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Performance of indicators for ecosystem structure and functioning against screening criteria

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Summary

This report provides the background information for the scoring of indicators against the criteria described in Rice and Rochet (2005) and the weighting of those criteria.
Therefore we will address the issue of operational objectives and present some of that background information. In doing this we attempted to collate information from several of the EU ecosystems. These ecosystems may differ in terms of their biological features, physico-chemical characteristics, type and level of exploitation, data availability or other factors. As these differences may affect the scoring and choice of indicators, a brief description of these ecosystems and their exploitation is provided. The choice of ecosystems was concomitant with the RAC areas.

The criteria for which this document aims to provide background information are: concreteness, theoretical basis, public awareness, cost, measurement, historical data, sensitivity, responsiveness, specificity. Below examples of how various indicators perform against these criteria will be presented and discussed.

For guidance on the criteria: **concreteness, public awareness** and **cost**, only little information was available. Most of the information on **theoretical basis** was provided in the first INDECO report.

Extensive information was available on the availability of **historical data** for indicators of both state and pressure. For state indicators we distinguished different ecosystem features (table 4.1) that need to be conserved in order for the whole ecosystem to be in a healthy state and attempted for each of those features to show time-series of potential indicators that reveal the information available.
Pertaining to the availability of historical data the biotic component for which most data are available is that of fish, both at the level of the population as well as the community. For fish populations there are broadly three categories of indicators: abundance, biological characteristics and genetic composition and we distinguish between the commercial species and the non-assessed species. For abundance indicators there is broad agreement on the indicator for the commercial species and historical data exist in all ecosystems. Only in the Mediterranean there are issues pertaining to the consistency of the data. For the abundance of the non-assessed species, two groups of indicators can be distinguished: abundance in numbers or weight of a suite of selected species or the decline indicator based on IUCN decline criteria. Both groups of indicators depend on Research Vessel monitoring programmes which exist in all European ecosystems and hence historical data are available. The most common indicators on biological characteristics often describe changes in age- or size structure where the former can only be determined for the assessed species while the latter can be determined for all species. The same level of availability of historical data applies for these indicators as for the abundance-type of indicators.

Because existing methods for genetic analyses of diversity such as microsatellite techniques or RAPDs are costly there are hardly any data available on the genetic composition of fish populations. However, a method has recently been developed that calculates an indicator for “genetic effects” for immediate application which requires the same type of data needed to age species. Therefore availability of historical data for this indicator is comparable to that of the other indicators for assessed species. The only difference may come from the requirements of this method on the number of individuals that need to be sampled.

Fish community indicators usually depend on the same type of Research Vessel monitoring programmes on which the non-assessed fish population indicators are based and hence the same level of availability of historical data applies. In these programmes fish need to be measured for the size-structure indicators, and the species identified for the species composition.

For the other biotic ecosystem components (e.g. seabirds, marine mammals or –reptiles, benthos or habitat) considerably less data are available and often there are no indicators, other than an abundance index for selected species, developed yet. Often the indicator is supposed to address the "sensitive" part of the component (e.g. sensitive benthic species, sensitive habitats) without any specification of how sensitive is defined.

For ecosystem functioning several indicators have been put forward that may differ pertaining to the availability of data as some are based on model output, others on more conventional type of data.
Physical/chemical features as well as the Plankton will not be directly affected by the fishery but may be of relevance in explaining (part of) the variation in those features of the ecosystem that may be affected by the fishery. Historical data of these features exist for most ecosystems but notably for the latter availability of time-series may be an issue as much of the scientific community traditionally involved in fisheries science does not have direct access to such data and only few regular monitoring programmes exist.

The ultimate indicator for pressure is the fishing-induced proportion mortality per time of a specific ecosystem component (e.g. commercial fish, benthic invertebrate or marine mammal). This type of information, however, is usually only available for commercial fish species. For all other ecosystem components, indicators such as effort per métier or fleet capacity are used as proxies. Data of the least informative indicator, fleet capacity, are available for all Ecosystems. For the more informative indicator, fishing effort, historical data are often incomplete, inconsistent or not available for all métiers and countries.

Different features of **measurement** are distinguished: variance and bias. For variance examples are given that show the variance within a measurement in time (e.g. based on differences between sampling points) and the variance between years. A high variance between years makes it more difficult to use the indicator for management purposes as it increases the time needed to determine whether or not a particular measure actually resulted in the expected change. Power analysis is used to show how the variance of several indicators determines their use to evaluate management measures.

Several studies exist that describe the **sensitivity** of indicators to fishing pressure. These studies cover different ecosystems and show at least for a number of fish community-, or population level indicators that both the actual value as well as the trend of the indicators may be sensitive to fishing.

An analysis of the **responsiveness** of state and pressure indicators showed that pressure indicators responded almost immediately to a management measure that was intended to reduce effort in a particular area while responsiveness of most state indicators was considered poor.

The Baltic Sea which is known to be heavily affected by environmental factors is used to provide examples of how indicators may be affected by factors other than fishing. This also exemplifies that in such ecosystems where **specificity** of indicators is relatively low, indicators that reflect these environmental factors but are not affected by fisheries themselves may be necessary.

Based on these examples and knowledge of the different ecosystems a suite of indicators is selected that will be formally tested against the above criteria to determine their suitability as part of an Ecosystem Approach to Fisheries Management.

1 Introduction

This report is intended to be the basis for the final phase within Work-packages 2, 3 and 4: the process of selecting indicators for ecosystem-based fisheries management in European waters. This selection process will be based on a framework that was developed specifically for the objective selection of a suite of indicators for use in fisheries management (Rice and Rochet 2005). The framework encompasses eight steps, and provides guidance on pitfalls to be avoided at each step. Step 1 identifies user groups and their needs, featuring the setting of operational objectives, and Step 2 identifies a corresponding list of candidate indicators. Step 3 assigns weights to nine screening criteria for the candidate indicators: concreteness, theoretical basis, public awareness, cost, measurement, historic data, sensitivity, responsiveness, and specificity. Step 4 scores the indicators against the criteria, and Step 5 summarizes the results. Steps 3-5 offer technical aspects on which guidance is provided, including scoring standards for criteria and a generalized method for applying the standards when scoring individual indicators. Multi-criterion summarization methods are recommended for most applications. Steps 6 and 7 are concerned with deciding how many indicators are needed, and making the final selection of complementary suites of indicators. Ordinarily, these steps are done interactively with the users of the indicators, thus providing guidance on process rather than technical approach. Step 8 is the clear presentation to all users of the information contained. This and the previous report of work-packages 2, 3 and 4 provide the background necessary for the final selection of indicators which will be presented in the 3rd and final report of the INDECO work-packages 2, 3 and 4.

In the previous report we presented a review of potential indicators and their theoretical background. In this review we distinguished pressure and state indicators (from PSR, Pressure, State, Response) and provided for each type of indicators a framework that may be used to assess the quality of the indicator in terms of representivity respectively ascertain that all features of the ecosystem that need protection are covered by indicators.

In this report we will address the operational objectives relevant for step 1 and background information to guide the scoring against the screening criteria that is done in step 3. When relevant and possible we will distinguish different regions when presenting our material.

In our first deliverable we adopted a regional approach to develop suites of indicators for an EAFM by using the division into RACs as adopted by the EU as the first basic reporting unit. Following up on this we distinguish four ecosystems concomitant with the RAC areas:

- Baltic Sea
- North Sea
- Bay of Biscay (representative for the SW waters)
- Mediterranean

We did not include the NW waters as there was too little expertise of this area within the group.

These ecosystems may differ in terms of their biological features, physico-chemical characteristics, type and level of exploitation, data availability or other factors. As these differences may affect the scoring and choice of indicators, a brief description of these ecosystems and their exploitation is provided.

1.1 Criteria for the screening process

For the weighting and scoring of potential indicators against the screening criteria we provide background information per criterion on all potential indicators that may guide this process.

The screening process is based on Rice and Rochet (2005) and below are for each of the screening criteria, the constituent considerations (sub-criteria) in conducting the scoring (H, high; F, fair; M, moderate; L, low) for an indicator (IND). Stars on items labelled H and L indicate that, if the consideration (or method of evaluation) is relevant, scoring high there is of high importance, and scoring low is a nearly fatal flaw, respectively.

Concreteness

- Concrete property of physical/biological world (H), or abstract concept (L)?
- Units measurable in the real world (H), or arbitrary scaling factor (L)?
- Direct observations (H), or interpretation through model (L)?

Theoretical basis (number of competing theories to allow contrast is important)

- (i) Not contested among professionals (H); (ii) basis credible, but debated e can account for patterns in many data sets (H-F, depending on how other models fit the same data); (iii) credible, but competing theories have adherents and empirical support is mixed (M); (iv) adherents, but key components untested or not generally accepted (M-L)
- If IND derived from empirical observations: (i) concepts readily reconciled with established theory (H); (ii) concepts not inconsistent with, but not accounted for by, ecological theory (M); (iii) concepts difficult to reconcile with ecological theory (L)**;
- Theory allows calculation of reference point associated with serious harm (M)*

Public awareness

- Is it a property with a high (H) or low (L) public awareness outside the use as an IND?
- Does public understanding correspond well (H) or poorly (L) with technical meaning of IND?
- If awareness high, is public likely to demand action that is: (i) proportional to IND value as determined by experts (H); (ii) disproportionately severe (M); (iii) largely indifferent (L)
- Does the nature of what constitutes ''serious harm'' (used to define a reference point) depend on values that are widely shared (H) or vary widely across interest groups (L)?
- Internationally binding agreements, national or regional legislation require that a specific IND be reported at regular intervals (H), to agreements/legislation require environmental status reporting, but IND not specified (M) to no such requirements (L)

Cost

- Uses measurement tools that are widely available and inexpensive to use (H), to needs new, costly, dedicated, and complex instrumentation (L) IC; TJ

Measurement

- Can variance and bias of IND be estimated? Yes (H); No (L)
- If variance can be estimated, is variance low (H) to high (L)
- If bias can be estimated, is bias low (H) to high (L)?
- If IND biased, is direction usually towards overestimating risk (H), or towards underestimating risk (L)
- If both can be estimated, have variance and bias been consistent over time (H), or have they varied substantially (L)
- Probability that IND value exceeds reference point can be estimated with accuracy and precision (H), to coarsely or not at all (L)**

- IND measured using tools with known accuracy and precision (H), to unknown or poor/ inconsistent (L)
- Value obtained for indicator unaffected by sampling gear (H), to sampling methods can be calibrated (M), to calibration difficult or not done (L)
- Seasonal variation unlikely or highly systematic (H) to irregular (L)
- Geographic variation irrelevant or stable and well quantified (H), through random (M) to systematic on scales inconsistent with feasible sampling (L)**
- Taxonomic representivity: IND reflects status of all taxa sampled/modelled (High), through ecologically predictable subset of species (M), to only specific species with no identifiable pattern of representivity (L)

Availability of historical data

- Necessary data are available for: periods of several decades (H) to only relatively recent period (M), to opportunistic or none available (L)
- Necessary data are: from the full area of interest (H), to restricted but consistent sampling sites (Moderate), to opportunistic and inconsistent sources, or none (L)**
- Necessary data have high contrast, including periods of harm and recovery (H), to high contrast but without known periods of harm and recovery (M), to uninformative about range of variation expected (Low)
- The quality of the data and archiving is known and good (H), to data scattered with reliability but not systematically certified, and archives not maintained (L) MP (e.g. environmental IND);
- Data sets are freely available to research community (H), to private or commercial holdings (L)

Sensitivity (length of time-series used for testing important)

- IND responds to fishing in ways that are: (i) smooth, monotonic, and with high slope (H)**; (ii) smooth, monotonic, and with low slope (M); (iii) smooth, monotonic over a restricted range of effort characteristics (M-F); (iv) unreliable (M-F, depending on when it fails to inform about fishing effects); (v) insensitive or irregular. Magnitude of response does not depend on magnitude of signal in effort (L)

Responsiveness (length of time-series used for testing important)

- IND changes within 1-3 years of implementation of measures (H), to IND only reflects system responses to management on decadal scales or longer (L)

Specificity (contrast in data set used for testing important)

- Is impact of environmental forcing on IND known, and small (H) or strong (L)?
- If environmental forcing affects IND, effect systematic and known (H), to irregular or poorly understood (L)**
- Relative to other factors, IND: (i) known to be unresponsive (H); (ii) responds to specific factors in known ways (M); (iii) thought to be unresponsive (F); (iv) responds to many factors in only partly understood ways (L)**

Background information for potential indicators will be presented per region in sections that are relevant to one specific criterion: e.g. historical data, measurement or sensitivity.

2 Operational objectives

Whether the indicators are intended to merely inform discussion or to support decision-making directly, the management objectives need to be clearly specified. Some jurisdictions are attempting to do this explicitly (Bergen Declaration, 2002; EC, 2003), in which case the operational objectives can be taken directly from the policy documents. Often, however, operational objectives do not exist, or are so general and vague that they provide little guidance for selecting appropriate indicators. In those cases, management bodies must formulate operational objectives first. In doing this it is necessary that there is agreement on the way that regional or sector-specific operational objectives nest within those operating at a higher level, and how they relate to the high level Vision and goals expressed by the EU or national governments. Finally, it is important that terms commonly used in the application of objectives in a wider management framework are clearly defined and understood. Below is an objectives hierarchy and definitions that were agreed as a suitable framework within which UK ecological objectives and goals could be structured (Table 1). This framework starts with a Vision overarching a number of Strategic Goals, each of which will be underpinned by objectives for the ecosystem, as well as Social and Economic Objectives. In turn, each Ecological Objective will be supported by many operational objectives, with appropriate indicators and reference points. These operational objectives could consider both state and pressure variables.

Vision: high level statement of 'how things should be',
Strategic Goal: high level statement of what is to be attained that is common to all regions, uses and sectors.
Ecological Objective: high level statement of what is to be attained for each ecological component. This should relate only to the state of ecosystem components.
Operational Objective: An objective that has a direct and practical interpretation and that is specific to regions, uses and/or sectors. This could include objectives for both the pressure generated by human activities and the state of components.
Indicator: something that is measured and used to track an operational objective.
Target / limit reference points: A 'benchmark' value of an indicator, usually in relation to the operational objective, such as desired targets, undesirable limits or triggers for specified management response.

Table 2.1. An agreed hierarchy for setting ecological objectives within a framework to address the UK vision, and supported by operational objectives and their indicators and target / limit reference points.

