 1987). All studies agreed that meiofaunal diversity was lower in polluted sites. On the other hand, nematode densities generally increase as they are able to exploit the food released from sewage discharge (Vidakovic 1983, Arthington et al. 1986, Bongers et al. 1991). *Especially Sabatieria* is an indicator of organic enrichment as it dominates in sediments with poor oxygen content (Vanreusel 1990, Vincx et al. 1990, Lampadariou et al. 1997).

Several nematode genera (e.g. *Daptonema* and *Viscosia)* seem to be insensitive to **oil (hydrocarbon) pollution** (Boucher 1980, Elmgren et al. 1980, Bonsdorf 1981, Frìcke et al. 1981, Gee et al. 1992), although some authors reported a decrease in nematode abundance with increased oil content in the sediment (Wormald 1976, Giere 1979, Danovaro et al. 1995, 1999). Some genera (e.g. *Chromaspirina*, *Hypodontolaimus*, *Onchalaimellus*) are highly resilient and disappeared immediately after an oil spill, but recovered soon. Danovaro et al. (1995) could not find significant changes in the Index of Trophic Diversity(ITD) in relation to an oil spill. Mahmoudi et al. (2005) proved that nematode response to hydrocarbon spills varied by species, be it intolerant, sensitive, opportunistic or resistant to the pressure, while for some species, the effect only appeared after a time lag (Renaud-Mornant et al. 1981). Copepods appeared to be very sensitive to the products that are utilized to treat oil spills (Bleakley and Boaden 1974, McLachlan and Harty 1982, Cross and Martin 1987). Several surfactants are proven to be extremely toxic to oligochaetes (Giere 1980), while dispersants can reduce the growth rate of archiannelids (Gray and Ventilla 1971).

The overall nematode diversity decreased after exposure to **heavy metal contamination** (Heip et al. 1985, Somerfield et al. 1994, Millward and Grant 1995). But also the diversity within different nematode genera decreased due to contamination with several heavy metals (Gyedu-Ababio et al*.,* 1999), except for Arsenic (Guo et al. 2001).

Nematode abundance decreased in response to **fish farm pollution** (Duplisea and Hargrave 1996, Mirto et al. 2002). Nematode biomass increased immediately after the installation of a new fish farm, but reverted to normal levels soon after farming ceased. Specific genera characterized the nematode assemblages in farm sediments, while other genera disappeared almost completely in farm sediments (Mirto et al. 2002). The Maturity Index (MI) of the nematode assemblage from fish-farm sediments dropped in areas with high organic matter deposition. The Index of Trophic Diversity (ITD) increased in farm sediments due to the increased abundance of non-selective deposit-feeding nematodes (Mirto et al. 2002).

Meiofaunal taxa are sensitive to **physical disturbance**, more specific to the disposal of dredged material (Somerfield et al. 1995). Some species (e.g. the *S. pulchra* group) might have an indicator value, as they are found in undisturbed conditions but also persist as dominants of an impoverished meiofaunal community (Tietjen, 1980; Somerfield et al*.,* 1995; Boyd et al*.,* 2000). Generalist nematodes such as *Sabatieria* spp. and *D. tenuispiculum* can evoke a rapid colonization of a disturbed area (Heip et al*.* 1984, Lambshead 1986, Somerfield et al*.* 1995). Physical disturbance by mechanical beach cleaning activities alters the structure of the whole nematode assemblages (Gheskiere et al*.* 2005a, b). Similar differences in sandy sediment nematode assemblages from continuously disturbed and unperturbed situations were also detected by Schratzberger and Warwick (1999).

2.2.5.5 Macrofauna

2.2.5.5.1 Updated table (from ICES, 2008) on some commonly used macrobenthic indicators

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
AMBI	Azti Marine Biotic Index	Index based on indicator species	Index is based on the model from Pearson and Rosenberg (1978) and adapted from Grall and Glémarec (1997) and uses the relative abundance of 5 ecological groups		Classification of sensitive/tolerance species Abundance of species	soft bottom habitats within European estuarine and coastal environments	Borja et al. 2000	in use worldwide (references in Europe, Africa, Asia, America)
APBI	Ascidian province benthic index					Gulf of Maine	Hale and Heltshe 2008	
BBI	Brackish Water Benthic Index		BQI, Shannon's diversity, Density, Richness			for Baltic Sea	Perus et al. 2007	in use MS Finnland
BCI	Benthic condition index	Multimetric index	The benthic index includes: (1) Shannon–Wiener diversity index adjusted to salinity; (2) mean abundance for Tubificidae; (3) percentages of abundance of the class bivalvia; (4) percentages of abundance of the family Capitellidae; and (5) percentages of abundance of the order amphipoda.	refinement of a previous attempt (Engle et al. 1994)		Gulf of Mexico	Engle and Summers 1999	
BENTIX		Index based on indicator species	BENTIX is based on the proportion of sensitive/tolerant species and relies on the reduction of macrozoobenthos data in relative abundance of 3 wider ecological groups.	AMBI	Classification of sensitive/tolerance species Abundance of species	designed to fit the Mediterranean benthic ecosystem	Simboura & Zenetos 2002, Simboura and Reizopoulou 2007	
BEQI	Benthic Ecosystem Quality Index	Index based on the diversity value and structure of the community (including network analyses)	A multimetric method distinguishing three scale levels (ecosystem, habitat, community) to asses overall ecosystem functioning. The BEQI multimetric integrates the information of the three levels and primarily aims at providing a signal that is capable of showing significant deviations from	Originating from the NIOO-method (Ysebaert & Herman, 2004).	Biomass Abundance data	Dutch marine and estuarine habitats	Ysebaert & Herman, 2004 Van Hoey et al. 2007	

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
			a defined reference state.					
BHQ	Benthic Habitat Quality	Based on the analysis of sediment profile images.	The BHQ is based on a Sediment Profile Imaging infaunal successional model; using a quantitative determination of the relative densities of surface and subsurface organisms. the results of that index are closely tied to traditional biological and geological sampling.	Based on the work by Rhoads and Germano (OSI)	Sediment profile images		Rhoads and Germano, 1982 Rhoads and Germano, 1986 Nilsson and Rosenberg, 1997	
B-IBI	Benthic Index for Biotic Integrity	Integrative index: integrating environmental information or more than one of the previous categories.	Developed to assess benthic community health and environmental quality in Chesapeake Bay. The B-IBI evaluates the ecological condition of a sample by comparing values of key benthic community attributes (metrics) to reference values expected under non-degraded conditions in similar habitat types. It is therefore a measure of deviation from reference conditions.	IBI transformed in B-IBI (1997) by Weisberg et al. (1997)	(1) Species diversity (H', S, Dominance) (2) Productivity (abundance and biomass) (3) Specific composition (% of abundance or biomass of sensitive taxa or pollution indicator species) (4) Trophic composition (% of abundance of carnivorous, omnivorous, deposit feeders, filter feeders) (5) Distribution of species under the water/sediment interface (% of taxa as a function of the depth).	Chesapeake Bay, USA	Weisberg et al. 1997, Alden et al. 2002, Llanso et al. 2003	
BITS	Benthic Index based on Taxonomic Sufficiency		Index based on family classification level.			coastal lagoons in the Italian Sea	Mistri & Munari 2008	

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
BO2A	Benthic Opportunistic Annelida Amphipod index	Index based on indicator species		BOPA	Abundance of amphipods and opportunist annelids including Oligochaeta and Hirudinea	French freshwater zones of transitional waters (i.e. up to the upper limit of the tidal range)	Dauvin et al. 2007, Dauvin and Ruellet 2009	
BOPA	Benthic opportunistic polychaetes amphipods index	Index based on indicator species	The BOPA index considers the total number of individuals collected in the samples, the frequency of opportunistic polychaetes, and the frequency of amphipods (except the genus Jassa)	Improved version of the Opportunistic Polychaetes/Amphipods ratio proposed by Gomez-Geistera & Dauvin 2000	Abundance of amphipods and opportunist annelids	French marine and estuarine habitats	Gomez-Gesteira and Dauvin 2000, Dauvin & Ruellet, 2007	There is the need to calibrate the thresholds between EcoQs classes as defined for these medium-to-fine sand communities, which are characteristics of shallow sublittoral soft-bottoms of the north-western Mediterranean Sea.
BQI	Benthic Quality Index		Index is based on the proportion of sensitive/tolerant taxa adjusted for species richness and low abundance a local scale is used to convert biotic index in an ecoQ (based on the highest value of BQI found in the studied area)	Based on the work by Rygg (assessment of sensitivity tolerance through the assessment of Esi) Modified in a technical report 2006 (in swedish), published in a legal regulation 2008	Proportion of sensitive/tolerant species (ES500.05 of each species) Number of species Number of individuals	Swedish coast	Rosenberg et al. 2004, Blomqvist et al. 2006	
BRI	Benthic response index	Multimetric index based on indicator species	Two-step method in which ordination analysis is employed to establish a pollution gradient, afterwards the pollution tolerance of each species is determined based upon its abundance along the gradient (Smith et al. 1998). The main goal is to establish the abundance-weighted average pollution tolerance of the species in a sample.	Marine analogue of the Hilsenhoff index used in freshwater benthic assessments (Hilsenhoff 1987)	Classification of species sensitive/tolerance	Southern California coastal shelf	Smith et al. 2001	

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
BTA	Biological Traits Analysis		taxonomic-free descriptor, biological-functional traits				Marchini et al. 2008	
Carlit	Ecological Quality index		This index, which completely fulfils the requirements of the WFD, is expressed as a ratio between the observed values in the sector of shore that is being assessed and the expected value in a reference condition zone with the same substrate and coastal morphology (Ecological Quality Ratio, EQR). Based on the cartography of littoral and upper-sublittoral rocky-shore communities				Ballesteros et al, 2007	
Daphne		Multimetric index for coastal waters	A multimetric index based on six characteristics of the community that do not require in-depth taxonomic expertise: number of mollusc species, % of bivalves, % of polychaetes, abundance of the opportunistic species Corbula gibba, % of amphipods and number of 'typical mollusc species' that are individuated by multivariate analysis			Italy, Northern Adriatic Sea	Forni and Occhipinti-Ambrogi, 2007	
DKI	Danske Kvalitet Indeks	Index based on indicator species	A multimetric approach which takes into account the proportion of sensitive/tolerantspecies, measured by the AMBI; a diversity component, Shannon–Wiener index;and a factor to compensate for low densities and species numbers. All variables have equal weight.		Proportion of sensitive/tolerant species Number of species Number of individuals	Denmark	Anon, 2007; Borja et al., 2007	
EQI	Ecofunctional Quality Index	Multimetric index based on macroinvertebrates *and aquatic flora*, for lagoons				lagoon environments	Fano et al. 2003	

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
FINE	Fuzzy INdex of Ecosystem Integrity	Multimetric index based on macroinvertebrates *and aquatic flora*				lagoon environments	Mistri et al. 2007, 2008	
IEI	Index of environmental integrity						Paul 2003	source Pinto et al. 2009
IQI	Infauna Quality Index	Index based on indicator species	A multimetric index, combining AMBI (disturbance/sensitive taxa), Simpsons (diversity), number of individuals (abundance), and number of taxa	A combination of existing indices.	Classification of sensitive/tolerance speciesAbundance of species	England	Prior et al., 2004 Borja et al. 2007	
ISD	Index of size distribution		taxonomic-free descriptor, body size distributions				Reizopoulou and Nicolaidou 2007	
ISI	Indicator species index			based on the improved version of the Hurlbert index (1971)			Rygg 2002	source Pinto et al. 2009
ITI	Infaunal trophic index	Infaunal Trophic Index	Based on trophic guilds		trophic guilds		Word 1978	source Pinto et al. 2009
M-AMBI	Multivariate-AMBI	Integrative index	Multivariate analysis based on the relative abundance of 5 ecological groups and species richness and diversity	AMBI	Classification of sensitive/tolerance species Abundance of species	soft bottom habitats in European estuarine and coastal environments	Borja et al., 2004 Muxika et al., 2007	in use in Europe, Africa, America and Asia
MarBit	Marine Biotic Index Tool	Multimetric index	A multimetric index based on four single indices: Proportion of sensitive species, proportion of tolerant species, abundance distribution (log-normal distribution), Taxonomic spread index (TSI). All variables have equal weight.		Classification of sensitive/tolerance species by reference list	Baltic	Meyer et al. 2008	in use MS Germany, Baltic

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
MEDOCC		Index based on indicator species	The MEDOCC index, presented in MED-GIG, 20007, is an adaptation to the Western Mediterranean area of the AMBI index developed for the Atlantic coast (Borja et al., 2000) based on sensitivity/tolerance of the species. The MEDOCC index is able to detect organic enrichment following communities succession The main differences with the original AMBI method proposed by Borja et al. (2000) are the following: 1) Change in the categories of the ecological groups in some species. 2)Four ecological groups have been considered (instead of five in AMBI) and 3)change in the algorithm for the calculation of the index.	AMBI	Classification of sensitive/tolerance species Abundance of species	Western Mediterranean Area	Pinedo et al. in GIG 2008	in use MS Spain
mIBI	Macroinvertebrate index of biotic integrity	Multimetric index				Calcasieu estuary (USA)	Carr and Gaston 2002	source Pinto et al. 2009
MISS	Macrobenthic index of sheltered systems	Multimetric index	MISS consisted in including a selection of existing benthic indicators with additional metrics describing the community and its trophic structure. MISS was inspired by the development of Indices of Biological Integrity conducted in North America.			France; Zostera noltii seagrass bed in Arcachon Bay	Lavesque et al., 2009	
MMI	Macrofauna monitoring index						Roberts et al. 1998	source Pinto et al. 2009
NQI	Norwegian Quality Index	Index based on indicator species	Sensitivity component: AMBI index based on the relative presence of pollution-sensitive species in the sample. Diversity component: SN diversity index (Rygg, 2006).	none	Proportion of sensitive/tolerant species. Number of species Number of individuals	Norwegian fjords and coastal waters	Rygg, B. 2002 Rygg, B. 2006	

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference	comments
OSI	Organism sediment index						Rhoads and Germano 1986	source Pinto et al. 2009
P-BAT	Portugese Benthic Assessment Tool	Multivariate	A multimetric approach which takes into account the proportion of sensitive/tolerantspecies, measured by the AMBI; a diversity component, Shannon–Wiener index; and Margalef index	AMBI, M-AMBI	Classification of species sensitive/tolerance	Portugal	Teixeira et al. 2008	in use MS Portugal
PLI	Pollution load index						Jeffrey et al. 1985	source Pinto et al. 2009
TWIN	TWo-stage INdex	Multimetric index	Use of hard bottom macrobenthos community. TWIN is calculated in two separate stages. First, the presence and abundance of macrobenthic species are used to define, through a fuzzy model, a station's membership grade in six pre-defined ecological sectors (Lagoon Mouth, Vivified Lagoon, Rough Eutrophic, Calm Eutrophic, Urban, Estuarine), each corresponding to characteristic communities. Then a formula links the membership grades to the five ecological status classes ranging from high to bad quality, as requested by the European Water Framework Directive.			lagoon environments (Italy, Adriatic Sea)	Marchini and Occhipinti-Ambrogi 2007	
VPBI	Virginia province benthic index						Paul et al. 2001	source Pinto et al. 2009
ZKI	Index of zoobenthos community	Index of zoobenthos community	assessment method: Biomass and Feeding guilds				Borja et al.,2009	in use MS Estonia

*For the full references not discussed further in this report, please see the report of the ICES WKBEMET workshop (ICES, 2008)

2.2.5.5.2 Applicability of some commonly used indicators*

Index	Is the index easy to calculate? What possible problems can occur?	Can the index be easily claculated in any kind of data set?	Does the index dependent of seasonal or year to year variablility?	Is the index dependent of sampling effort?	Is there an operational tool available for calculation of the index?	Is the index easy to communicate?	Reference
AMBI	Yes. Problems can occur with low number of spp (<3), high percentage of unassigned spp. (>20%), or erroneous assignations	Yes, abundance is needed	No	No	Yes a free software (http://ambi.azti.es)	Yes, very visual	Borja et al., 2000
BENTIX	yes	not in estuarine ecosystems	no	no	yes	yes	
BEQI	No several steps involved	Yes a representative number of reference and assessment samples per habitat is needed	Seasonal dependent; includes year-to-year variability	Yes but the method si constructed to handle datasets with different sampling effort	Calculation based on access database	Moderate	Ysebaert & herman, 2004; Van Hoey et al., 2007
BHQ	Subjectivity in the Interpretation of the images	Needs just images		No	Yes		
B-IBI	No Many metrics required to calculate B-IBI. The definition of reference condition needs physical data	Yes for data collected in transitionnal waters (oligohaline zone excluded)		No	Yes	No, probably no EQR sensu WFD.	Dauer et al., 1992; Daurer, 1993; Weisberg et al., 1997; Borja et al., 2007
BOPA	Yes Taxonomic reduction	Yes limitation when n < 20	Yes	No	Yes		
BQI	Calculation of index is fairly easy to calculate when you have the sensitivity values at hand. Calculation of species sensitivity in step one requires a lot of data from both disturbed and undisturbed sites. There might be some problems when calculating sensitivity walues for dominating species that are indifferent to disturbance.	Needs large data set to calculate ES50 values. Given you have the sensitivity values you can calculate BQI on any data set.	Low seasonal variability. The seasonal change was restricted to changes in abundance of the dominant species, which belonged to the dominant ecological groups.	Yes but not the adapted BQI_{ES50}	Easy to use in Sweden where sensitivity values, a small Excelfile for calculation and instructions for use exists. More complicated in other areas.	Yes, the index BQI is easy to communicate, the calculation of sensitivity values is not so easy to communicate.	

Index	Is the index easy to calculate? What possible problems can occur?	Can the index be easily claculated in any kind of data set?	Does the index dependent of seasonal or year to year variablility?	Is the index dependent of sampling effort?	Is there an operational tool available for calculation of the index?	Is the index easy to communicate?	Reference
IQI	Yes Simple formula	Yes, requires only species abundance data		No	Yes	Yes	Borja et al. 2007
M-AMBI	Yes. Need of reference conditions by habitat/type. Need of a certain number of samples	Yes, abundance is needed	At some extent, due to diversity and richness	No, due to richness	Yes a free software (http://ambi.azti.es)	Yes, very visual	Borja et al., 2004
NQI		Number of individuals should be larger than 5		Very slight			Rygg B. p.c.

*Taken from WKBEMET (ICES 2008, and references there in)

2.2.5.5.3 Regions/habitats and stressors tested forsome common indicators (contains both macrobenthic and macrophyte indicators).

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
NQI	Sweden/Denmark/UK/Spain	Kattegatt/Skagerrak/North Sea/NE Atlantic		Hypoxia	Good	Anon 2007
			soft bottom	Recovery from organic pollution	Good	Anon 2007
				Toxic pollutants	Good	Anon 2007
IQI	United Kingdom, Ireland	Used on data in the UK and Ireland				
	Portugal	Portugese coast and estuaries				
B-IBI	USA (Chesapeake Bay, Carolina and mid-Atlantic estuaries)	Atlantic ocean	estuarine habitats- Distinction of habitats by their salinity range of variation	organic matter and eutrophication	Good capacity of discrimination. Influenced by salinity, depth and hypoxia and not by chemical contaminants. Precautions in oligohaline zone	Dauer et al. 1992, Dauer 1993, Weisberg et al. 1997, Borja et al. 2007
BEQI	Belgium, Netherlands	Southern Bight of the North Sea, estuaries (Westerscheldt, Eems-Dollard), semi-enclosed sytems (Waddensea, Oosterscheldt), salt lakes (Veerse meer, Grevelingen)	soft-bottom estuary-coastal waters subtidal-intertidal	eutrophication, oxygen depletion	good	Van Hoey et al. 2007
				Dredge disposal	good	Van Hoey et al., 2010
				alien species	good	Ysebaert and Herman 2004
BHQ	France, Germany, Sweden, Ireland, Greece	Mediterranean, Atlantic	mud, subtidal	eutrophication, oxygen defiency	good	
				trawling/dredging	good	
BOPA	France, Spain, Algeria, Italy	English Channel, Atlantic,	subtidal, intertidal	Oil spill	good	

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
				organic matter enrichment	good	Dauvin et al. 2006, Dauvin and Ruellet 2007
				salinity gradient	good	
				Sewage	bad	J.A. de-la-Ossa-Carretero et al, 2009
				physical disturbance	bad	
AMBI	whole European Union, Norway, Turkey, north Africa, Greenland, USA, Canada, Mexico, Chile, Uruguay, Brazil, China, Hong-Kong, Indonesia, Iran, Reunion Island, etc.	Atlantic, Mediterranean, Norwegian Sea, North Sea, Baltic, Black Sea, Chesapeake, Florida, Canada, Pacific Ocean, Indian Ocean, Red Sea, etc.	soft-bottoms from subtidal, intertidal, estuaries, continental shelf, batial, lagoons, mangroves, etc.	eutrofication, organic matter enrichment, aquaculture, oil spill, drill cuttings, metal pollution, dredging, sediment disposal, engineering works, sludge disposal, etc.	good, in general. It is insensitive to natural disturbance	more than 220 SCI references
				aggregate extraction, physical disturbance, trawling	no clear response, probably because these pressures act as natural disturbance (i.e. as storms)	Muxika, I., A. Borja, W. Bonne, 2005. The suitability of the marine biotic index (AMBI) to new impact sources along European coasts. Ecol Indic, 5: 19-31.
M-AMBI	whole European Union, Norway, USA, Canada, China, etc.	Atlantic, Mediterranean, Norwegian Sea, North Sea, Baltic, Black Sea, Chesapeake, Florida, Canada, Pacific Ocean, Indian Ocean, etc.	subtidal, intertidal, estuaries, continental shelf, lagoons	eutrofication, organic matter enrichment, aquaculture, oil spill, metal pollution, dredging, sediment disposal, engineering works, etc.	good, in general. It is relatively insensitive to natural disturbance	more than 60 SCI references
Bentix	Greece	Mediterranean		Organic matter enrichment	Good	

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
			Soft bottom (coastal lagoon)	Pollution	Good	Simboura and Reizopoulou 2008
				eutrophication		Simboura et al. 2005
				aquaculture		Aguado-Giménez et al. 2007
			coastal area	anthropogenic impact from land-based sources	Good	Simboura and Reizopoulou 2007
			coastal area	metalliferous waste charge	Good	Simboura et al. 2007
			Subtidal	Oil spill	Bad	
				Aquaculture	Good	
	France	Atlantic	Bay of Seine	Contaminated environment		Dauvin et al. 2007
			Seine Estuary	Contaminated environment		Dauvin et al. 2007
	Spain	Mediterranean				
	Cyprus	Mediterranean				
	France					

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
	Turkey	Mediterranean	Izmir bay	Pollution	Good	Dogan, 2004
BQI see Gremare et al 2009	Sweden	Kattegatt/Skagerrak	Soft bottom	Recovery from organic pollution	Good	Anon 2007
	France	Bay and estuary of Seine	Soft bottom	Organic pollution, metal contamination	Good	Dauvin et al. 2007
	Sweden, Germany, France, Portugal, Algeria	Baltic North Sea, Mediterranean, English Channel	Soft bottoms, sublittoral, shelf, estuaries	Organic enrichment	Good	
				Eutrophication, oxygen deficiency	Good	
				Salinity gradient	Good	
	Finnland	Baltic Sea		Fish farming	Response to changes in oxygen saturation, organic matter and species richness	Perus et al. 2007
	Sweden	Kattegatt/Skagerrak	Soft bottom	Hypoxia	Good	Anon 2007
DKI	Sweden	Kattegatt/Skagerrak	Soft bottom	Recovery from organic pollution	Good	Anon 2007
BCI	Italy					
BITS		Mediterranean	Soft bottom, coastal lagoons	eutrophication	Good	Munari et al. 2009
				sediment organic matter quality	Bad	Munari et al. 2009
BO2A	Turkey	Marmara Sea				

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
ISI	Greece, Slovenia, Italy, Spain, France, UK, Norway					
ITI	UK, Shetland	North Sea, Mediterranean, Atlantic		aquaculture	to be improved by using a suite of benthic indicators (rather than a single indicator)	Borja et al. 2009
		North Sea		aquaculture		Nickell et al. 2009
MarBit	Spain					
MEDOCC	Italy, Slovenia, Spain					
MISS	France			Physical disturbance	good	Lavesque et al., 2009
PEEI	France	Mediterranean	Posidonia		good	Gobert et al, 2009 UNEP Congress
BiPo	Spain, France Italy	Mediterranean			good	Lopez y Royo et al, 2009
EEI	Greece	Mediterranean			good	

Index	In what geographic location is the index tested?			For what specific stressors (human and natural) is the index tested and what is the response)		Reference
	countries	seas	habitat	pressures	response	
	Spain	Mediterranean			good	
	Slovenia	Mediterranean			good	
	Cyprus	Mediterranean			good	
	Algeria	Mediterranean			good	
Carlit	Spain	Mediterranean	Hard bottom macroalgae		good	Asnaghi et al, 2009
	Greece	Mediterranean			good	Asnaghi et al, 2009
	France	Mediterranean			good	Mangialajo et al, 2007
	Italy	Mediterranean			good	Nocolic et al, 2009

*From WKBEMET, references there in.

2.2.5.5.4 Publications which compared different macrofauna indicators

REFERENCES	number of species	total abundance	biomass	margalef index	Pielou evenness	Simpson	dominance index	ES(100)	ES(50)	S-W Diversity	Taxonomic	ABC curves	Exergy	MDS	AMBI	EQR	EQR Uk approach	m-AMBI	BENTIX	BOPA	BQI	BQIw	BQIes	12EC	ITI	B-IBI	BBI(Finland)	BMI(UK?)	DKI(Denmark)	BO2A	RBI	NQI (Norway)	MEDOCC	TICOR
Rosenberg et al.,2004																					X													
Salas et al.,2004 *				X		X				X		X			X						X													
Reiss & Kroncke, 2005								X		X	X	X				X	X					X												
Labrune et al., 2006 *										X					X						X													
Quitino et al., 2006 *	X	X	X	X	X				X	X					X		X				X													
Blanchet et al., 2007 *										X					X				X	X	X													
Borja et al., 2007 *															X			X									X							
Chainho et al., 2007				X						X		X			X																			
Dauvin&Ruellet,2007																				X														
Dauvin et al.,2007 *	X	X					X		X	X					X				X	X	X			X	X									
Fleischer et al., 2007 *															X						X	X	X											
Perus et al., 2007															X						X						X	X	X					
Pranovi et al., 2007 *										X	X				X			X	X	X														
Zettler et al., 2007										X					X						X													
Borja et al., 2008															X			X								X								
Puente et al., 2008	X	X									X				X			X																
Chainho et al.,2008																										X								X
Bigot et al., 2008															X			X																
Borja & Dauer, 2008	X	X	X												X			X	X															
Josefson et al.,2009	X	X								X					X						X								X			X		

REFERENCES	number of species	total abundance	biomass	margalef index	Pielou evenness	Simpson	dominance index	ES(100)	ES(50)	S-W Diversity	Taxonomic	ABC curves	Exergy	MDS	AMBI	EQR	EQR Uk approach	m-AMBI	BENTIX	BOPA	BQI	BQIw	BQIes	12EC	ITI	B-IBI	BBI(Finland)	BMI(UK?)	DKI(Denmark)	BO2A	RBI	NQI (Norway)	MEDOCC	TICOR
Ambrogi et al.,2009															X			X															X	
Gremare et al., 2009															X						X													
Borja et al., 2009															X										X									
Dauvin et al., 2009																				X										X				
Simonini et al.,2009															X			X	X															
Ranasinghe et al., 2009																					X										X			
Ware et al., 2009	X	X						X		X	X				X										X									
Pinto et al., 2009															X			X	X	X	X	X			X									
Simboura et al., 2009															X			X	X															

* From WKBEMET, references there in

2.2.5.6 Fish

2.2.5.6.1 General description for some commonly used fish-based indicators

Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference
AFI	AZTI's Fish Index	multi-metric index	The Estuarine Demersal Indicators was developed for the Basque Country, using fish and crustaceans data.		Richness of species, trophic composition, invasive species, guilds of the species, opportunistic species, etc. (9 metrics in total)	Basque Country	Borja et al., 2004; Uriarte and Borja, 2009
BHI	Estuarine Biological Health Index	multi-metric index	Whereas the CDI measures the degree of dissimilarity (degradation) between the potential community and the actual community, the BHI modifies the CDI to incorporate a measure of the degree of similarity between the potential community and the actual community. The BHI is only based on presence and absence data and does not take into account the relative proportions of the various species present.	CDI			Cooper et al., 1994
CDI	Estuarine Community Degradation Index	multi-metric index	The CDI is based on a comparison of the fish community present within an aquatic system, to the community that would exist in the absence of, or prior to, degradation. The index assumes that differences between the potental commuity ant the present assemblage are due to habitat degradation.			South African estuary	Ramm, 1988
EBI	Fish-based Estuarine Biotic Index	multi-metric fish index	The fish-based Estuarine Biotic Index consists of a balanced set of metrics combining several aspects of the estuarine community such as trophic status, species richness, nursery function and presence of intolerant estuarine type species.			brackish section of the Scheldt River Estuary	Breine et al., 2007

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Index	Full name	Functional classification	Description of index	historical evolution	input information (eg. Classification of species sensitivity/tolerance)	Originally developed for?	Reference
EFCI	Estuarine Fish Community Index	multi-metric fish index	The index comprises 14 metrics or measures that represent four broad fish community attributes: species diversity and composition, species abundance, nursery function and trophic integrity.			South African estuary	Harrison et al., 2004
EQUATION	Estuarine QUAlity and condiTION	multi-metric index	EQUATION is presented for integrated evaluation of estuarine quality and is based on an aggregation of four different components: vulnerability, measured on the basis of physical characteristics; water quality, based on nutrients, primary production and dissolved oxygen; a composite index of benthic quality; and an index of quality for higher trophic levels. Each component is evaluated by means of a set of descriptors.			Is tested on 5 different estuaries in the US and Europe	Ferreira J.G., 2000
FHI	Estuarine Fish Health Index	multi-metric index	FHI focus on both qualitative and quantitatve comparisons with a 'reference' fish community. In the qualitative assessment, the number of species in each estuary is compared to the average number for the groups to which it belonged. In the quantitative assessment, the per cent abundance of the species within each estuary is compared to the per cent abundnce of the species captured in the group to which it belongs (using Bray-Curtis similarity measure).				Harrison et al., 2000
FRI	Estuarine Fish Recruitment Index	multi-metric index	Uses ichthyological information to assess changes in habitat integrity, especially the availability and suitability of estuarine nursery areas to marine migrant fishes.			South African estuary	Quinn et al., 1999
MFCI	Marine Fish Community Index	multi-metric fish index	The MFCI is divided into 4 typologies: Rocky subtidal; shallow, intermediate and deep soft-bottoms. It incorporates both functional and structural community information.				Henriques et al., 2008
TFCI	Transitional Fish Classification Index	multi-metric index	The TFCI uses 10 ecological measures to analyse fish populations. It incorporates both structural and functional attributes of estuarine fish communities.			The Thames	Coates et al., 2007

2.2.5.6.2 Publications which compared different fish-based indicators

REFERENCES	CDI	EFCI	EHI	EBI	TFCI	FHI	FRI	EBI (breine)	EDI
Cooper et al.,1994	X		X						
Henriques et al., 2008	X	X	X	X	X				
Martinho et al., 2008				X				X	X
Whitfield & Elliott, 2002	X		X	X		X	X		

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2.3 MONITORING MARINE POPULATIONS

2.3.1 MONITORING MARINE POPULATIONS: REVIEW OF METHODS AND TOOLS

Coordinator: Anke Weber (NIVA)

2.3.1.1 Plot sampling

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2.3.1.1.1 Introduction

Plot sampling is a comprehensive method used mainly for measurements of population size in terms of abundance. Furthermore, plot samples can be used for estimating other relevant parameters of a population, such as biomass or length structure. The key idea of plot sampling is to estimate population abundance by "scaling up" the counts of animals from the covered (surveyed) area to the study area (Borchers et al. 2004). Hence, abundance in a study area of surface *A* is estimated by $\hat{N} = \frac{nA}{A_C} = \frac{n}{P_C}$, where A_c is the surveyed area and $P_c = \frac{A_c}{A}$ is the 'coverage probability', which is the fraction of the study area that was covered by the survey.

So far the term "plot sampling" is not very common in marine science. For our purpose the term plot sampling comprises all the sampling methods that are based on working with plots on different spots in a community, and counting the number of organisms in each plot. There are many different types of plot sampling which differ with respect to the shape of the plot, which is usually a square, a strip, or a circle. However, all types of plot sampling are identical in terms of statistical treatment. In general, a major advantage of plot sampling that use grabs, trawls and dredges is that analyses on community level are possible, while other monitoring methods have a stronger focus at one population or individuals. In

In the marine environment a wide variety of sampling devices are used for plot sampling. These can be subdivided into *in situ* sampling equipment (grabs, trawls, dredges, nets) and imaging equipment (cameras or video recorders and underwater vehicles such as ROVs or AUVs).Plot sampling with underwater visual surveys (e.g., quadrat sampling or strip transects) do not need special sampling equipment but only equipment to define the plot (a frame for quadrats; a reel and a line for strip transects) and SCUBA gear.
For a summary and comparison of standard plot sampling devices, see table 2.3.1.1. The choice of a suitable device depends primarily on the characteristics of the target population or community (e.g. size, distribution of the animals), and in which habitat type it occurs. Target groups that can be monitored by plot sampling are populations or communities of fish, invertebrates, phytoplankton or macrophytes. *In situ* sampling tools for pelagic organisms include midwater or pelagic trawl, plankton nets (ring nets, Bongo nets, MOCNESS), neuston nets, drift nets, gill nets, fyke nets, and tools for benthic organisms include bottom grabs (Hamon grab, Van Veen grab), bottom trawl (beam trawling, otter trawling, shrimp trawling), dredges (Newhaven scallop dredge, Rallier-du-bathy dredge), corers (Reineck boxcorer, nulticorer), hyperbenthos sledge, hyperbenthos net.

A common method using visual and imaging techniques is quadrat sampling. A quadrat is a frame (square, rectangular or circular) employed to standardise the size of the sample area in order to ensure the comparability of results between samples and monitoring events. Quadrat sampling is commonly used in the inter- and subtidal zones. The choice of the quadrate design (e.g. size, shape) relates fundamentally to the characteristics of the population under investigation. Corers can be used for additional subsampling.

For plot sampling of specific groups detailed information is given in reviews about epibenthos (e.g. Coggan et al. 2007; Rees et al. 2009), endobenthos (Kröncke and Bergfeld 2003; Eleftheriou and McIntyre 2005; Rumohr 2009) and fish (Michalopoulos et al. 1993) and in the appropriate sections within this document.

Table 2.3.1.1. Comparison between devices/methods for monitoring populations with plot sampling.

Device	Target populations	Efficiency	Accuracy/resolution	Bias	Comment
Grabs	Endobenthos	Can be influenced by the penetration depth of the gear, weight of the gear, 'bite' profile, sediment type and the subsequently sampled volume of sediment; Variable depending on weather conditions, hydrodynamics, size of the vessel and the experience of the operator affect the quality of the samples and grab failures (grab bouncing, drift, pressure waves)	Quantitative	Collection of mobile epibenthic species limited Insufficient penetration depth can miss deep burrowing species	
Corers	Endobenthos	See grabs Longer handling time compared to grabs, unsuitable under rough weather conditions (more than 5 Beaufort)	Quantitative	See grabs Undisturbed sediment cores	Additional observation and measure devices can be added to the frame, cores used for boxcosm-experiments, benthic chamber measurements
Trawls and sledges	Epibenthos, Hyperbenthos, Nekton	Variable depending on weather conditions, hydrodynamics, size of the vessel and the experience of the operator affect the quality of the samples and failures (drift, pressure waves)	Semi-quantitative; Accuracy depending on the mesh size, tow length should be recorded with depth sensor. Shorter the tow length more accurate the information on spatial structure of population	Gear may skip over the seabed reducing the area sampled Nets may clog reducing the sample efficiency Catchability may vary depending on various factors, which often makes comparisons of relative abundance difficult	Trawl performance should be checked by electronic monitoring devices (e.g. video)
Dredges	Endobenthos, Epibenthos	Varying penetration depth for different sediment types See trawls and sledges	Semi-quantitative	Catchability may vary depending on various factors, which often makes comparisons of relative abundance difficult	

Nets	Nekton, Plankton		Semi-quantitative Accuracy depending on the mesh size	Nets may clog reducing the sample efficiency	
Visual (diving)	Epibenthos, Nekton	Non-destructive Appropriate for protected species/habitats Appropriate for all kinds of substrates/habitats (rocky and coral reefs, sandy/muddy bottoms, seagrasses etc) No permanent record	Quantitative	Imperfect detectability will lead to underestimation of abundance	When there is no assurance of perfect detectabilty, distance sampling methods may be preferable
Photocameras	Epibenthos, Nekton, Plankton, Fish	Non-destructive (specific) recognizable taxa Images in 2D or 3D	Quantitative (when scaled) Image resolution and visibility limiting		Illumination-depended (except in shallow and clear waters) “4D” (time advantage)
Drop-down videocameras a) ROV/submersible	Epibenthos, Nekton, Plankton, Fish	Non-destructive (specific) recognizable taxa Limited use in tidal currents or windy conditions Active control of position in small scale Transect observation	Quantitative (when scaled) Limiting accuracy: Image resolution and visibility		Speed, illumination, turbidity-depended
Drop-down videocameras b) Lander	Epibenthos, Nekton, Plankton, Fish	Non-destructive (specific) recognizable taxa Continuous monitoring possible	See ROV/submersible		Illumination, turbidity-depended “4D” (time advantage)
Towed videocameras	Epibenthos, Nekton, Plankton	Non-destructive (specific) recognizable taxa Limited flexibility for species recognition Inertia in direction changes	Towing speed influences image quality and positional accuracy		Speed, illumination, turbidity depended Attaching weights to the camera frame might be required to maintain position
Hydroacoustics	„reefbuilding“ Epibenthos, Nekton	Non-destructive Cost-effective method over large areas	Qualitative, low resolution Only specific ecosystem components		Ground-truthing needed

2.3.1.1.2 Assumptions

In plot sampling, the critical assumption is that all individuals present in the surveyed areas A_c are detected (or caught when referring to devices such as trawls, dredges, grabs etc.). However, this assumption cannot be tested using the survey data, and to ensure that it holds to a good approximation additional data is needed.

In underwater visual surveys with strip transects, to ensure perfect detectability it may be necessary to use narrow strips, which is problematic for scarce species (Burnham and Anderson, 1984; Buckland et al., 2001) and increases the variance of density estimators (Kulbicki and Sarramégna, 1999; Buckland et al., 2001). Kulbicki and Sarramégma (1999) found that the maximum width that assures detection of all individuals may vary even for closely related fish species and may even change for a given species at various habitats. When designing a strip transect survey, there is no way to estimate the maximum strip width for certain detection of the target species.If the assumption that all individuals present in the surveyed areas are detected is not met, there is underestimation of abundance, which is not uncommon in underwater surveys (Katsanevakis 2009).

In many cases, when the assumption of perfect detectability (or catchability) does not hold, an assumption of constant detectability (or catch efficiency) is made, which allows for the estimation of relative abundance (or index of abundance) such as 'catch per unit effort', which is commonly used in trawls and dredges. However, this assumption is also commonly violated. Catch rates of trawls and dredges may be quite low and variable, depending on various factors such as trawling speed, gear specifications, substrate, behaviour and life history of target species, size of the individuals, duration of the haul, time of the day, and season (e.g., Shafee 1979; Chapman 1980; McLoughlin et al. 1991; Bailey et al. 1993; Giguere and Brulotte 1994; Tuck et al. 1997; Hall-Spencer et al. 1999; Reis et al. 2006).

For community studies, catch efficiency of the sampling device is assumed constant for all species. This assumption is critical as soon as not only one species/population is targeted but an entire community including many different taxa. If the catch efficiency of the sampling device differs significantly between species, as e.g. found for trawls and dredges, the description of the community composition is biased. In case of sampling with grabs and corers expert judgement prior analyses is often sufficient to account for this bias, but in case of towed gear catch efficiency is rarely known.

2.3.1.1.3. Sampling design

In this section we describe some aspects of sampling design for plot sampling. For more comprehensive information about design and analysis in benthic surveys see Underwood and Chapman (2005). Plot sampling-surveys to assess fish population densities can be effectuated using underwater video systems assembled on sophisticated vehiclessuch as Remotely Operated Vehicles (ROVs) and Autonomous Underwater Vehicles (AUVs; Michalopoulos et al. 1993). For zooplankton sampling design the **ICES (International Council for the Exploration of the Sea) Zooplankton Methodology Manual**provides comprehensive coverage of modern techniques in zooplankton ecology (Harris et al. 2000)

Two important aspects of sampling design are sample location and replication. The allocation of sampling unit must ensure representativeness of the entire study area.

2.3.1.1.4. Choosing Plots

Several methods exist to choose plots in a larger study area. Plots can be chosen randomly in the community or at regular intervals. In either case, grid coordinates can be used to mark the location. If random sampling is desired, random numbers can be used to determine the coordinates. Four commonly used strategies for locating samples are judgement (also called selective), random, stratified random (according to strata based on depth, sediment type, etc.) and systematic (based on a user-defined grid) (Table 2.3.1.2).

Table 2.3.1.2. Summary of the advantages and disadvantages of different types of sample selection (in Davies et al. 2001).

Sample selection	Advantages	Disadvantages	Comment
Judgement	Can be quick and simple if knowledge of habitat/species is sufficient. Samples can be deliberately taken around e.g. a rare species or feature of particular importance. Useful when the locations of a rare habitat or species are known. Samples can be placed in areas subjectively considered homogenous or representative.	Extrapolation of results to the whole feature or site is not valid without strong justification. Comprehensive knowledge of the site is required. Statistical analysis is not valid and errors are unknown.	Efficient but dependent on quality of prior knowledge. Should not be used if there are any concerns over the quality/reliability of this prior knowledge.
Random	Requires minimum knowledge of a population in advance. Free of possible classification errors. Easy to analyse data and compute errors.	Locating sample observations can be time-consuming. Often larger errors for a given sample size than with systematic sampling. May not monitor what is required.	Only useful when a feature is spatially homogeneous throughout the SAC. Any restrictions on access will compromise the process.
Stratified random	Ensures that all the main habitat types present on a site will be samples (if defined as strata). Characteristics of each stratum can be measured and comparisons between them can be made. Greater precision is obtained for each stratum and for overall mean estimates if strata are homogeneous.	If strata have not been identified prior to monitoring, preparing can be time-consuming. The most appropriate stratification for a site at one time may have changed when repeat surveys are carried out. Monitoring efficiency may therefore also change.	The optimum approach for most SAC monitoring requiring a degree of randomness.
Systematic or grid	If the population or attribute is ordered with respect to some pertinent variable, a stratification effect reduces variability compared with random sampling. Provides an efficient means of mapping distribution and calculating abundance at the same time.	If sampling interval is correlated with a periodic feature in the habitat, bias may be introduced. Strictly speaking, statistical tests are not valid, though in practice, conclusions are unlikely to be affected.	This has the advantage of providing an estimate of extent and a random subsample can be taken for other analyses.

2.3.1.1.5 Replicate sampling

In general, taking many small samples is considered more informative and reliable than taking a few large samples. This is due to the following main advantages:

- broader coverage of the sample site
- better estimate of spatial dispersion of species
- higher number of degrees of freedom for statistical analyses

For statistical reasons, 2-3 replicates (e.g. grabs) are considered the minimum replication per sample. Accurate replicate sampling is especially important in order to detect ecologically important changes (Rogers et al. 2008).

For community investigations the sufficiency of a sampling effort can be estimated by empirical species- area- curves. These curves estimate the number of replicates (minimal sampling area) required to obtain an acceptable percentage of the total number of species (e.g. Gray and Pearson 1982; Beukema 1988; Hartung 1993). A more sophisticated way to test the sampling efficiency is the use of statistical distributions like the normal, binomial or Poisson distribution (e.g. Elliott 1971; Mühlenberg 1989; Hartung 1993; Pfeifer et al. 1995). These techniques imply an *a priori* accepted level of precision (commonly 0.2 or 0.4).

Clarke and Green (1988) pointed out, that pilot studies are often essential and always desirable in order to establish an appropriate sampling design. Assessing the accuracy and precision of biological effect studies they also introduced a cost function, accounting for limited financial resources.

Accounting to various scales of patchiness in population distribution, sampling should be nested hierarchically (Underwood 1981; Underwood 1991; Morrisey et al. 1992; Ellis and Schneider 2008) to increase the reliability of the samples. The spatial scale of patchiness in the measured variables is often unknown prior to sampling. Consequently, a spatial scale of patchiness be it sampling units (small scale) or sampling locations (large scale) would not necessarily be taken into account by the sampling design nor will it necessarily be revealed analyzing the samples later on. If the within-location variation has not been adequately estimated by sample replication, this prevents valid comparisons among locations because the data are pseudoreplicates (Hurlbert 1984).

Also large scale comparisons of sites can be confounded by the variance on a small scale. This also holds true for temporal variations. Long-term effects are difficult todisentangle from short-term variations (e.g. seasonal), and might be visible only on the long run, even if sampling was undertaken on the appropriate long-term scale (e.g. Buchanan and Moore 1986; Buchanan 1993; Frid et al. 1996). The larger the scale difference between sampling levels (e.g. replicate grab samples - locations), the larger the magnitude of patchiness that could be overlooked will be.

2.3.1.1.6. Additional sampling pre-requirements

- Number of plots monitored: A sufficient number of plots should be quantified so that adding additional plots to the sample does not reveal additional new species.
- Plot size (surface area): The size of the plot sample depends on the size of the species being sampled. One square meter or less is usually sufficient for sampling small species. Strip transects for surveys of fish or large invertebrates usually have a surface of some hundreds of square meters.
- Frequency of plot surveys through time: Monitoring is per definition a time-series sampling (repeated sampling can be equidistant or at biologically relevant time intervals).

2.3.1.1.7. Limitations

- The destructive plot sampling methods are not useful for monitoring large or fast species, such as mammals or endangered species in general.
- Most *in situ* sampling devices such as grabs and towed gears are disturbing the sampled area by removing the organisms and altering the substrate, which is problematic especially in sensitive habitats.

- Small sampling area
- Number of replicates limited due to high effort for sample processing
- Semi quantitative (in case of towed gears)

2.3.1.1.8. Statistics and modelling

Numerous univariate and multivariate techniques can be applied to analyse plot sampling data sets. Analyse procedures are extensively described e.g. for benthos in Clarke and Warwick (1994). Here we only refer to the most widely applied techniques and some new improvements.

Traditional univariate statistical data analysis of abundances, species numbers and biomass include typically the calculated mean and standard deviations, minimal and maximal values and are well suitable for population studies. These methods have, however, some constraints applied on community / multi species level.

Over the last decades, multivariate data analyses became the standard techniques for analysing community pattern (Field et al. 1982). They have the great advantage of using the information of the whole species composition. The basic idea of cluster analysis and ordination techniques is to compare different sites/sampling times by calculating similarities/dissimilarities measures.

Cluster analysis reveals pattern by hierarchical sorting illustrated in dichotom dendrograms giving the similarity levels or grouped samples on an axis. The reliability of dendrogramms can only be assessed appropriately in comparison with corresponding ordination techniques (Warwick and Clarke 1991; Warwick and Clarke 1994).
Ordination techniques like multidimensional scaling (MDS), principal component analysis (PCA), canonical correlation analysis (CCA) or two-way indicator species analysis (TWINSPAN) are exploratory methods to analyse the community pattern, visualising trends and changes in community structures. Correlation analyses are used to detect potential relationship commonly between community and environmental data, but also for inter-specific relationships. Correlations are visualised by vectors along which the data are grouped. The directions of the vectors indicate whether the factors might control the community in the same or opposite way.

Using MDS plots the BIOENV program of the PRIMER software package superimposes community data by other data sets and correlation analysis reveals the importance of single factors or factors in combination with the community structure (Clarke and Ainsworth 1993).

The non-metric MDS is a much more flexible tool than the metric PCA technique. For the MDS, analysis of the data is done on a (dis)similarities matrix expressed in distances between the samples in an n-dimensional space. Graphically, it is translated in Euclidean distances plotted in a 2 (or 3) dimensional diagram. The closer the samples lie to each other in the diagram the higher is their similarity. A stress measure indicates how representative the diagram reflects the multidimensional data.

The significance of the revealed pattern can be tested statistically by accompanying randomisation testing procedures, based on the dissimilarities of all pairs of subsamples. E.g. the ANOSIM program by Clarke (1993) uses the differences of the rank similarities between and within the samples. The randomisation has the advantage that it does not rely on a distribution model (see above, 'species curves') and does not require random sampling. Similar are the L-test (Smith et al. 1990) and the MST-test; the latter of which is accompanied by the 'minimum spanning tree' ordination technique (Schleier and Bernem 1996). All these techniques also provide a tool for assessing the importance of single species in terms of being a ‘characteristic species in a data set or not. This can be a crucial factor in assessing the similarity of different data sets.

Because of the complex structure of multivariate data, a general tool applicable for all types of investigations and data is not available. It is always advisable to experiment with several methods to throw light on the data from different viewpoints.

2.3.1.2 Distance Sampling

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2.3.1.2.1 Introduction

Abundance estimation is often confounded by detection probability, which is the probability of correctly recording the presence of an individual within the sampled area. Failure to properly account for detection probability often leads to biased abundance estimators and possibly false estimates of population status and trends. One of the most active areas of biometric and wildlife research is the development of methods and models to properly account for detection probability; this is reflected in the recent proliferation of books describing these approaches (e.g., Thompson et al. 1998; Buckland et al. 2001; Williams et al. 2002; Borchers et al. 2004; Buckland et al. 2004; Thompson 2004). One such widely used method that properly accounts for detection probability is Distance Sampling, which may be considered as an extension of Plot Sampling.

Distance sampling comprises a set of methods for estimating density and/or abundance of biological populations. The standard literature of distance sampling is the introductory book by Buckland et al. (2001) in which conventional distance sampling methods are described in detail and aspects of survey design are covered with several examples. The follow-up of this work is the 'advanced distance sampling' edited volume by Buckland et al. (2004), where automated survey design, multiple-covariate distance sampling, density surface modeling, mark recapture distance sampling and other advanced topics are covered. The standard software for distance sampling surveys is DISTANCE (Thomas et al. 2010). DISTANCE is a Windows-based freeware computer package that allows the researcher to design and analyze distance sampling surveys, aiming to estimate the density and abundance of a biological population.

The main distance sampling methods are line transects and point transects (the analogues of strip transects and point counts respectively). A standardized survey is conducted along a series of lines (in line transects) or points (in point transects) searching for the objects of interest. Objects could be either individuals or groups of individuals (termed clusters). The distance from the line or point is recorded for each animal (or cluster) detected,. A detection function is fitted from the set of recorded distances, which is used to estimate the proportion of animals missed by the survey and hence correctly estimate abundance. Hereafter, we focus on line transect sampling, which is the most widely used distance sampling method in the marine environment; more details on point transects may be found in Buckland et al. (2001).

When the detection of individuals is difficult, a distance sampling method is typically more efficient and cost-effective than a simple strip transect or point count sampling. This is because densities are corrected with the use of the detection function and the sample size is larger for the same amount of effort as all detected individuals may be recorded regardless of how far they are from the line or point.

Unlike plot sampling, distance sampling does not require all objects in the sampled plots (covered region) to be detected. If n animals were detected in a distance sampling survey, then an estimation of the mean density and the total number of animals in the study area is given by $\hat{D} = n / A_c P_a$ and $\hat{N} = n / P_c P_a$ respectively, where P_a is the probability that any particular individual that was in the covered region was detected, A_c is the covered area, A is the entire study area, and $P_c = A_c / A$ is the fraction of the study area that was covered by the survey ('coverage probability'). On average, P_a is the fraction of animals in the covered region that were detected. It has been assumed that the probability that an animal was in the covered region and the probability that an animal in the covered region was actually detected are independent and common for all individuals.

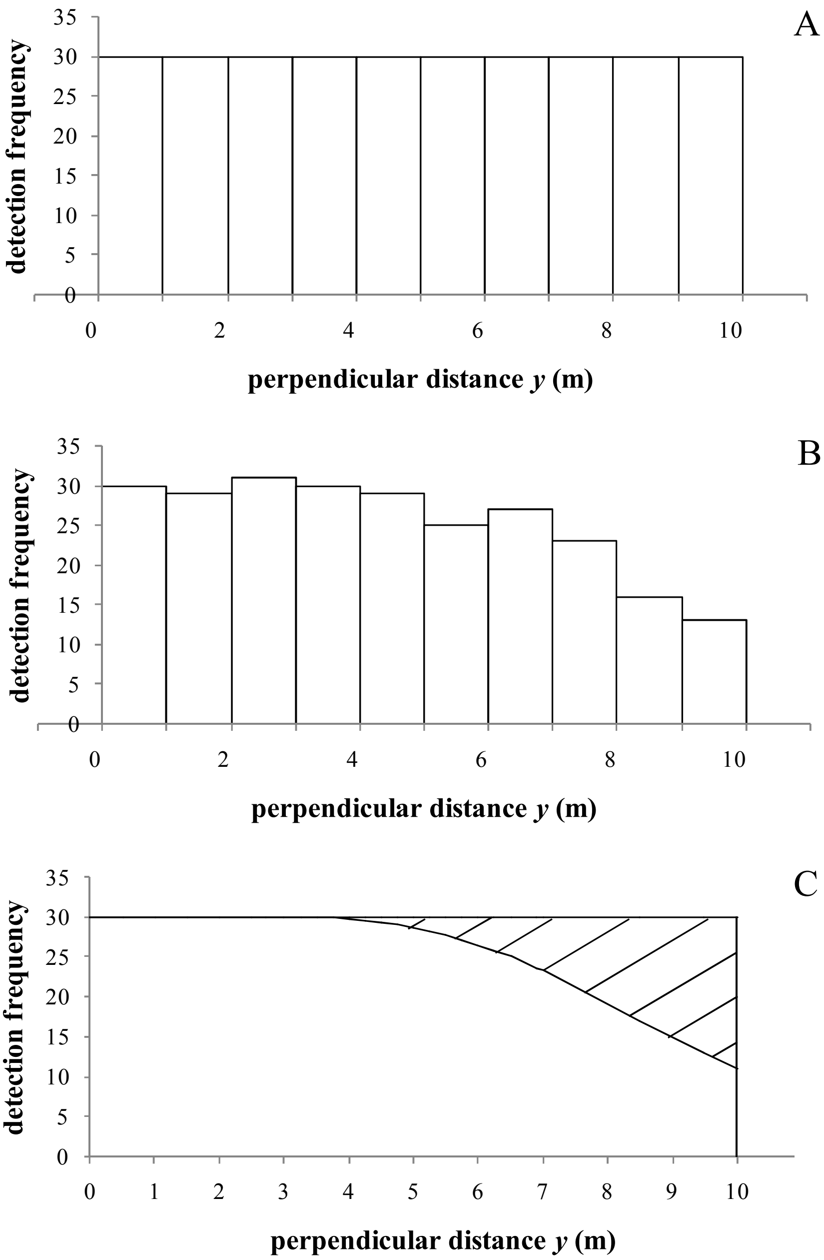

Fig. 2.3.1.1: (A) The expected average histogram of recorded distances, when no individuals are left undetected (or else the expected actual distribution of distances) in a line transect survey; (B) real histogram of distances, where a tendency to detect fewer individuals at greater distances is obvious; (C) a model of the distribution of distances of figure B; the lined area is proportional to the number of individuals that remained undetected.

The main task of the analysis of distance sampling data is to estimate the detection probability P_a (P_c is usually known by design). The fundamental concept behind the estimation of $P_a$from the distance data is straightforward, as illustrated in Fig. 2.3.1.1 (for line transect sampling). If all objects were detected, we would expect, on average, the histogram of the recorded distances y_i from the line to be uniform (Fig. 2.3.1.1A), provided that the lines are randomly placed within the study area. Any decline in the numbers of recorded individuals with increasing distance reflects a decline in the probability of an individual being detected (Fig. 2.3.1.1B). The proportion of individuals detected, which is identical to the detection probability P_a,may be estimated as the unlined area of Fig. 2.3.1.1C divided by the total area under the horizontal line; a naive estimation of the curve of Fig. 2.3.1.1C could be made e.g. by fitting with least squares the midpoints of each histogram class.

When standardized (i.e., divided by its value at y = 0), this curve is defined as the detection function *g(y)*, which gives the probability of detecting an individual that is at distance y from the line. The focus of the statistical analysis of distance sampling data is the modeling of the detection function *g(y)*.

In line transect surveys, the study area is sampled by placing k randomly positioned lines or a grid of k systematically spaced lines randomly superimposed on the study area (preferably k> 20), each of length l_i. An observer moves along each line, recording the perpendicular distance y_i of every target individual (or cluster of individuals) detected within a distance w from the line. The surveyed area is $A_c = 2wL$, where $L = \sum_{i=1}^{k} l_i$ is the total length of the line transects. Population density and abundance are estimated by $\hat{D} = \frac{n}{2wL\hat{P}_a}$ and $\hat{N} = \frac{nA}{2wL\hat{P}_a}$ respectively, when single individuals are recorded.

