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INDECO

Development of Indicators of Environmental Performance of the Common Fisheries Policy

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## **A review of the indicators for ecosystem structure and functioning**

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Dr.Gerjan Piet (RIVO), Dr.Fabio Pranovi (DSA-UNIVE)

Contributors: Dr.Eugene Andrulewicz (SFI), Dr.Magnus Appelberg(ICR), Dr.Angel Borja (AZTI), Dr.Otello Colloca (UR), Dr Otello Giovanardi and Dr.Sasa Raicevich (ICRAM), Dr.Simon Greenstreet (FRSMLA), Dr.Michele Gristina (IAMC-CNR), Dr.Astrid Jarre(DIFRES), Dr.Simon Jennings(CEFAS), Dr.Antti Lappalaninen(FGFRI), Dr.Simone Libralto(DSA-UNIVE), Dr. Mark Tasker (JNCC), Dr.Nikos Streftaris and Dr.Vassiliki Vassiloupoulou (HCMR)

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Lead name and organisation: Indrani Lutchman and James Brown, Institute for European Environmental Policy (IEEP)

[Final]

### **The INDECO project**

The purpose of this Co-ordination Action is to ensure a coherent approach to the development of indicators at EU level, in support of environmental integration within the CFP and in the context of international work on indicators. The principal objectives of INDECO are:

1. to identify quantitative indicators for the impact of fishing on the ecosystem state, functioning and dynamics, as well as indicators for socio-economic factors and for the effectiveness of different management measures;
2. to assess the applicability of such indicators; and
3. to develop operational models with a view to establishing the relationship between environmental conditions and fishing activities.

A consortium of 20 research organisations from 11 EU Member States is implementing INDECO. An Advisory User Group will provide a link between the researchers and policy makers, managers and stakeholders.

More information on INDECO can be found on the project's website:

<http://www.ieep.org.uk/projectMiniSites/indeco/index.php>

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## 1 INTRODUCTION

This report constitutes the deliverables number 5, 6 and 7 of INDECO. The aim of each of these deliverables was to provide a review of the available indicators at the levels of respectively, the population, the community, and the ecosystem. As there is a lot of overlap between these levels, both in terms of the available indicators as well as the data on which these indicators can be based, we felt that it would be most useful to combine the outputs of Work Packages 2, 3 and 4.

When implementing an Ecosystem Approach to Fisheries Management (EAFM), indicators are required (1) to describe the pressures affecting the ecosystem, the state of the ecosystem and the response of managers, fishers and society (2) to support management decision making, (3) to track progress towards meeting management objectives and (4) to communicate the effects of complex impacts and management processes to a non specialist audience. Indicators have for some time appeared to be accumulating opportunistically rather than through a structured approach to comprehensive coverage of the full range of ecological, social, and economic goals (ICES, 2001a). This situation has some undesirable consequences, including a management regime where managers, policy-makers, and science advisors would be struggling to support or apply decision-making constrained by (too) many indicators. Yet some of the constraints may be redundant or even contradictory, and some important ecosystem properties (or societal goals) may not be protected or advanced by the indicator-based management framework. Another reason for the seeming proliferation of indicators is that they have been developed without reference to the management frameworks they would support.

An international Working Group on ‘Quantitative Ecosystem Indicators for Fisheries Management’ (SCOR/IOC WG 119, [www.ecosystemindicators.org](http://www.ecosystemindicators.org)) was an important step towards co-ordinated efforts to structure what was fast becoming an indicator ‘jungle’. The Working Group began its work in 2001, membership included INDECO participants, and many of its results were presented at a symposium at IOC headquarters in Paris in spring 2004. The proceedings of this symposium were recently published (Daan et al., 2005), and an overview of the WG’s work, as well as key findings from the symposium are given by Cury and Christensen (2005).

Daan (2005) pleaded for a rigorous definition of what an indicator is and what purpose it is supposed to serve. He distinguished between metrics, which measure something specific, and indicators that are supposed to tell us something different from what they actually measure. In this review we did not make this distinction and used the term indicators for both indicators and metrics. This distinction will be made in the next stage of the project –the evaluation process, which should result in a comprehensive suite of indicators to be used for fisheries management.

Several methods have been proposed for classifying environmental management indicators, and a widely used framework is the pressure state response (PSR) system (Garcia and Staples, 2000). This framework uses pressure indicators (P) to measure the pressure impacting an ecosystem component, state indicators (S) to measure the state of the ecosystem component and response indicators (R) to measure the response of managers to the change in state. Since policy commitments and associated

objectives relate to state, the reference points, trajectories or directions needed to measure progress towards meeting objectives are initially set for state indicators. Achievement of these reference points, trajectories or directions will, by definition, mean that the operational objectives are being met. Once reference points, trajectories or directions have been set for state indicators, and the links between pressure, state and response are known, then corresponding reference points, trajectories or directions can be set for pressure and response indicators. Thus, an understanding of the relationship between state and pressure and how state reference levels translate into pressure reference levels is needed to make management decisions, since higher and lower level objectives both relate to state while pressure and response indicators are essential to manage state and often have the desirable properties of ease of measurement and rapid response times. As a result, guidance for year on year management decision making is often better based on pressure and response indicators, with changes in state assessed less frequently to confirm that pressure and response are affecting state as predicted (Nicholson and Jennings, 2004).

Initially, this review of existing indicators for Work Packages 2, 3 and 4 was expected to deliver an overview of state indicators that describe the structure and functioning of the ecosystem at different hierarchical levels (ie at the level of population, community and ecosystem). However, having established the relevance of pressure and response indicators within an EAFM and the importance of a thorough understanding of their link with state indicators, we felt that an overview of potential pressure indicators relevant for the Common Fisheries Policy (CFP) would also be within the remit of these work packages. Importantly, any suite of indicators should be tailored to fit the characteristics of the ecosystem in question, and the numbers and types of indicators used to support the EAFM will therefore vary among management regions (eg eco-regions). Any particular suite of indicators will further depend on actual and potential human impacts, as well as resources available for monitoring and enforcement (Degnbol and Jarre, 2004; ICES, 2005b). We therefore collated an overview of the data sources available by geographic area, and linked these with the type of indicators they may support.

## **2 STATE INDICATORS**

This review of literature on indicators showed that many different indicators exist, describing different components or set of components in the ecosystem, and each of which may provide a specific type of information to answer a specific question. Many classifications of these indicators exist, ranging from classifications based on the organisation level, along the gradient from single-species to ecosystem functioning (Link, 2002; Rochet and Trenkel, 2002) to classifications based on their properties (Rice, 2000) or nature (Link *et al.*, 2002) or ecosystem objectives (Gislason *et al.*, 2000).

In order to achieve a holistic framework for ecosystem management and/or protection it is necessary to provide a comprehensive suite of indicators that adequately cover the structure and functioning of the entire ecosystem and its components at different hierarchical levels.

In order to achieve this goal, we distinguished between structures and processes in the ecosystem (see Table 1). We distinguished indicators that describe the structure of the ecosystem in terms of its physical/chemical characteristics, while the biotic components are described according to the hierarchical levels: population and community. “Habitat” is defined as: “the total of all the environmental (i.e. physical/chemical) conditions present in the three-dimensional structural configuration occupied by an organism, population, or community”. Therefore it is considered a combination of the physical and chemical ecosystem components in combination with one or more of the biotic components (eg macrophytes for seagrass beds, benthic species for coral reefs etc.), which are strongly associated with this physical/chemical environment or may even be the structural habitat agents. The functioning of the ecosystem is described at the ecosystem level. Altogether this provides a generic framework that distinguishes the ecosystem components and processes that occur in the European waters and together cover all aspects of ecosystem structure and functioning. The assumption is that if all shaded cells of Table 1 contain one or more indicators all aspects of the ecosystem are covered. However, for application in individual regions, not all of the ecosystem components or hierarchical levels may require indicators for objective-based management of human activities to be guided effectively. Table 1 may be instrumental in the process of selecting a limited suite of indicators that adequately covers all aspects of the ecosystem that need protection within an EAFM.

Table 1 Ecosystem elements relevant for identifying the effects of human activities on an ecosystem.

The elements are based on a hierarchical level of organisation of ecosystem components resulting in a comprehensive coverage of the system by indicators provided at least the shaded cells are filled with one or more indicators.

Ecosystem components	Structural			Functional
	Population	Community <sup>2</sup>	Habitat <sup>3</sup>	Ecosystem <sup>3</sup>
Macrophytes <sup>2</sup>				
Benthos				
Cephalopods <sup>2</sup>				
Fish				
Phytoplankton				
Zooplankton <sup>1</sup>				
Seabirds <sup>2</sup>				
Marine mammals <sup>2</sup>				
Marine reptiles <sup>2</sup>				
Physical				
Chemical				

<sup>1</sup> including large pelagic invertebrates such as jellyfish

<sup>2</sup> for components consisting of few species the community level may not be applicable

<sup>3</sup> Habitat or ecosystem level indicators may be an aggregate of one or more components

Following Table 1 we will distinguish different sections for physical/chemical indicators, population level indicators, community level indicators, habitat indicators and ecosystem level indicators. Other groups which may include one or more of the ecosystem components distinguished in Table 1 and are often distinguished in other frameworks are “threatened and declining species” or “sensitive and opportunistic species”. These groupings may apply to benthos, fish, seabirds, marine mammals or marine reptiles (eg turtles) and would conceptually fall under the population level. However, they may be used as sentinels at the community level instead of often more complicated indicators. Definitions of these groupings are provided in section 2.2.

## 2.1 Physical/chemical indicators

While many of the physical/chemical indicators will not be affected directly by fisheries they may affect biotic ecosystem components that are subject to management under CFP. It was therefore considered relevant to incorporate some indicators that describe this aspect of the ecosystem.

### **Temperature;**

**Nutrient concentrations;** notably Nitrogen (N) and Phosphorus (P) as these may cause eutrophication



**Oxygen concentration;** both in the water column and on the sediment surface. This is notably relevant in the Baltic, where oxygen-limited conditions are regularly encountered.

## **2.2 Indicators at the Population level**

At the population level we distinguish indicators relating to the 'health' of the population (incidence of disease, condition etc.) or the population status of individual species or stocks. For indicators that describe the population status of individual species or stocks it is relevant to distinguish between indicators based on input- or output data from the assessment process which only apply to the assessed species (usually of commercial value) and indicators based on other sources of data (usually monitoring programmes) which may be used for both assessed and non-assessed species.

### *2.2.1 Health*

**Condition factor;** In fish ecology, condition is believed to be a good metric of the general well-being or fitness of the population under consideration (Adams and McLean, 1985). This can also be expected to apply at the level of the community. Several condition indices are used in fishery science as metrics of the length-weight relationship of a population. However, the conversion of a two-dimensional length-weight relationship into a single statistic results in a loss of information and, in many cases, an inaccurate representation of that relationship. After review of the most common condition indices by Bolger and Connoly (1989), Cone (1989) proposed the calculation of estimates of ordinary least squares regression parameters as the most accurate method of examining length-weight relationships for fish populations. However, since regression parameters are commonly heterogeneous and slope and intercept are often inversely related, valid interpretation of the results is difficult (Bolger and Connoly, 1989). An alternative, the estimated weights of fish of a particular species and length from regression equations specific to the groups under consideration (De Silva, 1985), has the disadvantage that it is dependent on an arbitrary choice of the length. For the community, one possibility would be to use the average condition of a theoretical community of fixed size-structure and species composition over time as an index of body condition. For each individual in this community, the condition is expressed as the weight calculated from the species-specific length-weight relationship per year and the mid-range length of the size-class. Considering that length-weight relationships are only determined annually for a subset of (commercial) species, this theoretical community will consist of a subset of species that are present in the actual community. Another possibility would be to use the full frequency distribution of condition factors (calculated correctly) across a suite of species, and compare the distributions themselves across space or time, or compare their ordinations.

**Incidence of disease, pathogens, parasites, and contaminants;** Considerations relating to the types and incidence of diseases and parasites are similar to those relating to body burdens of contaminants and other measures of body condition. If lower environmental quality affects the biological health of individuals, their

resistance to disease and parasites may be lowered. Hence, it is possible that metrics based on the incidence of disease or parasites across a full community could be developed.

### 2.2.2 *Assessed species only*

The status of the stocks of many commercial fish species is assessed and for these species indicators are available that are based on Sequential Population Analysis (SPA) output (Gulland, 1956). SPA requires reliable estimates of the age composition of the total international catches and allows an evaluation of the historic development of fishing mortality (F) and stock numbers by age group up to the present day. While these methods yield converged parameter estimates for year classes that have reached the end of their life, estimates for recent years will vary to some extent during subsequent assessments, because of uncertainty about the proportion still surviving. To obtain the best possible estimates, a variety of statistical methods has been developed that use additional information on catch per unit of effort (cpue) derived from commercial and/or research vessel data to improve estimates of fishing mortality and stock numbers. Essentially, the methods reconstruct population size on the basis of an exponential decay in numbers surviving as governed by removals by the fishery and by natural deaths. For the latter component no direct information is available and therefore a common assumption has to be made that natural mortality (M) is constant from year to year at some average level that may be age group specific. Inferences about this level may be made from independent data sets (tagging experiments in particular). However, in reality M may vary from year to year and the uncertainty introduced by the assumption of constant M depends critically on the fraction of the total mortality (Z) represented by fishing mortality (F): hence for the older fish assessments for heavily exploited fish stocks ( $F \gg M$ ) are inherently more accurate than for lightly exploited stocks ( $F \ll M$ ). Although F or Z are output from the assessment process and delivered as an indicator of stock status from the perspective of the stock and ecosystem they are essentially pressure indicators (see section 3.1)

**Recruitment (R);** The number of young fish produced each year, which survive from spawning to enter the adult fish stock or the fishery.

**Spawning Stock Biomass (SSB);** ‘Spawning stock biomass of commercial fish species’ was identified as an Ecological Quality element. The associated EcoQO is that the stock biomass should be ‘above precautionary reference points for commercial fish species where these have been agreed by the competent authority for fisheries management’. The relevant precautionary reference points are those for ‘spawning stock biomass, also taking into account fishing mortality, used in advice given by ICES in relation to fisheries management’. ICES has established  $B_{pa}$  and  $F_{pa}$  as the respective precautionary reference points for spawning biomass and fishing mortality for use in formulating advice. They are set on a stock-specific basis, and take account of both stock dynamics and uncertainties in the assessment.  $B_{pa}$  is the spawning biomass at and above which there is a low probability that true SSB is so low that productivity is impaired.

**Mean Age of the population;** This is calculated similar to the community level indicator Mean Weight (see chapter 2.3.1) where weight per specimen is substituted by age

**Age above which 50 % of the population is mature;** this index can be calculated directly from data (standard fish length, age at length and maturity status) collected for most commercial species on research vessel trips.

**Proportion of commercial fish stocks within safe biological limits (ie  $SSB > B_{pa}$ , and  $F < F_{pa}$ );** Rather than stating that the ‘spawning stock biomass of commercial fish species’ should be ‘above precautionary reference points...’ where the reference points are ‘those for the spawning stock biomass *also taking into account fishing mortality, ...*’, (Piet and Rice, 2004) suggest the EcoQO should explicitly be based on the proportion of stocks within safe biological limits, where  $SSB > B_{pa}$ , and  $F < F_{pa}$ . are considered together. Since the existing management approaches for individual stocks are all based on the objective to exploit stocks within safe biological limits (ie  $SSB$  above  $B_{pa}$ . and fishing mortality sustainable) the wording of the EcoQO could be changed so that it would simply condense this information into a form that gives an appropriate overview of the overall status of commercial fish stocks in a specific geographic area. Thus, the suggested EcoQ element would be the ‘proportion of commercial fish stocks within safe biological limits (ie  $SSB > B_{pa}$ , and  $F < F_{pa}$ )’ and the objective (EcoQO) should be that, for any given environmental regime, this EcoQ should be at or above a desired level. The desired level is a societal/political decision relative to the EcoQ reference level (ie where the anthropogenic influence on the ecological system is minimal). An advantage of this indicator is that it is easy to set a reference level as this is by definition equal to 100%.

**Fishing mortality (F);** A measure of the proportion of a fish stock taken each year by the fishery, see  $SSB$ .  $F_{pa}$  is the fishing mortality at and below which the true fishing mortality has a low probability of leading to stock collapse.