For European waters a number of objectives have been formulated in the Marine Strategy which should constitute a contribution to the Community Strategy for Sustainable Development. As indicated in the 6th Environment Action Programme (6th EAP), the overall ecological objective of the Marine Strategy is that it should promote the sustainable use of the seas and conservation of marine ecosystems, including sea beds, estuarine and coastal areas, paying special attention to sites holding a high biodiversity value.
This overarching objective should be made operational by setting specific (intermediate) sectoral or issue objectives which should include time-lines for their achievement. Achieving this will require an integrated approach to address all threats and a careful assessment of their negative impact on marine environment and an identification of emerging threats.
In endeavoring to achieve this, the regional diversity in the ecological characteristics of the different seas and their sub-regions, the actual quality status thereof, the pressures and threats acting on these seas, the political, social and economic situations in the different regions and existing international institutional arrangements should be recognized and taken into account.
Several specific objectives have already been agreed in EC policy from the Treaty and specific legislation as well as by regional marine conventions. These objectives which are in many cases of a political value or aspirational nature have been used as a basis in the following overall set of objectives. Implementation of these objectives should reflect the overall high level of ambition but recognize regional variation on the actual need and opportunities for remedial action.

Loss of Biodiversity and Destruction of Habitats

Objective 1 The European summit in Gothenburg in June 2001 concluded in the context of the debate on sustainable development that a political objective of the EU was to halt biodiversity decline by 2010. This is an extremely ambitious and challenging objective which will drive environmental policy over the next 8 years.

Objective 2 In the longer term, the objective is to ensure a sustainable use of biodiversity through the protection and conservation of natural habitats and of wild fauna and flora in the first instance in the European seas, *inter alia*, by restoring marine ecosystems and reestablishing certain trophic levels which have been affected by human activities and by preventing the human induced introduction of new non-indigenous species, genetically modified organisms and disease organisms.

Objective 3 In relation to the reform of the Common Fisheries Policy which is currently underway the environmentally relevant objectives have already been identified and included in the Commission’s proposal on this reform, namely a change in fisheries management to reverse the decline in stocks and ensure sustainable fisheries and healthy ecosystem, both in the EU and globally.

For the Common Fisheries Policy the above objective only provides an operational objective for the commercial stocks, i.e. reverse the decline in stocks. What a “healthy ecosystem” or a “sustainable fishery” constitutes is not specified and hence there are no operational objectives stated.

For this exercise of selecting indicators we put forward a number of operational objectives in line with the higher-level objective of "ensure sustainable fisheries and healthy ecosystem" that correspond to all features of an ecosystem that need to be conserved. The assumption is that if all objectives are met the ecosystem is considered "healthy". Table 4.1 in chapter 4 represents all these features comprehensively. For each cell in this table that corresponds to a particular ecosystem feature the operational objective would be to:
"halt and reverse the decline of [ecosystem feature x] so that [ecosystem feature x] is within the bounds of natural variability".

3 Description regions

3.1 Baltic Sea

The Baltic Sea is one of the largest brackish areas in the world extending from 54º to 66º N and 12º to 30º E, and covering an area of 422 000 km^2. It receives freshwater from a number of larger and smaller rivers while saltwater enters from the North Sea along the bottom of the narrow straits between Denmark and Sweden. This creates a salinity gradient from southwest to northeast and a water circulation characterized by the inflow of saline bottom water and a surface current of brackish water flowing out of the area. The bottom topography features a series of basins separated by sills. The Gulf of Bothnia and the Gulf of Riga are internal fjords, while the Baltic Proper and the Gulf of Finland consists of several deep basins with more open connections. The western and northern parts of the Baltic have rocky bottoms and extended archipelagos, while the bottom in the central, southern, and eastern parts consists mostly of sandy or muddy sediment.

The water column in the open Baltic is permanently stratified with a top layer of brackish water separated from a deeper layer of more saline water. This separation limits the transport of oxygen from the surface and as a result the oxygen in the deeper layer can become depleted due to breakdown of organic matter. A strong inflow of new saline and oxygen-rich water from the North Sea can lead to a renewal of the oxygen-depleted bottom water. Strong inflows can occur when a high air pressure over the Baltic is followed by a steep air pressure gradient across the transition area between the North Sea and the Baltic. Such situations typically occur in winter. Strong inflows were frequent prior to the mid-1970s, but have since become rarer and as a result salinity has decreased over the last 25 years.

The Baltic receives nutrients and industrial waste from rivers, and airborne substances from the atmosphere. As a result the Baltic has become eutrophied during the 20th century. The eutrophication has led to increases in primary production, changes in trophic flows through food webs, and intensified magnitude and frequency of algal blooms and oxygen-depletion events that occur due to the infrequent exchange of water with the Skagerrak and North Sea. The effects of eutrophication on the fisheries are complex and difficult to resolve, but any process leading to a reduction in oxygen concentration in the deep layers during cod spawning periods will affect cod egg survival, as well as the survival of benthic animals that are prey for demersal fish species. The Baltic Sea is also severely contaminated. Whereas DDT pollution has decreased substantially, the decline of PCB and dioxin concentrations has levelled off. Contaminant levels in northern Baltic herring and salmon are so high that consumption is being regulated

3.1.2 The fisheries

The distribution of the roughly 100 fish species inhabiting the Baltic is largely governed by salinity. Marine species dominate in the Baltic Proper, while freshwater species occur in coastal areas and in the northernmost parts. The main target species in the commercial fishery are cod, herring, and sprat. They form about 95% of the total catch. Other target fish species having either local economical importance or ecosystem importance are Baltic salmon, plaice, flounder, dab, brill, turbot, pike-perch, pike, perch, vendace, whitefish, burbot, eel, and sea trout.

The main fisheries for cod in the Baltic use demersal trawls, pelagic trawls, and gillnets. There was a substantial increase in gillnet fisheries for cod in the 1990s, and because of the change in stock age composition in the late 1990s and early 2000, the share of the total catch of cod taken by gillnets has decreased and demersal trawl catches increased. Pelagic fisheries in the Baltic are dominated by pelagic trawlers catching a mixture of herring and sprat. The proportion of the two species in the catches varies according to area and season. In addition, fisheries for predominantly herring are carried out with trapnets/poundnets and gillnets in coastal areas. The catches of the pelagic species are used for human consumption, for reduction to oil and meal, and for animal fodder. While feeding in the sea, salmon are caught by driftnets and longlines and during the spawning run they are caught along the coast, mainly in trapnets and fixed gillnets. Where salmon fisheries are allowed in the river mouths, set gillnets and trapnets are used. The coastal fishery targets a variety of species with a mixture of gears, including fixed gears (e.g. gill, pound, and trapnets, and weirs) and Danish seines. The main species exploited are Baltic herring, Baltic salmon, sea trout, flounder, turbot, cod, and freshwater and migratory species (e.g. whitefish, perch, pikeperch, pike, smelt, vendace, eel, and turbot). In addition there are demersal trawling activities for Baltic herring, cod, and flatfishes in some parts of the coastal area. Coastal fisheries are conducted along the entire Baltic coastline.

3.1.1 Monitoring programs

Cod in the Baltic Sea have been monitored annually since 1982 through bottom trawl surveys (Baltic International Trawl Survey; BITS) that are carried out by research institutes in most Baltic countries. The national research vessels each surveyed part of the area with some overlap in coverage but applied at that time different gears, sampling seasons and sampling designs. In 1985 ICES established formal study groups in order to standardize the surveys. A common standard trawl (the TV3 trawl) and standard sampling procedures were implemented in year 2000. A survey manual has been agreed (ICES 2000) that specifies the standard gear design, fishing methods, biological sampling and formats for data exchange. The manual is regularly updated.

The standard TV3 trawl come in two sizes for different sizes of research vessels, one 520 meshes in circumference and one 930 meshes. The small trawl is used for vessels up to around 800 HP and the larger trawl for vessels with higher engine power. The design and construction of the standard trawls are given in the BITS manual.

BITS is conducted as a depth-stratified random survey during May and December each year. The surveys cover only the southern half of the Baltic Sea (Baltic Sub-divisions 22-28; Figure 3.1). The strata are based on Sub-divisions and depth layers. Each year the necessary stations are randomly selected before the beginning of the international trawl surveys from a list of clear haul data. The standard haul is a 30 minute haul with a towing speed of 3 knots over ground and trawling is restricted to daylight conditions.

Data from the survey results are checked nationally but stored in the ICES international database DATRAS (http://www.ices.dk/datacentre/datras/datras.asp)

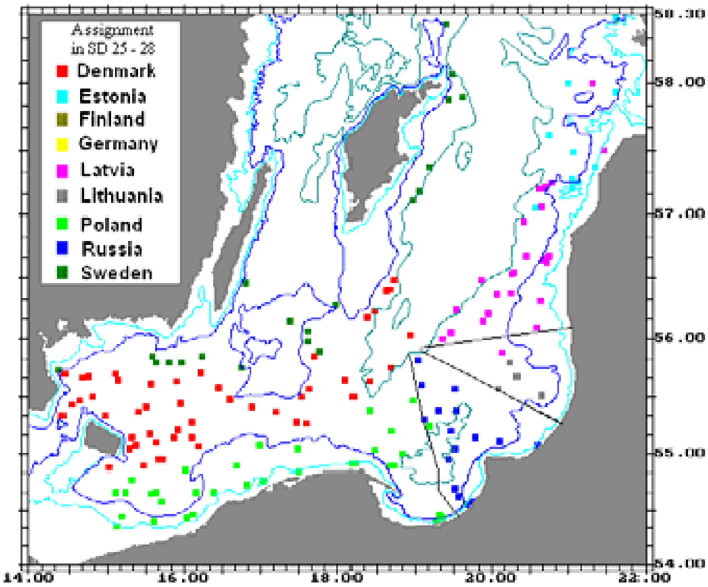


Figure 3.1 Example of the BITS coverage: Positions of planned stations in the Baltic Sub-divisions 25 to 28 in November 2005 by country (ICES 2006b).

In addition to sampling the open sea fish community in the Baltic, sampling of coastal fish has been performed since 1980s using multi-mesh gillnets (Thoresson 1993). Sampling of these coastal reference areas is performed annually in August at depths above 20 m as part of the HELCOM coastal fish monitoring programme (Ådjers *et al.* 2006). The selected areas are generally subjected to low fishing pressure and minor local anthropogenic impact. However, several of the HELCOM sampling sites are exposed to coastal fishing and may thus be tested as ecosystem indicator of fishing impacts.

3.2 North Sea

3.2.1 The ecosystem

The North Sea is a semi-enclosed, temperate sea which extends from 51° to 61° N and 3° W to 9° E and covers an area of 750,000 km^2 with a volume of 94,000 km^3 (Ducrotoy, *et al*., 2000). Topographically, the North Sea may be divided into three sections. These are the Southern Bight (51-54^{o}N) with water depths generally less than 40m, the central North Sea (54-57^{o}N) with water depths of 40-100 m, except for the shallower areas on the Dogger Bank and along the western coastline of Denmark, and the northern North Sea (north of 57^{o}N) which includes an area of shelf water 100-200m deep, and the Norwegian Channel with water depths from 200 to >700m in the Skagerrak between Denmark and Norway (Holligan *et al*, 1989). The North Sea is connected to the open seas in three areas. The main connection is through the wide northern boundary, but exchanges also occur through the narrow Dover Strait to the English Channel, and via the Fair Isle Current through the Shetland and Orkney Isles. The inflow of water through the northern boundary of the North Sea is estimated to be 1,300,000 m^3 s^{-1}, through the Dover Straits at 150,000 m^3 s^{-1} and between the Shetland and Orkney Isles at 300,000 m^3 s^{-1} (The Marine Forum, 1990). The currents of the North Sea tend to form a counter-clockwise cyclonic circulation pattern, as waters which enter from the north are transported to the western coast, whilst waters which enter through the Channel move up the Dutch/Belgian coast (Turrell *et al*., 1992). The majority of waters eventually flow through the Skagerrak and leave via the Norwegian Coastal Current. The salinity and temperature of water in the North Sea is influenced by the localised effects of freshwater input and heat exchange with the atmosphere. On average temperature varied between 4-16 ^{o}C but notably in the past decade a gradual increase was observed. Due to high nutrient loadings of the main rivers eutrophication was an issue in several of the coastal waters with a peak in the 80s but decreasing since then due to waste water treatment.

Different communities of both fish and benthos are distinguished within the North Sea. Their distribution is mainly determined by physical characteristics like depth, temperature or currents. Over time considerable changes have been observed in the biota. These may be caused both by anthropogenic factors (e.g. fisheries) as well as other factors like climate.
The North Sea is probably one of the most data rich areas in the world with regular analytical assessments of most exploited fish stocks based on regular market sampling of landings in all major ports, several standardized annual RV surveys that also register non-target fish species, the Continuous Plankton Recorder that monitors plankton in a part of the North Sea and often less extensive monitoring programmes of benthos, seabirds, or marine mammals that are often not conducted on an annual basis.

3.2.2 The fisheries

The North Sea has been fished intensively for centuries but notably in the early to mid 20th century there has been a rapid expansion and mechanisation of the fishing fleet (Rijnsdorp and Millner, 1996) resulting in a considerable increase in impact. Historical data describing the North Sea ecosystem before the mechanisation of the fishing fleet hardly exist (if at all) and so it is difficult to assess or quantify accurately the effects of fishing activity on the target species, non-target species and habitats. There are several types of fisheries in the North Sea, targeting different species and hence using different gears and covering different areas. The two main types of demersal fishery are otter trawl in the North-Western part of the North Sea targeting roundfish such as cod or haddock and beam trawl in the South-Eastern part of the North Sea targeting flatfish, notably plaice and sole. The main target species of the pelagic fishery is Herring and there is an industrial fishery that mainly targets sandeel. There is also an important fishery targeting invertebrates, mostly in the coastal zone there are fisheries directed at shellfish or shrimp, in the offshore waters the fishery targets Nephrops. Within each of these fisheries there is a great variety in types of gear or ways of rigging making it extremely difficult to characterise the different fisheries.

As the ecological status of most commercial stocks has deteriorated, increasingly stringent measures have been adopted to reduce fishing effort. This has resulted in a gradual decrease in terms of capacity and fishing effort (expressed in days-at-sea or hours fished) but with increased efficiency due to technical creep it is unclear to what extent the effective pressure of the fisheries on the ecosystem and its components (i.e. mortality or habitat damage) has changed.

3.2.3 The surveys

There are several bottom trawl surveys conducted in the North Sea suited to assess the relative abundance of populations that are not regularly assessed. Of these surveys we describe those surveys that are often used to create time-series for the indicators: one beam trawl survey (BTS), two otter trawl surveys International Bottom Trawl Survey (IBTS) and the English

Ground Fish Survey (EGFS) and finally the Scottish August Ground-fish Survey (SAGFS) using a 48-foot Aberdeen Otter Trawl.