When the objects are detected in clusters and a count is also made of the number of individuals (s) in each observed cluster, population density is estimated by $\hat{D} = \frac{n}{2wL\hat{P}_a}\hat{E}(s)$, where $\hat{E}(s)$ is an estimate of the average cluster size in the population. The simplest estimate of the average cluster size is $\bar{s}$. However, as detection may be a function of cluster size, other approaches for estimating $\hat{E}(s)$ have been developed (Buckland et al. 2001).

The detection probability is given by $\hat{P}_a = \frac{\int_0^w \hat{g}(y)dy}{w}$. The integral $\mu = \int_0^w g(y)dy$ is called 'effective strip half-width' and represents the half-width of the strip extending either side of a transect centerline such that as many objects are detected outside the strip as remain undetected within it. Thus, an alternative form of the density estimator is $\hat{D} = \frac{n}{2L\hat{\mu}}$. Noting that the probability density function of the distance data *f(y)* is simply the rescaled detection function *g(y)* so that it integrates to unity, it is implied that $f(y) = g(y)/\mu$, and thus $f(0) = 1/\mu$ (because g(0) = 1, i.e. detection probability of an individual that is on the line is assumed to be 1). Hence, $\hat{D} = \frac{n\hat{f}(0)}{2L}$. Such a density estimator, expressed in terms of a probability density function, is convenient due to the large existing literature on fitting probability density functions (Buckland et al. 2001).

The variance of the estimated population density is well approximated using the delta method (Seber 1982: 7–9) by the equation $\hat{\text{var}}(\hat{D}) = \hat{D}^2\left[\frac{\hat{\text{var}}(n)}{n^2} + \frac{\hat{\text{var}}(\hat{P}_a)}{\hat{P}_a^2}\right]$. The $\hat{\text{var}}(n)$ is generally estimated from the sample variance of the encounter rates n_i/l_i, weighted by the line lengths l_i. When f(0) is estimated by maximum likelihood, its variance is estimated by evaluating the Fisher information matrix (Buckland et al. 2001). Bootstrapping (Efron and Tibshirani 1993) is a good alternative for variance and confidence interval estimations. The simplest procedure is to sample with replacement from the replicate line transects using the nonparametric bootstrap. A model selection criterion, such as AIC, may be used to select the best model among a set of candidate models for each resample and in this way incorporate model selection uncertainty into the estimate of precision, as an alternative to the model-averaging approach proposed by Burnham and Anderson (2002).

In the marine environment, distance sampling techniques have been applied in population surveys of a variety of taxa such as marine mammals (e.g., Hedley and Buckland 2004; Scheidat et al. 2008; Herr et al. 2009), sea birds (e.g., Ronconi and Burger 2009), marine fish (e.g., Thresher and Gunn 1986; Kulbicki and Sarramégna 1999; Issaris et al. 2009), bivalves (e.g., Katsanevakis 2005, 2006a, 2007), and sea anemones (e.g., Katsanevakis and Thessalou-Legaki

2007). Line transect surveys in the marine environment include shipboard, aerial, and underwater surveys with snorkels or SCUBA gear as well as with occupied submersibles (e.g. Yoklavich et al. 2007).

2.3.1.2.2 Assumptions

First of all the survey must be competently designed and conducted, otherwise the estimates may be of little value. Although it is assumed that the population of interest comprises objects (animals or clusters) that are distributed in the study area according to some stochastic process, a Poisson distribution is not assumed. It is, however important that the lines are placed randomly with respect to the distribution of the animals or clusters in order to safely assume that the statistical distribution of distances of objects from the transect line is uniform.

There are three essential assumptions for reliable density or abundance estimations from line transect sampling (Buckland et al. 2001; Thomas et al. 2010). Extensions to the conventional distance sampling methods allow one or more of these assumptions to be relaxed at the expense of some extra effort during the survey (Buckland et al. 2004).

The most critical assumption is that all animals that are located on the line are detected with certainty, i.e., *g*(0)= 1. In practice, detection on or near the line should be nearly certain and survey design must consider ways to assure that this assumption is met (Buckland et al. 2001). Detection probability *g(y)* is assumed to fall off in a smooth manner out to some distance *y* = *w* from the line. When the '*g*(0) = 1' assumption is violated, estimates of abundance are negatively biased in proportion to *g*(0), e.g., if *g*(0) = 0.75, estimated abundance will be on average 75% of true abundance. This assumption is relaxed in mark-recapture distance sampling (Laake and Borchers 2004).
The second basic assumption is that individuals are detected at their original location, prior to any movement in response to the observer (this is particularly important when surveying highly mobile animals such as bentho-pelagic fish or cetaceans). Random movement of animals (independently of the observer) might cause a (usually small) bias of the estimated abundance upwards because moving animals are more likely to be detected when they are close to the line, biasing detection distances down. Movement in response to the observer can cause a large bias in abundance estimation (Fewster et al. 2008). This bias will be positive if the animals are attracted by the observer, and negative if the observer is avoided. If there is undetected movement ahead of the observer and the same animal is recorded several times while traversing a single transect, bias can be quite large.

The third essential assumption is that distance measurements are exact. Although line transect estimators are fairly robust to random measurement errors, they are sensitive to systematic bias in distance measurement such as rounding to zero distance (Buckland et al. 2001; Borchers et al. 2004). Careful measurements with the use of tape lines, laser rangefinders or other means are always preferable than rough estimates by eye, which are often very poor.

2.3.1.2.3 Modelling the detection function

There are three main desired properties for models of the detection function *g(y)*: model robustness, the 'shape criterion', and estimator efficiency (Buckland et al. 2001). Model robustness means (1) that the model is a general and flexible function that can take a variety of plausible shapes, and (2) that data can be pooled over many factors that affect detection probability and still yield a reliable estimate of density or abundance (pooling robustness concept). The desired shape criterion suggests that the detection function *g(y)* has a 'shoulder' near zero distance, which means that detection remains nearly certain at small distances from the line. This is expressed as $g'(0) = 0$. For models that are model robust and have a shoulder near zero distance, efficiency is a desired property, i.e., selecting a model that provides relatively precise estimates.

The detection function is modelled in the general form:
$g(y) = \frac{key(y)[1 + series(y)]}{key(0)[1 + series(0)]}$, where *key(y)* is the key function and *series(y)* is a series expansion used to adjust the key function (Buckland et al. 2001). Three models for the key function are implemented in program DISTANCE that have the desired properties of model robustness, shape criterion, and efficiency: the uniform function

$key(y) = 1/w$ (0 parameters), the one-parameter half-normal function $key(y) = \exp\left(-y^2/2\sigma^2\right)$, and the two-parameter hazard-rate function $key(y) = 1 - \exp\left[-\left(y/\sigma\right)^{-b}\right]$. The cosine series $\sum_{j=1}^{m} a_j \cos\left(j\pi y/w\right)$, simple polynomials of the form $\sum_{j=1}^{m} a_j \left(y/w\right)^{2j}$, and Hermite polynomials of the form $\sum_{j=2}^{m} a_j H_{2j}\left(y/\sigma\right)$ are considered as series adjustments, where σ (scale term) and a_j are best fit parameters (Buckland et al. 2001). Some generally useful models are the uniform key function with a cosine or a simple polynomial series expansion, the half-normal key function with a cosine or a Hermite polynomial series expansion, and the Hazard-rate key function with a cosine or a simple polynomial series expansion. These series-expansion models are nonparametric in the sense that the number of parameters used is data-dependent (Buckland et al. 2001). Typically, an adequate model for *g(y)* will include no more than three parameters and quite often the key function will be adequate with no need for a series expansion. Usually, distance data are truncated prior to analysis (i.e., a distance *y* = *w* is specified beyond which detected objects are ignored) so that outliers that make modelling *g(y)* difficult are deleted; otherwise extra adjustment terms may be needed to fit the long tail of the detection function.

For each dataset, several models should be considered. Model selection may be based on the information theory approach (Burnham and Anderson 2002) and Akaike's Information Criterion (AIC) (Akaike 1973) or the small-sample, bias-corrected form AIC_c (Hurvich and Tsai 1989) of Akaike's criterion (when the ratio of the sample size to the number of model parameters is <20). The number *j* of parameters in each series expansion may also be defined using AIC or AIC_c between models of increasing order. Quite often more than one detection function model is supported by the data. When the data support evidence of more than one model, model averaging the predicted detection function or the estimated parameters (e.g., P_a or μ) across models is advantageous in reaching a robust inference that is not conditional on a single model. This procedure is termed multi-model inference (MMI) and has several theoretical and practical advantages (Burnham and Anderson 2002).

2.3.1.2.4 Multiple Covariate Distance Sampling

Detection probability and the shape of the detection function may depend on several factors other than distance. Provided that animals at zero distance are detected with certainty,distance sampling estimators of population density and abundance are unbiased (or nearly unbiased) due to the 'pooling robustness' property of the detection function models, even though heterogeneity in the detection probabilities due to covariates other than distance is ignored (Marques and Buckland 2004).

However, inference might be improved in some cases by estimating a detection function $g(y,\mathbf{z})$, which apart from distance *y* from the centerline, also depends on a vector of covariates **z**. This might be true when a large component of the variance of the abundance estimate is due to the estimation of the detection function and this variance can be 'explained' by variables other than distance or when detection probability changes across strata but there are inadequate detections in some strata to allow separate estimation of detection probability within each stratum (Borchers and Burnham 2004; Marques and Buckland 2003, 2004).
Marques and Buckland (2003, 2004) proposed that the covariates are incorporated into the estimation of the detection probabilities via the scale term σ (that appears in the half-normal and hazard-rate models), according to the relationship $\sigma = \exp\left(\beta_0 + \sum_{i=1}^{r} \beta_i z_i\right)$. In this formulation the covariates are assumed to affect the rate at which detectability changes as a function of distance. The covariates may relate to the environment (e.g., habitat type, visibility, sea condition), the observer, or the individual detections (e.g., cluster size or individual size), and can be either continuous or qualitative factors. Abundance is estimated by the 'Horvitz-Thompson-like' estimator

$\hat{N}=\sum_{i=1}^{n}\frac{1}{\hat{P}_a(\mathbf{z})}$. Variance of the abundance estimator may be estimated either empirically (Marques and Buckland 2004) or by using the bootstrap (Efron and Tibshirani 1993), taking the line transects as the sampling unit.

2.3.1.2.5 Density Surface Modeling

Significant methodological advances are expected in the near future in the integration of modelling with Geographic Information Systems (GIS) and model-based inference from sample survey data (Buckland et al. 2000; Hedley et al. 2004). Conventional distance sampling is founded on a combination of model-based and design-based inference. The detection function is modelled with a semi-parametric approach, according to Buckland et al. (2001) (see Section 2.3.1.2.3) and model-based inference is drawn within the covered (sampled) area. To extend inference to the entire study area, conventional distance sampling relies on design-based methods (i.e. line transects are placed randomly or based on a systematic grid randomly placed over the study area) ensuring equal coverage probability, i.e. that all portions of the study area have an equal probability of being included in the sample. However, for practical reasons the equal coverage probability assumption is often violated. In such cases, entirely model-based inference for abundance and density estimation in the study area is a good alternative. Furthermore, wildlife managers increasingly wish to extract more than an abundance estimate from their surveys and frequently want to relate animal density to spatial variables, reflecting various factors like topography or habitat (Hedley et al. 2004).

Density surface modelling (based on distance sampling data) aims to developing models that predict the population density of the target-species in a unit area in relation to spatial predictors. A first step towards this direction was made by Hedley (2000) and Hedley and Buckland (2004), who developed methods for improving abundance estimation of cetacean abundance, allowing heterogeneity in the spatial distribution of cetaceans to be modelled from standard line transect data. Two approaches were suggested that enable spatial variation in animal density to be modelled using standard generalized linear modelling (GLM; McCullagh and Nedler 1989) or generalized additive modelling (GAM; Hastie and Tibshirani 1990).In the first approach (count model), the transect lines were divided into smaller discrete units (segments), and the expected number of detections in each unit was modelled using explanatory spatial covariates. In the second approach (waiting distance model), the response was derived from the observed waiting times (or distances) between detections (Hedley 2000; Hedley and Buckland 2004; Hedley et al. 2004). The count model is the one mostly applied in studies of marine animal abundance (e.g., Williams et al. 2006; Katsanevakis 2007, 2009; Katsanevakis and Thessalou-Legaki 2009; Herr et al. 2009) and is presented hereafter.

According to the count model, each transect is divided up into small segments of length $l_{seg,i}$ and area $a_i = 2wl_{seg,i}$. The number of animals n_i detected in each segment *i* is recorded. Additionally, spatial covariates such as longitude, latitude, depth, habitat type, and other spatially referred environmental variables are recorded for each segment. The total number of animals within segment *i* is estimated from the Horvitz-Thompson-like estimator $\hat{n}_i=\sum_{j=1}^{n_i}\frac{1}{\hat{p}_{ij}}$, where $\hat{p}_{ij}$ is the probability of detection of animal *j* in segment *i*, obtained from the fitted model for the detection function. The expected values of n_i can be related to spatial covariates using GLMs or GAMs. The general formulation for a GLM is $f\left(\mathrm{E}[\hat{n}_i]/\alpha_i\right)=c+\sum_m \beta_m z_{mi}+\sum_r F_{ri}$, and for a GAM $f\left(\mathrm{E}[\hat{n}_i]/\alpha_i\right)=c+\sum_m s_m(z_{mi})+\sum_r F_{ri}$, where *f* is the link function, *c* denotes the intercept, β_m is the coefficient for spatial covariate *m*, $s_m(\cdot)$ is the one-dimensional smooth function for spatial covariate *m*, z_{mi} is the value of spatial covariate *m* for segment *i*, F_{ri} are categorical predictors, and $a_i = 2wl_{seg,i}$ is the covered area of the segment. Using the selected model for n_i, predictions can be made for regions of the study area that were not surveyed, provided that there is a known distribution of the spatial covariates that were included in the model. If the whole study area is divided into a prediction grid of sufficiently small cells, the abundance of the target species in the study area may be estimated as the sum of $\mathrm{E}[\hat{n}_r]$ at each cell *r* of the prediction grid, i.e., $\hat{N} = \sum_r \mathrm{E}[\hat{n}_r]$. Based on the predictions $\mathrm{E}[\hat{n}_r]$, a distribution map of the target species in the study area may also be produced.

Density surface modelling (DSM) with GLMs or GAMs employs model-based inference for abundance and density estimation in the study area. With DSM, abundance is related to spatial covariates and the inferred relationships between abundance and ecologically meaningful covariates might have great biological significance and could be of more importance than simple abundance estimation. With DSM, the distribution of the target-species may be visually depicted by easily producing distribution maps. There are several examples of DSM for marine species based on the 'count' model, mainly focusing on marine mammals (e.g., Hedley and Buckland 2004; Williams et al. 2006; Herr et al. 2009) but also on benthic invertebrates (e.g., Katsanevakis 2007, 2009; Katsanevakis and Thessalou-Legaki 2009); these surveys were based on aerial, shipboard, or underwater line transect sampling.

2.3.1.2.6 Study design and field methods

Distance sampling methodology relies on a number of assumptions (see Section 2.3.1.2.2), of which one of the most important ones is a good study design and field methods.

In the study design two points are of particular importance: a) the collection of enough data to be able to calculate a detection function and to achieve a desired level of precision and b) the placement of the sample units (e.g. lines). We will provide a brief overview of these two points, but further details on how to design a distance sampling survey can be found in Buckland et al. (2001, 2004). For reliable estimation of the detection function the absolute minimum sample size (e.g. sightings of groups of whales) should be 60 – 80 and more if added variance in the sample is to be expected (e.g. when having a stratified survey area). The total line length of a survey depends on the expected encounter rate of the target species and the precision required from the survey. This information is best obtained by pilot studies or derived from existing research (literature). The placement of the sample units, such as transect lines, is also very important for a successful survey. At least 10 to 20 replicate lines should be surveyed to provide a basis for an adequate variance of the encounter rate and a reasonable number of degrees of freedom for constructing confidence intervals. Additionally, lines should be placed using some form of randomization (e.g. random parallel transect lines, randomly chosen starting point of a systematic grid of lines). It is important that all portions of the study area have an equal probability of being included in the sample ("equal coverage probability"). In some cases the study area can be stratified into several smaller areas, for example if abundance estimates are required for different habitat types or management blocks. Smaller areas might be easier to survey, but stratification is most useful if the sub-areas differ considerably in density. The actual configuration and orientation of the transect lines depends on the survey region, logistics and efficiency as well as knowledge of density gradients of patterns in the survey region. In marine line transect surveys the most typical layouts are parallel or zigzag transects. In a parallel design the survey effort on the transit between two lines should generally not be used as this would cause unequal coverage probability at the edges of the study area. In aerial surveys (see chapter 1.2.3a, section 2.7.2) these transits can be used for observer rest or rotation. More efficient, in terms of using the survey time, are zigzag (or saw-tooth) designs as they allow continuous data collection. To ensure equal coverage probability the software program 'DISTANCE'(Thomas et al. 2010) has an option for creating such a design. To avoid obtaining an unrepresentative sample, transect placement should not be parallel to a physical or biological feature (e.g. depth contours). In addition, they should follow any potential density gradients of the target animals to avoid a high variance in encounter rate for the replicated lines.

The use of good field methods include the application of a survey protocol, the defining of searching behaviour as well as accurate data measurement and recording, including correct determination of cluster size, species, behaviour and other parameters. In the following sections study design and field methods for the application of distance sampling methodology will be described particularly forshipboard, aerial and underwater surveys.

2.3.1.2.6.1 Shipboard surveys

Shipboard surveys probably are the most-often used tool to make at-sea inventories of distribution patterns and densities of marine mammals and seabirds. Ships can stay at sea longer than aircraft, and operate in remote areas. On the other hand, (large) ships are more expensive to operate than planes and cover less ground per unit of time.

Ships were first used, and still are to a large extent, as platforms of opportunity. As ships sail almost anywhere at sea anyway, and often have some extra space for researchers, many ships have been used across the globe to make first inventories of marine animals and seabirds at sea. Many such "piggy back" surveys have yielded interesting results, but usually should not be considered monitoring because of their unstructured nature. However, with sufficient time, effort and endurance, whole seas or whole national jurisdictional areas may be studied during longer periods and several atlases of marine mammal or seabird distribution patterns throughout the year have been produced from databases collected from ships of opportunity (e.g. Camphuysen and Leopold 1994; Durinck et al. 1994; Skov et al. 1995; Stone et al. 1995; Offringa et al. 1996; White et al. 2001; Reid et al. 2003a, 2003b). Admittedly, most of the extensive databases underlying these atlases are only partly based on ships of opportunity surveys, and partly on so-called dedicated surveys in which ships are hired for specific survey goals.

Dedicated surveys are expensive and mostly used for several specific tasks: filling in gaps (e.g. for mapping work for atlases); targeting animals that are difficult to survey, such as harbour porpoises (Hammondet al. 2002; Hammond 2006); targetting specific areas, such as offshore windfarm sites or larger areas where such sites may become operational in the future (Camphuysen et al. 2004; Garthe and Hüppop 2004; Leopold et al. 2004, 2009; Leopold and Camphuysen 2009; Vanermen and Stienen 2009). Note, however, that such exercises can also be done by plane (e.g. Dierschke et al. 2006; Petersen et al. 2006; Petersen and Fox 2007; Gilles et al. 2009).

Both marine mammals and seabirds may move in response to a moving vessel, which violates one of the assumptions of distance sampling (see Section 2.3.1.2.2). Field protocols should try to avoid such responsive movement, e.g. by using high-powered binoculars to detect animals before the vessel is close and triggers their response. Marine mammals are only visible when they are at the surface and thus for many species the critical assumption of certain detectability at zero distance (g(0)=1) is violated. This assumption is relaxed in mark-recapture distance sampling, which is commonly applied for marine mammals (Laake and Borchers 2004).

2.3.1.2.6.2 Aerial surveys

Aerial surveys applying distance sampling methodology have been used for a variety of different marine species, including jelly fish (Houghton et al. 2006a), fish (Blaylock 1993; Houghton et al. 2006 a, b), sea turtles (Goméz de Segura et al. 2003, 2006), seabirds (e.g. Briggs et al. 1985) and marine mammals (e.g. Bonnell and Ford 1987; Hammond et al. 2002). Many different aircraft types have been used, including planes (Hammond et al. 2002), helicopters (e.g. Southwell 2005), microlights (Rowat et al. 2009; Jean et al. 2010) and blimps (Hain et al. 1999). To guarantee successful aerial surveys it is important to consider the special circumstances of aerial surveys and to make sure that the assumptions of distance sampling theory are not violated.

One of the main assumptions is that all animals on the transect line are detected (g(0) = 1) (for more detail see Section 2.3.1.2.2). In reality, a number of factors can influence this probability. The two main points are the availability bias (animals are there, but are not visible) and the perception bias (animals are there, but are missed by the observer). Some species spend most of their time close to the surface (e.g. Cliff 2007) or occur in shallow areas and, providing water clarity is good enough, can be recorded while they are under water (e.g. Pollock et al. 2006). However, most marine species spend a considerable amount of time further underwater, thus not visible to an observer (not "available"). The perception bias can be reduced by training the observers to focus their search effort on the area around the transect line (Buckland et al. 2001) and to take sufficient breaks to avoid observer fatigue. Several approaches have been taken to consider availability and perception bias and to estimate g(0) in aerial surveys. Although the g(0) = 1 assumption is crucial for accurate density estimates, only few studies using standard distance sampling have estimated g(0) in the field; most of those are focused on cetaceans (Bächler and Liechti 2007). To determine the availability bias, Barlow et al. (1988) used breath rate data to estimate the time harbour porpoises were at the surface. Laake et al. (1997) tracked harbour porpoises from land concurrently with aerial surveys to obtain an overall g(0) value. Palka (1995, 2005) used a combination of aerial and shipboard surveys to estimate g(0) values for the same species. In some studies information on availability of animals can be derived from time depth recorders or telemetry data (Westgate et al. 1995; Thomson et al. 2006). Perception bias can be investigated by using two independent observer teams in the aircraft (Thomson et al. 2006). For harbour porpoises the circle-back method has been developed by Hiby (1999), which is an adaptation of the tandem flights (Hiby and Lowell 1998; Hammond et al. 2002) and has been applied in European aerial surveys for porpoises (Hammond et al. 2002, Scheidat et al. 2004).

In shipboard surveys animal movement prior to detection can be a problem, in particular for species that are either approaching or avoiding the vessel (e.g. Au and Perryman 1982; Turnock and Quinn 1991; Hyrenbach 2001; Palka and Hammond 2001; Clarke et al. 2003; Lloyd et al. 2003). In aerial surveys this is generally not a concern because of the speed of the survey platform; nevertheless it is useful to make sure that the aircraft noise is as low as possible to limit possible impact on the animals.

It is also important to make sure that during aerial surveys visibility of the transect line from an aircraft is not restricted. The most common solution is to use high winged aircraft and bubble windows that allow an unrestricted view of the transect line. If the transect line is not visible some analytical solutions are available, such as the left truncation of the detection function (Buckland et al. 2001).

Another aim during a survey is to have a large enough sample size for distance sampling analyses. Despite the fact that aircrafts can cover large areas in a short time period, which is helpful when trying to use periods of good weather and get fast coverage of an extensive survey area, airplane minimum speed is generally quite high (80 to 100 knots) which limits the time for detecting and identifying animals. The height of the aircraft also impacts the sighting probability of the target species. It is important to consider that a higher survey height gives observers more time to scan the transect area. However, the perceived size of the animals is reduced and potential cues for identifying sightings or species is reduced, making them easier to miss. Therefore the optimal survey height can range widely depending on the species. For example, blue whales have been successfully surveyed at a height of 500m (Hucke-Gaete et al. 2003), whereas seabirds have been successfully surveyed at a height of 60m (e.g. Henkel et al. 2007).

A further important assumption of distance sampling is that distances are measured exactly. In aerial surveys the distances are ideally measured when the animals are perpendicular to the transect line. With a known survey height this can be done by measuring the vertical angle to the sighting and calculating the distance post-survey. Distances can also be measured in intervals, using specific angle intervals, for example by placing marks on the window (e.g. Bengtson et al. 1995). It is also possible to interrupt the survey, fly to the sighting and take a direct measurement of the position (and thus distance to the transect line) with a GPS.

Throughout the aerial survey information on environmental conditions are recorded, including any condition that could impact the sighting probability of an animal (e.g. sea state, glare). Information recorded during a sighting can include species, cluster size, behaviour and group composition. For the distance sampling analyses it is of particular importance to record the cluster size accurately. Cluster sizes can be underestimated if the sighting occurs at a larger distance. When using a helicopter or airplane, surveys can be interrupted to approach such a sighting. Photographs can be taken to get accurate cluster size estimates and also to help identify species.

Data recording should be done in a way that it does not distract the observers from searching. Options are the use of an additional person for data recording or continuous or voice activated audio recording, which then has to be transcribed at a later time. In any case, a clearly defined survey protocol and thorough training of the observer team is necessary to ensure consistent quality of the survey. Finally, when planning distance sampling aerial surveys safety should be considered, in particular in remote areas, and training in emergency procedures should be given.

2.3.1.1.6.3 Underwater surveys

Underwater distance sampling surveys in the marine environment have been conducted to study populations of fish (Thresher and Gunn 1986; Letourner et al. 1998, 2000; Arnal et al. 1999; Kulbicki and Sarramegna 1999; Kulbicki et al. 2007; Preuss et al. 2009; Issaris et al. 2009), marine bivalves (Katsanevakis 2005, 2006a, 2007; Katsanevakis and Thessalou-Legaki 2009), and sea anemones (Katsanevakis and Thessalou-Legaki 2007).

In line transect surveys by SCUBA diving, a nylon line with distance marks that is deployed using a diving reel seems the most efficient way to define the transect line. Another option might be not to physically define the line but to move along an imaginary line using a compass or following depth contours. When no real line is deployed, the length of the transect could be estimated by marking entrance and exit points with a GPS or by GPS-recording the entire transect. Using an underwater scooter would increase the length of the transect that could be surveyed within the time limits of a dive, which might be of importance for sparsely distributed species.

Although not deploying a real line with a diving reel would save diving time (the extra time to deploy and reel back the line), a possible problem would be that distances from the line would not be recorded accurately. It is possible that a large percentage of detected individuals would be recorded as on the line, i.e., at $y = 0$. This is because the line is not sufficiently defined and when the observer detects an object within a few cm from the (imaginary) line, he cannot accurately measure its distance and so records it as being on the line. Modelling such distance data is problematic and would give unreliable estimations of density and abundance. When the line is not marked with the use of a diving reel, an effort should be made in the survey protocol to avoid bias due to heaping at zero distance. One option would be to have an assistant diver swimming along the transect using a compass to follow the (imaginary) line and towing a rope (Katsanevakis 2009). The observer could swim behind and measure the distance of any detected object from the moving rope.

Accurate measurement of distance should be a principal goal of the sampling protocol. Biased distance measurements such as ‘heaping’, i.e., rounding to convenient values, will lead to biased estimates of density and abundance. Distances should be measured directly and ocular estimation should be avoided. The range of measured distances greatly depends on the visibility in the surveyed region. In underwater line transect surveys, the measured distance are usually in the order of a few centimeters or a few meters. A meter rule, a measuring tape, or a graded plastic rod could be used to measure distances directly.

Visual estimation of distances invites heaping of distances and/or biased estimates of distance. Underwater, the bias in visual estimates of distance is expected to be greater in relation to land surveys due to distance distortion (Katsanevakis 2009). Visually perceived distortions of size and distance are produced by the diver's facemask, which introduces an air-water interface between the eye and the object of regard. The effect of this interface is that the diver sees a virtual image, which is optically located at about 3/4 of its physical distance, i.e., the object’s apparent distance is decreased by about one-fourth. A negative bias in distance measurements will result in a positive bias in density.

The most important assumption of distance sampling is that $g(0) = 1$ and detectability is about 100% certain ‘near’ the line and falls off smoothly at greater distances. Thus, the observer must try to optimize the detection of individuals in the vicinity of the line. If there is a pair of divers/observers, each diver could search independently on one side of the line to increase overall efficiency. When surveying sessile or slow-moving species, the best way to satisfy the ‘$g(0) = 1$’ assumption, is by swimming slowly just above the line and preferably moving stepwise in small length increments than continuously (Katsanevakis 2009). If it is not possible to satisfy the ‘$g(0) = 1$’ assumption by applying adequate field protocols (e.g., when surveying very cryptic species) then the mark recapture distance samplingmethod (Laake and Borchers 2004) should be applied.

When surveying mobile species such as fish, the diver should be careful to satisfy the second important assumption of distance sampling: that individuals are detected at their original location, prior to any movement in response to the observer. To satisfy this assumption, the diver should move along the line but instead of searching below, he should focus forward and near the line, trying both to detect individuals that are in front of him before they move and to ensure that $g(0) = 1$. He should also move relatively fast (in comparison to surveys of sessile species), ideally faster than the speed of the target species. The observer should be careful to locate the point of first sighting of each individual and to measure the perpendicular distance from that point to the line. Depending on the conditions and the surveyed species, it might be useful to move at 1-2 m above the sea floor (and the line) in order to increase the field of view. To reduce the range over which individuals might respond to the observer, the use of a rebreather (closed-circuit SCUBA) instead of an open-circuit breathing set would be beneficial. Rebreathers produce far fewer bubbles and make less noise than open circuit breathing devices, thus making it possible to reduce the distance beyond which marine animals respond to the observer.

2.3.1.3 Occupancy Estimation and Modelling

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2.3.1.3.1 Overview

When monitoring animal populations, the most commonly used state variables are population density or abundance. However, estimation of density or abundance is often costly and requires substantial effort (e.g., Lancia et al. 1994; Pollock et al. 2002; MacKenzie et al. 2006) or may be unfeasible for various reasons (e.g., in the case of rare or very cryptic species). Alternatively, species occupancy, defined as the proportion of area, patches, or sampling units occupied (or alternatively as the probability of presence in a sampling unit) may be seen as a low-cost surrogate of abundance. Moreover, there are cases when occupancy is the appropriate state variable and would be chosen in the first place. Occupancy would be the state variable of choice in studies of distribution and range (Scott et al. 2002), alien invasions (Issaris et al. 2010), metapopulation studies (Moilanen 2002), community studies (Martinez-Solano et al. 2003; Weber et al. 2004), and large-scale monitoring (Hall and Langtimm 2001; Manley et al. 2004).

By detecting a species at a sampling unit, its presence is confirmed with certainty. However, the non-detection of the species may result either from the species being genuinely absent, or from the species being present at the site but undetected during the survey (MacKenzie et al. 2006). Hence, the true absence of a species from a sampling unit is often impossible to be inferred solely from presence/absence data. 'False absences' may lead to incorrect inferences about the system and erroneous management decisions if the imperfect detection of the species is not accounted for; occupancy is underestimated, colonization rates are biased, and habitat relationships may well be misleading, particularly if detectability also changes with different habitats (Moilanen 2002; Tyre et al. 2003; Gu and Swihart 2004; MacKenzie 2005; MacKenzie et al. 2006).

To address this issue, a set of methods that permit inference about occupancy based on presence-absence data while taking into account the imperfect detection of the target species have been developed (Geissler and Fuller 1987; Azuma et al. 1990; Nichols and Karanth 2002; MacKenzie et al. 2002, 2003, 2005, 2006; Dorazio et al. 2005). All methods involve multiple visits to each site, where the target species is either detected, with probability p, or not detected, with probability $1-p$. The goal is to estimate the proportion of sites that are occupied, ψ, accepting that the species is not always detected.

Under a general sampling scheme, s number of sites are surveyed for a K number of times for the target species. For every survey, the appropriate methods (visual, aural, indirect traces, etc.) are used to detect the presence of at least one individual of the species. It is assumed that the target species is not falsely detected because of misidentification or other reasons. The sites are closed to the changes in occupancy during the survey, which means that no new sites are becoming occupied after the survey has started and no sites are abandoned until the survey is over. A species may or may not be detected when present at a site and its detection is assumed to be independent of detecting the species at other sites. The resulting presence/absence data for each site surveyed is a sequence of detections (1) and non-detections (0) that form a detection history H_i of the target species at each site. The set of these detection histories is used to estimate the proportion of sites occupied by the species (MacKenzie et al. 2002).

A presentation of some of the available *ad hoc* methods of estimation as well as the detailed description of a more robust and general model-based approach is provided in the book by MacKenzie et al.(2006), which can be considered standard literature for occupancy estimation and modeling. With the latter approach, occupancy ψ is jointly modeled with probability of detection p. In the more general and widely applicable model-based approach, models are built on straightforward probabilistic arguments describing the underlying processes that may have caused a given detection history to be observed (MacKenzie 2005). Covariates that affect ψ and p are incorporated in the models. The model parameters are estimated with standard maximum likelihood techniques.Researchers can develop a set of candidate models to fit to the data, each representing a different hypothesis about the system under study. The various alternative hypotheses can be challenged following a model selection approach based on information theory (Burnham and Anderson 2002).

Detection rates often vary considerably among species, owing to their different characteristics and behaviour. Different observers may have quite dissimilar capability of detecting a species, and the observer-effect should normally be considered. In the marine environment several factors may affect detectability such as water turbidity, habitat structure, weather conditions, and season. Moreover, the probability of detecting the species in a location is often a direct function of the species abundance. As a consequence, occupancy will go undetected more frequently for rare than for abundant species (MacKenzie et al. 2006).

2.3.1.3.2 Single-Species, Single-Season Occupancy Models

On the basis of the conceptual model, there are two stochastic processes occurring that affect the outcome of whether the species is detected at a site (MacKenzie et al. 2006). A site might be either occupied (with probability ψ) or unoccupied (with probability 1- ψ) by the species. If the site is unoccupied, the species will not be detected. If the site is occupied, at each survey j the species will either be detected (with probability p_j) or pass undetected (with probability q_j = 1-p_j).

The series of detections and non-detections for each site is recorded as a sequence of 1s and 0s respectively (a detection history). For example, consider the detection history H_i=101 (denoting site i was surveyed three times, with the species being detected in the first and third surveys). The probability of this history would be Pr(H_i=101) = $\psi p_1 q_2 p_3$. For sites where the species is never detected there are two possibilities, either the species is present but never detected (a 'false absence') or the species is genuinely absent. Thus, Pr(H_i=000) = $\psi q_1 q_2 q_3$ + (1-ψ). By deriving such expressions for each of the s observed detection histories, assuming independent observations, the likelihood of the data will be $L(\psi, \mathbf{p} \mid H_1, H_2, ..., H_s) = \prod_{i=1}^{s} \Pr(H_i)$, where **p** is the vector of detection probabilities.

The potential relationships between the model parameters (occupancy and detection probabilities) and characteristics of the sites (e.g., habitat type, depth, slope) or environmental (e.g., annual average surface temperature, current intensity, water turbidity) and geographical variables (longitude, latitude) may be investigated. Covariates are incorporated by using the logistic model $\theta_i = \exp(\mathbf{Y_i \beta}) \cdot (1 + \exp(\mathbf{Y_i \beta}))^{-1}$, where ϑ_i is the probability of interest (occupancy or detection probability), $\mathbf{Y_i}$ are the covariates to be modelled, and $\boldsymbol{\beta}$ denotes the vector of the covariate coefficients to be estimated (MacKenzie et al. 2006). Standard maximum likelihood techniques are applied to obtain estimates of the model parameters.

An information theory approach may be followed for inference. According to the information theory approach, data analysis is assumed to be the integrated process of *a priori* specification of a set of candidate models (based on the science of the problem), model selection based on the principle of parsimony according to Akaike Information Criterion (AIC; Akaike 1973), and the estimation of parameters and their precision (Burnham and Anderson 2002). The principle of parsimony implies the selection of a model with the smallest possible number of parameters for adequate representation of the data, a bias versus variance tradeoff. Furthermore, rather than estimating parameters from only the 'best' model, parameters (i.e., occupancy and detection probabilities) can be estimated from several or even all the models considered. This procedure is termed multi-model inference (MMI) and has several theoretical and practical advantages (Burnham and Anderson 2002; Katsanevakis 2006b).

2.3.1.3.3 Assessing Model Fit

To assess the goodness-of-fit (GOF) of the models, MacKenzie and Bailey (2004) proposed a procedure under which a simple Pearson chi-square statistic is calculated and a parametric bootstrap procedure is used to determine whether the observed statistic is unusually large. Hence the overdispersion parameter ĉ is estimated so that in situations where even the most global model (the model with all parameters) is found to be a poor fit of the data, the estimate of ĉ may be used as a variance inflation factor to adjust model selection procedures and standard errors (Burnham and Anderson, 2002). Steps of the overall procedure are detailed in MacKenzie and Bailey (2004).

Let O_h be the number of sites observed to have detection history *h*, and E_h be the expected number of sites with history *h* according to the current model. Generally, E_h equates to the sum of the estimated probabilities of observing *h*

$$E_h = \sum_{i=1}^{N} \Pr(X_i = h)$$

These probabilities may be site-specific depending upon the model that has been fit to the data.

Assuming no missing observations, there are 2^T possible detection histories that may be observed. The test statistic for assessing the fit of the model can be calculated as

$$X^2 = \sum_{h=1}^{2^T} \frac{(O_h - E_h)^2}{E_h}$$

The parametric bootstrap is used to determine whether the observed value is unusually large. Assuming that the fitted model is correct it is implemented as follows:

1. Fit model to the observed data and estimate parameters $\hat{\psi}_i$ and $\hat{p}_{ij}$ (which may be functions of covariates);
2. calculate the test statistic for the observed data, X^2_{Obs}, using the model fit in Step 1;
3. for each site generate a pseudo-random number (*r*) between 0 and 1. If $r \le \hat{\psi}$, then the site is occupied, hence generate *T* further pseudo-random numbers (r_j) between 0 and 1. If $r_j \le \hat{p}_{ij}$, then the species was "detected" and the corresponding bootstrapped observation is a "1", otherwise "0". If $r > \hat{\psi}_i$, then the site is unoccupied and the bootstrapped observations will all be "0" for that site;
4. fit a model with the same structure as in Step 1 to the bootstrapped dataset.
5. calculate the test statistic for the bootstrapped data, X^2_B, using the model fit in Step 4, and store the result;
6. repeat Steps 3–5 a large number of times to approximate the distribution of the test statistic, given the fitted model is correct.
7. Compare X^2_{Obs} to the values of X^2_B to determine the probability of observing a larger value (the *p* value).

Hence, the overdispersion parameter ĉ may be estimated as suggested by White et al. (2001):

$$\hat{c} = X^2_{Obs} \Big/ \overline{X}^2_B$$

where $\overline{X}^2_B$ is the average of the test statistics obtained from the parametric bootstrap.

If ĉ = 1, then no overdispersion occurs and the model adequately fits the data. If ĉ exceeds 1, then there is indication of overdispersion (there is more variation in the observed data than expected by the model); values < 1 (which may suggest underdispersion) or> 4 often hint inadequate model structure.

The estimate of ĉ should be computed only for the global model; it is generally assumed that if the global model fits the data adequately, then a reduced parameter model will also fit the data because it originates from the global model (Burnham and Anderson 2002).

2.3.1.3.3 Violation of Model Assumptions

There are four essential assumptions for reliable occupancy estimation using the above modelling approach (MacKenzie et al. 2006):

1) Sites are ‘closed’ to changes in occupancy during the survey season, i.e. occupancy status remains constant. This may be reasonable over a relatively short time interval (e.g., within a single year), but is unlikely to hold for longer studies (i.e., across multiple years).
2) ψ is either constant across sites or it is appropriately modelled with covariates Y_i.
3) *p* is constant across all sites and surveys or is appropriately modelled using site and/or survey covariates.
4) Detection of species and detection histories are independent across sites.

If the above assumptions are not met, some or all estimators may be biased and inferences regarding the factors that influence occupancy and its dynamics as well as the detection of species might be false. The impact of the violation of each one of the above assumptions has not yet been thoroughly investigated; insights however can be extracted from violations of similar assumptions in the context of mark-recapture population studies (MacKenzie et al. 2006).

In the case of violations of the closure assumption, it is expected that the occupancy estimator will be unbiased if species move in and out of the sampling unit in a random way, although occupancy will now refer to proportions of sites “used” by the target species (MacKenzie 2005). However, if movement in and out of the sampling unit is not random, occupancy will be biased (i.e. immigration movement, when sites are becoming occupied during the course of the survey). The bias is less predictable if a model with constant *p* is used; allowing detection probabilities to vary among surveys absorbs some of the effect of emigration and immigration. In fact, the systematic decrease or increase of detection probability estimates during a season may indicate that emigration or immigration is taking place respectively (MacKenzie et al. 2006).
In the case of unmodelled occupancy heterogeneity (variation of occupancy probability ψ among sites), the bias is relatively unknown compared to other model assumptions and more simulation studies are required.

Unmodelled heterogeneity in detection probability *p* generally leads to negatively biased occupancy estimations. Low detection probabilities coupled with large variations (among sites or surveys) tend to increase the bias (Royle and Nichols 2003). Bias is further exaggerated in studies involving a small number of sites or a few repeated surveys at each site (MacKenzie et al. 2006).

Finally, violation of the independence of detections produces overstated precision of occupancy estimates. Non-independence is a common issue at studies where no adequate spatial separation exists between sites and animals’ movement causes multiple simultaneous detections. In that case, the number of independent sites or detection histories (“effective sample size”) is smaller than the number of sites surveyed and the estimated standard errors obtained from the model are too small (MacKenzie and Bailey 2004).

2.3.1.3.4 Survey Design

The ultimate goal of an occupancy survey is to achieve a desired level of precision in the estimation of occupancy with minimal effort. Therefore, the number of repeated surveys that is required to be conducted emerges as a key aspect of designing occupancy studies. It has been proved that when detection probabilities are high, it is more efficient to survey more sampling units (sites) than increase the number of surveys per sampling unit, but as the detection probability decreases, more surveys per site are necessary (Tyre et al. 2003). MacKenzie and Royle (2005) as well as MacKenzie et al. (2006) provide specific advice on the number of repeat surveys per sampling unit considering the variance of the occupancy estimator under specific sampling schemes.

The two main probabilistic sampling schemes that could be designed are: (a) random selection of sites to survey from throughout the entire population of available sites (region); (b) stratification of the region (i.e considering habitat type) and random selection of the sites only from the strata of interest.

Repeated surveys of the sites can be conducted as multiple discrete visits (e.g. on different days); however, discrete visits may not always be necessary. Other options include conducting multiple surveys within a single visit; using multiple observers to conduct independent surveys, either on the same or a different visit; surveying multiple plots within a larger site on a single visit. The decision of which approach is most practical depends upon the study objective, whether the model assumptions are likely to be satisfied given the biology of the target species, and the logistical considerations of sampling (MacKenzie and Royle 2005). In a recent application in the marine environment, Issaris et al. (2010) studied range and habitat relations of benthic alien species and used multiple observers (free divers) conducting independent time-limited surveys on the same visit at each site to estimate occupancy status.

Regarding the time available for an occupancy study, it is reasonable to opt for completing the survey of all sites as fast as possible to ensure the closure of the target species' population to changes in occupancy status; it is generally our intent to provide a snapshot of the system at a given point in time.

2.3.1.3.5 Multiple Season Occupancy Models

In metapopulation studies or species-habitat studies, the rate of change often has a greater importance than the absolute value of the occupancy state. Multiple season occupancy models have been developed to provide estimates of rates of change or "trends" in occupancy of target species, allowing the investigation of the effects of environmental variables and management actions by incorporating proper covariates, as described in detail in the relative chapter of the book by MacKenzie et al (2006).

Two approaches to modelling the detection/non-detection data exist, each with different assumptions regarding the processes that govern the target species population dynamics. In the implicit dynamic model approach, the probability that the target species does not go extinct locally at a previously occupied site is equal to the probability of colonization of a previously unoccupied site. Under this approach, a single season model is effectively applied to the data collected in each season, regardless of the underlying processes of changes in occupancy. In the second approach, the models therein explicitly account for the processes of colonization and local extinction by considering that the probability of a site being occupied in season *t* depends upon the occupancy state of the site in the previous season *t-1* (first-order Markov process occupancy state dependence) (MacKenzie et al 2006).

2.3.1.3.6 Further extensions

The occupancy models have also been extended to investigate whether the occupancy of one species influences the occupancy of a second species (i.e. co-occurrence). This development is detailed in MacKenzie et al. (2004). The paper by Royle and Nichols (2003) investigates the relationship between 'patch' or 'site' level detection probabilities and individual detection probabilities. If a site-level detection probability can be obtained and an individual detection probability can be modelled, a latent estimate of abundance can be obtained from detection/apparent absence data.

More recent research focuses on incorporating both false positive and false negative observations into the estimation (Royle and Link 2006), dealing with heterogeneity in detection probabilities (Royle 2006) and applying these methods to estimating species richness (MacKenzie et al. 2006; Dorazio et al. 2006). Nichols et al. (2007) extend these methods to a large class of questions that seek to classify sites by different categories of occupancy (e.g. to classify occupied sites by whether young animals are observed at the sites).

Nichols et al. (2008) developed occupancy models for multiple detection methods that permit simultaneous use of data from all methods in order to gain information about method-specific detection probabilities. Moreover, the approach permits estimation of occupancy at two spatial scales: the larger scale corresponds to species' use of a sample unit, whereas the smaller scale corresponds to presence of the species at the local sample station or site.

2.3.1.3.7 Software tools

Occupancy models can be fitted with the freeware software PRESENCE (Hines and MacKenzie 2004), MARK (White and Burnham 1999), and GenPres (Bailey et al. 2007).

PRESENCE (http://137.227.242.23/software/presence.html) is a software that has been primarily developed to fit occupancy models to detection/non-detection data. This software was developed to enable estimation of the proportion of area occupied, or similarly the probability that a site is occupied, by a species of interest according to the model presented by MacKenzie et al. (2002). Currently, there are nine types of models that can be fit to detection/non-detection data within Program PRESENCE, including single-season, single-species, multiple seasons, multiple species and multi-state models. A user can fit multiple models to their data set and PRESENCE stores the results for each model and presents a summary of how well the models rank according to a model selection metric (AIC is used as the default). Models are fit using maximum likelihood techniques, hence parameter estimates are known as maximum likelihood estimates.

MARK
(http://www.phidot.org/software/mark/docs/book/ or http://warnercnr.colostate.edu/~gwhite/mark/mark.htm),developed by Gary White (White and Burnham 1999; White et al. 2001), is a Windows-based program for capture-recapture, occupancy modeling and other types of analysis. Occupancy models implemented in Mark include MacKenzie et al. model (2002), plus a robust-design extension, (MacKenzie et al. 2003). In addition, the single-season occupancy model of Royle and Nichols (2003), plus some extensions, have been implemented. Other occupancy models include the multi-site occupancy model (Nichols et al. 2008), and single-season and multi-season occupancy models with multiple states and state uncertainty (Nichols et al. 2007; MacKenzie et al. 2009).

The program **GENPRES** is written in C and RAPIDQ (a visual BASIC variant) and can be downloaded from the Patuxent Wildlife Research Center website (http://www.mbr-pwrc.usgs.gov/software).
This program simulates presence/absence data to be used as input to programs MARK or PRESENCE. It can be used to get an idea of how precise the estimates are for a given sample effort or design. Additionally, it reports the bias of estimates if heterogeneity is an important factor. Nine model-types are available in GENPRES.
Input information required for program GENPRES includes the generating model type (single- or multi-season) and structure, parameter values, analysis model(s), and evaluation method (either simulations or expected values). The greatest value of program GENPRES is its flexibility to examine a wide variety of design options, tailored to a given biological system, and subject to economic constraints; thus, giving investigators an extremely useful tool during the critical planning phase of a study.

Another resource for self-help in Occupancy modelling is the on-line publication http://www.uvm.edu/envnr/vtcfwru/spreadsheets/occupancy/occupancy.htm) by Donovan and Hines (2007). The exercises presented in the e-book are designed as a teaching tool to explore the occupancy models described by MacKenzie et al. (2006). Each exercise consists of a spreadsheet application, followed by an analysis in programs PRESENCE, MARK, or GenPres.

2.3.1.4 Mark-recapture techniques

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2.3.1.4.1 Introduction

Mark and recapture (MR) is a common technique used to estimate the size of populations, to study movements and migration of individuals and to provide information on birth, death and growth rates of species (Krebs 1999).

MR methods are based on capturing and marking individuals from a population and then resample the same population to count the number of marked and unmarked individuals. The size of the entire population can be estimated from the proportions of marked and unmarked individuals. This procedure was first applied by Petersen on marine fishes and by Lincoln on waterfowl populations, and it is often referred to as the Lincoln-Peterson (LP) method (Krebs 1999). The main assumption of the LP method was that the population does not change in size between mark and recapture measurements. This single marking and recapture procedure has also been the base for the developing of other methods based on multiple MR samples (Southwood 1978).

The development of sophisticated mark-recapture models over the last four decades has provided fundamental tools for more accurate estimates of the size of wildlife populations. Moreover, historically the field implementation to estimate population size in animals has led to a wide variety of indices proposed by numerous authors (reviewed by Southwood 1978, and Krebs 1999). Such indices attempt to correct potential biases in estimating size population of the several sampled taxa.

MR has been used to study the populations of species belonging to different taxa such as molluscs, mammals, crustaceans, echinoderms, fishes, turtles and birds.

Despite assumptions and statistical aspects, the success of MR technique is also related to tools and equipment used for capturing, marking and recapturing the specimens of a given population. In the following sections a review of the main assumptions, models, tools and limitations related to the MR methods is reported.

2.3.1.4.2 General assumptions of the mark-recapture methods

Mark-recapture methods have been widely used for estimating the size of animal populations, but such techniques are valid only under certain restrictive assumptions which can be summarise in four points (Southwood 1978):

1) Animals are not affected by marks.
   This means that survival, catchability, ability to migrate, reproductive ability in the time interval between capture and recapture are all unaffected by the marks. Many references dealt with various marking methods applied to animals (Seber 1982; Southwood 1978; Nielsen 1992; Sutherland 1996, see also section 4.3). These authors explained that there is no perfect mark, since all tags may interfere with the animal's life cycle.

2) There is no change in the ratio between marked and unmarked animals during the interval between samplings.
   This means that during the interval between marking and the subsequent recapture period, nothing has happened to imbalance the proportions of marked to unmarked animals, that is, no new individuals were born or immigrated into the population, none died or emigrated, and no mark was lost or overlooked by observers. One of the fundamental assumptions in mark-recapture studies is that no tag is lost. The latter assumption is fundamental in MR studies and, if violated, parameter estimates are biased. Studies on tag detection highlighted lost tagging in several species and its effect on the estimation of population parameters (Cowen and Schwarz 2006; De Graaf 2007; Vu and Kohlhorst 2003). Tag-induced mortality and tag loss reduces the size of the marked population. Experiments under controlled conditions have been carried out to assess tag induced mortality (Green et al. 1985; Ludwig et al. 1990) and the rate of tag loss (Montgomery and Brett 1996; Sánchez-Lamadrid 2001). The most common method to estimate rates of tag loss in the field is double tagging experiments (Barrowman and Myers 1996; Reñones and Goñi 2000).

3) All animals have the same chance of getting caught; the catchability of all individuals is equal.
   This assumption has two aspects: firstly, all individuals of different age groups and of both sexes are sampled in the proportions in which they occur; secondly, all individuals are equally available for capture irrespective of their location in the habitat. Moreover, the chances for each individual to be caught must remain constant during the marking and recapture period. This means, marked individuals must not become either easier or more difficult to catch (individuals don't change their behaviour in response to sampling equipment). Furthermore, time variations (daily, seasonal) must not affect catchability. Gilbert et al. (2001) reported evidence from MR experiments that tagging induced trap shyness in snapper. They observed that tagged fish had a reduced probability of recapture by the method by which they had originally been caught. Equal catchability is the most important assumption of the MR models and is discussed in more statistical detail in section 4.2.

4) The marked animals are homogeneously distributed among the population.
   This means that the time spent between the first marking and the recapture period must allow all marked individuals to disperse homogeneously throughout the population.However, the time period must not be so long as to allow for a significant increase by immigration or reproduction. This assumption is true if there is uniform

mixing of marked and unmarked animals and all individuals are catchable. However, uniform mixing is unlikely among most animals thus, where possible, a random sampling should be adopted in MR studies (Kendall et al. 2009).

The above-mentioned assumptions are often violated in natural populations,which leads to biased estimations of population size. If this is the case, tests must therefore be carried out to reveal to which degree the assumptions are met and, where possible, to make appropriate allowances if they do not. The second assumption is relaxed in the Joly-Seber method (see below), which was designed for open populations and does not assume the absence of recruitment and mortality.

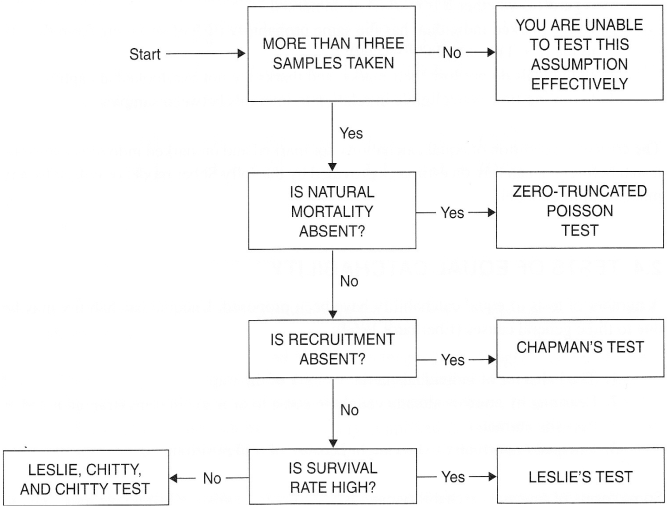
2.3.1.4.3 Statistic context and models

MR methods are used to monitor open and closed populations. In a closed population, no mortality, birth, or migration takes place, so that the basic assumption is that the population does not change between mark and recapture times. An open population changes in size composition from births, deaths and movements hereby violating assumption 2.Therefore, different methods must be applied to estimate the size of open and closed populations. The Lincoln-Peterson, Schnabel and Schumacher-Eschmeyer methods are mainly employed for closed populations while the Jolly-Seber method is mainly applied to open populations (Krebs, 1989).

Test of equal catchability

The most critical assumption for all MR models, which must be tested *a priori* to sampling in order to ensure good estimates, is the equal catchability for marked and unmarked individuals (Krebs 1999). Eberhardt (1969) reported that unequal catchability may be mainly due to modified behaviour of individuals in the proximity of the trap, to learning by animals (trap-addicted or trap-shy animals) and to trap positions.

There are a number of decisions that have been developed to test the assumption of equal catchability (Fig. 2.3.1.3; Krebs 1999). One of the most efficient tests of equal catchability is the Zero-truncated Poisson Test. It can be used if there are four or more trapping occasions and if the period between trapping occasions is short enough to ensure little or no mortality. Krebs (1999) provides a detailed explanation of Zero-truncated Poisson test which is based on an analysis of the capture frequency of animals in the population. Marten (1970) developed a regression method for MR estimation of population size with unequal catchability.



Closed populations

The Lincoln-Peterson (LP) model is based on a single episode of marking, and a second, single episode of recapturing individuals. The basic principle is, that if a proportion of the population was marked in some way, returned to the original population and then, after complete mixing, a second sample was taken, the proportion of marked individuals in the second sample would be the same as was marked initially in the total population. That is, R (marked recaptures) / C (total captures in second sample)=M (marked initially) / N (total pop. size). Therefore $\hat{N} = \frac{CM}{R}$ where $\hat{N}$ is the estimateof population size at time of marking.

The assumptions of the LP method are that the population size is constant, without recruitment or losses, that sampling is random, and that all individuals have an equal chance of capture in a given sample. As LP formula tends to overestimate the population and is based on sampling without replacement, several other formulas have been suggested to reduce the bias (Bailey 1952; Seber 1992). Several techniques of obtaining confidence intervals and accuracy for LP estimates are available (see Seber 1982; Robson and Reiger 1964; Pollock et al. 1990; Zar 1996).
The Schnabel method is an extension of the LP model to more than 2 capture occasions. It treats the multiple samples as a series of LP samples, and a population estimate is obtained as a weighted average of LP estimates.This method works well in closed populations such as the fish in a lake. The Schumacher-Eschmeyer method can also be used with more than two capture occasions. As for the LP estimates, several statistics have been applied to reduce the estimation biases. The assumptions of the Schnabel and Schumacher-Eschmeyer methods are the same as for LP. The main advantage of these multiple sampling is the use of linear regression techniques to obtain an estimate of population size but only when assumptions are true (Krebs 1999).
The program CAPTURE is the most general approach for analyses of mark-recapture data for closed populations. The program calculates population estimates using a variety of different models and then chooses the one that "fits" best. This approach works well if population size and probability of capture are both high.