**Total mortality (Z);** is fishing mortality (F) + Natural mortality (M)

**Exploitation rate (F/Z);**

### 2.2.3 All species

For the non-assessed species only indicators based on surveys or commercial by-catch data are available. Several of the community level (eg size-structure) indicators (chapter 2.4.3) may also be used at the population level.

**Total Biomass** is the sum of weights of all individuals of one species, when this is done across species it becomes a community level indicator. A common proxy for this is the CPUE of a survey/monitoring programme.

**Total Number** is the sum of all individuals of one species (see Total Biomass).

**Size above which 50 % of the population is mature;** this index that can be calculated, at least for many fish species, directly from data (standard fish length and maturity status) collected on national research vessel trips.

**Genetic diversity** is a fundamental component of biodiversity and is as critical to sustainability of our natural resources as are diversity of species and ecosystems.

Virtually all species are composed of populations that exist somewhat independently of each other, and thus genetic diversity exists both within and among populations of one species. Levels of genetic diversity in any one population are determined primarily by four forces: (1) mutation, the ultimate source of all genetic diversity; (2) migration, the exchange of individuals between populations; (3) natural selection, the removal of 'unfit' individuals from the population; and (4) genetic drift, random changes in gene frequency of each generation due to limited numbers of breeding adults. Mathematical tools have been developed that allow diagnosis of the relative strengths of the four genetic forces and, indirectly, properties of populations, such as population size, breeding structure, and dispersal abilities. Measurement of genetic diversity with molecular markers can add value to assessments of ecological condition derived from other ecological indicators, such as landscape and species assemblage indicators. Population parameters can be effectively estimated with molecular markers and used to characterize the geographic structure and connectivity of populations critical to interpreting data for ecological assessments. Genetic diversity also serves as an independent indicator of environmental condition as environmental stressors typically reduce genetic diversity, primarily through the forces of selection and genetic drift. A reduction of genetic diversity reduces long-term sustainability of the population. The molecular technologies that can be used for genetic diversity analysis include allozyme, DNA fingerprint, microsatellite DNA, and mitochondrial DNA fragment or sequence analysis (Bagley, Franson *et al.* 2002).

**Presence of “indicator”, “charismatic”, or “threatened and declining” species;** Societal concerns about the environment often focus on a limited number of organisms that are in some way attractive. Such charismatic species, including dolphins, killer whales, large sharks, and a variety of seabirds, are often viewed as sentinels of the health of the ecosystem. The scientific justification for such a view varies with the species, but as many are higher predators and long-lived they will often be more sensitive to human impacts and more liable to be affected by cumulative impacts over time. Indicator and sensitive species are selected on the grounds of criteria that explicitly use their known response to impacts and the definition of such species is: “A species easily depleted by human activity and, when affected, is expected to recover over a long period or not at all”. As such, the term “sensitivity” takes into account both the tolerance to and the time needed for recovery (largely species dependent) from the stressor. Fragile species are considered to be especially susceptible to physical/mechanical disturbance. Many examples of such indicator taxa exist in the pollution literature (Pearson and Rosenberg, 1978) and a considerable number of benthic taxa have also been suggested as being sensitive to direct effects of fishing (ICES 2003). Development of this approach is often more difficult than it at first appears as lists of sensitive/indicator taxa are rarely transferable between regions and developing the list from the impacted system studied leads to circularity. Although, some of these approaches are being used in the implementation of the Water Framework Directive (Borja *et al.*, 2000; 2003, 2004; Rosenberg *et al.*, 2004). Diaz *et al.* (2004) provide a complete review of this issue. Several criteria exist for “threatened and declining species” and therefore the most pragmatic approach is to assume that strictly protected or endangered/vulnerable species previously identified by IUCN, the Bern and Bonn Conventions, OSPAR and other national programmes should provide a comprehensive species list.

**Presence of opportunistic species:** These are defined as species (second- and first-order, based on Borja *et al.*, 2000, ecological groups IV and V) that follow the reproductive r-strategy (*sensu* Pianka, 1970), with short life-cycle (<1 year), small size, rapid growth, early sexual maturity, planktonic larvae through the year, and direct development.

### 2.3 Indicators at the Community level

There are two important aspects of the community: size-structure and species composition. Size-selective fishing targeting the larger fish should result in a change in size-structure of the community. In a fisheries context the change in species composition will be driven mainly by life-history characteristics of the species in the community as the typical K-selected species that grow slowly to large sizes, with a late size and age at which they mature and that produce few offspring will be stronger affected by fisheries than the r-selected species that show the opposite. Any change in species composition or extinction of species will be reflected in the species diversity measures which have become an important topic in many policy documents.

#### 2.3.1 Size structure

**Slope size-spectra;** (Sheldon *et al.*, 1972) showed a log-linear relationship between fish biomass and size. In spite of the differences in numbers and size between species, the community as a whole shows a log-linear decrease of biomass with increasing size. The slope of this relationship is assumed to reflect the efficiency of energy transfer and the mortality rate and can be used as a metric of the size-structure. Although several alternatives have been suggested since its introduction (Borgmann, 1987; Boudreau *et al.* 1991; Boudreau and Dickie, 1992; Thiebaut and Dickie, 1992; Sprules and Goyke, 1994), the conceptual basis is widely recognized (Rice and Gislason, 1996). The general formula for the log-linear relationship between size and biomass is:  $\ln(y) = a * \ln(x) + b$  where:  $x$  = size,  $y$  = biomass or number,  $a$  = slope,  $b$  = intercept. A disadvantage is that slope and intercept are not independent, which makes it difficult to interpret a time series of either one. Also, an arbitrary choice must be made about the minimal size of fish that should be incorporated in the linear regression; depending on the mesh-size of the gear and towing speed, certain size-classes will be under-represented and thus disturb the relationship. As a way around the problem of dependency of slope and intercept (Daan, *et al.*, 2005) suggest to use mid-length height (or intercept of the centred spectrum) instead of intercept as this appears to be independent from the slope. Rice and Gislason (1996) studied the log-linear relationship for the North Sea fish community and observed a change in slope caused by a decrease in large fish. This change was attributed to the impact of fisheries. (Gislason and Lassen, 1997) showed that a linear relationship between fishing effort and the slope of the size spectrum can be expected. WGECO (ICES, 1998) reported that there is now sufficient theoretical and empirical evidence to be confident that changes in fishing mortality should result in a long-term change in the slope of the size spectrum. Provided that growth and relative recruitment of the constituent species do not change, the change in the slope should be directly proportional to the change in exploitation rate of the community.

**Mean weight or Mean length;** are usually based on surveys and calculated per haul. Mean Weight is calculated as:  $\bar{W} = \sum_k W_k / N$  where  $W$  is the body mass of an individual and  $N$  is the total number of individuals.

**Proportion of large fish;** proportion of weight or numbers of fish larger than a specific size

**Length-frequency distribution;** the length-frequency distribution of the community is determined by summing up the number of individuals caught per size class. In most cases these size classes will be cm-classes. A relevant metric to represent the length-frequency distribution may be the total number or weight of the community above a specific length threshold. Another relevant metric that may be derived from the length-frequency distribution is the percentage composition of groups that cover certain size ranges.

**k-dominance curves** are graphical representations of abundance where size-groups are ranked according to their abundance (Rochet, 1998; Rochet and Trenkel, 2003)

**Multi-dimensional ordination;** For studies involving complex tabular data (commonly  $i$  rows as sampling sites,  $j$  columns containing species or size classes and cell entries of (transformed) abundances of species or size-class  $j$  at site  $i$ ), ordination methods can be used to reduce this complexity to a small number of (usually) orthogonal (ie, not correlated) gradients (reviewed in (Jongman *et al.*, 1987). Several ordination methods exist such as Principal Components Analysis (PCA), Correspondence Analysis (CA), and Non-metric Multidimensional Scaling (MDS). Of these methods, MDS has become the preferred technique for ecological ordinations of fish communities because of its increased robustness in the face of irregular distributions of abundance and high sampling variance (Clarke and Ainsworth, 1993; McRae *et al.*, 1998). Although this technique may reveal patterns or trends that would otherwise remain obscured, interpretation or linking them to useful management information is difficult. Here, ordinations are listed under size structure, but ordinations on the basis of species abundances as well as frequencies of size classes are also common.

### 2.3.2 *Species composition*

**Species presence / abundance;** there are several informative measures of community structure that do not take into account the species identities of the community. It is conceivable therefore that changes to species presence or absence may go undetected unless reference is made to lists that include the relative abundance of species.

**Index of rare species;** variability in abundance of the uncommon species in a survey can illustrate underlying patterns of change that are not evident from analysis of the dominant parts of the community. For example, the presence of unexpected migrants or the decline in population size of less common species can be used as metrics of previously unobserved adverse human impact. (Daan 2001) proposed a spatial and temporal diversity index that was based on species rarity.

**Index of declining or increasing species;** a variety of metrics are available based on the proportion of species in the community which are showing increases or decreases

in abundance (biomass). These measures are at best coarse and may provide little information about causes of the changes, but are readily interpreted and understood by non-specialists. They might therefore be best used as indicators that could trigger research into the causes of change.

**Proportion of sensitive of threatened species;** threatened species are those occurring on the IUCN red lists.

**Non-indigenous species;** the presence of non-indigenous species, used here to mean species introduced by anthropogenic activities rather than natural invasions/range expansions is, by definition, a failure to maintain natural levels of biological diversity. For larger organisms, the presence of non-indigenous species is easily recorded; for lower organisms, our lack of knowledge of pristine fauna (and of natural invasions) makes this more difficult (Eno *et al.*, 1997).

**Species turnover/loss rates;** the rate at which species composition changes from year to year in samples taken in a consistent manner and location is a widely used metric in terrestrial conservation biology. It requires consistent and reliable sampling where sampling is expected to detect most of the species that are present. Measures of turnover rates are most effective at local scales, and may be less effective at the scales of large marine ecosystems when many samples are pooled.

**Theoretical Distribution Metrics Log-Series and Log-Normal:** Parameters derived from these distributions have the advantage of being relatively sample-size independent (Kempton and Taylor, 1974). Also, there has been considerable debate in the ecological literature regarding the theoretical reasons as to why distributions of species relative abundance should follow either one of these models (Fisher *et al.*, 1943; Preston, 1962; 1980; Kempton and Taylor, 1974; May, 1976). One major difficulty with using these indices lies in the necessity to fit the data to the distributions, to estimate parameters of the distribution for subsequent use. Generally this tends to require a substantial amount of data, rather negating the advantage of sample-size independence. Often fitting the data to the distribution proves to be difficult, and in testing the significance of any fit, one hopes not to disprove the null-hypothesis, which is unsatisfactory from a statistical perspective.

**Abundance Biomass Comparison (ABC) curves** Abundance Biomass Comparison curves, first proposed by Warwick (1986), were initially used to demonstrate the effects of pollution on communities, but have subsequently been used to examine the effects of various anthropogenic activities, particularly on marine benthic invertebrate communities (Warwick *et al.*, 1987; Agard *et al.*, 1993; Warwick and Clarke, 1994). ABC curves compare species ranked dominance plots in terms of both abundance and biomass. They have their theoretical basis in classical evolutionary theory of *r*- and *k*-selection. In an undisturbed state, the ranked dominance by biomass plot should lie above the abundance curve. In a disturbed state, the ranked dominance by biomass plot should lie below the abundance curve (Clarke and Warwick, 2001). Yemane *et al.*, (2005) use the W statistic as an index. This statistic is used to compare the two plots and holds positive values when the biomass plot lies above the abundance plots (ie an undisturbed state) and negative values when the abundance plot lies above the biomass plot (ie a disturbed state). Values close to zero are considered to characterise a moderately disturbed community. Only relatively recently have ABC curves been

used to study the effects of fishing on marine communities, and generally these have shown the expected response to increasing fishing disturbance (Bianchi *et al.*, 2001; Blanchard *et al.*, 2004; Jouffre and Inejih 2005; Yemane *et al.*, 2005).

**Multi-dimensional ordination** see size structure, section 2.3.1

**Mean Trophic level** see ecosystem level, section 2.4

**Life history composition** is an important aspect of species composition. There is extensive theoretical literature that distinguishes K-strategists from r-strategists, that is, species whose life history characteristics adapt them to living in undisturbed, stable environments vs. those adapted to living in frequently disturbed, variable environments. Particular life history characteristics can be used to place species somewhere along this continuum, and thus provide an indication of vulnerability to disturbance by additional fishing mortality. Correspondingly, the life history character composition of communities may provide a metric of the past impact of fisheries on that community. Values for one or more of the parameters are available for many species from the literature. This list, however, is far from comprehensive and for several of the parameters, values are available for only a few species. Therefore, unless we have much better tabulations of life history traits for large numbers of species, establishing the relationship with fishing impact may suffer from circularity. Community metrics based on these parameters are calculated per year by weighting the community species' biomasses with the value of that particular life history parameter. Other potential metrics might be derived from sex ratio, lifetime reproductive output, or growth rates. Possible life history characteristics that might be used as such metrics include:

**Mean maximum length** is indicative of the composition of the fish community in terms of life history types, was calculated per haul as:  $\overline{L_{\max}} = \sum_j (L_{\max j} N_j) / N$  where

$L_{\max j}$  is the maximum length obtained by species  $j$ ,  $N_j$  is the number of individuals of species  $j$  and  $N$  is the total number of individuals.

**Mean maximum age** is calculated like Mean maximum length but with age substituted for length

**Size above which 50 % of the population is mature;** this is an index that can be easily calculated, at least for many fish species, directly from data (standard fish length and maturity status) collected on most national research vessel trips.

**Age above which 50 % of the population is mature;** the data used to derive the index above, when combined with age at length keys, can also be used to calculate the age at which 50% of the population(s) sampled are mature.

**Elasmobranch/teleost ratio;** this index partially resembles and could be integrated by those indexes proposed by Link (2005), which consider the ratio of (top) predators vs. lower trophic levels species. Link (2005) proposes the use of a) BLT4+ : Biomass of all species at trophic level 4 and above; b) Bpisc: Biomass of all piscivores. The Elasmobranch versus teleost (bony fish) ratio standardizes the biomass of high trophic level species to lower level preys, moreover it considers species very sensitive to fishing activities. In fact, elasmobranchs are considered generally to be a group highly



vulnerable to fishing disturbance, being often characterised by large maximum size, low reproductive output, high age and size at maturity (K-strategists). Moreover, many elasmobranchs (eg sharks) occupy the highest level of the trophic web and are usually top predators. Several studies highlighted the threat posed by fishing activity to this taxonomic group (for a review, see Stevens *et al.*, 2000) and, due to the high occurrence of elasmobranchs as by-catch, the concerns about their eventual decline goes beyond the smaller group of targeted species (Stobutzki *et al.*, 2002). Since teleosts (bony fish) comprise mainly species characterized by high reproductive output, relatively smaller age at maturity and smaller size, the elasmobranch/teleost ratio is intended to summarize the resulting changes in the fish community between groups that are (generally) characterized by contrasting life-histories (and vulnerability).

**Fecundity;** number of eggs per female or number of eggs per body weight

**Mean  $k$  and/or  $L_{\infty}$  of von Bertalanffy growth curve;**  $k$  and  $L_{\infty}$  are parameters of the von Bertalanffy growth curve, which has been established for many species under both experimental and natural conditions. These parameters have been used as metric to illustrate specific types of change in the species composition of fish in the groundfish community of the north-western North Sea. Parameter values were available for 23 species within the sampled groundfish assemblage. Although not representative of the entire community, nevertheless, these 23 species accounted for 99% of all the individuals sampled. Analysis of the data suggested a long-term decline in the average  $L_{\infty}$  and increase in the average growth rate ( $k$ ) of fish in the community (Jennings *et al.*, 1999). The community was increasingly dominated by faster growing fish which grew to smaller ultimate length.

**Biodiversity;** The concept of species diversity has a long history in the ecological literature; countless different metrics have been devised and utilised in numerous different studies covering taxa from just about every phylum in the plant and animal kingdoms (Brown, 1973; Connell, 1978; Davidson, 1977; Death and Winterbourn, 1995; Eadie and Keast, 1984; Heip *et al.*, 1992; Huston, 1994; MacArthur and MacArthur, 1961; Magurran, 1988; May, 1975; Rosenzweig, 1995; Washington, 1984). Despite this long tradition, and perhaps in part due to the proliferation of different metrics, species diversity as a concept has been questioned (Hurlbert, 1971). Hill (1973), however, argued that much of the perceived difficulty with the concept lay in the fact that it combined the two characteristics of richness and evenness. The theoretical underpinning of the concept has been discussed (May, 1975; 1976). The ability of the different indices to actually detect environmental and anthropogenic influences has on occasion been questioned (eg, Robinson and Sandgren, 1984; Chadwick and Canton, 1984), however, in general these problems have usually been associated with inadequate sample size (Soetaert and Heip, 1990). Several species diversity metrics can be considered as candidate indicators.