The IBTS survey covers the whole North Sea, Skagerrak and Kattegat, within the 200 m isopleth. We have only used 1st quarter data from 1980 onwards, the year when the same survey gear, a GOV-trawl (Grande Ouverture Verticale), was adopted by all participating nations and excluding Skagerrak and Kattegat. For gear specifications see ICES (1999).

The BTS was initiated in 1985 to estimate the abundance of the dominant age groups of plaice and sole including pre-recruits. The survey is carried out in the south-eastern North Sea by RV Isis and in the central and eastern North Sea by RV Tridens. Both vessels use a pair of 8 m beam trawls rigged with nets of 120 mm and 80 mm stretched mesh in the body and 40 mm stretched mesh cod-ends. A total of 8 tickler chains are used, 4 mounted between the shoes and 4 from the groundrope. RV Tridens is also equipped with a flip-up rope. The survey was designed to take between one and three hauls per ICES rectangle (boxes of 0.5° latitude by 1° longitude). The stations are allocated over the fishable area of the rectangle on a "pseudo-random" basis to ensure that there is a reasonable spread within each rectangle. No attempt is made to return to the same tow positions each year. Towing speed is 4 knots for tow duration of 30 minutes and fishing occurs during daylight only.

3.3 Bay of Biscay

3.3.1 The ecosystem

The Bay of Biscay extends from 44° to 48º N and from 11º W to the coastlines of France and Spain and can be considered as a subtropical/boreal transition zone. The topographical diversity and the wide range of substrates result in many different types of coastal habitat. This diversity is reflected in the biological richness of the region, which includes a wide range of fish species, many of these of commercial interest.

On the continental shelf, the transport is driven by tides and wind, with buoyancy important off major rivers during periods of high run-off. The most conspicuous upper layer mesoscale features are a poleward-flowing slope current in autumn and winter, and wind-induced coastal upwelling in spring and summer.

Eutrophication does not appear to be a problem in this region although an apparent increase in the occurrence of harmful algal blooms has been reported in recent decades. Mariculture is mainly confined to the cultivation of bivalve molluscs (mussels, oysters and clams) and its impact is usually minimal.

In the Bay of Biscay, changes in species composition since the eighteenth century are identified and effects of both fishing and warming effects were observed. In the last 20 years, a significant warming of the sea surface temperature has been reported as well as increased records of tropical species.

There are regular analytical assessments of most exploited fish stocks, a standard annual RV survey, regular market sampling of landings in all ports, but there are no regular monitoring programmes of plankton, seabirds, or sea mammals.

NOTE: in northern Spain there are several coastal monitoring networks which include plankton survey, some of them starting in the 80's and 90's. Moreover, regular surveys of ichthyoplankton have been undertaken since 1985, covering the area between Santander and Nantes (coastal and offshore) in spring. There are also data from CPR. Regarding sea mammals, there are several studies and projects, including monitoring of these species from ferry lines (Bilbao-Portsmouth, etc.) or specific surveys.

3.3.2 The fisheries

Most of the demersal fisheries in this area have a mixed catch. Although it is currently possible to associate specific target species with particular fleets, various quantities of cod, whiting, hake, anglerfish, megrim, sole, plaice, and *Nephrops* are taken together, depending on gear type. Since the 1930s, hake has been the main demersal species supporting trawl fleets on the Atlantic coasts of France and Spain. Hake are caught throughout the year, the peak landings being made in spring-summer months. The three main gear types used by vessels fishing for hake as a target species are lines, fixed-nets and trawls, mostly bottom trawls, a few pelagic ones, and recently also Very High Opening trawls. *Nephrops* are an important component of the fisheries in this area. These fisheries developed in the 1970s and 1980s. Fishing effort has decreased continuously since the early 1990s. However, gear efficiency has increased in recent years and this may have helped maintaining LPUE at relatively high levels. Bay of Biscay anchovy is also the target of an important pelagic fishery (pelagic trawls, seines).

3.4 Mediterranean

3.4.1 The ecosystem

The Mediterranean Sea lies between Europe, Asia and Africa (about 46°N, 30°N, 6°W and 36°E). Excluding the Black sea it covers an area of approximately 2.5 million km^2, with an average depth of about 1500 m (the deepest recorded point is 5267 m in the Calypso Deep in the Ionian Sea) and a total volume of 3.7 million km^3; the coastline extends for 46,000 km.

The Mediterranean Sea is an enclosed basin, being connected to the Atlantic Ocean only through the shallow (300 m) and narrow (14 km) Strait of Gibraltar. The continuous inflow of surface water from the Atlantic Ocean is the major source of replenishment and water renewal for this ecosystem. The major feature of this surface current system is the movement of water from the Atlantic Ocean toward East combined with numerous spin-off eddies along the way. The return of Mediterranean waters to the Atlantic Ocean takes place through deep waters flowing from East to West and spilling over the sill of Gibraltar into the Atlantic Ocean. The Mediterranean circulation system also includes strong vertical convection currents that determine the distribution of salinity and provide for vertical recycling of nutrients and other dissolved substances.

Ocean waters take over a century to be completely renewed through the Strait of Gibraltar. The scarce fresh water inflow from rivers coupled with high evaporation processes, makes the Mediterranean to be a 'concentration basin', much saltier than the Atlantic. Exceptions to these features are the Adriatic Sea and the Black Sea, due to the water discharge of big rivers, namely Po River for Adriatic and Danube, Don and Dnieper Rivers for the Black Sea.

A shallow submarine ridge (the Strait of Sicily) between the island of Sicily and the coast of Tunisia divides the sea in two main sub-regions: the western and the eastern Mediterranean (respectively 0.85 million km² and 1.65 million km²).

The Mediterranean region is climatically speaking a transitional area: rainfall throughout the region is irregular both within and between years and in particular in the southern parts where these irregularities are relevant. It is characterized by the so-called Mediterranean climate.
Except in very few areas, like the Gulf of Gabes in Tunisia and the Northern Adriatic, the Mediterranean is characterized by very weak tides, with tidal amplitudes that are very small compared to standards for mid latitude areas (smaller than 1 m); this feature has major consequences on the characteristics for the shorelines and the pollution.

The Mediterranean Sea has relatively low concentrations of nutrients even in deeper waters. These chemicals are exported in the flow of deep water through the Strait of Gibraltar that in turn receives nutrient poor surface Atlantic water. No deep nutrient-rich Atlantic waters take part in the Mediterranean circulation, and the input of nutrients is mostly due agricultural run-off through rivers input and direct discharge.

While the Mediterranean Sea exhibits a low level of biological productivity, it is characterized by a relatively high degree of biological diversity. According to a rough estimate, more than 8500 species of macroscopic marine organisms live in the Mediterranean Sea, corresponding to somewhat between 4% and 18% of the world's marine species (Bianchi and Morri, 2000). This is a conspicuous figure if one considers that the Mediterranean Sea is, compared to the world ocean, only 0.82% in surface and 0.32% in volume. The high biodiversity of the Mediterranean Sea may be explained by historical (its tradition of scientific studies dates older than any other sea), paleogeographic (its tormented geological history through the last 5 million years has been determining the occurrence of distinct biogeographic categories), and ecological (its variety of climatic and hydrologic features within a single basin has probably no equals in the world) reasons. The Mediterranean Sea is also characterised by a high number of endemic species, averaging more than one quarter of the whole Mediterranean biota (Tortonese, 1985; Fredj et al., 1992; Giaccone, 1999) and is significantly higher than that for the Atlantic Ocean.

At present the Mediterranean biodiversity is undergoing a warring alteration under the combined pressures of climate change and further anthropogenic sources of ecological disturbance (*e.g.* over-exploitation, introduction of alien species, pollution, land-use reclamation) but protection measures, either at species or ecosystem level, are still scarce and mainly ineffective.

The continental shelf is usually very narrow, but the coastal marine area of the Mediterranean, which stretches from the shore to the outer extent of this continental shelf, shelters rich ecosystems and the few areas of high productivity in the sea. Whereas central zones of the Mediterranean are low in nutrients, coastal zones benefit from terrestrial nutrients that support higher levels of productivity. Among coastal marine areas, rocky intertidals, estuaries, and, above all, seagrass meadows (mainly *Posidonia oceanica*) are ecosystems of significant ecological value.

The Mediterranean hosts a huge number of critically endangered species (IUCN, 2006), but the assessment of the status of some vulnerable and poorly known marine groups, like sharks and rays is still lacking (e.g. Coll and Palomera, 2006). So far, the attention has been focused on flagship and charismatic species, like the Monk Seal or marine turtles, but a first assessment of the status of sharks has shown that some species are thought to be extirpated from the Mediterranean Sea, and others threatened with extinction risk.

Finally, the Mediterranean basin hosts important wetland and lagoon systems, in Spain (Valencia), France (Languedoc and Giens), Italy (Sardinia, Tuscany, Puglia, and Venice), Central Greece, Cyprus, Morocco (Nadar), Algeria, Tunisia, and across the entire Nile delta in Egypt. These areas are of great significance to the conservation of biological diversity (RAMSAR Convention, 2002) and are also highly productive. They support numerous other functions related to flood control, recreation, tourism, fisheries and agriculture as well as nutrients and chemicals retention (Jickells et al., 2000). They also act as nursery areas for many fish specie and as breeding and wintering areas for a great variety of birds, being essential stopover points on the migratory routes of numerous species.

Gulf of Lions

The sub area Gulf of Lions is an open gulf in the Northwest of the Mediterranean sea which expands from the Cap Cerbère (western part) to the Cap Sicié (eastern part). The continental shelf of the gulf is nearly monotonous. From the coastal line to 200 meter depth, muddy deposits including pelits are dominating. In some places, sandy deposits exist along the coast and far from the shore. Very few rocky outcrops occur in this area. The upper slope is carved with a lot of canyons where bedrocks can be found around the valleys.

The most characterized water flow is the Liguro-Provencal stream which passes alongside the continental slope westward, from the Gulf of Genoa to the Catalan Sea. From spring to autumn, a well designed thermocline divides the surface waters with an average temperature of 20 °C from the deeper layer which is about 13 °C all during the year. The depth of the thermocline varies from 50 to 100 meters.

An important feature in the Gulf of Lions is the possibility of upwellings induced by the wind. Two types of wind, mistral and tramontana blow from the coast to the open sea, inducing a rising of cold deep water towards the sea surface. These winds are active mainly in winter and sometimes in summer. Within a short time, they bring about a decreasing of sea surface temperature of 5 to 10 °C. In the whole area, the salinity is high; it can reach 38.5 psu with a slight oversaltness in the deep water.

Taking into account the muddy characteristics of the seabed on the shelf, wide areas inhabited by species of concern for fishery are dominated by *Ophiotrix*. In the Gulf of Lions, there are some important populations of species like poor cod (*Trisopterus minutus capelanus*), which is lacking along the east coast of Corsica, and red mullets. A lot of cephalopods with high fishery value, as the common octopus, are living on the shelves.

Large individuals of some species able to live on the shelves and along the slopes are often more represented in the latest areas. It is particularly the case for the big anglerfishes, adults of hake, spiny lobsters and big shrimps. Finally, one important characteristic of the existing distribution of some of the main fishery target species in the Gulf of Lions is the separation of the populations into two components, the juveniles (on the upper shelves for hake and anglerfishes) and the breeding adults (mainly along the upper slope for these two species, for instance). This partition between shelf and slope has to be considered taking into account the distribution of fishing effort in the area. Indeed, as the French trawlers fishing in the Gulf are out of the port for periods of less than one day, and due to the distance from the ports to the limit of the shelves, they have not the possibility to fish along the slope where they could find bigger individuals.

The Strait of Sicily

The Strait of Sicily, which connects the Western to the Eastern Mediterranean basins, has complex bottom morphology. Along the southern coasts of Sicily, the shelf is widest in the Westernmost (Adventure Bank) and Easternmost (Malta Bank) sectors (about 50 nautical miles wide). A narrow shelf in the central part (about 15 nm wide) separates these two large banks. The shape of the slope is extremely irregular, incised by many canyons, trenches and steep slopes. From an oceanographic point of view, the Strait of Sicily is characterized by a two-layers flux model: the upper layer, or "Modified Atlantic Water" (MAW), with a relatively low salinity waters flowing from the Western to the Eastern Mediterranean basins and the lower layer flux, or "Levantine Intermediate Water" (LIW), characterized by relatively higher salinity waters and richer in nutrients and flowing in the opposite direction. The surface circulation promotes the establishment of permanent upwellings towards the left side of the stream in certain places, possibly reinforced by wind-induced upwellings, which may sharpen the density front due to the offshore Ekman transport. The complex bathymetry influences the current features in the region, resulting in mesoscale patterns such as eddies, meanders and filaments in the upper layer. The main source of nutrients in the area is associated with coastal upwellings and the doming of intermediate waters inside the cyclonic vortexes.

The Northern Adriatic Sea

The Adriatic Sea (800 km of length; 180 km of medium width; surface of 139000 km^2) is an epicontinental sea, constituted by a series of three basins that increase in depths from north to south. The northernmost part, which constitute the Northern Adriatic Sea, has an average depth of 33.5 m and a surface area of 18900 Km^2 (Fonda Umani *et al.*, 1992). The north-western part of the Adriatic is represented by a sedimentary shoreline, which is characterized by a system of deltas and lagoons between the Po and Isonzo rivers. In contrast, NE Adriatic coast is hilly and has predominantly rocky shore.

The Northern Adriatic is subjected to strong forcing functions, atmospheric and river run-off, producing a clear seasonal variability both in the circulation (Artegiani *et al.*, 1997) and the primary productivity (Fonda Umani *et al.*, 1992). During winter the water column is completely mixed due to the surface heat loss induced by low air temperature and wind, that leads to the formation of the North Adriatic Deep Water (NAdW); in contrast, during summer, the water column is stratified. The river runoff, especially the contribute of the River Po discharge, is particularly important in this basin and affects circulation since this flow deviates towards south following the western coast determining the typical counterclockwise circulation of the Adriatic (Zavatarelli *et al.*, 1998). River run-off is also responsible of the input of nutrients, which enhance the productivity of this basin, considered eutrophic if compared to the general oligotrophic, nutrient limited, waters of the Mediterranean Sea.
Due to its geological history and climate features, the Northern Adriatic Sea, is considered, biogeographically speaking, as a separate subunit of the Mediterranean Sea, characterized by some individual properties. These peculiar conditions influence the distribution of species, thus in the North Adriatic Sea endemic Mediterranean species cohabit with species of Atlantic origin (Gamulin-Brida, 1967; Bianchi and Morri, 2000).

Urbanization and agriculture in the Po river catchment basin, coupled with the population increase and tourism pressure, lead to eutrophication problems in the Northern Adriatic Sea with early signs in the 1930. Eutrophication reached its maximum during the 80's, coupled with dystrophic events and mass mortalities due to benthic anoxia (Barmawidjaja *et al.*, 1995; Sangiorgi and Donders, 2004).