Open populations

The Jolly-Seber method considers a multiple capture-recapture survey in which there is the possibility of a gain in population numbers through recruitment or immigration, or a loss in population numbers through death or emigration. This method requires that more than two captures are done and that the marks, applied on one date, are

different from those applied at another time. Marks identify not only that it has been captured previously, but also when it was first captured. As animals are tagged individually, data on movement can also be obtained simultaneously with a population estimate.
The samples are usually point samples of short duration, and separated by a long duration from the next sample. The time interval between samples does not need to be constant, and any number of samples can be accommodated, so that a series of data extending over many years can be used in this method.

The assumptions of the Jolly-Seber method are: (i) probability of capture in the t^{th} sample is the same for all animals (marked and unmarked), (ii) probability of survival from *t* to *t+1* is the same for all marked animals, (iii) no errors occur in identification of individuals and no tag losses and (iv) sampling time is negligible in relation to the interval between samples, i.e. the population size does not change during the sampling event.Besides the population size, the Jolly-Seber method provides also estimates for the probability of survival and of the recruitment (dilution) rate *sensu* Jolly (1965). However, some sources of variation affect the estimates from the Jolly-Seber method. Therefore several formulas are needed to correct the biases (Krebs, 1999) and to obtain reliable confidence limits.

Some statistical approaches and software (such as MARK) have been developed to provide parameter estimates from marked animals when they are re-encountered at a later time (White et al. 1999). Such programs enable also to plan *robust design* recommended for MR long-term studies (Kendall 1999; Kendall and Hines 1999; Dupuis and Schwarz 2007).

2.3.1.4.4 Limitations

Mark-recapture techniques are a powerful technique to estimate population size and to gain information on population dynamics and patterns of species movement. However, it is only accurate under certain assumptions (see section 4.1) and MR methods suffer several limitations.

1) Despite the statistical approaches to correct the bias in estimating population size, experiments have to be carried out to evaluate some of the MR assumptions. As no ideal tag or mark has been produced to date (Lucas and Bares 2000), the choice of mark/tag and of the marking/tagging procedure should be tested before starting MR studies (Ludwig et al. 1990; Wallin et al. 1997; Montgomery et 1995; Sánchez-Lamadrid 2001). This sampling design is important to the quality of the study, and should therefore be undertaken with care, even though it is expensive and time-consuming.
2) Dispersal and migration of tagged animals are often underestimated because of spatially limited recapture effort. Information is obtained obviously only from recaptured animals, not from missing individuals which moved outside the sampling area or died.
3) The tag reporting rate is generally low, and reports are obtained mainly from professional or recreational fishermen. Since the estimation of the reporting rate is important for the quality of MR studies, a greater effort should be made to optimise the knowledge about recapture reporting rates, as well as about the position and date of recovery (Reñones and Goñi 2000).
4) Movement pattern can only be estimated on the basis of a straight distance between the recaptures and the release sites. No information about the animal’s behaviour between the two capturing events, either spatially or temporally, can be obtained between (Lucas and Baras 2000).
5) The sampling calendar can be delayed or interrupted by bad weather conditions due to seasonality, or fish can change their dispersal pattern in relation of uncontrolled environmental features (sea current, temperature, food availability, etc…) resulting in a lack of spatial information.

2.3.1.4.5 Tools used for mark-recapture application

The success of the MR technique depends also on tools and equipment used for capturing, marking and recapturing the specimens of a studied marine population. Generally, MR experiments involve a sequence of operations, such as capture, handling and holding of the animals, marking process, the release of marked individuals, the subsequent recapture of the animal and the detection or recovery of the mark.

Capture methods

Capture methods are strongly dependent on the aims of the study so that the equipment has to be accurately chosen according to: the size of individuals, where the animal lives (open water, rocky habitat, sandy bottom, etc…), when it is active or inactive (i.e. looking for food, moving between shelter and refuge, staying quite inside refuge, etc.) (Nielsen, 1992). Nonetheless, the sampling equipment has to be designed to collect specimens alive and in good health thus reducing any stress prior to the marking or tagging operation.

Common capture equipment includes:

- Netting, nest invasion, live trapping and sedation are mainly used to catch marine birds and mammals. Highly-selective tools to collect fish in good condition are: traps, purse seine, hook and line, long-line and electrofishing (Rodney 1979; Cowx 1990)
- Baited traps can be used to collect benthic fish and crustaceans like lobster and crabs.
- Purse seines are useful to encircle fish shoals, concentrate the fish in a small bunt of netting close to the vessel, and catch individuals by means of a scoop net.
- Hook and line can be used to catch large fish such as tuna or shark.
- A long-line can be a high-selective method depending on the kind of bait used. It is often applied to collect nectobenthonic fish.
- Electrofishing is becoming on the most popular methods of catching fish in shallow fresh waters (streams, rivers and lake shores) (Paukert 2004; Schoenebeck and Hansen 2005).

In several cases, the use of fishing equipment requires the application of recovery procedures. Captured fishes often suffer barotraumas or injuries on the buccal and pharyngeal apparatus because of the hook. In these cases, it's necessary to punch the swim-bladder and to remove, where this is possible, the hook from the mouth of the fish (Nostvik and Pedersen 1999)

Mark and tag choice

The appropriate tag depends on the aims of the study, the size and species to tagged, the expected time between realise and recapture and whether the tag is likely to be recovered by fishers or fisheries scientists (Table 2.3.1.3).

Especially with endangered species or brood stock and trophy animals, the tag must be as non-intrusive as possible. In order to reduce the stress by capture and handling, underwater tagging methods have been developed (Matthews and Reavis 1990; Davies 1995).

The choice of the most suitable mark/tag is depending on:

- Aim and duration of the study
- Number and size of target individuals
- Potential effect of mark/tag on animal growth and behaviour
- Potential effect of mark/tag on animal predation and vulnerability to fishing gear
- Cost of marking/tagging technique

Based on these considerations we can choose among the available marking and tagging techniques.

Marks typically infer alterations to the animal's appearance that enable the animal to be identified externally. The most common marks are: tattoos, fin clipping (partial amputation of fins), pigments, shell notches or dyes. These techniques are useful in short-term studies and restricted geographic areas. For example, tattoos allow the marking of groups of individuals by adopting different tattoo schemes. Unfortunately, the efficacy of a tattoo is strongly dependent on colour pattern of individuals and how this pattern can change as fish grow up. In recent research, such marking methods are used less frequently.

Tags, internal or external, are physical devices attached to the animals' body. They sometimes protrude out of the skin and are easily visible, even underwater i.e. from Visible Implant Elastomer tag suitable for small fishes (<10 cm) to the

Petersen disc or T-bar tag which can be univocally coded or numbered and applied to large individuals (McFarlane et al. 1990). This kind of tag can be carried by animals for months due to a very low tag loss rate. The type of external tag must be carefully chosen, according for the shape of the individuals, their swimming speed and the habitat where they live (algae can grow up on the tag). Among internal tags, the coded wire microtag (a magnetized stainless steel part) has been widely employed to investigate marine populations (Nielsen 1992). When the animal is recaptured, tag can be removed and the data downloaded (Nielsen 1992). In recent years, electronic tags (data storage tags) have also been used to collect information on some environmental parameters, that a tagged individual experiences.

Recapture of tagged animals is usually performed according to a specific sampling design using a research vessel and/or based on fishermen's returns, often rewarded, of tagged animals. Either way, the marks or tags subsequently have to be dispatched and interpreted. External tags are easily to read and decode, but miniaturized tags such as elastomers, and marks as tattoos and fin clipping can be difficult to interpret if they are old and word.

An additional way to supplement recapture results is to track individuals in the field using underwater visual census techniques. Also scuba divers can easily recognize external tags applied to fish or crustaceans and retrieve important information about their location, behaviour and activity on a very precise scale (Davies 1995; D'Anna and Pipitone, 2000).

2.3.1.5. Removal and Catch-Effort Methods

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2.3.1.5.1. Introduction

The main objective of these methods is the estimation of the population density and abundance of animal populations.

Sampling methodologies provide the scientific foundation for gathering the required data, while statistical models allow for inferences from these data sets (*estimation*). Finally, statistical robustness is evaluated on the basis of the models' predictive power and the acceptable limits of error around the estimate (*uncertainty*).
Some commonly used methods to estimate animal abundance are:

- Removal method
- Change-in-ratio method
- Catch-effort method

All methods require at least two samplings (surveys) and are based on the following concept: changes in detection rate, following removal of a number of animals, provide valuable data that allows for estimation of the population size.

In brief:

Removal Method

A number n_1 of animals is removed (captured, detected) on a first survey, and an additional number n_2 is removed on a second survey. The reduction between the two surveys ($n_1 - n_2$) provides valuable information for the estimation of the true population (N).

Change-in-ratio method

This method depends on the presence of observed differences in the variables at the animal level (size, sex, etc.) If there are two types of organisms in a population (e.g.: males-females; juveniles-adults) and surveying results in the removal of one type compared to the other, the calculated *change-in-ratio* allows for estimation of population size.

Catch-effort method

If sampling effort is positively correlated to sampling size, then the proportion of the population removed will be higher if the effort is greater. The method extends the Removal method and is generally applied in harvesting. If *catch-per-unit-effort* (CPUE) declines with time and this decline is linear, then regressing accumulated removals upon CPUE allows for estimation of the population size at the beginning of exploitation (initial or pristine or un-exploited population).

2.3.1.5.2. Statistical context

Notations:
N = populations size or abundance
$\hat{N}$ = estimator of N
n = number of animals removed (detected, sampled)
p = probability of detecting an animal

Detection probability:
The proportion of animals that would be detected, after a large number of surveys, is called the detection probability *p*.

In the simplest case: $\hat{N} = \frac{n}{p}$

p is limited to values between 0 and 1. These values depend on:

- the difficulty of detecting an animal,
- how much effort we invest on looking for an animal,
- external ambient conditions
- animal behaviour

Likelihood:
In the general approach we must find an expression of $\hat{N}$ which will be the closest to N. This is done by estimating the maximum likelihood of N, using a mathematical formula called the likelihood function.

The idea is to imagine the population as a series of N independent trials. A trial is a 'success' when an animal is detected, and a 'failure' when it is missed. These conditions set the scene for applying the binomial distribution. The probability density function gives the probability of *n* successes, out of a sample of N, occurring with probability *p*:

$$L(N \mid n, p) = \binom{N}{n} p^{n} (1-p)^{N-n} \tag{1}$$

Solving this likelihood function in order to find its maximum, leads us to the Maximum Likelihood Estimator (MLE) which is, our ultimate goal. The formulation of this likelihood function depends on the specific method applied.

Uncertainty

The MLE of any population parameter under study is a random variable. This implies that it may vary significantly among surveys. The largest the variation, the less confident we are about our estimator.
To quantify this, we need a measure of uncertainty around our estimates. *Confidence intervals* (CI) include the range of values around which the true population parameter lies *k*% of the time. *k* is usually a pre-set probability of 90, 95,

99%. CI's allows for evaluation of our estimators, usually derived from the estimators' variance (or standard error). Wide CI's imply poor approximation, while narrow CI's indicate a good approximation.

Removal Method

Recall the concept: A number n_1 of animals is removed (captured, detected) on a first survey, and an additional number n_2 is removed on a second survey. The reduction between the two surveys (n_1 -n_2) provides valuable information allowing for estimating the true population (*N*).

N: true population
S: survey occasions
n_s: animals detected (removed) on occasion *s*
R_s: total animals removed before occasion *s*
p: detection probability

Assumptions:

- Population is closed
- all animals are detected (removed) with same probability p
- detection events are independent
- detections are independent between occasions (surveys)
- p is the same on all occasions (surveys)

The likelihood function takes the following form:

$$L(N \mid n, p) = \begin{pmatrix} N \\ \\ n \end{pmatrix} p^{n} (1-p)^{N-n} \tag{2}$$

where: {n,R}=(n_1, R_1), ..., (n_s, R_s)^2, N $\equiv N_1$, R_1= 0

Maximizing this function we obtain the MLE of N ($\hat{N}$).
For a two sample survey:

$$\hat{N} = \frac{n_1^2}{n_1 - n_2}, \quad \hat{p} = \frac{n_1 - n_2}{n_1}$$

CI estimation depends on the population size, sample size~~s~~ and the detection probability *p*. For large samples and large values of *p*, assuming a normal distribution for $\hat{N}$ and $\hat{p}$ a satisfactory CI approximation is achieved using the variance estimators (*q* = 1-*p*):

$$Var[\hat{N}] = \frac{N(1-q^S)q^S}{(1-q^S)^2 - (pS)^2 q^{S-1}} \tag{3}$$

$$Var[\hat{p}] = \frac{(qp)^2 (1-q^S)}{[N[q(1-q^S)]^2 - (pS)^2 q^S]} \tag{4}$$

In any other case, more complicated approaches may perform better (profile likelihood, bootstrapping) (Borchers et al. 2002).

Change-in-ratio method

Recall the concept: The method depends on the presence of observed differences in animal-level variables (size, sex, etc.) If there are two types of organisms in a population (e.g.: males-females; juveniles-adults) and surveying removes more of one type than the other, the calculated *change-in-ratio* allows for estimation of population size.

x: level variable (e.g.: sex)
N: true population
S: survey occasions
$N_s(x)$: number of animals of level *x* in the population before survey occasion *s*
N_s: population size before survey occasion *s*
n_s: sample size on occasion *s*
$R_s(x)$: number of animals of level *x* removed before survey occasion *s*
R_s: total number of animals removed before survey occasion *s*

Assumptions:

- Population is closed
- the two types of animals are equally catchable (detection probability does not depend on animal-level variable)
- effort put into each survey is not taken in account

Applying the methodology on two samples (*s* = 1, 2) with level variable *sex* (m: males; f: females) we obtain:
Proportion of males in the population just after the first survey (sample) is

$p_2(m) = N_2(m)/N_2$or

$$p_2(m) = \frac{p_1(m)N_1 - R_2(m)}{N_1 - R_2} \Rightarrow N = \frac{R_2(m) - R_2 p_2(m)}{p_1(m) - p_2(m)}$$

Based on the assumption that both sexes are equally catchable:

$$\hat{p}_1(m) = \frac{n_1(m)}{n_1}, \quad \hat{p}_2(m) = \frac{n_2(m)}{n_2}$$

$$\hat{N} = \frac{R_2(m) - R_2 \hat{p}_2(m)}{\hat{p}_1(m) - \hat{p}_2(m)} \tag{5}$$

Assuming that *N* is normally distributed, we can calculate CI's from the variance estimators:

$$\hat{V}ar[\hat{N}] = \frac{\sum_{s=1}^{2} \hat{N}_s^2 \hat{V}ar[\hat{p}_s(m)]}{(\hat{p}_1(m) - \hat{p}_2(m))^2} \tag{6}$$

$$\hat{V}ar[\hat{p}_S(m)] = \frac{\hat{p}_S(m)(1-\hat{p}_S(m))}{(n_S-1)}[1-\frac{n_S}{\hat{N}_S}] \qquad (7)$$

In the absence of normality for *N*, likelihood profile or non-parametric bootstrapping may not perform well in this method, due to the assumption of equal catchability among animal-levels (Borchers et al. 2002). Refer to Pollock et al. (1987) for a more complex alternative dealing with unequal catchability.

Catch-Effort method

Recall the concept: If more effort is put into removing animals, we would expect to remove a higher proportion of the population when we use more effort. The method extends the Removal method and is generally applied in harvesting. If catch-per-unit-effort (CPUE) declines with time and this decline is linear, then regressing accumulated removals upon CPUE allows for estimating the population size at the beginning of exploitation (pristine or un-exploited population).

c_i: catch (individuals removed at sample *i*)
K_i: accumulated catch just before sample *i*
f_i: effort used to obtain sample *i*
F_i: accumulated effort just before sample *i*
CPUE: catch-per-unit-effort c_i/f_i
$\overline{K}$: mean $K_i = \frac{\sum K_i}{s}$
s :total number of samples (*i*=1,2,3, ..., *s*)

Assumptions:

- Population is closed
- probability of an animal to be caught depends on the level of effort exerted
- all individuals have equal probability of being caught in sample *i*

The method is applied on exploited populations. However, it is efficient only if removals are significant enough to cause a detectable decline in the CPUE. Under all the aforementioned assumptions, CPUE is proportional to the actual population size. Regressing accumulated catch (K_i) upon CPUE (c_i/f_i) gives us an estimate of population size and catchability (see Fig. 2.3.1.4).

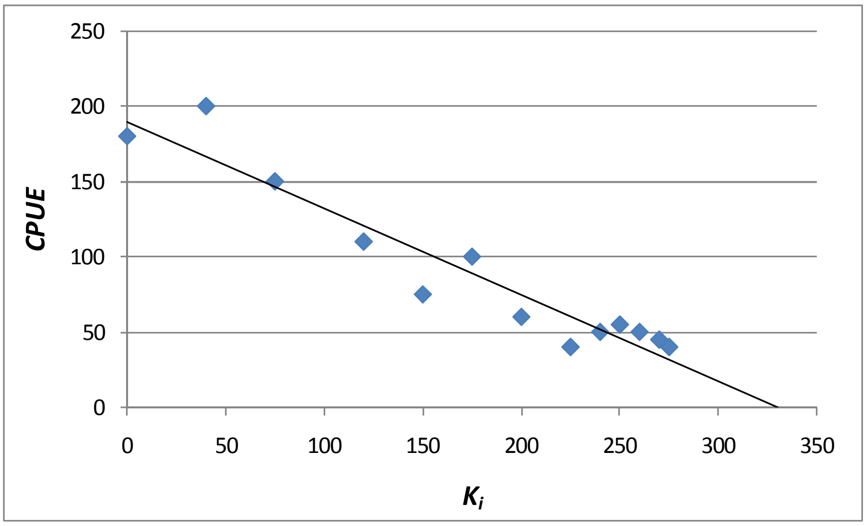

Fig. 2.3.1.4: Example plot of catch-effort data (source: Fischler (1965))

Population size ($\hat{N}$) is given by the x-axis intercept of the regression line, while the probability of an animal being caught in one unit of effort is given by the slope (*catchability*) (Krebs 1999).

The likelihood approach notation becomes:
n_s: number of individuals sampled in sample *s*
l_s : effort exerted on sample *s*
CPUE: n_s/l_s

We need a model to express detection probability as a function of effort. A simple form would be:

$$p(l_s) = 1 - e^{-\theta\, l_s} \tag{8}$$

The unknown parameter θ is estimated by fitting the function to observed data points.

The likelihood function extends the Removal method described previously, with probability $p(l_s)$ being expressed as a function of effort (l_s):

$$L(N, \underline{\theta} \mid \{n, R\}) = \prod_{s=1}^{S} \binom{N_s}{n_s} p(l_s)^{n_s} (1 - p(l_s))^{N_s - n_s} \tag{9}$$

However, functional forms for $p(l_s)$ are numerous and depend on the assumptions of the way/ method animals are detected. The above formula (8) is widely used in fisheries science (Borchers et al. 2002). In this method, MLE calculation of population parameters requires excessive computations and was not applicable until recently in ordinary desktop computers. Some approaches requiring less computational effort have been proposed by Seber (1982).

Catch-effort methods have been applied in harvested populations for decades, especially fisheries, although some of them are known with a variety of names:

- Biomass dynamic models (Schaefer 1954),
- Catch-At-Age methods
- Virtual Population Analysis(Gulland 1965)
- Doubleday's method (Doubleday 1976)

Hilborn and Walters (1992) provide a comprehensive introduction on these methods while Quinn and Deriso (1999) deal with the technicalities of the statistical approaches.

2.3.1.5.3 Limitations

The real world is too complex to model in a finite series of formulas. The most common way to approach nature is by creating simple abstract models incorporating key aspects of the problem under solution. Incomprehensive complexities are left out, till scientific knowledge allows us to include them in our models.

Population estimation methods could not deviate from this general rule, and all aforementioned techniques assume ideal conditions when studying a population. Some of the fundamental assumptions driving all of these methods are:

- Number of removals (detections) is known with certainty.

This is not always the case. Bias or uncertainty in observations will always cause serious concern. Observers' abilities (e.g.: experience, visual acuity, zeal) may vary significantly and the assumption of '*all individuals have equal probability of being detected* on *sample i'* may be violated.

- The population is closed

This is the most overwhelming assumption made herein. The reasons for making use of it, is the extreme complexity arising in the models if we do not. In general, populations are replenished either through reproduction, or through adjacent migrating populations (transient or permanent). Seber (1982) discusses a number of more complex methods dealing with open populations.

- Heterogeneity is ignored

The assumption of '*all animals are equally detectable*' does not take into account animal behaviour. Most populations are *heterogeneous*, meaning that not all individuals act in a similar predictable way. Put in statistical terminology, '*catchability*' is not constant among different animal-levels. E.g.:

- Larger sized animals are more easily detected than smaller ones;
- Adults frequently inhabit different areas than juveniles;
- Males and females may not be equally represented in a population;
- Seasonal migrations (due to reproduction, wintering or feeding) may give dissimilar outputs for surveys conducted in different time periods; etc…

As a result the most 'detectable' animals are seen first, while 'undetected' animals boost their proportions in the remaining population, resulting in biased estimations.
Otis et al. (1978) and Scharwz and Seber (1999) reason on how to model heterogeneity in 'general removal' models, incorporating animal-level stratification.

CPUE from commercial surveys may give large biases if changes in '*catchability*' over time are not taken in account. There are documented cases where although CPUE remained fairly unchanged, the population was rapidly declining. This phenomenon is the nightmare of every manager and it is called '*hyperstability*' (Hilborn and Walters 1992). Harvesting techniques become gradually more efficient and are capable of catching more animals applying the same unit of effort. These new techniques may involve technological improvements or increase of experience gained through time. Whilst the former may be easy to detect, the latter is extremely difficult to quantify and model. E.g.: over time, an experienced fisherman may figure out the patchy distribution of some commercial fish populations and direct his effort towards certain areas of high abundance. In this case, officially monitored catches, when analysed, will falsely depict a stable population status, due to the failure to incorporate in the model spatial changes in effort and catchability.

2.3.1.5.4. Tools

- ***MARK-RECAPTURE*** is a group of programs developed by C.J. Krebs in Delphi and Fortran to run on desktop PC's. Among others, it covers Catch-effort models and change-in-ratio method. It can be purchased through Exeter Software (http://www.exetersoftware.com/cat/ecometh/ecomethodology.html).

- ***WiSp*** is a package of functions to be used in conjunction with R software (http://www.r-project.org/). This software allows one to investigate the behaviour of a variety of sampling schemes and estimators of population density and abundance. The package can be downloaded from the above R site or from the RUWPA contract-funded research group (http://www.ruwpa.st-and.ac.uk/estimating.abundance/WiSP/index.html). It is based on the work of Zucchini et al. (2007). It covers removal, catch-effort and change-in-ratio methods through a series of functions (e.g. point estimates are given through: *point.est.rm*(); *point.sim.ce*(); *point.sim.cir*())

- Specifically for catch-effort methods on fisheries, there is a large number of available softwares. The most recent one covering a wide variety of statistical approaches is the free access ***FLR*** library (http://flr-project.org/) in conjunction with R software. It contains a series of packages, among others:
  - FLCore Core package of FLR, fisheries modelling in R
  - FLAssess: methods for stock assessment models
  - FLHCR Harvest Control Rules

  The FLR library is still under development by researchers across a number of laboratories and universities in various countries (http://flr-project.org/index.php?q=node/13).

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2.3.2 Marine Mammals

2.3.2.1 General

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The term ‘marine mammals’ describes a very diverse group of taxa, including cetaceans, pinnipeds, sirenians, otariids and polar bears. Some of these animals spend their entire life cycle in the water (e.g. cetaceans and sirenians), whereas othersspend most or part of their life cycle on land or ice (e.g. polar bears, pinnipeds).

Effective management and conservation of marine mammal populations requires reliable information on population size and spatial and temporal changes in abundance. Absolute abundance estimates for marine mammals are often difficult to obtain and therefore relative abundance indices may be used.

Compared to many other animal groups, the monitoring of marine mammal abundance is particularly difficult. They are generally only visible when they come to the surface to breathe, or when they haul out on ice or land. Some species are distributed over large areas and migrate long distances. Marine mammal population sizes can be estimated by a variety of techniques. For absolute abundance estimates, the main methods are distance sampling, mark-recapture, migration counts and colony counts. Other approaches, providing mostly relative abundance estimates, are acoustic methods such as towed or stationary hydrophones.

2.3.2.1.1. Distance sampling

is the most widely used technique for estimating the abundance of animal populations (Buckland et al. 1993, 2001). The primary method of distance sampling applied to marine mammals is line transect sampling (e.g. Marsh 1995; Hammond et al. 2002; Southwell et al. 2005) and cue counting, a method specifically developed for populations of large whales (Buckland et al. 2001).

2.3.2.1.2. Mark-recapture

can be used on marine mammal species that can be identified individually (or marked individually) and that ideally aggregate in specific locations each year. For Polar bears the approach of capture, mark (e.g. ear marks and/or tattoos), release and recapture have been used (e.g. Gilbert 1976). A different method to identify individuals is to take photographs of natural markings (Hammond et al. 1986) and use those for mark recapture calculations. For example, humpback whales can be identified by the unique patterns of their flukes and they aggregate annually in their reproduction of feeding grounds. Many bottlenose dolphin populations worldwide are resident in coastal areas and can be identified by the differences in their dorsal fins (e.g. Wilson et al. 1999). Pinnipeds, such as grey seals, are individually identified by there head-patterns (e.g. Hiby and Lovell 1990).

2.3.2.1.3. Migration counts

are applied to large whales that migrate past coastal watch points where observers can record the animals. Correction factors then need to be applied for that proportion of the population passing outside watch hours, that are not detected, or that pass outside the watch area. This has been done mainly for gray whales off California (e.g. Buckland et al. 1993) and Bowhead whales off Alaska (e.g. George et al. 2004).

2.3.2.1.4. Colony counts

are mainly applied to populations of pinnipeds that aggregate at certain periods of the year for breeding and pupping. In most cases only a proportion of the overall population will be present, but serial colony counts can be used to determine the rate of increase, e.g. of pups, to determine the time when same proportion of animals is present each year at the colony when the counts are made. Counts can be made directly of the colony (e.g. from a cliff top), by taking aerial photographs (including thermal imagery) (e.g. Burn et al. 2006) or by using a ship following the coastline (e.g. McCann and Rothery 1988). This method has also been used on Sea Otters (Udevitz et al. 1995).

In the last decades the use of acoustic monitoring methods of marine mammals has increased. Visual surveys are dependent on and influenced by sufficient daylight, adequate weather conditions and the visibility of animals (e.g. because they are not diving). Passive acoustic methods can overcome some of these limitations and might have a higher detection rate than visual surveys (e.g. McDonald and Moore 2002; Barlow and Taylor 2005). Passive acoustics are either applied as a mobile or a fixed system. Hydrophones can be towed behind a ship, the advantages being that it can be combined with visual surveys and cover large areas (e.g. Fristrup and Clark 1997). Fixed passive acoustics can be left in an area for a prolonged period of time and data will not be affected by the presence of a towing vessel or its noise. In European waters both towed hydrophone arrays (Gillespie and Chappell 2002; Gordon et al. 1998) and stationary passive acoustic monitoring systems are used to investigate the occurrence of harbour porpoises. The development of methods to translate acoustic activity to absolute abundance estimates is under way (e.g. Tougaard et al. 2006). For other species, several studies have used passive acoustics to estimate the absolute abundance of cetaceans (McDonald and Fox 1999; van Parijs et al. 2002).

2.3.2.2 Phocid seal survey techniques♦

Contributor: Gerry Sutton (UCC Cork)

2.3.2.2.1 Aerial, boat or land based counts at seal haul-out sites during moulting or breeding seasons:

a. Population estimates of harbour seals

The standard methodology across the harbour seals global geographical range for estimating population size is via fixed-wing, occasionally helicopters, aerial surveys of haul-out sites during the pupping or molting periods when a larger fraction of the population of seals are hauled out (Heide-Jørgensen and Härkönen, 1988; Thompson and Harwood, 1990; Reijnders et al*.,* 1997, 2003; Frost et al*.,* 1999; Huber et al., 2001; Jeffries et al*.*, 2003). While breeding season counts provide reliable estimates of abundance as well as valuable pup production data, Härkönen et al*.* (1999) concluded that in non-stable age-structured populations the influence of the differential haul-out behaviour on estimating abundance is likely to be greater during the breeding period than during the moult period. Reijnders et al.

♦ The techniques described pertain primarily to grey seal (*Halichoerus grypus*) and harbour seal (*Phoca vitulina vitulina*), the two most common species occurring in European waters and not all the techniques are applicable to monitoring other phocid seals that occur in more northern European latitudes e.g. bearded, harp and ringed seals and in Mediterranean waters e.g. monk seal

(2003) recommended future use of moult count data to obtain a reliable and consistent index of population abundance of harbour seals in the Wadden Sea, while Thompson et al*.* (1997) suggest that counts made during the August moult provided more reliable population estimates for harbour seals hauling out on rocky shores in the UK. Large-scale surveys of harbour seal populations occurring in rocky-shore habitats in the northeast Atlantic and northeast Pacific are generally conducted during the annual moult (Reijnders et al*.,* 1997; Huber et al*.,* 2001; Small et al., 2001, Boveng et al., 2003; Duck et al., 2005).

Conventional aerial photography of harbour seal haul-out groups is effectively used to obtain seal counts on sandy or muddy haul-out substrate and is used at many haul-out sites throughout the species range e.g. the Wash and surrounds in the UK, East and West coast US, Canada, Scandinavia and Holland (Reijnders et al*.,* 1997; Härkönen et al*.,* 2002; SCOS, 2005; Gilbert and Guldager, 1998). Hauled out seals are counted visually or from reading photographs taken with 35mm still or digital cameras with 70-300mm telephoto lens and high speed color slide film. On rocky or seaweed covered rocks harbour seals are difficult to detect. Thermal imaging provides a means of detecting otherwise well-camouflaged seals on rocky or seaweed-dominated shores as well as on sand or mud-banks. This technique has been used to survey the Scottish coast for harbour seals since 1988 (Hiby et al., 1993, 1996) and was adopted for the harbour seal survey of Northern Ireland in 2002 (Duck and Thompson, 2003) and the Republic of Ireland in 2003 (Cronin et al., 2007). Since the thermal imaging camera operates in the infra-red spectrum, it is not influenced by light conditions and seal haul-outs can easily be detected from distances of up to 3km (Duck, C., SMRU, *pers. comm*.) minimising disturbance to the animals. The technology also enables the detection of the heat-shadow or thermal footprint of animals that have entered the water, improving the accuracy of aerial-counts over those conducted by eye. Biases due to differences in land-based observer ability are avoided and any errors in aerial survey data (e.g. misidentification of harbour seals in mixed species groups) can be assessed by ground-truthing at a number of accessible sites. The use of a helicopter allows for maximum area coverage in a short period of time thereby reducing the manpower required to conduct such an extensive survey. In addition, aerial surveys can operate in certain weather conditions that would impede boat surveys such as moderate to strong wind and high sea swell.

The population estimate obtained during a survey can only be considered a minimum population estimate as a fraction of the population will be at sea and not available for counting. Minimum population estimates are sufficient for assessing long-term population trends, however an assumption must be made that the proportion of animals at sea during the count does not vary between years or geographical areas (Thompson and Harwood, 1990). Alternatively, the proportion of the population at sea during surveys can be estimated (by e.g. telemetry) and the count corrected to obtain an estimate of 'absolute abundance'. Such estimates are necessary for incorporation into ecological models and assessing predation pressure by seals on commercially important fish stocks.

Determining the variation in harbour seal haul-out behaviour over time and what factors influence this allows the approximation of what proportion of the population is ashore during counts. This information can be used to devise a correction factor for counts at haul-out sites, improving the accuracy of population estimates. A variety of approaches have been used to estimate this proportion, including telemetry (Yochem et al., 1987; Thompson et al., 1989, 1997; Ries et al., 1998; Huber et al., 2001; Simkpins et al., 2003; Sharples, 2005), a bounded count method (Olesiuk et al., 1990), time lapse photography (Stewart, 1984; Thompson and Harwood, 1990) and photo-identification of individuals (Moran, 2004).

b. Population estimates of grey seals

There are two possible strategies for estimating the number of grey seals in a population or sub-population: (1) systematic sampling (e.g. mark-recapture estimation) or (2) direct counting of a definable population cohort and extrapolation to population size based on the age-structure of the population and age-specific fecundity rates (Harwood and Prime, 1978). These strategies are conducted during the breeding season and the cohort that is counted is pups of the year. This is done by ground sampling at the haul-out sites (using mark-recapture to estimate pup production) or aerial techniques (when there is a wide geographical area to cover) and digital imagery of breeding colony obtained and pups counted from images in the laboratory. A series of counts throughout the breeding season are conducted to estimate total pup production. Once reliable pup count data are available the estimation of total pup production then depends on modelling the observed birth rate against an established statistical framework that describes how the numbers of whitecoat and moulted pups vary over the season (Hiby et al., 1988; Myers et al., 1997). The production estimation model designed for this process has been used for UK grey seal pup production estimation since 1984, delivering pup production estimates and associated 95% Confidence Intervals (C. Duck et al., SMRU, *unpubl*.). The model allows various parameters (e.g. degree of pup misclassification, time to moulting, time to leaving the breeding site) to be fixed or freed in order to deliver the most accurate model fit to the observed counts, thereby reducing the error (i.e. coefficient of variation or CV) of each production estimate

The use of photo-identification for mark-recapture estimation of population size has also proven itself a useful monitoring tool and one which could be used to independently assess pup production-based estimates. Other useful grey seal population monitoring methods include point counts of animals hauled out ashore outside the breeding season, such as that performed around Ireland in the summer 2003 (Cronin et al., 2007) and during the 2007 moult (Ó Cadhla and Strong, 2007). While the data generated give an indication of the seasonal distribution and relative abundance of grey seals utilising haul-out sites outside the breeding season, their value as long-term monitoring tools will depend on the accumulation of data from telemetry and other sources to determine the relationship between numbers and age-structure of grey seal sub-group(s) counted ashore and the population at large.

2.3.2.2.2 Telemetry (for estimating habitat use, home range, foraging areas and haul-out behaviour)

Telemetry techniques used for studying seal habitat use, home range, foraging areas and haul-out behaviour include radio-tags, satellite tags (ARGOS SRDL), GSM tags. Some of the tags used to date include dive computer and wet/dry sensors and other environmental sensors (e.g. temperature, salinity). Information that results from the telemetry technologies include location, dive and haul-out duration information and used to study animals range, foraging behaviour and haul-out behaviour. Most of these are attached to the pelage of the seal, some of the smaller radio and satellite tags can be flipper mounted, those attached to the pelage fall off when the animal moults. A major shortcoming of using telemetry to estimate the proportion of the population ashore during counts is the tag loss associated with the moult and the resulting gaps in information on haul-out behaviour during this period. However, certain approaches have been applied to try to overcome this limitation. Ries et al. (1998) developed a maximum likelihood estimator to infer rate of radio-tag loss in the Dutch Wadden Sea and to estimate the size of the local pre-pupping population. Flipper mounted telemetry devices have been used to obtain information on the haul-out behaviour of harbour seals during the moult in northwest US (Huber et al*.,* 2001; Simpkins et al*.,* 2003). Sharples (2005) combined telemetry data with ground counts over the year until tags fell off prior to moult to devise a mean harbour seal population estimate for St. Andrews Bay, Scotland; this was compared to a minimum population estimate obtained by aerial means during the moult and the proportion of seals ashore during this period subsequently estimated.In telemetry studies that have used large samples of tagged seals the proportion of seals hauled out during counts or during 'ideal conditions' has been used to devise correction factors for count data (Thompson et al., 1997; Huber et al*.,* 2001, Simpkins et al., 2003). Recent telemetry efforts in southwest Ireland have provided valuable data on the haul-out behaviour of harbour seals (Cronin, 2010).

2.3.2.2.3 Mark recapture – photo identification

Mark-recapture techniques are powerful tools for estimating demographic parameters (Seber, 1982). Photographing the natural markings of animals is a non-invasive means of 'marking' large number of individuals and has been used to study population parameters of various cetacean and pinniped species. It is also an inexpensive technique relative to other more traditional tagging methods (Karlsson et al., 2005). The success of photo-identification application to identify individuals and to examine demographic parameters of a population over long periods of time depends, amongst other things, on the uniqueness and consistency of individual markings.Photo-identification of individual harbour seals over several years has been used successfully in British Columbia for the purposes of assessing predation on fish stocks (Olesiuk et al., 1996). The technique has proven useful for examining population parameters of harbour seals in Alaska (Crowley et al*.,* 2001; Hastings et al., 2001; Moran, 2004) and Scotland (Mackey, 2004; Cunningham and Duck, 2007) and site fidelity of harbour seals in southwest Ireland (Cronin, 2007). The system was effectively used to provide mark-recapture histories from photo-identification data to estimate the number of grey seals around the Farne Islands and other major North Sea haul-outs in summer in the early 1990's (Hiby, 1997). The technique has also been utilised to provide information on site fidelity and movement of female grey seals between haul-outs in the Irish Sea and Celtic Sea (Kiely et al., 2000; Lidgard, 1999, 2002) and in studies in the Baltic Sea (Karlsson et al., 2005).

Until relatively recently photographs for photo identification purposes were taken with the single lens-reflex (SLR) camera using standard or slide film. Advances in digital photography have resulted in high performance professional digital SLR cameras which eliminate the need to handle rolls of film in the field. Hastings et al. (2001) recommended the use of digital photography in photo-identification studies of harbour seals to obtain larger sample sizes and reduce photograph processing time and costs.

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2.3.3 MONITORING SEABIRDS

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Many different aspects of seabird biology are monitored and hence, many different survey and monitoring techniques are used. At sea, many seabirds disperse widely and are thus hard to follow. Densities and distributions are normally mapped by ship-based or aerial surveys. Such mapping exercises are normally limited to single seas or areas under national jurisdiction, i.e. rarely will cover the entire ranges of seabirds at sea. Moreover, as monitoring requires repeated surveys, not all mapping at sea should be seen as monitoring, although the methods used are fit for these purposes as well.While mapping these parameters, additional information on seabird behaviour at sea may be gathered (Camphuysen et al. 1995; Camphuysen and Garthe 1997; 2004; Garthe et al. 2008).

2.3.3.1 Mapping seabird distributions and densities at sea

At-sea distributions of seabirds may be mapped with a variety of methods, largely depending on research questions and available resources. In some situations, seabirds on the water may be mapped from land, e.g. around breeding colonies. Further offshore, shipboardand aerial surveysare used (see also Sections 2.3.1.2.6.1 and 2.3.1.2.6.2). Each of these two platforms thus has its specific strengths and weaknesses, but combination of ship and aircraft are rare, due to the costs involved.

Survey techniques used during ship-based surveys vary widely and depend on the target species (its abundance and visibility) and research questions. Ships (or small boats) may be used to get total counts of local populations, such as numbers of breeding black guillemots on rocky shores (Ewins 1985; Ewins and Tasker 1985), or flocks of seaduck in coastal waters (Leopold et al. 1995); to study specific feeding behaviour at sea, e.g. in relation to human offshore activities (Garthe et al. 2008; Mendel et al. 2008); or to map and follow populations at sea. For seabirds the strip-transect method is commonly used (Tasker et al. 1984; Komdeur et al. 1992), in which seabirds are grouped into a few pre-determined, perpendicular distance bands. Normally, all targets (further referred to as birds) in a 300 m wide strip on one side of the moving ship are counted. Birds are divided in swimming and flying birds as both categories pose different problems to the observers. All swimming birds (including birds touching the water only briefly, e.g. during plunge-diving) will be assigned to a distance class (sub-strip adjacent to the vessel; Fig. 2.3.3.1). The standard distance-bands in ESAS (European Seabirds At Sea) methodology are: A (transect line to 50m perpendicular); B (50-100m); C (100-200m) and D (200-300m).

Swimming birds near the transect line will be avoiding collision and swim away perpendicularly. At the time of detection, some of the “A” birds will be detected in sub-band B and for this reason, sub-bands A and B are often combined to a band AB. Using relative numbers of birds noted in AB, C and D, respectively, makes it possible to estimate the numbers of birds missed in the more distant bands C and D, as numbers in all three bands should be the same, assuming birds are evenly spread over the sea surface. This is done in a similar fashion as during analyses of line transect data, in which each target is given a specific distance (e.g. Skov et al. 1995). G(0), in this case the proportion of birds present in band AB is not normally determined, and simply assumed to be 1. Correction factors to compensate for missed birds in bands C and D are based on this assumption, and the relative numbers seen in all sub-bands.

For flying birds passing through the 300 m strip a so-called snap-shot method is used (Tasker et al. 1984). This allows estimating true densities of flying birds from a moving vessel, provided that birds are only included in the densities estimates if they:

1) Fly by at a perpendicular distance of no more than 300 m on the side of the ship (portside or starboard) where the count was conducted;
2) do so no further than a distance ahead of the observers, that is covered by the moving ship in one minute (as an example: this is circa 300 m at a sailing speed of 10 knots) and
3) do so at the start of a full minute (snap-shot).

This snap-shot method for flying birds may seem more complicated than the method for swimming birds. The problem here is, that we need to get the correct densities for flying birds in the airspace above the area surveyed, from a platform that moves at a speed similar to that of the birds. Consider that the ship moves at a standard speed of 10 knots, so that it would move forward circa 300 m per minute. With a total band width to be surveyed, a surface area of 300x300 would be surveyed in that one minute. In that area and in that minute, a given number of flying birds would be encountered. Now consider that this same ship would be moving at indefinitely slow speed, in other words, would take forever to cover the same 300 m of transect length. In that hypothetical case, an indefinite number of flying birds would pass through the square of 300x300 m and if all these birds would be classed as “in transect” the resulting calculated density would be infinite per km^2. The correct density would only be achieved (if all birds passing through the 300x300 m square would be counted) if the ship would travel at the speed of light. In other words, counting all birds at any speed lower than the speed of light would overestimate densities and the bias is related to vessel speed. The solution is to only count flying birds at the onset of each subsequent counting period (1 minute), while all birds passing through the 300x300 m square at other moments (later) should be ignored. If counts are grouped into longer segments of transect, flying birds should still be counted in a shapshot very whole minute.

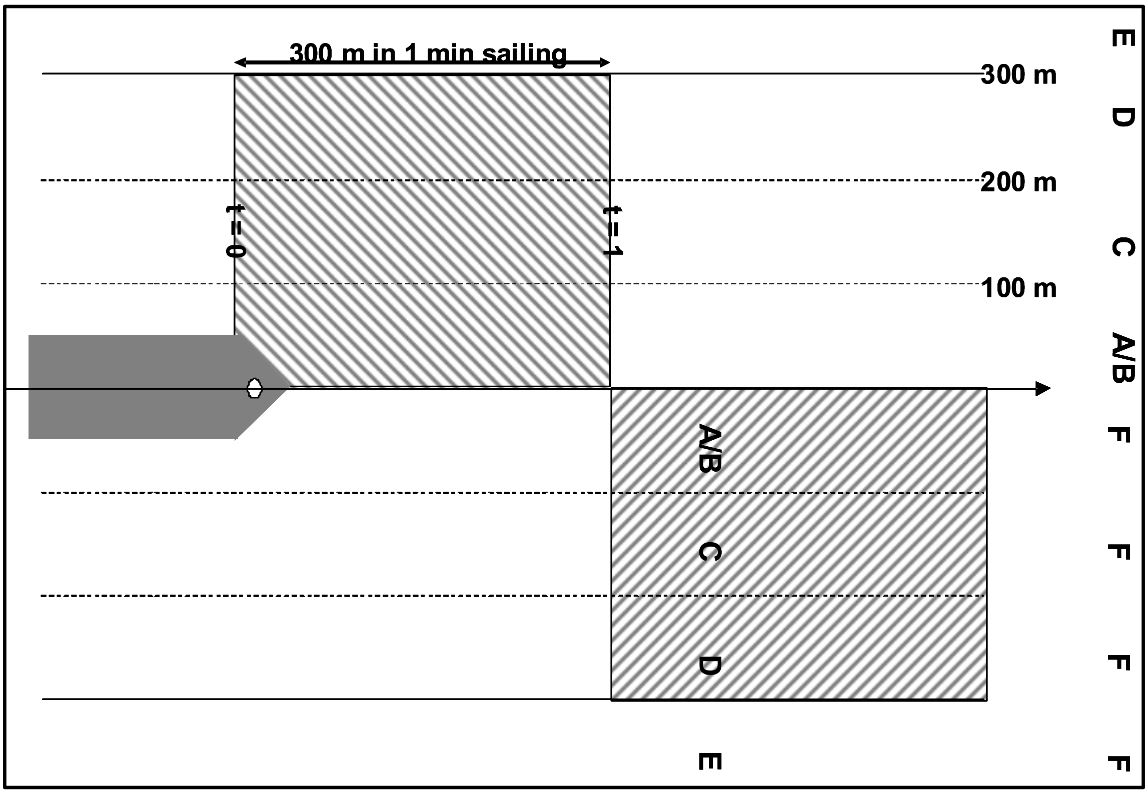

Fig 2.3.3.1. Schematic overview of the two available counting strips to the left and right of the survey vessel (not drawn to scale). Only the side with the best viewing conditions is normally surveyed. The two shaded 300x300 boxes are examples of two subsequent areas used for snap-shots of flying birds. The first box, to the left of the ship's course line, represented by the central arrow, and marked from t=0 to t=1 (minute) is used for the first snap-shot of the bird count for flying birds. At some point in time, in this case after one minute, the observers decide that viewing conditions are better on the right side and move their attention there accordingly. Now, a second box is used for the next snap-shot, on starboard (and presumably the observers keep watching here for a longer period of time, always broken up into one minute snapshots). Thus, the t=1 moment becomes the t=0 for the next snapshot, etc. Swimming birds are registered continuously and marked as "in transect" when swimming on the snapshot side of the ship and within sub-bands AB, C or D. Counting results are grouped in 5 minute intervals. All birds seen flying are noted as "F" on the field sheets, and later in the database. Swimming birds are noted as swimming in one of four sub-bands on either side of the ship: AB (0-100 m perpendicular), C (100-200m), D (200-300m), W (anywhere from 0-300 m perpendicular, but no further data) or E (outside the counting strip at more than 300 m perpendicular distance). Note that only the birds seen in A/B, C and D plus those in W and F when "in transect" will be used for density calculations. The approximate positions of the bird observer(s) preferably is on the top of the bridge (white oval).

Cross-over techniques between line-transect and strip-transect methodology have been used for marine mammals seen during (strip-transect) seabirds surveys (Leopold et al. 1992; 1997). Additional perpendicular bands, >300 m from the trackline were used for mammals in such surveys, and detection functions were calculated following distance sampling theory (Buckland et al. 2001).

New survey techniques

Survey work at sea is labour-intensive and there is always a drive to find technical alternatives for human observers. An interesting new development is the use of high resolution video during aerial surveys of seabirds (and marine mammals). While still no alternative of most surveyors, such techniques may become more widely used in the future (Hexter 2009).

Seawatching

Seawatching (counting numbers of seabirds passing a vantage point on the coast, per hour) is an old monitoring technique to follow numbers of seabirds over time. Long-term datasets exist for certain sites, but it is not always easy to see what exactly is being monitored. Some studies, however, have provided insights in local area use, total population sizes, changing migration patterns, and effects of severe weather on seabirds (den Ouden and Stougie 1990; Wheeler 1990; Waltho 2005; Camphuysen and Leopold 1996; Oliver 1999; Camphuysen 2009).

2.3.3.2 Breeding numbers, breeding output

Several questions regarding seabirds are more easily studied on land, than at sea. Several seabirds that are widely dispersed during most of the year, concentrate in few and well-known breeding colonies in the breeding seasons. Population size (adult breeders only) and population development is often used by colony counts. Whole colonies, or even all colonies of a given species may be counted (e.g. Nelson 2002), or much smaller, monitoring plots may be followed from year to year (Walsh et al. 1995; Irwin 2005; Heubeck 2006). Shear numbers of birds in a colony only provides crude information on how (well) the birds are doing at sea. Seabirds are generally long-lived and important parameters to seabirds, such as at-sea food availability may take many years to become manifest in numbers of breeding adults (e.g. Durant et al. 2003). Feeding conditions at sea are better monitored by following parameters on breeding success or growth of chicks of seabirds as these provide a direct link to the situation in the marine environment (Cairns 1987; 1992),or biological parameters such as the amounts of forage fish in the sea (Hatch 1992; Furness and Greenwood 1993; Furness 1996; Harris and Wanless 1997; Litzow et al. 2000; Rowe et al. 2000; Davoren and Montevecchi 2003; Tasker and Furness 2003).

2.3.3.3 Remote census techniques

As seabirds spend most of their time at sea, out of sight from land, several different remote census techniques have been used to monitor their whereabouts and behaviour at sea. Tracking devices, using radio (VHF), more sophisticated satellite or GPS loggers and relatively low-tech devices that estimate position from daylight data have been used successfully on a wide variety of seabirds, ranging in size from storm-petrels to albatrosses (Nygard and Einvik 1991; BirdLife International 2004). Many studies using tracking devices are aimed at finding out where birds feed at sea or how they migrate, and may not be considered monitoring studies. However, studies aimed at following the birds' behaviour in relation to changing environmental or feeding conditions are monitoring in a sense, even if such studies only last for (part of) one season. Different examples may be found in Wanless et al. (1990); Watanuki et al. (1999); Bugoni et al. (2005); Whittier and Leslie 2005; Perrow et al. (2006); Okes et al. (2009); Paiva et al. (2010); Votier et al. (2010).

2.3.3.4 Monitoring the environment through seabirds

Seabirds may be used to monitor the state of the environment, as they accumulate certain pollutants in their tissues, eggs, feathers or stomachs (Furness and Camphuysen 1997). By default, this type of environmental monitoring also monitors the state of seabirds. Oil pollution at sea is relatively easily measured by following numbers of dead birds that wash up on beaches, and the percentages of oil-contaminated birds among these (Camphuysen and van der Meer 1996; Camphuysen and Heubeck 2003). Contaminant levels of even harder to spot pollutants, such as man-made chemicals or heavy metals are monitored by following levels in bird eggs and feathers. Time-series may go back a long time if museum material is added to recent samples (Berg et al. 1966; Frank et al. 1983; Goede and de Bruin 1984; Appelquist et al. 1985; Thompson et al. 1992; Becker et al. 1992, 1997, 1998; Becker and Muñoz Cifuentes 2004; OSPAR Commission 2007), and plastics and other floating debris levels are followed by monitoring seabird stomach contents and numbers of entangled birds washing up (van Franeker et al. 2009; Ryan et al. 2009; Phillips et al. 2010).

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2.3.4 MONITORING POPULATIONS OF MARINE TURTLES

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There are many different methods for monitoring turtle populations (Table 2.3.4.1), but universally, nest counts are the most standard technique for assessing sea turtle population. For example, turtles on a nesting beach can be individually marked with small electronic PIT (Passive Integrated Transponder) tags which are injected into the muscle of the shoulder and remain in place for the lifetime of the turtle and then detected with a small handheld scanner upon return (Reina et al. 2002). Although nest counts only provide a good assessment of the number of adult females (males to not nest), nest counts are the primary response variable for assessing changes in sea turtle population size (Heppell et al 2003). Nest counts also provide additional information such as number of eggs per nest, number of hatchlings and number of nests per females during a breeding season. As nest counts are often highly variable from year to year (e.g. inter-annual variation may be driven by environmental variability) it is important that long term monitoring programmes are established to improve population estimates (Heppell et al 2003). Additionally it is important to standardise the methodology and effort. Simple regression of such nest count data is generally insufficient due to the high variability with even in stable populations there may be years when almost no turtles nest and alternatively, years when much higher than normal numbers nest (Hays 2000). However, by incorporating mark-

recapture information on remigration intervals using statistical methods, such trend analysis can improve estimates (Heppell et al. 2003).
Additional methods for monitoring turtle populations include visual surveys from boats and aircraft (Marsh & Sinclair 1989). Aerial surveys can provide important estimates of turtle populations at foraging grounds (Marsh & Sinclair 1989). For example, Houghton et al. (2006) carried out aerial surveys throughout the Irish and Celtic Seas to determine the abundance of leatherback sea turtles. Such estimates can be extremely useful when compared with comparable aerial survey estimates from other coastal foraging grounds. Indeed, Doyle et al. (2007) suggested that turtle densities in Irish and UK waters are an order of magnitude smaller than turtle densities along the east coast of America and Canada. Roos et al. (2005) used aerial surveys to estimate turtle numbers foraging in sea grass beds and reef flats. In the Mediterranean, Cardona et al. (2005) used aerial surveys to determine habitat use of loggerheads around the Balearic Islands. When conducting aerial or boat surveys for turtles it is important to use appropriate distance sampling techniques (Buckland et al 2001) and to remember that marine turtles spend more than 90% of their time underwater (Hochscheid et al. 2010). Indeed, if possible it is important to correct surface estimates of turtles using known behavioural data collected from tracking studies (e.g. probability of detecting a turtle at the surface). Also, aerial surveys may only be useful if there are high densities of turtles. Once densities are too low it may become difficult to detect trends.

Mark recapture techniques such as PIT tagging on nesting beaches and satellite tracking of individuals can help provide important data on internesting intervals and remigration intervals. Such information is critical in determining accurate population estimates and correcting beach count data. For example, large inter-annual variation in nesting numbers can be attributed to synchrony in the remigration intervals of individuals (Solow et al. 2002). This synchrony can be driven by environmental variation that affects the feeding conditions (Solow et al. 2002).

Long term satellite tracking of individuals can provide useful insights on factors determining the length of remigration interval (e.g. location of foraging grounds). At present, the ability to track some turtle species (e.g. leatherbacks) over entire reproductive cycles (2-3 years between successive nesting periods) is not feasible, however for some species such as loggerheads, individuals have been tracked over two remigration intervals (Schofield 2009). Satellite tracking can also be used to infer mortality in turtles (Hays et al. 2003). In addition to PIT tagging and satellite tagging, there are less invasive mark-recapture techniques such as photo ID that can provide information on both males and females (Schofield et al. 2008).

Table 2.3.4.1 Common methods for assessing population size and trends in sea turtles (adapted from Heppell et al 2003)

Census type	Life stage	Data	Examples
Nesting beach – nest counts	Adult females, eggs and hatchlings	Abundance and trend data, eggs per nest, egg and nest survival	Bjorndal et al. (1999); Hays and Speakman (1991)
Nesting beach – tagging (PIT)	Adult females	Abundance and trend data, remigration interval, adult female survival	Chaloupka & Limpus (2002);
In-water surveys with tagging/photo ID	Juveniles to adults (male/female)	Abundance and trend data, size and age distribution, determination of sex ratios	Roos et al. (2005); Schofield et al. (2008); Schofield et al. (2009)
CPUE fishing recoveries	Juveniles to adults (male/female)	Abundance and trend data, size and age distribution, by-catch mortality	Carreras et al. (2004); Casale et al. (2007)
Aerial or boat surveys for individuals, aerial surveys for beach tracks	Juveniles to adults (male/female)	Abundance and trend data, size and age distribution	Marsh & Sinclair (1989); Cardona et al. (2005); Roos et al. (2005); Houghton et al. (2006); Witt et al. (2009)
Strandings	All	Size and distribution, mortality rates	Witt et al. (2007)

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2.3.5 MONITORING POPULATIONS OF FISH (INCLUDING STOCK ASSESSMENT)

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2.3.5.1 Introduction

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Fish populations and assemblages can be monitored using various techniques, depending on the species, its habitat, biology and ecology as well as on the monitoring objectives. Fish monitoring is mainly - but not exclusively - carried out for stock assessment purposes, and can be done on different parts of the stock or on different life stages of the species investigated.

Basically, monitoring techniques can be split in two broad categories: indirect and direct methods.

Indirect methods are based on fishery-dependent data, such as catch and effort statistics and demographic (size/age) structure of the catch. Such methods are widely used and allow the estimation of the abundance/biomass of fish stocks at sea, either through pool dynamics (VPA and similar methods) andsurplus production models (Hilborn and Walters, 1992).

Direct methods are based on research surveys and are aimed at avoiding the biases deriving from the analysis of commercial catches. They are traditionally used to provide fishery-independent data on abundance/biomass and on the distribution by size/age of fish and shellfish. Moreover, such data are used in the "tuning" of VPA and similar methods (Hilborn and Walters, 1992).

More recently, survey-based models have been used for providing fishery-independent trends in stock abundance and demography (Needle 2003), and for investigating stocks dynamics under different exploitation patterns (Lembo et al. 2009).

Surveys are also widely used for providing data on biological parameters of stocks, and increasingly for providing information in support of an ecosystem approach to fisheries management. Since survey data are geo-referenced, they are also useful for spatial management of fishery resources.

From a different perspective, direct methods can also be split in removing (e.g., catch based methods) and non-removing (e.g., acoustic or visual techniques) methods.
The quality of the estimates obtained from survey data depends on the suitability of the method for the single species or life stage, and on the statistical design of sampling. In any case, when monitoring fish, it is important to realize that patterns seen in the monitoring will be dependent on the monitoring technique(s) used. No technique is suitable to cover the complete fish population at once.

2.3.5.2 Monitoring of abundance, biomass and demographic structure of fish populations based on commercial fisheries data (indirect methods)

Two methods will be analyzed in this section: (i) VPA and other similar methods, (ii) methods based on catch and effort data.