**$k$ -Dominance Curves** The simplest representation of the species relative abundance data, on which any metric of species diversity is based, is the straightforward graphical representation of relative abundance on species abundance ranking. The most commonly used representation of this type is the  $k$ -dominance curve (Lambhead *et al.*, 1983; Clarke, 1990). This index was endorsed by WGECCO (2001)

because of the simple, easily comprehensible way that it conveyed the information, avoiding the problems of trying to convey both aspects of species diversity in a single numeric parameter. Well-defined statistical methods for determining differences between samples have been developed (Clarke, 1990).

**Hill's N0 N1 N2;** Hill (1973) suggested that several of the most commonly used diversity indices were mathematically related, forming a family of indices varying in their sensitivity to species richness and species evenness (Peet, 1974; Southwood, 1978).  $N_0$  is species richness, a simple count of the number of species in the sample,  $N_1$  is the exponential of the Shannon-Weaver diversity index (Shannon and Weaver, 1949, sometimes referred to as the Shannon-Wiener index),  $H$ , computed as  $H = -\sum p_i \cdot \ln(p_i)$ , effectively the number of abundant species, and  $N_2$  is the reciprocal of Simpson's diversity index,  $d$ , computed as  $d = \sum p_i^2$ , effectively the number of very abundant species. These indices are all affected by sample size, which is a major disadvantage with regard to monitoring change in marine ecosystems where sampling is logistically difficult and expensive. As the Hill number notation increases, the index moves from being a measure of species richness to one of species dominance. Low N number metrics, eg,  $N_0$  and  $N_1$ , are consequently the most affected by variation in sample size. When the problem of variable sample size can be addressed, these metrics have been used to demonstrate long-term temporal and spatial trends in species diversity that have been associated with differences in fishing activity (Greenstreet and Hall, 1996; Greenstreet *et al.*, 1999).

**Species-Effort Index;** many scientists have argued on theoretical grounds that species richness (eg,  $N_0$ ) is the most important aspect of species diversity, but the sampling effort required to estimate this adequately from the data normally available from fish or benthic surveys is usually prohibitive. A species-effort index derived from the parameters of the function describing the rate of increase in the number of species recorded, as samples from a survey are increasingly aggregated, may offer a solution. This function is exactly equivalent to the species-area relationships of the form  $S=cA^z$ , which describes species richness in habitats of varying size, eg, islands, continents (Rosenzweig, 1995). The two parameters,  $c$  and  $z$ , could perhaps be derived from a much smaller number of trawl samples to provide a relatively sample-size independent estimate of species richness.

**Margalef's Species Richness Index;** this index provides an alternative solution to an index of species richness that takes account of variation in sampling effort. Margalef's index,  $d$ , is calculated as  $d = (S - 1)/\ln(N)$ , where  $S$  is the number of species in the sample and  $N$  the number of individuals (Clarke and Warwick 2001).

**Pielou's Evenness Index;** Pielou's index,  $J$ , provides a measure of the evenness of the distribution of the individuals in a sample across the species sampled, calculated by  $J = H / \ln(S)$ , where  $H$  is the Shannon-Weaver index of diversity and  $S$  is the number of species in the sample.

**Taxonomic Diversity Indices;** taxonomic diversity indices were developed by Warwick and Clarke (1995; 1998). They are closely related to the Shannon-weaver Index, but they also provide additional information with respect to the level of phylo-

genetic relationship present in samples. As such they were considered to convey some information on the genetic diversity aspect of biological diversity. They have been demonstrated to be relatively sample-size independent, and to be sensitive to ecological perturbation in circumstances where other species diversity metrics, such as the Shannon-weaver, or Simpson's Indices, fail to respond. They are, for example, particularly sensitive to situations where a group of particularly vulnerable, closely related species may be in decline and being replaced by alternative, unrelated species. The impact of fishing on elasmobranch fish species is an example of this (Rogers *et al.*, 1999). However, in circumstances where Hill's N1 and N2 are varying, these taxonomic indices may convey little additional information (Hall and Greenstreet, 1998).

**Elasmobranch/bony fish ratio;** the elasmobranch vs. bony fish ratio is intended to summarize the eventual changes occurring in the fish community between groups which are (roughly) characterized by contrasting life-histories (and vulnerability). In fact elasmobranchs are considered to be species highly vulnerable to fishing disturbance, being often characterised by large maximum size, low reproductive output, high age and size at maturity (K-strategists). Several studies highlighted the threat posed by fishing activity to this taxonomic group (for a review, see Stevens *et al.*, 2000) and, due to the high occurrence of elasmobranchs as by-catch, the concerns about their eventual decline goes beyond the smaller group of targeted species (Stobutzki *et al.*, 2002). In contrast bony fish comprises mainly species characterized by high reproductive output, relatively smaller age at maturity and smaller size.

The definition of a baseline for this indicator could be hampered by the lack of information in areas that have long histories of fishing and poor markets for elasmobranchs.

This index could partially resemble and be integrated by those indexes proposed by Link (2005), which consider the ratio of (top) predators vs. lower trophic levels species since many elasmobranchs (eg sharks) occupy the highest level of the trophic web and are considered top predators. Link (2005) proposes the use of a) BLT4+ : Biomass of all species at trophic level 4 and above; b) Bpisc: Biomass of all piscivores. Nevertheless it must be stated that the presence of constant scaling of elasmobranchs biomass to biomasses at different trophic levels has not been demonstrated.

**Pelagic/Demersal fish ratio;** the dynamics over time of fish community composition can provide useful insights on the food source dynamics or habitat state in marine environments. With this aim, the Pelagic/Demersal ratio (P/D) was defined. The pelagic species are positively influenced by nutrient enrichment (Caddy, 1993) because the enhanced primary production (phytoplankton production) has a positive effect on this group while the possible negative effects like anoxia caused by eutrophication mainly affects the bottom habitat. Indeed, many European coastal areas and semi-enclosed basins have increasingly been subjected to this phenomenon over the last century (Diaz, 2001). Accordingly, demersal species, which are more influenced by the dynamic of the benthic community should be more sensitive to the negative consequences of excessive nutrient enrichment. Thus moderate nutrient enrichment may have a positive effect on the demersal biomass, while increasing

levels of nutrient availability and occurrences of anoxia may have serious negative effects on the demersal biomass (Caddy, 2000). Therefore, positive trends over time of P/D ratio are related to increasing levels of nutrient enrichment, and extremely high values of P/D are signs of eutrophication or even anoxia (Caddy, 2000; De Leiva Moreno et al., 2000). Therefore the P/D ratio can be used in a suite of indicators in order to distinguish the influence of eutrophication from the effects due to fishing that are often combined in other indicators. However, this indicator, although mainly related to nutrient availability in the ecosystem, might also be affected by changes in the structure of fish community due to the removal of top predators which, through a top-down control, could reflect in an increase of prey abundance. Hence, in order to disentangle eutrophication and fishing induced changes in the fish community, the P/D ratio should be applied in addition with estimates of eutrophication trends in the considered areas.

**Discard/catch ratio in a commercial fishery;** discards are defined as the fraction of the total catch that is discarded at sea by fishermen during sorting operations driven by legal, economic or personal considerations (Hall, 1999). It includes both target (undersized individuals, catches exceeding TAC quotas) and non-target species. Discarded biomass was evaluated as high as 27% of the global catch (Alverson et al., 1994). However, since this estimate is mainly based on undersized commercial species data, it should be considered an underestimate. The incidence of discarding in a commercial fishery is often described by means of discarded biomass/catch ratio (discard ratio), which varies by orders of magnitude depending on the considered fishing gear or activity, the exploited habitat, and the target species density. As an extreme example, the discard ratio reaches values higher than 10:1 in the shrimp trawl fisheries (Alverson et al., 1994). The mortality rate of the discarded species can vary (both within the short and long-term) according to their physiology (Bergmann et al., 2001), fragility (lethal and sublethal injuries (Pranovi et al., 2001) and to the pressure and temperature shock they are subjected (Gamito and Cabral, 2003). This additional mortality may result in changes in community structure and, in some cases, in increased risk of local or regional extirpation for species more vulnerable to the effects of fishing disturbance (mainly large, fragile and long-living species—ie K-strategic species). The metric usually employed is the discard ratio (Discarded Biomass vs. Commercial Catch Biomass; Alverson et al., 1994); but also the total discarded biomass could be analysed. Finally, it should be better acknowledged that discards should be included in the computation of several of the above mentioned indicators derived from the commercial catches, (eg, PPR; Pauly and Christensen, 1995), in order to improve the link between fishing activities and the changes in ecosystem structure and functioning (Zeller and Pauly, 2005).

**Scavengers;** an important side effect of discarding is the role played by this process in providing a source of energy subsidies through additional carrion supply (Britton and Morton, 1994). All this can result in enhancing the role played by those facultative scavengers that are able to exploit this 'new' source of energy. This effect is particularly important for those species having physiological features or behavioural adaptations that enable them to survive the capture and discarding processes. Ramsay et al. (2000) showed for *Asterias rubens*, that scavengers can increase their abundance at intermediate levels of fishing disturbance. Such phenomena, have been proposed as

an explanation of the homogenisation of benthic habitat in heavily exploited ecosystems (Pranovi et al. in press; Raicevich et al., submitted). The change of the scavenger abundance has been recently proposed as an ecosystem indicator (Link, 2005) even though the definition of warning threshold and limit reference point can still be considered rather speculative..

#### **2.4 Indicators of Habitat size and quality**

The health of the ecosystem may also be determined by the occurrence, size or quality of specific habitats. The effect of fishing on these habitats may be quantified based on occurrence or size of the habitat when fishing activity changes the habitat to the point where it does not classify as such a habitat anymore (eg coral reef) or if the effect is more subtle by an index of overlap with fishing activity and/or a change in quality of the habitat. Potential indicators are:

**Size of the habitat;** size of the area covered by a specific habitat.

**Proportion of the habitat fished;** or number of fishing registrations (eg from VMS) in that area.

For the assessment of habitat quality there are two relevant aspects of our definition of habitats (see section 2). One emphasizes the physical and chemical nature of the environment, including any role that biological components may have, while the other centres on a chosen species or life history stage. The former, what we might refer to as ‘conceptual-habitat’, would include descriptions such as, for example, coral reef, sandflat, and continental shelf muds. The latter, the ‘species-habitat’ concept, would include descriptions such as ‘cod spawning habitat’ and ‘plaice nursery habitat’. The EC Habitats and Species Directive (EC, 1992) uses both definitions, formally stating them as,

- ‘terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features’
- ‘an environment defined by specific abiotic and biotic factors, in which the species lives at any stage of its biological cycle’.

But even though habitat quality appears intuitively an attractive concept, providing a scientific, non-subjective assessment of these multidimensional phenomena for conceptual habitat quality is challenging (ICES, 2004). Firstly there are considerations of the parameters to be included and then decisions need to be taken about how this multivariate data set should be combined into a simple measure or index (ie weightings for different parameters). Deeper reflection on the issue of natural variation in habitat quality shows that in fact changes in habitat quality within a habitat will often lead, at some point, to us redefining the habitat into a new class. For example, gravel beds will contain finer sediments, including settled organic matter and silt particles and faecal material produced in situ, in the interstices between the gravel. Variation in this material may be one element of the quality of that gravel habitat. However, at some point the increasing level of fine material would lead to the habitat being considered a ‘muddy-gravel’.

The most logical way of assessing the suitability of an area for a particular species, or life stage, is to measure the density of that species or life stage. Thus the quality of the habitat is a direct reflection of the abundance of the species/stage there, and the concept of habitat quality is thereby reduced to species abundance. It thus provides no additional scientific information or ecological insight beyond that related to the species concerned.

Current measures of habitat quality use indicator species and reference sites to assess the status of habitats. Although the use of indicator species and communities is controversial, and reference sites may not be suitable for areas which have been impacted for centuries, they are a central component of several indices of habitat quality which are currently in use.

**Index of biotic integrity (IBI);** the IBI was introduced by (Karr, *et al.*, 1986) and assesses habitats based on the biological communities they support, with an emphasis on species richness and indicator species. The index compares the assessment sites to a reference site, which is considered relatively un-impacted. The IBI was developed originally for freshwater systems but has evolved to include marine systems. A NOAA workshop was held to develop an IBI for marine benthic and pelagic habitats for the purposes of assessing essential fish habitat and concluded that although IBI was a suitable index to measure benthic habitat quality, it was not suitable for measuring the quality of water column habitats as they were too variable, too dynamic and too transient in quality (Hartwell, 1998). The method devised for measuring marine benthic habitat quality was: 1) categorise the benthic habitat as having soft bottom, hard bottom or live bottom substrates, 2) categorise the area as estuarine (submerged or intertidal), coastal shore zone or offshore, 3) divide the assessment area according to geographical boundaries based on large scale oceanographic and geological features, 4) measure the 'health' of the biological community. The latter was to be assessed by:

- Infauna community structure, composition, number of organisms and biomass by taxa Shellfish, epibenthic fish, benthic foraging fish community structure, composition, number of organisms and biomass by taxa
- Percent spatial extent of 3-D refugia
- Percent spatial extent of living refugia verses total refugia
- Dominance by selected species (opportunistic verses equilibrium)
- Changes in dominance
- Biomass of fish food
- Contaminant impact (eg incidence of disease, dominance of pollution tolerant species). The age structure of selected species (as a measure of physical disturbance/chemical impact)

Specific to estuaries:

- Measures of resident verses migratory species
- Functional parameters of selected species (eg, filtration capacity)

The characteristics of healthy and degraded habitat were identified as:

Degraded	Healthy
Low diversity	High diversity
High dominance by selected species	Low dominance
High proportion of immature individuals	Stable age structure
High proportion of tolerant species	Low proportion of tolerant species
High proportion of r-selected species	High proportion of K-selected species
High chemical body burdens	Low chemical body burdens
High disease/lesion incidence	Low disease/lesion incidence
Low coverage by biological refugia	High coverage by biological refugia

**Organism-sediment index (OSI);** the OSI was introduced by (Rhoads and Germano, 1986) and is more process orientated than the IBI and uses images to record the end products of biological and physical processes that structure benthos (Diaz *et al.*, 2003). Data are collected by sediment profile images to estimate the depth of the apparent colour redox potential (RPD) layer, the successional stages of the macrofauna, the presence of gas bubbles in the sediment (an indication of high rates of methanogenesis), and the presence of reduced sediment at the sediment water interface that would indicate current or recent low dissolved oxygen conditions to assess the quality of the benthic habitat (Diaz *et al.*, 2003).

**Benthic Habitat Quality Index (BHQ);** the BHQ was introduced by (Nilsson and Rosenberg, 1997) and uses sediment surface and sediment profile images to assess sediment characteristics (texture, oxic/anoxic conditions, lamination) which can be related to functional properties of macrofauna (burrows, tubes, feeding voids, reworked sediments) which will give an indication of habitat quality. The BHQ was developed in relation to benthic faunal successional models developed by Pearson and Rosenberg (1976) and OSI (Rhoads and Germano (1986).

**Habitat Affinity Indices (HAI);** the HAI was introduced by (Nelson and Monaco, 1999) and defines habitat affinity based on the relative concentration of a species in a particular habitat compared with the availability of that habitat in the study area. Measurements include dissolved oxygen, temperature, salinity, depth, substrate type, sediment contaminants and toxicity and the size and species present in that area.

**Submerged Aquatic Vegetation (SAV) Habitat Quality Index;** the U.S. Chesapeake Bay restoration program (<http://www.epa.gov/bioindicators/html/marinetidal.html>) has focused on SAV for the Bay grasses, as they require light and suitably low nutrient levels in the water. They have set a goal of providing adequate habitat to 1m depth for SAV. To develop this indicator, Chesapeake Bay Program Bay segments were assessed using 1994 to 1996 data and were scored as passing, failing or borderline for SAV habitat requirements: Secchi depth (a measure of water clarity), dissolved inorganic nitrogen, dissolved inorganic phosphorus, chlorophyll a (a measure of algae), and suspended solids. In some areas only four habitat requirements apply; dissolved inorganic nitrogen habitat requirements do not apply in tidal fresh and oligohaline, or very low salinity, areas. Scores for each segment are a composite based on all applicable habitat requirements. Scores are adjusted to range between 1 and 10 (1 being most degraded, 10 representing the best condition).