Furthermore there was an increase of the frequency of mucillages events, a phenomenon which was first described in the begin of the 18th century, and whose causes seems to be related to the changes in the water circulation pattern in the Adriatic Sea, climate conditions and further anthropogenic disturbance as like nutrient loads (Deserti *et al.*, 2005).

The Aegean Sea

The Aegean Sea is characterized by highly irregular coastline, the presence of many islands (about 2000) and the presence of an extended plateau and deep basins such as the North Aegean Trough with a maximum depth of 1600 m, and the Cretan Sea with a maximum depth of 2561 m. The main water masses in the Aegean are: a) Black Sea Water, b) Levantine Intermediate Water, c) modified Atlantic Water and d) Eastern Mediterranean Deep Water. Intensive vertical mixing takes place over large areas of the Aegean Sea and in areas such as the Cretan Sea and the North Aegean Sea it leads to dense water formation and, hence, to the ventilation of intermediate and deep layers (Stergiou *et al.*, 1997). The concentration of nutrients in the Aegean Sea is 12 times lower than that in the Atlantic Ocean. The surface layer is separated from the intermediate and deep-water layers by a transitional layer of 100-200 m thickness, within which the concentration of nutrients increases rapidly. In relation to oxygen concentration, surface layers are almost saturated in oxygen, while a sharp decline is observed in the transition layer reaching in the deep water layer about 4.2.ml/l. Annual primary production in certain areas of the Aegean exhibit the lowest values recorded in the Mediterranean. Secondary productivity was estimated to be 12 to 18 times lower than that in the Black Sea and four times lower than in the Adriatic Sea. The total number of fish species in the Aegean is lower than that reported for the whole Mediterranean, indicating the impoverishment of the eastern part of the Mediterranean.

A great number of small scale fishery vessels operate in the Aegean Sea, 13114 of which are smaller than 12m and 420 have a larger size. Moreover, there are 298 trawlers and 271 purse-seiners. Overall, fishery production is inadequate to meet local needs. Moreover, the total sea fishery production of Greek waters has shown a declining trend during the last years; from 140000 tonnes in the beginning of last decade it dropped to about 90000 tonnes. In general, the oligotrophic nature of the Aegean Sea, and of the Greek waters in general, is also clearly reflected in fish catch densities which, on average, are lower than those in other areas of the world, although in this oligotrophic environment relatively eutrophic areas do exist.

Tyrrhenian – Liguria Sea (GSA 9)

The Tyrrhenian Sea is generally considered as a distinct entity within the western Mediterranean basin, because it is semi-enclosed between islands (Corsica, Sardinia and Elba) and mainland (Italy), and separated from the rest of the western basin by a channel of moderate depth, ca 1500 m.

Along the central-western Italian coasts the Tyrrhenian Current or Eastern Corsica Current (ECC) flows northward through the Corsica Channel into the Ligurian Sea. The Corsica Channel is a passage between the Corsica and Capraia Islands, connecting the Tyrrhenian Sea to the south with the Ligurian Sea to the north. It is about 40 km wide at the surface and significantly narrower below 100 m depth, steeping deeply up to 450 m . It plays a key role for water circulation in north-west Mediterranean Sea as the water exchange through it involves the whole water column (80% MAW and 20% LIW; Astraldi and Gasperini, 1992). North to the Capraia Island, the ECC merge the colder Western Corsica Current (WCC) which flows northward on the western side of Corsica Island, forming the Ligurian Current (Béthoux *et al.* 1982). This latter substains a basin-wide cyclonic circulation in the Ligurian Sea involving both the Modified Atlantic Water (MAW) at the surface and the Levantine Intermediate Water (LIW) at depth.

In the northern and central Tyrrhenian Sea, the circulation is organized in a series of cyclonic and anticyclonic gyres determined by the wind effect. The current fluxes (except in winter) are mostly concentrated at the boundaries of the existing gyres. In winter the current at the frontal region increases and there is a westward shift and an intensification of associate upwelling. This is the only season in which a direct connection exists between the Tyrrhenian and Ligurian seas through the Corsica channel. In the other seasons the water transport is confined to the frontal regions of different gyres. Due to the occurrence of the gyres, the northern part of the basin exerts a crucial role in the general water mass budget on the Tyrrhenian Sea. A principal effect is that the associated upwelling provides a mixing of the MAW (Modified Atlantic Water) and the LIW (Levantine Intermediate Water) below, with a corresponding modification of the water properties (Artale et al., 1994).

The general pattern of phytoplankton seasonal dynamic was typical to subtropical areas with a maximum in cold season from October to April and a minimum in summer months. The intensity of winter spring bloom (March-April) significantly varied during different years. In the Ligurian Sea a substantial positive correlation links the intensity of spring bloom of phytoplankton with strong autumn-winter water turbulence (mainly driven by winds) and reduced wind mixing in March (Nezlin et al., 2004).

3.4.2 The fisheries

The Mediterranean Sea is one of the most diverse and stable Large Marine Ecosystems in terms of species groupings and their share in the total catch (more than 100 species are commercially exploited).

The fishing activity is characterized by a high diversity both in terms of catches (multi-target) and fishing techniques/gears (multi-gear). Fisheries can grouped into three main categories: demersal fisheries (trawling and seining fisheries), pelagic fisheries (which targets both small and large pelagic resources) and small-scale fisheries.

The FAO 10-year capture trend (1990-1999) shows stable catch trends in recent years, with a moderate increase in shelf catch, from under 1 million tons in 1990 to 1.1 million tons in 1999 (FAO, 2003). Moreover, no significant differences between the western to the eastern basin can be observed (Papacostantinou and Farrugio, 2000). Clupeoids (sardines, anchovies and herrings) form the most important group in the catch (38%) followed by miscellaneous coastal fishes (18%) and molluscs (16%).

According to Caddy (1990) the increase in total landings recorded during the last decades could be due, at least partially, to a bottom-up effect related to increased river basin nutrient run-off. In any case, most of the exploited stocks must be considered fully or heavily exploited. Moreover, some areas seem to evidence a fishing-down-the-food-web effect due to fisheries. These disruptive effects and decrease in species diversity might leave more space for invading alien species.
In general, fisheries management in the Mediterranean could be considered at a relatively early stage of development, with a lack of reliability of time-series data (official statistics). Quota systems are generally not applied, mesh-size regulations usually are set at lower levels relative to scientific advice, and effort limitation is not usually applied or, in the case it is usually not based on a formal resource assessment.

The management measures applied by the Mediterranean countries can be broadly separated into two major categories: those aiming to keep the fishing effort under control and those aiming to make the exploitation pattern more rational. Moreover the rapidly growing population and urbanisation along the littoral, the high number of tourists, the scarcity of effective marine protected areas, render the objective of reaching sustainable fisheries particularly daunting.

Furthermore most of exploited stocks are shared between different countries and this increase difficulties on enforcing effective management.

Finally, due to I) the huge diversity of fishing gears and practices, II) the very high intensity of fishing effort, III) the high diversity of habitats (from the shallow-waters to the deep-sea and the oceanic domain), IV) the important biological diversity, a wide variety of impacts of fishing activity is recorded in the Mediterranean (Tudela, 2004). These impacts range from local effects on the sea bottom caused by trawling to large-scale to impacts on Cetacean populations driven by driftnet by-catch.

The Gulf of Lions
Trawling has been used for fishing since a few centuries in the Gulf of Lion. During the last three decades, it has been developed to such an extend that it is now one of the main components of the fishery techniques in the area (Meuriot *et al.*, 1987). Indeed, owing to technical innovations (notably from bottom to pelagic trawling), the trawling fleet is able to fish in the whole water column. Nowadays, this fleet can be divided into two main components, one directed to few small pelagic species (*Engraulis encrasicolus* and *Sardina pilchardus*), and the other characterised by the exploitation of a great diversity of demersal and pelagic species. Finally, trawlers contribute to the exploitation of the main species caught in the area. Landings from trawlers would represent more than

three quarters of the demersal species landings in the area (C.G.P.M., 1988) and close to 80 % of small pelagic caught (Taquet *et al.*, 1997).

The Strait of Sicily

The fleet operating in the Strait of Sicily consists of about 350 trawlers, with an overall GRT amounting to 27100 and a total power of 94496 KW, based in Sicilian harbours (IREPA source).

The Sicilian trawlers, operating mainly on a short-distance trawl fishery, are based in seven main port (Mazara del Vallo, Sciacca, Porto Empedocle, Licata, Gela, Scoglitti, Pozzallo) along the southern Sicilian coasts. Excluding the Mazara fleet, trawlers usually perform daily trips, starting in early morning and coming back in the afternoon. Normally they carry out two 4-5 hour hauls per day. Mazara del Vallo represents the main commercial fleet of trawlers of the area and one of the most important of the Mediterranean, with 147 trawlers, having an overall GRT amounting to 20211 and a total power of 59970 KW . Differently from the other Sicilian fleets, about the 80% of the Mazara trawlers, the largest ones, usually are employed on long fishing trips (15 – 25 days) mainly in international waters of the Strait of Sicily, operate both on the continental shelf and on deep bottoms (down to 700 – 800 m depth).

Considering the Sicilian fleet, two main type of trawling fisheries could be identified:

- the inshore trawling, operating closely to Sicilian coast, including the whole fleet of Sciacca, Porto Empedocle, Licata, Gela , Scoglitti e Pozzallo and about the 25% of the Mazara del Vallo trawlers. This fishery is a typical mixed species Mediterranean;
- the distant trawling which is formed by the most of Mazara del Vallo fleet, operating in a wider area. This fishery is targeted, according to areas and season, to three key species (*P. longirostris, A. foliacea* and *Mullus* sp.).

Studies conducted on the demersal resources shown that the fishing capacity in the Strait of Sicily, corresponding to the maximum sustainable yield for all demersal species, was overcome during the late seventies – early eighties. A first evident sign of sufferance of the demersal resources on the northern side of the Strait of Sicily resulted from the analysis of the catch rate of commercial species from seventies onwards. A further signal of overfishing could arise from the analyses of the discard practices in the Mazara fleet. During the last ten years many species once discarded, such as *Clorophtalmus agassizi*, *Argentina sphiraena* and the small sized shrimp of genus *Plesionika,* are now landed (Anon., 2000). More in detail from 1996 to 2000, the percentage of the total catch of Mazara trawlers, which was discarded, decreased from 50 % to 20 %.

The Northern Adriatic Sea

Due to the enormous fresh water flow and the subsequent contribution of nutrients the Adriatic Sea shows a high primary production. 55% of Italian commercial fisheries catch comes from the Adriatic Sea. The Northern Adriatic Sea fisheries contributes to about 24% of the Italian landings, which is a conspicuous figure if we consider the features of the Northern Adraitic fishing fleet, which constitute respectively the 15% of the total number of Italian fishing vessels, 12% of its gross tonnage and 18% in terms of engine power (IREPA, 2003).

The main fishing activities (in terms of number of fishing vessels) are represented by small scale-fisheries, demersal trawlers (either otter trawl and rapido-trawl), hydraulic dredges and mid-water pelagic trawlers. Pelagic species (sardines –*Sardina pilchardus*- and anchovies -*Engraulis encrasicholus*) represents the most important species landed in the Northern Adriatic Sea (respectively 25,200 and 5,600 tonnes) followed by the clam *Chamelea gallina* (9,800 tonnes), and other molluscs (7,500 and 6,700 tonnes respectively).

Tyrrhenian – Liguria Sea (GSA 9)

In the central Tyrrhenian-Ligurian sea the fishing fleet is made up by 1875 vessels using different fishing techniques. Small-scale fishery using passive gears, is the main sector as number of vessels (1253) followed by trawlers (344 vessels) and polyvalent vessels (173). Trawling is the main fleet segment as total engine power and gross tonnage as well as total annual yield. Vessels perform daily trips to exploit resources both on the continental shelf and slope up to 700 m depth.

Since 1996 the fleet decreased of about 32% and 27% as total tonnage and total engine power respectively, due to the demolition facilities within the EU Multi-Annual Guidance Programme - MAGP IV. The fleet is widespread along the coasts between several fishing harbours. The total annual landing was estimated around 22.000 tons in 2004 of which 8400 tons came from trawl fishery, 6810 from purse seiners exploiting small pelagics and 5500 tons from small-scale vessels.
The most important commercial species in 2004, as total value of annual landing, were in ranked order of importance, hake (*Merluccius merluccius*), swordfish (*Xiphias gladius*), Norway lobster (*Nephrops norvegicus*), cuttlefish (*Sepia officinalis*), red mullet (*Mullus barbatus*), deep-sea pink shrimp (*Parapenaeus longirostris*) and anchovy (*Engraulis encrasicolous*).

3.4.3 The surveys

The analyses have been done using the data coming from the French part of the Medits surveys (International bottom trawl in the Mediterranean). These surveys cover all the trawlable areas over the shelves and the upper slopes from 10 to 800 m depth in the Gulf of Lions (13860 km²). The stations were distributed applying a stratified sampling scheme with random drawing inside each stratum. The target sampling rate was one station per 60 square nautical miles. The same sampling gear (GOC 73) was used for the whole survey. Its codend mesh size is 20 mm (stretched mesh), and its vertical opening was about 2 meters. One Medits survey was carried out every year since 1994, during the spring-summer period.

During the surveys, all macrofauna species were usually identified, counted and weighted (total weight), with a special attention to a list of 56 fish, cephalopods and crustaceans (long Medits list). For a selection of these, the length frequency distribution, sex and maturity stages were recorded (short Medits list). The community indicators were calculated based on the species of the long Medits list, while the population indicators were calculated based on the species included in the short Medits list. In order to remove rare, poorly sampled species, species with a low occurrence were excluded. An occurrence threshold of 5 % (average across years and shelf and slope area) was usually applied, with some adjustments according to the geographical unit.

Most of the biological information on groundfish gathered by IAMC-CNR Sez. di Mazara del Vallo in the Strait of Sicily was obtained within the framework of two main programs of assessment of demersal resources: the GRUND program, started in 1985 and founded by the Italian government and the International program MEDITS, started in 1994 and supported also by the European Union.
Both the programs cover all the trawlable areas from 10 to 800 m depth of the Italian side of the Strait of Sicily in autumn (GRUND) and spring (MEDITS).

Samples were collected with the professional trawler “S. Anna” and two standard gears, with a fine mesh in the cod-end (20-30 mm opening).

Data were processed to obtain abundance indices in term of number (Density Index) and biomass (Biomass Index) per square kilometre of the whole catch and biological information on the target species (length-frequency distributions, sex ratio, maturity stage, age composition, growth, mortality).