2.3.5.2.1 VPA and other similar methods

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Fishing affects fish stocks in two basic ways – reducing the overall abundance of the stock, and changing the stock demographic structure. With higher fishing mortality rates, there will be relatively fewer older fish in the stock. The age structure will thus be shifted towards younger fish, and the length structure will be shifted towards smaller fish. Catch composition data are thus required to estimate the relative abundance of different age classes or cohorts. This information is primarily used to determine the current mortality rate in the stock, which is an indicator of the level of fishing pressure. When raised to the total catch composition of the fishery, these data can also be used in the so called "VPA (Virtual Population Analysis) and similar methods" to estimate the numbers of fish in each age class or cohort. The latter is routinely used to fit the relationship between the spawning stock and the subsequent recruitment. Catch composition data also reveal the exploitation pattern (fishing mortality at size or age) which reflects the selectivity of fishing gears. Finally biological information from catch can be used to estimate the growth rates and other demographic features (sex ratio, reproduction pattern, maturity ogives, and so on).

Catches are sampled in order to characterise their composition in size, age, sex, maturity and fecundity. In general sizes are sampled according to a stratified random sampling or a clustered sampling approach, while for age, sex, maturity and fecundity a two-stage sampling is used (Cadima et al. 2005). Guidelines on sampling schemes to estimate length compositions (stratified across different fleets, fishing grounds, ports and time periods) are provided by Sparre et al. (1989) and Sparre (2000). Samples are usually taken on a monthly basis, and from all varieties of gear and capture locations. Sample size should be large enough to determine the levels of variance within each stratum. Length frequencies must then be raised to the whole commercial catch using catch and effort data. Where fish can be aged (e.g. using otoliths or scales), sub-samples of the length frequency data are usually taken to estimate the proportions in each length class in each age group. This is known as an *age length key* (ALK). Fish for ageing are usually selected as a stratified sample (e.g. picking the first five fish in each length class), so that sampling is evenly allocated across both small and large fish. ALKs are usually re-sampled every year since the number of fish in each age class varies with the annual recruitment strengths, and because growth and mortality rates can also change over time.

Other biological data needed for the VPA based assessments include size (or age) at maturity, fecundity, and average weight-at-length. All of these biological characteristics may change to some extent between years reflecting the overall health of the environment, or the availability of food. Special sampling programs are usually conducted every few years to estimate these characteristics. Information on spawning seasons and feeding patterns may also be useful to understand the seasonality of growth and recruitment, and in order to evaluate the possible value of closed seasons in managing the fishery.

The VPA, which is also known as *sequential population analysis* or *cohort analysis* in its more approximate form, has a long history. Derzhavin (1922) combined catch statistics with age data of fish for the first time, and the method was developed by several scientists, including most notably Fry (1949), Gulland (1965) and Pope (1972). A very comprehensive review of VPA and other similar age-structured methods was provided by Megrey (1989), and a detailed practical manual on using VPA was provided by Lassen and Medley (2000) and Darby and Flatman (1994).

The VPA as a method uses commercial fisheries data to make calculations about the population that must have existed at sea in order to sustain the recorded catches. The analysis is based on information on the contribution of each cohort to the catch, a cohort being defined as a group of fish all of the same age belonging to the same stock (Sparre and Venema 1998). Essentially, if the number of fish caught from a cohort is known for one year, it is possible to have a minimum estimate of the number of fish which were alive the previous year by adding those fish which succumbed to natural mortality (Fig. 2.3.5.1).

By estimating annual survivor numbers and natural as well as fishing mortality rates backwards scientists are able to reconstruct the 'virtual' population.

The main assumptions of a VPA are (Hilborn and Walters, 1992):

- there are no fish alive in the stock that is older than the oldest of the catch;
- the natural mortality rates (not necessarily constant) are known;
- there is no net immigration and emigration.

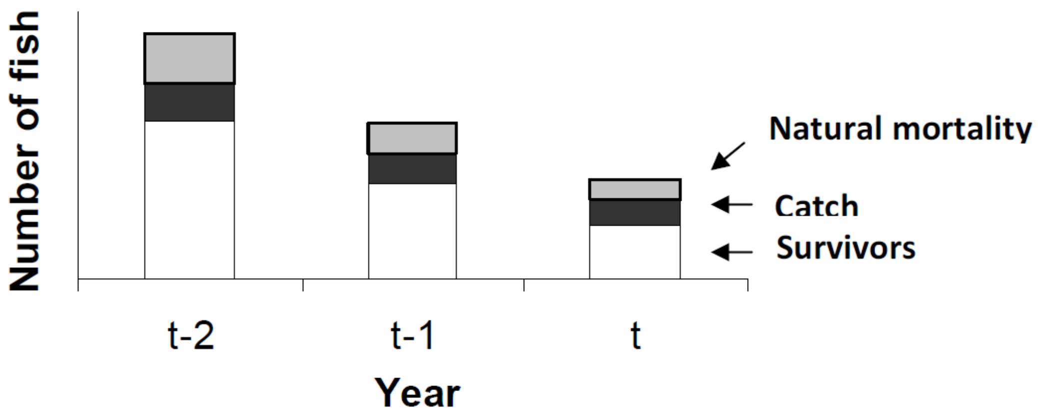

Figure 2.3.5.1: VPA for a hypothetical fish population with three age cohorts (adapted from Jennings et al. 2001).

VPAs require two key estimations: terminal fishing mortalities of the oldest age groups in all years and natural mortality rate. Terminal fishing mortality rates are often difficult to estimate, however VPA calculations will be nearly independent from this parameter as long as fishing mortality rates are high relative to natural mortality (Christensen 1996). In mathematical terms a basic VPA involves solving the equation $C_{a,y}/ N_{a+1,y} = F_{a,y} / Z_{a,y} \{\exp(Z_{a,y})-1\}$, where C represents the catch and Z is the coefficient of instantaneous total mortality (fishing mortality F + natural mortality M). The equation has to be solved for each age *a* in year *y* for every year class (Cotter et al. 2004). As a result of its apparent simplicity and the few assumptions which is based on, the VPA has become a very common stock assessment method for species where catch-at-age data is relatively easy to obtain.

There is however a large body of literature regarding the effects that errors in input parameters may have on VPAs and related methods (Pope 1972; Saila et al. 1985; Christensen 1996). More specifically, studies have been carried out on the impacts of errors in cohort size due to inaccurate landings and discards data (Sampson 1987; Kimura 1989), inaccurate age determinations (Bradford 1991; Reeves 2003; Punt et al. 2008) and erroneous natural mortality rate estimations (Sims 1984; Hilden 1988; Schnute and Richards 1995; Mertz and Meyers 1997; Clark 1999).

Age-reading errors are made when estimates of age which are based on reading hard structures such as otoliths or scales differ from the actual age of the animal concerned. Bradford (1991) found that such errors can lead to substantially biased time series of recruitment estimated from a sequential population analysis. Simulations carried out by the same authors indicated that even mild inaccuracies in ageing can lead to halved estimates of recruitment variance, and as a result introduce potentially important autocorrelations in the time series of recruitment. Reeves (2003) carried out simulations of ageing errors in Baltic cod stocks, attempting the quantification of the effects of age reading errors on the perception of stock trends and on the short-term management advice given by scientists as a result. Inaccuracies in ageing in particular affected estimates of required and effective fishing mortality calculated for catch forecasts and advice based thereupon. Overall ageing errors in Baltic cod generally lead to overly optimistic advice on total allowable catches (TACs), and thus potentially endangered stock conservation. Punt et al. (2008) proposed a method for constructing age reading error matrices which account for both age reading inaccuracies and bias, in order to enable scientists conducting stock assessments to incorporate these errors into their analyses. The authors found that the method was able to accurately estimate bias and imprecision as long as sample size was large and the person estimating the ages unbiased.

A poor estimation of natural mortality rates will cause bias in the calculation of stock size and in yield estimations. Such a bias can subsequently lead to over- or underestimation of harvest rates and quotas recommended to managers. Clark (1999) found that an underestimation of natural mortality rates did not seriously affect long-term yields and resulted in lower recommended real harvest rates, making a conservative estimate of natural mortality a better choice for management purposes. The fact that precise estimates of natural mortality rates are often lacking combined with the sensitivity of VPA output results to this particular parameter led Cotter (2004) to the suggestion that VPA outputs should be considered relative rather than absolute values, as is often the case.

The VPA looks at fish populations from a historical perspective, under the premise that once the historical evolution of catches can be explained, a prediction of future catches becomes possible by using analytical approaches (Shepherd and Pope 2002). The analysis thus works retrospectively, which in fact is a further important limitation of the method: VPA provides the least information on stock size and fishing mortality (F) in the previous year, which is the most important to predict the future performance of the fishery (Cotter et al. 2004).

In order to address the problem of retrospective operation, calibrated or tuned VPAs were developed (Pope and Shepherd 1985). In this approach, analyses of catch-at-age data are tuned using abundance indices from scientific research surveys, commercial landings or catch per unit effort data. Methods vary from what are known as *ad hoc* tuning methods (e.g., Laurec and Sheperd 1983) to integrated statistical methods such as for example CAGEAN (Deriso et al. 1985) and ADAPT (Gavaris 1988).

Due to the large number of methods available, Pope and Shepherd (1985) tested the performance of various VPA tuning methods using effort data. To achieve such a comparison, the authors estimated the results of repeated data analyses modified by randomly generated noise and subsequently compared them to the original situation which had been used to construct the data sets. Such an approach allowed for the evaluation of both variability and bias in the results. The methods investigated included Saville's method (Saville 1981), Hoydal-Jones method (Anonymous 1982), Gamma method and variations thereof (Anonymous 1982), Partial Exploited Biomass method (Anonymous 1981), Laurec-Shepherd method (Laurec and Shepherd 1983), Rho and Log Rho method (Anonymous 1982), Armstrong-Cook method (Armstrong 1985, pers comm in Pope and Sheperd 1985), and the Hybrid method (Pope and Shepherd 1985). A detailed review of each of the tested tuning methods is given by Pope and Shepherd (1985), and lies outside the scope of the current review. The study concluded that all methods worked well when catchability was constant, even in the presence of considerable noise in catch and effort data. However, those methods which assumed a constant catchability were easily biased when the real catchability varied systematically in any of the fleets from which the effort data were obtained. Moreover, the authors found that different methods were more or less advantageous when applied to data containing different errors, and suggested the Laurec-Shepherd method as well as a Hybrid method based on the results of the analyses (Pope and Shepherd 1985).
Generally speaking, the *ad hoc* Laurec-Shepherd method was extensively used by ICES scientists in Europe, whereas the ADAPT integrated statistical method was more widely applied on the east coast of North America. In order to compare the results of the two methods, Patterson and Kirkwood (1995) used Monte-Carlo simulations and known error structures to analyse the performance of both methods in situations of uncertainty. Results showed that the ADAPT method gave more precise and less biased population estimates under the simulated error distributions. Moreover when simulating management advice, the ADAPT method resulted in improved fixed fishing mortality targets as well as reduced variability in TACs between years. However, Patterson and Kirkwood (1995) also found that both methods resulted in fishing mortalities which were higher than the management targets when indices of stock size were highly uncertain.

Doubleday (1981) developed a method called Survivors, which essentially combines and builds on some of the features of separable VPA (Pope and Shepherd 1982) and conventional tuning methods such as the Laurec and Shepherd (1983) method discussed above. The method attempts to estimate the number of survivors of each population cohort at the end of the time period represented by the catch data.An underlying model assumes that fishing mortality can be separated into the fleet levels used in the assessment, the catch per unit effort or survey data are incorporated consistently and the data for the final year have similar errors as data for previous years (Shepherd 1999). Through an iterative algorithm the final estimates are weighted averages of all the data available. However, the method developed by Doubleday was never widely used before being further developed by Shepherd (1999), who called the resulting method *eXtended Survivor Analysis* or XSA. XSA is an improvement over the original Survivors method in that multiple abundance indices can be incorporated, and the method allows for variable catchability between age groups, thus accounting for the related problem of predicting year-class recruitment strength (Shepherd 1997). Moreover, XSA uses an algorithm which never yields negative results, and calculates the abundance of survivors with a higher level of consistency (Shepherd 1999). A more detailed description of the XSA mathematical computations are given in Lassen and Medley (2000).

However, Shepherd (1999) pointed out that XSA relies on the following assumptions:

- the VPA and catch-at-age matrices are treated as being exact
- scientific survey data / catch per unit effort data are collected separately so observations errors are assumed to be independent
- correlations among errors are not the result of errors in age-reading or across ages as a result of year effects.

Cotter (2004) gives several examples where the assumptions would be invalid, thus pointing out the limits to the applicability of XSA. Moreover, an independent evaluation of XSA as a method is difficult despite its widespread popularity at ICES and to an extent at SGMED working groups, in part due to the fact that there are many different ways of performing an XSA. There are for instance options of down-weighing or exclusion of historical data, using alternative catchability methods, including an additional estimate of survivors in the final year (Darby and Flatman 1994). While such possibilities allow a skilled analyst to bring assessment results in line with perceived stock status, they also make the results highly subjective (Cotter et al. 2004).

Integrated catch analysis (ICA) is based on a similar concept as VPA, requiring age-disaggregated catch data as well as survey tuning indices (Patterson and Melvin 1996; Patterson 1998). The method relies on an integrated analysis model, and as such it circumvents the over-parametrization of classical VPA approaches. It does this through an algorithm which minimizes the weighted sum of squared residuals of modeled compared to observed catch and tuning data, making the assumption that catches and survey tuning indices are log-normally distributed. Moreover, fishing mortality is modeled by incorporating both an age-specific selection pattern and a year-specific factor, thus taking into account potentially inaccurate estimations of terminal fishing mortality rates. ICA has been applied to a number of pelagic stocks by both ICES and SGMED, lately through the FLR framework (Kell et al. 2007), possibly due to the fact that ICA allows for the inclusion of a total biomass index for tuning. This means that relative abundance indices coming from acoustic surveys, frequently available for pelagic species, can be incorporated into the stock assessment.

A further alternative to VPA for assessing stock status are statistical catch at-age-methods, which essentially attempt to develop a model of population dynamics and then attempt relating model predictions to actually observed data. Statistical techniques are used to determine the best set of model parameters to fit the observations (Jennings et al. 2001). The method is based on catch curve analyses and was developed by several authors, including Doubleday (1976), Paloheimo (1980), Fournier and Archibald (1982), Deriso et al. (1985), Kimura (1990). Useful reviews are provided by Megrey (1989) and Hilborn and Walters (1992), the latter authors in particular present a detailed chronology of statistical catch-at-age methodological developments and comparisons between the various approaches developed by the authors listed above.

Statistical catch-at-age analysis overcomes the main problems with VPA, the fact that abundance estimates for cohorts that have not completely disappeared from the fishery are the most imprecise in a VPA analysis. Moreover, the majority of recent catch-at-age methods do not require the estimates of natural mortality needed for VPA, and thus eliminate this considerable source of error on stock assessment results (Jennings et al. 2001; Hilborn and Walters 1992). The method proposed by Deriso et al. (1985), is a particularly well-known early example of statistical catch-at-age analysis, widely used in part due to the accompanying computer software called CAGEAN (Catch-AGE Analysis) developed by the authors. Like VPA, the model makes use of the traditional catch and exponential decay equations. However, instead of starting the calculations with the terminal age group, annual recruitments and numbers at age are estimated from the first year of the analysis forward in time. Fishing mortality is assumed to be separable, in the sense that it can be approximated by taking into account the age-dependent fishing vulnerability as well as a year-specific factor (Jennings et al. 2001). CAGEAN is able to use fishing effort as well as catch at age data, and in an important contrast to VPA the method does not assume that catches are known exactly (Deriso et al. 1985). A further advantage is the ability to quantify the uncertainty surrounding key model outputs and a high flexibility to incorporate additional information, for instance estimates of natural mortality. However, a major disadvantage of statistical catch-at-age methods is the complex nature of the computational procedures, which make the approach less transparent (Jennings et al. 2001).

2.3.5.2.2 Methods based on catch and effort data

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Catch and effort data are among the most important information to obtain from a fishery. In fact, catch per unit effort (CPUE) is the most traditional index of fish stock abundance (Hilborn and Walters 1992; Hoggart et al. 2006).Catch and effort data are usually obtained from interviews with fishermen, usually collected when they land their catch, or from logbooks. Landings are usually subsampled and raised to total catches within different strata, based on a frame survey of the number of active vessels. Logbooks may help in estimating the complete census of the fleet catch, or at least of a sample of cooperative fishers. However, although expensive, the most reliable method for collecting commercial catch and effort data is to use trained observers onboard fishing vessels to directly record data. Through this approach bycatch and discards, which might otherwise go unrecorded, can also be sampled.

Specific guidance for the collection of catch and effort data is given by Gulland (1983). Guidelines for resource mapping, frame surveys and data collection in manpower limited situations are given by Caddy and Bazigos (1985). Stamatopoulos (2002) describes the alternative types of fishery surveys, using different combinations of complete enumeration (census) and sub-sampling across space and time.

Catch rates from commercial fisheries (Catch Per Unit Effort, CPUE) are traditionally used to provide abundance indices of exploited stocks. However indices estimated from CPUEs may give an unreliable view of stock abundance for at least two main reasons:
i) fishing tends to be concentrated on the larger densities of the stock, thus CPUEs give an optimistic estimate of the standing stock;
ii) catching power or “catchability” of commercial vessels tends to change with time due to technological creeping in fishing equipments, leading to a change in the relationship between catch and biomass at sea.

In general, survey-based abundance indices are less biased for spatial effects and effort changes because the survey track and the fishing gear used can be kept constant over the years. However data from the commercial fishery are also essential for estimating the total catch and fishing effort, and for raising the samples taken at landings for analytical models. Samples from commercial fisheries are usually cheaper and easier to obtain in large quantities than research vessel survey data. As a result, sample sizes are usually larger, and variances lower. Where the fishery is exploited by more than one gear type, or where catchability has changed over time, commercial catch and effort data should be standardized (Bishop 2006), e.g. using Generalized Linear Models (GLMs) or Generalized Additive Models (GAMs) (Hilborn and Walters 1992; Bigelow et al. 1999).

Biomass dynamic models and their properties

Age-structured analyses of catch data may not be practical for assessing all fish stocks. Many fish species are difficult to age, in particular tropical fish, and in many data-poor situations scientists are forced to rely solely on catch and effort data. Biomass dynamic models, also known as surplus production models, production models, stock production models and surplus yield models are the simplest fish stock assessment models available, and the only option in such data poor situations. The term *biomass dynamic model* can be considered to be the most accurate since the concept of surplus production or yield can also be considered in the context of an age-structured model. These models deal with the entire stock as one large unit of biomass, without entering into details and attempting to model age or length frequencies. Biomass dynamic models were first used by Graham (1935), and several potential forms have been developed and discussed in the literature (Schaefer 1954; Schaefer 1957b; Pella and Tomlinson 1969; Fox 1970; Walters and Hilborn 1976; Uhler 1980; Csirke and Caddy 1983; Roff 1983; Schnute 1985; Garcia et al. 1989; Lleonart and Salat 1989; Prager 1994). The theory underlying the models has been reviewed by several authors, including for instance Ricker (1975), Caddy (1980), Gulland (1983), Pauly (1984), Hillborn and Walters (1992) and Jennings et al. (2001).

The underlying concept of biomass dynamic models centres on the need to find the maximum sustainable yield (MSY), i.e. the highest catch that can be sustainably harvested in the long term without decreasing the productivity of the fished stock. If a given level of fishing mortality is to be sustainable, there must be a balance between mortality (which reduces the biomass of a population) on one hand and reproduction and growth, which increase it on the other hand (Russel 1931). In other words the two sources of biomass increase are gain in weight by individuals present in the population and recruitment of new individuals after birth, while catch and natural mortality are sources of biomass loss (Hilborn and Walters 1992):

new biomass = last biomass + recruitment + growth – catch – natural mortality.

This scenario of course does not take into account potential immigration and emigration from the fish population being analyzed. For an unexploited fish population, recruitment and growth can be combined into a single term called *production* (Hilborn and Walters 1992):

new biomass = last biomass + production – natural mortality.

In situations where production exceeds natural mortality rates, population growth will occur, and the term surplus production is used to quantify the difference between production and natural mortality. Mortality and reproduction are however not entirely independent, but instead fluctuate within limits dictated by a combination of biological factors (e.g. predation, competition) and abiotic factors (e.g. weather, temperature). Biological factors are essentially regulated by density dependence, which is the relationship between population density, and birth, growth and death (Jennings et al. 2001). Such density dependence gives populations the resilience to sustain increases in mortality rates due to fishing, so that (Hilborn and Walters 1992):

new biomass = old biomass + surplus production – catch.

The first and most widely used model of population growth was developed by Schaefer (1954; Schaefer 1957a; Schaefer 1957b), who used a classical logistic equation of population growth based on work concerning the potential production of North Sea fish stocks carried out by Graham (1935). There was an attempt to apply the equation to a multispecies fishery (Lord 1971), however it is usually applied to single species stocks and written as:

$$dB / dt = rB (1 – B / k) – C$$

where B is the stock biomass, r is the intrinsic rate of population increase (i.e. the difference in per capita death and birth rates in the absence of density dependence), k is a parameter corresponding to the unfished equilibrium stock size, and C is the catch rate. The catch rate C is assumed to be proportional to fishing effort and stock size:

$$C = qfB$$

where q is a parameter describing the catchability, f is the fishing effort and B is the stock biomass. Finally, this model for catch implies that the catch per unit effort (CPUE) is an index proportional to stock abundance. The Schaefer model is essentially characterized by a symmetric relationship between biomass and surplus production, with surplus production equal to zero both at a low and high level of stock biomass.

An alternative to Schaefer's logistic model of population growth is the Fox curve (Fox 1970), which is more appropriate when used for biomass measurements as opposed to abundance measurements in terms of numbers of individuals, for which the logistic curve has traditionally been used. In practice the difference between the two models is the fact that while in the Schaefer model an effort level where yield per unit of effort can be zero, in the Fox model yield per unit of effort is greater than zero for all levels of fishing effort. The choice between the two models thus only becomes important when relatively large values of fishing effort are being considered. A detailed comparison between the two models, including a worked example can be found in Sparre and Venema (1998).

Pella and Tomlinson (1969) extended the Schaefer model by adding an extra parameter, to the logistic model:

$$dB / dt = rB – r / k B^m – C.$$

When m = 2, this equation is identical to Schaefer's original equation. The reasoning behind this parameter is the notion that the surplus production curve is perfectly correlated with stock size in the Schaefer model, whereas the addition of m allows the production curve to be skewed to the left of to the right in the Pella and Tomlinson approach. If m < 2 the model produces a maximum to the left, and if m > 2 the model produces a maximum to the right of the classic logistic curve. Either of these scenarios is possible (Hilborn and Walters 1992), and thus this approach offers greater flexibility in the shape of production curves. The Pella and Tomlinson approach has been used extensively in practice, in part due to the computer programme GENPROD (Pella and Tomlinson 1969), which allows the estimation of parameters using assumptions of equilibrium.

Model parameter estimation

While the choice of production curve to use is pretty straightforward, fitting these models to real data through an appropriate choice of model parameters is not. All methods of fitting biomass dynamic models to data rely on the assumption that true abundance and a recorded index of abundance such as CPUE can be related. The underlying idea is that CPUE can be related to population biomass by taking into account catchability, i.e. the parameter q. Catchability will change both in response to the distribution and behaviour of a population and due to changes in technology, and is thus a parameter which is difficult to estimate. There are two major categories of methods for estimating the parameters of biomass dynamic models: *equilibrium methods* and *non-equilibrium methods*; useful reviews are given by Hilborn and Walters (1992), Polacheck et al. (1993) and Jennings et al. (2001).

Equilibrium methods

Equilibrium models are popular when attempting to fit the Schaefer model (Schaefer 1954), assuming that in an equilibrium state the relationship between effort and CPUE is linear. The underlying notion is that each year's catch and effort data represent a steady-state or equilibrium situation, where the catch is simply equal to the surplus production at that specific level of fishing effort (Hilborn and Walters 1992). However this assumption is a very dangerous one to make since CPUE will only in very rare cases be a simple reflection of density-dependent responses of a fish population to the amount of fishing mortality being exerted. Instead, CPUE will also reflect a decrease in the overall standing stock, especially in a developing fishery where fishing effort frequently increases on a yearly basis. The great danger with using equilibrium methods is that in many fisheries effort is steadily increased over time, and stocks are given credit for a much higher level of resilience against exploitation than is actually the case (Jennings et al. 2001).

In order to account for the fact that fish stocks are only very rarely in equilibrium, Gulland (1961) suggested using an *equilibrium approximation* approach. More specifically, the author suggested relating the annual CPUE with the fishing effort averaged over years, with n equal to the number of age classes of fish being harvested. This approach thus estimates the amount of fishing effort which in a state of equilibrium would produce on average the observed level of annual CPUE in the fishery. A full explanation of this approach, including a discussion of rationale and performance of the approach can be found in Gulland (1969). A similar approach to overcome the difficulty that stocks are very rarely in a state of equilibrium was taken by Fox (1975). Fox used a weighted average to approximate equilibrium fishing effort: the CPUE of the incoming year class j in year i $CPUE_{ij}$ is related to the amount of effort in year i, that of the previous year class $CPUE_{ij-1}$ is related to the fishing effort in years i and i-1, that of the year class which entered 2 years previously $CPUE_{i,j-2}$ is related to the fishing effort in years i, i-1, and i-2 and so forth. In order to perform these calculations Fox (1975) provided a computer program written in Fortran, GENPROD, which has been widely distributed and used by fishery scientists.

Non-equilibrium methods

Process-error methods

Process-error methods are also known as regression methods since they transform production curves into linear forms, and subsequently attempt to fit models to the data (Walters and Hilborn 1976; Schnute 1977; Lleonart et al. 1985; Lleonart and Salat 1989). These methods still use catch and effort data, however the assumption that the population is in equilibrium is no longer made. Schnute (1977) for instance developed a method to integrate the Schaefer model over one-year time steps, thus transforming the model into a dynamic equation. Lleonart and Salat (1989) suggested a different approach, based on the concept of *stock inertia*, where stock inertia represents the stock's inherent resistance to changes in fishing effort.

According to Walters and Hilborn (1976), the Schaefer's model may be expressed in a discrete form (difference equation model) as:

$$Bt+1 = Bt + rBt(1-(Bt / k)) - Ct$$

Since C=$qfB_t$and $B_t=U_t/q$, with f=fishing effort, q= catchability coefficient and U= C/fthe model can be rearranged in terms of CPUE (U) and effort (f) as:

$$(U_{t+1} /U_t) -1= r –(r/kq)U_t - qf_t$$

Having time series of catch and effort data the r and k parameters of the model can be obtained solving the equation as a standard multiple linear regression of the form:

$$Y= b0 + b1X1 + b2X2$$

Although the attractive due to the easy computational approach, the major drawback of process-error methods is related to the fact that they assume catch and effort have been measured without errors, and that all error is due to the interaction of population size and growth rates (Hilborn and Walters 1992; Polacheck et al. 1993; Quinn and Deriso 1999). In practice process-error methods depend on a large amount of contrast in the time series of catch and effort data in order to obtain reasonable parameter estimates (Jennings et al. 2001).

Observation-error methods

Observation-error methods, which were first proposed by Pella and Tomlinson (1969), are generally recognized to be the best methods for estimating production model parameters. This approach assumes that the underlying production relationship is correct, and that errors only occur in the relationship between the index used to measure abundance and true stock size. In order to test this, an estimate of stock size at the beginning of the available time series is made, and a model is subsequently used to predict stock sizes for the entire time series. Observed catch or population sizes are then compared to expected ones, and statistical methods used to adjust estimated parameter values in such a manner that they provide the best fit in the predicted vs. observed time series.

In the case of the Schaefer's model expressed in terms of difference equation, for example, the parameters (r and k) may be estimated using time series of biomass estimates and catch or catch and effort data, with some assumption on catchability coefficient(Hilborn and Walters, 1992).

The statistical methods used are usually nonlinear parameter estimation techniques that frequently involve the minimization of the squared deviations between observed and predicted CPUE. Observation-error methods were developed by several authors, including Fletcher (1978), Butterworth and Andrew (1984), Ludwig et al (1988), Punt (1990). Detailed reviews of the techniques involved are given by Hilborn and Walters (1992), Polacheck (1993) and Quinn and Deriso (1999).

When commercial information is limited but long time series of Z and abundance indices from trawl surveys are available, a variant of the non-equilibrium surplus production model can be fitted (Abella, 2007).

The classical model requiring time series of index of abundance (Bt) and effort(f) is:

$$Bt+1 = Bt + rBt(1-(Bt / k)) - qfBt$$

Since *qfBt =C*,catch in weight *(Ct)* can be substituted by the classic Baranovcatch equation:

$$C=(F/Z) B(1-exp(-Z_t)$$

and the model can be written as:

$$B_{t+1} = B_t + rB_t(1-(B_t / k)) – (F/Z) B_t(1-exp(-Z_t))$$

Z can be estimated by analysing the size structure of the surveys catches and F computed by subtraction if an estimate of M is available.

2.3.5.3 Monitoring of abundance, biomass and demographic structure of fish populations based on scientific survey data (direct methods)

Four methods will be analyzed in this section: (i) bottom otter trawl surveys, (ii) beam trawl surveys, (iii) ichthyoplankton surveys, (iv) underwater visual surveys.

2.3.5.3.1 Bottom otter trawl surveys

Contributors: Carlo Pipitone (CNR-IAMC), Fabio Fiorentino (CNR-IAMC)

Among scientific survey techniques, the bottom otter trawl is used worldwide to collect such data. Due to the possibility of keeping the gear, vessel, catch processing protocols and sampling design constant from year to year, observed changes in biological parameters, size/age structure and abundance are assumed to reflect actual changes in living populations without any bias (or with known constant bias) arising from the fishermen behaviour. Furthermore, scientific trawl surveys allow to collect data unobtainable from commercial catches, like e.g. sexual maturity and data on sublegal-sized fish (Ocean Studies Board 2000).

The bottom otter trawl

Bottom otter trawls are widely used in shelf and slope demersal fisheries. Being highly effective at catching benthic and nektobenthic fish and invertebrates, they are considered a good sampling tool for the monitoring of demersal stocks on “trawlable” grounds i.e., soft bottoms uniform enough to let the net be towed without damage.

The trawl net is shaped more or less like a loose pyramid (Fig. 2.3.5.2), which is towed on the seafloor by means of two steel *warps* of variable length wrapped around a double winch. The warps are paid out as much as about 4-8 times the bottom depth (the deeper the bottom, the longer the warps), which is necessary to keep the net laid down on the seafloor. When the vessel is underway the net mouth is kept open horizontally by two metal or wooden *otter boards* (or *trawl doors*) due to the friction produced against the water mass (Fig. 2.3.5.3). Each door is connected to one end of the mouth with a *sweepline*. The vertical opening of the mouth is ensured by floats arranged along the *headrope* (or *floatline*). The *footrope* (or *groundrope*) is provided with sinkers and/or chains and keeps the mouth adherent to the seafloor. The groundrope may be equipped with *bobbins* (or *rockhoppers*) that allow the trawl net to be towed over rough bottoms. The net vertex or *cod end* is kept close with a short rope that must be opened to let the catch fall down on the boat deck.

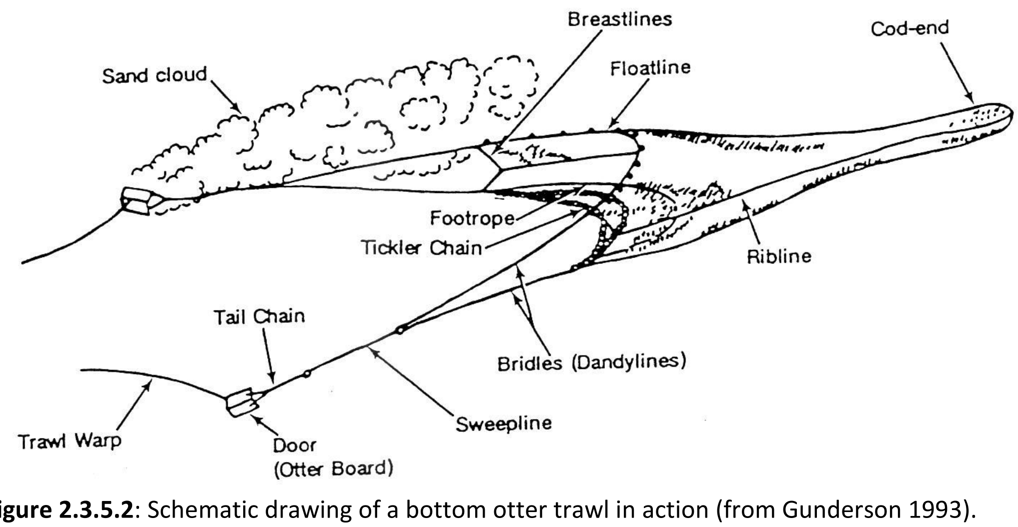

Figure 2.3.5.2: Schematic drawing of a bottom otter trawl in action (from Gunderson 1993).

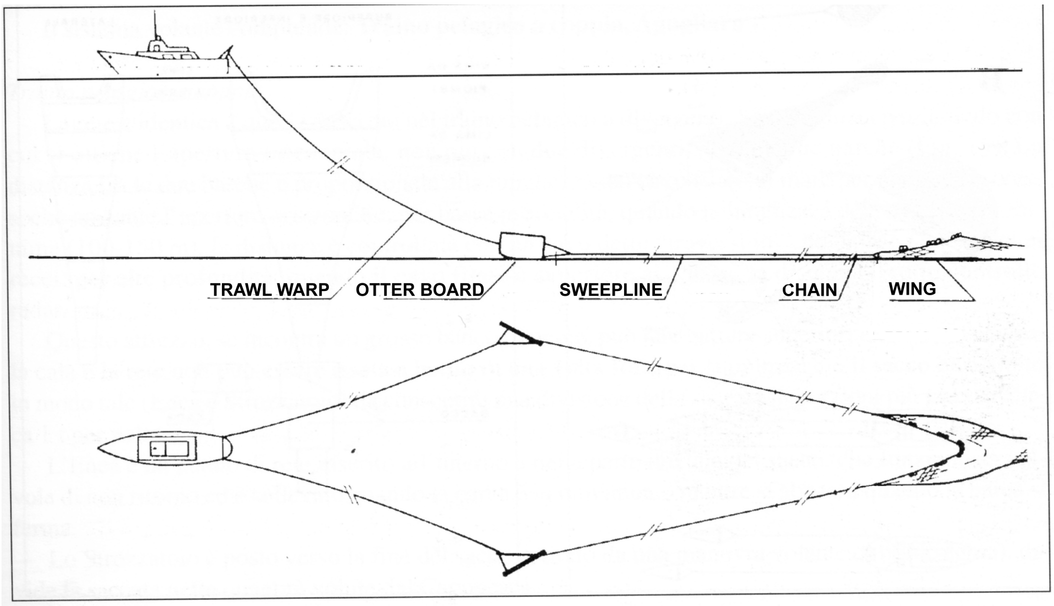

Figure 2.3.5.3: Schematic lateral and upper views of a bottom otter trawl in action (typical Mediterranean trawler, from Scaccini 1974).

The mesh size decreases from the mouth to the cod end: the selectivity of the net is determined almost exclusively by the mesh size at the cod end.
The trawl net is towed at about 2.5-3.0 knots and the duration of each haul depends on the sampling plan, usually 30 to 60 minutes.
Also the vessel used in the survey has an effect on the quality of data. Be it a research vessel or a hired commercial trawler, the quality of navigation and positioning equipment needs to be high to allow for exact measurements of the swept area. The experience of the captain and its crew are crucial at ensuring flawless onboard operations and at identifying non-trawlable areas.

Survey planning

A trawl survey may fall into several different categories, each with its own objectives (Saville 1977). We may identify two broad survey types:

- exploratory surveys, aimed at gathering baseline data on demersal stocks in an unknown (or poorly known) area;
- stock assessment surveys, aimed at collecting data on biology and population dynamics and at estimating the standing stock at sea. If collected as a time series, such data allow to follow temporal trends in the main population parameters - especially biomass and size structure - in order to evaluate the results of management or protection.

The area and timing of a survey depend on the research objectives, the type of resource under investigation and the available funds. Ideally the survey area should include the distribution area of the investigated stock(s); at the very least it should include the operational area of a fishery (e.g., a gulf, a bank, etc.). The timing should include the four seasons, or at least include the recruitment season and the reproductive rest period for the main commercial species.

Sampling design

Since trawl surveys are normally carried out over areas large enough to be spatially heterogeneous, the sampling design is stratified according to depth or habitat criteria, with boundaries and size of each stratum mapped prior to the survey. Abundance and demographic indexes are calculated separately for each stratum. The stratification provides a gain in precision of estimates whenever the variation among observations within each stratum is less than expected from samples taken randomly over the whole survey area (Hilborn and Walters 1992).

Random versus systematic sampling within strata:
There are two main approaches in sampling design for trawl surveys: the random stratified and the systematic stratified.
Classically trawl surveys selected sampling sites within a stratum completely at random, with every possible site having the same probability to be included in the sample each year (Fogarty 1985). However a systematic sampling will give a more accurate estimate of the mean density provided that the systematic transect or grid is oriented so as to cut across the spatial features of the stratum. In this way the surveyed area is more variable than by randomly choosing sample points while ignoring gradients (Hilborn and Walters 1992).
According to Hilborn and Walters (1992), in modern trawl surveys the use of a systematic grid should be preferred because:

- It is more efficient in term of ship time and sample point location;
- It permits the most accurate possible mapping of spatial patterns of density (total or by sex/age/length fractions of population) indices and boundaries of distributions;
- It minimizes the risk of missing density concentrations that are of roughly the same diameter (or larger than) the distance between grid points;
- It makes comparisons of distribution and density patterns over time easier.

As far as accuracy is concerned, it is better to have a low sampling density to make sure that the survey covers a large area so that the whole distribution of the investigated stock is sampled.

Catch sampling and processing

Both the vessel crew and the scientific staff should be adequately informed and trained about the importance of keeping all species and individuals in the sample without discarding those considered "less important" (e.g., non-commercial species or smaller individuals).

Time is always a limiting factor in onboard and laboratory processing. For this reason, large catches are generally sampled in order to collect the needed data from a representative portion of each trawl haul (Gunderson 1993; Sparre and Venema 1998). Several methods can be used to obtain representative samples from survey catches (Sparre et al. 1989).

Regardless the adopted protocol, two main steps are generally recognized: onboard processing and laboratory processing.
The following example of standardized procedure comes from the trawl surveys carried out in the Strait of Sicily during the MEDITS and GRUND programmes (Fiorentino et al. 2004).

Onboard processing

When the catch is on the deck, after the net has been checked to certify the haul validity, a photo is taken. Each shot contains date, survey code, number of the haul and stratum. Specimens remarkable for their size or for their rarity are photographed individually. Thereafter, the catch is processed according to the following procedure (Fig. 2.3.5.4):

1) remove all large specimens, regardless of the species
2) remove all bulky non-living materials (e.g. boulders, car tyres)
3) sort the catch by categories:
    - *target species:* bony and cartilagineous fishes, cephalopods, and decapodcrustaceans
    - *non-target species:* bony and cartilagineous fishes, cephalopods, and decapodcrustaceans
    - *other biological material (trash)* – other invertebrates, algae, seagrass, andbiological debris
    - marine litter.

Depending on the size of the haul, target species and/or other categories may be sub-sampled. In order to allow a good reconstruction of the length composition of target species when different length groups are present, the catch by species may be divided into fractions homogeneous by size. Secondly, samples are randomly extracted from these fractions(Fig. 2.3.5.4).

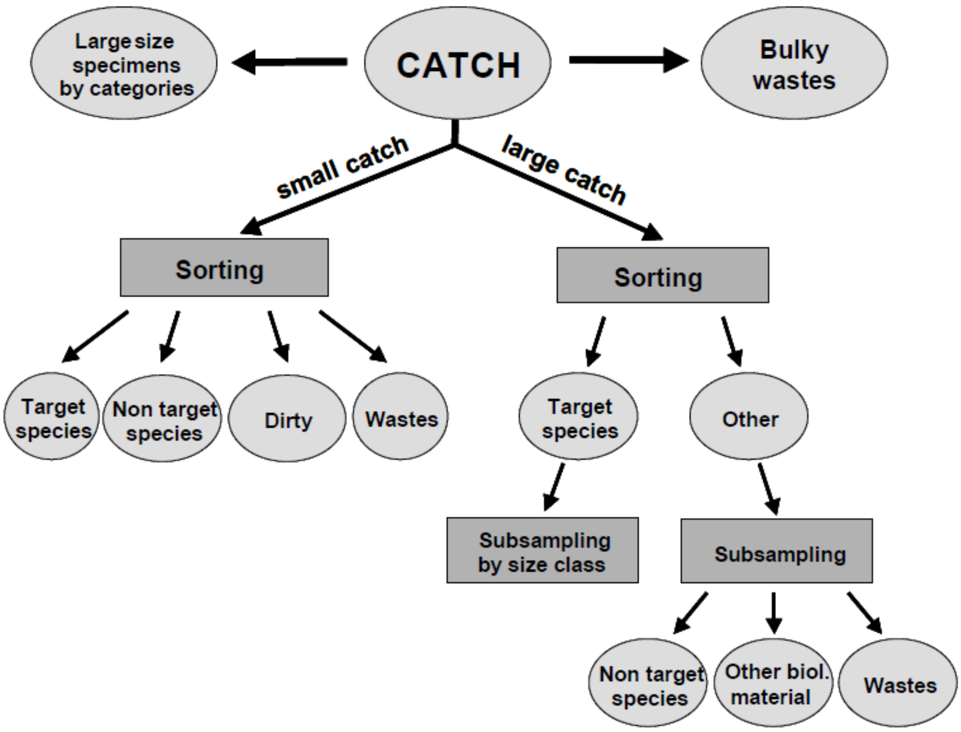

Figure 2.3.5.4: Flow chart showing a procedure for processing catches by haul in trawl surveys (sorting and sub-sampling) (from Fiorentino et al., 2004).

After sorting, specimens are stored in plastic boxes with a label reporting a code for the haul and survey and immediately deep-frozen (–40°C). After 24 h, each box is placed in a cardboard box and stored at –18°C.

The number of individuals and total weight in grams of non-target species are recorded. Samples of non-target species may be preserved for further study, within the framework of specific research programmes.
Trash and litter are weighed. Samples of trash can be preserved for lab analysis in order to gain a rough idea of benthic assemblages.
Regardless of the category, weight is always recorded on board with a steelyard.

Target species

The lists of target species should be combined by the following criteria:

- defined as such by current legislation
- importance for the demersal fisheries in the area
- easiness of routine identification
- vulnerability to fishing pressure

The main collected data concern catch rate as total, taxonomic groups and species, expressed in terms of trawling time (number/biomass per hour) or swept area (number/biomass per km^2).

More detailed data on target species demography can be collected onboard after sampling (e.g., length frequency distribution), while other data (e.g., individual size and weight, sampling of hard part for ageing, sexual maturity) can be collected in the lab; all of them are recorded in *ad hoc* forms following pre-determined protocols.

Laboratory processing

At the end of the trawl survey, the frozen material is stored at the lab. Before processing, specimens are defrosted overnight at room temperature. Each specimen is measured and weighed. Sex of target species is identified and the macroscopic maturity stage and gonad weight are recorded.

Morphometric measurements

In each sample or sub-sample of target species, length and weight are measured for the following taxonomic categories:

- Fishes: total length
- Cephalopods: mantle length
- Crustaceans: carapace length.

The length of fishes and cephalopods is measured with ichthyometers, whereas callipers are used for crustaceans. These measurements can also be taken onboard after the catch.

The body weight, measured with analytical scales, is expressed in grams rounded to the nearest 0.1 g in cephalopods and to the nearest 0.01 g in crustaceans. The weight of fish able to swallow large prey is measured after emptying the stomach.

Hard structures

Hard structures used in the age estimation are taken from adequate sub-samples of each haul. Sub-samples are selected in order to cover uniformly the length-range present in the catch. For each haul, the hard structures of at least two specimens, by sex and for each size class are removed: first rays of dorsal fin (*illicia*) for *Lophius* spp. and otoliths (*sagittae*) for the other teleosts.

2.3.5.3.2. Beam trawl surveys

Contributors: Ingeborg de Boois (IMARES), Carlo Pipitone (CNR-IAMC)

Fish monitoring programs using beam trawls in Europe are mainly carried out in the North Sea and its estuaries, the Celtic Sea, the English Channel and, since 2007, the Bay of Biscay. WGBEAM (Working Group on Beam Trawl Surveys) is a five-country working group coordinating beam trawl surveys in the ICES area. A manual with survey procedures and details is available at ICES (ICES 2009). Beam trawl surveys were used originally for the stock assessment of common sole, *Solea solea* and plaice, *Pleuronectes platessa*. Indices of standardized numbers per yearclass by species are derived from yearly surveys and used in the assessment. In the near future, indices from other flatfish species like turbot (*Psetta maxima*), brill (*Scophthalmus rhombus*), dab (*Limanda limanda*), lemon sole (*Microstomus kitt*), witch flounder (*Glyptocephalus cynoglossus*) and flounder (*Platichthys flesus*), could be derived from beam trawl surveys (ICES 2007).
Additionally, when systematically carried out over a long time period, beam trawl surveys can be used to identify changes in demersal fish assemblages related to such factors like climate changes (van Hal et al. 2010; Tulp et al. 2008).

Besides flatfish, which are the main group of target species, beam trawl catches containother demersal fish and macrobenthic fauna since the gear works in close contact with the seabed. For this reason the gear damages bottom habitats and communities to some extent, the entity of the damage depending on the heaviness of the rigging.

The beam trawl

A beam trawl is made of a conic net attached to a frame that keeps the net mouth open. The net vertex or *cod end* is kept close with a short rope that must be opened to collect the catch on the boat deck. The frame is composed of a metal tubular *beam* that gives fixed horizontal opening and two *heads* with *shoes* that give fixed height (30-50 cm). The gear is rigged with either tickler chains (connected to the beam and/or to the net) or a chain mat (Figs. 2.3.5.5 and 2.3.5.6).

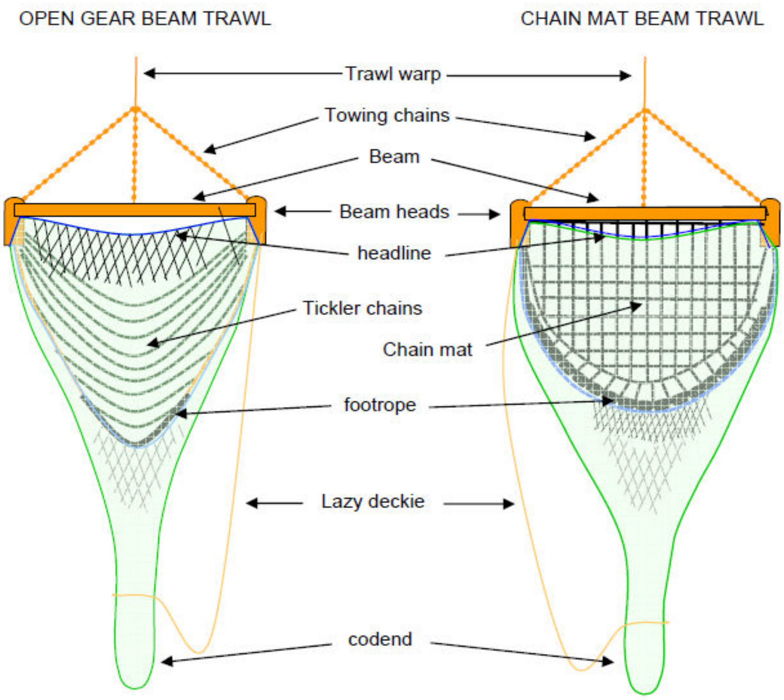

Figure 2.3.5.5: Schematic drawing of open gear beam trawl with tickler chains (left) and chain mat beam trawl with chain mat (right) (from Seafish, 2010).

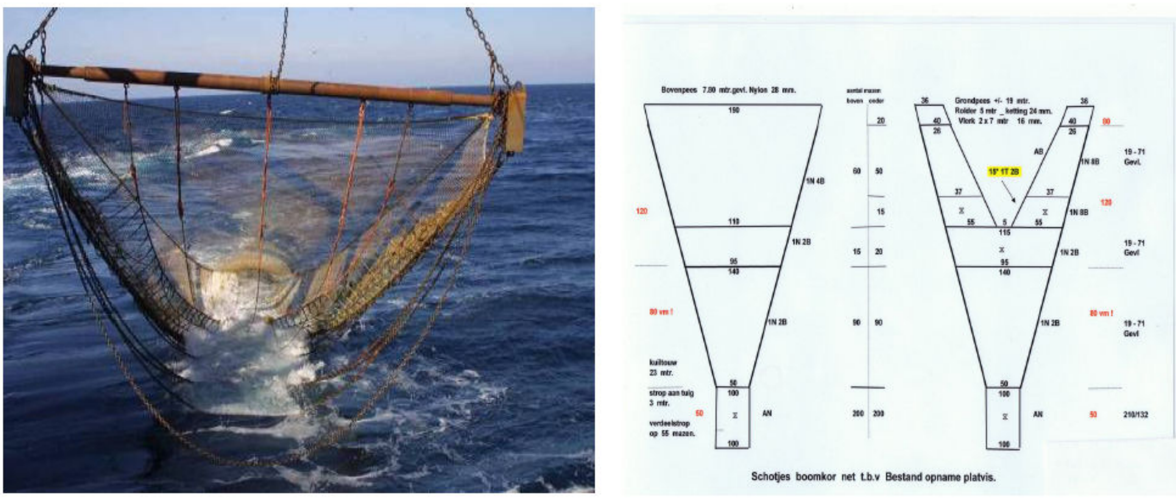

Figure 2.3.5.6: 8m beam trawl at the start of the haul (left); scheme of the beam trawl net used in the Dutch beam trawl survey (right).

The optimal rigging depends on the seabed structure. Beam trawls with tickler chains are used on smooth sandy bottoms. The ticklers help to disturb the fish from the seabed, causing them to rise and be caught by the net. The optimal number of ticklers depends on the substance of the sand: on soft sandy bottoms less ticklers are needed to get the fish out of the substratum than on more compact sandy bottoms.

In rougher areas (e.g. southwestern North Sea) a beam trawl with a chain mat is recommended to prevent gear damage by rocks or boulders. The purpose of this mat is to guide the net over any rough ground, thereby minimising damage to the gear. When operating a beam trawl in a sandy area with scattered boulders, a flip-up rope might be sufficient to keep boulders out of the net. This is a rope mesh fence towed ahead of the footrope to lift the footrope over any obstacles on the seabed.

The beam width can vary between 2 and 12 meters. For fish monitoring purposes, beam trawls from 2 to 8 meters are commonly used. Commercially sized beam trawls vary between 4 and 12 meter, depending on ship size and power.

Beam trawls can be operated on all depths (as long as bottom contact is possible), but fishing at depths >100 m might cause problems, especially in lighter gears since the increase of the trawl warp length and weight might influence the gear functioning.
Most modern trawlers tow two beam trawls from long derricks projecting over each side of the vessel.

The mesh size used in a beam trawl net depends on the targeted fish. In scientific surveys a mesh size of 20 or 40 mm is used for 0-1 year and older flatfish, respectively. In commercial beam trawls, mesh sizes vary between 80 and 120 mm in the cod-end, depending on the geographical area.

Current monitoring programs tend to use - despite the slight variability in the substrata covered - a standard gear since changing gear or rigging is time-consuming and causes intercalibration issues.

Survey planning

WGBEAM surveys are designed on a grid of fixed rectangles where the hauls tend to remain fixed in number and position across years (ICES 2009). Each haul lasts 30 minutes at a speed of 4 knots.

Three main problems are related to the quantification of catches in beam trawl surveys:
1) the exact measure of the sampled (towed) area. The ship log is used commonly to identify the sampled area. There are also options to fix a wheel or other measuring instruments to the gear;
2) the estimation of the duration of bottom contact during the haul. To fish properly, a beam trawl needs to be in constant contact with the seabed. Due to seabed structure, wind and sea conditions the gear might rock and lose contact with the seabed. To estimate the time of bottom contact during a haul, video devices attached tothe net might be used;
3) the often unknown catch efficiency of the gear. Realistic estimates of catch efficiency and hence of abundance or biomass of fish and invertebrate epifauna in an ecosystem are particularly important to determine e.g. secondary production or consumption rates (Harley et al. 2001). Therefore, the limitations of a sampling gear have to be considered in assessing how well an assemblage is described.

The influence of the above issues on fish stock estimates is reduced by standardising the gear and sampling operations. Also taking weather conditions into account, like e.g. not fishing with wind >6 Bft, helps to reduce potential errors. In this way, time-series from beam trawl surveys can be used to assess quantitative trends in fish stocks.

Catch processing

Total number and weight of all species plus length frequency distribution for commercial fish and shellfish species are collected during WGBEAM surveys (ICES 2009). Biological data (age and/or sexual maturity) are collected from selected species. So age and size structure information as well as fish assemblage structure for fish species living up to about 50 cm above the sea bed can be obtained from beam trawl monitoring programs.

2.3.5.3.3 Ichthyoplankton surveys

Contributor: Ingeborg de Boois (IMARES)

Monitoring of ichthyoplankton is carried out either to measure spawning stock biomass, or to measure the youngest year class of fish species. Ichthyoplankton includes fish eggs and fish larvae.

Main purposes and coordination

In the Atlantic, the Baltic Sea and the North Sea, long-time ichthyoplankton surveys have been carried out for anchovy and sardine eggs, mackerel and horse mackerel eggs, herring larvae in order to do stock assessment on those species.

Within the ICES area, for mackerel and horse mackerel the surveys are coordinated by ICES WGMEGS (Working Group on Mackerel and Horse Mackerel Egg Surveys), for anchovy and sardine by WGACEGG (Working Group on Acoustic and Egg Surveys for Sardine and Anchovy in ICES areas VIII and IX), for herring by WGIPS (Working Group for International Pelagic Surveys) and IBTSWG (International Bottom Trawl Survey Working Group). Outside the ICES area, some ichthyoplankton sampling has been carried out, but never in a consistent way to provide information for stock assessment (Somarakis et al. 2004). An experimental survey was carried out at Porcupine Bank and Rockall Trough (Dransfeld et al. 2005).

For plaice and cod eggs, surveys are carried out and are coordinated by ICES PGEGGS. Those surveys give the possibility to compare the stock assessment based on catch and effort surveys with the assessment based on the annual egg production (Damme et al. 2009).

Manuals are available for the mackerel and horse mackerel egg survey (ICES 2009) and for the herring larvae sampling made by IBTSWG (ICES 2005).

Gear and fishing techniques

Various techniques can be used to monitor ichthyoplankton. The two main categories of sampling tools are *towed gears* and *continuous samplers*. Towed gears are hauled from a ship, while continuous samplers collect water at a constant speed and sieve out the plankton. For both methods information on the volume of water filtered is collected with a flow meter, hence density estimates for ichthyoplankton can be made. All methods use a fine meshed net to retain all plankton.

During ichthyoplankton sampling, other variables like water temperature, salinity, and weather conditions are collected routinely.

Sampling of ichthyoplankton for assessment purposes is mostly done by fishing on fixed stations at regular intervals. The interval chosen depends on the area fished and on the available ship time.

Towed gears

Towed gears are conic nets shot and hauled from a ship. The mesh size varies according to the life stage sampled. For egg sampling, a mesh of 150-250 µm is often used, for larvae sampling a 280 µm mesh is adopted.

The depth sampled is dependent on the species targeted. Towed gears often do an oblique haul, which means that the net is shot and hauled at a constant winch speed up to a certain depth. Sometimes vertical hauls are taken (Uriarte and Motos 1998).
Examples of towed gears are (Fig. 2.3.5.7):

- Gulf high speed plankton sampler: it is towed behind the vessel, slowly dropped to within 5 m from the bottom and then returned to the surface again. A net with a fine mesh inside the sampler collects small organisms floating in the water column. Nosecones of different diameters regulate the amount of water entering the net while flow meters measure how much water is being filtered.
- Pairovet: a small, fine mesh net that is shot into the water to 70 meters from a motionless ship, then towed vertically to the surface. It is used primarily to sample fish eggs.
- Bongo: the name is chosen because the nets look like bongo drums. The net is towed obliquely through the water while the ship is underway from 212 meters to the surface, effectively sampling the layer of water where nearly all the ichthyoplankton occurs.
- Manta: the net is dragged at the sea surface while the ship is underway. It is used to sample fish larvae that occur typically at the surface, such as dolphin fish (*Coryphaena hippurus*), grunion (*Leuresthes* spp.), and flying fish (Exocoetidae).