All the habitat indices measure the physical, chemical and biological characteristics of habitats and use indicator species, in some part, to assess quality. The use of indicator species is much debated and inconsistencies in the response of species to stressors are well documented (Jones and Kaly, 1996; Linke-Gamerick *et al.*, 2000; Mendez *et al.*, 2000; Forbes *et al.*, 2001; Bustos-Baez and Frid, 2003). A comparison of the IBI and the OSI was conducted in Chesapeake Bay, USA (Diaz *et al.*, 2003). The results showed significant differences in the assessment of habitats as stressed or of good quality. When IBI indicated poor conditions, the OSI tended to indicate good quality habitat. The authors argued that this result was to be expected as the benthic habitat quality (as measured by the OSI), would improve before biotic integrity (as measured IBI). The IBI and HAI are reliant on comparisons to control sites and therefore may not be the most suitable index for areas impacted by human disturbance for several centuries. A review of the above indices concluded there was no theoretical basis for favouring any one index and subjective selection of 'natural' or high quality sites was often required to make the index operational (ICES 2004, 2005b).

Before any progress with habitat indicators can be made, the habitat types to which it is to be applied and their distribution will need to be determined. European habitat types have been classified by EUNIS (<http://eunis.eea.eu.int/habitats.jsp>), which is a hierarchical system that uses both physical descriptors and characterizing species to identify habitat types.

## **2.5 Indicators at the Ecosystem level**

In this section we attempt to provide a comprehensive review of possible ecosystem indicators, but it need to be mentioned that these indicators have a range of properties that may or may not make them suitable for supporting an EBFM. Moreover, many of the indicators are based on model output and thus error in the models may bias any assessment of their relationships with F.

The ecosystem represents the highest hierarchical level of complexity of natural systems, which summarises interactions among all different components, both biotic and abiotic, expressing them in terms of functional processes.

All this could have important implications in a management context, since the lower hierarchical levels appear to be more sensitive to external disturbance/stress than ecosystem processes (Vitousek, 1990), that is ecosystem under stress apparently keeps much of its functions even though species composition changes (Holling, 1992).

Moreover, from the combination between structures and processes can also emerge 'new' features, or emergent properties, not present before in the different components individually, contributing to further increase the system complexity.

All this complexity, characterized by high variability and unpredictability, represents one of the main difficulties in identifying ecosystem indicators. As stated by Rice and Rochet, (2005), in the context of the evaluation of ecosystem effects of fishing, marine ecosystems have so many properties of concern and so few proven general state measures that, at present, there is generally no shortage of proposals for indicators (eg CSAS, 2001; ICES, 2001; Link *et al.*, 2002).



Since the main processes that occur in the ecosystem are production, consumption, and respiration of the system and cycling and transfer of energy, one useful way to better understand relationships among different ecosystem components is the food web analysis. Indeed, trophic interactions result to be the most important interactions between ecosystem components, and a great number of studies have been dedicated to this issue during the past two decades, including the development of trophic flow models.

In spite of this, many ecosystem indicators (eg trophodynamic indicators) resulted to be still descriptive, and reference points have not yet been clearly identified (Mayer et al., 2004). Recent analyses (see Fulton et al., 2005), however, highlighted that, with regard to overall performance and robustness ecosystem-level indicators, together with the community- ones, resulted to be the most informative. Although models only tentatively can mime the real world, they represent a valid tool for capturing, in a coherent manner, ecosystem processes as a whole and giving answers to management questions (Yodzis, 2001; Peck, 2004).

Network and model-dependent indicators are often proposed in theoretical works as useful summaries of system state or system dynamics, because they are commensurate with ecosystem properties such as trends in species richness, resource internalization, trophic specialization, and succession (Christensen, 1995; Ulanowicz and Abarca-Arenas, 1997). But a detailed analyses of the relationships among different ecosystem indicators in terms of redundancy and correlation, in identifying alternate stable states and/or regime shifts in real ecosystems is still lacking, having been explored mainly in a theoretical framework (Pimm, 1980; Loreau, 2001; Fath et al., 2001).

All these indicators need to be tested against agreed criteria (eg Rice and Rochet, 2005) to assess their value in supporting an Ecosystem Approach to Fisheries (see Annex 2), but, as addressed by Jennings (2005), in the EAFM framework, even if till now scarcely or never applied to real systems management, it is necessary to implement and test indicators also for this highest hierarchical level, since managers would benefit from receiving decision tables that describe the expected status of various ecosystem components alongside the expected status of target stocks in relation to reference points.

**The mean Trophic Level (TL);** TL identifies the position of an organism within the food web and it is defined as the number of food interactions (passages) that allow transfer of energy from primary producers and detritus to the given species. Originally defined as integer values (Lindeman, 1942; TL=1 primary producers and detritus, TL=2 herbivores, TL=3 carnivores), it was extended to fractional values for accounting the omnivoricity that characterize living organisms (Odum and Heald, 1975). Therefore, the trophic level of a species is defined as one for primary producers and detritus, and for consumers, as the average trophic level of its preys weighted by their proportion in the diet (Stergiou and Karpouzi, 2002). In accordance with this definition, the Trophic level at the level of the population is a real number ranging from 2 (detritivorous or herbivorous) up to 5 (large top predators, ie tuna and sharks), which is calculated by means of the following formula:

$$TL_j = 1 + \sum_i DC_{ij} \cdot TL_i$$

where the Trophic level of the predator  $j$  ( $TL_j$ ), is calculated as a function of the fraction of the preys  $i$  in its diet ( $DC_{ij}$ ), and their trophic levels ( $TL_i$ ). Likewise the mean Trophic level can be determined at the community or ecosystem level where the latter is a measure of the average number of passages, and gives an idea of the development and complexity of the trophic web. Mean Trophic level can then be calculated as:

$$\overline{TL} = \frac{\sum(TL_{ij} \cdot W_{ij})}{\sum W_{ij}}$$

where  $W_{ij}$  and  $T_{ij}$  are respectively the mass and trophic level of species  $i$  in length class  $j$ . The mean Trophic level of the landings is often used as a proxy of the mean Trophic level at the communities or ecosystem level. For this the mean Trophic level of the catch (Pauly et al., 1998) is estimated by weighting the Trophic level of the caught species ( $TL_j$ ) by their proportion in total landings ( $Y_j/\sum Y_j$ ) instead of the catches ( $W_j/\sum W_j$ ), thus:

$$mTL = \frac{\sum_j Y_j \cdot TL_j}{\sum_j Y_j}$$

Since fishing activities target large individuals with high Trophic level thereby decreasing the mean trophic level resulting in an effect known as Fishing Down the Food Web (Pauly et al., 1998). Therefore the mean trophic level of the catches is widely and efficiently used as an indicator of fishing impact, although some confounding factors and accuracy in estimating species TL need to be evaluated (Pauly and Palomares, 2005).

**The Connectance Index (CI);** CI relates to the food web and is defined as the ratio between the number of actual interactions and the maximum number of the possible interactions that the network can have (Christensen 1995, Christensen et al. 2000). The degree of connectedness is representative of the complexity of the trophic web and a measure of the development of the ecosystem since one expects that trophic structure changes from linear chain (low CI) to web-like trophic interactions (high CI) in developing ecosystems (Odum, 1969). Although there is still need for a clear evaluation, complexity and connectivity of marine trophic web can be influenced and depleted by disturbances and stresses, such as those produced by fishing activities. The CI can be estimated directly on the basis of diet studies or through models of the trophic network: in both cases CI is highly influenced by the detail used to describe the ecosystem (Christensen and Pauly 1993; Christensen 1995).

**System Omnivory Index (SOI);** SOI in order to avoid the dependence of the Connectance Index from the web description (thus from the number of compartments or species described), Christensen et al. (2000) recommend the use of the System Omnivory Index (SOI), which describes the interactions in the food chain through the variance in the trophic levels of the preys for groups of predators (Pauly et al., 1993). Such an index can vary in an interval between 0 and 1: a value close to 0 indicates that consumers of the system are specialized, while values close to 1 indicate that consumers feed on many trophic levels (Christensen and Pauly, 1993; Christensen et

al., 2000). Following theoretical analyses and considerations (Odum, 1969) disturbances should reduce connections and therefore should be expected a reduction of SOI in ecosystem deeply affected by fishery. However this has not proven already and also theoretical and modelling applications on that are resulting as being controversial (Christensen, 1995).

**The Primary Production Required (PPR);** PPR represents the amount of primary production needed to support the secondary production that is exported as catches (Pauly and Christensen, 1995). It is proposed as an indicator quantifying the pressure of the fisheries since it can be easily compared and scaled with Primary Productivity of the system (Pauly and Christensen, 1995; Tudela et al., 2005). PPR can be estimated from ecosystem models through back calculation of the trophic flows that, from a considered species, go down to the primary producers from all possible pathways (Christensen et al., 2000). However, knowing the trophic level of the species, it is possible to have rough estimates of PPR through a formula proposed by Pauly and Christensen (1995):

$$PPR = \frac{1}{9} \cdot \sum_i \left[ Y_i \cdot \left( \frac{1}{TE} \right)^{TL_i-1} \right]$$

thus PPR is estimated using landings for different species ( $Y_i$ ) and their trophic level ( $TL_i$ ), while TE is the average transfer efficiency in the ecosystem and the factor  $1/9$  is the average conversion coefficient from gC to wet weight. PPR can also be estimated at the community level, where it takes, indirectly, the inefficiencies along the trophic web into account that link the given species to the primary producers. Although the PPR has a simple meaning that favour its application, its calculation is subjected to several approximations, the most important of which is the assumption of constant TE and the conversion factors from mass to carbon (Pauly and Christensen, 1995).

**The Fishing in Balance index (FiB);** FiB represents the ratio between the energy required to sustain the fishery landings and a baseline value, and it was proposed in order to assess whether a certain level of exploitation can be sustained by a given marine ecosystem and to detect bottom-up effects (Pauly et al., 2000; Pauly and Palomares, 2005). The FiB is estimated for time series of landings, using the first year as a reference and it takes into account both the landings and their mean Trophic level, which are combined in the following expression:

$$FiB_k = \log \left( \frac{Y_k \cdot \left( \frac{1}{TE} \right)^{mTL_k}}{Y_0 \cdot \left( \frac{1}{TE} \right)^{mTL_0}} \right)$$

in which  $Y_k$  and mean Trophic level<sub>k</sub> represent the total landing and its mean Trophic level for the year k,  $Y_0$  and mean Trophic level<sub>0</sub> the total landing and its mean Trophic level for the first year of the time series. The mean Trophic level is at the exponent of the inverse of the Transfer Efficiency (TE), which is set initially as 10%, found to be the average value by an analysis of a suite of different marine ecosystems (Pauly and

Christensen, 1995). A positive trend in the FiB time series may be caused by an increase in the fishing effort (expanding fisheries) or by an increase in the nutrients availability, which, in turn, leads to an increase in the productivity of the ecosystem and then in the landings (Pauly and Palomares, 2000). Constant values of FiB index over time identify periods during which the fishing pressure and the carrying capacity of the ecosystem have been stable, or that fishing effort has changed accordingly to the changes in the carrying capacity (balanced exploitation, Pauly et al., 2000).

**Production/Respiration;** the Net Primary Production on Total Respiration ratio estimated at ecosystem level is considered an important index of the maturity of ecosystems (Odum 1969, Perez-Espana and Arreguin Sanchez 1999). Such as for growing organisms, mature ecosystems should evidence a steady state, thus production should be balanced by equal amount of respiration. Therefore, values of this ratio close to 1 are indicative of a mature ecosystem, where fixed energy is balanced by the maintenance energetic cost. For the same reason the Net Production of the system, that corresponds to the difference Net Primary Production and Total Respiration, in a mature, stable ecosystem should be balanced: therefore Net Production of the system should be zero in pristine non impacted ecosystems (Odum 1969). However, the suitability of indicators based on total Production and total respiration on respect to fishing impact was evaluated theoretically only (Christensen, 1995) and no definitive conclusions are achieved.

**Finn Cycling Index;** the cycling is considered an important indicator of the ecosystem ability to maintain its structure through positive feedbacks (Ulanowicz 1986, Monaco and Ulanowicz 1997). The cycling level is used as an index of stress (Ulanowicz 1986, Christensen and Pauly 1993) and as an indicator of the maturity stage of the ecosystem (Odum, 1969; Christensen, 1995; Vasconcellos et al., 1997). The cycling should be quantified from experimental measures of flows in the ecosystem, which is prohibitive because time-consuming and highly expensive. However, cycling can be estimated from ecosystem models that estimates flows of the trophic web. These tools provide a comparable estimation of cycling through the Finn Cycling Index (FCI) that represents the fraction of total flows of the system that are cycled (Finn 1976). However, this index is a broad estimate of the cycling including also the 2 step cycling through detritus-detritivores-detritus (through death): when these short processes are dominating, although the ecosystem is not considered at high maturity stage the value of the index is high, therefore Christensen (1995) proposed an index accounting only for cycling through predators.

**The Predatory Cycling Index (PCI);** PCI is calculated similarly to FCI but excluding the cycling through detritus that generally is the dominant part (Christensen 1995). Therefore PCI is usually lower than FCI: very different values of these two cycling indexes indicate predominance in the web of cycling through detritus, while very high PCI (relative to FCI) indicate the prevalence of cycling in the upper part of the trophic web. This last situation is indicative of a mature state since a complex web is required to maintain cycling in the higher part of the trophic pyramid.

**Mean Path Length of cycles** is necessary to consider if such cycling results from short and fast cycles, typical of disturbed systems, or long and slow ones, typical of complex trophic structures (Odum 1969, Kay et al. 1989, Christensen 1995). The

number of groups that a cycling flow crosses is used to quantify the length of the cycles, which is measured through the Finn's Mean Path Length and the Finn's straight-through path length with and without detritus: they provide measures of the average number of links in the cycles and are estimated by trophic network models (Christensen et al., 2000). They are used as maturity indexes, since the cycling is lengthy and slow in mature and complex ecosystems and short and rapid in perturbed ones (Odum, 1969; Christensen, 1995).

**Total System Throughput (TST);** the TST is the total amount of energy flowing through the ecosystem. It is estimated from trophic network models and it is highly dependant on the primary productivity that sustains the ecosystem. However, when normalized by primary production, the TST is indicative of the size of the system in terms of flows and can be used as a measure of the processes and factors that hamper the development of the ecosystem (stress, impacts, degradation, pollution) where high values are indicative of a developed, complex and mature ecosystem (Odum, 1969; Ulanowicz, 1986). Although theoretically significant, there is no direct evidence of effects of fishing on total dimension of ecosystem flows.

**Biomass/Throughput;** the ratio between total Biomass (B) and Throughput (T) is an indicator of the biomass necessary to sustain a unit of energetic flow in the system (Christensen 1995). It can be regarded as a measure of dominance, at ecosystem level, of K- versus r-strategy species: high values of B/T are characteristic of systems dominated by large organisms, while low values are typical of ecosystems dominated by small individuals with high turnover rates. Therefore, the B/T ratio is low in development phases of an ecosystem and increases in value as the system matures and tends to store biomass in its components (Odum 1969; Ulanowicz 1986; Christensen et al., 2000).

**The Transfer Efficiency (TE);** TE summarizes all the inefficiencies due to respiration, excretion, egestion and natural mortality at ecosystem level. Along an idealized linear trophic chain, TE can be estimated for each integer trophic level as the ratio between the production of the actual and the precedent Trophic level (Ulanowicz 1993, Lalli and Parsons, 1993). Disturbances can increase the inefficiencies thus TE is often used as indicative of the state of the ecosystem. Fishing can have two opposite effects on TE: it represents a stress and thus can decrease the TE of the ecosystem through increase of inefficiencies; however the removal of biomass of fished species can also have positive effects, reducing intraspecific competitions and thus increasing productions and TE. Although the balance between these two effects is unclear, there is some evidence that highly fished ecosystems have higher values of TE compared to unfished or lightly fished systems (Jarre-Teichmann et al. 1998, Pranovi et al., 2003; Libralato et al., 2004).