3.4.4 The MEDITS survey

MEDITS is an international survey conducted annually in all Northern Mediterranean (Table 3.1, Figure 3.2). From 1994 to 2004 overall 11,741 hauls were performed (Table 3.2). These data are currently being processed to estimate indicators along the same lines as in the Gulf of Lions, a report should be available by the end of 2006.

Table 3.1. The geographical units in the Mediterranean.

FAO SUBAREA	GFCM MANAGEMENT UNITS (SAC-GFCM, 2001)	CODES of the geographical units	MEDITS strata	Surface (km², from Medits)	Comments
WESTERN	6. Northern Spain	6	112 & 113	32506	
	1. Northern Alboran Sea	1	111	12753	
	3. Southern Alboran Sea	3	114		
	7. Gulf of Lions	7	121	13860	
	8. Corsica Island	8	131	4562	
	9. Ligurian and North Tyrrhenian Sea	9	132	42410	The boundary between the areas 9 & 10 is ~ 0.5° southern than the GFCM limit.
	10. South and Central Tyrrhenian Sea	10	134a-b	20255	
	11. Sardinia	11	133	26975	
CENTRAL	15. Malta Island 16. South of Sicily	16	134c & 135	59278	
	17. Northern Adriatic	17	211	92261	
	18. Southern Adriatic Sea	18	221e-h	24008	
	19. Western Ionian Sea	19	221a-d	13520	
	20. Eastern Ionian Sea	20	222	16823	
EASTERN	22. Aegean Sea (including Crete)	22	223, 224 & 225	155674	

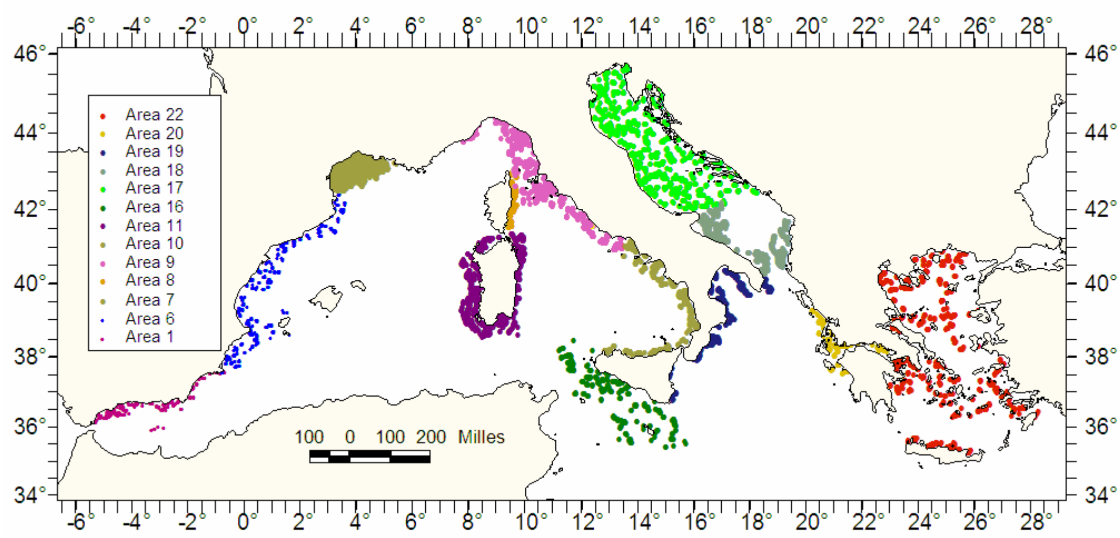


Figure 3.2. Haul position of the Medits survey in the thirteen geographical units surveyed since 1994.

Table 3.2. Number of hauls per year and area.

Year	1	6	7	8	9	10	11	16	17	18	19	20	22
1994	28	55	70	24	153	84	124	56	86	72	74	12	98
1995	28	83	68	22	153	85	108	57	88	72	74	15	105
1996	34	73	65	24	153	85	125	56	137	112	74	22	135
1997	36	66	76	13	153	85	126	56	139	112	74	18	146
1998	34	60	71	24	153	85	123	56	138	112	74	32	146
1999	39	78	66	25	153	85	124	56	86	112	74	32	146
2000	39	75	69	23	153	85	123	61	135	112	74	31	142
2001	40	85	67	22	153	85	123	65	136	112	74	31	141
2002	46	90	64		120	70	99	76	181	90	70		
2003	47	93	73	23	120	70	99	120	181	90	70	32	142
2004	43	85	67	25	120	70	95	118	182	90	70	32	149
Total	**414**	**843**	**756**	**225**	**1584**	**889**	**1269**	**777**	**1489**	**1086**	**802**	**257**	**1350**

4 Background information for the scoring of potential indicators against criteria

In this chapter we will attempt to provide background information that can be used as guidance for the evaluation of potential indicators against the Rice and Rochet (2005) criteria.

For state indicators the starting point is table 4.1 in which each shaded cell represents an ecosystem feature that needs to be conserved in order for the whole ecosystem to be in a healthy state (see Piet & Pranovi 2005). For each shaded cell one or more indicators may be considered, preferably for different European regions. In addition to these state indicators we also identified the need for pressure indicators (see Piet & Pranovi 2005). In this chapter we aim to provide background information to score potential indicators against the criteria identified in chapter 1.1:

- Concreteness
- Theoretical basis

- Public awareness
- Cost
- Measurement
- Availability of historical data
- Sensitivity
- Responsiveness
- Specificity

This will not comprehensive but should allow a better informed evaluation and scoring of the indicators.

Table 4.1 Ecosystem features 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[2]	Habitat[3]	Ecosystem[3]
Macrophytes[2]				
Benthos				
Cephalopods[2]				
Fish				
Phytoplankton				
Zooplankton[1]				
Seabirds[2]				
Marine mammals[2]				
Marine reptiles[2]				
Physical				
Chemical				

[1] including large pelagic invertebrates such as jellyfish
[2] for components consisting of few species the community level may not be applicable
[3] Habitat or ecosystem level indicators may be an aggregate of one or more components

4.1 Concreteness

For the different sub-criteria of concreteness indicators exist that can be classified as one or the other:

- For the criterion “concrete property of physical/biological world, or abstract concept” applies that population level, community level or physical/chemical indicators are usually concrete properties of the physical/biological world while the ecosystem level indicators are abstract concepts
- For the criterion “Units measurable in the real world, or arbitrary scaling factor” applies that most state or pressure indicators are measured in Units measurable in the real world, not arbitrary scaling factors.
- The criterion “Direct observations, or interpretation through model” appears to be related to the first as most of the concrete properties are based on direct observations while the abstract concepts are mostly model-based.

4.2 Theoretical basis

For most indicators the theoretical basis was described in INDECO (2005) “A review of the indicators for ecosystem structure and functioning”.

4.3 Public awareness

The Public is usually more aware of issues dealing with the larger and more easily observed marine components e.g. marine mammals, seabirds and some of the commercial fish species. There is less interest in the smaller ecosystem components such as benthic invertebrates or the more abstract concepts of community or ecosystem. An exception to general lack of public awareness of community level issues may be that of biodiversity. The Convention on Biological Diversity, Agenda 21 and Annex V of OSPAR all place a legal obligation on signatories (including the EC and all European states that border and exploit the biological resources of the North Sea) to conserve biodiversity and to restore it in degraded systems. This has resulted in a relatively high public awareness of biodiversity.

4.4 Cost

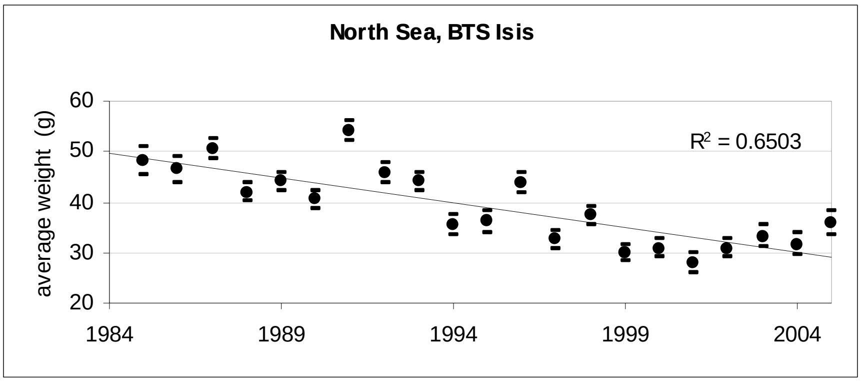
For many of these indicators it is important to make the distinction between fixed and marginal costs as in most cases the indicators can be calculated at only little additional expenses (marginal costs) from e.g. monitoring data or assessment results that are collected at considerably higher expenses (fixed costs), of the funded under the Data Collection Regulation, in support of other indicators or EU services.

An example of the fixed costs of two surveys are provided here as an indication. As EVHOE survey covers both the Bay of Biscay and Celtic Sea, and MEDITS the Gulf of Lions and East Corsica, the last line indicates the cost for the sub-regions considered elsewhere in this report, based on the number of hauls (EVHOE) or number of days at sea (MEDITS) in each sub-region.

Costs in euros	EVHOE		MEDITS	
Vessel costs	RV Thalassa	596,000	RV Europe	176,000
Onboard staff cost		246,000		126,000
Total		842,000		302,000
Sub-region	Biscay (½)	421,000	Lions (19/26)	226,500

4.5 Measurement

According to Rice & Rochet (2005) there are two main features of “Measurement” that determine the suitability of an indicator: variance and bias.
Variance can be determined for each yearly value in a time-series if this is based on several measurements each year (e.g. hauls in a RV survey) or between years for a specific time period.



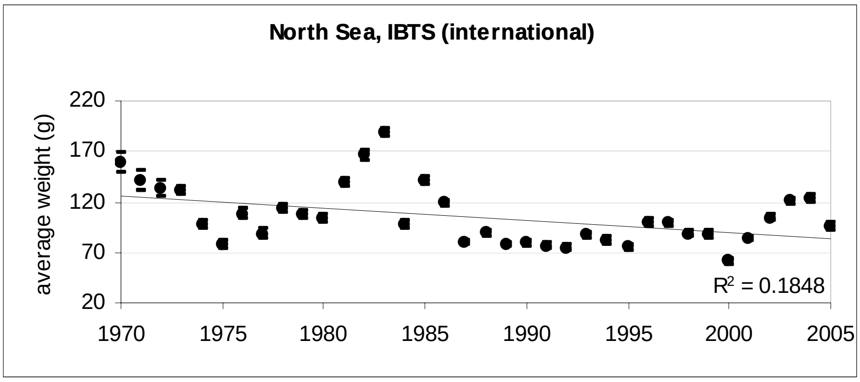


Figure 4.1. Time-series and variance within year in the average weight of the fish community in two North Sea surveys.

For two surveys (MEDITS and EVHOE) covering sub-areas of EU waters (respectively Gulf of Lions and Bay of Biscay) the mean and standard deviation of the Coefficient of Variance (CV) is given for some population and community level indicators of which the abbreviation is explained below:

Level	Indicator	Description
	ln(N)	Log abundance population i
Population	Z	Average total mortality rate for species i
	Wbar	Average weight of population i
	Lbar	Average length of population i
	Lvar	Variance in length in population i
	$L_{\%}$	Percentiles of length distribution of population i
	Lmat	Length at maturity (50% mature)
	Btot	Total biomass in community
	Ntot	Total abundance in community
Community		
	Delta	
	Lbcomm	
	pmoy	
	PropGx	Proportion of large individuals in total community abundance > x cm

Table 4.2. Indices of variance for various indicators based on two surveys (MEDITS and EVHOE) covering sub-areas of EU waters (respectively Gulf of Lions and Bay of Biscay)

Population level indicators: Mean CV (across species and years)

Survey	L5	L25	L75	L95	Lbar	Ln(N)	Lvar	Wbar
MeditsLion	0.108	0.075	0.056	0.045	0.03	0.018	0.142	0.505
EVHOE_Biscay	0.102	0.05	NA	0.022	0.023	0.023	0.131	0.526

Population level indicators: Standard deviation of CV

Survey	L5	L25	L75	L95	Lbar	Ln(N)	Lvar	Wbar
MeditsLion	0.122	0.11	0.072	0.054	0.041	0.01	0.13	0.262
EVHOE_Biscay	0.227	0.103	NA	0.037	0.027	0.012	0.12	0.258

Community level indicators: Mean CV

Survey	Btot	Delta	Lbcomm	Ntot	pmoy	PropG15	PropG20	PropG25	PropG30
MeditsLion	0.13	0.076	0.008	0.145	0.198	0.009	0.016	0.022	0.05
EVHOE_Biscay	0.18	0.1	0.0004	0.197	0.27	0.001	0.003	0.006	0.014

Community level indicators: Standard deviation of CV

Survey	Btot	Delta	Lbcomm	Ntot	pmoy	PropG15	PropG20	PropG25	PropG30
MeditsLion	0.077	0.076	0.009	0.064	0.093	0.001	0.003	0.003	0.005
EVHOE_Biscay	0.051	0.046	0.0001	0.06	0.068	0.0004	0.0006	0.002	0.004

The population level indicators with the best (1-3%) and least varying precision are log-abundance and average length. On the other hand, length variance has a low precision (12-14%), and average weight has a very low precision due to problems with weighing onboard vessels and rounding in data-bases. At the community level as well, length-based indicators are estimated with the best precision. Overall the indicators examined here are well estimated and their precision does not vary much from year to year.

Total mortality rate Z is very sensitive to the growth parameters. This was specifically analysed in the Gulf of Lions. For *Chelidonichthys gurnardus*, the only parameters available for the Von Bertalanffy Growth Function (VBGF) in the Mediterranean were from Stergiou *et al*. (1997), with: $t_0 = -1.99$, $k = 0.219$ and $L_{inf} = 26.4$. The application of these parameters suggested a number of year classes much higher than expected from the length distribution of this species in the Gulf of Lions (Figure nn). Indeed, considering the strength of the first two year classes, an assessment of the VBGF parameters has been done using the Bhattacharya and NORMSEP method (Gayanilo *et al.*, 2002). The application of the new parameters drastically reduced the estimated number of year classes (from about 20 to about 10).

The *Merluccius merluccius* caught during Medits in the Gulf of Lions are mainly young individuals (age 0-1). Consequently, the abundance is strongly related to the recruitment. So, the low value of Z age 0-2 in 1997 (Figure 4.3) is related to the very high recruitment of this species in 1998.