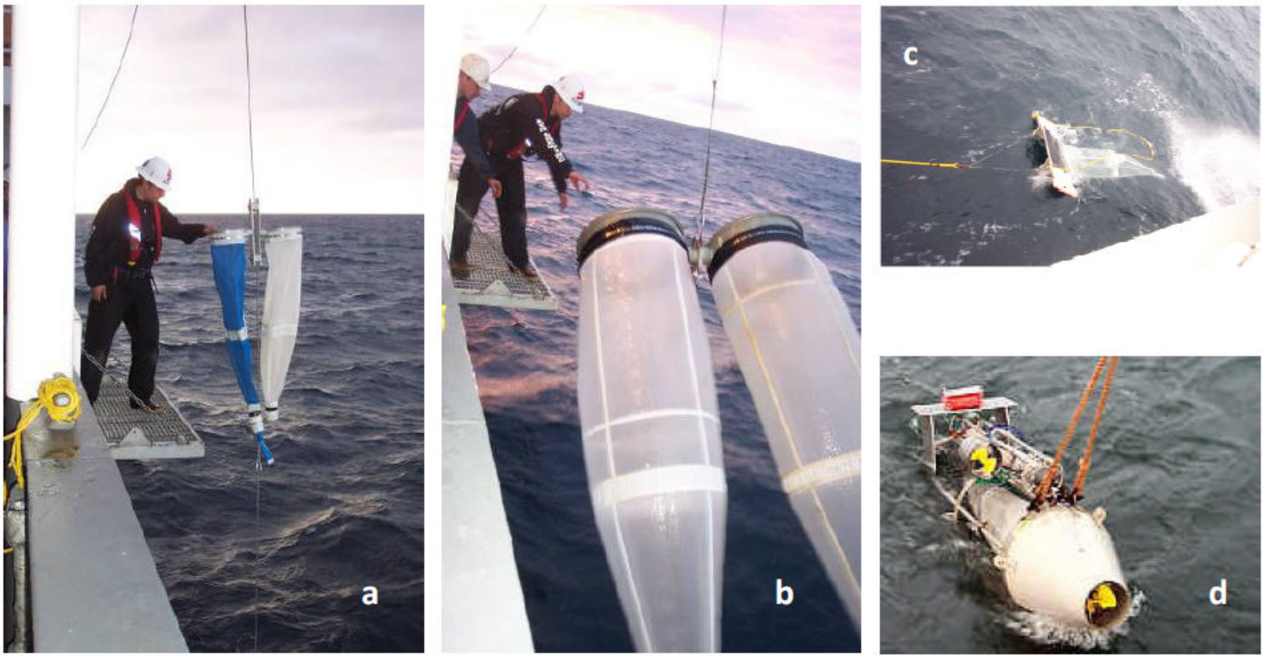

Figure 2.3.5.7: Pictures of ichthyoplankton nets. a: Pairovet; b: Bongo; c: Manta; d: Gulf VII. (Photos taken from: http://swfsc.noaa.gov (a-c) and www.cefas.co.uk(d)).

Continuous samplers

The Continuous Underway Fish Egg Sampler (CUFES) is used to collect pelagic eggs of fish, and ancillary data, at about 3 m depth from a moving research vessel (Checkley et al. 2000). These samples and data are, in turn, used to investigate the spawning habitat and estimate spawning biomass. Pelagic fish eggs are often aggregated in time and space. Continuous sampling from a ship moving at full speed is an effective means to assess aggregated distributions. Water is continuously pumped at ca. 600-700 liters per minute to the concentrator. Particles are then concentrated by an oscillating net (e.g. with a 500 µm Nitex mesh) in approximately 3% of the flow. The filtrate is discharged overboard. The concentrate passes to the sample collector where all particles are retained over sequential sampling intervals (e.g., 5-30 min) on a cod end of the same size mesh as used in the concentrator.

One of the advantages of CUFES is that it operates under nearly all sea conditions, providing a real-time estimate of the volumetric abundance of pelagic fish eggs at pump depth. CUFES-derived estimates of volumetric abundance are regularly compared with estimates of areal abundance made using vertically-towed nets such as Calvet or bongo (Bez 2000).
The main disadvantage is that continuous samplers only sample at one fixed depth.

Processing the catch

Sorting

When the sampling gear comes on board the research vessel, the net is rinsed with sea water, flushing the plankton to cod-end of the net.

Sample processing is dependent on the species and the life stage collected. All samples are fixed (generally with 4% formalin in sea water) and stored. Larvae are sorted manually from the fresh catch and then fixed. For the eggs, fixation is often done prior to sorting since eggs turn opaque after fixation and will only then be recognizable.

Sorting can be done manually by eye or under a stereolens. For sorting mackerel and horse mackerel eggs the so-called 'spray method' is used, based on the difference in density for fish eggs and other zooplankton present in the sample (Eltink 2007).

Since all samples (the sorted and the unsorted part) are fixed and stored, it is possible to check the quality of the sorting after the survey.

The species sorted from the sample depend on the research needs. Generally, only the eggs/larvae of target and other similar species are collected from the catch. Fish larvae are identified to the species level and measured. For fish eggs, the species and the development stage of the egg will be identified. All sorted larvae and eggs are counted.
It is possible to sort the samples for other (ichthyo)plankton besides the target species after the survey, if required by the research program.

Species identification
The main identification criteria for fish larvae are number of vertebrae, fin development, pigmentation.

Fish eggs are identified mainly by their size (different species have different size range of their eggs) and by the presence or absence of an oil globule.

Abundance estimates
The ichthyoplankton abundance is basically calculated as the number of eggs/larvae per volume of filtered water, which is measured by means of a flow meter installed at the mouth of the net.
The Daily Egg Production Method (DEPM) is used to calculate the spawning stock biomass (SSB) of a species starting from the number of eggs. The DEPM SSB estimate for horse mackerel is calculated as:

$$SSB = P/(F \times S \times R)$$

where P is the daily egg production for the total area, F is the female batch fecundity per ton, S is the spawning fraction of females and R is the sex-ratio, taken here as a constant of 0.5. The variance of the SSB estimate is approximately calculated with the expression:

$$var(SSB) = (F \times S \times R)^{-2} \times var(P) + P^2 \times (F^2 \times S \times R)^{-2} \times var(F) + P^2 \times (F \times S^2 \times R)^{-2} \times var(S)$$

assuming that the covariances between P, F and S are zero. The estimates obtained are dependent on the criterion chosen for estimating the spawning fraction.

2.3.5.3.4. Underwater visual surveys
Contributor: Stelios Katsanevakis (HCMR)

Abundance estimation of demersal fish populations is a by-product of several of the standard analytic fisheries stock assessment techniques, such as catch-at-age analysis, virtual population analysis, and depletion methods. However, these techniques rely on a time history of exploitation of the stock to determine abundance and may not provide short-term abundance estimations (Hilborn and Walters 1992). Monitoring of demersal fish stocks is commonly conducted with surveys using commercial fishing gear such as dredges or trawls, e.g. the International Bottom Trawl Survey (IBTS) in the North Sea (Heessen et al. 1997) and the International Bottom Trawl Survey in the Mediterranean (MEDITS) (Abelló et al. 2002). Although these surveys are valuable to monitor population trends based on time series of relative abundance indices, they may not always provide unbiased short-term density or abundance estimations, while they are inappropriate for many habitats such as rocky and coral reefs or seagrass meadows.

Catch efficiency and size selectivity issues with trawls and dredges are well documented. A major problem with such surveys is how to estimate catch efficiency, i.e. the percentage of the individuals present in the swept area that were actually captured by the gear. Trawl and dredge surveys normally take the area swept by the gear and expand the catches by the proportion of the study area that was surveyed (Hilborn and Walters 1992). In this way it is assumed that all individuals present in the swept area are captured (100% catch rate or catchability = 1), which is far from reality. Catch rates of trawls and dredges may be quite low and variable, depending on various factors such as trawling speed, gear specifications, substrate, behaviour and life history of target species, size of the individuals, duration of the haul, time of the day, and season (e.g., Shafee 1979; Chapman 1980; McLoughlin et al. 1991; Bailey et al. 1993; Giguere and Brulotte 1994; Tuck et al. 1997; Hall-Spencer et al. 1999; Reiss et al. 2006).

On rocky or coral reefs, where trawls and dredges may not operate, several other techniques have been used to study the abundance of fish populations, such as nets, traps, spears, explosives, and ichthyocides. These techniques should be considered as qualitative rather than quantitative. Relative abundance might be reported with such techniques, although variability in catch efficiency among sampling sites is quite likely to give questionable results. Furthermore, these techniques are destructive (especially explosives and ichthyocides) and have adverse impact on species and communities. Thus at the very least, they should not be applied to monitor protected species or species in protected areas or sensitive habitats.

With the low and variable catch rate of fishing-related methods, in many cases it is difficult to reach unbiased conclusions regarding absolute or even relative abundance and spatial distribution of demersal fish based exclusively on such data. Underwater visual surveys (UVS) are often advantageous in certain habitats compared to fishing surveys, to monitor fish populations. In addition, direct visual surveys are non-destructive, which should be of concern, especially when dealing with endangered species or protected habitats such as seagrass meadows or coral reefs. UVS are the standard monitoring tool in many cases, e.g. to monitor the ichthyofauna of coral reefs.
The main methods of UVS applied to fish are: (1) plot sampling, including strip transects and point counts; (2) distance sampling, including line transects and point transects; (3) fixed-time transects; (4) occupancy estimation with repetitive sampling; and (5) rapid visual techniques. All these methods can involve diver-based surveys (SCUBA or snorkelling) or *post hoc* examination of video or photo records.

2.3.5.3.4.1 Plot sampling with SCUBA or snorkelling

Contributor: Yiannis Issaris (HCMR)

Plot sampling methods are the most common and widely used UVS methods for surveying reef-fish communities. They typically consist of strip transects and stationary point counts and rely on trained observers recording fish presences, lengths, and abundances observed within a fixed volume of the water column. Surveys are usually conducted in shallow water and can be conducted while snorkelling or SCUBA diving. The choice depends mainly on the clarity of the water and the depth that observations are to be made. Observations deeper that 3-4 m in clear waters will generally require SCUBA (Cote and Perrow 2006), although due to time restrictions imposed by the physics and physiology of SCUBA diving (extensive decompression obligations and breathing gas availability), such surveys are usually conducted at depths <30 m. The use of enriched NITROX (i.e., nitrogen-oxygen mixtures with higher oxygen concentration than in air) instead of air as a breathing gas increases the available no-decompression time but is not applicable at high depths due to oxygen toxicity. Recent developments in technical scientific diving have allowed safer access at depths >30 m using special mixtures of breathing gases (heliox, trimix, e.t.c.) and sometimes even special breathing apparatuses such as Closed Circuit Rebreathers (CCRs). However, fish plot sampling UVS methods have not been applied by divers in deeper waters yet.

Stationary point counts

With the stationary point count method, fish are recorded within a specific circular area. The observer visually fixes a central point on the bottom to mark the centre of the stationary point count, while simultaneously estimating the radius of the count area (e.g. 7 m from the central point), noting features of the substratum to mark the circular boundary of the census (Samoilys and Carlos 2000). In some cases, in order to avoid false distance estimations, the observer may first lay down a measuring tape (e.g. 15 m in length) and then take position in the middle (Bohnsack and Bannerot 1986). Counting is done in a fixed time (usually 15 minutes) and all fish seen within the boundaries are recorded.The larger more mobile species are usually counted first and the smaller less mobile species are counted later.

Under modified field protocols, the observer may use the first 5 minutes to make a list of all species of fish in the circular area of survey and then use the other 10 minutes to count numbers and sizes of all fish in his list (Bohnsack and Bannerot 1986), or even count the most visible species around him for 5 minutes while remaining on the centre of the survey area and then actively search the whole area for more conspicuous species for the remaining 10 minutes. The best search radius may also differ when studying different fish species that have different behavior and in different visibility conditions (Kimmel 1993), although the 7 or 7.5 m radius has been proved to be the appropriate point count dimension for counting a range of cryptic and mobile species simultaneously (Bohnsack and Bannerot 1986; Samoilys and Carlos 2000).

Strip transects

With the strip transect methodology, fish density is estimated by a diver swimming or being towed along a strip of known or estimated length and width. The observer swims along a fixed line, measuring tape or chain marked at 1-5 m intervals previously laid on the substratum, along a depth contour. After waiting for 5-15 minutes for disturbance created by the setting of the transect line to settle, numbers and sizes of each target (or all) species occurring within a given distance of the transect line are recorded (Cote and Perrow 2006). Like in stationary point counts, the larger or more mobile species are usually counted first and the smaller or less mobile species are counted later. This can be done either by completing the transect once, while only recording mobile species, and then return to record the cryptic ones or simultaneously by incorporating a different search strategy while swimming along the transect, like e.g. looking forward for mobile species and closer for cryptic ones.

Area dimension, observer speed, and replication have the greatest effect on census estimates (Sale and Sharp 1983; Thresher and Gun 1986; Fowler 1987; Lincoln Smith 1988). The most appropriate area dimension for a census will be a compromise between an observer’s ability to count fish rapidly, and the behavior, distribution and density of the fish. The smaller the census area, the more thoroughly it can be searched, so narrow transects are likely to yield better counts. On the other hand, a narrow transect of practical length may be too small to yield adequate numbers for population assessment of the more mobile fish species typical of reef fisheries (Samoilys and Carlos 2000). Mapstone and Ayling (1993) have found that very short (25 m), very long (100 m), and very wide (20 m) transects are unsuitable for many reef fishes, so usually, mid-range transects sizes are typically used, i.e. 50 or 75 m lengths, and 10 or 5 m widths.

Moreover, transect speed should be as rapid as possible without compromising search efficiency, and observers must by trained not to count individuals that enter the census area after the census has started (Samoilys and Carlos 2000; Watson et al. 1995).

Limitations/Problems

A number of biases may influence the accuracy and precision of density estimates when using the strip transect technique (Cheal and Thompson 1997). Observer biases (including swimming speed, distance from the substratum and search technique) can generally be reduced by training (Sale and Sharp 1983; Thompson and Mapstone 1997), whereas biases induced by the distribution and habits of fishes will vary with size and placement of transect units. Historically, a wide variety of transect dimensions have been used (Sale and Sharp 1983; Choat and Ayling 1987; Kingsford and MacDiarmid 1988; Carpenter 1990; Chua and Chou 1994). A trend has been noted for increased density estimates from narrower transects as well as a variation of precision among the different strips width (Cheal and Thompson 1997). The same has been shown to apply for stationary point count methods, where species density appears to be inversely related to the length of the sampling radius (Bohnsack and Bannerot 1986). Ideally, observed density should not change with sampling radius if fishes are randomly distributed in a uniform habitat. Obviously, not every individual of every species is detected and this always causes an underestimation of density (Franzreb 1981). Therefore, the results of these methods should be seen as indices of abundance instead of absolute abundance estimates (Bohnsack and Bannerot 1986).

Often, the fundamental variable of interest is not density itself, but rather differences in density across some independent variable like community disturbance from fishing pressure. However, it has been shown that fish behaviour towards the surveying diver also differs with changes in the fishing pressure of the population under study (Harmelin-Vivien 1986; Bohnsack and Bannerot 1986; Bellwood 1998). In heavily fished areas, fish tend to move/stay away from the observer, while in protected areas, their behaviour is more relaxed and the secure distance in which they remain away from the observer is smaller. Using traditional UVS methods (strip transects and stationary point counts) in studies of the effectiveness of the protection in marine reserves, the differences in fish density estimates and even more so in biomass estimates can be biased. The reserves might appear to have larger estimated values than disturbed areas for identical real values, clearly due to different fish behaviour (Kulbicki 1998).

Computer simulation experiments by Watson and Quinn (1997) have shown that strip transect and stationary point count methods perform equivalently in terms of bias and variability when fish do not move, the amount of sampled area is the same, and all fish within the sampled area are observed. Comparison of the two methods in the field has also shown that they are equally effective, although point counts are more efficient due to their faster deployment times (Samoilys and Carlos 2000).

2.3.5.3.4.2 Plot sampling-surveys with ROV/AUV/UW video

Contributor: Giovanni D'Anna (CNR-IAMC)

UVS involving SCUBA or snorkelling are restricted to shallow waters. In SCUBA diving surveys there are substantial time constraints due to decompression sickness and gas availability. Such issues are overcome with underwater video surveys. Such surveys are based on image data acquisition and have been employed to get estimates of fish abundance, diversity and size in both shallow (Willis and Babcock 2000; Harvey et al. 2002) and deep waters (Priede andMerret 1998; Yau et al. 2001).

The first applications of video surveys used movie films (cinetransects), still photography and remote-controlled underwater television to estimate fish abundance in kelp beds and on reefs (Myrburg 1973; Alevizon and Brooks 1975). However, the low definition of images made the identification and counting of fish difficult, which was the main reason for the limited use of video surveys for quantitative studies in the past (Russel et al. 1978; Harmelin-Vivien 1985). More recently, video and associated technologies have greatly increased the ability of underwater devices to acquire high resolution quantitative image data to study fish populations (Michalopoulos et al. 1993).

Several types of video camera systems have been implemented depending on the objectives of the study and on environmental factors. They include single or stereo video cameras operated by a SCUBA diver (Davis and Anderson 1989; Bortone et al. 1991), towed by a boat (Schaner et al. 2009) ormounted on a stand that has a fixed field of view over the seabed (Willis and Babcock 2000).Cameras may be equipped with infrared lighting for studies at night or at depths exceeding light penetration.

Strip transects by divers provided more accurate and precise density estimates than the video method (Davis and Anderson 1989).Bortone et al. (1991) stated that the field of vision of video recording was lower than that of human eye resulting to a relatively lower number of recorded species and individuals. A stationary video method (baited underwater video) was applied for the density detection of carnivorous fish (Willis and Babcock 2000) but such method was less efficient than strip transects conducted by divers to estimate abundance and size of shallow-water reef fish in the Mediterranean (Stobart et al. 2007).
Harvey et al. (2001; 2002) showed that the stereo-video camera system (two video cameras calibrated together) provided much more accurate and precise fish length estimates if compared with those determined visually by an experienced diver.Measurements were made using digitally captured images and suitable software programs.

Nowadays, plot sampling-surveys to assess fish populations can be effectuated using underwater video systems assembled on sophisticated vehiclessuch as Remotely Operated Vehicles (ROVs) and Autonomous Underwater Vehicles (AUVs).
A ROV is a vehicle manoeuvred by an operator aboard a vessel. It is linked to the ship by a tether that carries electrical power, video and data signals back and forth between the ship and the vehicle.

An AUV is a self-propelled, untethered, and unmanned vehicle that can operate wholly underwater beyond the control and communication of any support facility (Fernandes et al. 2003).

Both ROVs and AUVs are normally classified into categories based on their size, weight, ability or power and are able to work from shallow to deep waters. They have been largely used in a variety of marine habitats especially where it is not possible to use more conventional methods (Pacunsky and Wayne 2000). ROVs and AUVs have been employed to study marine benthic communities and their habitats (Auster et al. 1995; Parry et al. 2002) and the behaviour of marine animals (Spanier et al. 1995; Patel et al. 2004) but they still are scarcely applied to assess fish population abundance.

Video surveys conducted from underwater vehicles, have to use plot-sampling methods to estimate population densities. However, variations in altitude, pitch, and roll of the vehicles produce video frames of unequal and unknown size, thus making the use of video surveys inaccurate to estimate fish density (Michalopoulos et al. 1993). In fact, an implicit assumption of plot-sampling by ROVs and AUVs is that vehicles maintain a constant altitude from the bottom that allows the estimation of the surveyed area.

To reduce the variability due to the use of ROVs and AUVs for assessing abundance and size structure of fish, some technical expedients have been adopted. Many of the technical aspects related to designing and conducting video surveys forestimating fish density with a small ROV have been discussed by Pacunsky et al. (2008) for rockfish and lingcod: a ROV system was configured to acquire high-quality video, with suitable buoyancy and an efficient umbilical storage and management (Fig. 2.3.5.8). ROV surveys were carried out using a habitat-based, random-stratified design and strip transects was conducted assuming that all organisms within the strip were detected with the same probability (Pacunsky et al. 2008). Stewart and Auster (1989) and Csepp (2005) discussed some transect strategies (transect perpendicular to the shoreline, line and radial transect) for studies using small ROVs. They highlighted advantages and disadvantages of such methods and the importance of some environmental factors (current and habitat complexity) for choosing the more appropriate strategy.

Two red diode lasers mounted on a ROV provide accurate estimations of transect width and size of individual fish (see Tusting and Davis 1992). Pacunsky et al. (2008) reported a very high accuracy (Fig. 2.3.5.9) of this method tested by driving the ROV at different altitudes over a measured grid deployed on a flat bottom. However, as measurements assumed a flat substrate with the ROV flying at a fixed altitude, such method may not be appropriate in high-relief habitats where large variation in camera height can produce significant bias in density and fish size estimates.

Remote video systems provide an alternative or complementary method to plot sampling with SCUBA diving to estimate the relative abundance of marine species and solve the problem of difficult access to divers due to adverse environmental conditions and depths. But, despite the cost of ROVs and AUVs systems, several other constraints limit the use of video surveys to fish study populations. Lighting conditions and scattering particles in the water reduce the visibility to a range of a few meters. Fish response to ROVs and AUVs systems is still poorly known and it could affect quantitative estimates of fish. Imaging processing and data analysis of fish size should be improved in order to obtain more accurate estimates of the demographic structure of fish populations.

2.3.5.3.4.3 Distance sampling

Contributor: Stelios Katsanevakis (HCMR)

Estimation of population density or abundance of fish populations with UVS is often confounded by imperfect detectability of individuals within the surveyed region (e.g., Sale and Sharp 1983; Harmelin-Vivien et al. 1985; Thresher and Gunn 1986; Kulbicki and Sarramégna 1999; Edgar et al. 2004). Distance sampling (DS) is a group of methods for estimating the abundance and/or density of biological populations, properly accounting for detection probability (Buckland et al. 2001, 2004). DS methods are extensions of plot sampling methods: line transects are the extension of strip transects and point transects are the extension of point counts. In DS the extra effort of sampling is to record the distance of each detected individual from the line (in line transects) or point (in point transects). This set of recorded distances is used to estimate detection probability in the surveyed region (Buckland et al. 2001), which is a prerequisite for unbiased density/abundance estimations. Details on the assumptions and modelling of DS data are given in Chapter 2.3.1.

Although for many groups of animals DS methods are the standard approach for abundance estimations, this is not the case for marine fish. The advantages of DS methods in UVS of fish populations (accounting for detectability, providing evidence for responsive movement of fish due to the presence of the divers, abundance estimators with higher statistical power than plot sampling estimators) have been demonstrated in several comparative studies (e.g., Ensign et al. 1995; Kulbicki and Sarramégna 1999). However, plot sampling (mainly strip transects) is still by far the most commonly used UVS approach for density/abundance estimations of marine fish populations, and application of DS methods is rather the exception (e.g., Thresher and Gunn 1986; Letourneur et al. 1998, 2000; Arnal et al. 1999; Kulbicki and Sarramégna 1999; Kulbicki et al. 2007; Preuss et al. 2009; Issaris et al. 2009). Detectability issues in UVS are commonly ignored and usually no evidence is provided to justify the assumption of perfect detectability.

A basic assumption of DS is that individuals are detected at their original location. Conceptually, DS requires a 'snapshot' of fish distribution at the time of the survey. The mobility of fish often causes a violation of this assumption, which can be the source of substantial bias in the estimation of abundance. Random movement of fish (independently of the observer) could cause a small upward bias of the estimated abundance because moving animals are more likely to be detected when they are close to the line, biasing detection distances down. However, if the same individuals are counted more than once on the same sampling unit because of random movement, a substantial upward bias of estimated abundance could occur.

Movement in response to observer is the most problematic and can cause a substantial bias in abundance estimation (Buckland et al. 2001; Fewster et al. 2008). Kulbicki (1998) studied the behaviour of fishes in response to divers surveying line transects, based on a large dataset of 110,000 records of 293 tropical reef species. He classified fish species into four major categories of behaviour: neutral, shy, curious and secretive. Neutral fishes were not apparently affected by the presence of an observer; shy species tended to stay away from the observer and larger numbers of fish would be observed further away from the transect line than directly on the transect path; curious species tended to be attracted by the diver; secretive species were well camouflaged or tended to hide in holes or crevices as an observer approached.

Shy and curious species are of most concern (especially the latter) for their potential to give biased results in DS surveys. Evasive movement prior to detection (shy species) will often be apparent from an examination of the histogram of distance data (Fig. 2.3.5.10B). However, if some fish move a considerable perpendicular distance, while others remain in their original location, it is possible that the effect will not be noticeable from the histogram of distance data. When evasive movement occurs, the estimation of fish abundance will be biased low. If evasive movement is not severe and is such that only animals close to the line move away but are still seen then the bias is expected to be small when the estimator is based on a monotonically decreasing model, because the curve fitting will "smooth" and "average" the detection curve (Burnham et al. 1980). When fish are attracted by the divers (curious species) and are detected after they have moved towards the line, a high frequency of small distances will be evident (Fig. 2.3.5.10C). However, only from a histogram of distance data it is usually impossible to distinguish between a case of responsive movement and a case of a rapid drop of detection probability with distance. When fish are attracted by the divers, the estimation of fish abundance will be biased high. To reduce bias due to fish movement, field procedures should be developed to minimize the amount of such unobserved responsive movement (e.g., the use of rebreathers instead of open circuit breathing devices).

The behaviour of fish species may vary among sites and is greatly affected by disturbance levels. Kulbicki (1998) compared the behaviour of fish among sites of varying disturbance, ranging from highly disturbed areas where divers go frequently and actively chase fishes (spear fishing or netting) to very low disturbance levels where divers are absent or do not disturb fish (natural reserves). Substantial differences were found in the behaviour of fish. The tendency of shy fish to stay away from divers drastically increased with disturbance levels; even species classified as neutral at low disturbance level exhibited evasive movement in highly disturbed areas. This could be problematic in comparisons of population density among regions and may lead to false conclusions regarding the effectiveness of marine reserves (Kulbicki 1998).

Another assumption of DS is that distance measurements are exact. Buckland et al. (2001) stressed the importance of careful and accurate measurements of distance especially near the line. However, in fish UVS distances are often visually estimated and not actually measured. Thresher and Gunn (1986) found important differences in distance estimates between divers. Kulbicki (1998) mentioned that estimations of distance may change with turbidity; in clear waters distances tend to be underestimated, while in turbid waters distances tend to be overestimated. Although with good training divers may achieve sufficiently good accuracy in estimating distances, it might be better to actually measure distances with a measuring tape or a graduated rod. Nonetheless, measuring distances underwater would lead to a substantial deceleration of the divers' speed, which might cause an intensification of problems related to responsive movement of fish (movement of fish should be slow relative to the speed of the observer). Thus, the selection of whether to measure distances or visually estimate them depends on the target species; it seems that for species of low mobility it is beneficial to measure distances, while for species of high mobility it is better to estimate distances visually so that the observers keep moving with a relatively high speed.

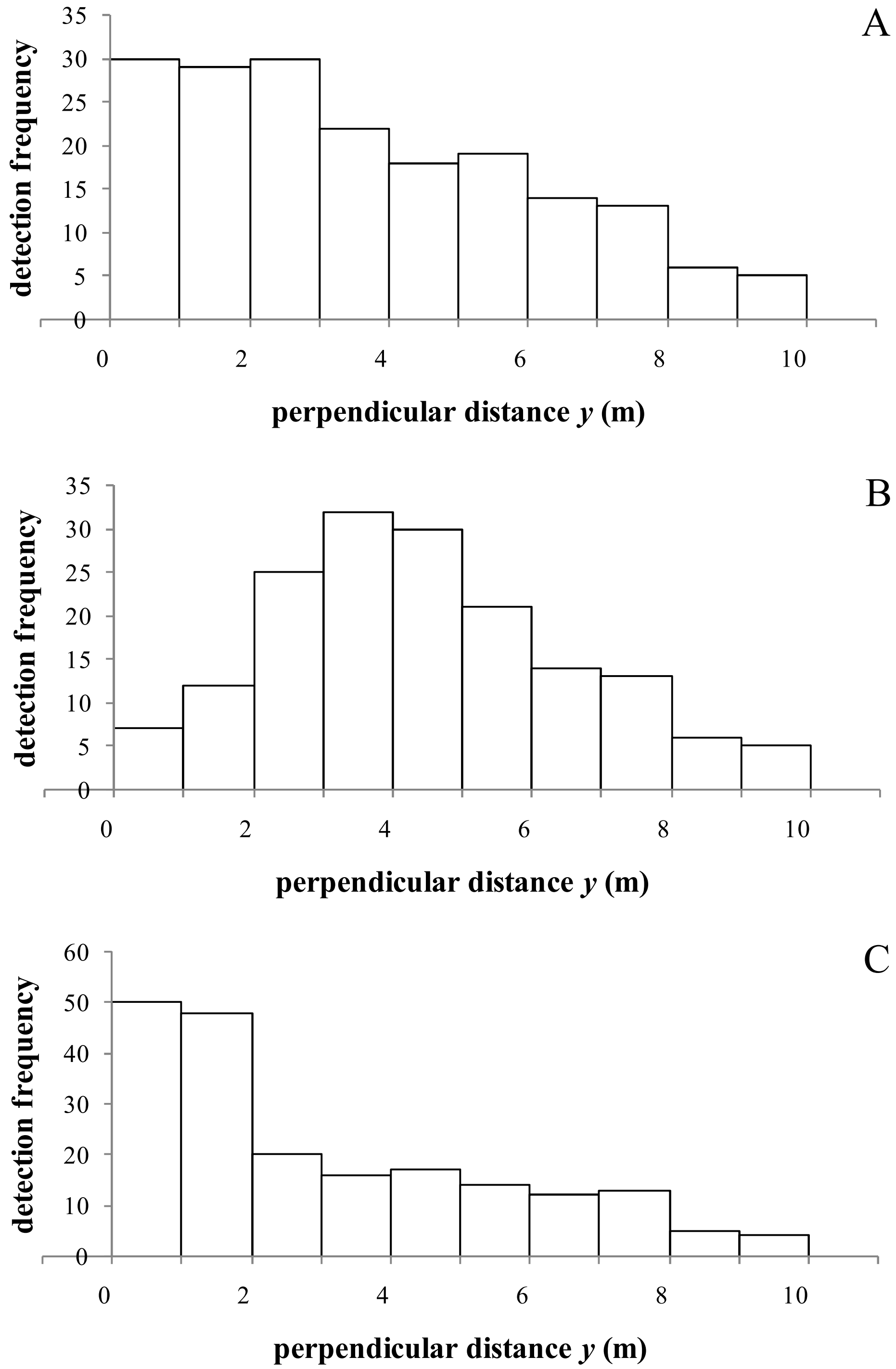

Figure 2.3.5.10: The expected average histogram of recorded distances in a line transect survey, (A) when no movement prior to detection occurred (neutral species); (B) when there was apparent evasive movement prior to detection (shy species); (C) when fish were attracted by the diver (curious species).

The most important assumption of DS is that detectability is certain on the line (i.e., for zero distance) and the field protocols of DS surveys must consider ways to assure that this assumption is met. However, when studying the populations of highly cryptic species, this assumption may be violated. In such cases classical DS underestimate true abundance. Such detectability bias can be overcome in some cases by the combination of mark recapture methods with DS methods (Laake and Borchers 2004). The first study that applied a mark-recapture distance sampling approach for estimating abundance of fish populations was conducted by Issaris et al. (2009) who studied the abundance of two temperate seahorse populations (*Hippocampus hippocampus* and *H. guttulatus*) in eastern Mediterranean. In that study two independent observers undertook the counting of animals and the common detections were treated as recaptures. With that approach, detectability on the line was estimated and accounted for in the estimation of detection probability and abundance.

2.3.5.3.4.4 Fixed-time swims or SCUBA surveys

Contributor: Giovanni D'Anna (CNR-IAMC)

Fish mobility and their quick adjustment to environmental factors affect the distribution of fish populations making the sampling procedures and the unbiased estimation of fish abundance very difficult. To make such unbiased estimations of fish abundance/density with UVS, the diver has to move along a predetermined transect and either strive to record all individuals within a predefined distance from the line (plot sampling) or put some extra effort to record the distance from the line of each detected individual and thus estimate detectability (distance sampling) (see paragraphs 2.3.5.3.4.1and 2.3.5.3.4.1 for details).

During fixed-time swims or SCUBA surveys (FTS) divers do not have to follow defined transect lines but are free to record all fish encountered during a predetermined time. Such methods involve counts of fish species for a fixed duration on random paths across the selected area. The idea is that the elimination of time-consuming procedures gives the diver more time and the opportunity to census more species (Kimmel 1985).

FTS are mostly used to provide information on species composition and spatial distribution of fish assemblages but they are not recommended for the estimation of fish abundance(De Martini and Roberts 1982; Harmelin -Vivien et al. 1985; Kimmel 1985; Sanderson and Solonsky 1986).

Comparisons among UVS methods highlighted that divers using FTS collected the highest number of species per survey but strip transects were more efficient in estimating abundance (Bortone et al. 1986, 1989; Michalopoulos et al. 1993). These results show the inaccuracy of FTS for assessing faunal abundance but put in evidence the important role these methodscould play in descriptive studies of fish assemblages.

FTS methods have been adopted to monitor the fish assemblage composition (qualitative inventory) in coral reefs (GBRMPA 1979), *Posidonia oceanica* meadows (Francour 1999), marine protected areas (Harmelin 1999; Vacchi and La Mesa 1999) and artificial reef areas (Charbonnell et al. 1996).

In the Scandola marine reserve, Miniconi et al. (1990) recorded 34 species considered rare or infrequent and highlighted that65% of themwere sampled using FTS. Similar results were reported by Camus et al. (1987) in their fish inventory of Lavezzi Island. La Mesa and Vacchi (1999) used random swims and SCUBA surveys to collect additional qualitative data on the fish assemblage at the Ustica Island marine reserve. FTS were adapted to the characteristics ofartificial reefs in Portugal to describe the associated fish fauna (Santos et al. 1995).

Fixed-time random counts have also been employed to characterize the microhabitats used by littoral fish (Harmelin et al. 1995; La Mesa et al. 2006) and to gain qualitative information at sea on the spatial distribution and behaviour of released tagged hatchery-reared fish (D'Anna et al. 2004). Despite of the effectiveness in depicting species composition of fish assemblages, fixed-time counts provide only qualitative data insufficient to assessfish populations.

2.3.5.3.4.5 Occupancy estimation and modelling

Contributor: Stelios Katsanevakis (HCMR)

When studying the status of a fish population, the most commonly used state variable is population density or abundance. Unbiased abundance estimations of fish populations often require substantial effort or might be extremely difficult for practical reasons, e.g. for rare or highly cryptic species. Another useful state variable for single-species studies is occupancy, defined as the proportion of an area or sampling unit that is occupied by the species (MacKenzie et al. 2006). Estimation of occupancy is often still possible when abundance estimation is very effort-intensive, costly or impracticable (MacKenzie et al. 2004, 2005). Thus, sometimes occupancy is viewed as a surrogate for abundance, although it is the natural state variable to be used in studies of spatial distribution or alien-species invasions (MacKenzie et al. 2006). Methods for estimating occupancy are also valuable in community studies of species richness, in which the number of fish species that are present at any point in time or space is the state variable of interest (Dorazio and Royle 2005; Dorazio et al. 2006).

Simple presence/absence underwater surveys (either by plot sampling or fixed-time swims or dives) have been used in the past to estimate occupancy. However, these surveys routinely assume that detectability is perfect (i.e., equal to one) and thus commonly underestimate occupancy and species richness of fish communities (MacKenzie et al. 2006; MacNeil et al. 2008a, 2008b). In an assessment of the structure of coral reef fish communities in Tanzania, detection probabilities of 47 reef fish families varied substantially among fish family groups and among sites, and ranged between 0.05 and 0.54 (MacNeil et al. 2008a). That study also found that schooling behaviour, reef fish functional group, and fish length can affect reef fish detectability. The very low detectability figures reported by MacNeil et al. (2008a) highlight the need to account for fish detectability in presence/absence surveys, to make unbiased estimations of species occurrence. Such methods that account for imperfect detection in occupancy estimations are based on replicated sampling of the same sites and have been recently developed by MacKenzie and his colleagues (MacKenzie et al. 2002, 2004, 2005, 2006).
Although the need to account for detectability in fish occupancy or species richness surveys has been demonstrated, the developed methods for estimating occupancy when detection probabilities are less than one (MacKenzie et al. 2006) have not yet been applied for fish populations, except for a study by Issaris et al. (2010), in which the occupancy of the alien fish *Siganus luridus* was estimated in the National Marine Park of Zakynthos (Ionian Sea, Greece). In that study, occupancy models according to MacKenzie et al. (2006) were developed that accounted for detectability and the effect of covariates (observer, protection zone) on detectability or occupancy.
This is a promising method for monitoring fish populations and communities, as it is a substantial improvement of current present/absence surveys, which often fail to provide unbiased estimates of occupancy or species richness.

2.3.5.3.4.6 Rapid visual techniques

Contributor: Yiannis Issaris (HCMR)

The Rapid Visual Technique (RVT) determines species diversity and provides information on relative abundance, presented as frequency of occurrence. Observers conduct this survey at a constant speed for a fixed time (e.g. 50 mins), swimming randomly around the survey area, usually limited down to specific habitats under study and recording as many fish species as possible. The survey period is divided into intervals (e.g. 10 mins). Each species sighted is recorded once when first seen in the defined time interval.

This method is based on the assumption that the probability of encountering a species increases with its abundance. The more common the species, the sooner the observer is likely to encounter it, therefore species occurring in early time intervals are the most abundant in the community.

To estimate their relative abundance, each species is given a score based on the time interval within which it was seen. For example, in the case of 5 intervals of 10 minutes each, fishes occurring in the first 10-minute interval receive a score of 5, those occurring in the second interval are scored 4 and so on with the fifth interval fish scoring 1. To obtain reliable data, replicate sample censuses are conducted at each site. Species scores among replicate 50 min samples are summed to indicate frequency of occurrence. Variations of this technique appear in Jones and Thompson (1978) and in Kimmel (1985).

Fish RVT have low equipment requirements and are time-efficient compared to other UVS methods. They are useful for preliminary surveys of species diversity, providing valuable information for the design of long-term abundance monitoring studies. However, this method over-emphasises the importance of widespread albeit rarer species, while underestimates patchy but abundant species (DeMartini and Roberts 1982, Hill and Wilkinson 2004). Moreover, species diversity is usually underestimated, since many cryptic species remain undetected and are usually overlooked.

2.3.5.4 Monitoring of spatial distribution of fish populations

Two methods will be analyzed in this section: (i) spatial analysis of geo-referenced data, (ii) acoustic/radio telemetry.

2.3.5.4.1 Spatial analysis of geo-referenced data

Contributor: Stelios Katsanevakis (HCMR)

Until recently, fisheries management was almost exclusively based on a single-species approach utilizing models devoid of spatial variation and assuming that fish distribution and exploitation patterns were spatially homogenous (Hilborn and Walters 1992; Caddy 2009). Recent research directions and new legislation are introducing the concepts of spatial and ecosystem-based fisheries management, to take into account the heterogeneity of the marine environment, the patchiness of fish distribution and the processes at multiple scales that influence fish stocks (Browman and Stergiou 2005; EC 2008). Knowledge of the spatial component of marine biological resources is now recognized as a prerequisite for their sustainable exploitation and effective fisheries management (Booth 2000; Valavanis 2002, 2008; Wilen 2004). Special focus has been given, among others, to the mapping of relative abundance of fish, identification of essential fish habitats, and mapping of species migration corridors (Valavanis 2002, 2008).

Geo-referenced raw catch and effort data from commercial fisheries, fisheries surveys or visual surveys may be analyzed to provide insight on fish spatial distribution and essential fish habitats. The simplest analysis of spatial distribution of fish populations is based on stratified sampling (with depth being the most common stratification factor) and the aim is to infer abundance differences among strata (usually variation of abundance among bathymetric strata) (e.g., Machias et al. 1998; Kallianiotis et al. 2000; Abelló et al. 2002). However, greater insight is gained when fish abundance is functionally related to environmental and spatial variables that are expected to affect directly or indirectly fish distribution. Relevant environmental data may be acquired by interpreted satellite images (see Valavanis et al. 2008 for a table of sources of such data, such as sea surface temperature, sea surface chlorophyll-a, photosynthetically active radiation, sea surface salinity) or by data collected during survey trips in the study area (e.g., sediment characteristics, sea bottom temperature and salinity, benthic habitat types).

Extrinsic factors influencing the spatial distribution of demersal fish are depth, which has been stated to be the main gradientalong which faunal changes occur when analyzing shelf and upper slope assemblages (Moranta et al. 1998; Demestre et al. 2000; Kallianiotis et al. 2000; Katsanevakis and Maravelias 2009; Katsanevakis et al. 2009), type of substratum (Mahon and Smith 1989; McCormick 1995; Gaertner et al. 1999; Demestre et al. 2000; Katsanevakis et al. 2009), and physical characteristics (such as temperature and salinity) of the water masses (Mahon and Smith 1989; Maravelias et al. 2007; Katsanevakis et al. 2009). Depth is important mostly because of its indirect effect on fish abundance as it is correlated to many crucial environmental parameters such as light intensity, temperature, nutrient concentration, primary and secondary productivity; in the absence of such data in studies of fish spatial distribution, depth is concluded to be the main predictor variable of population density (Katsanevakis et al. 2009). Intrinsic biotic interactions among assemblage members also affect the abundance of demersal species but seem to be of less importance (Mahon and Smith 1989; Sale et al. 1994).

The spatial distribution of pelagic fish has been correlated to sea surface temperature (Laurs and Lynn, 1977; Carey and Scharold 1990; Daskalov et al. 2003; Damalas et al. 2007; Giannoulaki et al. 2008), thermal fronts/gradients (Podesta et al. 1993; Maravelias and Reid 1997), sea surface salinity (Maravelias and Reid 1997), depth (Giannoulaki et al. 2008), sea currents (Podesta et al. 1993), zooplankton biomass (Maravelias and Reid 1997), bottom topography (Bigelow et al. 1999), distance from coast (Strasburg 1958; Megalofonou et al. 2009), lunar cycle (Bigelow et al. 1999; Damalas et al. 2007; Megalofonou et al. 2009), photosynthetically active radiation (Giannoulaki et al. 2008), and wind intensity (Bigelow et al. 1999; Daskalov et al. 2003).

A variety of analytical tools exist to infer relationships between abundance or relative abundance and environmental variables. Frequently used analytical tools, such as multiple linear regression, assume linearity, normally distributed error terms, and independent observations. These assumptions are quite restrictive, as the dependence of population density on spatial and environmental parameters is expected to be more complex (Lehmann et al. 2002; Venables and

Dichmont 2004; Katsanevakis et al. 2009), and thus there is a need for appropriate non-linear flexible modelling techniques. Such statistical methodologies have been developed in recent years and are now widely applied to model the spatial distribution of fish populations. Such methods that allow the fitting of statistical models that better agree with ecological theory and are not restricted by convenient mathematical formulas include generalized linear models (GLM; McCullagh and Nedler 1989), generalized linear mixed models (GLMM; Breslow and Clayton 1993), generalized additive models (GAM; Hastie and Tibshirani 1990), multiple adaptive regression splines (MARS; Friedman 1991), boosted regression trees (BRT; Friedman 2002), and artificial neural networks (ANN; Rumelhart et al. 1986).

Among them, GAM is the most common and well developed method for modelling fish abundance and spatial distribution, and is widely applied in fisheries science (Valavanis et al. 2008). GAM are increasingly used in ecological studies to study spatial distribution and abundance of demersal (e.g., Maravelias et al. 2007; Katsanevakis et al. 2009), small pelagic (e.g., Maravelias and Reid 1997; Daskalov et al. 2003; Giannoulaki et al. 2008) and large pelagic fish (e.g., Bigelow et al. 1999; Damalas et al. 2007) as the ecological interpretability of the non-parametric response curves and the flexibility of GAM to fit the data closely are advantageous characteristics (Fig. 2.3.5.11). Several evaluations of alternative approaches to model fish abundance demonstrated the good performance and properties of GAM as well as their advantages in ecological interpretability, although in some cases other methods may have some comparative advantages (Lehmann et al. 2002; Leathwick et al. 2006; Valavanis et al. 2008).

Maps of relative abundance or potential essential fish habitats are produced by the application of the developed fish abundance predictive models within GIS grids. At each cell of the grid, a prediction of fish abundance (or index of abundance or probability of presence, depending on the kind of model) is produced, utilizing the known values of the predictor variables as input to the model. The predicted values can be directly mapped (Fig. 2.3.5.12), whereas the extent and resolution of the predicted area depend on the dataset used to build the model and the range and resolution of the predictor variables within the study area.

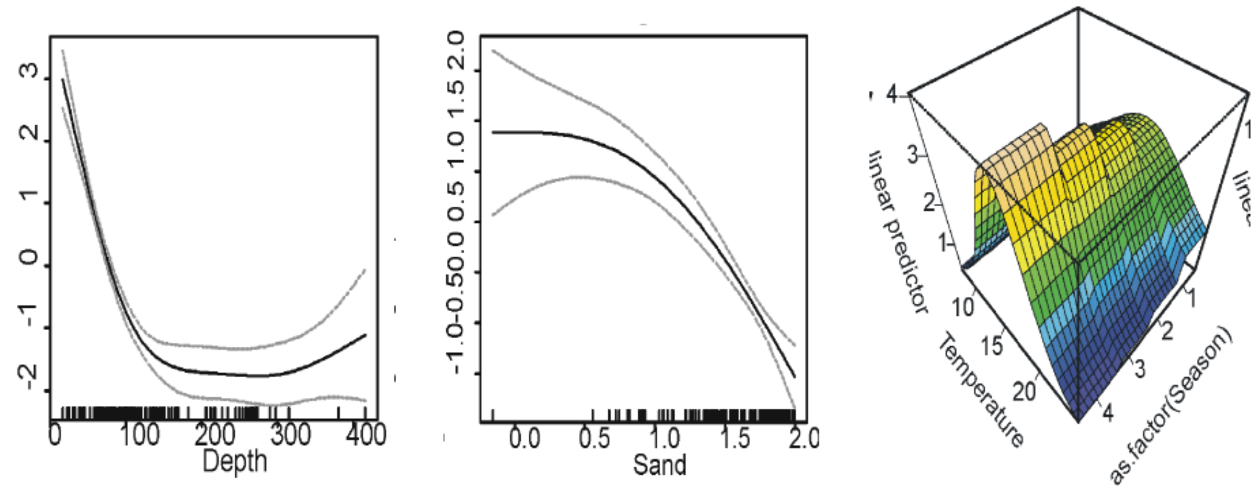

Figure 2.3.5.11: An example of the output of a GAM. The relative abundance of whiting (*Merlangius merlangus euxinus*) in the Aegean Sea has been modelled using depth, sand content (Sand: log of the dry weight percentage of sand in the sediment) and bottom temperature (interacted with season: 1 – winter, 2 – spring, 3 – summer, 4 – autumn) as predictor variables (from Katsanevakis et al. 2009). There is a clear preference of the species for shallow, sandy bottoms and temperatures around 15 ^oC.

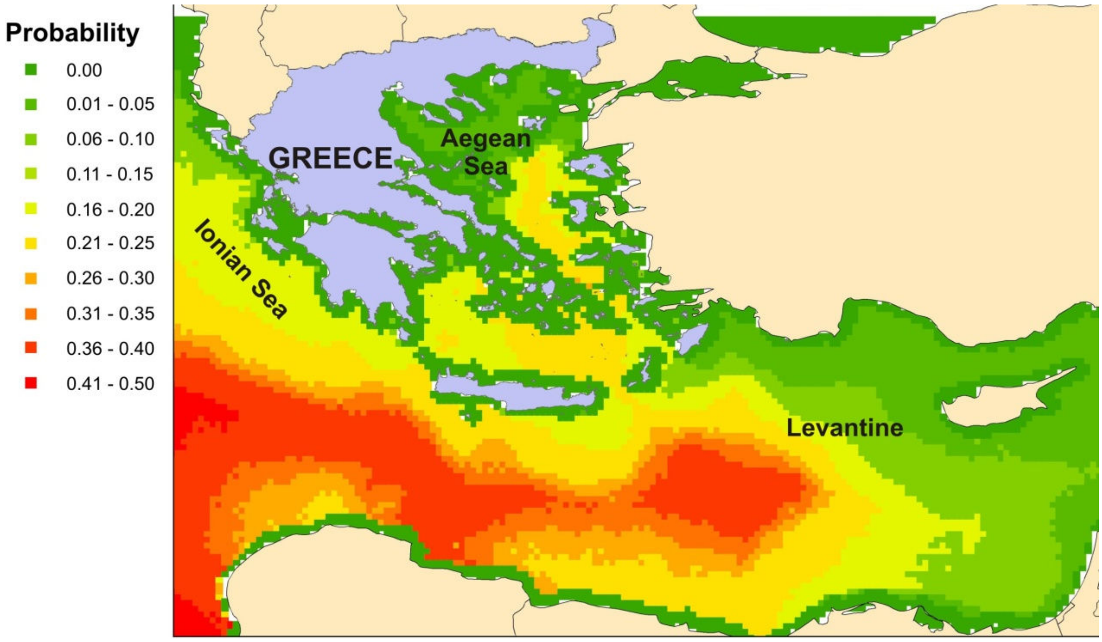

Figure 2.3.5.12: GAM predicted spatial probability of catching a blue shark (*Prionace glauca*) in the surface longline fishery of eastern Mediterranean Sea (which may be viewed as an index of abundance) during August (resolution is 1/10 of a degree - probability refers to a unit effort of 1000 hooks) (from Damalas 2009, with permission).

2.3.5.4.2 Telemetry systems

Contributor: Giovanni D'Anna (CNR-IAMC)

The pattern of spatial dynamics in individuals can be considered one of the most important demographic variables influencing the structure of fish populations (Jones 2005). Ignoring these spatial trends can often provide inaccurate relative abundance estimates and lead to misleading interpretations of the various aspects of a species biology, such as its distribution, growth, reproductive and feeding patterns. Moreover, measurement of the use of space through time by fish enables understanding of their home range, homing and pattern of activity which are crucial to assisting management and conservation of fish stocks (Kramer and Chapman, 1999; Ormond and Gore 2005; Chateau and Laurent Wantiez 2008; Knight et al. 2008).

Mark-and-recapture with conventional tags has been one of the most commonly used methods to study fish movements but it has several limitations (Nielsen 1992). Such methods do not allow measurement of the exact distance travelled by fish thus no quantitative analysis of fish movement patterns can be made (Zeller 1999). Better estimates of true movement can therefore be obtained if continuous tracking methods are employed or if fish are monitored frequently over time.

The use of electronic tags is one of the most important developments for studying the behaviour and spatial distribution of fish. Underwater telemetry makes use of transmitters (*pingers*) attached to animal bodies to track them in their natural environment by means of manual or automated receivers (also called *hydrophones* when placed underwater) (Stasko and Pincock 1977; Lucas and Baras 2000). Advances in telemetry equipment and techniques enable the precise and simultaneous positioning of fish as well as remote monitoring of physiological and behavioural variables (biotelemetry) (Cooke et al. 2004).

Several telemetry techniques exist to study the distribution and movements of fish in marine and freshwater environments. They differ each other by the means of production, transmission and detection of signals which can be acoustic or electromagnetic (McCleave 1978; Collins et al. 1997). The most appropriate method depends on the specific limitations of the species and its habitat. In the following sections, the application of ultrasonic, radio and electromagnetic telemetry systems are reviewed.

The general assumptions behind the use of such techniques are (i) that tags do not change the physical condition or behaviour of tagged fish, (ii) that fish remain tagged for the duration of the study, and (iii) that individuals carrying transmitters are representative of the population as a whole (Nielsen 1992).

2.3.5.4.2.1 Ultrasonic/acoustic telemetry

Contributor: Giovanni D'Anna (CNR-IAMC)

What isultrasonic/acoustic telemetry and how does it work?
Ultrasonic/acoustic telemetry (UT) uses sound waves with a frequency range (20-300 kHz) which travels well through fresh, brackish and salty water (Prince and Maughan 1978). Due to its strong signals, UT works well also for tagged fish that travel far, live in deep water, or make wide vertical movements. Variations in the strength and pulse rate of ultrasonic signals are easy to detect allowing very precise location of tagged animals (Stasko and Pincock 1977). The use of ultrasonic systems requires suspended underwater receivers (hydrophones) and of sophisticated transmitters that are generally expensive.
Two main methods are used to track fish: manual tracking (MT) and automated tracking (AT).

Manual tracking

MT normally involves the use of a hydrophone mounted to a boat to follow a signal from an ultrasonic transmitter attached to an individual fish (Fig. 2.3.5.13). Omnidirectional hydrophones allow to detect only the presence of the tagged individual, while direction receivers allot to track its position and movements. The localization of tagged fish is recorded and a track of fish movements is compiled (Holland et al 1992). The path of fish is recorded by GPS way-points or landmarks. However many factors can affect the accuracy of fish localization. MT can provide tracks of large fish (e.g. marlin, tuna, sharks) with a spatial error of <100 m during time periods of <10 days and over a distances of <100 km (Klimley et al. 2001). The error of fish position using MT reduces to only a few meters when tracking demersal fish moving in complex habitats.

Figure 2.3.5.13: Ultrasonic receiver for manual tracking with an omnidirectional hydrophone (from www.vemco.com).

Automated tracking

Recent advances in acoustic telemetry (Voegeli et al. 1998; 2001) allow simultaneous collection of data on multiple tagged animals without the requirement of constant contact with tagged animals. These systems generally consist of moore-mounted receivers equipped with at least three omnidirectional hydrophones (Fig. 2.3.5.14) which can be radio- or cable-linked and controlled by a base-station computer (Voegeli et al. 2001). By measuring arrival times of the signal at receivers, the position and movement of the tagged fish can be determined (Heupel and Heuter 2001). Performance of acoustic positioning and telemetry systems are discussed by O'Dor et al. (1998). Studies using cable-

or radio-linked systems reported location accuracy of 1-2 m when waves are detected by at least three hydrophones (Cote et al 2003).
The introduction of relatively low-cost, moored data-logging acoustic receivers has provided opportunities for tracking marine organisms over small (hundreds of metres) and large (hundreds of kilometres) scales and for long-term monitoring movements. Such omnidirectional receivers are single-frequency independent units planned only to record the presence of a tagged fish within the range of the receiver along with the date and time of signal reception (Lacroix and Voegeli 2000). They are also able of recording telemetered data such as temperature and depth, when appropriate transmitters are employed.

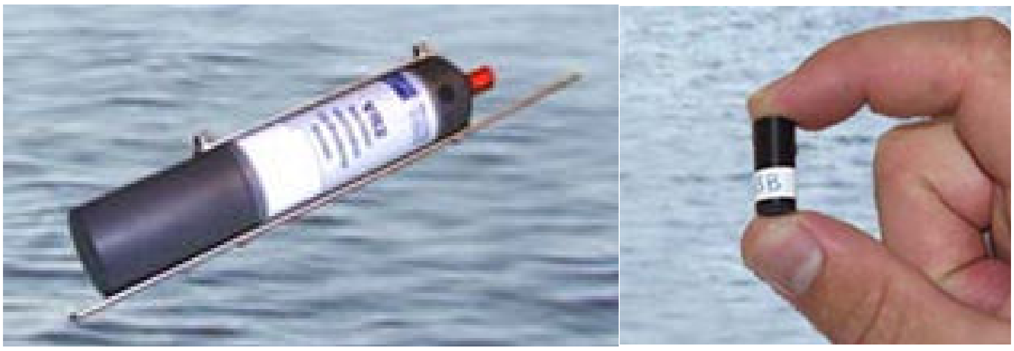

Figure 2.3.5.14: Automated omnidirectional receiver (left); miniaturized ultrasonic transmitter (*pinger*) (right) (from www.vemco.com).

How is ultrasonic/acoustic telemetry used to monitor the spatial distribution of fish?
Since 1960s, UT has been mainly applied to define the fine scale movements of marine and freshwater fish. MT has been used to study highly migratory pelagic fish such as tunas, carangids, as well as home range, diel movements and use of habitat of sharks, flatfish and other demersal fish (Wirjoatmodjo and Pitcher 1984; Holland et al. 1990; Brill et al. 1993; Morrissey and Gruber 1993; Dagorn et al. 2000; Lutcavage et al. 2000; Heithaus et al 2002; Windle and Rose 2005). MT has been used in coastal areas to follow fish in shallow-water complex rocky habitats, where they provided information on home range, habitat use and diel activity (Matthews et al., 1990; Matthews 1992; Holland et al. 1993; Zeller 1997; Lowry and Suthers 1998; Arendt et al 2001; Lowe et al 2003; Meyer and Holland 2005).

In the Mediterranean Sea, Jadot et al. (2006) used MT to investigate diel activity patterns and home range of *Salpa salpa* in a *Posidonia oceanica* meadow, while the homing behaviour of groupers in a marine protected area (MPA) was reported by Spedicato et al. (2005). A constraint encountered studying fish movement in such complex habitats is the considerable reduction of the detection range when a tagged fish is hidden behind rocks and/or is associated with high-relief rocks.