**Emergy;** it is the Embodied Energy that represents the amount of energy needed for the construction of living organisms and biomass, including all the energy needed to sustain metabolic functions (Odum, 1997). Emergy is estimated through back-estimation of the energy (from primary producers and solar energy) needed to build up a biomass and is therefore often reported in terms of solar energy equivalents. The total amount of energy can be estimated by means of the number of passages of the energy from the autotrophs to the given organism and is thus based on the trophic

level of the organisms. Calculation is similar to the estimation of the Primary Production Required (Pauly and Christensen, 1995).

**Exergy**; it is a concept derived from thermodynamics and it represents the amount of work which can be obtained from a system (Fath, 2002). Exergy reflects the quality of the energy: since high quality energy (solar energy or energy in the food) can be transformed efficiently in other forms of work, it has high Exergy content. Thus Exergy represents both the content of energy and its quality, and it is often proposed as a measure that is comprehensive of quantity and informational content embedded within chemicals and organisms (Muller and Leupelt, 1998). The state of reference for the calculation of exergy in biological systems is the ecosystem at the thermodynamic equilibrium: thus the proposed reference is the ecosystem without living forms and with all energy and compounds forming an ‘inorganic, primitive soup’ (Jorgensen et al., 1995). Systems that develop tend to increase the exergy storage through increasing complexity and biomass during development (Jorgensen, 2000), therefore Exergy storage is often used as an indicator of ecosystem maturity, or level of development. Although it is mainly used as a goal function in ecological modelling (Jorgensen and Nielsen, 1998; Fonseca et al., 2000), Exergy is also used as an indicator of stresses like pollution (Bastianoni and Marchettini, 1997; Bastianoni, 1998), eutrophication (Marques et al., 1997) and fishing impacts (Pranovi et al., in press) that may occur in an ecosystem. According to Jorgensen et al.(1995) Exergy is defined as:

$$Ex = RT \cdot \sum_{i=0}^N (C_i \cdot \beta_i)$$

where  $C_i$  is the concentration of the species (i) in the system (biomass per unit of area), and  $\beta_i$  = the weighting coefficients expressing the information contained within the (i) species estimated by means of the genetic information (number of genes). Although this indicator has little potential for a wide public communication, a first evaluation on the effect of fishing on benthic community demonstrate good results that are, however, still preliminary (Pranovi et al., in press).

**Fisher information index (I)**; the application of the information theory to ecological systems goes back to the Shannon-Weaver Index (see section 2.3.2) which combines the concepts of evenness and richness and provides a useful measure of biodiversity. All species are ‘equal’ and weight in the same way irrespective of their order. However, in some situations there is a notion of ordering, and in such cases the probability density takes on a characteristic local ‘shape’. A primary example of such a situation is data linked to temporal dynamics, since time is a naturally ordering variable. Fisher Information is a local measure of dispersion and therefore warrants consideration as a measure of ecological organization. Fisher Information, as developed by Fisher (1922), has been interpreted as a measure of the state of disorder of a system (Frieden, 1998). Fisher Information, I, for a single measurement of one variable is calculated as follows:

$$I = \int \frac{1}{P(\varepsilon)} \cdot \left( \frac{dP(\varepsilon)}{d\varepsilon} \right)^2 d\varepsilon$$

where  $P$  is the probability density function for sampling a particular value of  $\epsilon$ , and  $\epsilon$  is the deviation from the true value of the variable. Eq. (6) is valid for systems of one variable, where the deviation is invariant to the size of the variable being measured. Fisher Information is sensitive to changes in probability distribution shape since it involves a derivative term, and is in these terms a local measure. For highly ordered systems, in which deviations from a particular value of a variable are rare, the probability density  $P(\epsilon)$  is steeply sloped around zero. Such systems have a high Fisher Information. Conversely, for systems in which deviations are likely to be observed, the probability density  $P(\epsilon)$  has a more uniform or ‘unbiased’ shape, and the Fisher Information is correspondingly lower (Fath and Cabezas, 2004).

**Ascendency (A);** the  $A$  has been proposed to characterize the degree of development and maturity of an ecosystem, since it takes in account both the size of the ecosystem in terms of flows (Total system Throughput,  $T$ ) and the organization (through information content,  $I$ ) of the flows (Ulanowicz, 1986). The Capacity ( $C$ ) is the upper limit of  $A$  and their difference is called the System Overhead. According to Ulanowicz (1986), the  $A/C$  ratio is a measure of the system’s maturity as well as its ability to withstand perturbations. Ascendency can be calculated from trophic flows, usually estimated by means of trophic network models (Christensen, 1995; Christian and Luczkovich, 1999). Relative Ascendency ( $A/C$ ) is the fraction of the potential level of organization that is actually realized (Ulanowicz 1986). High values of this index are related to low level of stress in the system and vice versa. Hence disturbing activities, like eutrophication and fishing, should produce a decrease of  $A$  (Wulff and Ulanowicz 1989). For example, the whole system effects of eutrophication could be indexed by using a combination of  $A$  and TST, comparing the change of flows in the whole ecosystem as they change during and after eutrophication (Mann et al. 1989). In fact the eutrophication may be defined at the ecosystem level as an increase in  $A$  due to a rise in TST that more than compensate for a concomitant fall in  $I$  (Ulanowicz, 1986). Internal Relative Ascendency ( $A_i/C_i$ ) represents the balance between the efficiency of carbon flows and the system redundancy and is most suitable to compare different ecosystems, (Mann et al 1989). Environments with a relatively high value of  $A_i/C_i$  have a significant internal stability (Monaco and Ulanowicz, 1997).

**Overhead;** the complement to Ascendency is the System Overhead, which represents the cost to an ecosystem to circulate matter and energy the way it does (Monaco and Ulanowicz 1997). Thus, overhead represents the degrees of freedom a system has at its disposal to react to perturbations (Ulanowicz 1986), and Christensen (1995) showed overhead to correlate better with other model-derived metrics of maturity sensu Odum (1959).

**Mean Path Length (MPL);** mature systems are believed to be characterized by high retention and use of energy in the trophic web, resulting in a ‘slow’ energy flow in mature ecosystems (Odum, 1969). Conversely, stressed or disturbed ecosystems, or even ecosystem during development stages, are characterized by fast and inefficient use of energy. In order to quantify these characteristics the (MPL) was proposed, as a measure to quantify the number of passages that, on average, the energy is subjected to in the ecosystem when passing from primary producers to high trophic levels, eg consumers or top predators (Christensen et al., 2000).

**Mixed Trophic Impact (MTI)** is a measure of the relative impact of a change in the biomass of one component on other components of the ecosystem (Ulanowicz and Puccia, 1990). Through matrix calculations, TI quantifies the net effects of one species (or a fishery) on every other species in a system by accounting as positive effects those of a prey species on its predator (weighted relative to its proportion in the diet), and as negative effects those of a predator on its prey (weighted according to the fraction of the production of a prey that is consumed by the predator). Hence, TI includes direct effects that propagate along the trophic web at all levels, as well as the indirect effects that one species may have on another through trophic interactions. A major drawback, however, is that it is built on static diet compositions, and can therefore be used in sensitivity analyses, but not in projections.

**Ecosystem stability**; the stability of ecosystem is an essential emergent property determining the sustainability of all of its functions and the services they convey to society. Ecosystem stability can be assessed through two traits, the resistance of the ecosystem to external forcing and disturbance, and the resilience of the ecosystem, referring to its capacity to return to the original state after this has been altered by significant disturbance. In graphical terms, the two concepts can be defined as the depth (resistance) and the width (resilience) of an attracting basin (Holling, 1996).

Ecosystem stability is an essential feature to consider in the framework of renewable resources management, since ecosystems that have low resistance and resilience will be particularly vulnerable to external forcing. Vulnerability is the flip side of resilience: when an ecological system loses resilience it becomes vulnerable to change that previously could be absorbed (Kasperson and Kasperson, 2001). Anthropogenic pressures, such as fishing activities, directly and indirectly affecting different ecosystem components, can induce deep changes in functioning, thereby making the system more vulnerable to other disturbances (both natural and human-induced), due to the reduction in resilience. Moreover, once the system has been shifted to another stable state, it is possible that the new state results to be less desirable in terms of services, but more stable in terms of resistance, than the previous one, making quite impossible to recover to the previous situation (Sheffer *et al.*, 2001). The importance of stability in a management context struggles with the absence of appropriate metrics. Indeed, which metrics reflect these ecosystem features, especially in real systems, still remain an unsolved issue. According to Cury *et al.* (2005), a model approach could be used to explore dynamic stability (Vasconcellos *et al.*, 1997; Perez-Espana and Arreguin-Sanchez, 1999). Possible indicators for the above concepts of stability are:

**Persistence**; can be expressed as the time required for biomass change  $> 10\%$

**Resistance**; is equal to  $1/\text{relative B change}$

**Resilience**; resistance/recovery time (to initial state)/models (according also to resilience definition reported by Pimm, 1984).

### 3 PRESSURE INDICATORS

Pressure indicators describe human activities such as fisheries, aquaculture or the discharge of pollutants that may affect the ecosystem. Thus far very little effort has



been directed to the development of pressure indicators for fisheries or aquaculture. Nevertheless, for each topic we will present either an approach to address this issue or a brief introduction to existing efforts.

### **3.1 Fisheries**

WGECO (ICES, 2005) showed a framework to structure pressure indicators of fishing disturbance on the ecosystem but their usefulness and accuracy depended on the amount of information available. When considering this example it should be realized that it specifically applies to a fishery that targets demersal/benthic species or habitats with a mobile gear (trawl or dredge). Nevertheless it is useful to structure our thinking on how pressure indicators can be quantified and what type of information is needed for that. In time such a framework may be developed for other fisheries/métiers as well. According to the example, the ultimate measure of fishing pressure (ie destruction of habitat, mortality of ecosystem components) should, in terms of its units, ideally be related directly to the State indicators (area covered by a specific habitat or abundance of the ecosystem component). In the case of fisheries this sometimes occurs for some ecosystem components when using eg landings and discards data, but often a proxy such as days-at-sea or hours fished must be used. In this section we will explore how the different indicators of fishing pressure are related through their information content and what type of information is necessary to select pressure indicators for the fishery that relate, in terms of their units, to the State indicators. For this, we will present potential pressure indicators at different levels of information content. The accuracy of these indicators, and their suitability as measures of pressure, increases with increasing levels of information (ICES, 2005). In order for these indicators to be used within a management framework the manageable unit should be that of the fishery where métier is used as the subunit that shows a relatively homogeneous impact on the ecosystem and its components. The definition of what can be considered a specific type of fishery (ie métier) is always arbitrary and increasingly smaller sub-units can be created (Figure 1). Every variation in (rigging of) the gear may alter its impact on the ecosystem, so subdividing the fishery many times, thereby creating many subdivisions, potentially allows the most accurate assessment of the ecosystem effects of that fishery. However, if data on effort and on the impact per unit of effort by métier are not available there is no point in subdividing beyond the limitations of the data. For data to be useful they should at least be available on a yearly basis and have a spatial component. The usefulness of data per métier increases with spatial and temporal resolution.

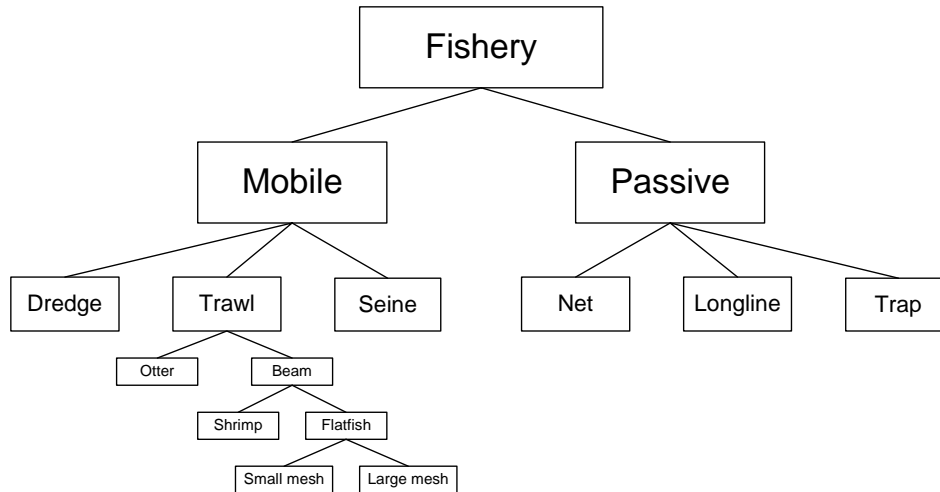


Figure 1 Example of how a fishery can be sub-divided into increasingly smaller métiers

Figure 2 distinguishes five levels at which we can describe fishing pressure. The first level is that of the number of vessels operating in a specific geographic area and effectively describes the capacity. If information exists on the activity of the vessels, for example the time away from port the next level of pressure can be quantified eg effort in days-at-sea. This level of information should be available for most EU fleets of larger vessels. Information on fishing practices and gear characteristics allows the calculation of third level indicators, such as the frequency with which the seabed is swept or a volume of water trawled. The impact of fishing on a habitat, fish- or benthic community is not only determined by the measure of effort (eg days-at-sea or frequency) but also how this effort is distributed within that area, both currently and in relation to the historic distribution of effort. An even distribution of effort will have an overall bigger impact than a patchy distribution where the same amount of effort is concentrated on a relatively small area, leaving the remainder unaffected. Using information on the micro-scale distribution of effort results in the 4<sup>th</sup> level indicators: micro-scale frequency distribution of swept area or volume trawled. Finally, by combining the 4<sup>th</sup> level indices with information on the effects of the gear and the State of the ecosystem components (ie abundance) we reach the highest (5<sup>th</sup>) level indicators which actually give the area of a habitat that is fished or the mortality of an ecosystem component induced by fishing. This is considered the ultimate indicator of fishing pressure as it can be linked directly to a State indicator such as the total area of that habitat or the abundance of that ecosystem component.

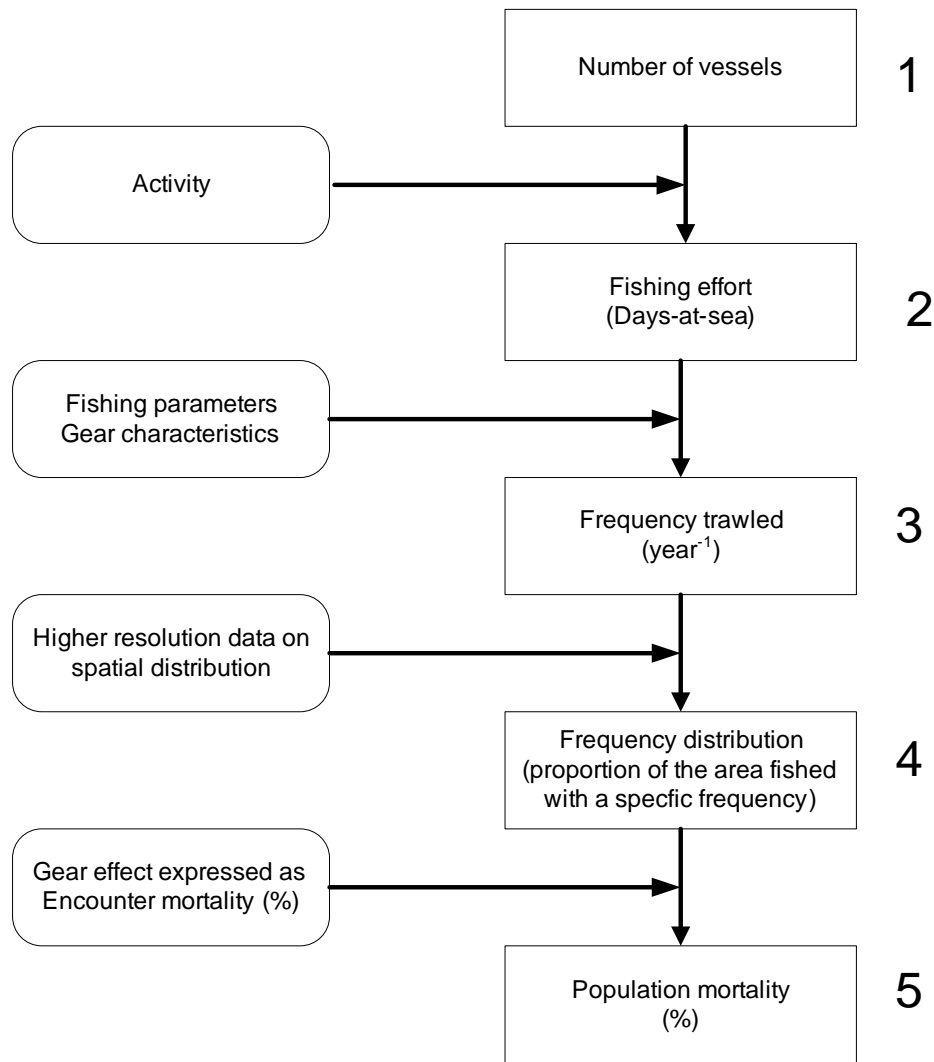


Figure 2 An example framework of fishing pressure indicators at different levels of information content (slightly modified from ICES 2005b).