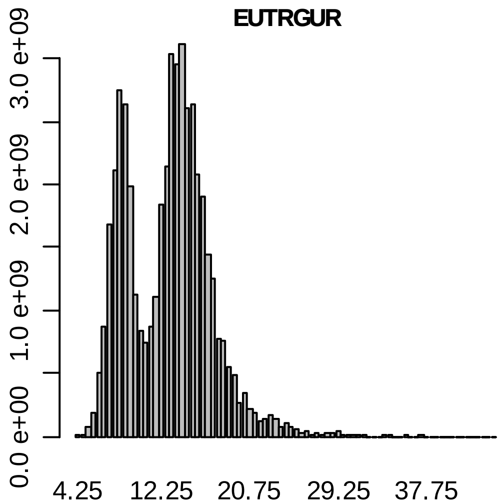


Figure 4.2. Cumulative length distribution of *Chelidonichthys gurnardus* in the Gulf of Lions (1994-2004).

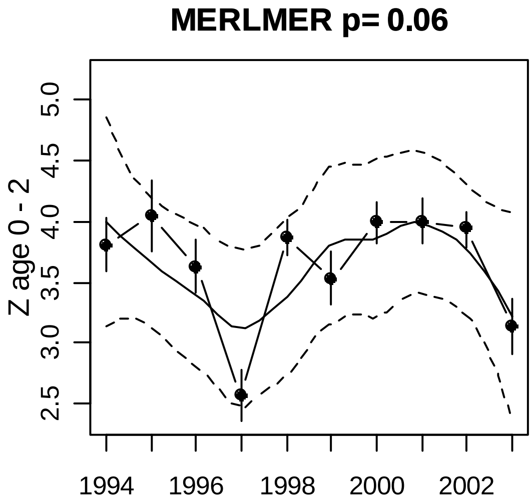


Figure 4.3. Cumulative mortality rate of *Merluccius merluccius* for ages 0 to 2 in the Gulf of Lions(1994-2004).

In conclusion, estimating Z from length-frequencies requires reliable estimates of growth parameters, and should not be attempted otherwise.

Another example of the variance between years for several indicators in a certain time period and depending on the level of fishing effort is shown in figure 4.4.

Power analysis is often used to determine how this variance between years affects our ability to detect trends in indicator time-series. Power analyses have been conducted on a variety of indicators. The results of this are shown e.g. in chapter 4.6.10 and INDECO (WP5 2nd deliverable insert ref).

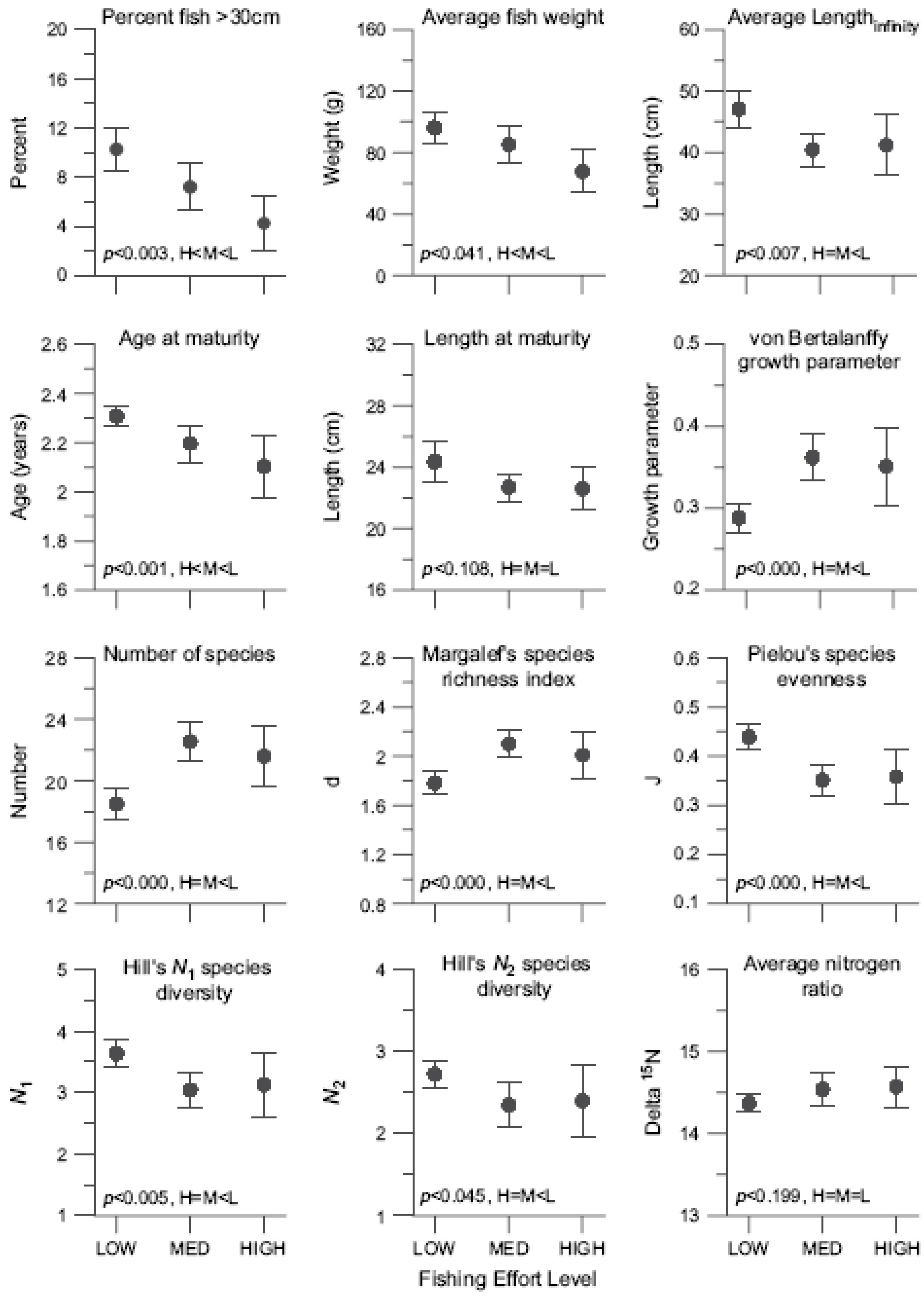


Figure 4.4. Mean fish community metric values (±95% confidence limits) for three fishing effort treatments in the period 1983-1996. ANOVA probability values are shown together with the ranking indicated by Tukey post hoc comparisons.

4.6 Historical data

The availability of historical data was explored for indicators of both state and pressure indicators. The sub-headings refer to ecosystem components and features shown in table 4.1.

4.6.1 Fish populations

In this chapter we present indicators that describe different features of fish populations that may be affected by fishing:

- Abundance

- Biological parameters
- Genetic changes

For indicators that describe the population status in terms of abundance of individual fish species or stocks it is relevant to distinguish between indicators based on 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. For the former we present the historical data for the indicator "Proportion of commercial stocks that are within safe biological limits" developed by Piet & Rice (2004), for the latter we present the historical data for the indicator "Threat indicator" developed by Dulvy et al. (2006) which essentially integrates the change in a group of populations.

4.6.1.1 Abundance of assessed fish species

For quantification of the indicator "Proportion of commercial stocks that are within safe biological limits" different approaches were applied for the regions for which ICES provides advice (e.g. North Sea, South-western waters and Baltic Sea) as opposed to the Mediterranean.

Regions covered by ICES advice

In these regions the indicator can be interpreted as 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 (SSB), also taking into account fishing mortality (F), 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 stock 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. F_{pa} is the fishing mortality at and below which the true fishing mortality has a low probability of leading to stock collapse. To evaluate the performance of fisheries management advice Piet & Rice (2004) identified the state of these stocks using precautionary reference points. Therefore three criteria were used to determine whether a stock was within safe biological limit, and hence that the objective was met:

- SSB was above the precautionary reference point (SSB>SSBpa)
- F was below the precautionary reference point (F<Fpa)
- Both the above (SSB>Bpa and F<Fpa)

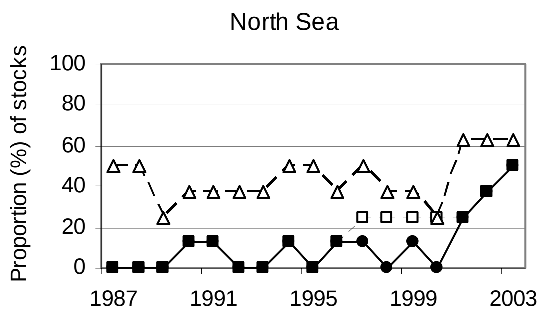
The suggested indicator is the proportion of commercial fish stocks within safe biological limits. The objective is that this indicator should be at a target level relative to a reference level which by definition is 100%.

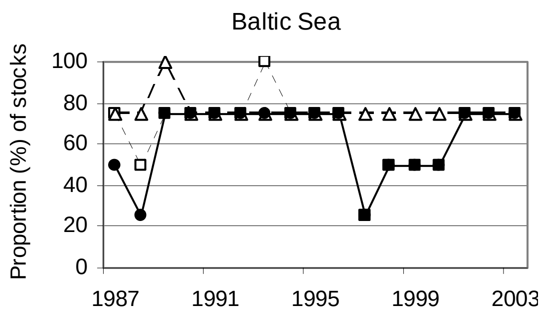
For indicators relating to spawning stock biomass (SSB) or fishing mortality (F), the appropriate source of information for the different stocks is the regular assessments by the ICES Working Groups reporting to ACFM (http://www.ices.dk/committe/acfm/comwork/report/asp/advice.asp) . Quantification of these indicators was based on the most recent ACFM advice available. Stock estimates are given per ICES area (figure 4.5). These ICES areas were attributed to the appropriate RAC areas. Where stocks cross boundaries of the RAC areas they will be attributed to the geographical areas based on the following interpretations:

- Stocks in ICES IIIa that are assessed along with stocks in the Baltic will be attributed to the Baltic region;
- Stocks in ICES IIIa that are assessed along with stocks in the North Sea will be attributed to the North Sea region;
- Southern Channel stocks (ICES area VIId) that are assessed along with North Sea stocks will be attributed to the North Sea region;
- Stocks in ICES-area VI of the eastern Atlantic that are assessed along with stocks in the North Sea will be attributed to the North Sea region;
- Bay of Biscay stocks that are assessed along with stocks in ICES area VII of the eastern Atlantic will be attributed to the North Western waters (e.g. Megrim);
- Stocks that are assessed throughout the Bay of Biscay, the eastern Atlantic and the North Sea will be attributed to the North Western waters (e.g. Hake).

When ICES assessments provide values of fishing mortality as F, F_{HC}, F_{disc} and F_{IB}, the general F was used for analysis. Of stocks that fall within the defined areas and for which ICES provides quantitative scientific analysis, stocks were excluded if:

- They are not assessed, and estimates of SSB are not available, even though they may be fished commercially;
- Values of SSB and F were displayed in graphs only;
- No precautionary levels of SSB of F were available.





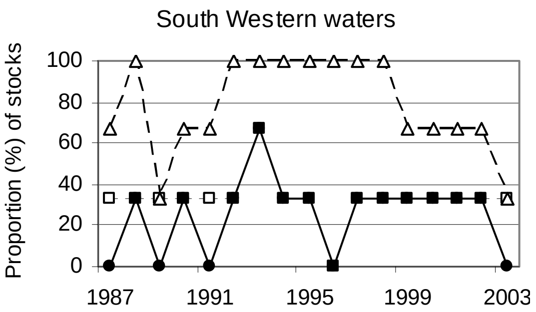


Figure 4.5. Proportion of stocks within safe biological limits, based on F (--□--), SSB (--Δ--) and SSB&F (−●−)

Baltic Sea

The open sea pelagic fish community in central Baltic Sea constitutes only few marine fish species and therefore it could also be possible to show the status of each of the populations separately instead of through the indicator "Proportion of stocks within safe biological limits" presented earlier. Herring and sprat make up the principal prey items of the piscivores cod and salmon. Both clupeid species feed on zooplankton and herring also consumes nektobenthos. Thus, they comprise an intermediate link between the higher and lower trophic levels in the sea. The Baltic Sea is subjected to a high fishing pressure, mainly directed towards cod, herring and sprat. Due to natural variation and anthropogenic pressures to the fish community, the sea has undergone severe changes during the last decades. Cod biomass has been reduced from above 1,000,000 tons in early 1980s, to less than 200,000 tons in early 1990s, and the stock has not recovered since then (Figure 4.6, Casini 2006). Also herring biomass has decreased since mid 1970s, from 3,000,000 tons to below 300,000 tons in late 1990s. In contrast, sprat biomass increased in late 1980s and peaked in mid 1990s.

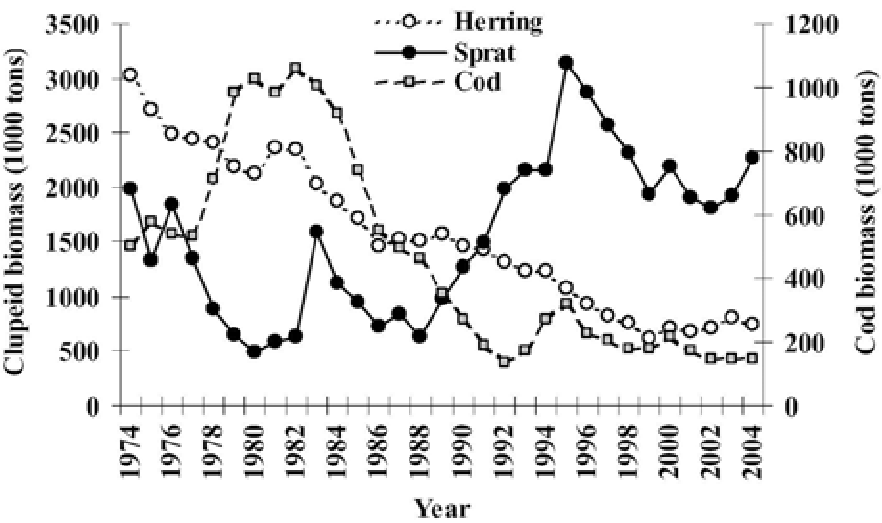


Figure 4.6. Biomass of cod, herring and sprat in central and western Baltic Sea 1974 to 2004. Cod refers to ICES SD 25-32, sprat SD 22-32 and herring SD 25-29 and 32, Gulf of Riga excluded. (From Casini 2006).

Causes to the substantial changes in fish biomass over time possibly relates to several, partly confounding, factors. Climatic variations resulting in an unusual warm period in the 1990s, low inflow of marine water from the North Sea, as well as an ongoing eutrophication possibly play an important role. However, fishing is regarded to be one of the major causes. For cod, fishing mortality (F) decreased in the 1970s, followed by an increase in the 1980s and a dramatic drop in 1992 (Figure 4.7). Fishing mortality of herring increased significantly until late 1990s, where after it has decreased to low levels again during the last six years.