In situ receivers have been applied worldwide for automated tracking of fish. AT with unlinked receivers has been mainly used to continuously monitor long-term movement patterns, home range and site fidelity of fish within a variety of habitats (Huepel et al. 2004; Bruce et al. 2005; Humston et al. 2005; Garla et al. 2006; Lindholm et al 2006; Abecasis and Erzini 2008; Collin et al. 2008; Abecasis et al. 2009). AT has been recently applied to study movement patterns of white seabream in an artificial reef area (D'Anna et al. in press) (Fig. 2.3.5.15) after prior estimate of the error position from an array of automated receivers (Giacalone et al. 2005).

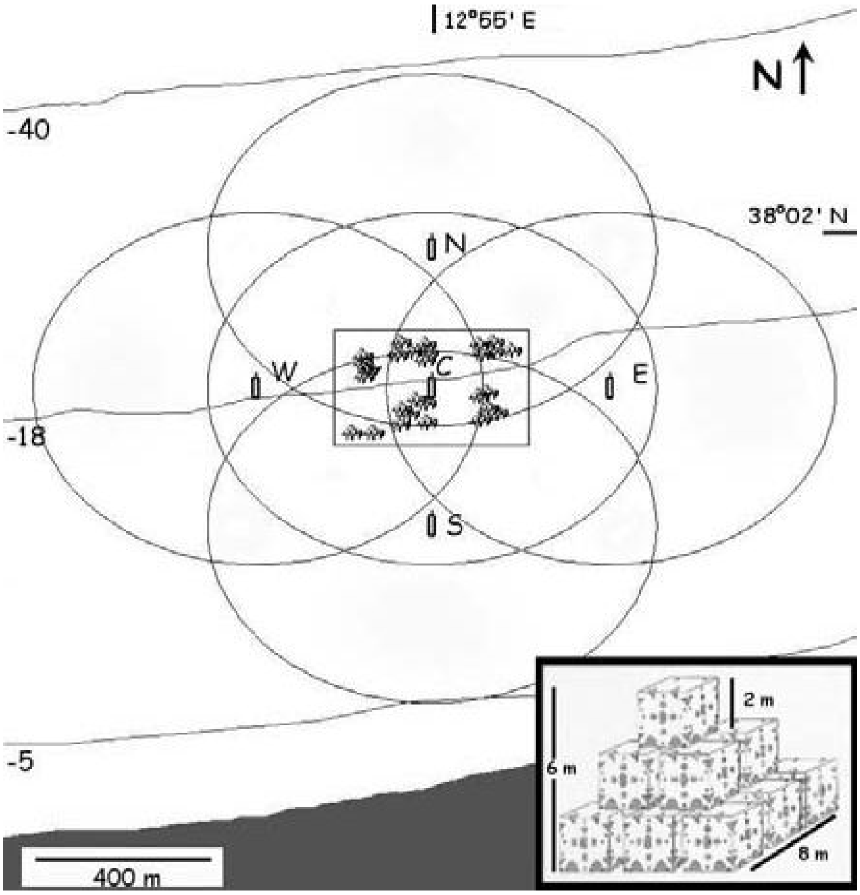

Figure 2.3.5.15: Receivers positioning design using automated omnidirectional hydrophones to study fish movement in an artificial reef area: circles represent the receivers (C, N, E, S, W) detection range, the rectangle indicates the artificial reef area (from Giacalone et al. 2004).

In situ, both independent and linked receivers have been applied to studies inside MPAs. AT has been employed to examine the interactions between fish populations of protected and unprotected areas (Egli and Babcock 2004; Chateau and Wantiez 2009), to study behavior and territoriality of endangered fish (Lembo et al 2002; Pastor et al. 2009), to follow diel and seasonal movements and home range of young and adult fish in relation to MPA borders (Parsons et al. 2003; Garla and Chapman 2006; Meyer et al. 2007). Generally, results from these studies show the inadequate design of MPAs to preserve nursery, foraging and spawning areas of the surveyed fish.

The opportunity offered by AT to get continuous data on long-term movement patterns of fish and the use of MT for a fine-scale fish tracking (Bellquist et al. 2008), can contribute to a better design and location of MPAs, and to assess their effectiveness (Kramer and Chapman 1999; Halpern 2003; Botsford et al. 2003; MPA News 2004; Chapman et al 2005; Ormond and Gore 2005).

Issues for consideration with ultrasonic/acoustic telemetry

MT requires transmitters that emit a signal (or "ping" sound) every few seconds, which results in a relatively short battery life. AT involves transmitters that ping less frequently, allowing for a battery life of one year or more. However, physical obstacles, hydrostatic pressure, thermal stratification, turbulence, turbidity and habitat complexity interfere with ultrasonic signals reducing the range of reception (Giacalone et al. 2005). Studies using MT tend to be more labour- and time-demanding, permit to follow only a small number of individuals and are not able to provide data on long-term movements (Heupel and Hueter, 2001).

Despite the higher location accuracy, AT with linked, in-situ acoustic receivers is not used very often, mainly due to the high cost and the limited number of receivers that can be linked in the array. Moreover, location estimates are mostly limited to the area between the linked receivers (Klimley et al. 2001; Voegeli et al 2001) and, if fish swim outside the range of one of the linked receivers, fish localization is inaccurate or impossible (Simpfendorfer et al 2002).

Heupel et al (2006) reported and discussed the main approaches which have been adopted to track fish movements and the importance of an appropriate design and deployment of the independent receivers arrays. Recent applications, based on the detection ranges of overlapping automated receivers, has increased the accuracy in locating fish throughout small areas (Arendt et al 2001). Simpfendorfer et al. (2002) estimated the location of tagged fish by means of a weighted mean algorithm which uses the number of detections recorded by overlapping automated receivers. One implication of this method is that the mean position estimate falls within the minimum convex polygon described by the receiver locations. The same authors highlighted the importance of a correct arrangement of receivers throughout the study area to increase the positional accuracy.

2.3.5.4.2.2 Passive Integrated Transponder telemetry

Contributors: Carolyn M. Knight (NIVA) and Kate L. Hawley (NIVA)

What is PIT telemetry?

Passive integrated transponder (PIT) telemetry has become increasingly widely used to monitor fish movements. PIT telemetry combines active telemetry through powered detectors with small, cheap, passive tags enabling substantial numbers of large or small fish to be continuously monitored with automated receivers. This form of telemetry has enabled researchers to monitor movements and spatial distributions of individuals as well as whole populations on a finer spatial and temporal scale than is possible with traditional tagging methods.

How does it work?

A PIT tag is an integrated circuit chip and coil antenna normally encapsulated in a glass cylinder (Whitfield Gibbons and Kimberly 2004) (Fig. 2.3.5.16). Each tag stores a unique alphanumeric code, allowing the tagged individual to be identified, even on a global scale. The term *passive* indicates that there is no battery in the tag, thus they have a theoretically indefinite lifetime. The tag is powered by an external energy source, usually a handheld or stationary reader (Fig. 2.3.5.17) that generates a low radio-frequency electromagnetic field used to momentarily activate the tag (Prentice et al. 1990a; 1990b; 1990c). Once activated the tag transmits information to the reader where the code is displayed or stored (Prentice and Park 1983).

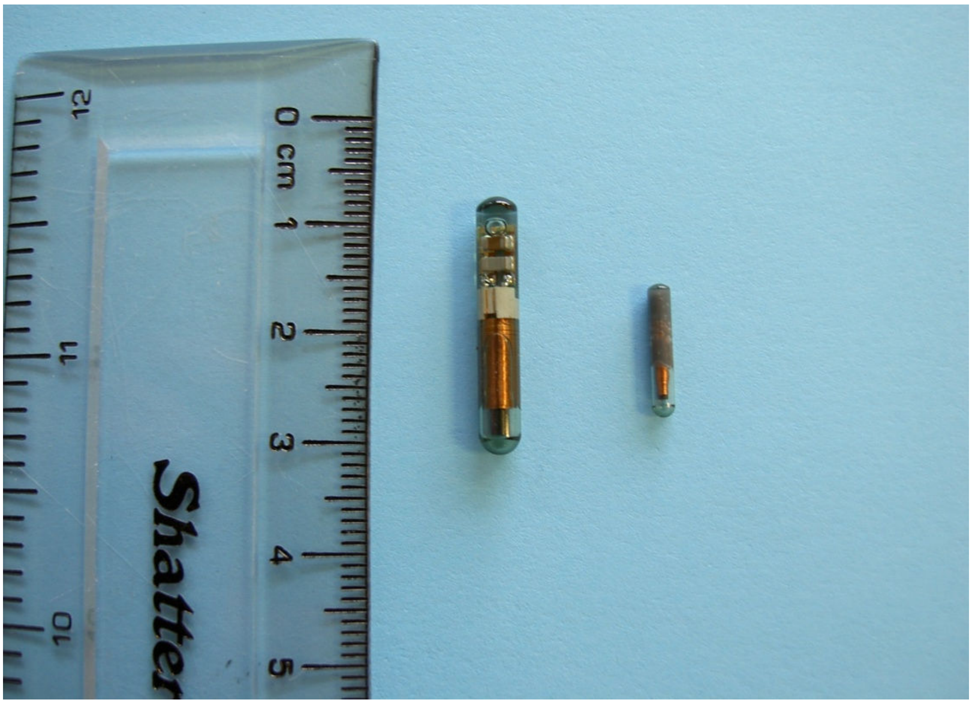

Figure 2.3.5.16: Typical PIT tags.

Figure 2.3.5.17: PIT detector station in a Norwegian stream. The wires seen crossing the river form part of the loop antennae.

PIT tags are generally cylindrical in shape and their size can vary from 12 to 32 mm in length and from 2 to 3.5 mm in width (Prentice et al. 1990a; 1990b; 1990c). The tag detection range depends on the size and orientation of the tag antenna coil and on the overall size of the tag (Roussel et al. 2000) but can reach a maximum of approximately 1 m either side of the antenna under optimum conditions.

How is PIT telemetry used to monitor the spatial distribution of fish?

PIT tags were first implemented in fisheries studies as a method of identifying individual fish and monitoring their movement (Prentice and Park 1983). Use in fisheries research has diversified from early work using tags merely as identifiers upon recapture. Studies now use automated detectors to monitor fish movements constantly, and without the invasion of recapture (Armstrong et al. 2001; Aarestrup et al. 2003; Riley et al. 2003; Scruton et al. 2003; Bolland et al. 2009). This enables assessment of diel activity and differences between species by registering the movement of tagged individuals past fixed monitoring stations. Currently, the PIT tag must pass very close to a detector (within <90 cm) and so this limits use to shallow streams, areas which can be interlaced with readers without disturbance, or areas in which the animal is likely to stop and rest.

PIT tags are currently used to provide information at the individual, population and community levels (Lucas et al. 1999; Gardeur et al. 2001; Sloman et al. 2002; Paulsen and Fisher 2003) and can do so on a long-term basis. At the individual level they can be used as identifiers to provide information on growth (Gardeur et al. 2001), individual migrations and movements since last recapture (Smithson and Johnson 1999) and information of individual movements in and out of monitored areas (Lucas et al. 1999). Individually marked animals can provide information on habitat use (Teixeira and Cortes 2007), temporal information on movement at the population level (Metcalfe et al. 1999) and interactions of different species (Collis et al. 2001), in addition to elucidating social relationships (Sloman et al. 2002) and spatial distributions of the population (Pope and Matthews 2001) while taking into account individual life history (Skalski et al. 1998). Through all this information and more, PIT telemetry has the potential to explain more widespread and complex phenomena within an ecosystem such as food web information (Petersen and Barfoot 2003; Whitfield Gibbons and Andrews 2004) or anthropogenic pressures (Armstrong et al. 1998; Calles and Greenberg 2007) and thus influence management practices and conservation objectives (Pope and Matthews 2001; Scruton et al. 2003).

The use of PIT telemetry to monitor the spatial distribution of fish is one of the most effective applications of this method. In-river migrations can be followed providing valuable insights for fisheries managers and researchers using either a series of automated detectors(Bolland et al. 2009) or a portable detector (Breen et al. 2009). On a large scale PIT telemetry has been utilised to assess the impact of bird predation on salmonid populations in the Columbia River. Information was gathered from the expelled PIT tags found within the predatory bird colonies (Collis et al. 2001).

PIT telemetry has also been applied to monitor the spatial distributions of small, young-of-year fish where radio or acoustic tags would be too large. This technique has facilitated novel insights into Atlantic salmon (*Salmo salar*) parr migrations (Pinder et al. 2007) and habitat use of young-of-year pike (*Esox lucius*) (Cucherousset et al. 2009), where previously only traditional recapture methods have been possible with small fish.

Advantages of the technique

PIT telemetry provides a relatively quickly administered, reliable and cheap method of marking and, in particular, constantly monitoring large numbers of fish. PIT tags are a non-destructive sampling technique and tags can theoretically last forever and provide information throughout the lifetime of the animal. PIT telemetry is able to provide more spatial and temporal information about the movements of individual fish than simple external tagging, without the prohibitive time and economic expenses of radio telemetry. Due to the small size of tags the minimum fish size for individuals to be tagged can be very low. Cucherousset et al. (2009) for example, tagged young of the year pike to a minimum size of 42 mm without apparently experiencing any negative effects.

An advantage of PIT tagging as opposed to other tagging methods is that tags can be internal rather than external and thus do not suffer the drawbacks of external marks such as healing of scars and damage of tags (Whitfield Gibbons and Andrews 2004). Additionally, PIT tags are well suited to monitoring fisheries as tags are invisible to anglers, thus avoiding the problem of non-reporting of tags (Parker and Rankin 2003). PIT telemetry enables the behaviour and social interactions of individuals to be monitored without the alteration of their behaviour due to visible tags (Sloman et al. 2002).

Issues for consideration with PIT telemetry

A limitation to PIT technology is that collisions in signal sending may occur when a large number of fish pass through at the same time, for example in a shoal, and some tags may be missed (Castro Santos et al. 1996). The reason for this is that though PIT readers have a high read rate (of several times per second) they are only able to record one tag at a time (Lucas and Baras 2000). Thus, if two tags simultaneously pass a reader only one may be recorded (albeit several times over), while the other could pass through unrecorded.

The chief restraint limiting expansion of the uses of PIT telemetry for monitoring spatial distribution of fish populations however, is the requirement for the detector and tag to be in very close proximity. This limits the breadth of situations in which it can be used (Armstrong et al. 2001). Possibilities expand as the technology develops (Ibbotson et al. 2004), but will never reach the distances achieved with active forms of telemetry such as radio and acoustic telemetry (Whitfield Gibbons and Andrews 2004). However, because the automated monitoring of PIT tags is passive, it is much less labour- and cost- intensive than radio-tracking and enables the possibility of marking fish at the population level, while still providing reasonably detailed information (Whitfield Gibbons and Andrews 2004).

2.3.5.4.2.3. Radio telemetry

Contributors: Carolyn M. Knight (NIVA) and Kate L. Hawley (NIVA)

What is radio telemetry and how does it work?

Radio telemetry (RT) has a relatively long history as a tool for monitoring wild animal populations. It has been used on many animal groups to provide data on spatial behaviour and demography (White and Garrott 1990; Kenward 2001; Hodder et al. 2007). In its simplest form, RT is based on a VHF radio signal emitted from an active tag which, when within range, is detected by a radio-antenna receiver (Fig. 2.3.5.18). This provides the user with the spatial and temporal information of an identified individual without the need for recapture. The advantage of RT over traditional recapture methods is that data can be collected more rapidly and accurately and with less experimental bias (White and Garrott 1990). These advantages are greatest for elusive species and this method has been effectively applied to study the spatial distribution of fish.

Figure 2.3.5.18: Researcher with radio-antenna receiver.

How is radio telemetry used to monitor the spatial distribution of fish?

RT has been used to monitor fish populations in lake and river systems with both potamodromous and diadromous species (Erickson et al. 2002; Koehn 2009; Serrano et al. 2009). However in contrast to UT, RT only functions effectively in freshwater and thus cannot be used for coastal populations or studies of migrations between fresh and saltwater. On the other hand an advantage of RT over UT is that, generally speaking, it can be used to provide finer scale spatial and temporal data, as manual tracking (opposed to automated logging) is more easily conducted (with use of a radio antenna and not with an underwater hydrophone).

RT has been applied to studies of home range (Knight et al. 2009), movement patterns (Hilderbrand and Kershner 2000), migration (Lucas 2000; Behrmann-Godel and Eckmann 2003), habitat use (Aebischer et al. 1993; Allouche et al. 1999) and diel activity (Crook et al. 2001). Due to the relatively high tag cost this method is not commonly used for marking at the population level, potentially reducing the power of observations made (Zeller 1999) but inferences gained from marking a sample can be used to guide population scale monitoring with other methods or directly for practical application.

The information on spatial and temporal fish movements gained with RT can be used in assessment of pressures on populations, design and valuation of mitigation measures and in directing population monitoring. Pressures on populations can include physical barriers e.g. weirs, water intakes or hydropower turbines that physically affect the spatial distribution of fish (Caudill et al. 2007), ecological challenges e.g. behaviour and ecology of endangered species (Cavalli et al. 2009), or the impact of non-native species introduction that provides competition affecting population spatial structure by predation or resource competition (Jackson and Zydlewski 2009). RT is often used in collecting information used for design of management plans and protected areas (Tiffan et al. 2010) and mitigation measures such as assessment of the suitability and success of fish passes or habitat rehabilitation work (Parsley et al. 2007; Evans et al. 2008). Another application of RT is in recommending how best to focus population monitoring or identifying key or vulnerable life stages of a species.

RT tags are also on the market that are capable of making both environmental and internal physiological measurements. Tags with capabilities to measure water temperature and depth (pressure) are now commonly available and used in a wide variety of studies (Koehn 2009; Tiffan et al. 2009). Electromyogram (EMG) tags are also frequently used for energetic cost assessments, a widespread use being fishway passage evaluation (Pon et al. 2009). Motion sensor radio tags which, when calibrated, can assess the physical activity of a fish are used for a broad range of applications from spawning activity and behaviour (Karppinen and Erkinaro 2009), to river reconstruction evaluation (Makiguchi et al. 2008). The use of such radio biotelemetry methods provides increasingly greater and more detailed insight into the basis and consequences of fish spatial distribution.

Advantages of the technique

Radio telemetry provides reliable and fine-scale spatial and temporal information on the spatial distribution of fish. As discussed, more and more information is becoming available. New tagging technology is now providing environmental and physiological measurements to help understand the cause and effect of spatial distribution of fish and increasingly smaller tags are available to enable marking of smaller species and life stages. RT is possibly the best telemetry method for manual tracking, thus providing the most detailed spatial and temporal information (as well as linking to habitat recording carried out when fish are located, for example), though 3D systems of acoustic telemetry are now providing very detailed information for studies in limited areas. A recent development of VPS and VRAP 3D systems by Vemco (Amrix Systems, Halifax, Canada) is now enabling near-continuous monitoring of 3D positions of fish in restricted freshwater and coastal systems (e.g. small lakes, small reefs or marine protected areas).

Issues for consideration with radio telemetry

RT is limited to freshwater systems. For that reason UT is more suited to studies where migratory fish are to be followed during both marine and freshwater phases. The cost of tags generally prohibits tagging of very large numbers of fish and thus monitoring of a whole population, something that is possible with PIT telemetry. Battery size constrains tag life and is limited by the size of the fish and by the weight of the tag that can be ethically carried. This creates rather short study durations for the smallest tags (around three weeks). As PIT tags do not contain batteries their size is much reduced (currently 12 mm) and they can be inserted in much smaller individuals.

A combination of PIT and radio or acoustic telemetry thus allows ‘cradle to grave’ marking of the fish, first with a PIT tag and later with the addition of an active telemetry tag (for a recent example see Ovidio et al., 2009). This provides much potential for monitoring of the spatial distribution of fish populations because, through the combination of two or more techniques, the full population and all life stages (except the smallest juveniles) can be tagged with fine-scale information available for a sample of that population.

2.3.5.4.2.4 Notes on other telemetry systems

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Pop-up satellite tags, or PSATs are widely used to track the movements of large pelagic animals. PSATs have been successfully applied in such animals as tunas (Gunn et al. 2001; Bestley et al. 2008), sharks (Pade et al. 2009; Weng et al. 2007), and marine turtles (Pelletier et al. 2003) that live close to the ocean surface and that typically travel hundreds or thousands of miles. As the pop-up tag is mounted on the animal (Fig. 2.3.5.19) it can store data such as temperature, depth and light. At a predetermined time, the tag initiates an onboard release mechanism and pops off the animal. The tag is buoyant and rises to the surface where it relays its stored data to the ARGOS satellites by means of UHF signal (170-300MHz). Data can then be downloaded via the ARGOS system. Unfortunately, the size of most PSATs limits the number of species on which they can be applied. Size reductions have very recently permitted smaller species such as eel to be monitored, providing substantial advances in the understanding of their ecology (Aarestrup et al. 2009).

Figure 2.3.5.19: PSAT installed in the dorsal region of a sturgeon.

PSATs have been used together with micro-data loggers, better known as archival or data storage tags (DSTs). These tags are suitable for long-term studies (up to several years) and enable collection of physiological and environmental data such as: water temperature, salinity, pressure (depth), fish position (GPS), movement direction (compass) or tilt (in 3D). Data are stored in the internal flash memory of tag and can be retrieved only after recapturing the tagged animal. Due to the reduced size of transmitters, DSTs have been applied from tunas (Bestley et al. 2008) to salmons (Johansson et al. 2009).

2.3.5.5 References

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2.3.6 MONITORING POPULATIONS OF INVERTEBRATES

Coordinator: Stelios Katsanevakis (HCMR)

2.3.6.1 Monitoring macro-zoobenthos (soft substrata)

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Introduction

Numerous recommendations for the collection and treatment for monitoring of macro-zoobenthos of soft substrata exist. Since there are recommendations on different areal levels, such as national (e.g. JNCC 2004), regional (e.g. Todorova and Konsulova 2005) and international (e.g. Rumohr 2009), efforts have been made for example by ICES to standardize the methods for monitoring macro-zoobenthos of soft substrata (ICES 2004; Rumohr 2009). Especially the focus on quality assurance has intensified over the past years. The current basis guideline for quality assurance in the collection and treatment of macro-zoobenthos samples was published by ISO 16665: 2005 (ISO 2005). Internationally Rumohr (2009) refers to that guideline as do Todorova and Konsulova (2005) on a regional level for the Black Sea Integrated Monitoring Programme.

While the ISO 16665: 2005 (ISO 2005) guidelines aim at standardising monitoring surveys carried out for commercial purposes (fisheries, aquaculture, oil) or in connection with the EU Water Framework Directive, the guidelines of Rumohr (2009) aim at assisting individual scientists.

Definitions

- *Soft substrata* are defined as sediments ranging from mud to sand. According to ISO 16665: 2005 (ISO 2005) 'soft-bottom areas of sea floor consist of loose deposited particles including clay, mud, sand and gravel, shells and maerl, also including mixed substrata with gravels, small stones and pebbles scattered on a bed of finer material, but excluding cobbles'.
- *Macro-zoobenthos* are bottom-dwelling animals retained on a mesh screen of 1 mm or 0.5 mm size (ISO 2005).

The following information on methods for monitoring macro-zoobenthos is based on reviews by Eleftheriou and McIntyre (2005), Kröncke and Bergfeld (2003) and Rumohr (2009) and amended with current information.
The main content of this review is a summary of standard sampling and analytical techniques for benthic parameters including a short description and assessment of the techniques in terms of accuracy and applicability.

Sampling strategy

The design of the sampling programme largely depends on the specific aim of a study. The temporal and spatial scales have to be considered as well as the local abiotic factors. Major constraints arise from limited resources like time, money, manpower and laboratory facilities.

Detailson the various options for sampling strategy and the implications on subsequent statistical analysis are given e.g. in Cochran (1977), Elliott (1977), Green (1979), Rees et al. (1991), Gray et al. (1991), Warwick and Clarke (1993), Underwood (1996), van der Meer (1997), Rees (1999), Armonies (2000), Underwood and Chapman (2005). A combined stratified random sampling approach has turned out to be a very effective sampling strategy especially for large-scale, long-term monitoring as conducted on the Dutch continental shelf (Essink 1995; Holtmann et al. 1996; van der Meer 1997).
Some basic sampling strategies used in macro-zoobenthos investigations include:

- time-series sampling (equidistant, at biologically relevant time intervals, BACI design (e.g. Stewart-Oaten et al. 1986; Underwood 1992)

- stratified sampling (according to strata, depth, sediment type, etc.)
- area sampling (grid sampling, randomized sampling)
- single-spot (station) sampling
- transect sampling (usually along a biological, physical or chemical – contaminant or nutrient – gradient)
- nested sampling (e.g. reviewed in Ellis and Schneider 2008)
- Related design details include:
- number of samples
- sample size (surface area)
- position(s) of all samples relative to each other and/or biotic/abiotic factors (current, pollution source etc.)
- precision of results

Replicate sampling

In general taking many small samples is considered more informative and reliable than taking a few large samples. This is due to the following main advantages:

- broader coverage of the sample site
- better estimate of spatial dispersion of species
- higher number of degrees of freedom for statistical analyses

For statistical reasons, 2-3 replicates (grabs) are considered the minimum replication per sample while in benthic studies up to 5 replicates are common. Accurate replicate sampling is especially important in order to detect ecologically important changes (Rogers et al. 2008).

The sufficiency of a sampling effort can be estimated by empirical species- area- curves. These curves estimate the number of replicates (minimal sampling area) required to obtain an acceptable percentage of the total number of species (e.g. Gray and Pearson 1982; Beukema 1988; Hartung 1993). A more sophisticated way to test the sampling efficiency is the use of statistical distributions like the normal, binomial or Poisson distribution (e.g. Elliott 1971; Mühlenberg 1989; Hartung 1993; Pfeifer et al. 1995). These techniques imply an *a priori* accepted level of precision (commonly 0.2 or 0.4).

Clarke and Green (1988) pointed out that pilot studies often are essential and always desirable in order to establish an appropriate sampling design. Assessing the accuracy and precision of biological effect studies they also introduced a cost function, accounting for limited financial resources.

Accounting to various scales of patchiness in benthic distribution, sampling should be nested hierarchically (Underwood 1981; Underwood 1991; Morrisey et al. 1992; Ellis and Schneider 2008) to increase the reliabilility of the samples. The spatial scale of patchiness in the measured variables is often unknown prior to sampling. Consequently, a spatial scale of patchiness, be it sampling units (small scale) or sampling locations (large scale) won't necessarily be taken into account by the sampling design nor will it necessarily be revealed analyzing the samples later on. If the within-location variation has not been adequately estimated by sample replication, this prevents valid comparisons among locations because the data are pseudoreplicates (Hurlbert 1984).

Otherwise large scale comparisons of sites can be confounded by the variance on a small scale. This also holds true for temporal variations. Long-term effects are difficult todisentangle from short term variations (e.g. seasonal), and might be visible only on the long run, even if sampling was undertaken on the appropriate long-term scale as revealed by the Northumberland long-term series (e.g. Buchanan and Moore 1986; Buchanan 1993; Frid et al. 1996). The bigger the scale difference between sampling levels (e.g. replicate grab samples - locations), the bigger the magnitude of patchiness that could be overlooked.

Gear and methods

A wide variety of sampling gears and methods has been developed for macro-zoobenthos sampling in soft substrata (Eleftheriou and McIntyre 2005). The choice of a suitable sampler depends on the scientific objective (e.g. deep burrowing target species), specific requirements in the study area (e.g. different sediment regimes), good handling characteristics at sea in bad weather conditions, suitability for various ships, financial limitations and sampling tradition. In addition, the time required for processing the samples and the required level of sampling precision influences the choice of sampling gear (Jensen 1981; Kingston 1988).

A summary of standard sampling- and analytical techniques for benthic parameters including a short description and assessment of the techniques in terms of accuracy, precision and applicability is given in Table 2.3.6.1.

Table 2.3.6.1: Summary of standard sampling gear for macro-zoobenthos in soft-substrata. For more gears and details see Eleftheriou and McIntyre (2005) and Rumohr (2009).

	Gear	accuracy	source of errors	comments	Information source
Grabs	van-Veen grab, modified	quantitative, accuracy depends on penetration depth, sample volume, number of samples / space; experienced personal	varying penetration depth for different sediment types, (can be balanced by adding weight) ; upper sediment layer might be flushed away	routine, easy to handle, even from small ships; in different sizes, covering areas from 0.025 – 1.0 m²; unsuitable for sediments coarser medium sands	Eleftheriou and McIntyre (2005), Rumohr (2009)

	Petersen grab	see above	see above	scientific, 0.2 m^2, only little penetration depth in fine sands, inferior to Van Veen grab	Ziegelmeier (1978), Ursin (1954)
	Campbell grab	similar to Petersen			Eleftheriou and McIntyre (2005)
	Okean grab	similar to Petersen			Eleftheriou and McIntyre (2005)
	Smith-McIntyre grab	see above	see above	routine, 0,1 m^2	Eleftheriou and McIntyre (2005)
	Day grab	similar to Smith-McIntyre		greater simplicity than Smith-McIntyre	Eleftheriou and McIntyre (2005)
	Hamon grab	quantitative		gives representative results when sampling a relatively constant surface	Kenny and Rees (1994)

Corers	Reineck box corer	see above, species numbers estimates about 15% better compared to van Veen grab (van der Meer and Seip 1986)	varying penetration depth for different sediment types, (can be balanced by adding weight)	routine; undisturbed sediment cores, longer handling time compared to grabs, unsuitable under rough weather conditions (more than 5 Beaufort), additional observation and measure devices can be added to the frame, cores used for boxcosm-experiments, benthic chamber measurements; mobile epifauna not sampled efficiently	Eleftheriou and McIntyre (2005)

	Reineck box corer, NIOZ-type	see above	see above	routine, round box, see above	Rumohr (2009)
	GOMEX box corer			comparatively easy to use, safe in rough seas, and modest in cost compared to the cumbersome but widely accepted USNEL spade corer.	Boland and Rowe (1991)
	HAPS box corer (Danish Institutes)	see above	see above	routine, round box, see above	Kanneworf and Nicolaisen (1973)
	UNSEL spade or box corer	see above	see above	routine, large sample volume up to 0,25 m², removable spade reduces handling time; see above	Rumohr pers. comm., Hessler and Jumars (1974)
	MUC Multicorer	see above	see above	scientific, (up to 12) undisturbed sediment cores (10 cm diameter) with overlying bottom water	Barnett et al. (1984)
	Craib corer	quantitative	no vertical penetration	scientific, single core of 5-7 cm in diameter, hydraulic bumper and automatic spherical closing device ensure minimum disturbance of superficial sediment layer operated from small ships, unsuitable for bad weather conditions and uneven bottoms	Rumohr pers. comm., Heip et al. (1985), Eleftheriou and McIntyre (2005)
Dredges	Triple-D dredge	quantitative (in combination with a velometer)	varying penetration depth for different sediment types	scientific, sediment top layer cut off by a blade, closure device, designed for the quantitative collection of the large and rare epifauna and infauna	Bergman and van Santbrink (1994)

Benthic lander	benthic chamber (profiling) lander	quantitative		*in situ* (remote) and *in vitro* (ship, mesocosm, boxcosm)	Tengberg et al. (1995)

Grab sampling

Types of grabs and corers

Petersen Grab and descendants

The Petersen Grab (Petersen 1913) is basically the prototype of all bucket grabs from which many modifications and improvements have been made. Nowadays most widely used are the van Veen Grab (van Veen 1933, with the modifications described by Dybern et al. 1976), which possesses long arms and endless warps. Other modifications are found in the Smith-McIntyre grab (Smith and McIntyre 1954): an added frame, springs to drive the jaws into the sediment and trigger plates at opposite corners activating the grab at the instant it is sitting squarely on the bottom. The Day grab (Eleftheriou and McIntyre 2005) is a simpler version of the latter, without springs. These grabs usually cover a surface area of 0,1m² or 0,2m² (although smaller and larger versions are available) and have weight ranges between 25 kg and 200 kg.

Box corers

Box corers have been successfully employed in a variety of design in benthos research, most of them based on the Reineck box corer (Reineck 1963). In principle, a corer supported by a metal frame penetrates the bottom (assisted by added weights) and a hinged cutting arm swings down when the warp is hauled, closing the bottom of the core. The 'spade corer' by Hessler and Jumars (1974) has been modified by a removable spade from the lever arm, reducing handling time and disturbances during further processing. Also round box corers are in use like the Netherlands Institute for Sea Research (NIOZ) type with a flat spade, or the HAPS designed by Danish institutes (Kanneworf and Nicolaisen 1973).

Box corers are large, heavy gears weighing generally around 1000 kg or more. Sample sizes of 0.25 m² are reached, but are generally around 0.1 m² or smaller. 0.25 m^2USNEL corers are mainly used at greater depths.

Also multicorer-systems (MUCs), e.g. the SMBA corer (an elaborated Craib corer) equipped with up to 12 cores are frequently used (Barnett et al. 1984; Holme and McIntyre 1984), saving much time by the simultaneous replicate sampling. Because of a rather narrow tube diameter (5 to 10 cm) these corers are primarily suitable for meiobenthic surveys.

One great advantage of box corers is that the sample is almost undisturbed in its structure and orientation, reflecting *in situ* conditions very well. After sampling, the box can usually be removed with the sample and its overlying water intact, allowing detailed studies of the sediment surface and its vertical zonation. Furthermore such core samples can be used for microcosm experiments, e.g. with integrated biogeochemical measurements.

Benthic chamber lander

Benthic chamber lander systems (Tengberg et al. 1995) are virtually a further development of the box corer technique, using a closed system for the in situ measurement of the benthic metabolism, eg. nutrient fluxes, respiration rates (Hall et al. 1996).

Efficiency of grabs and corers

Efficiency of grabs and corers is largely correlated with the penetration depth of the gear, weight of the gear, bite profile, sediment type and the subsequently sampled volume of sediment (Ursin 1954; Birkett 1958; Christie 1975; Ankar 1977; van der Meer and Seip 1986; Kingston 1988; Riddle 1989).

Additionally, weather conditions, hydrodynamics, size of the vessel and the experience of the operator affect the quality of the samples and grab failures (grab bouncing, drift, pressure waves).

Kingston (1988) recommended a penetration depth of at least 5 cm by which in most sediments about 90% of benthic macrofauna in terms of number of species and abundances is sampled (Ankar 1977). Large deep burrowing species / older individuals of some bivalves (*Chamelea gallina*, *Ensis* spp. and tellinids) and tube worms (*Nephtys hombergii*, *Lanice conchilega*) found down to 20 cm depth will be underestimated (Birkett 1958; Beukema 1974; Salzwedel 1979).

Aware of this restriction Gray et al. (1991) recommended a penetration depth of at least 10 cm, that marks almost the maximum penetration depth of grabs whereas box corers can provide samples far deeper (15-30cm) (Beukema 1974; van der Meer and Seip 1986; Kingston 1988; Riddle 1989).

Van der Meer and Seip (1986) calculated a relative underestimation of 15% in total species numbers by a van Veen grab compared to a Reineck box corer. No statistical differences were revealed for the two gears on silty sediments by the Texel Intercalibration workshop (Heip et al. 1985).

For increasing penetration depth the edges of buckets of framed grab samplers (Smith and McIntyre and Day grab) should extent below the frames and additional weight should be added to the van Veen Grab, e.g. to the upper edges of the jaws (Ankar 1977; Kingston 1988; Rumohr 1998).

Blomquist (1991) addressed the disturbance or even loss of the sediment surface by sampler induced pressure wave as a major problem in quantitative sampling.

In conclusion the boxcorer technique is generally recommended for sampling North Sea benthos, giving most reliable results by high penetration depth and least disturbed samples for a variety of integrated analysis and experimental approaches. Major disadvantage is the applicability of this expensive and heavy gear restricted to relatively calm weather and large vessels (see also Blomquist 1991; Rumohr 1998).
Therefore, for monitoring surveys on macro-zoobenthos the van Veen grab seems to be an adequate alternative choice, easy to handle and without the technical limitations mentioned for box corers. By further improvements in design and use (Rumohr 1998) the van Veen grab might become an even more reliable sampling gear. For the Baltic Sea by an international agreement the van Veen grab is the standardised gear for infaunal monitoring studies.

On coarse sediments (gravel, coarse sands) where sampling by most grabs is insufficient, the Hamon grab turned out to be very effective, giving representative quantitative results by sampling a relatively constant surface (Kenny and Rees 1994). In comparison the anchor dredge commonly used on such sediments gives just semi-quantitative results, catching to a higher proportion epifaunal species.

As discussed by Jensen (1983) who revealed underestimated abundances of copepods and nematodes by 30 % to 50% in the top cm for the HAPS corer, data might not be reliable for all taxonomic groups. Nevertheless, the subsample method from box corers might still be advantageous for direct comparisons of macro- and meiofaunal distributions. More precise and accurate samples are provided by the more sophisticated Craib corer (Craib 1965) consisting of a frame mounted tube, which is driven into the sediment by weights, controlled by a hydraulic damper ensuring minimum disturbance of the light deposit surface. Samples of 15 cm length are obtained and a closing device prevents losses from the sample. By the later elaborated SMBA multicorer version (Barnett et al. 1984; Holme and McIntyre 1984) replicate samples can easily be obtained.

Diver-operated samplers
Scuba diving is a very useful method for sampling soft substrata in shallow water (Rumohr 2009). Scuba sampling can be done with tubes (Jensen 1983). Diver-operated boxcorers (Rumohr and Arntz 1982) or suction samplers (Hiscock and Hoare 1973) can also be used on mud to sand substrata. Further references to sampling by scuba diving are found in Eleftheriou and McIntyre (2005).

Other methods
Remote sensing methods by satellite but also shipbased scanning sonar systems in combination with substrata sampling have a very limited applicability (Rumohr 1995).

Sample treatment and analysis

Sieving and sorting

Benthos samples are sieved and sorted to separate the fauna from sediment. As an international standard the use of a 1mm sieve for macrofauna is recommended by the ICES Benthos Ecology Working Group (Rumohr 2009) for routine surveys. For many studies an additional 0.5 mm sieve is advisable accounting for a variety of small individuals / species (mainly polychaetes and crustaceans).

Generally during the sieving process the whole sample is suspended in sea water over sieves of desired mesh width. The danger of damaging animals during the sieving process is high, and damaged specimens complicate the subsequent sorting, identification and biomass determination. Therefore, highmost care must be taken during the rinsing process. Although most work has to be done by hand, special washing devices with sprinklers and sieve cascades, suitable for large sample volumes, partly automate and standardise the sieving and sorting process.

Sometimes it is practical to subsample large sample volumes for pragmatic reasons but in that case rare species might not be sampled representatively.

Generally the sampling design, sampling techniques, treatment and analysis of samples in long-term monitoring studies must be carried out as constant as possible over time. If any aspect is changed for any reason, intercalibration studies should be carried out alongside old- fashioned procedures to ensure the replicability, reliability and comparability of the produced data.

Where it is impossible to sieve material prior to fixation, sievingthe already fixed material is a solution. However, the sorting characteristics of fixed material are different from those of live fauna and apparently result in higher abundance and biomass figures. An intercalibration of both procedures indicated that sieving procedure affects macro-zoobenthos studies at the level of diversity, density, as well as community structure (Degraer et al. 2007). This effect is particularly important when dealing with area studies, such as nearshore environments dominated by small, interstitial and/or larger, slender polychaetes. Mainly polychaetes are negatively impacted by alive-sieving, with relative losses up to 80%. Especially small, interstitial polychaetes (e.g. *Hesionura elongata* and *Spio filicornis*) tend to actively escape from the sieve which results in a relative loss up to 100%. Next to animal size, behaviour, the presence of head appendages, the depth of the sampling stations and sampling season are believed to influence the susceptibility to the sieving procedure. Combining data obtained after different sieving procedures can be useful, but its reliability mainly depends on the type of questions that need to be answered. Therefore, in publications and in databases it should always be stated whether the sieved material was alive or fixed.

Fixation and conservation

Standard primary fixation agent is a 4% seawater-formalin solution, which should be buffered by a hexamine (eg. Borax), in order to prevent acidisation and the dissolving of calcareous structures. This 4% seawater-formalin solution (including a hexamine) can also be used for conservation, but 70-80% alcohol might be preferred for being less hazardous.

In most animals fixation entails body changes e.g. by contraction, rejection of appendages, bleaching. Therefore, identification should ideally be done using living animals and (if necessary) anaesthetic substances, like MgCl (4%).

Fixation agents like Formalin, Kohrsolin and alcohol are known to (significantly) reduce the body weight by dilution of tissue (e.g. Hamilton and Kingston 1985; Brey 1986). Notable losses occur in the first 4-6 weeks, after which the weight remains constant (Brey 1986) allowing reliable and comparable biomass data.

If genetic analyses are required, macro-zoobenthos should be fixed in ethanol. Another fixation agent for preserving morphology or and extractable DNA is DESS, a versatile solution (Yoder et al. 2006).

Biomass determination

Biomass determination can be done using wet weight, dry weight, shell-free dry weight or ash free dry weight, from either fresh or fixed material. For additional information, energy content and equivalents of carbon, nitrogen or phosphorus may be determined, using calorimeter techniques or elementary analysers (e.g. HPLC).
Most reliable is the determination of ash free dry weight (AFDW) (see Rumohr 2009). It gives a reliable estimate of the amount of organic matter available as food for higher trophic levels and can be used for productivity estimates. Fresh-wet weight gives a better estimate ofwet weight as compared to formalin-wet weight, but if the latter has to be used, weighing should be done no earlier than three months after fixation (Brey 1986). This is because preservation could produce weight changes, especially in annelids and bivalves (Ricciardi and Bourget 1998). The main advantage of formalin-wet weigh is that organisms are not destroyed. Consequently, they can be preserved for future verifications and quality assurance.

Shell-free dry weight is estimated after shells and exoskeletons of molluscs and echinoderms are removed. This can be done manually, or the tissues can be dissolved using chemicals (e.g. HCl, NaClO, etc.).

Using parallel determination of wet and dry weight allows an easy comparison of the results to other biomass data. Conversion factors can be used to estimate biomass from weight, length or size measurements (Rumohr et al. 1987; Brey et al. 1988; Ricciardi and Bourget 1998). However, Ricciardi and Bourget (1998) conclude that 'it is preferable for researchers to produce their own conversion factors from subsamples'.

Taxonomy

Species identification work needs highly skilled and experienced staff, because accurate identification is essential for the outcome of further statistical analysis and for the interpretation of the species abundance data. Identification keys exist for almost all benthic taxonomic groups occurring in the North Sea (for review and publication lists see Heip and Niermann 1988; Howson and Picton 1997). A detailed North Sea specific species inventory does not exist, but it is referred to the comprehensive work for British waters, covering also most of the North Sea species (Howson and Picton 1997).

Experiences from the NSBS revealed how problematic it is to ensure high taxonomic quality standards on an international level (Künitzer et al. 1992). Taxonomy changes continuously, new species are described, species are renamed, and synonymes are used. In standard identification guides these changes are often not updated for decades. Therefore voucher specimens should be kept for later re-examination.

Quality assurance by continuous training of staff and quality control by taxonomic workshops and intercalibration exercises should be carried out in a regular scheme.
Often new insight in taxonomy takes too long to be considered in everyday identification work, which relies mainly on a few (sometimes fairly old) identification guides. Even when taxonomic publications are considered, the publication process itself can take very long and distribution of information by these cannels might be fairly limited. Databases in the world wide web are a more current and up-to-date basis for taxonomic information exchange (European Register of Marine Species -ERMS; http://www.marbef.org/data/index.php- or World Register of Marine Species -WoRMS; http://www.marinespecies.org/index.php).
Since classical morphology-based techniques of taxonomic analyses are sometimes inefficient in many groups of organisms, the application of molecular biological methods in taxonomy and ecology provides promising tools to gain an insight into the structure of species communities in natural environments. Molecular methods are able to generate data independently of the morphological organisation of an organism. A variety of molecular biological techniques has been developed to distinguish between species or between different genotypes (populations) within a species.

Conversely, some authors argue that higher taxonomic level than 'species' level could be use at least for pollution assessment. In the current literature, 'family' is generally accepted as sufficient taxonomic level to investigate pollution gradients. However, when describing habitats it is recommended to identify the organisms to the lowest taxonomic level possible.

Data analysis

All kinds of univariate and multivariate techniques are applied to analyse benthic data sets from field surveys, lab experiments or even taxonomic studies. Analyse procedures are extensively described in Clarke and Warwick (1994). Here we only refer to the most widely applied techniques and some new improvements, especially in the context of (bio)-diversity measures.

Univariate techniques

Traditional univariate statistical data analysis of abundances, species numbers and biomass include typically the calculated mean and standard deviations, minimal and maximal values and are well suitable for population studies, but have some constraints applied on community / multi species level. Values are either related to the sampled area, or extrapolated to a square meter.

Multivariate techniques

Over the last decades multivariate data analyses became the standard techniques for analysing community pattern (Field et al. 1982). They have the great advantage of using the information of the whole species composition. The basic idea of cluster analysis and ordination techniques is to compare different sites/sampling times by calculating similarities/dissimilarities measures.

Cluster analysis reveals pattern by hierarchical sorting illustrated in dichotom dendrograms giving the similarity levels or grouped samples on an axis.
Dendrograms have four main disadvantages (Field et al. 1982):

- The hierarchy is irreversible - once a sample has been placed in a group its identity is lost
- Dendrograms show only intergroup relationships. The level of similarity is the average inter-group value
- The sequence of samples in a dendrogram is arbitrary and 2 adjacent samples are not necessarily the most similar. The dendrogram has to be thought of as a suspended mobile, the samples holding thread free to rotate in a horizontal plane
- Dendrograms tend to over emphasize discontinuities

The reliability of dendrogramms can only be assessed appropriately in comparison with corresponding ordination techniques (Warwick and Clarke 1991; Warwick and Clarke 1994).

Ordination techniques like multidimensional scaling (MDS), principal component analysis (PCA),canonical correlation analysis (CCA) or two-way indicator species analysis (TWINSPAN) are exploratory methods to analyse the community pattern, visualising trends and changes in community structures. Correlation analyses are used to detect potential relationship commonly between community and environmental data, but also for interspecific relationships. Correlations are visualised by vectors along which the data are grouped. The directions of the vectors indicate whether the factors might control the community in the same or opposite way.

Using MDS plots the BIOENV program of the PRIMER software package superimposes community data by other data sets and correlation analysis reveals the importance of single factors or factors in combination with the community structure (Clarke and Ainsworth 1993).

The non-metric MDS is a much more flexible tool than the metric PCA technique.
For the MDS, analysis of the data is done on a dis/similarities matrix expressed in distances between the samples in an n-dimensional dimension. Graphically, it is translated in Euclidean distances plotted in a 2 (or 3) dimensional diagram. The closer the samples the higher is their similarity. A stress measure indicates how representatively the diagram reflects the multidimensional data.

The significance of the revealed pattern can be tested statistically by accompanying randomisation testing procedures, based on the dissimilarities of all pairs of subsamples. E.g. the ANOSIM program by Clarke (1993) uses the differences of the rank similarities between and within the samples. The randomisation has the advantage that it does not rely on a distribution model (see above, 'species curves') and does not require random sampling. Similar are the L-test (Smith et al. 1990) and the MST-test; the latter of which is accompanied by the 'minimum spanning tree' ordination technique (Schleier and Bernem 1996). All these techniques also provide a tool for assessing the importance of single species in terms of being a ‘characteristic species in a data set or not. This can be a crucial factor in assessing the similarity of different data sets.

Because of the complex structure of multivariate data, a general tool applicable for all types of investigations and data is not available. It is always advisable to experiment with several methods to throw light on the data from different viewpoints.

Diversity indices

Furthermore, communities can be described by calculating diversity indices, of which a variety of species numbers and their relative contribution to total abundances by various mathematical formulas is related. Hills diversity numbers (Na) of various orders (Hill 1973) are mathematically related to a variety of commonly used indices. The Shannon-Wiener Index (Shannon and Weaver 1963, also refered to as the Shannon index or the Shannon-Weaver index; Krebs 1989) is mostly used in connection with the Evenness Index by Pielou (1966), a measure of the distribution of individuals among the different species. The Shannon-Wiener index can be also estimated from biomass data (Wilhm 1968). Soetaert and Heip (1990) tested the sample size dependency of indices, stating highest precision for the Shannon-Wiener index. Although diversity indices are very useful for comparisons between sites and in this sense widely used in impact studies, their use for comparisons between studies is hampered by their sample size dependency, which requires a standardisation of sample size (e.g. by randomisation) (Soetaert and Heip 1990). The sample size independent ES value, a calculation of the number of expected species (Hurlbert 1971), has the drawback of accounting only for species numbers not for abundances.

Information on diversity can also be gained graphically by the k-dominance curves (Lambshead et al. 1983), in which the species are ranked in order of dominance (of abundance or biomass) on the x-axis (log scale) with percentage dominance on the y-axis (cumulative scale). The steeper the curves the higher the diversity. Further elaboration led to the Abundance - Biomass Comparison (ABC)-method applied to detect (pollution-induced) disturbance effects in communities (Warwick 1986; Warwick et al. 1987; Gray et al. 1988; Austen et al. 1991; Kröncke et al. 1992; Kröncke and Rachor 1992). The ABC- method describes the degree and direction of separation of the k-dominance curves by a single statistics (Beukema 1988; Clarke 1990) conceptually based on the Pearson and Rosenberg model. Using 'partial dominance curves' (Clarke 1990) the problem of the overweighing of the first ranks (Beukema 1988; Ibanez and Dauvin 1988) might be overcome, but caution in interpretation of ABC-graphs is advisable.

A new approach in measuring diversity is the taxonomic distinctness measure which is a univariate (bio-) diversity index (Warwick and Clarke 1995; Clarke and Warwick 1999). It captures the community structure not only of the distribution of abundances amongst species but also assumes the taxonomic relationship of the species. In its simplest form using just presence/absence data, the taxonomic distinctness measure calculates the average distance between all pairs of species in a community sample, where the distance is defined as the path length through a standard Linnean or phylogenetic tree connecting these species. This index is robust to variation in sampling effort, making it a good tool for comparing different studies. The result can be tested statistically and it seems to be more closely linked to functional diversity, although no higher sensitivity to environmental changes might be derived. Taxonomic artefacts might be caused by the somewhat inconsistency of the Linnean classification in the way it defines taxonomic units across different phyla. A better metric of species relatedness might be gained from genetic distance, or from a phylogeny combining molecular tools with traditional morphology.

Nevertheless, such diversity indices might overcome the problem that comparable data on species diversity require a highly skilled and complex analysis of species and an unusually high degree of standardisation with respect to the degree of taxonomic rigour applied to the sample analysis.

Diversity measures are much less sensitive than multivariate methods in detecting community changes (Gray et al. 1990; Warwick and Clarke 1991; Warwick and Clarke 1993).

Websites for quality assurance
National
Germany: http://www.umweltbundesamt.de/wasser/themen/blmp/index.htm
UK - National Marine Biological Analytical Quality Control Scheme: http://www.nmbaqcs.org/

Regional
Within the International Council for the Exploration of the Sea (ICES) are two relevant Steering Groups on Quality Assurance of Biological Measurements in the Northeast Atlantic and Baltic Sea, respectively http://www.ices.dk/iceswork/workinggroups.asp

2.3.6.2 Monitoring epibenthos

Contributors: Sandra Vöge (SGN), Henning Reiss (SGN), Ingeborg de Boois (IMARES)

Introduction

Epibenthos are animals living on or immediately above the seafloor. Some species are attached to the substrate, other are mobile. The size range of epibenthos is usually greater than the size of endobenthos (and also of hyperbenthos). On these grounds the term megabenthos (or megafauna) is often used instead of epibenthos in the literature. It has to be mentioned here that the definition of epibenthos, but also of endo- and hyperbenthos, are not clear-cut and several taxonomical groups belong to more than one category depending on their behavior, sex, size, life stage or simply on the gears used to sample the species. For example, many species especially crustaceans such as amphipods or cumaceans, remain in the sediment during daylight but migrate into the water column during night, and thus, changing from an endobenthic to a hyperbenthic or even pelagic life mode.

However, in this chapter we describe the most common strategies to sample epibenthic organisms on soft-substrate. For more comprehensive reviews see Coggan et al (2007) and Rees et al. (2009).

Sampling

Sampling epibenthos might be done by using gears as described in Chapter 2.3.6.1, but especially for the larger sized epibenthic species these gears are not very efficient. Thus, towed gears are preferably used for the monitoring of epibenthos, sampling larger areas than by using grabs or corers. An extensive review of sampling techniques is done by Rees et al. (2009), providing guidelines for epibenthos monitoring in subtidal environments.

Other methods for monitoring epibenthos next to towed gears are diver operated sampling, camera sledges, ROVs and acoustic methods (see Sections 2.1, 2.3.6.4, and 2.3.6.5). The following description will focus on towed gears since they are most widely used for monitoring epibenthos in EU waters.

One major problem with all towed gears for sampling epibenthos is the quantification of the sampled fauna (abundance or biomass per sampled area). Generally, epibenthic sampling is found to be semi-quantitatively, which means that the numbers of individuals or biomass values cannot precisely be related to the sampled area. Therefore, the following sub-chapter will address this general problem before discussing the individual sampling gears.

Catch efficiency of towed gears

There are two aspects related to the problem of quantifying the epibenthic fauna with towed gears; firstly the difficulties in precisely recording the sampled (towed) area, and secondly, the often unknown catch efficiency of the gear. The term 'catch efficiency' is generally defined as the number of individuals or the biomass of epibenthic species, expressed as the proportion of the total population in a sampled area caught with the gear (Allenet al. 1992; Kaiseret al. 1994). Realistic estimates of catch efficiency and, thus, of abundance or biomass of fish and invertebrate epibenthos in an ecosystem are particularly important to determine e.g. secondary production or consumption rates (Harleyet al*.* 2001). Therefore, the limitation of a sampling gear has to be considered in assessing how well an assemblage is described. But in most cases the catch efficiency of a towed gear is not known, since these gears often represent the sole method to monitor the standing stock of epibenthic organisms.

Estimations of the catch efficiency of towed gear was done by combined underwater video or photographic analysis and trawl catches (Yeh and Ohta 2002, Sonntag pers. comm.), by experimental approaches such as repeated sampling (Fonds 1994; Loneraganet al*.* 1995) or by removal experiments (Reisset al*.* 2006). These studies have shown that catch efficiency differs remarkably between gears, species and sediment type of the study sites. For example, for the standardized 2 m beam trawl, a widely used trawl for epibenthos monitoring in the North Sea, catching efficiency varied between 9 % and 72 % depending on the species and was below 50 % for most of the species (Reisset al*.* 2006). This highly variable catch efficiency of towed gears has to be considered when data are analysed and especially when absolute values are important e.g. for estimating secondary production.

However, several attempts have been made to at least improve the sampling procedure in order to generate more quantitative results, which include gear modifications to increase the catch efficiency of the gear (Rogers and Lockwood 1989; Kaiseret al*.* 1994; Jenningset al*.* 1999), estimations of the towing distance by using a meter wheel or a net probe to better quantify the towing distance (sampled area) (Carney and Carey 1980; Eleftheriou and Moore 2005) and standardization of trawling duration and speed to make the data comparable (Zühlkeet al*.* 2001).

Generally, when monitoring epibenthos, the group of species targeted for should be considered to use the best suitable technique

Towed gear and methods

Trawls, dredges and sledges towed over the sea floor are used for sampling epibenthos. Species caught depend mainly on the type of the gear, the mesh size, towing speed and sediment characteristics. Towed gears are destructive sampling devices and normally towed for short distances to obtain representative samples. The duration and speed of tow depends on the nature of the substratum and the gear being used. Small trawls and dredges are usually towed at 1-2 knots but speeds of up to 4 knots may be used for larger otter trawls and beam trawls targeting demersal fish species (Coggan et al. 2007).

Trawls and dredges are designed to be towed over the seabed. Different types of ground-gear, such as tickler chains, chain mats, rubber rollers etc., are attached to the trawl to increase its efficiency or selectivity.

Beam-trawls are mostly equipped with tickler chains or a chain mat to disturb the fauna in front of the net opening and consequently increase its catch efficiency. Different types of beam trawls are most often used to sample epibenthos. Smaller beam-trawls (e.g. 2m beam width) are mainly used to sample the invertebrate epibenthos, whereas larger ones (up to 8m) are predominately used to sample demersal fish with invertebrates as “qualitative” by-catch (see also Section 2.3.5.3.2). The Agazzi Trawl is a modified beam-trawl which can sample epibenthos with both sides of the trawl. This double-sided trawl is mainly used in deep waters, where it is difficult to control the orientation of the trawl at the sea bottom.

Otter trawls are mainly used to monitor demersal fish and few large mobile invertebrate species (see for description of the gear Section 2.3.5.3.1).

Dredges are simple heavy metal frames fitted with a bag or an (iron) net, predominately designed for sampling hard substrate, but also suitable for limited penetration and sampling of soft sediments. Especially the Anchor dredge type (Sanders et al. 1965) is used for (at least) semi-quantitative sampling of infauna, digging down to a defined depth of about 10 cm (Eleftheriou and Holme 1984). In the North Sea it has been applied in particular on gravel (Kennyet al. 1998; Kenny andRees 1994a, b). Examples of dredges are in Table 2.3.6.2.

Sledges typically consist of frame-supported samplers on runners to maintain the height or aspect of the front end for the collection of the fauna at or just above the seabed. Sampling is therefore more controlled than with simpler dredge designs (Rees et al. 2009). Examples of sledges and its characteristics are given in Table 2.3.6.2.