The boxes on the left describe the type of information required, the level is indicated to the right. Encounter mortality is the % mortality caused by the singular passing of a specific type of gear. Direct mortality is the % mortality of an ecosystem component in an area caused by a known amount of effort of a fishery that operates that gear type.

Following from the above we can distinguish the following pressure indicators for fishing activities

**Fleet capacity;** Number of vessels or total engine power (Hp) or quantity of gear such as miles of longline, number of hooks or gill-net area

**Fishing effort;** This is when capacity becomes operational and can be expressed using various indicators eg Days-at-sea, Hp Days-at-sea, Hours fished or number of hooks set per day or gill-net length-soak time (m.days).

**Frequency with which an area is trawled;** Specific for trawl fishery

**Proportion of the area trawled with a specific frequency;** Here information is available on the spatial micro-distribution in that area.

**Total catch or catch per species**

**Total landings or landings per species**

**Total discards or discards per species;** especially information on protected species is relevant here

**Total fisheries-induced mortality or direct mortality;** see also population indicators for assessed species, section 2.2.2

### **3.2 Aquaculture**

Aquaculture produces effects on marine environment at a range of different scales: Zone A (local), Zone B (delimiting a bio-geographical feature such as an estuary, bay or loch) and Zone C (delimiting the regional scale seawards to a defined hydrodynamic and ecosystem limit). Most aquaculture environment interaction studies take place within Zone A, and it is therefore mainly here that a strong signal may be observed. This would imply that ecosystem degradation may be only locally important and, provided that effects at Zone B and C are minimal, wider ecosystem damage may be avoided. But this is not the case if the local area affected is so degraded that it compromises ecosystem services that are important at wider levels eg nursery areas for fish. There is a natural tension between avoiding local degradation and minimizing wider impacts and this tension is driven by physical processes that disperse wastes (and diseases). Thus, there is a need for indicators of ecosystem change, and modelling tools that predict these indicators, to be operational in a nested sense, to reflect the fundamental physical processes that determine the degree and extent of interactions. Furthermore, local (Zone A) changes may be aggregated through effects on multiple zones within a (Zone B) region, in such a way that the only possibility for sustainable management at the local level is to begin to evaluate the carrying, assimilative and holding capacities for the region as a whole, both in terms of target species and environmental sustainability. Several indicators of the effects of aquaculture on the environment have been proposed, eg Infaunal Trophic Index or AMBI (Borja *et al.*, 2000) but these are usually state indicators and often not well developed across different ecosystem types. The only true pressure indicator for aquaculture that has been put forward is the

**Aquaculture production per area or body of water;** This is a measure of the pressure that aquaculture exerts on the specific areas it occurs in. Depends on space available and assimilative capacity of a body of water.

At present, the development of indicators to quantify the effects of aquaculture activities on the environment is one of the main tasks of a European research Project (ECASA, [www.ecasa.org.uk](http://www.ecasa.org.uk)). Because it has only recently begun there are no results yet.

#### 4 MANAGEMENT- OR ECOREGIONS

DG Environment of the European Commission is in the process of finalising a Thematic Strategy on the Marine Environment under the sixth Environmental Action Programme (Decision 1600/2002). The overall aim of the Marine Thematic Strategy (MTS) is to promote sustainable use of the seas and conserve marine ecosystems. The Commission is expected to formally adopt the MTS during 2005. In March 2005 an open consultation on the MTS was launched which suggests the disclosure of two key documents by the Commission later this year: a Communication on the marine environment and a proposal for a Marine Framework Directive.

One of the most important goals within the MTS is the development of an EAFM for which an accurate definition of the European marine eco-regions is considered essential.

The INDECO project is thus related to this broader marine policy development process, as well as EU fisheries management more specifically, as the project is expected to provide a generic indicator framework that may be used as part of an EAFM and should be applicable to all EU waters. The choice of indicators will, however, differ between regions as this is dependent on the availability of data as well as the way the ecosystem is structured in terms of the abundances of components and the processes that determine its functioning. The choice and definition of regions should therefore be based both on environmental considerations and the unit of decision-making. Some examples of relevant divisions are below.

(Sherman and Alexander 1986) established a system of Large Marine Ecosystems (LMEs) in the world's oceans, of which five are relevant to EU waters and hence fall within the scope of this project. These were apparently based on environmental considerations, but the environmental parameters were not defined and in most non-US parts of the globe appear largely arbitrary (see <http://www.edc.uri.edu/lme/>):

1. Baltic Sea,
2. North Sea,
3. Celtic/Biscay shelf
4. Iberian coastal
5. Mediterranean Sea

Another forum that brought forward a division of EU waters was the new ICES Working Group on Regional Ecosystem Descriptions (WGRED, ICES 2005a), acting as an advising group in the implementation of the EMS. This working group established the following eco-regions (see Figure 3):

- A. Greenland and Iceland Seas,
- B. Barents Sea,
- C. Faroes,
- D. Norwegian Sea,
- E. Celtic Seas,
- F. North Sea,
- G. South European Atlantic Shelf,

- H. Western Mediterranean Sea,
- I. Adriatic-Ionian Seas,
- J. Aegean-Levantine Seas,
- K. Oceanic northeast Atlantic

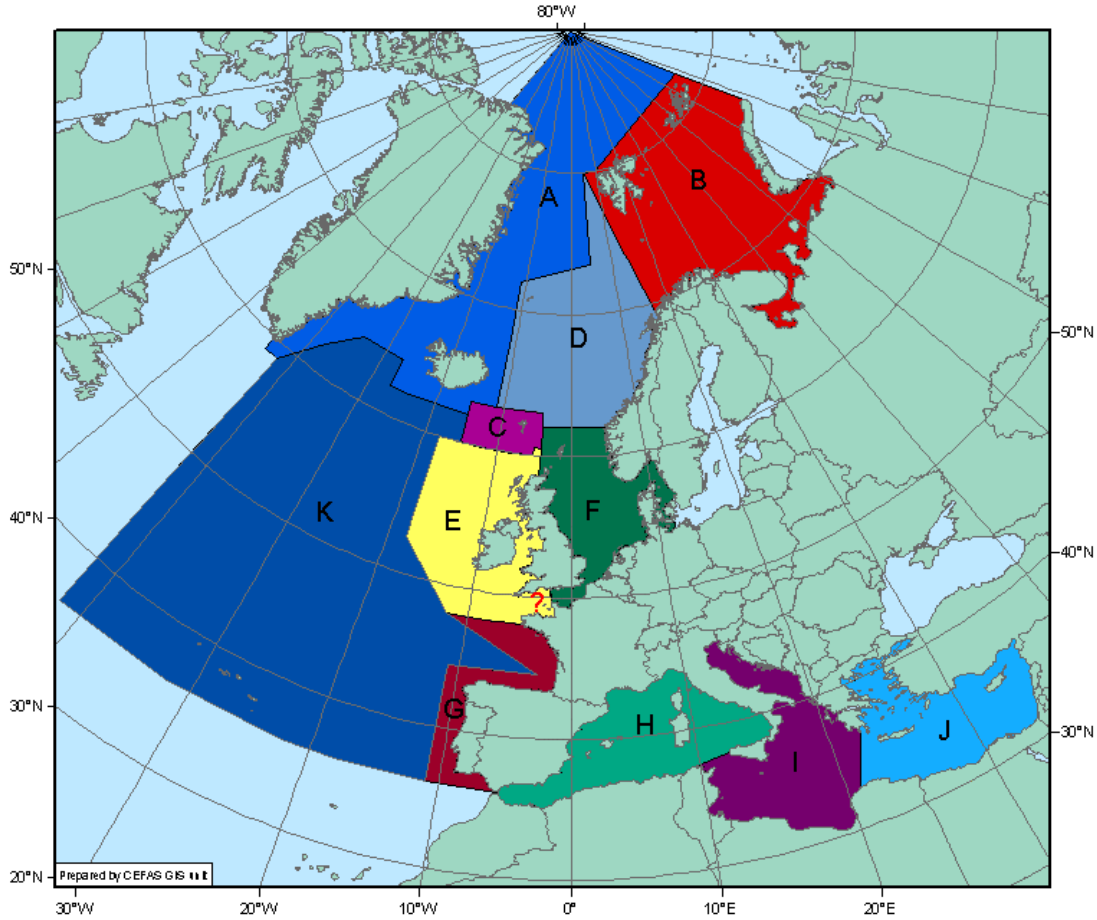


Figure 3 Eco regions according to the ICES Working Group on Regional Ecosystem Descriptions (WGRED, ICES 2005a).

A third set of divisions of EU waters was established by the EU for Regional Advisory Committees (RACs, Council Regulation (EC) No 2371/2002) and is shown in Table 2 and Figure 4.

Table 2 RAC boundaries

Name of the Regional Advisory Council	ICES areas, CECAF divisions and General Fisheries Commission for the Mediterranean
---------------------------------------	--

Baltic Sea	IIIb, IIIc and IIId
Mediterranean Sea	Maritime Waters of the Mediterranean of the East of line 5°36' West
North Sea	IV, IIIa
North Western waters	V (excluding Va and only EC waters in Vb), VI, VII
South Western waters	VIII, IX and X (waters around Azores), and CECAF divisions 34.1.1, 34.1.2 and 34.2.0 (waters around Madeira and the Canary Islands)
Pelagic stocks (blue whiting, mackerel, horse mackerel, herring)	All areas (excluding the Baltic Sea and the Mediterranean Sea)
High seas/long distance fleet	All non EC-waters

Source: Council Decision 2004/585

It should be noted that the differences between the RAC areas and the MTS eco-regions are not large. The main differences are that the whole Mediterranean Sea is covered by the Mediterranean RAC while the area it is split into three MTS eco-regions. The boundaries between the North Sea and North Western Waters RAC, the Arctic and Faeroes Islands Eco-Regions and the Distant Water RAC will not be relevant to INDECO. Within the INDECO project we will use the division into RACs as adopted by the EU as the first basic reporting unit to develop suites of indicators for an EAFM. Because these regions are still very heterogeneous in some cases we may need sets of indicators that apply to geographical subdivisions of these areas.

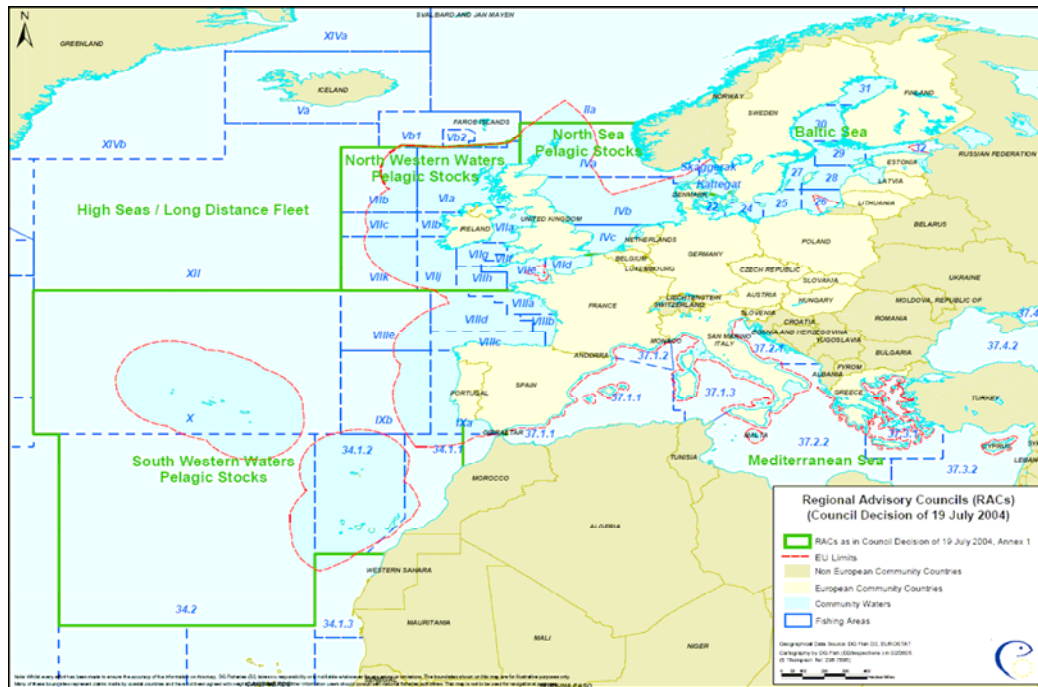


Figure 4 Regional Advisory Councils (RACs).

## 5 DATA SOURCES

The data sources available will be assessed for those management or eco-regions for which there is expertise within the INDECO consortium. These data sources may consist of routine data gathering activities such as trawl- or acoustic fish surveys, benthic grab surveys, seabird colony surveys, seabirds at-sea surveys, or observer schemes to monitor cetacean by-catch in fisheries. Useful information also includes the output of models such as those used in stock assessments (eg Virtual Population Assessment models), or ecosystem models (eg ECOPATH or ERSEM) (Christensen and Walters, 2004).

### 5.1 Baltic Sea

An inventory of routine surveys undertaken in the Baltic Sea, including the Kattegat, is given in Annex 4. Some results of the fish surveys feed into stock assessments carried out routinely by ICES, who also provided the structure for survey co-ordination. Non-fish monitoring is standardised through the HELCOM Monitoring Programme ([www.helcom.fi](http://www.helcom.fi)), and results have routinely been summarized in the HELCOM Periodical Assessments of the State of the Baltic Sea Environment since the 1980s.

### 5.2 North Sea

An inventory of routine surveys that are undertaken in the North Sea to monitor the state of the various ecosystem components is shown in Annex 5.



### **5.3 South-western waters**

An inventory of routine surveys that are undertaken in the bay of Biscay to monitor the state of the various ecosystem components is shown in Annex 6.

### **5.4 Mediterranean**

An inventory of routine surveys that are undertaken in the Mediterranean to monitor the state of the various ecosystem components is shown in Annex 7.

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Annex 1 Definition of size-based indicators, objectives and reference directions of change (RD) under fishing pressure based on theory basis and empirical evidence

(*B*: total biomass; *N*: abundance; *i*: population index; *L*: length; *W*: weight). Empirical evidence refers also to models fitted to observations.