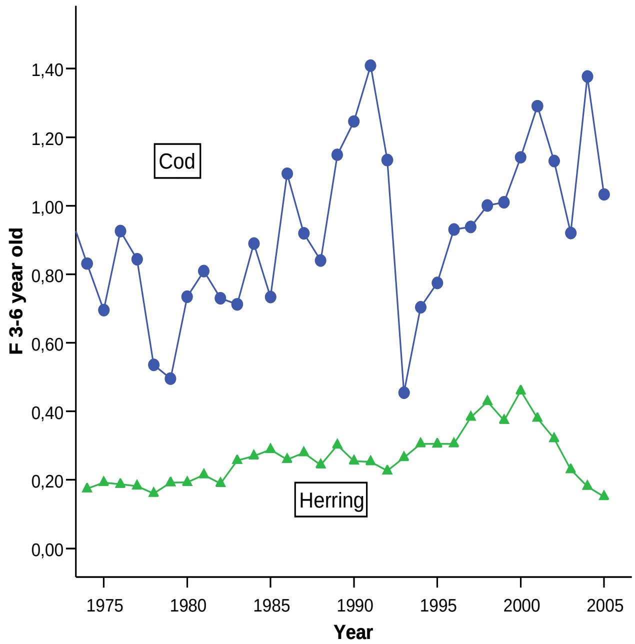


Figure 4.7. Fishing mortality (F) in 3-6 y old cod (in SD25-32) and herring in SD25-29 (excl. Gulf of Riga) years 1974-2005. Data from ICES 2006a.

Mediterranean

The General Fisheries Commission for the Mediterranean (GFCM) of the FAO, through its Scientific Advisory Committee (SAC), provides a forum for countries to attempt to coordinate assessment and management activities, and is the principal decision-making body. The implementation of the decisions of the GFCM is aided by a series of sub-regional fisheries projects

Examining the GFCM-SAC assessments over the past 5 years (2000-2004) (GFCM-SAC, 2003), it is apparent that only a limited number of stocks is assessed and not even in a consistent and systematic manner. An example is given in figure 4.8 using two wider areas covering the EU waters of the Western Mediterranean corresponding to FAO's Balearic and Sardinia areas. In the first area only 8 species are assessed and in the latter 6. Assessment of demersal species is poor in the first area and in the second assessment of pelagics is missing. In Sardinia demersal stocks in sub-areas are characterised either as Overfished or Fully fished. Note: In Sardinia where stocks have been assessed both as OF and FF we considered them overfished.

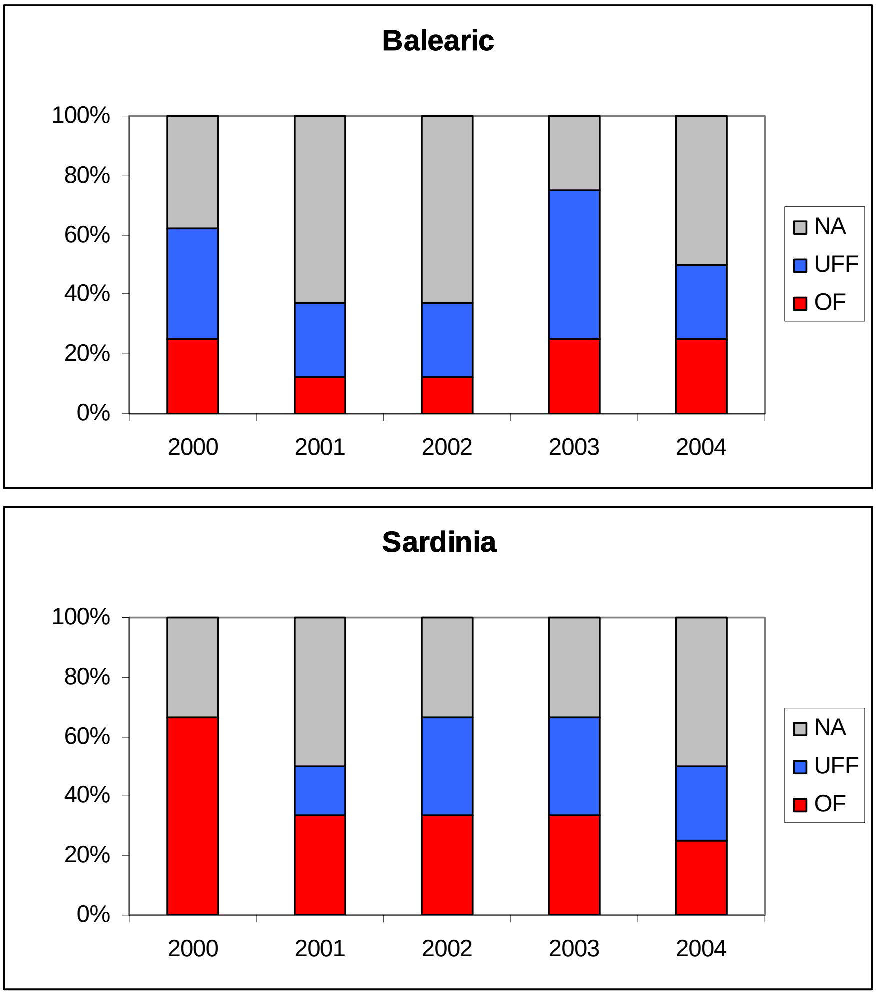


Figure 4.8. Time-series for two areas in the Mediterranean. OF: Overfished UFF: Underfished or Fully fished, NA: No Assessment in particular year

4.6.1.2 Abundance of non-assessed fish species

For the non-assessed species we present two indicators: The first indicator the proportion of hauls in a particular survey (here the IBTS) in which fish at a specific higher taxonomic level were observed, the second is the "threat indicator" (Dulvy et al. 2006).
For the first we used the elasmobranchs as these are known to be one of the most vulnerable groups to fishing because of their life-history characteristics.
Time-series of the two elasmobranch indicators based on BTS and IBTS showed no distinct trend in either time-series, and lots of variation (Figure 4.9).

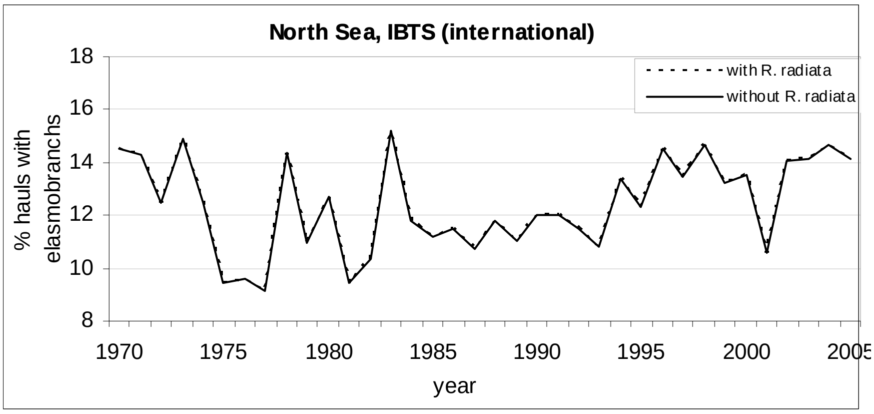


Figure 4.9. Time-series of the proportion (%) of hauls per year with elasmobranchs in it. We distinguished between hauls with any of the elasmobranch species or excluding *R. radiata*.

The second indicator, the threat status of a suite of marine fishes, was based on the English Ground Fish Survey (EGFS). The suite of species represents the section of the fish fauna likely to be most vulnerable to the effects of fishing and also to represent a key area of public concern for the larger bodied more charismatic megafauna. Relative abundance is converted to an internationally recognised measure of threat for each species over time by applying the World Conservation Union (IUCN) A decline criteria to fisheries-independent survey abundance data. A composite threat index is derived from averaged weighted species threat scores in each year.

The North Sea English groundfish survey (EGFS) data were used as a measure of abundance of adult fishes. A survey grid of 75 stations is fished annually. Stations were fished with a Granton demersal trawl until 1990, but from 1991 a Grand Ouverture Verticale (GOV) demersal trawl was used. Tow duration up to 1991 was 60 min, for 1992 onwards the tow duration was 30 min (B. Harley pers. comm.). The Granton trawl gear was fitted with a cod-end liner of 14 mm stretched mesh and the GOV trawl was fitted with a cod-end liner of 20 mm stretched mesh. Both gears were towed at a speed of approximately 4 knots. All fishes caught were identified and measured. Catch rates were raised to numbers or biomass caught per 60 min tow. Not all stations in the survey grid are fished every year due to poor weather, equipment damage or ship failure, and in the earlier surveys more stations were sometimes surveyed (for more details see Maxwell and Jennings 2005).

Species were excluded if they were known to be poorly sampled by the gear, rare or found in peripheral North Sea habitats and had a maximum length of <40 cm (Sparholt 1990; Knijn et al. 1993; Maxwell and Jennings 2005). Specifically, species were excluded if <150 individuals had been caught in the history of the survey, or if morphology, behaviour and habitat preference was expected to lead to very low and variable catchability. Individuals <40 cm have increased in abundance in recent years, possibly as a result of the depletion of their larger predators, so we restricted this analysis to species with a maximum size greater than 40 cm total length (Daan et al. 2005). The twenty-three species retained for analysis were representative of the breadth of morphology, life histories, ecology and taxonomic diversity of the larger bottom dwelling fishes sampled on the English groundfish survey in the North Sea. The average age of maturity of all species in this suite of fishes was 4.9 years.

Threat was assessed using IUCN A decline criteria which are based on the reduction in population size over the greater of 10 years or three generations where causes are reversible, understood and have ceased (IUCN 2004). The qualifying decline thresholds are *Critically Endangered* - ≥90% decline, *Endangered* - ≥70% decline and *Vulnerable* - ≥50% decline. We measured threat retrospectively over the time series using the 'extent of decline'. Extent of decline was calculated by comparing subsequent changes in abundance to a fixed start date of 1982. A linear model was fit to the first 10 years of data, $t_1 – t_{10}$ and to each successive year, i.e. $t_1 – t_{11}$, $t_1 – t_{12}$,…, $t_1 – t_{maximum}$. The percent change in abundance was calculated from the start (t_1) and end (t_{10} to $t_{maximum}$) abundances as predicted from the least squares linear model fit (IUCN 2004). Species that had met one of the decline criteria qualifying as threatened (Critically Endangered, Endangered, Vulnerable) in any year were not delisted (categorized as not threatened) unless their abundance had increased beyond a preset threshold. As an example, we chose a preset threshold of the mean catch rate averaged over the first three years of the time series. This 'baseline' is a compromise, as the real baseline is unknown and also a three year span was chosen simply to provide a reasonable estimate of threshold abundance, as individual annual abundance estimates can be highly variable (Maxwell and Jennings 2005).

The composite threat indicator was calculated for each year as the average of weighted species threat scores. Individual species threat categorisations were weighted as *Critically Endangered* = 3, *Endangered* = 2 and *Vulnerable* =1 (following Baillie et al. 2004), and allocated to the final year of the period over which the decline was measured. A composite threat indicator was calculated as the average threat score of all species for each year. This indicator is readily interpreted, the scores can vary from 0 to 3, such that a score of 0 is equivalent to no species meeting any of the threat criteria to a score of 3 is equivalent to each species being *Critically Endangered*.

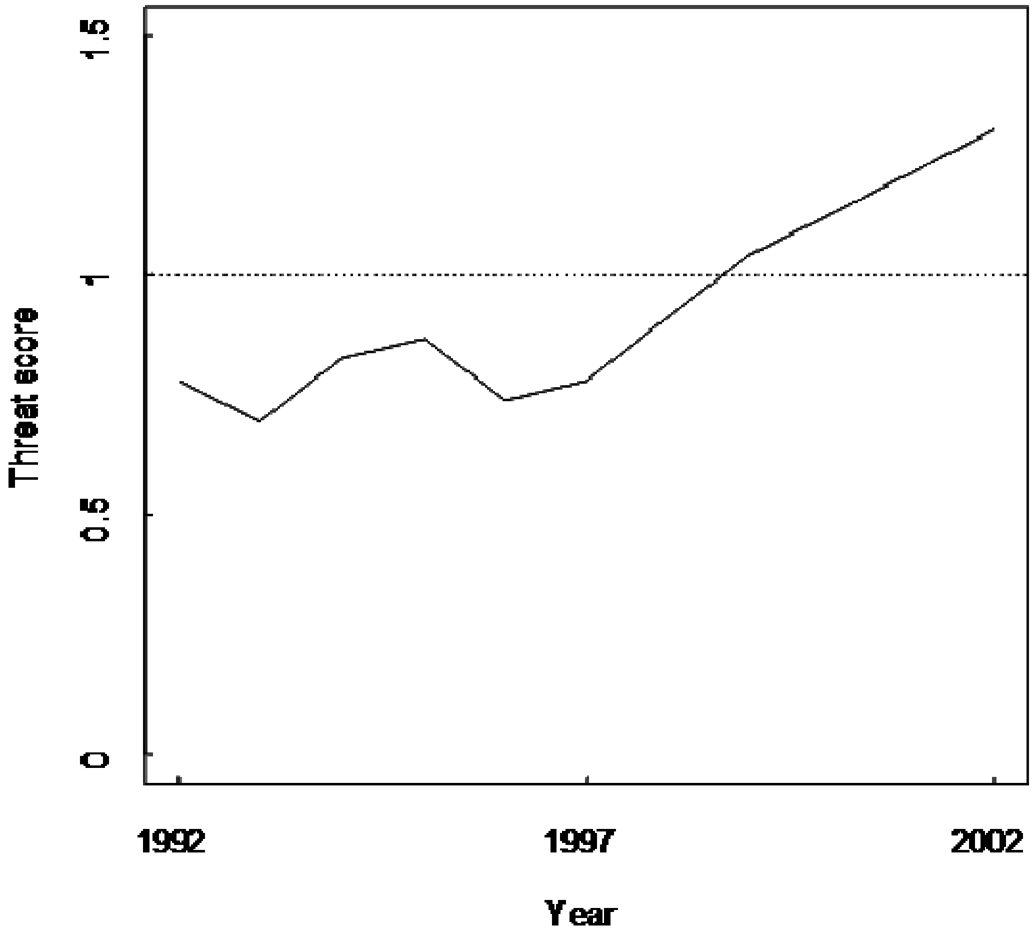


Figure 4.10. An indicator of threat over time for a suite of 23 North Sea demersal fishes measured as ‘extent of decline’ from the start of the time series. A score of 1 is equivalent to each species meeting the *Vulnerable* criterion and is indicated with a dotted line.

4.6.1.3 Biological parameters

For fish populations that are caught in Research Vessel (RV) surveys a number of other population indicators can be calculated (see table 4.3).

Table 4.3. Description of selected indicators and the expected effect of fishing on them.