Table 2.3.6.2: Summary of standard sampling techniques epibenthos (towed gear) mainly based on Rees et al. (2009) and Coggan et al. (2007) if not mentioned otherwise.

	gear	accuracy	source of errors	comments	Information source
Trawls	Beam trawl (2m – 8m width)	Semi-quantitative; Accuracy depending on the mesh size, tow length should be recorded with depth sensor	Variable efficiency under poor weather conditions	Used in BTS to sample demersal fish (8m Trawl); 8m trawl only with large vessels	
	Standard beam trawl (2 m width)	See above; mesh size: 20 mm and cod-end 4mm stretched; standardized sampling procedure (tow duration 5 min; 1 knot towing speed)	Variable efficiency under poor weather conditions	Easy to deploy even from small vessels; widely used in the North Sea; enables comparison of data	(Jenningset al. 1999; Callawayet al. 2002)
	Agassiz trawl (1.5 – 3m width)	See above; double sided	See above; Limited number of tickler chains;	Often used in the deep sea	
	Otter trawl	See above	Less suitable for invertebrate sampling because of large mesh-size and type of ground-gear (e.g. rollers)	Only large vessels; used for sampling demersal fish e.g. IBTS (GOV)	(ICES 2006)

	gear	accuracy	source of errors	comments	Information source
Dredges	Anchor dredge (0.3-0.5m width)	Semi quantitative; allows sampling of coarser sediments; penetrating the sediment (endofauna); double sided	Variable efficiency under poor weather conditions	Easy to deploy even from small vessels	
	Scallop dredge (0.8m width)	Semi-quantitative; allows sampling of coarse sediments; can be deployed as multiple gear (up to 3 in a row); can be used to explore unknown seabed before using less robust gear; double sided	Variable efficiency under poor weather conditions	Heavy gear; multi gear can only be deployed from larger vessels	
	Rock dredge (0.6m width)	Semi-quantitative; allows sampling of coarse sediments; can be used to explore unknown seabed before using less robust gear; double sided	Variable efficiency under poor weather conditions		
	Kieler Kinderwagen (1m width)	Semi-quantitative; sampling of sandy and muddy bottom; double sided	See above; irregular bottom contact in stony areas	Easy to deploy	
	Triangular dredge (0.8m width)	Semi-quantitative; mainly with 30mm net; also on coarse grounds	See above	Easy to deploy	

	gear	accuracy	source of errors	comments	Information source
	Triple-D (0.2m width)	Semi-quantitative, but relatively reliable quantification of free living infauna and epibenthos; samples mainly large endofauna by penetrating the sediment; possible to sample a preset distance	See above	Heavy gear; only large vessels	(Bergman and van Santbrink 1994)
Sledges	Ockelmann sledge	Semi-quantitative; designed to skim sediment surface			
	Hyperbenthos sledge	See chapter “hyperbenthos”			
	Epibenthos sledge (1m width)	Semi-quantitative, but better controlled than other towed gears; opening/closing mechanism; sampling epibenthos and hyperbenthos		Heavy gear; only large vessels, but a small versionalso possible	(Brenke 2005; Rothlisberg and Pearcy 1977)

Sample processing (towed gear)

Most of the methods for processing epibenthos samples coincide with the methods for endobenthos samples (see Section 2.3.6.1). Thus, we only address here a few issues, which are specifically important for epibenthos sampling. For further details see Callaway et al. (2007), Coggan et al. (2007) and Rees et al. (2009).

- Sieving: the choice for a sieve mesh size is mainly depending on the mesh size of the gear deployed. Nevertheless, standardized trawl-sampling procedures (standard 2m beam trawl) were used and recommended for North Sea wide epibenthos surveys; with a sieve cascade of 5mm and 2mm (the later for qualitative data).
- Colonial animals: for registration of colonial animals the most suitable way to estimate is presence/absence registration. In addition, the weight of the colony might be taken.
- Biomass (wet weight on board): when weighing species on board it has to be taken into account that this will always be wet weight. A motion compensated balance is required.
- Measuring species: when measuring epibenthos species (for example for production estimates) it is recommended to use a standardized way of measuring, e.g. Callaway et al. (2007), since the way to measure is dependent on the species.

Acoustic methods

In recent years hydroacoustic methods were used more and more frequently to determine not only sediment types and bed morphology, but also macrofaunal distribution patterns (Hewittet al*.* 2004; Birchenoughet al*.* 2006; Degraeret al*.* 2008). Up to now, the detection of benthic species by means of acoustics is mainly restricted to species creating distinct biogenic structures or to epibenthic species occurring in dense mats on the sea floor. The main acoustic methods used are multibeam sonar systems and side-scan sonar. The details are given in Section 2.1 (Habitat Mapping).

2.3.6.3 Monitoring hyperbenthos

Contributor: Lene Buhl-Mortensen (IMR)

Introduction

The hyperbenthos is a term applied to the association of small sized bottom-dependent animals (mainly Crustaceans) that have good swimming ability and perform, with varying amplitude, intensity and regularity, seasonal or daily vertical migrations above the seabed (Brunel et al. 1978). Beyer (1958) was the first referring to "hyperbenthos" that has to be used in preference to "suprabenthos", because the Greek noun benthos should be preceded by a Greek prefix (hyper-), rather than by its Latin equivalents (supra-, super-). Most of the hyperbenthic species are present in much higher densities than in either the overlaying water layers or in the adjacent sediment and most of them are not there accidentally.

Beyer (1958) discussed the species found in abundance in the hyperbenthos but which were rare or absent elsewhere. More recently, there has been recognition of a distinction between the truly hyperbenthic species and a variety of "visiting" or "immigrant" animals that can be classified as endobenthic, epibenthic or planktonic. There has been also increased interest in the role of hyperbenthos in the functioning of marine ecosystems, mostly because it has been found that many demersal fish and epibenthic crustaceans feed, for at least part of their life, on hyperbenthic animals.

There is renewed interest in pre-recruit studies as many larval and early post-larval fish and crustaceans have a hyperbenthic phase. On the other hand, studies of benthic-pelagic coupling related to energy fluxes rarely include samples taken within a few centimeters above the seabed and may, therefore, underestimate significantly the flux of particulate organic material (Mees and Jones 1997). The often highly mobile hyperbenthic animals living immediately above the seabed are only occasionally caught by conventional benthic or pelagic sampling gears thus a plethora of hyperbenthic sampling devices have been constructed and used with varying success. The choice of sampling equipment in use depends largely on local conditions *e.g.* size of the ship, power and capabilities of the lifting gear, degree of exposure, depth, bottom relief and sediment structure and the type of sample required for the specific research topic under investigation (Eleftheriou and Holme 1984).

Hyperbenthic species (consisting of mysids, isopods, or amphipods) are characterized by their swimming capacity (see Sainte-Marie and Brunel 1985; Mees and Jones 1997). Until recently, hyperbenthic species were not included in benthic production models because they were not well represented in benthic samples taken with traditional endobenthic–epibenthic gear (cf. Huberdeau and Brunel 1982), i.e., grabs or box corers, which correctly estimate only the density of infauna or non swimming epifauna (Buchanan and Warwick 1974; Sainte-Marie and Brunel 1985).

However, there is increasing evidence that the hyperbenthos both contributes an important part to the overall benthic fauna diversity and composition (Buhl-Jensen 1986; Buhl-Jensen and Fosså 1991; Buhl-Mortensen 1996; Miskov-Nodland et al. 1999; Buhl-Mortensen et al. 2009) and in addition plays a key role in energy transfer in the benthic boundary layer (e.g., Sorbe 1981; Mees and Jones 1997; Cartes and Maynou 1998).

High *P*/*B*s have been reported for hyperbenthic peracarids (San Vicente and Sorbe 1993, 1995; Cartes and Sorbe 1999) from coastal, shelf, and bathyal communities, leading to the question of whether hyperbenthic species show generally higher *P*/*B*s than infaunal species (Cartes and Sorbe 1999). Failure to address these presumably high *P*/*B*s of hyperbenthic species would imply ignoring a nontrivial part of the benthic production if hyperbenthos is not sampled, resulting in biased models of energy flow in trophic webs. Fig. 2.3.6.1 shows a comparison of P/B values for hyperbenthos and infauna crustacean and and other invertebrats (Cartes et al. 2002).

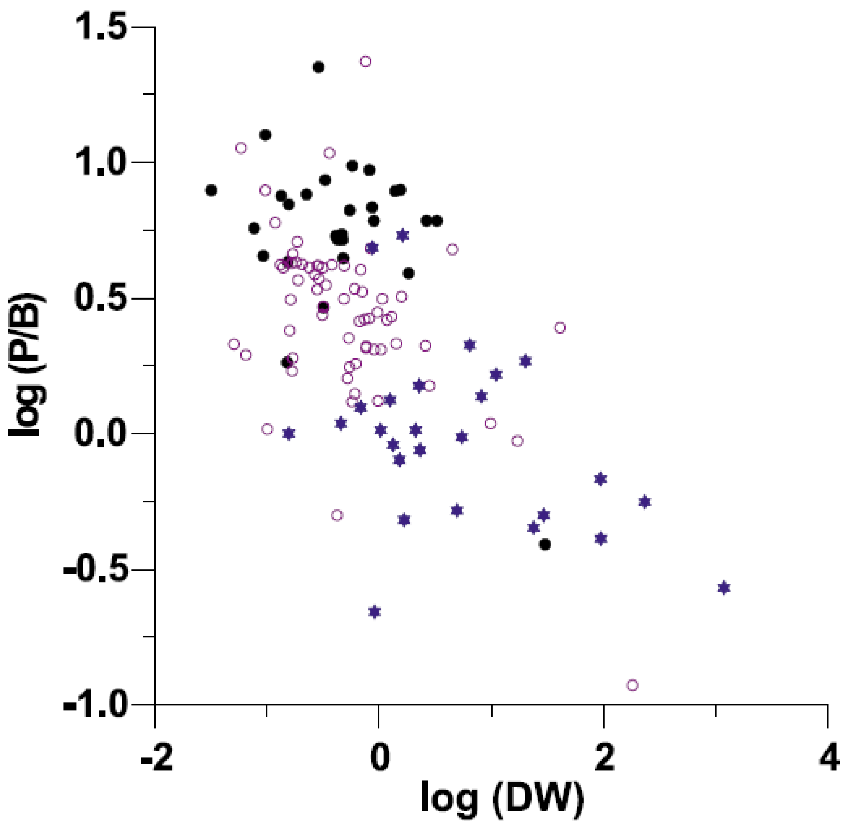

Figure 2.3.6.1: Log P/B (production–biomass ratio) vs. log mean body mass (dry weight, DW): hyperbenthic crustaceans (●); infaunal– epifaunal crustaceans (○); other marine invertebrates (*) (derived from Brey 1990) published in Cartes et al. 2002.

Sampling of hyperbenthos

Normally species are classified as belonging to either the hyperbenthic or the infaunal–epifaunal groups according to the gear with which they are collected: grabs or corers to sample benthos and sledges (Fig. 2.3.6.2) to sample hyperbenthos. Both types of gear capture quite dissimilar fauna and they are complementary methods, as indicated by the different dominance of peracarid crustacean species captured with box corers and sledges (Sainte Marie and Brunel 1985; Elizalde 1994; J. Cartes, unpublished data).
Hyperbenthos is only correctly sampled using sledges because of its swimming capacity, which is the main distinctive feature of hyperbenthos compared with benthos.

Presently, because of the incomplete knowledge of the biology and ecology of most species, no biological criteria are available to separate benthic and suprabenthic/hyperbenthic species.

Sledge-mounted designs are used for sampling the hyperbenthos (see e.g. Rothlisberg and Pearcy, 1977; Fosså et al. 1988; Brattegard and Fosså, 1991; Mees and Jones, 1997; Dauvin et al., 2000; Eleftheriou and Moore, 2005). These typically employ fine meshed collecting nets (down to 0.5 mm), flowmeters, and opening/closing mechanisms to facilitate quantification, and several of them (e.g. Brunel et al., 1978; Brandt and Barthel, 1995; Dauvin et al., 1995) support additional frame mounted nets to determine vertical zonation. Hyperbenthic organisms, especially crustaceans, can represent a significant food source for fish and are therefore an important target in resource assessment surveys. Such surveys must allow for the diurnal changes in swimming activity exhibited by many species (e.g. Kaartvedt, 1986). An example of a hyperbenthic sampler is illustrated in figure 2.3.6.2 (see also Rees et al. 2009). Epibenthic sledge provides semi-quantitative samples and does not sample benthic organisms equally well. Organisms living deep in the sediment are less likely to be caught than those inhabiting more superficial layers. Nevertheless they are known to provide large and rich samples of soft bottom benthos. The samples are semi-quantitative, however if the gear is carefully handled the samples are comparable (Battegard and Fosså 1991).

Figure 2.3.6.2: Details of a hyperbenthos sledge (see Buhl-Jensen, 1986; Brattegard and Fosså, 1991; Miskov-Nodland et al., 1999) showing (top) the fine mesh collecting net extending back from the Sledge-mounted rectangular sampling box, and (bottom) the front end of the sampler with Spring-loaded opening/closing mechanism.

Hyperbenthos should be sampled during daytime unless the scope of the study is not to involve diurnal migration (hyperbenthos migrate). When sampling with an epibenthic sledge, the speed during the 10 - 20 min long hauls is kept at around 1 knot to avoid organisms escaping at lower towed speed and a pressure wave in front of the grear at higher speed. Samples are sieved through a 0.5 mm screen (or finer) and preserved in 4% neutralised formalin. Before identification, the samples are transferred to 96% alcohol.

2.3.6.4 Monitoring mega-zoobenthos with visual techniques: SCUBA or free diving

Contributor: Stelios Katsanevakis (HCMR)

Plot Sampling with SCUBA or free diving

Plot sampling is the most commonly used method to conduct underwater visual surveys (UVS) for estimating density and/or abundance of populations of mega-benthic invertebrates. The basic idea of plot sampling is to "scale up" the counts of animals from the covered (surveyed) area to the study area (Borchers et al. 2004). Similarly, instead of density or abundance, the percent of species cover may be estimated for sessile species such as corals, sponges, and encrusting bryozoans. The key assumption of plot sampling surveys is that all individuals present in the surveyed region are detected. There are many different types of plot sampling that vary based on the shape of the plot, which is usually a square, a strip, or a circle, but they are all identical from a statistical point of view. The selection of plot shape and size largely depends on the size and density of the target species, its behaviour, the habitat type, and the necessity to satisfy the assumption that all individuals within the covered region are detected.

In general, for small-sized sessile or of very low-mobility species in high densities, quadrat sampling is more practical. Such sampling is conducted using small metal (usually aluminium) or plastic square frames that are easily manipulated by one diver (Fig. 2.3.6.3). For large-sized species in low densities, strip transects (synonym: belt transects) are preferred to increase the surveyed area and thus the sample size. The surveyed area of a strip transect may be increased by increasing its length and/or width. However, the wider a strip transect the more probable to violate the assumption of perfect detectability. In such a case the abundance estimator is negatively biased by a factor equal to the detection probability. Imperfect detectability is usually not an issue in small quadrats but it might cause serious bias in strip transects.

Figure 2.3.6.3: Underwater quadrat sampling. The square frame may be subdivided in smaller squares to facilitate counting (or even to collect separate data for each sub-quadrat and also analyze small-scale variation of counts). Photo: Yiannis Issaris.

Abundance estimation of small invertebrates on rather flat surfaces may also be conducted by photo quadrats or high-resolution digital photography and digital image analysis. This is also a plot sampling approach but instead of counting individuals or percent species cover *in situ*, photos of quadrats are taken and analyzed later in the lab. In high-resolution digital photography, images are usually obtained in multispectral mode, making it possible to eliminate radiometric distortion caused by panoramic effects like shadows and changes in rock surface coloration, and thus achieving better species identification and abundance estimates in comparison to traditional photo quadrats (Pech et al. 2004).

UVS with plot sampling have been conducted to estimate density and/or abundance of a great variety of invertebrate taxa, such as gastropods, cephalopods, bivalves, holothurians, sea urchins, barnacles, crabs, lobsters, hermit crabs, sponges, and bryozoans in a variety of habitats such as rocky or coral reefs, sandy bottoms, and seagrass beds (e.g., Keough 1989; Edgar and Barrett 1997; Tuya et al. 2000; Guidetti et al. 2004; Katsanevakis and Verriopoulos 2004; Oigman-Pszczol and Creed 2006; Shears et al 2006; Barnes et al. 2007; de Voogd et al. 2009). An important limitation to the application of UVS with SCUBA or free diving is depth. When air or enriched air (NITROX) is used as a breathing gas, a depth of ~40 m may be considered as the lower limit. By using Helium-Nitrogen-Oxygen mixes (TRIMIX) the operational depth is shifted deeper and may even exceed 100 m; however, the cost and effort increases substantially. Thus, to survey deep-dwelling invertebrates other alternative methods, such as ROV surveys, are selected.

Distance Sampling with SCUBA

Distance sampling comprises a set of methods for estimating density and/or abundance of biological populations (Buckland et al. 2001, 2004). Line transect sampling is the most widely used distance sampling method in the marine environment (see Section 2.3.1 for details on the method). Underwater distance sampling has not been widely used to estimate population density and/or abundance of benthic invertebrates. Despite its advantages in relation to plot sampling, the latter has been the selected method in the majority of underwater surveys for abundance estimation of benthic invertebrates. The extra effort of distance sampling is the collection of the perpendicular distances of all detected individual from the line or point, which are used to estimate detectability. Until now, distance sampling with SCUBA has been applied to study populations of the Mediterranean fan mussel *Pinna nobilis* (Katsanevakis 2006, 2007; Katsanevakis and Thessalou-Legaki 2009), the Mediterranean scallop *Pecten jacobaeus* (Katsanevakis 2005), and the sea anemone *Alicia mirabilis* (Katsanevakis and Thessalou-Legaki 2007). Line transect sampling has been applied in all the above mentioned cases; there is no application in the literature of underwater point transects to survey invertebrate populations.

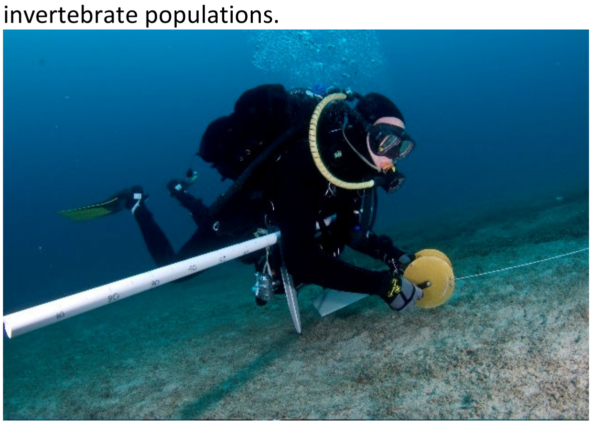

Figure 2.6.3.4: Underwater line transect sampling. The researcher rolls the line with the use of a diving reel after completing a line transect. A two-meter graduated plastic rod was used for distance measurements. Photo: Yiannis Issaris.

In underwater distance sampling, a nylon line with distance marks that is deployed using a diving reel seems the most efficient way to define the transect line (Fig. 2.3.6.4). In this way, the transect line is physically defined and its length is easily and accurately measured. Another option might be to not physically define the line but to move along an imaginary line using a compass but there are difficulties in accurately estimating the transect length and, more important, in accurately measuring the perpendicular distances of the target animals from the line.

The most important assumption of distance sampling is that detectability is certain 'near' the line and falls off smoothly at greater distances.Thus, the observer must try to optimize the detection of individuals in the vicinity of the line. When surveying sessile or slow-moving invertebrates, the best way to satisfy this assumption, is to swim slowly just above the line, preferably moving stepwise in small length increments than continuously, and for every length increment to make a thorough scan directly on the line and close to it and then look at greater distances on both sides of the line (Katsanevakis 2009).

Detectability and the shape of the detection function in an underwater benthic survey may depend on various factors such as habitat type, water turbidity, weather conditions, sun position, the ability of the surveyor, the target species and its brightness, contrast, age, sex, or size (Fig. 2.3.6.5). Buckland et al. (2001) recommended a minimum sample size of 60-80 detections for reliable estimation of the detection function. However, especially for endangered, rare, or cryptic species, it is possible that this sample size will not be achieved in a survey. If the rare species has a similar size and provides similar visual cues as a more abundant species, then their detections could be pooled to estimate a common detection function and detection probability. This would give increased precision to the estimated detection function, conditional on the assumption of equal detectability, which should be tested and supported by the data.

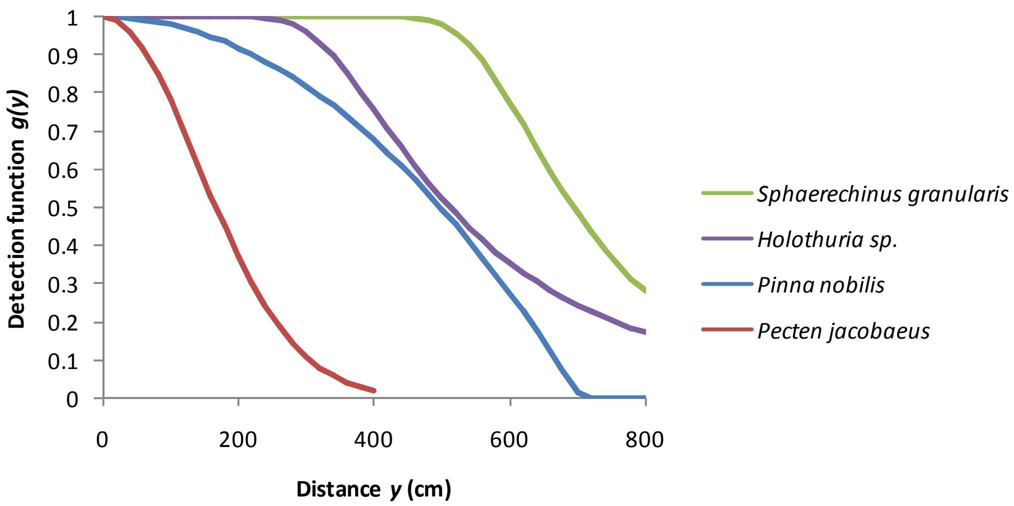

Figure 2.6.3.5: Variability of the detection functions of four benthic species (the Mediterranean scallop *Pecten jacobaeus*, the fan mussel *Pinna nobilis*, a sea cucumber Holothuria sp., and the purple sea urchin *Sphaerechinus granularis*) as estimated by line transect surveys conducted in the same location (Lake Vouliagmeni, Greece) at the same time (summer 2004) [figure taken from Katsanevakis 2009b].

A topic for which significant methodological advances are expected in the near future is model-based inference from distance sampling survey data and the integration of modelling with Geographic Information Systems (GIS) (Buckland et al. 2000). A first step towards this direction was made by Hedley (2000) and Hedley and Buckland (2004), who proposed two approaches that enable spatial variation in animal density to be modelled using standard generalized linear modelling (GLM; McCullagh and Nelder 1989) or generalized additive modelling (GAM; Hastie and Tibshirani 1990).In the first approach (count model), the transect lines are divided into smaller discrete units, and the expected number of detections in each unit is modelled using explanatory spatial covariates. The only applications of this approach for benthic invertebrates have been reported by Katsanevakis (2007) and Katsanevakis and Thessalou-Legaki (2009) who modelled the population density of the fan mussel *Pinna nobilis* in two different areas. In these cases, apart from abundance estimates, the species density was related to spatial and environmental variables, such as depth and habitat type, and distribution maps were produced.

With such a model-based approach, it is not required that the line transects are located according to a formal and restrictive survey sampling scheme. This is quite important, as in underwater benthic surveys, it is not always easy or feasible to strictly follow a survey design that ensures uniform coverage probability. Provided that the line transects give a good spatial coverage of the study area and the spatial model is a good approximation of reality, spatial models could offer more reliable estimation of abundance than traditional distance sampling (Katsanevakis 2007). Another benefit of using a density modelling approach is that abundance may be easily estimated in any subset of the study area by simply numerically integrating under the relevant section of the fitted density surface, while conventional distance sampling restrict abundance estimations to the whole study area or to predefined (at the design stage) strata (Hedley et al. 2004). Density modelling approaches are expected to provide an improvement in the precision of the abundance estimate, as part of the spatial variability is actually modelled (Katsanevakis 2007).

Other methods based on SCUBA or free diving

Line Intercept Transects, Point Intercept Transects, Chain Transects

Line intercept transect (LIT) or point intercept transect (PIT) surveys (Loya 1972, 1978) are applied to estimate the percent cover of sessile species such as corals, sponges or encrusting bryozoans (e.g., Beenaerts and Vanden Berghe 2005; Nadon and Stirling 2006; Leujak and Ormond 2007; Pawlik et al. 2008). In these methods a transect line is laid over the bottom (usually coral or rocky reef). In LIT the length of line overlying various kinds of organisms is then measured (usually with an accuracy of 1 cm), either by graduations on the transect line or by a hand-held meter tape or stick. Some researchers have used chain (chain transects) instead of transect line (Zea 1993), counting the number of chain links overlying each kind of substrate instead of measuring distance of coverage. Chain transects is the only method that provides information on substrate rugosity. However, it can damage fragile and fan shaped or branching organisms and should be avoided in such cases. In PIT, the line is marked at fixed points and sessile benthic organisms or substrate categories directly beneath the marks are recorded.

LIP and PIT methods tend to be quicker than plot sampling methods; in comparison to quadrats they are less sensitive to small scale spatial variation (Beenaerts and Vanden Berghe 2005). PIT is much faster than LIP (twice as fast according to Beenaerts and Vanden Berghe 2005) and appears to be more time- and cost-efficient (Nadon and Stirling 2006; Leujak and Ormond 2007), which is important when there are field time limitations. LIT, in addition to percent cover, can provide population densities as number of individual colonies per unit area (Lucas and Seber 1977).

Fixed-time swims or SCUBA surveys

This is a semi-quantitative approach as it does not provide an estimate of absolute density/abundance but rather relative values (i.e., abundance indices) or only presence/absence data. It is rather the simplest and fastest underwater visual monitoring method. In such kind of surveys, the observers swim or dive freely (i.e., not necessarily following a specific transect) for a predefined length of time within the habitat or region of interest recording species seen (either only presence/absence or counts of individuals). In surveys of species composition, replicate timed swims could be searched until the habitat has been adequately sampled (e.g., the point at which a species accumulation curve levels off). This method has the advantage of being fast and therefore inexpensive. However, caution is needed in comparisons of relative densities from such surveys as detected differences might be due to differences in detectability and not due to real density differences.

Nearest-neighbor and point-to-nearest-individual

The key idea is that when animal density is high, distances to the nearest animals (from a randomly selected animal or point) are shorter than when density is low (Borchers et al. 2004). In the nearest-neighbor method, individuals within the survey region are randomly sampled and the distances to their nearest neighbor are measured, while in the point-to-nearest-individual method, the distances from randomly selected points within the survey region to the nearest individual are recorded. These methods have been used to estimate abundance of some marine benthic invertebrates such as abalones (e.g., Officer et al. 2001). However, Borchers et al. (2004) recommended against the use of these methods for various reasons: they are impractical for sparsely distributed animals, the methods are inefficient and inappropriate for mobile objects, and the state model, which specifies that animals are uniformly distributed throughout the survey region is usually unrealistic. If the distribution of individuals tends to be more aggregated than a random pattern (which is usually the case for marine invertebrates) abundance is underestimated by the point-to-nearest-individual method and overestimated by the nearest-neighbor method (Borchers et al. 2004). McGarvey et al. (2005) investigated the performance of these methods, when applied to estimate greenlip abalone (*Haliotis laevigata*) density. They found substantial bias in the estimates (between 18% and 55%), which was attributed to the clustered distribution of the species.

2.3.6.5 Monitoring mega-zoobenthos with visual techniques: applications with submersibles/ROVs/drop cameras

Contributor: Pål Buhl-Mortensen (IMR)

Introduction

This summary is largely based on the Norwegian standard for visual seabed surveys (NS 9435:2009).The English version of this standard is currently in review for European Standardisation Committees. Information on the habitats, biotopes and biodiversity on the sea floor is an important part of ecosystem-based environmental management, and necessary in order to evaluate the consequences of various anthropogenic activities. Implementing the EU's Water Framework Directive and required monitoring of biodiversity will require documentation and monitoring of different types of sea floor types using inter-comparable and non-destructive methods. Many sea-floor areas are difficult to investigate using traditional sampling methods such as grabs and dredges. Visual surveillance using geo-referenced positions which allow locations to be revisited allows a documentation of conditions and changes in indicator organisms that otherwise would be difficult to achieve. The standard present methods particularly suitable for mapping and monitoring of distribution and diversity of habitats and species on hard bottom substrates below the depth of SCUBA diving. They are also suitable for description of distribution and occurrence of large and scattered organisms on soft bottom, where sampling with grab do not provide representative results. Remotely Operated Vehicles (ROVs) and passive tethered observation platforms are used for mapping and environmental surveys of the sea floor. However, the methods used and the results obtained have been rather variable, with respect to geographic positioning, taxonomic precision and quantification.It is therefore important that the methods used are standardized in order to compare results.

Definitions

drop camera	video- or still image camera that is lowered to the sea floor, and which takes a photograph at a single position
geographic precision	accuracy with which a given point can be relocated within a geodetic reference system
geographic resolution	the lowest unit of measurement at which a geographic distribution can be reproduced
locality	geographic description of a place or an area where samples are collected covered by one or more sampling stations
macrofauna	animal species > 1 mm
megafauna	animal species > 20 cm
monitoring	investigation to record environmental conditions and any changes over time
reference location	location representing presumed natural environmental conditions in the monitoring area
remotely operated vehicle ROV	remotely operated motorised underwater vehicle equipped with video camera and often also possibilities for mounting additional equipment such as sonar, environmental probes, manipulator arms and sampling equipment
sample	Still photograph or video sequence
sampling station	area where photographs or video recordings are taken
still photograph	photograph of a delimited area
towed video platform	
tethered video platform	passive sampling gear comprising a supporting construction onto which a video camera and lights (as required also still camera and various environmental sensors) are mounted
transect	defined and continuous line or belt of pictures or video sequences across a delimited area
video sequence	part of a video film

Survey programme

A survey programme should be designed in accordance with the aims of the investigation, required precision of results, local topographical and hydrographic conditions in the survey area, information on local sources of pollution, experience from previous investigations and any other factors that can be of significance for the surveys. The survey programme should be established before commencement of the survey, but appropriate modifications may be made during the fieldwork, as judged necessary.
Visual surveys of the sea floor may be carried out by still photography or video, and may be divided into three main types as follows:

- Pilot survey
- Mapping
- Trend monitoring, investigations over time

Pilot survey

This is a general mapping of the bottom conditions and habitats, including dominant organisms. A pilot survey may be used as a reconnaissance survey for future more comprehensive recording and surveying of environmental conditions. This type of survey is not suitable for recording changes over time, with the exception of major changes in dominant organisms and bottom conditions (underwater sediment slide, bottom trawling tracks etc.). The requirements for methodology and reproducibility generally are relatively simple.

Mapping

This is a survey to describe the distribution of bottom types and organisms in a given area. Visible organisms should be identified to lowest possible taxonomic level and quantified.Bottom types should be described in accordance with Table 2, and the relative composition should be determined.

Video recording or still photography along transects is the most appropriatemethod for mapping. Transects can be located as distributed single lines or in a pattern of parallel lines. The distance between transects and still photographs is determinedby the demand for geographical resolution in accordance with the aims of the survey. The degree of spatial variation of habitats and species composition is determining for how dense transects and still photographs should be to provide reliable material for production of areal maps. For areal mapping where this variation is not known, transects should be parallel separated with a distance of maximum 50 m. Information from more widely scattered transects can be used for generating areal maps combined with prediction and verification. Still photographs should be more than 20 m apart, corresponding to one photograph per minute at a stable speed of 0.7 knot. The total length of transects is determined by the aims of the mapping. If the aim is a representative description of species diversity of observed fauna (large macrofauna and megafauna), the total length of transects on a location should be at least 600 m.

For mapping of smaller areas or single habitats the length of individual transects should be adjusted to fit the shape of the area or the extent of the habitat. For areas smaller than 600 m in longest direction the required minimum distance of 600 m transect length is not relevant. In addition to positioning of start and end points geographical positions should be recorded continuously along the transects. Start and end point noted on a log sheet for each transect, and continuous recording of positions should be logged by a computer or recorded as a text overlay on the video records.

Trend monitoring

General

The aim of this type of survey is to investigate bottom types and assemblages of organisms, or one or more selected bottom-dwelling taxa over time, in order to document natural variations and any changes over time. Changes may be detected in the abundance and distribution of individual species, selected indicators such as individual size, and observed mortality, or the composition of bottom substrates. Monitoring can be carried out at fixed locations or by means of repeated parallel transects within defined areas. The survey should be carried out using semi-quantitative or quantitative methods following a predefined programme. A reference location or reference area should be sampled to ensure that the data can be used for comparison with the environmental conditions in adjacent areas.

Trend monitoring at fixed stations

Trend monitoring of fixed stations is carried out by collecting still photographs from a unit area that can be identified and relocated by means of markers or naturally-occurring points of reference, such as large rocks or bedrock features. The sampling area to be monitored may be marked using positioned air-filled glass buoys anchored to the sea floor with lead weights (gas-filled objects are clearly detected by sonar, and therefore facilitate relocation). The geographical position of the marker should be noted in order to re-visit the exact area. An area in the immediate vicinity of the marker should be photographed to help relocate the sampling location.

Trend monitoring requires the use of an ROV in order to take photographs from exactly the same spot on the sea floor during each visit.When photographing the fixed locations, the ROV should be positioned at the same place and in the same direction relative to the marker, for each sampling session. Photographs or video films from previous field surveys can be used as references for exact positioning. Information concerning each photograph should be noted in the sampling logbook, as described under the section for recording. A minimum of three fixed locations should be photographed at each station in order to ensure that these are representative of the area at large.

Trend monitoring using video transects

Trend monitoring using transects is suitable for documentation of changes within areas with only slight variations in habitat/ bottom type. Parallel transects give representative observations and also are suitable for mapping of local distributions of organisms and bottom types. A minimum of three transects is required.The minimum combined length of the transects is dependent on the aims of the individual survey. If the aim is to detect changes in the diversity of organisms, the combined transect lengths should be at least 600 m. If the area under investigation is less than 200 m across, the number of transects should be increased, to achieve a total length of 600m. The total transect length can be shorter if the aim of the survey is to detect changes in local distribution limits for habitats or selected species. In such cases the number and length of transects should be adjusted in accordance with the aim of the investigation. In all other aspects the survey is carried out as described for transect survey.

Reference location

As part of surveys of areas affected by natural or human caused factors, a reference location outside the impacted area should be selected. Reference locations should to a largest possible degree represent the natural state without influence of any local sources, and should provide a measure of natural temporal and spatial variation in benthic communities. Reference locations should be included in surveys where comparison of the fauna outside the influenced area is needed, or where knowledge about the natural variation is crucial. If reference locations are used these should be comparable with the impact locations, and the investigations should be performed in similar ways under similar conditions.

Survey execution

Equipment

The technical specifications for the equipment used should be described when reporting the results. The requirements made of the equipment are dependent on the aims of the survey. For mapping and monitoring, colour camera should be used, together with underwater positioning equipment with an error margin of ≤ 2 m + 5 %.

For the purpose of mapping shallow coastal areas using a drop camera the ship’s GPS can be used without hydro acoustic positioning.In open sea areas and in areas with strong currents, the ROV should be equipped with a sufficiently strong motor or ”garage” to avoid drift from the targeted locality (at a fixed position or between two fixed positions).
Still photographs for use in trend monitoring should document an area of between 0.25 m^2 and 1 m^2 with a good image quality (focus and contrast) with a minimum resolution of 1080x1560 pixles (HD-format, equivalent to 300 DPI at 9x13 cm).

For quantitative surveys, the geographic position of the observation platform on the sea floor is determined using a known error margin. When using a towed observation platform, the vessel's positions may be used, after correction for the known deviation, provided that the vessel's course and speed are stable.

Positioning

General

Geographic references for observations should be given at an accuracy appropriate to that of the limitations of the equipment/method.

Positioning should be carried out with reference to a grid-net system (for example European Datum: ED-50, European Reference Frame 1989: EUREF89, World Geodetic system: WGS-84, or in accordance with the UTM-system of grids).

Geographic references (beyond general locality: +/- approx. 100 m) should be based on hydro-acoustic positioning. When using a towed video platform or drop camera, its position at the bottom can be estimated from the vessel's position by correcting for deviations in relation to the observation platform (cable length, angle, direction). In all cases, the method used should be documented.
There are several sources to errors in positioning of underwater equipment. The main components in underwater positioning provides transmission of satellite signals to the ship and calculation of the distance and direction to the observation platform. The quality of underwater positioning is mainly depending on how the ship is equipped, but the setting and calibration of this equipment is also very important.

For mapping and monitoring the hydro acoustic positioning equipment needs to be calibrated at an annual basis. If a calibration has been performed for instance by comparison with a transponder placed on the sea bed, values for the error should be provided in the report. If such a calibration has not been made the errors provided by the producers of the equipment should be used instead.

Filtering of navigational data can significantly reduce noise. The recommended method for this is Kalman filtering.

A simpler method for filtering navigational data is to remove deviant recordings that are obvious outliers from the remainder of the recordings. Deviant values can be replaced by a value derived from the running mean of five records (two before and two after the point of the deviant record) in the series of navigational recordings. If filtering of navigational data is used, the method used should be documented when reporting the results.

The geographic resolution can be obtained by comparing the distances covered by video sequences of similar lengths with the distance as calculated using speed.

Positioning for pilot surveys

For pilot surveys, positioning may refer to the position of the vessel. The positions of video transects should as a minimum be defined by the start and end-positions. The precision of positional information should fulfil the requirements of Order-2 in S-44 (20 m + 5 % of water depth).

For drop camera a calculated position for where it hit the bottom is satisfactory. This position is estimated based on the offset between the location of the ship positioning system's centre point and the location on the ship where the drop camera is deployed.

Positioning for mapping

For mapping, positions should be recorded at regular intervals (at least one record per 10 s) during the video recordingsThe precision of the positional information should as a minimum be ± 2 m + 5 % of water depth.

Positioning for trend monitoring of fixed stations

For trend monitoring using still photography, the positioning should be accurate enough to allow relocation of the exact location on the sea floor in order to be able to follow developments in individuals/populations using positional data for markers and/or photographs/video recordings from previous surveys.

Transect surveys

In general, the camera should be at least 3 m or closer to the bottom in order to identify organisms with a size <10 cm, or to estimate percentage composition of bottom substrates. Records from a greater height can only be used for mapping of dominating species > 10 cm or qualitative registration of coarser bottom substrates (cobble, boulder and bed rock). The elevation above sea bed is measured by an altimeter (acoustic instrument measuring the elevation above bottom) or by using trigonometry.

Estimation of height using trigonometry demands that the distance from camera lense to the centre of the image and the camera's inclination angle is known. The distance from the lense to the centre of the image from the width of the field view (scaled by parallel laser points) and the angle of view.

As far as possible, an even height (1 m - 3 m for mapping) and speed (0.5 - 2 knots for pilot surveys and 0.5 - 1 knot for mapping) should be maintained. The optimal distance from the bottom is dependent on the type of survey and field of vision as given by the angle of the lens, light conditions, image resolution (contrast and focus) and visibility. The image quality for video is reduced by high speeds, which additionally complicates identification of organisms and density estimates. The maximum speed for mapping is 1 knot, but to ensure good image quality, the speed should not exceed 0.8 knot.

For quantitative faunal surveys, the areas covered should be estimated based on the field of vision and the distance travelled. For calculation of observed area, the photographic field should be scaled using laser points or calculated using the lens and camera angles together with height above the bottom. Height should be recorded to the nearest decimetre. The angle of the lens and the angle of the camera relative to the bottom should be recorded in degrees.

On a sloping or uneven seabed calculation of areas based on trigonometry using height above the seabed is complicated and not practical. In such cases laser scaling points are needed.

For mapping where the field of vision is 1.5 - 3 m, a distance of 10 cm between the laser points is recommended. For pilot surveys conducted at a greater height above the bottom and a wider field of vision, the distance between the laser points can be increased to 20 cm.

The length of transects to be covered in order to ensure comparable results is dependent on the aims of the survey. For mapping of biodiversity and abundance distribution, the transect length as a minimum ensure that a further 10 % increase in transect length would not result in more than a 10 % local (within the transect) increase in recorded taxa. In practice, this is equivalent to approximately 400 m areas with a uniform habitat. In areas of habitat heterogeneity, this length should be increased to 600 m. It can be difficult to control the distance covered in ”real time” and therefore it may be practical to calculate the distance covered based on the speed of the ROV, or vessel in the case of towed observation platforms.

Information on geographic position, depth and time, together with height above the bottom and camera angle if appropriate, should be recorded with reference to the image material in the form of time codes. This may be stored either as text on the video recordings or as a separate date file. For stored data files, it is essential to include a record of time of sampling, given as GPS-time, for the start of the video recording.

Processing and recording

Image analysis can be divided into two types:
analyses of video sequences;

analyses of still images (photographs or video ”frame grabs”).

Analyses should be standardised with respect to distance covered, for example by counting the numbers of organisms recorded per unit time. For colonial or encrusting organisms, the most common means of quantification is an estimate of the percent coverage of a specified unit area.

For quantitative analyses of video, the sequence lengths (sample size) should not be less than four times the error margins of the navigational data. For a navigational uncertainty of > 5 m, the use of video sequences < 20 m will carry an error margin of over 50 % of its actual length.

For quantitative image analysis, the pictures should have been taken perpendicular to the surface, to ensure a correct calculation of area and best possible identification of organisms. Calculation of area also may be carried out on sloping surfaces if the image can be scaled (for example by parallel laser points at known distances) and if the angle of the camera relative to the bottom is known. Quantitative data is presented as numbers of individuals or colonies per unit area, or as percent coverage. The percent coverage may either be measured directly, or by means of a ”point count” method with 100 points evenly distributed along parallel lines placed both horizontally and vertically on the picture. The extent of coverage of organisms or habitats is then given as the percent of the points that coincide with organisms or substrate. In cases where the image is not taken perpendicular to the bottom, the network of points should be adjusted such that the distance between the points reflects the perspective of the image. For analysis of macro-fauna, the standard unit area is given as a 50 cm x 50 cm frame. This area can be marked as a central field on the images after photographing/recording.

Bottom substrates

Bottom substrates should be classified in accordance with the Udden-Wentworth standard [3] for granulometric composition, see Table 2.3.6.3.

Table 2.3.6.3: Udden-Wentworth scale and categories of bottom substrate for use in visual surveys of the sea floor

Udden-Wentworth scale		**Survey type and min. category**	
Grainsize	**Seabed substrate**	**Pilot**	**Mapping/Trend**
0.6 µm – 3.9 µm	Clay	Mud/sand	Mud
3.9 µm – 63 µm	Silt		
0.063 mm – 2 mm	Sand		Sand/Granule
2 mm – 4 mm	Granule		
4 mm – 64 mm	Pebble	Gravel	Pebble
6.4 cm – 25.6 cm	Cobble		Cobble
25,6 cm – 410 cm	Boulder		Boulder
> 4 m	Bedrock	Bedrock	Bedrock

For the finer sediment fractions, silt and clay, which are difficult to distinguish on video recordings, the combined term “mud” should be used. In the same way, sand and granule are combined as sand/granule. Recognition of mud and sand/granule is based on structure in the image and not identification of single grains. Identification of grainsize can be aided by letting the camera platform/ROV touch the seabed to stir up some sediment. In cases of uncertainty about the seabed substrate category a sediment sample should be taken for ground truthing. This can be made using a grab from the ship or using a small sediment corer mounted on the ROV or the video platform.

If the percent composition of bottom types is to be recorded, this can be calculated as described above for analyses of macrofauna from still images.

Identification and quantification of organisms

For mapping and monitoring surveys of biological communities and diversity of megafaunal species organisms should be identified to lowest possible taxonomic level. Abundances should be recorded as numbers per unit area, for example per 10 m^2 for mapping and per $m^2$for trend monitoring using still images covering a smaller area. A qualitative abundance scale (Table 2.3.6.4) can be used if quantification is of little relevance, for examples for colonial organisms and in areas with mass occurrences of organisms.

Table 2.3.6.4: Scale for calculation of extent of coverage for qualitative recordings of organisms

Interval-code	Coverage (colonial/encrusting organisms)	Mass-occurring organisms (number of individuals or colonies·1m^2)
5	75 % - 100 %	>100
4	50 % - 75 %	50-100
3	25 % - 50 %	25-50
2	5 % - 25 %	5-25
1	< 5 %	1-5
0	Not present	Not present

Alternatively, abundance quantification may be carried out on sub-samples (”frozen” video images).It is often not possible to identify specimens with certainty, and also there may be considerable variation between individuals within the same taxon. In such cases, ”morphological species” may be used if their form, colour and size is described. For this purpose, a photographic reference collection should be included in the reporting of results.

2.3.6.6 Monitoring meiofauna (soft substrata)

Contributor: Simone Mirto (CNR-IAMC)

Introduction

The term meiobenthos was introduced by Mare (1942) to define organisms displaying a size intermediate between the smaller micro-benthos (i.e., bacteria, diatoms and a large fraction of protozoa) and the larger macro-benthos. The term meiobenthos includes both animal and vegetal components whereas meiofauna is referred only to animal organisms. From a dimensional point of view meiofauna are composed by all organisms of size ranging from 30 μm to 1 mm. Such size class includes both protozoa of large dimensions and metazoans, but in operational terms, generally, we refer only to the latter component. From a functional point of view meiofauna can be defined as small sized benthic metazoans whose individual biomass ranges from 0.01 to 50 μg (dry weight) and having life histories and feeding habits that allow defining meiofauna as a well defined entity different from largermacrofauna (Warwick and Gee 1984). Meiofauna are composed by organisms that live within the sediment, displaying morphological and functional adaptations in relation with the characteristics of the sediment. Meiofaunal taxa that account both interstitial and endobenthic forms (nematodes, copepods, turbellarians) display large morphological differences between sandy and muddy sediments. Sand meiofauna tend to be elongated to better dwell in the interstitial space among sediment grains, while muddy meiofauna are not restricted to a specific morphology and are generally larger. Meiofauna represent the most abundant class of organisms of marine benthos, reaching densities comprised from 10^5 to 10^6 individuals per m^2 (100-1000 ind. 10 cm^{-2}) and a biomass of 1-2 g DW m^{-2} in coastal systems at depth > 100 m (Coull and Bell 1979). Such high values of abundance and biomass might change as a function of season, latitude, depth, tides and sediment grain size.

Vertical distribution of meiofaunal taxa within the sediment is generally confined to the depth of oxygen penetration (which can be also measured from the Redox Potential Discontinuity depth, RPD). Most meiobenthic species are generally found in the top 2 cm of the sediment, which typically display oxygenated condition with Redox potential values >+400 mV.

Nematodes are the taxon numerically dominant in most meiobenthic assemblages, accounting for > 90% of total density. Harpacticoid copepods are generally the second group in terms of abundance, followed by polychaetes and turbellarians.

Meiofauna present strong temporal fluctuations. The reproductive cycle typical of meiofaunal organisms is annual, but studies demonstrated that some meiobenthic species possess life cycles of >3 years (Herman and Heip 1983) and can be found as temporarily inactive stage (cysts, resting stages; Pati et al. 1999). Moreover certain species are characterized by recruitment periods postponed in respect to other species to minimize competition. Meiobenthic organisms are mostly deposivorous, but are abundant also diatom grazers and, especially in deep-sea environments, bacterivorous. Nematodes display a sclerotized cuticle, which allows identifying, from the morphology of the mouth shape and size, the main food source. This allows a rather easy classification of nematodes into trophic guilds, which in turn are useful to investigate also trophic and functional diversity. Generally, nematodes are divided into 4 trophic groups on the basis of Wieser's classification (Wieser 1953): selective deposit feeders (or bacterivorous, labeled as group 1A), which feed mostly on bacteria; non-selective deposit feeders (group 1B), which feed on organic detritus; epistrate-feeders (group 2A), which feed on benthic diatoms and predators-scavengers-omnivorous (group 2B), which can prey upon other meiofauna and macrofaunal juveniles. Jensen (1987) proposed a new classification unifying deposit feeders in a single group, keeping epistrate-feeders as proposed by Wieser and dividing the group 2B in predators and omnivorous. Recently, Moens and Vincx (1997) developed Wieser's classification, further distinguishing the trophic group 1A in nematodes mostly feeding upon ciliates (ciliate feeders) and those feeding upon bacteria (microvores). Moreover, the trophic group "predators" (2B) is divided into obligatory and facultative predators, which prey only occasionally. Trophic characteristics of the different species within the same family are generally conservative and therefore, from a taxonomic point of view, the taxonomic level of genera and family is sufficient to investigate meiofaunal response to environmental changes (Herman and Heip 1983, Danovaro and Gambi 2002).

Despite the reduced individual size, meiofaunal secondary production plays a key role in benthic energetics. The small size, is indeed coupled with a high metabolic activity and a rapid turnover (i.e., as the ratio of meiofaunal production to meiofaunal biomass, P:B), which is on average 5 time higher than for macrofauna, thus yielding similar production values even in systems dominated by macrofauna in terms of biomass (Gerlach 1978).

During the last years, meiofauna has been largely used as collective indicator of alteration of the functioning of marine ecosystems. Due to their high sensitivity to environmental perturbations, to the high number of individuals (which make easier statistical analyses and reduce sampling volumes, Moore and Bett 1989), to the lack of larval dispersion and to the short life cycle, meiofauna are becoming a common target to evaluate the disturbance and re-colonization of marine environments (Danovaro et al. 1995, Mirto et al. 2002). In particular, meiofauna can represent a useful tool to assess the impact of several disturbance events on benthic domain, not only in case of pollution events (Sandulli and De Nicola 1991), but also in the case of introduction of"artificial reefs" (Danovaro et al. 2002) and mussel or fish farms (Mirto et al. 2000, 2002, La Rosa et al. 2001). Moreover, meiofauna can be utilized to investigate benthic response to particulate fluxes (e.g., river "plume"; Danovaro et al. 2000). The short generation time and the large number of organisms produced would lead to the conclusion that meiobenthic organisms are generally opportunists. In facts, whether from one side the lack of larval dispersion of meiofaunal taxa would delay substrate colonization time, from the other side is an advantage in pollution studies since the impact is not masked from larval immigration from other areas. However, due to the small size meiofaunal organisms require a strong sorting and identification effort. Only organisms that present a cuticle or an chitin exoskeleton ("hard body meiofauna") allow the identification even after preservation, whereas other groups, such as turbellarians require the observation of living organisms for their identification.

Sampling and data analysis

Methodologies and sampling equipments on soft substrates

When sampling on soft substrates, the use of corer is the best sampling method because corers allow analyzing a given surface area (Elmgren and Radziejewska, 1989) to collect undisturbed sediments. Meiofauna present a patchy spatial distribution so that the size of the corers is important. The use of several corers of reduced internal diameter (for samples or sub-samples) allows defining better than the use of a smaller number of larger corers (Elmgren and Radziejewska 1989). Corers are generally made of transparent Plexiglas (inner diameter ranging from 2 to 10 cm) with thin walls and a sharp end to facilitate their penetration into the sediment. The corer is inserted into the sediment to a certain depth and subsequently gently extracted from the sediment to keep it undisturbed, avoiding the mixing and the re-distribution of organisms along the vertical profile. Closing the corer *in situ* allows the transport in laboratory in undisturbed conditions. Transparent corers allow visualizing the depth of the redox potential discontinuity on the basis of changes in sediment colour. Small corers collected manually can be obtained cutting syringes of 50-60 ml (Chandler and Fleeger 1983). Manual coring can be carried out in both intertidal and subtidal areas.

In deep-sea systems multiple corers are used to collect several samples simultaneously (es. 4-12 corers). These instruments allow collecting undisturbed samples by penetrating slowly into the substrate. In this way, several cores allow a better analysis of the horizontal and vertical distribution of benthic organisms. Multiple corers possess liners with variable internal diameter (es. 2.5-10 cm) and a length up to 60 cm.

Another common sampling instrument is the box-corer (Fleeger et al. 1983). The weight of a box corer after sampling can range from 150 to >750 kg and collects a surface sediment of 0.02 - 1.0 m^2 down to a depth of 50 cm. However, during sampling recovery most box corers allow a surface sediment mixing, which might alter the quality of the sample. Both Bett et al. (1994) and Shirayama and Fukushima (1995), comparing sampling efficiency of multiple corers and box corers in deep-sea environments, pointed out that meiofaunal density collected using box corers was significantly lower.

At shallow depth often soft substrates are sampled using a benthic grab. One of the most utilized is the Van Veen grab that, if of adequate weight and size, allows sediment sampling down to 600 m depth. A benthic grab can contain from 0.5 to >60 L of sediment and can penetrate down to 15-30 cm. Once recovered on board, if the benthic grab is equipped with a top window, it allows the insert of corers for subsequent sub-sampling (Della Croce et al. 1997, Cognetti et al. 1999). However, several investigators questioned the use of this equipment for the low quality of the samples due to sediment mixing during retrieval (Bett et al. 1994, Shirayama and Fukushima, 1995).

Modalities of sample preservation

Once collected sediment samples can be processed immediately or, when this is not feasible, can be kept in conditions that allow the survival of meiofaunal specimens untill extraction. If samples are extracted within a short time lag, they can be stored at low temperature (lower than the ambient temperature) without being frozen. It has been estimated that muddy samples can be mainteined for 24-48 hours while sandy sediments can be preserved for longer periods. Normally, samples collected for quantitative analyses must be processed within few hours from collection, because the short generation time and the presence of voracious predators (nematodes and turbellarians) might alter the state of the sample. If the vertical distribution of meiofauna is of interest a vetrical sectioning is required immediately after collection, to avoid that alteration of abiotic parameters might affect the distribution of specimens characterised by a high mobility. When sectioning is not possible all corers should be frozen (-20 °C) and in this way they can be preserved for a longer period of time (Pfannkuche and Thiel, 1988).

An alternative to sample freezing is fixation with buffered formalin (using $Na_2B_4O_7$, pH ca. 8.2). The quantity of formalin to be supplemented to obtain a final concentration of 4% is calculated on the basis of the overal volume of sediment plus prefiltered seawater or distilled water containing $MgCl_2$ (80 g L^{-1}). Samples fixed with 4% formalin can be stored for a long time before being extracted and analysed. Soft-body taxa such as gastrotrichs, turbellarians and nemertins are subjected to major morphological alterations, but are generally conserved intact. Usually few drops of a solution of Bengal Rose (0.5 g L^{-1}) are added to the sample. The Bengal Rose is a protein-binding dye which is generally used to facilitate organisms sorting and counting.

Modalities of sample extraction

Meiofauna extraction from the sediment is carried out by means of mesh sieves. Such operation must be carried out gently and carefully to avoid damaging of organisms. In this way large portions of the sample are removed and organisms extraction is facilitated. To increase extraction efficiency, before passing the sediment through the sieve, the sample can be treated with ultrasounds (1' for 3 times with 30" intervals) in order to detach organisms from grain particle surface. However, the effectiveness of this treatment to prevent sample degradation must be accurately tested because longer treatments ensure higher preservation but can damage meiofaunal organisms.
Mesh size of the sieves employed for meiofaunal extraction are:

- 1000 μm as upper limit of meiofaunal size when also macrofauna is collected using mesh sives of 1 mm;
- 500 μm as upper limit of meiofaunal size when also macrofauna is sampled using a 0.5 mm mesh;
- 30 μm as lower limit of meiofaunal size. Some studies use mesh sieves of 45 μm, but the use of smaller mesh sizes is recommended in deep-sea studies, in which frequent and evident size miniaturization is observed.

Extraction of meiofauna in vivo

Meiofauna can be extracted from sandy sediments by decantation and after being anaestesised with a solution of 6% $MgCl_2$.

From sandy and muddy samples, meiofauna can be initially concentrated by means of mesh sieves: the larger (0.5-1 mm) is used to retain macrofauna and larger debris, and the small of 30 μm is used to concentrate meiobenthic organisms. Once concentrated, if the sample is "dirty"for the presence of a large number of detrital particles can be "cleaned" with di high density solution of glucose or using silica-gel (e.g., PERCOL; Schwinghamer et al. 1986).

Another extraction method is decantation after resuspension (see below) or using the phototactic response of meiofaunal organisms to a light source (in particular for nematodes and copepods). The sample of sediment and water, placed on a Petri dish, is exposed to a light source that recall organisms to the top of the sediment surface. Once reached sediment surface, meiofaunal organisms are withdrawn using a needle or by means of a Pasteur and placed in a Petri with water (Couch 1988, 1989).