Indicator/ Notation	Description	Units	Objective	RD	Theoretical basis	Empirical evidence
Mean L (W) in community / $\bar{L} (\bar{W})$	$\bar{L} = \sum L / N, \bar{W} = B / N$	cm, mm, (g)	Quantifies relative abundances of large and small individuals (including species composition).	↘	Rochet and Trenkel, 2003	Nicholson and Jennings, 2004; Bellail <i>et al.</i> , 2003; Dulvy <i>et al.</i> , 2004
Mean L (W) in population / $\bar{L}_i (\bar{W}_i)$	$\bar{L}_i = \sum L / N_i, \bar{W}_i = B_i / N_i$	cm, mm, (g)	Quantifies relative abundance of large and small individuals (recruitment).	↘	Beverton and Holt, 1957	Haedrich and Barnes, 1997; Bellail <i>et al.</i> , 2003; Babcock <i>et al.</i> , 1999
Mean L-at-age <i>a</i> in population <i>i</i> / $\bar{L}_{i,a}$	$\bar{L}_{i,a} = \sum L / N_{i,a}$	cm, mm	Reflects size- and age-structure of population, as well as differential growth rates caused by density-dependent effects and environmental conditions	↗	Beverton and Holt, 1957; Parma and Deriso, 1990; Walters and Post, 1993	Rijnsdorp and van Leeuwen, 1996; Overholtz, 1989; Overholtz <i>et al.</i> , 1991; Bowering, 1989; Ross and Almeida, 1986; Shin and Rochet 1998
Mean maximum L in community / $\bar{L}_{max}$	$\bar{L}_{max} = \sum N_i \overline{L_{max,i}} / N$ ( $\overline{L_{max,i}}$ , or alternatively $L_{inf,i}$ , is fixed.)	cm, mm	Quantifies relative abundances of large- and small-sized species.	↘		Jennings <i>et al.</i> , 1999; Nicholson and Jennings, 2004
Maximum L in population <i>i</i> / $L_{max,i}$	Direct observation	cm, mm	Quantifies depletion of large fish within population	↘		
Mean L-at-maturity in population <i>i</i> / $\bar{L}_{mat,i}$	Length at which 50% of the population has attained maturity	cm, mm	Reflects differential growth rates caused by genetic variability, density-dependent effects and environmental conditions	↗ ↘	Hutchings, 1993; Reznick, 1993	Rochet, 1998; Beacham, 1983; de Veen, 1976 Bowering, 1989; Rijnsdorp, 1989, 1993; Beacham, 1983; Hempel, 1978; Rowell, 1993; Olsen <i>et al.</i> , 2004
Fulton's condition index in population <i>i</i> / $K_i$	$K = (W/L^3) \times 100$	10 <sup>2</sup> g.cm <sup>-3</sup>	Reflects overall habitat quality for growth and reproduction.	↗		Winters and Wheeler, 1994
Slope and intercept of L spectra (ls) / slope <sub>ls</sub> int <sub>ls</sub>	Represented in log scales, ls and ws are approximated by decreasing linear functions characterized by their slopes and intercepts.		Quantifies relative abundances of small and large fish and overall productivity of system.	↘ ↗	Gislason and Rice, 1998; Shin and Cury, 2004	Rice and Gislason, 1996; Bianchi <i>et al.</i> , 2000; Gislason and Rice, 1998; Dulvy <i>et al.</i> , 2004

Indicator/ Notation	Description	Units	Objective	RD	Theoretical basis	Empirical evidence
Slope and intercept of W spectra (ws) / slope <sub>ws</sub> int <sub>ws</sub>				↘ ↗		Pope and Knights, 1982; Pope <i>et al.</i> , 1988; Murawski and Idoine, 1992; Duplisea <i>et al.</i> , 1997; Jennings <i>et al.</i> , 2002a
Slope and intercept of size diversity spectra (ds) / slope <sub>ds</sub> int <sub>ds</sub>	Distribution of diversity (eg, Shannon index) against fish size.		Reflects species diversity along energy flow	?		Hall and Greenstreet, 1996; Rice and Gislason, 1996
Proportional and relative stock density / PSD RSD	$PSD = (N_{i \geq \text{quality length}}) / (N_{i \geq \text{stock length}}) \times 100$ $RSD = (N_{i \geq \text{specified length}}) / (N_{i \geq \text{stock length}}) \times 100$		Quantifies proportion of large fish in population	↘	Willis <i>et al.</i> , 1993	

Annex 2 Definition of ecosystem indicators and reference directions of change (RD) under fishing pressure based on theory basis and empirical evidence

<i>metrics</i>		<b>Concreteness</b>	<b>Theoretical basis</b>	<b>Public awareness</b>	<b>Cost</b>	<b>Measurements</b>	<b>Availability of historical data</b>	<b>Sensitivity</b>	<b>Responsiveness</b>	<b>Specificity</b>
Elasmobranch/bony fish ratio		H	H	M	L	M	M	M	M	M
Mean trophic level		H	M	M	M	M	M	M	M	M
Pelagic/Demersal fish ratio		H	H	M	L	M	M	M	M	M
Scavengers		H	M	M	M	M	L	L	L	M
Discard/catch ratio		H	M	M	L	M	M	M	L	M
Connectance Index		L	M	L	M	H	L	M	L	L
System Omnivory Index		L	M	L	M	H	L	M	M	L
Primary Production		H	M	M	L	H	M	M	M	L
Primary Production Required		M-L	M	M	L	H	M	H	M	H
Fishing in balance Index		M	M	L	L	H	M	H	M	H
Respiration/Production			M	L	M	H	L	M	M	L
Finn Cycling Index		L	M	L	M	H	L	M	M	L
Predatory Cycling Index		L	M	L	M	H	L	M	M	L
Production/Biomass ecosystem			M	L	M	H	L	M	M	L
Transfer Efficiency			M	L	M	H	M	H	M	M
Emergy		L	M	L	M	H	L	L	M	L
Total System Throughput (TST)		L	M	L	M	H	M	L	M	L

<i>metrics</i>		<b>Concreteness</b>	<b>Theoretical basis</b>	<b>Public awareness</b>	<b>Cost</b>	<b>Measurements</b>	<b>Availability of historical data</b>	<b>Sensitivity</b>	<b>Responsiveness</b>	<b>Specificity</b>
Throughput (TST)										
Exergy		L	L	L	M	H	L	L	M	L
Fisher information		L	M	L	M	H	L	L	M	L
Ascendency		L	M	L	M	H	M	M	M	L
Overhead		L	M	L	M	H	M	M	M	L
mean path length, MPL		L	M	L	M	H	M	M	M	L
1/relative B change		L	L	M	n/a	L	L	n/a	L	L
resistance/recovery time (to initial state)		L	L	M	n/a	L	L	n/a	L	L
time required for biomass change >10%		L	L	M	n/a	L	L	n/a	L	L
Mixed Trophic Impact		L	M	M	M	H	M	M	M	M

Annex 3 Definition of ecosystem indicators and reference directions of change (RD) under fishing pressure based on theory basis and empirical evidence.

<i>Ecosystem features</i>	<i>Metrics</i>	<i>Data required</i>	<i>RD</i>	<i>References</i>
Diversity	Elasmobranch/bony fish ratio	Landings, survey data	↘	Link, 2005
Trophic interactions	Mean trophic level	Landings, survey data, model	↘	Pauly <i>et al.</i> , 1998; Pauly and Palomares, 2000
	Pelagic/Demersal fish ratio	Landings, survey data	↗	De Leiva <i>et al.</i> , 2000; Caddy, 2000
	Discard/catch ratio	Landings, survey data	↘	Alverson <i>et al.</i> , 1994
	Scavengers	Landings, survey data	↗	Link, 2005
Connectivity	Connectance Index	model	↘	Christensen <i>et al.</i> , 2000
	System Omnivory Index	model	↗	Christensen <i>et al.</i> , 2000
Production	Primary Production Required	Landings, survey data, model	↗	Pauly and Christensen, 1995; Tudela <i>et al.</i> , 2005
	Fishing in balance Index	Landings, survey data	↗	Pauly <i>et al.</i> , 2000
Respiration	Production/Respiration	model	↘	Christensen <i>et al.</i> , 2000
Decomposition	Flows to detritus (?)	Survey data, model	↗?	
	Labile:refractory detritus biomass ratio	Survey data	↗?	Fulton <i>et al.</i> , 2003
Cycling	Finn Cycling Index	model	↗	Finn, 1976; Christensen and Pauly 1992



<i>Ecosystem features</i>	<i>Metrics</i>	<i>Data required</i>	<i>RD</i>	<i>References</i>
	Predatory Cycling Index	model	↘	Christensen and Pauly 1992
	Mean Path Length of cycles	model	↘	Christensen and Pauly 1992
	Total System Throughput (TST)	model	↘	Ulanowicz, 1986; Christensen and Pauly 1992
Turn-over	Biomass/ Throughput	model	↘	
	Production/Biomass ecosystem	model	↘	Odum, 1969; Christensen and Pauly 1992
Efficiency	Transfer Efficiency	Landings, survey data, model	↘	Lalli and Pearson, 1993; Pauly and Christensen, 1995
	Emergy	Landings, survey data	↘	Odum, 1997
Maturity	Exergy	Landings, survey data	↘	Jorgensen <i>et al.</i> , 1995; Jorgensen, 2000
Complexity	Fisher information	model	↘	Fath and Cabezas, 2004
Complexity of flows	Ascendency	model	↘	Ulanowicz, 1986
	Overhead	model	↘	Ulanowicz, 1986
Number of passages (path)	mean path length, MPL	model	↘	Christensen and Pauly 1992
Interaction strength	Mixed Trophic Impact	model	↘ ↗	Ulanowicz and Puccia, 1990; Christensen and Pauly 1992
Resistance	1/relative B change	model	↗?	Cury <i>et al.</i> , 2005

<i>Ecosystem features</i>	<i>Metrics</i>	<i>Data required</i>	<i>RD</i>	<i>References</i>
Resilience	resistance/recovery time (to initial state)	model	✚	Cury <i>et al.</i> , 2005
Persistence	time required for biomass change >10%	model	✚	Cury <i>et al.</i> , 2005

Annex 4 Catalogue of routine surveys undertaken in the Baltic Sea that deliver data that can be used to quantify indicators.

For each of the biotic components it is indicated if the data apply for Population (P) or Community (C) type of indicators, otherwise an 'X' is given. The Sampling Codes link to the next table of this annex where a detailed list of the sampling equipment used and data collected is provided. <sup>1</sup> These population indicators apply only to commercial species

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code	
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles		Physical/ Chemical
Demersal trawl	BITS- Baltic	1 <sup>st</sup> . quarter	1982-present		P,C				X	1
	BITS-Baltic	4 <sup>th</sup> . quarter	1997-present		P,C				X	1
Demersal trawl	BITS- SD 21-23	1 <sup>st</sup> quarter	1996 - present		P,C				X	1
	BITS- SD 21-23	4 <sup>th</sup> quarter	1994-present		PC				X	1
Demersal trawl	BITS- SD24	1. quarter	1994-present		P,C				X	1
	BITS- SD22-24	4. quarter	1981-present		P,C				X	1
Gill-net	COBRA	Summer	1991-present		P,C					2
Acoustic	Kattegat- Skagerrak	Summer	1980s-present		P,C				X	3
	Baltic- International	Autumn	1983-present		P,C				X	3
	Southern Baltic	May	2002-present		P,C				X	3

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles	
Landings	WGBFAS	Year	1974-present		P,C				
	EU-data collection	Year	2001/2-present		P <sup>1</sup>				6
By-Catch	Discards	Year	1996-present		P				7
Salmon smolt	BSTAWG	Summer	1979-present		P <sup>1</sup>				4
Ichthyoplankton	Central Baltic	Spring-summer	1985-present			X		X	5
Remote Sensing	Modis Terra	Year	2005-...			P,C			8
	Algaline	Year	1992-			P,C		X	9
Oceanographic	HELCOM/COMBINE	Year	1979-	P,C		P,C		X	10
Seabird	At sea	Spring	1984-present				P		11
	NOVANA	Winter/Summer	2000-present				P		12
Seal Survey	Grey	Spring	1975-present				P		13
	Ringed	Spring	1975-present				P		13
Harbour porpoise	Observers	Year	from 2005 on?				P		14

Annex 4 continued. Sampling equipment used and type of data collected in the surveys listed in the previous table.

Sampling Code	Sampling Equipment	Data collected
1	Bottom- and combined bottom-pelagic trawls (16 mm mesh in codend): Granton, TV3, HG20/25	Expo, Abundance, length, weight, age, maturity. Uncoordinated surveys (national) since 1960s.
2	Multimesh gill-net, gill net series	Relative abundance of coastal warm-water fish species, growth, age/length of maturity
3	38 kHz, 120 kHz, EY500; Trawls: Expo	Fotø; Abundance/biomass, length, weight, age, maturity
4	Smolt trapping, electrofishing	Annual natural salmon smolt production, all Baltic countries included
5	Bongo: 500 µm, 300µm mesh, BabyBongo 150 µm mesh; BioMoc/Multinet 0.5m <sup>2</sup> , 355 µm mesh. IKS-80	Fish eggs and larvae; database can partly be extended back in time to 1954s.
6	Sampling from commercial catches	Numbers commercial fish at length in landings, weight- and age-length keys.
7	Observers onboard commercial vessels	EU/ICES-coordinated. Persistent problem: unreported landings. New EU-regulation expected to solve historic problem with pelagic fisheries catch compositions.
8	Terra Modis satellite	Remotely sensed daily chl-a maps over the Baltic Sea
9	Water sampler on 9 ferry routes	Nutrients, turbidity, algal biomass, blue-green algae
10	Water samples, plankton samplers, 0.1 m <sup>2</sup> van Veen grab for macrobenthos	Hydrography (temperature, salinity, oxygen), nutrients, distribution and abundance of zoobenthos and plankton. Areas with oxygen deficiency. HELCOM co-ordinated national surveys, plus some national extensions of international sampling programme, eg, Latvian zooplankton data series.
11	Binoculars/ Telescope	Counts of breeding seabirds/occupied nests, records of egg/chick production
11	Binoculars/ Telescope	Bird numbers in Danish waters by species, sampling-based survey every 3 <sup>rd</sup> year in mid-winter, and every 7 <sup>th</sup> year in summer during moult.

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Sampling Code	Sampling Equipment	Data collected
		Data series can be extended back to 1960s (partly annual, even seasonal observations), but these series are based on a different survey-design. Annual observations during 1980s and 1990s.
12	Aerial survey, observation from boats	Minimum population size estimates. Data from different countries combined
13	Observers on commercial vessels	Numbers, individual information.

Annex 5 Catalogue of routine surveys undertaken in the North Sea that deliver data that can be used to quantify indicators.

For each of the biotic components it is indicated if the data apply for Population (P) or Community (C) type of indicators, otherwise an 'X' is given. The Sampling Codes link to the next table of this annex where a detailed list of the sampling equipment used and data collected is provided. <sup>1</sup> These population indicators apply only to commercial species

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code	
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles		Physical/ Chemical
Demersal trawl	Q1 IBTS	Winter	1971-present	P,C	P,C	P,C			X	1,6,7
	Q3 IBTS	Summer	1991-present	P,C	P,C	P,C			X	1,6
	DBTS	Summer	1985-present	P,C	P,C				X	2
	SNS	Summer	1969–2003	P,C	P,C				X	3
	DFS	Summer	1970-present	P,C	P,C				X	4
	EBTS	Summer	1988-present	P,C	P,C				X	5
	2mBTS	Summer	1999-present	P,C	P,C					16
Acoustic	Herring	Summer	1992-present		P <sup>1</sup>	P,C				8,6
	Mackerel	Autumn	2000-present		P <sup>1</sup>	P,C				8,6
Ichthyoplankton	Cod/Plaice	Spring	-present			P,C				9,6
	Mackerel	Spr/Sum	-present			P,C				10,6
Landings	Markets	Year	1963-present	P <sup>1</sup>	P <sup>1</sup>					11
By-Catch	Discards	Year	1976-present		P					12
Remote Sensing	MERIS	Year								
	ENVISAT	Year								

Survey Type	Name	Season	Time series	Ecosystem components				Sampling Code	
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds		Marine mammals/reptiles
Oceanographic	Ferry Box	Year				P,C		X	13
	CPR	Year	1930-present			P,C		X	14
Benthic Grab	Infauna	Summer	1986 and 2000	P,C					15
	Colonies	Spring	1970, 1985-present				P		17
Seabird	Beached	Year	1975-present				P		18
	At Sea	Year	1980-present				P		19
Seal Survey	Grey	Summer	1970s/1980s-present					P	20
	Common	Summer	1970s/1980s-present					P	21
Cetaceans	Beached	year	Intermittent since 1930s					P	18
	Observer	Year	Some from 1997 onwards					P	22
	abundance	Summer	1994 and 2005					P	23



Annex 5 continued. Sampling equipment used and type of data collected in the surveys listed in the previous table.

Sampling Code	Sampling Equipment	Data collected
1	GOV demersal trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships
2	8m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
3	6m sole beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
4	3m beam trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
5	4m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
6	CTD profiler	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples
7	Plankton sampler, MIK or Isaak Kidd	Numbers of herring and sprat larvae (at length)
8	EK500, Pelagic Trawl	Acoustic echo integration, numbers fish at length in catch, weight- and age-length relationships
9	Gulf VII (Gulf III/Bongo) 40cm diameter 270µm mesh, 100-200m <sup>3</sup> filtered	Number of cod and plaice eggs per volume filtered
10	Gulf III (20cm)/Bongo (60cm), 250-280µm mesh, water column oblique tow	Number of mackerel and horse mackerel eggs per volume filtered
11	Landed catch sampled	Numbers commercial fish at length in landings, weight- and age-length relationships
12	Catch brought aboard sampled	Effort and Catch per trawl quantified, related to vessel and location. Numbers commercial fish at length in catch, age-length relationships. Proportion catch at length discarded recorded
13	CTD profiler, water sampler on 9 ferry routes	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, transmissometry, nutrients, pH, O <sub>2</sub> , algal composition
14	Standard CPR body with Autonomous Plankton Sampler	Geo-referenced 'snapshots' of filtered plankton collected on silk band filter, 'greenness' index
15	0.1m <sup>2</sup> van Veen/Day Grabs/0.1 m <sup>2</sup> or 0.25 m <sup>2</sup> Box Core	Abundance of benthic invertebrate species. Sediment particle size distribution, organic content.

Sampling Code	Sampling Equipment	Data collected
16	2m Epibenthic beam trawl	Numbers at length benthic invertebrate infauna and epibenthos, weight at length relationships, mean individual weight for productivity
17	Seabird colony visits, Binoculars/ Telescope	Counts of breeding seabirds/occupied nests, records of egg/chick production
18	Coastline survey	Location and counts of dead seabirds/seals/cetaceans by species. Recovery of bird leg-rings.
19	Vessel or Aircraft based survey	Geo-referenced counts of seabirds in transects of fixed width, abundance/distribution estimates.
20	Aerial Survey/Population Modelling	Counts of grey seal pup production, models used to estimate population size that would give rise to observed number of pups
21	Aerial Survey	Count of numbers of moulting harbour seals – minimum population size estimate
22	Fisheries Bycatch Observer schemes	Counts of numbers of marine mammals taken by different fisheries related to effort.
23	Cetacean monitoring	Counts of numbers of marine mammals

Annex 6 Catalogue of routine surveys undertaken in the Bay of Biscay that deliver data that can be used to quantify indicators for different Ecosystem components (see Table 1).

For each of the biotic components it is indicated if the data apply for Population (P) or Community (C) type of indicators, otherwise an “X” is given. The Sampling Codes link to next table in this annex where a detailed list of the sampling equipment used and data collected is provided

Survey Type	Name	Season	Time series	Ecosystem components						Sampling Code
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles	Physical/ Chemical	
Demersal trawl	Q1 IBTS	Winter	1971-present	P,C	P,C	P,C			X	1,6,7
	EBTS	Autumn	1988-present	P,C	P,C				X	5
	RESSGASCS	???	1987-2002		P					5
	EVHOE	???	1997-present		P,C					5
	Epifauna	Summer/aut	1986-present	P,C	P,C					16
Acoustic	Mackerel	Autumn	-present		P <sup>1</sup>	P,C				8,6
	Anchovy	Spring/autu	1987-present		P	P, C				6,7
	Sardine		2003-present		P					7
Ichthyoplankton	Anchovy	Spring	1987-present		P	P,C			X	9,6
	Mackerel and Horse mack.	Triannual	1986-present			P,C			X	10,6
	Sardine	Triannual	1986-present			P, C			X	10, 6
Landings	Markets	Year	1953-present	P <sup>1</sup>	P <sup>1</sup>					11
By-Catch	Discards	Year	1988-present		P					12
Remote Sensing	AVHRR	Year	1987-present						X	
	MODIS	Year	2004-present			X			X	
	SEAWIFS	Year	1998-2004			X				

Survey Type	Name	Season	Time series	Ecosystem components					Physical/ Chemical	Sampling Code
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles		
Oceanographic	Ferry Box	Year	2002-present						X	13
	CPR	Year	1951-present			P,C			X	14
	Standard series	Year	1986-present			P,C			X	9, 6
Benthic Grab	Infauna	Winter	1995-present	P,C					X	15
Seabird	Colonies	Spring	1996-present				P			17
	Beached	Winter	1990-present				P			18
	At Sea (ferry)	Year	1995-present				P			19
Cetaceans	Beached	year	1901-present					P		18
	At Sea (ferry)	Year	1995-present					P		19
	Observer	Year	2004-present					P		22
	SCANS	Summer	2005					P		23

<sup>1</sup> These population indicators apply only to commercial species

Annex 6 continued. Sampling equipment used and type of data collected in the surveys listed in the previous table of this annex.

Sampling Code	Sampling Equipment	Data collected
1	GOV demersal trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships
2	8m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
3	6m sole beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
4	3m beam trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
5	4m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
6	CTD profiler	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples
7	Plankton sampler, MIK or Isaak Kidd	Numbers of herring and sprat larvae (at length)
8	EK500, Pelagic Trawl	Acoustic echo integration, numbers fish at length in catch, weight- and age-length relationships
9	Gulf VII (Gulf III/Bongo) 40cm diameter 270µm mesh, 100-200m <sup>3</sup> filtered	Number of cod and plaice eggs per volume filtered
10	Gulf III (20cm)/Bongo (60cm), 250-280µm mesh, water column oblique tow	Number of mackerel and horse mackerel eggs per volume filtered
11	Landed catch sampled	Numbers commercial fish at length in landings, weight- and age-length relationships
12	Catch brought aboard sampled	Effort and Catch per trawl quantified, related to vessel and location. Numbers commercial fish at length in catch, age-length relationships. Proportion catch at length discarded recorded
13	CTD profiler, water sampler on 9 ferry routes	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, transmissometry, nutrients, pH, O <sub>2</sub> , algal composition
14	Standard CPR body with Autonomous Plankton Sampler	Geo-referenced “snapshots” of filtered plankton collected on silk band filter, “greenness” index
15	0.1m <sup>2</sup> van Veen/ Box Core	Abundance of benthic invertebrate species. Sediment particle size distribution, organic content, pollutants
16	2m Epibenthic beam trawl	Numbers at length benthic invertebrate infauna and epibenthos, weight at length relationships, mean

Sampling Code	Sampling Equipment	Data collected
		individual weight for productivity
17	Seabird colony visits, Binoculars/ Telescope	Counts of breeding seabirds/occupied nests, records of egg/chick production
18	Coastline survey	Location and counts of dead seabirds/seals/cetaceans by species. Recovery of bird leg-rings.
19	Vessel or Aircraft based survey	Geo-referenced counts of seabirds in transects of fixed width, abundance/distribution estimates.
20	Aerial Survey/Population Modelling	Counts of grey seal pup production, models used to estimate population size that would give rise to observed number of pups
21	Aerial Survey	Count of numbers of moulting harbour seals – minimum population size estimate
22	Fisheries Bycatch Observer schemes	Counts of numbers of marine mammals taken by different fisheries related to effort.

Annex 7 Catalogue of routine surveys undertaken in the Mediterranean Sea.

(H, I, J refer to the three Mediterranean eco-regions proposed by the new ICES Working Group on Regional Ecosystem Descriptions) that deliver data that can be used to quantify indicators for different Ecosystem components (see Table 1). For each of the biotic components it is indicated if the data apply for Population (P) or Community (C) type of indicators, otherwise an 'X' is given. The Sampling Codes link to the next table of this annex where a detailed list of the sampling equipment used and data collected is provided

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles	
	ION-PATR (J)	Seasonal	1983-1985		P,C				1
	EUB-PAG (J)	Seasonal	1986-1988		P,C				1
	N-AEGEAN (J)	Seasonal	1990-1991		P,C				1
	THERM-THRA (J)	Seasonal	1991-1993		P,C				1,6
	TRIBE (J)	Monthly	1995-1996	C	P,C			X	1,15
	FGE 1 (Pagasisitikos) (J)	2 seasons	1994	C	P,C			X	1.6, 15.16
	FGE 1 (Gouves) (J)	seasonal	1995-1996	C	X			X	6.15.16
	OTIP (Gouves) (J)	seasonal	1999-2000	C	(only 2000)			X	1, 6.15.16
	OTIP (Dia) (J)	seasonal	1999-2000	C				X	6.15.16
Demersal trawl	COST-IMPACT (Dia fishing ground) (J)	2 seasons	2002, 2003	C				X	6.15.16

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code	
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles		Physical/ Chemical
	AQCESS (Evia, Xios, Lesvos) (J)	2 seasons	2001, 2002	C	P,C				X	1, 6.15.16
	CYC-DOD (J)	Seasonal	1995-1996		P,C					1
	DEEP-F (J)	Monthly	1996-1997		P,C					1
	PAGAS (J)	Seasonal	1999-2000		P,C					1
	INTER (J)	Seasonal	1999-2000		P,C					1
	RESHIO (J)	Seasonal	2000-2001		P,C					1
	ART-R (J)	Seasonal	2004-2005		P,C					1,6
	MEDITS (J)	Summer	1994-present		P,C					1,6
	GRUND (H,I)	Autumn	1985-present	P,C	P,C				X	23, 25
	MEDITS (H, I)	Spring	1994-present	P,C	P,C				X	24, 25
	Clam (I)	June	1984-2001	P <sup>1</sup>						38
Acoustic	FISH-POP (J)		1986-1989							
	FISH-POP II (J)		1989-1991							
	BIOMASS (J)		1991-1993							
	STRIDE-153 (J)		1991-1994							



Survey Type	Name	Season	Time series	Ecosystem components						
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles	Physical/ Chemical	Sampling Code
Ichthyoplankton	ACOUSTICS/AIR (J)		1995-1998							
	AVITIS/FAIR (J)		1997-2001							
	NATIONAL (J)	Summer	2003-present							
	NATIONAL (J)	Summer	2003-present							
	ANCHEVA (I)	Summer	1998-present		P <sup>1</sup>					26
	JUVENILE (I)	Autumn	1998-present		P <sup>1</sup>					26
	Anchovy (I)	year	1975-present		P <sup>1</sup>					36
	Sardine (I)	year	1975-present		P <sup>1</sup>					36
	ANCHOVY-THRA (J)	Seasonal	1999-2000							
	PAGAS-I (J)	Seasonal	1999-2000							
ANSIC (I)	Summer	1998-present				P,C		X	27, 31	
Anchovy (I)	June-July	1976-present				P <sup>1</sup>			37	
Sardine (I)	Nov-March	1976-present				P <sup>1</sup>			37	
Landings	MEDLAND (J)	Seasonal	1998-2000							

Survey Type	Name	Season	Time series	Ecosystem components					Sampling Code
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptiles	
By-Catch	PS-LAND (J)	Monthly	2000-2002						
	NATIONAL (J)		2003-present						
	ESDAP (J)	Monthly	1995-2002						
	NATIONAL (J)		2003-present						
	Campbiol (H,I)	Seasonally	2002-present	P <sup>1</sup>	P <sup>1</sup>				28
By-Catch	Discards I (J)		1997-2000						
	NATIO-S-DISC (J)	Seasonal	2001						
	NATIONAL-DISC (J)		2003-present						
	Discard (I)	Seasonally	2003; 2005		P				29
Large Pelagic	BLUEFIN-F (J)		1999-2001						
	Swordfish (J)	Monthly	1998-2002						
	Observer (J)	Year							
	SCANS (J)	Summer	1994 and 2005						

Survey Type	Name	Season	Time series	Ecosystem components						
				Benthos/ Macrophytes	Fish/ Cephalopods	Plankton	Seabirds	Marine mammals/reptile s	Physical/ Chemical	Samplin g Code
	ANSIC (I)	Summer	1998-present						X	30
	MAT (I)	Monthly	1999-2002			P,C			X	34
Oceanographic	INTERREG2 and 3 (I)	Year	1999-present			P,C		X	X	35
Cetaceans	(I)									32, 33

<sup>1</sup> These population indicators apply only to commercial species

Annex 7 continued. Sampling equipment used and type of data collected in the surveys listed in the previous table of this annex.

Sampling Code	Sampling Equipment	Data collected
1	GOV demersal trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships
2	8m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
3	6m sole beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
4	3m beam trawl, 20mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
5	4m beam trawl, 40mm codend	Numbers fish at length in catch, weight- and age-length relationships, no. and weight benthos
6	CTD profiler	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples
7	Plankton sampler, MIK or Isaak Kidd	Numbers of herring and sprat larvae (at length)
8	EK500, Pelagic Trawl	Acoustic echo integration, numbers fish at length in catch, weight- and age-length relationships
9	Gulf VII (Gulf III/Bongo) 40cm diameter 270µm mesh, 100-200m <sup>3</sup> filtered	Number of cod and plaice eggs per volume filtered
10	Gulf III (20cm)/Bongo (60cm), 250-280µm mesh, water column oblique tow	Number of mackerel and horse mackerel eggs per volume filtered
11	Landed catch sampled	Numbers commercial fish at length in landings, weight- and age-length relationships
12	Catch brought aboard sampled	Effort and Catch per trawl quantified, related to vessel and location. Numbers commercial fish at length in catch, age-length relationships. Proportion catch at length discarded recorded
13	CTD profiler, water sampler on 9 ferry routes	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, transmissometry, nutrients, pH, O <sub>2</sub> , algal composition
14	Standard CPR body with Autonomous Plankton Sampler	Geo-referenced 'snapshots' of filtered plankton collected on silk band filter, 'greenness' index
15	0.1m <sup>2</sup> van Veen/Day Grabs/0.1 m <sup>2</sup> or 0.25 m <sup>2</sup> Box Core	Abundance of benthic invertebrate species. Sediment particle size distribution, organic content.

Sampling Code	Sampling Equipment	Data collected
16	2m Epibenthic beam trawl	Numbers at length benthic invertebrate infauna and epibenthos, weight at length relationships, mean individual weight for productivity
17	Seabird colony visits, Binoculars/ Telescope	Counts of breeding seabirds/occupied nests, records of egg/chick production
18	Coastline survey	Location and counts of dead seabirds/seals/cetaceans by species. Recovery of bird leg-rings.
19	Vessel or Aircraft based survey	Geo-referenced counts of seabirds in transects of fixed width, abundance/distribution estimates.
20	Aerial Survey/Population Modelling	Counts of grey seal pup production, models used to estimate population size that would give rise to observed number of pups
21	Aerial Survey	Count of numbers of moulting harbour seals – minimum population size estimate
22	Fisheries Bycatch Observer schemes	Counts of numbers of marine mammals taken by different fisheries related to effort.
23	Otter trawl with low vertical opening, 20/28 mm codend.	Abundance (biomass and number) by taxa. Length, weight, sex maturity and age by target species;
24	GOV demersal trawl, 20mm codend	Abundance (biomass and number) by taxa; length, sex and maturity by target species.
25	CTD	Sea surface and sea bottom Temperature (starting from 2000).
26	EK500, Pelagic Trawl	Acoustic echo integration, numbers fish at length in catch, weight- and age-length relationships
27	Bongo 40cm diameter 200µm mesh oblique tow	Number of anchovy ( <i>Engraulis encrasicolus</i> ) and gilt sardine ( <i>Sardinella aurita</i> ) eggs and fish larvae per volume filtered
28	Landed catch sampled	Numbers commercial fish at length in landings, weight, maturity and age-length relationships.
29	Catch brought aboard sampled	Effort and Catch per trawl quantified, related to vessel and location. Numbers commercial fish at length in catch, age-length relationships. Proportion catch at length discarded recorded

Sampling Code	Sampling Equipment	Data collected
30	CTD profiler	Temperature and salinity at 1m depth intervals
31	Standard CPR body with Autonomous Plankton Sampler	Geo-referenced 'snapshots' of filtered plankton collected on silk band filter, 'greenness' index
32	Coastline survey	Location and counts of dead cetaceans by species.
33	Fisheries Bycatch Observer schemes	Counts of numbers of marine mammals taken by different fisheries related to effort.
34	CTD profiler, water sampling	Temperature and salinity at 0.5 m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, turbidity, nutrients, pH, O <sub>2</sub> , phytoplankton and zooplankton, primary production, bacterial carbon production, enzymatic activities, dissolved and particulate organic carbon
35	CTD profiler, water sampling	Temperature and salinity at 0.5m or 1m depth intervals, calibration salinity samples, fluorometry and chlorophyll calibration samples, turbidity, nutrients, pH, O <sub>2</sub> , phytoplankton and zooplankton, primary production, dolphins count
36		Landings by main fishing ports, population structure by age and size, acoustic echo integration
37	Bongo 20, 236-335 micron	Number of eggs and larvae/m <sup>2</sup>
38	Hydraulic dredge	Abundance/Biomass of commercial bivalves (1 m depth intervals between 3 and 10 m)