Meiofauna extraction from muddy sediment fixed with formalin:

1. The whole sample is passed through of mesh of 0.5-1 mm, washed several times with water and collected in a 2 L becker.
2. The collected material is then filtered onto a mesh of 30 μm. The residual sediment is placed into a 50-ml Falcon and supplemented with LUDOX (HS 30, HS 40 or TM, arranging the density to 1.15-1.18 g cm^{-3}; i.e. equal or higher than those of meiofaunal organisms, but lower than sediment density). Falcons are then centrifuged per 10' at 3000 rpm (800 x g). The super natant is filtered on a 30 μm mesh net and then washed again with water. The pellet sediment is again supplemented with LUDOX and then centrifuged per 10' at 3000 rpm. This procedure is repeated 3 times to ensure an extraction efficiency of meiofaunal organisms >98%. The material collected on the top of the 30 μm mesh sieve is preserved in a falcon with 4% buffered formalin.
3. Once the extraction is completed the pellet sediment has to be checked under binocular to test extraction efficiency. A few drops of Bengal Rose (0.5 g L^{-1}) are added to stain meiofaunal organisms.

Extraction of meiofauna from sandy sediment fixed with formalin

The extraction of meiofauna from sandy sediments is based on decantation.

1. The whole sediment sample is filtered onto a mesh of 0.5-1 mm and collected into a large (2 L) becker.
2. The samples is mixed and resuspended in filtered water.

3. After a few seconds (to let the coarse sediment fraction to settle) the water is filtered on the 30 μm mesh sieve. This procedure is repeated 10 times to ensure an extraction efficiency of meiofaunal organisms of 98-100%. The material collected onto the filter is then fixed with 4% formalin and stained with Bengal Rose.

Sorting and identification of Meiofaunal Taxa

Once the extraction of organisms from the sediment is completed, it is necessary to proceed with the sorting in order to identify and count meiofaunal organisms.

Meiofaunal sorting is extremely time-consuming and laborious. It must be carried out only if we are interested in identifying components of the meiofauna which cannot routinely be identified in whole mounts, such as harpacticoid copepods. It may be useful however, to look at extracted samples, in water, under a binocular microscope with about 250x magnification, in order to check the state of preservation of animals, count individuals of major groups, or to carefully pick out larger pieces of detritus. A small Petri dish with lines scored on the bottom is ideal for this purpose. The use of Rose Bengal is recommended to facilitate the identification of the individuals. Usually, 15 minutes are sufficient to properly dye most meiofaunal organisms, however, for certain taxa bearing impermeable exoskeletons (Kinorhynchs, Halacarids) up to 2 days might be required for a perfect staining. The collection of coarse material can be carried out by means of stainless steel forceps, while tungsten needles mounted on wooden handles, or Pasteur pipettes with a thin capillary tip, may be useful to collect the smallest and most fragile organisms. Another practical tool is represented by an eyebrow hair glued on a needle by means of some nail varnish.

Generally, sediment samples contain large numbers of meiobenthic organisms, and it would be impossible to count and identify all of them. However, as meiobenthic assemblages are patchily distributed at small spatial scales, the collection and study of a small fraction is not correct. It is necessary to collect larger samples and then routinely identify a proportion, extracted at random from the whole sample, which we refer to as a subsample. As a general rule, a subsample containing at least 200 specimens is adequate for standard community analyses. It is helpful to keep the subsample size (defined as a fraction of the whole sample) constant within a particular study.

The simplest method to properly fractionate a meiofauna sample is to pour the whole sample into a plastic container, and add water until the total volume is equivalent to 1 litre; then agitate using a spoon with a circular movement for twenty seconds, so to resuspend the animals and collect one (or more) subsample using a ladle of known volume. Once the subsampling is completed, the remaining sample must be retransferred in formalin, and the removed volume noted on the container.

Sample examination and counting of individuals

These operations require the employment of a stereomicroscope at a minimum of ×25 magnification factor. A higher magnification factor may be necessary for the smallest meiobenthic organisms and larvae (×50, ×90). The enumeration of fixed and stained (Rose Bengal) specimens represents the most commonly used method for counting total meiofauna.

Sample examination and counting are eased by using cuvettes with internal subdivisions (e.g. 200 cells like in the Delfuss cuvette). Alternatively, Petri dishes with retinated bottoms may be used, but the lack of separating septa make the counting much more problematic compared with the use of Delfuss cuvettes. The identification to the specific level of various taxa requires the collection of individuals and the preparation of slides for the observation under the compound microscope.

Data Analysis

The abundance of meiofaunal assemblages is expressed as number of individuals per surface unit. Meiofauna display a typical patchy distribution and the analysis of the distribution of meiofaunal assemblages requires replicate samplings. Four to five replicates are needed to provide a reliable quantitative information for the sampled area. Abundance values are obtained as average of the values of each analyzed replicate, together with standard deviation.

Average ± standard deviation

Meiofaunal density is referred to a known sediment surface, generally 10 cm^2. In general, a sampling corer of 3.6 cm in diameter allows sampling a sufficient number of organisms and to have data normalized to 10 cm^2. if corers of a different diameter are used, it is necessary to normalize abundance values to the standard surface.

ind. 10 cm^{-2} = number of individuals $\times$ 10/S

where S = is the corer surface.

Generally, it is reported total meiofaunal density, the abundance of the main taxa and the total number of taxa, whereas other taxa are reported cumulatively as "others".

To define the structure and composition of meiofaunal assemblages the relative contribution of each taxon is calculated on total meiofaunal density.

Es: (n° nematodes/n° ind. of total meiofauna) $\times$ 100 = % nematodes

$\sum$ % each taxon = 100%

In general the percentages of the 7 more abundant taxa are reported, whereas the contribution of the other taxa is reported cumulatively as "Others".

The analysis of vertical distribution of total meiofaunal density and that of the main taxa is reported for each sediment layer analyzed (es. 0-1 cm; 1-2 cm; 2-5 cm; 5-10 cm; 10-15 cm), whereas if we refer to total meiofaunal density, the sum of meiofaunal densities in all sediment layers (i.e. 0-15 cm) is reported.

Here below is reported an example of data sheet to annotate sorting and counting of all meiofaunal taxa, together with abiotic environmental parameters:

Programme:
Station: Station depth (m):
Coordinates: **Depth of the sediment core (cm):**

N° Replicates: **Layer (cm):**

TAXA	Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5	Average
1. Nematoda						
2. Copepoda						
Nauplia						
3. Polychaeta						
4. Bivalvia						
5. Ostracoda						
6. Kinorhyncha						
7. Turbellaria						
8. Oligochaeta						
9. Tardigrada						
10. Gastrotricha						
11. Cumacea						
12. Amphipoda						
13. Isopoda						
14. Tanaidacea						
15. Acarina						
16. Nemertina						
Incertae sedis						
Others (*)						
Number of other taxa						
Total number of taxa						

Notes:

Determination of meiofaunal biomass

The determination of meiofaunal biomass is carried out in two different ways. The first, defined as 'gravimetric', allows the direct determination of the organism weight (after washing in distilled water, and dried at 60 °C per 24-48 h) through the use of micro analytic balance of adequate accuracy (±0.1 µg). This approach is generally utilized when a large number of organisms is available and pooled together (generally from 10 to 100-200) or when organisms are of large dimensions (always referring to meiofauna). The second method, defined as ‘volumetric’, is based on indirect estimates of meiofaunal biomass extrapolating organism dry weight from volumetric estimates. This method is applied associating a meiofaunal organism to a given geometric shape and this can be done for the majority of meiofaunal taxa which display a regular shape. Volume estimates (V) can be obtained from body length (L) and maximal length (W) at the microscope (for faster determinations it is possible to acquire the image and using a software for image elaboration). For instance, the biovolume of nematodes can be approximated using a cylinder using the following formula proposed by Andrassy (1956):

$$V = L\ W2\ 0.063 \times 10\text{-}5\ ;$$

where V is expressed in nl (10^{-9} l) whereas W and L are expressed in μm. Alternatively the formula proposed by Warwick and Price (1979) can be used:

$$V=C\ L\ W2$$

where V is expressed in nl, L and W in mm, while C is the conversion factor specific for each taxon.

This later formula derives from a procedure applicable also to organisms having an irregular shape, such as copepods. Models are constructed, in scale, based on shapes drawn at the microscope or based on photos taken from different perspectives in order to respect the body dimension (L and W max). The volume is determined drawing the model in a graduated cylinder and measuring water displacement. Reporting this volume with the one of the parallelepiped of length (L) and base (W^2) identical to those of the model it is possible to estimate the C conversion factor (Table 2.3.6.5; Warwick and Gee 1984)

Table 2.3.6.5: list of the C conversion factors:

Taxon	C
Nematodes	530
Ostracods	450
Kinorynchs	295
Turbellarians	550
Tardigrades	614
Idroids	385
Polychaetes	530
Oligochaetes	530
Tanaidaceans	400
Isopods	230
Copepods	(*)

(*) For copepods the factor C varies depending on the shape of the animal, as suggested by Warwick and Gee (1984), from a minimum of 230 for “discoid” forms (“scutelliforms”) to a maximum of 750 for “cylindrical” forms.

In general, the volume can be approximately converted into dry weight assuming a specific gravity of 1.13 (Wieser 1960). For nematodes, dry weight is generally 20-25% of the wet weight (Myers 1967, Wieser 1960). For further conversions it is useful to consider that C content of the biomass of meiobenthic organisms is ca 40% of its dry weight, while organic N is ca 10% of dry weight (Feller and Warwick 1988). The average calorific equivalent of meiofaunal biomass can be estimated through the equation: 1g dry weight = 5.26 Kcal.

2.3.6.7 Monitoring zooplankton

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Introduction

Zooplankton – types and size

Plankton is by definition ”drifters” that float more or less passively with the ocean currents, as opposed to nekton which is fish and other organisms that swim and position themselves more or less independent of currents. ***Zooplankton*** is mostly small animals but includes also large forms (such as medusa and colonial siphonophores) which may be a meter or more in size. They represent a wide range of taxonomic groups including crustaceans, mollusks, cnidarians, ctenophores, polychaetes, chaetognaths, rotatorians, tunicates, and others (Lenz 2000). Crustaceans are typically a dominant component of the zooplankton, represented by numerous species of copepods, ostracods, amphipods, euphausids (krill), and others. Cnidarians and ctenophores are important components of the so-called gelatinous zooplankton which are forms with high water content. They range in size from small (mm) hydromedusae to large scyphomedusae (m).

Many zooplankton live their whole lives in the water column and are true plankton, or *holoplankton*. Others live only part of the life as plankton and are called *meroplankton*. This includes eggs and larvae of a wide range of benthic invertebrates such as echinoderms, polychaetes, and crustaceans, and also of many fish. Meroplankton has often a strong seasonal pattern, with most forms present during spring and summer which is when most species reproduce. Holoplankton also show large seasonal variation related to the cycle of production of their main food base which is phytoplankton. Many zooplankton also reproduce in spring and summer with one to several generations during the productive season.

Zooplankton span a size range over about 6 orders of magnitude in linear dimension, from about 2 um for small unicellular, heterotrophic flagellates to about 2 m for very large medusa. It is common to denote fractions of the zooplankton according to individual size as: *nano*- (2-20 um), *micro*- (20-200 um), *meso*- (200 um-20 mm), *macro*- (2-20 cm), and *mega*-plankton (20 cm-2 m) (Lenz 2000). The smaller fractions (particularly nano) overlap to large extent the size distribution of unicellular or colonial phytoplankton (e.g. chain-forming diatoms). The “traditional” zooplankton is in the meso size range. This includes copepods (1-10 mm in length) that are often a dominant component of zooplankton both in terms of numbers and biomass. Macrozooplankton includes forms such as krill and amphipods which are crustaceans that can be a few cm long.

There is no sharp distinction between plankton and nekton but rather a wide and fuzzy boundary. Thus many of the macro-zooplankton like krill may have considerable swimming ability and are able to aggregate to form swarms and also perhaps to position themselves in relation to the flow fields of ocean currents. Smaller forms such as copepods in the meso-zooplankton can swim and position themselves vertically in the water column. In this way they can get into layers with different currents and through this influence their horizontal distribution. Larger forms such as krill may vertically migrate several hundred meters per day. Feeding usually takes part in the upper layers at night. At day time most forms are positioned in deeper layers to avoid predation.

What to measure?

There are a number of properties or attributes of zooplankton that can be characterized and quantified in a monitoring program. These include:

- Biomass
- Abundance
- Distribution
- Species or community composition
- Size composition
- Demographic parameters and rate measurements

- Production

These properties or attributes can be obtained in two principally different ways:
1) Collecting the zooplankton by filtering them from the water using a towed net (or pumping water through a stationary net), or
2) Recording the organisms in situ by visual (photography, video) or acoustic techniques.

Biomass can be determined in zooplankton samples obtained with nets and measured with various techniques as volume, wet weight (or wet mass), dry weight (or dry mass), ash-free dry weight, caloric content, and elemental weight (e.g. as carbon and/or nitrogen) (Postel et al. 2000). Biomass can be determined for individual animals, sorted fractions (species and groups), size fractions, and total sample. Results can be expressed per volume of water (m^{-3}) or per m^2 for the depth interval from which the sample was obtained.

Abundance can be determined by counting the individuals according to taxonomicgroups, species, stage, size category, etc. As for biomass the results can be expressed as number of individuals m^{-3} or m^{-2}.Abundance can be determined in samples collected with nets and can also be recorded in situ with optical and/or acoustic techniques (Foote 2000, Foote and Stanton 2000). In situ recorders include the OPC (Optical Plankton Counter) which records the size of particles as they pass a beam of light, and the VPR (Video Plankton Recorder) which records the organisms with video, allowing identification of species and stage and/or size in many cases. Abundance data can be used to estimate biomass by using recorded or average size by species and stage or size groups.

Distribution has at least 3 aspects: horizontal, vertical, and patchiness. Horizontal distribution is usually the broader-scale pattern of zooplankton in relation to currents, water masses, and topography (e.g. basin vs. shelf). Vertical distribution is the pattern of zooplankton versus depth, where some species are found in the surface layer (epi-pelagic), other at the bottom of the euphotic zone, and others again deeper down in the semi-dark or dark part of the water column (bathypelagic). Zooplankton species are usually found aggregated in patches at various scales resulting from small-scale physics (e.g. Langmuir cells) or behavior (swarming) or a combination of the two mechanisms. Vertical tows with simple nets may be an effective way of determining broad-scale horizontal distribution. Depth-stratified samplers (such as MOCNESS or Multi-Net) may be effective in determining both horizontal and vertical distribution patterns. Optical in situ recorders may provide detailed information on fine-scale patchiness, while acoustic techniques have their greatest strength in providing similar information at somewhat larger scale for macrozooplankton such as krill. Distributional changes are likely with climate variations, e.g. northward shifts are to be expected in most areas as a result of warmer climate (e.g. Beaugrand et al. 2002).

Species or community composition is an attribute that provides much information. The focus may be on the ecologically dominant or key species of zooplankton (such as *Calanus* species in arctic, sub-arctic and boreal waters), on rarer species or indicator species that may reveal changes in currents and distribution patterns, or on the overall species composition as reflected in diversity, species richness or other parameters or metrics for community composition. Analyses of samples collected with nets provide the most accurate determination of taxonomic composition. In situ recorders based on visual observation (video or photography) may also provide useful information on taxonomic composition, particularly if ground-truthed with simultaneous net sampling. Determination of zooplankton abundance may be a component part of determination of species or community composition.

Size composition of zooplankton assemblages or communities reflects the species or community composition. Data on the latter may be used to describe the size composition both in terms of abundance or estimated biomass in size classes. Size distribution can be obtained for biomass by using size-fractionation (screening) of collected samples, and also for abundance and estimated biomass by in situ recording of zooplankton organisms by optical and acoustic techniques. Size measurements are useful to differentiate species, e.g. *Calanus* copepods which otherwise are very similar morphologically (e.g. Kwasniewski et al. 2003). Size cohorts can also be used to segregate age classes of zooplankton (Dalapadado and Skjoldal 1996).

Demographic parameters such as rates of recruitment, growth and mortality can be derived from repeated sampling over time from a zooplankton community based on changes in abundance of stage, size or age groups of zooplankton species (classical population dynamical analysis). There are many inherent difficulties in this, including advective changes in communities (by not sampling the same water body), and non-representative sampling (bias over the range of sizes or stages). There are a number of ***rate measurements*** of metabolic and/or ecological processes that can be carried out to derive estimates of ecological functions and which may be used along with estimated demographic parameters. *Feeding rates* can be determined from analysis of stomach contents of collected animals. Feeding rates can also be determined experimentally (e.g. as a function of concentration of food) by incubating zooplankton that have been collected live. *Metabolic rates* e.g. of respiration and excretion can be determined in a similar way. There are a number of biochemical techniques (e.g. enzymes, RNA, immuno-fluorescence) that also can be used to derive estimates of rates based on measurements of collected samples of zooplankton. See Båmstedt et al. (2000), Runge and Roff (2000), and Ikeda et al. (2000) for reviews of methods for determining feeding, growth, reproduction, and metabolism.

Production of zooplankton is an important feature in trophic, food-web and ecosystem contexts. Production can be estimated from biomass using P/B ratios (e.g. Banse and Mosher 1980, Skjoldal et al. 2004), or from abundance data using population dynamical calculations (Runge and Roff 2000). Production can also be estimated from determination of rates of egg production, growth and moult (of copepods and other crustaceans) based on incubation or biochemical techniques (Runge and Roff 2000).

ICES Zooplankton Methodology Manual

The International Council for Exploration of the Sea (ICES) produced in 2000 a comprehensive methodology manual that provides overviews, reviews and recommendations of all major aspects of sampling and analyses of zooplankton (Harris et al. 2000). Reference to chapters in this book has been included in the brief overview in the preceding subsection. The Zooplankton Methodology Manual was prepared by the ICES Working Group on Zooplankton Ecology (WGZE) over a time period of about 5 years in the 1990s.

As part of the preparatory work for the manual, the ICES WGZE convened several workshops on methods comparisons. One of them was a sea-going workshop with two ships (RV Johan Hjort and RV A. V. Humboldt) to compare zooplankton sampling systems in a Norwegian fjord (the Storfjord in the Møre and Romsdal County). Results from this workshop were presented in a report in 2002 including 4 CD-ROMs containing all raw data (Wiebe et al. 2002; http://www.ices.dk/pubs/crr/crr250/CRR250.PDF). A manuscript is now being finalized for publication (H.R. Skjoldal, P. Wiebe, L. Postel, T. Knutsen), and some main conclusions from this work regarding sampling and biomass determination are included in a subsequent section.

The ICES WGZE is still active and considers various aspects of zooplankton methodology and ecology that are reported in annual reports available from ICES (their latest report from 2009 can be found here: http://www.ices.dk/products/CMdocs/CM-2009/OCC/WGZE09.pdf). As part of their work they produce an annual status report on zooplankton in the North Atlantic (http://www.ices.dk/pubs/crr/crr292/ICES292-SCREEN.pdf). Their latest product is a report from a workshop to compare zooplankton ecology and methodologies between the Mediterranean and the North Atlantic (WKZEM) (http://www.ices.dk/pubs/crr/crr300/CRR%20300-Final-web.pdf).

Sampling design

Zooplankton is a very diverse component of marine ecosystems and the attributes that can be measured in a monitoring program are also wide-ranging, as outlined in the Introduction. The purpose of the monitoring needs therefore to be clearly defined before any program is designed. The purpose of monitoring may be any (or a combination) of the following, illustrative and non-exhaustive, list:

1) Determination of the horizontal distribution of zooplankton biomass and abundance within a large marine ecosystem, e.g. as a characterization of the grazing field for plankton-feeding pelagic fish (such as herring, anchovy, mackerel).

2) Determination of the seasonal pattern of occurrence and vertical distribution of zooplankton biomass and abundance.
3) Determination of the population dynamics of one or more key species of zooplankton within the whole or part of a large marine ecosystem.
4) Determination of the long-term changes in species and community composition in relation to climate and ecosystem variability and change.
5) Determination of the zooplankton grazing impact on phytoplankton and documenting its (changing) role in ecosystem regulation and pelagic-benthic coupling.

It is usually desirable and necessary to combine approaches and techniques for monitoring zooplankton (Skjoldal et al. 2000). If a wide spectrum of zooplankton is to be included (e.g. micro-, meso- and macro-zooplankton), a combination of sampling gears must be applied. This may be different nets with fine and coarse mesh to collect small and large zooplankton, and also combinations of net sampling and optical and acoustic techniques. It may include a combination of infrequent broad-scale horizontal surveys and frequent sampling and observations to determine seasonal patterns and temporal variability at selected locations. It may also include a combination of approaches such as modeling (e.g. 3D circulation and transport of zooplankton), experimental rate measurements, and field sampling. Some further design considerations were presented by Skjoldal et al. (2000).

Two cases of sampling design can be mentioned as examples to illustrate the general issue. One is the monitoring by IMR in collaboration with the Russian PINRO of zooplankton in the Barents Sea ecosystem that covers an area of >1 million km^2 (Stiansen et al. 2009). A broad scale survey is carried out in the autumn (August-October) when open water has the maximum extent and much of the northern, arctic part of the Barents Sea is available for sampling. Zooplankton is sampled with vertical net hauls (WP-2 net) in two depth intervals (upper 100 m or deeper than 100 m). Sampling is also done with a MOCNESS which is a depth-stratified zooplankton sampler with opening and closing nets, operated over the water column. The zooplankton biomass (as dry weight) in three size fractions (<1, 1-2, and > 2 mm screen mesh opening) is determined and showed as horizontal maps (Fig. 2.3.6.6). The biomass values are also integrated over large areas and shown as time series (Fig. 2.3.6.7).

The broad-scale mapping of the horizontal and vertical distribution of zooplankton in autumn is supplemented by repeated sampling along oceanographic transects that run north from the Norwegian coast through the southern and central parts of the Barents Sea. This provides information on the seasonal development of plankton (including phytoplankton as reflected in chlorophyll and nutrients).

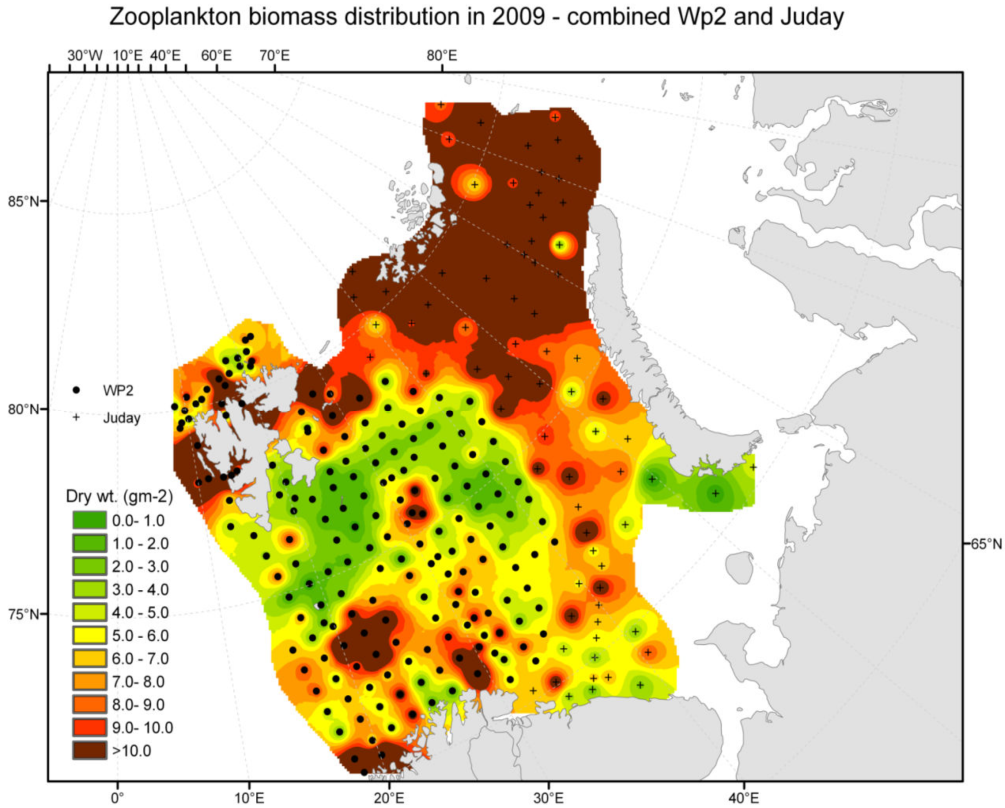

Figure 2.3.6.6: Horizontal distribution of zooplankton dry weight biomass in the Barents Sea. Results obtained with vertical net hauls over the water column with WP2 and Juday nets in joint cruises in the autumn 2009 by IMR Norway and PINRO Russia. This is an updated version for 2009 of results presented by Stiansen et al. (2009).

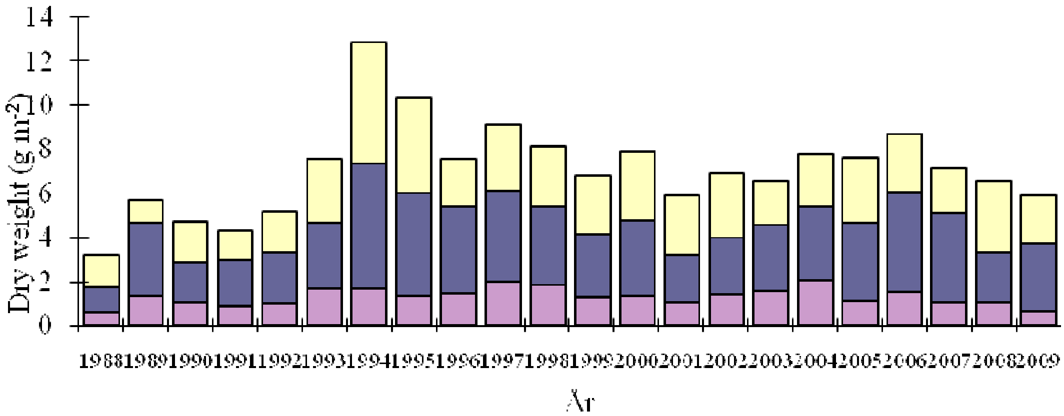

Figure 2.3.6.7: Time series of dry weight biomass of zooplankton in three size fractions (<1, 1-2, and >2 mm – pink, purple, yellow) depth-integrated over the water column and averaged over the surveyed area of the Barents Sea in autumn from 1988 to 2009. Results from joint surveys by IMR Norway and PINRO Russia (see Stiansen et al. 2009).

Horizontal and vertical distribution of zooplankton biomass provide information on the food and feeding conditions for pelagic plankton-feeding fishes, which in the Barents Sea are young herring (*Clupea harengus*), capelin (*Mallotus villosus*) and polar cod (*Boreogadus saida*). Inverse relationship between zooplankton biomass and stock size of capelin has been demonstrated, suggesting top-down influence of the fish back on their own feeding environment (Dalpadado et al. 2002). The zooplankton samples are routinely split in two halves (see Fig. 2.3.6.8), one for determination of biomass and the other for taxonomic analysis. Analysis of macrozooplankton collected with the MOCNESS sampler has revealed pronounced changes in abundance and size and age composition of krill and amphipod species, reflecting changes in predation from a fluctuating capelin stock (Dalpadado and Skjoldal 1996, Dalpadado et al. 2001, 2002).

The second example of a monitoring program of zooplankton is the CPR (Continuous Plankton Recorder) survey operated by the Sir Alistair Hardy Foundation in the UK. In this survey, CPRs are towed by commercial vessels on routes between ports in European seas and other places globally. CPR is a high-speed sampler with a 1 cm^2 opening and the samples are collected on gauze which is moved in steps (equivalent to a given towed distance) across the cod end of the zooplankton net within the encased CPR box. The CPR is typically towed at 10 m depth and basically samples a 1 cm thick line across the distance of the route. While this is obviously a limitation if one considers the wide and patterned vertical distribution of zooplankton species, the CPR time series have offered very important documentation of shifts in abundance and distribution of zooplankton in relation to oceanographic shifts and climate change (Beaugrand et al. 2002, Beaugrand 2003, Reid et al. 2003).

Zooplankton sampling systems

Types of samplers

Zooplankton has been sampled using towed nets since the 1880s. A variety of nets have been designed and used from the fist simple net made by Victor Hensen in 1887. There have been two main lines of development of nets: 1) samplers designed to be towed fast, and 2) samplers with larger openings and often with opening and closing nets. A thorough account of the history of zooplankton nets and other sampling systems was given by Wiebe and Benfield (2003). An overview of samplers and their operation is also given by Sameoto et al. (2000).

Zooplankton nets are towed vertically, obliquely or horizontally and may be simple nets, multiple nets, and opening and closing nets. Some commonly used nets are mentioned in the following (see Sameoto et al. 2000, for more details on these and other samplers).

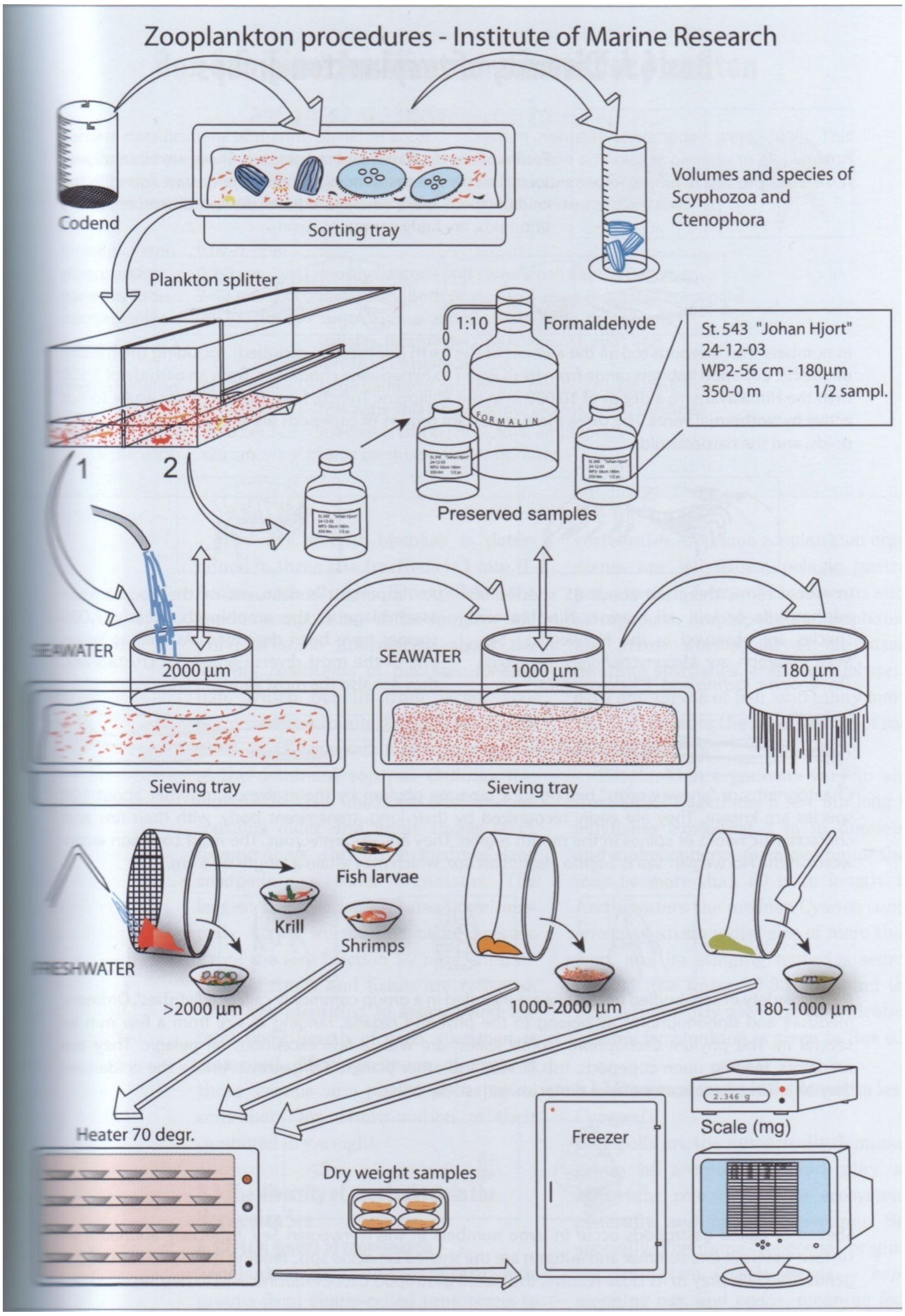

Figure 2.3.6.8: The standard procedure used at IMR in Norway for analyses of dry weight biomass and taxonomic composition of zooplankton samples. See text for further details. From Melle et al. (2004).

The ***WP-2 net*** is a simple net which is towed vertically. It can be operated with a closing device so that samples may be taken from a given depth interval, e.g. 500-100 m. The WP-2 net is a standard net designed and recommended by a group of experts established by UNESCO in the 1960s (the name WP-2 stands for Working Party 2). ***MOCNESS*** is a multiple, opening and closing net sampling system which is generally towed obliquely, which each net sampling a defined depth interval (e.g. 250-300 m). The nets have usually an opening of 1 m^2 and there are 8 of them in the sampler. ***MultiNet*** is a similar type of sampler that can be operated both with oblique and vertical tows. It comes in various sizes and is often used with 0.25 m^2 opening and 5 separate nets that can be opened and closed in different depth intervals. ***Gulf III*** is a high speed sampler with one or a set of rotating nets within a hydrodynamic casing and a nose-cone to reduce the flow of water into the net. ***LHPR*** (Longhurst Hardy Plankton Sampler) is another high speed sampler with nose-cone and a multiple cod-end system that enables samples to taken from depth intervals with high resolution (many discrete samples) as the sampler is towed obliquely through the water column.

Main sources of variability and errors

Zooplankton is often distributed fairly uniformly in water masses over large areas although there are usually pronounced vertical zonation patterns and changes associated with oceanographic structures such as frontal systems. There is also usually a high degree of patchiness in distribution on finer scales. This can be related to physical features associated with winds, waves and currents or to biological phenomena such as swarming behavior, or a combination of physical and biological factors.

Patchiness induces variance in zooplankton sampling. The extent of this depends on the scale of patchiness in relation to the scale of sampling. Small-scale patchiness (e.g. swarms at the scale of meters) can be averaged out by long net tows that integrate over many such structures. Conversely, large-scale patchiness may not be revealed with a low number of short tows that have a low probability of hitting infrequent large structures. Often the variance associated with zooplankton sampling is such that the coefficient of variation (SD/mean) for replicate samples is close to one. Larger samplers such as MOCNESS towed over the column has often less variance for the estimated depth-integrated zooplankton biomass, probably due to a “smoothing” or averaging effect on smaller scale patchiness (including vertical shifts in distribution within the water column).

Besides patchiness which induces variance, there are two major sources of error associated with zooplankton sampling. These are escapement or extrusion of small zooplankton forms though the mesh of the net and avoidance of larger and more agile forms that can swim or jump away from the path of the approaching net (Sameoto et al. 2000).

Small organisms pass through the mesh of zooplankton nets for two reasons. They are either too small to be retained by the net, or they are small enough to be squeezed through the net. The first is a more passive loss by escapement due to small size and the latter is more active extrusion though the mesh. ***Extrusion*** is a function of the shape and flexibility of the zooplankton organisms and of the speed of passage of water through the net. Generally, higher towing speed leads to more extrusion. This is usually countered by have a nose-cone that reduces the opening and a long net that provides a large filtering area in high-speed samplers. The body width is usually the dimension that determines extrusion rather than the body length. As a rule of thumb, zooplankton typically passes through mesh with size (rectangular) about 70 % of their own body width. This means that a zooplankton (e.g. copepod) with a width of 0.3 mm may pass through a mesh of 200 um. The cut-off is usually not very sharp but gradual from 100% loss of organisms (of a given type) smaller than a given minimum size to 0% loss (100% retention) of organisms larger than a given size (Skjoldal et al. 2010/In prep.).

Avoidance can be pronounced for large forms such as krill (Barkley 1972, Sameoto et al. 2000). They can see or otherwise sense the approaching net and avoid being caught by moving to the side. High speed has been one of the methods to overcome avoidance, catching the mobile zooplankton by surprise. It is uncertain how effective this strategy is since the plankton may still sense and get away from the usually narrow opening of high-speed samplers. Another approach has been to increase the opening of the net to make it more difficult for the zooplankton to swim out of the path of the approaching net. Again the effectiveness is uncertain since the larger net may be visible or otherwise sensible at a longer distance, allowing a longer response time for the zooplankton trying to escape. The color of the net and frame should be one that makes the sampler less visible in the water and thus less difficult to see and detect by e.g. krill which apparently rely on the visual sense to avoid being caught by an approaching net. Use of a flashing bright light which apparently blinds the organisms, has been found to increase the capture of krill (Wiebe et al. 1982, Sameoto et al. 1993). Usually avoidance is less for sampling in darkness at night.

The density of zooplankton as numbers of individuals per volume of water is one factor that influences sampling results. The dominant meso-zooplankton species such as the copepod *Calanus finmarchicus* in boreal waters are often present with several hundred individuals m^{-3}. A net haul with a 0.25 m^2 WP-2 net through the upper 100 m will therefore collect many thousand individuals. However, some of the larger macro-zooplankton such as krill may be present with 1 individual per 10 m^3 or one per 100 m^3 or even less. At such densities the likelihood to catch one of them would be small with small nets that filter a small volume of water. Even larger nets such as MOCNESS may have low probability of catching some of the larger forms due to their low densities, independent of the avoidance problem.

Gear intercomparison

The ICES sea-going workshop on comparison of zooplankton sampling systems showed that different nets including WP-2, MOCNESS and LHPR produced similar estimates of zooplankton biomass when operated with comparable mesh-sized nets (180-200 µm) (Wiebe et al. 2003, Skjoldal et al. 2010/in prep.). This suggests that data for mesozooplankton biomass can be broadly compared across studies and geographical areas if similar mesh size is used even though the gear differ. The mesh size on the other hand had a major influence on the result, and a 400 um mesh net collected only about half the amount of biomass compared to a 200 um net in the conditions of the workshop (neritic zooplankton community). Towing speed also was found to be important, with loss of small and medium-sized mesozooplankton through the mesh with faster towing speed. Higher towing speed on the other hand led to better catch of macrozooplankton such as krill and shrimps.

Analyses of zooplankton samples

A review of methods and methodological aspects for analyses of zooplankton samples collected with nets was given by Postel et al. (2000). It is common to determine biomass and/or taxonomic composition of samples. There are a variety of methods in use in different laboratories. There are no standard methods but Postel et al. (2000) give some recommendations that will contribute to harmonization and better comparability of results.

We will not repeat a review of methods here and refer to Postel et al. (2000) for general considerations on the topic. Instead we present the procedures used at IMR in Norway as an example of methods used in routine monitoring of zooplankton in large marine ecosystems. The overall procedure of sample handling and analyses is shown in Fig. 2.3.6.8.

The samples are rinsed from the net and collected in the cod-end. The total sample content is transferred to a splitter (Motoda plankton splitter) and divided in two halves: one for biomass determination and the other for taxonomic analysis and species enumeration. The sample half for biomass is screened successively through three meshes: 2 mm, 1 mm, and 180 um. The content on each screen is briefly rinsed with freshwater (to remove salt) and transferred to pre-weighed Aluminum trays. They are then dried at 60°C for a minimum of 24 h (to achieve constant dry weight), stored frozen in dessicator, and later weighed to obtain dry weight biomass. By taking into account the volume of water filtered in the net and the sampling depth interval, the results are expressed as dry weight biomass per m3 of seawater, or m2 of water column, for the <1 mm, 1-2 mm, and >2 mm size fractions. Large forms (macrozooplankton) such as krill, shrimps, amphipods, mesopelagic fish and fish larvae are removed from the sample before fractionation and are identified, length measured and weighed individually or as group to determine their biomass (and abundance) separately.

The sample half for taxonomic analysis is fixed with formalin and stored for later analysis. Before they are examined under the binocular (at 10-50 X magnification), the samples are rinsed with seawater. Subsampling is used in the counting procedure. Large and scarce forms are counted in the whole sample (half fraction of the total). Small forms are counted in subsamples dependent on the abundance and sample size. For MOCNESS samples that typically have filtered around 300 m3 of water, small copepods such as *Oithona* or early copepodites of *Calanus* are often counted in subsamples representing from 1/16 to 1/64 of the sample. Most forms are identified to species, and for copepods also to stage and sex for common species. Copepodite stages (CI - CVI) and *Calanus* species are determined based on prosome length (Kwasniewski et al. 2003) and other characteristics. Some forms such as chaetognaths are counted in size classes. Many invertebrate larvae of the meroplankton (e.g. polychaetes, echinoderms) are counted by groups and type of larvae.

Sample handling and subsampling generate error and variability in the results. However, the overall procedure has been found to be fairly robust and reproducible. It was used as the basic procedure in the sea-going workshop for gear intercomparisons referred to earlier. Determination of dry weight biomass was found to be associated with coefficient of variation (CV = SD/mean) of about 20% when there were little inherent patchiness at the scale of sampling. This includes all sources of variance from in situ sampling, sample treatment (rinsing of net, splitting, size fractionation), and weighing (Wiebe et al. 2003, Skjoldal et al. 2010/in prep.).

Biomass measured as dry weight has advantages over biomass as wet weight or displacement volume. Wet weight to dry weight conversion factor is typically about5 for copepods and many other mesozooplankton (Postel et al. 2000, Skjoldal et. al. 2004). Many gelatinous forms have high water content which may contribute to wet weight biomass. The comparability across samples from seasons and areas are therefore less than for dry weight biomass. The high water content of gelatinous forms leaves salt when dried. This may result in an increased dry weight compared to samples consisting of forms with lower water content. Ash-free dry weight is therefore a better measure of the organic biomass since salt is included with the ash after combustion of the samples at high temperatures. At IMR we ashed all the samples and used ash-free dry weight as the biomass measure for some years in the late 1980s in quite comprehensive sampling for both monitoring and research purposes. We found that dry weight and ash-free dry weight usually corresponded well, reflecting a rather constant ash content. The main exception was samples from the upper layer in late summer when biomass tended to be low and there were relatively more gelatinous forms. However, since determination of ash-free dry weight represented a considerable amount of additional work (combustion in a special oven, cooling in dessicator, and reweighing of samples), we decided to use dry weight on a routine basis as this was relatively precise as a measure of biomass and less expensive in terms of labor costs.

Subsampling generates inaccuracy in determination of species abundance and taxonomic composition of samples. There is a general relationship between the accuracy of the determination of the abundance of a taxon e.g. species on the one hand, and the density (abundance) of the species in situ (reflected in the total number of individuals collected in a tow) and the number of individuals of the species counted (reflecting subsampling and the fraction of the total sample counted), on the other. The fewer individuals counted and the lower fraction it represents of the total, the less accurate will the result be. This is a predictable relationship and the error (low accuracy) associated with low counts can be estimated (Cassie 1971, Skjoldal et al. 2010/in prep.).

2.3.6.8 Shellfish stock assessment

Contributor: Jeroen Jansen (IMARES)

Introduction

Shellfish fisheries and ecological status studies require insight in the size and development of commercial or ecologically relevant shellfish populations. In the Netherlands, shellfish stock assessments started for Blue Mussels (*Mytilus edulis*), Common Cockles (*Cerastoderma edule*) and Triangle Shells (*Spisula subtruncata*). These were three commercially interesting bivalve species and annual stock assessments were necessary for the Ministry to decide on fishery permits (Kesteloo et al. 2009; Goudswaard et al. 2009). After the collapse of the *Spisula subtruncata* population in the Dutch coastal zone in 2003 it was observed that the population of Razorshell Clams (*Ensis directus*) was growing exponentially (See for instance Goudswaard et al. 2009b). Several permits were given and the shellfish monitoring program in the Dutch coastal zone continued with a focus on Ensis directus. Together with Blue Mussels, Common Cockles and Razorshell Clams, all other bivalves that were sampled along were measured and registered as well. Some of these, such as Soft Shelled Clam (*Mya arenaria*), the Baltic Telling (*Macoma balthica*), the Common Otter Shell (*Lutraria lutraria*) or the Pacific Oyster (*Crassostrea gigas*) are of ecological interest for their role and importance in the ecosystem.

Ecology

The species mentioned in the preceding paragraph are filter feeding bivalve species, filtering suspended food particles, mainly algae, from the water column. In nutrient-rich Dutch coastal waters there is plenty of food for large populations to develop. *Macoma balthica* is facultative deposit feeder, therefore occurring in less dens aggregation as the other species do. All these species are broadcast spawners. In the course of spring and early summer they release millions of gametes into the water column. After fertilization larvae develop that have the ability to drift with water currents for several weeks after which settlement takes place. If settlement is successful, large seedbeds come into existence that are usually visible by the end of summer and for some species in autumn.
Most shellfish and filter feeders in particular, occur in relatively dense aggregations. These aggregations start as densely packed seedbeds that vary in size from a couple of square meters to several, sometimes hundreds of acres. New seedbeds are though of as vulnerable structures. In addition to huge density-dependent mortality, there are many factors that threaten the survival of seedbeds, e.g. predation, changes in water current velocity, oxygen depletion, sever winter conditions, etc. As such, seedbeds may disappear again within months or become patchy, irregularly distributed clusters of individuals. The structure of shellfish aggregations is an important consideration when planning a sampling campaign.

Sampling strategy

Stratified sampling

To estimate the population size of a shellfish species in a certain area with relatively high accuracy, stratified sampling is advisable.Stratified sampling means that the area of interest is subdivided into two or more sub-areas that differ (presumable) in the abundance of the species of interest. The idea about the sub-areas is that the errors can be calculated for the corresponding sub-stocks and then be summed up. This results in narrower confidence bands than when no sub-areas are analyzed. Usually, a higher density of sampling stations is placed in the area where the species is expected, and less where it is absent. In this way, the total stock will be assessed more accurately.

Stratified sampling can only be conducted when information is available on the distribution of shellfish beds and reefs. In many monitoring programs such information is limited to observations in previous years. Unexpected sampling of individuals from the (presumed) empty subarea (see preceding paragraph) will introduce large errors in the final analysis, especially when samples account for relatively large area's. When this occurs, adaptive sampling or a stock assessment with and without this particular sampling station may be sensible.In some cases, knowledge on the distribution of beds and reefs is available. For the commercial bivalve species, this information may come from the fishery industry that usually carries out inspections in order to plan their fisheries.

A special case is the assessment of littoral Mussel beds and Oyster reefs. The Wadden Sea is a shallow sea between the Dutch mainland and its Islands. It is characterized by extensive intertidal sand and mud flats that are abundant in beds and reefs of several shellfish species. Mussel beds and Oyster reefscan be seen from anairplane. After an inspection flight, littoral beds and reefs can be visited by foot and their contours can be mapped using a handheld GPS. Subsequently, the area can be visited by a research vessel during high tide to take samples at prefixed stations/locations.

New techniques

As an alternative for GPS track contours, several attempts have been made to draw contours from geo-referenced aerial photographs using Ecognition software (Fig. 2.3.6.9). So far, the software program recognized about 60% of the mussel beds. About 65 % of the beds recognized by the program are judged as “unlikely to be a mussel bed” by human eye.

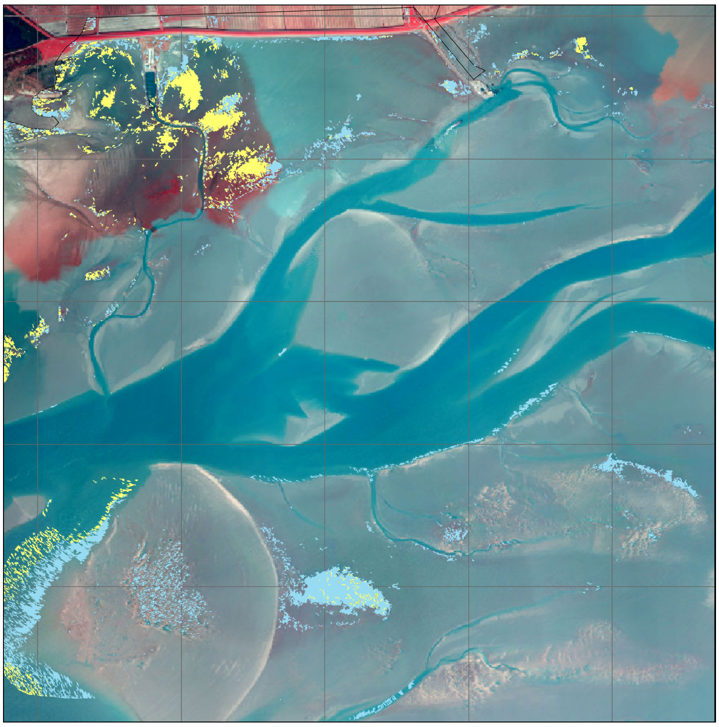

Figure 2.3.6.9: Aerial photograph of area near the Island Schiermonnikoog with several mussel beds. Mussel occurrences recognized by computerized analysis are colored blue (light colored mussel bed) or yellow (dark mussel bed)

For the subtidal areas Sidescan Sonar imaging is believed to be a powerful tool to estimate the contour of reef-like structures (Fig. 2.3.6.10). For sonar imaging, however, it is even more important to have some idea of the location of the sublitteral reefs and beds and ground truth is required to tell patches of bivalves from other structures on the sea floor.

When there is no information on the distribution of shellfish beds available, habitat modeling can be a suitable tool to define different strata, i.e. to reduce the area where the species of interest is expected. In this way a map can be drawn on which the possible occurrence of the species is categorized (See for instance Kater et al. 2003) and sampling stations can be chosen for each category. When the stations are distributed randomly per categorized area, this approach is particularly suitable for spatial interpolation techniques as Kriging (See for instance Bowling 2005).

Timing of a survey

Stock assessments that are used for fishery permits are carried out in spring. This is done for two reasons. First, the results of the survey have to be available in time for policy makers to decide on permit requests. Secondly, the assessment should not include newly formed seedbeds, because their faith is rather unpredictable. Seedbeds that survived their first winter will develop or decay less stochastically.

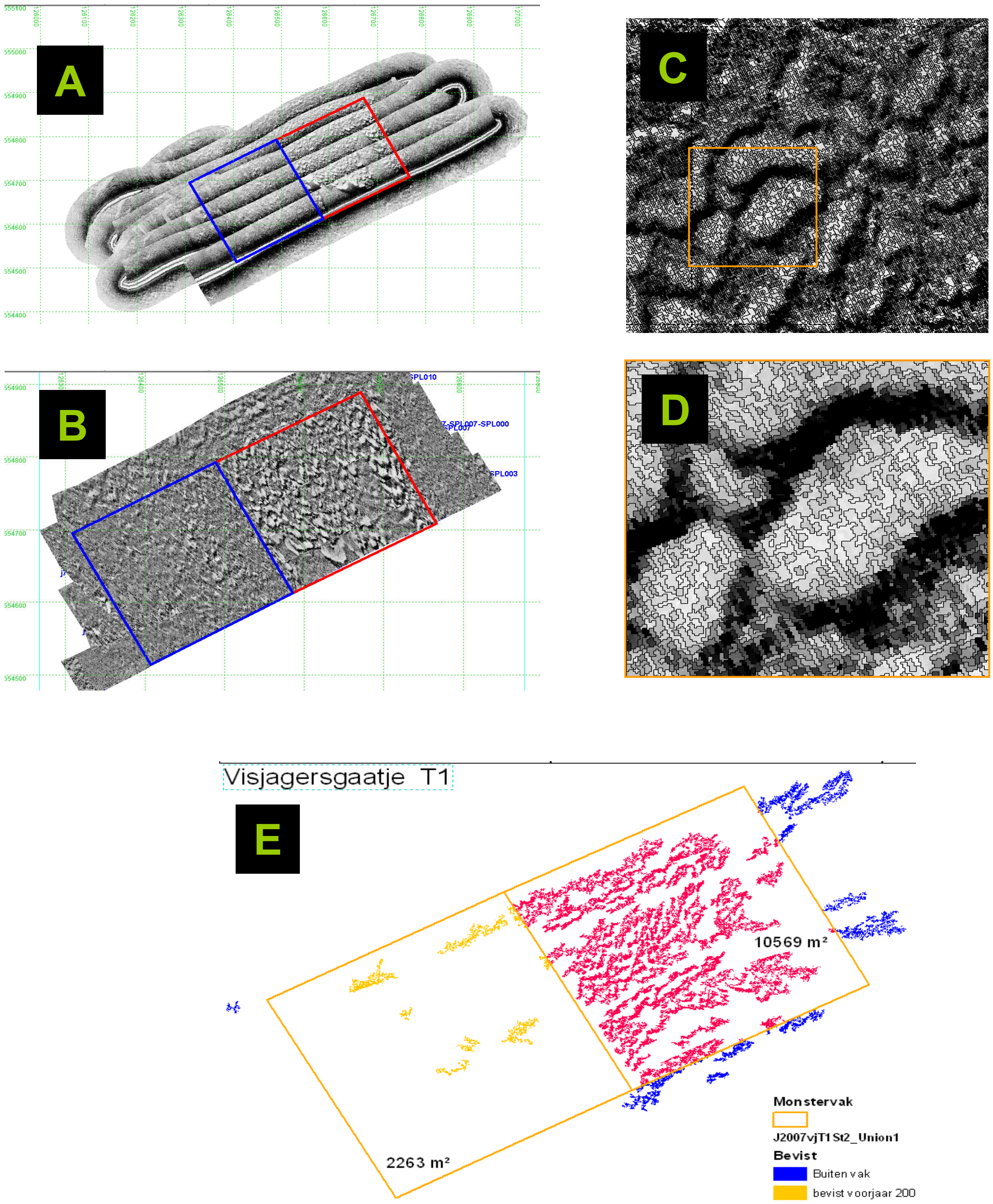

Figure 2.3.6.10: Analysis of area covered with Blue mussels from a sidescan sonar image. A shows elongated sonar images that result from parallel tracks that together cover the area of interest. In this particular case, the blue and red box represent experimental plots that were “open” and “closed” for mussel seed fisheries, respectively. In a computer the parallel sonar images are stitched together (B) and corrected for several factors, e.g. drift, cruising direction, etc. The result is further analyzed with Ecognition software that aims to identify mussel ridges (C and D for close up). Base on the analysis the program will deliver the area covered with structure/mussels and where it is situated (E).

Sampling gear

Integrating spatial variation

As suggested above, shellfish beds and reefs should not be imagined as homogeneous structures. Even when their position is known point-sampling with grab-like techniques (read Van Veen, Boxcorer, etc.) should be avoided when possible. Dredge samples are preferable because they integrate the spatial variation at a 10-100 meter scale. Both regular dredges and hydraulic dredges can be used (Fig. 2.3.6.11). With the dredges in figure 2.3.6.11 we sample tracks of 150 m. Since the regular dredges sample 10 cm wide tracks and hydraulic dredges 20 cm wide tracks, these devices sample 15 m^2 and 30 m^2 of sea floor, respectively.

Figure 2.3.6.11: (Left) Dredge and (Right) an hydraulic dredge, specifically designed for shellfish surveys. While the regular dredge accumulates the sample in a cage (5 mm^2 mesh size), the hydraulic dredge sucks the sample up and "washes" the sample on board of the vessel.

Sampling from protected areas

In protected area it is often found too destructive to use dredges to monitor the quality and development of shellfish beds and reefs. Still, sampling tracks is preferable above grabs, i.e. about 70 Van Veen grabs are required to estimate mussel density on a bed with the same accuracy as one or two dredge samples. For epibenthic bivalves, however, video surveys are a powerful, non-invasive method to integrate the spatial variation in sublittoral shellfish beds (Fig. 2.3.6.12).

Figure 2.3.6.12: Image of a mussel bed in the Wadden Sea. The image is sampled from a video track. Video surveys in the turbid Wadden Sea are made possible by constructing the camera in the top of a pyramid-shaped lens that is filled up with tap water (in this case the lens has a base of 50x50cm). The glass window at the base of the pyramid glides just cm above the sea floor and the camera films trough less than 2 inches of turbid sea water.

Deep burying shellfish species

Some bivalve species are difficult to catch. For instance Soft Shelled Clams and Razorshell Clams bury deep in the sediment. Dredges sample in the range of 5 – 15 cm deep, by which only the upper parts of these bivalves are sampled (mainly the siphons) and number of missed individuals is unknown. To assess the population size of these two species more accurately, IMARES developed a device that samples 25 cm deep in the sediment, still sampling an area of 0.4 m^2 (Fig 2.3.6.13). Applying dredging techniques and the new device at the same time demonstrated that dredging underestimates the abundance of these deep burying bivalve species with a factor 5 to 15.

Figure 2.3.6.13: The semi-grab, a device designed to sample deep burying bivalve species.

Stock evaluation

Often, together with information on the size of a shellfish population comes the question on the development of this population for the following period. A prediction on the (long-term) development, natural mortality and evaluation of the uncertainties are necessary point of discussion. Although a time series of the stocks development over many years is the most useful to help predicting its fait, the abundance and distribution of age classes is a good second. The 1-year old and (depending on the species) 2-year old individuals demonstrate the potential for growth in the population, while the subpopulation of older individuals has a relatively predictable mortality rate, increasing with age. For example, the natural mortality (without predation) is in the range of 10-20% in Common Cockles from the Wadden Sea for which age > 2 years and winter conditions are average. The impact of bird or starfish predation on the development of the stock is worth estimating separately (Rappoldt et al. 2003).

2.3.6.9 References

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