Level	Indicator	Description	Expected effect of fishing
	ln(N)	Log abundance population i	decrease
Population	Z	Average total mortality rate for species i	increase
	Wbar	Average weight of population i	decrease
	Lbar	Average length of population i	decrease

Lotka's intrinsic population growth rate r has been suggested for fish populations (Quinn and Deriso, 1999). It can be estimated as the slope of log abundances against time. If r is significantly lower (respectively higher) than 0, the population is decreasing (increasing). The expected and undesired effect of fishing is to slow population growth - although many other factors might also have the same effect. Taking r = 0 as the target reference point assumes that without any noticeable impact of fishing the population would be stable although varying between years.

Total mortality rate Z is classically written as the sum of natural and fishing mortality rates M and F and will clearly increase under exploitation. It can be estimated from numbers at length data, and has been suggested as a robust indicator for exploited populations (Die and Caddy, 1997). The proposed estimation method for total mortality is based on a simple age-structured population dynamics modelt. Pseudo-ages are obtained using the von Bertalanffy growth function. The proposed indicator consists of the total mortality for a given age range (age_min to age_max), i.e. the mortality rate of all individuals aged a_min to a_max-1 between years t-1 and t.

Age at maturity has been found to decrease under the effect of fishing (Trippel, 1995; Rochet *et al.*, 2000). If individual growth remains similar under the impact of fishing, length at maturity will decrease in a similar manner. Compensatory growth will to some degree reduce the impact of fishing on the observed reduction in length at maturity but strong signals should still be detectable.
Average length in the population is expected to decrease in exploited stocks (Beverton and Holt, 1957), and this has been observed in various situations (Haedrich and Barnes, 1997; Babcock *et al.*, 1999).

For herring in the Baltic, normalised mean weight of two-year-old individuals was found to decrease over time (Fig 4.11).

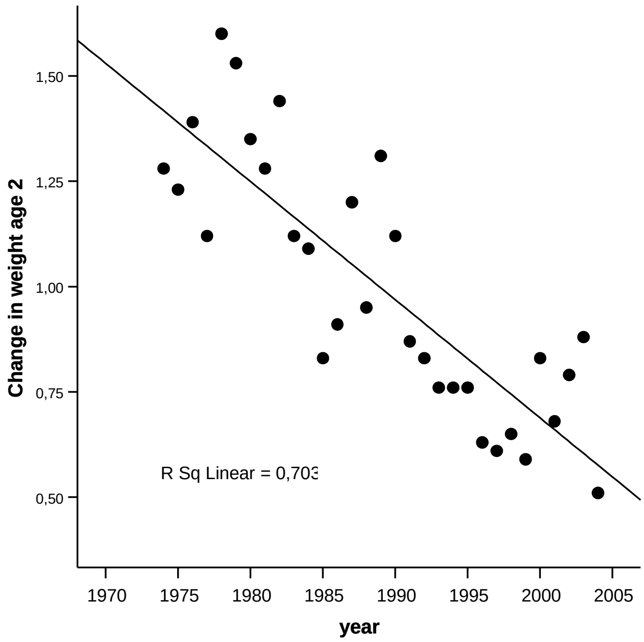


Figure 4.11. Normalised mean individual weight of two-year-old Baltic herring in SD25-28. Data from ICES 2005b.

To what extent this decrease in mean weight is related to fishery is not clear, and several contrasting explanations have been put forward (Sparholt 1996). Among them, it has been suggested that biomass of the zooplankton *Pseudocalanus elongatus* have declined due to reduced inflow of saline North Sea water to the Baltic Sea. Being a main prey item for herring, this decline is suggested to cause reduced condition of herring (Möllman *et al*. 2003, Figure 4.12).

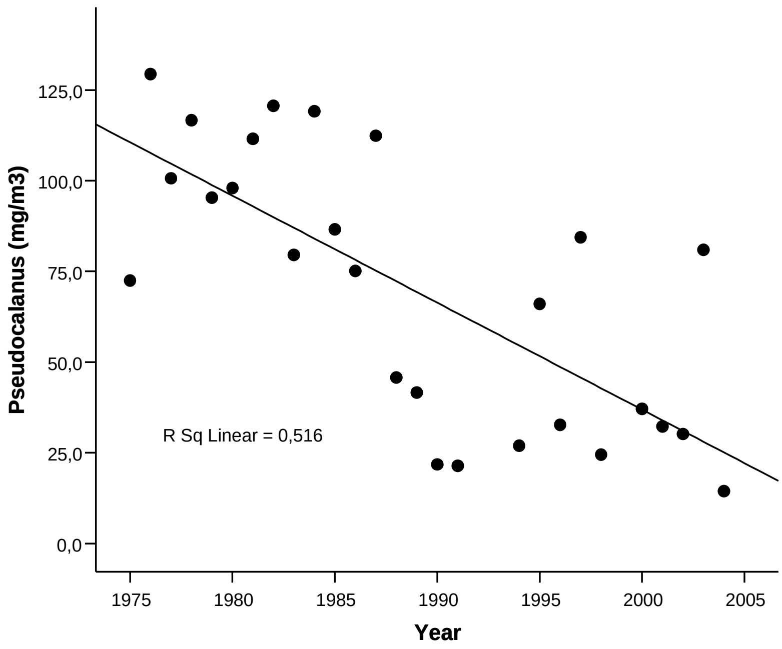


Figure 4.12. Summer biomass of Pseudocalanus (mg/m^3). Data from Christian Möllman, DIFRES.

However, recent studies suggest that a decline in herring condition could be related to a density dependent top-down regulation on a common food resource by the clupeid species (sprat and herring) in the sea (Casini *et al*. 2006). Thus, the observed change in individual mean weight of two year old herring may be a cascading effect of the substantial reduction of cod biomass, relaxing predation pressure on sprat biomass, thereby increasing inter-specific interaction between sprat and herring. Both hypotheses are to some extent supported by the positive relation between *Pseudocalanus* volume during summer and individual mean weight of two-year-old herring (Figure 4.13).

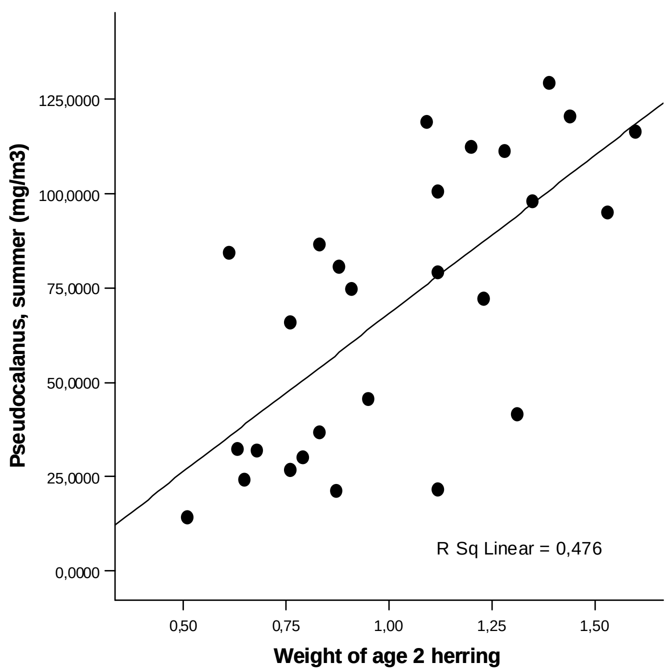


Figure 4.13. Relation between Pseudocalanus biomass (mg/m3) in central Baltic and normalised weight of two- yr-old herring (data from Christian Möllman, DIFRES, and ICES 2005b)

Another example of a size related metrics is the reduction in cod mean size over time in the Baltic Sea. Maximum mean weight of cod was estimated for the period 1988 to 2004 in the areas SD25-28 using fishery independent data from BITS (ICES 2005a, Figure 4.14)

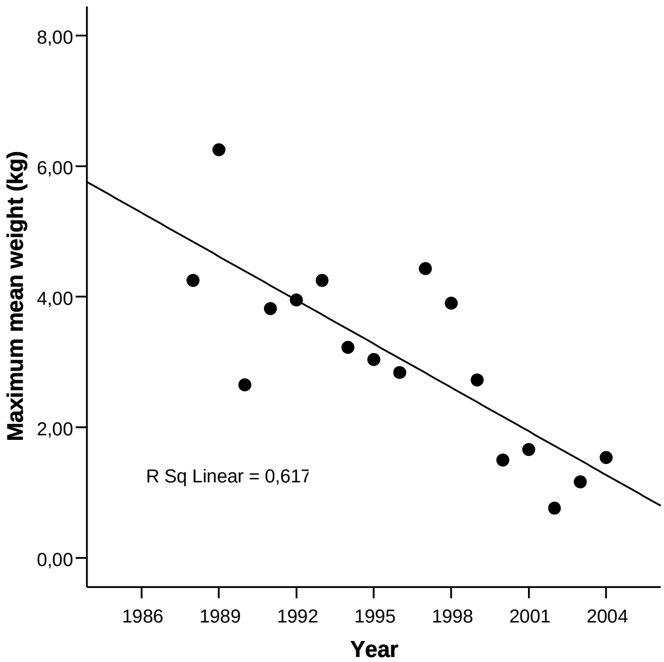


Figure 4.14. Maximum mean weight of cod in the Baltic Sea, years 1988-2004. Data from Swedish R/V Argos surveys in BITS, (ICES 2005a).

As fishing mortality of cod has been high and sprat biomass, which is the major prey item for cod, has increased over the time period, it seems plausible that the intense fishing pressure on cod is the main cause to the reduction in cod size over time.

The above examples point to the problem in finding straightforward relations between pressure and state indicators. Changes in size related population metrics may not only be the direct result of fishing pressure (as in the cod example), but also reflect indirect effects caused by environmental changes and/or more complex cascading effects.

4.6.1.4 Genetic effects

Genetic diversity represents the existence of variants (alleles) of individual genes. The alleles of a particular gene may occur in different frequencies in different populations, and the genetic variation of a particular species is therefore distributed both within populations (as different allele combinations between individuals) and between populations (differences in occurrence and frequency between populations). Exploitation can cause loss of genetic diversity, “genetic erosion”, through selective mortality on genotypes (e.g. fast growing fish) or eradication of local populations.

Exploitation can not only reduce genetic diversity, but can also change gene frequencies and hence the traits that these genes code for. Because of size-selective fishing practices commercial fishing selectively removes fast growing and/or late maturing individuals from exploited populations. Fishing can thereby alter the natural genetic variation in growth rate and maturation within populations, and hence induce evolutionary changes in these life history characteristics (Law 2000). Individual growth and maturation are also strongly influenced by the environment and population size, and can vary considerably without any changes in the underlying genetic components (so called plastic changes). Any indicator of the genetic variation in exploited populations thus needs to separate trait variation occurring because of plastic effects from those caused by changes in the genetic basis for that trait.

The recently developed probabilistic reaction maturation norm (PMRN) method is one way to disentangle plastic changes in maturation from possible genetically based changes in maturation (Heino et al. 2002, Barot et al. 2004a). The PMRN method is a probabilistic extension of the maturation reaction norm concept. A reaction norm describes the phenotypic expression of a genotype in different environments (Schmalhausen 1949). The *maturation* reaction norm concept is based on using variation in individual growth as a proxy for environmental variation to obtain sizes and ages of maturation, supposedly representing one and the same genotype (Stearns and Koella 1986). Although there is most likely no simple genetic basis for maturation and nothing such as a maturation genotype, the concept of the maturation reaction norm allows filtering away some of the plastic changes in maturation from possible genetic ones. The original maturation reaction norm concept (Stearns and Koella 1986) was determinate, whereas the PMRN is a probabilistic extension of it. The PMRN method derives the *probability* that an individual of a certain age and size *will mature* before the next season. This probability of maturing is then, by definition, independent of the probability that an individual will reach a certain age and size (Heino et al. 2002). That is, the probability of maturing is independent of variations in growth and survival. Changes over time in the probabilistic maturation reaction norm can thus reflect changes in maturation that are not caused by changes in growth or survival, such as possible genetic changes in maturation. Application of this technique to e.g., the Northern cod stocks (Olsen et al. 2004) suggest that changes in PMRNs could serve as a warning signal of changes in important life history characteristics even before drastic declines in population size occur.

The probabilistic maturation reaction norm should not be confused with what is known as maturity ogives. The PMRN describes the probability of *maturing* at a given age and size, whereas the maturity ogive describes the probability of *being mature* at a given age and size. Maturity ogives, which at the population level represent the proportion of individuals that are mature at a certain age (or size), are commonly estimated and used in stock assessments. The probability of *being* mature (the ogive) does not only depend on the maturation process but also on growth and survival before and after maturation, and hence cannot be used to disentangle plastic and possible genetic changes in maturation. In contrast, the PMRN method accounts for such confounding effects of mortality changes and growth-mediated phenotypic plasticity. Thus, the appropriate indicator to use is the age and size at maturation, as derived using the PMRN method.

The probabilistic maturation reaction norm (i.e. the probability of maturing) is derived from the maturity ogive (i.e., the probability of being mature) and from the mean annual growth at age as

$$m(a,s)=(o(a,s)-o(a-1, s-\Delta s(a)))/(1-o(a-1,s-\Delta s(a)))$$

where a is age, s is length, $o(a,s)$ is the maturity ogive, and $\Delta s(a)$ is the length gained from age a-1 to a. Estimation of the probabilistic maturation reaction norm thus involves (i) estimation of maturity ogives, (ii) estimation of growth rates (from length at age), (iii) estimation of the probabilities of maturing, and (iv) estimation of confidence intervals around the obtained maturation probabilities (see Heino et al. 2002, Barot et al. 2004 for further details).

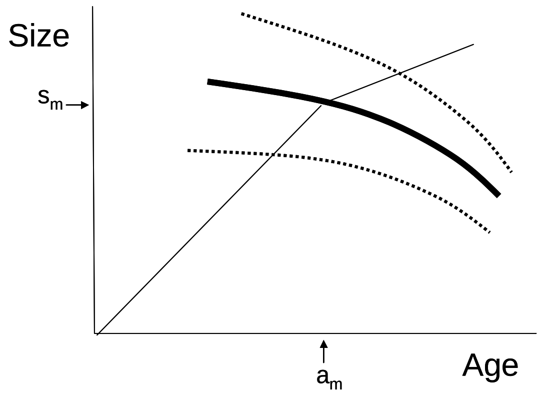


Figure 4.15. The probabilistic maturation reaction norm, i.e., the probability that an individual of a given age and size will mature before next season. Dashed lines indicate 25% and 75% probability to mature, whereas the intermediate thick line is the 50% probability to mature, also known as the maturation reaction norm midpoints. As an example, the thin line illustrates the growth rate of an individual, and its corresponding size and age at which it has a 50% probability to mature is indicated arrows on the corresponding axes.

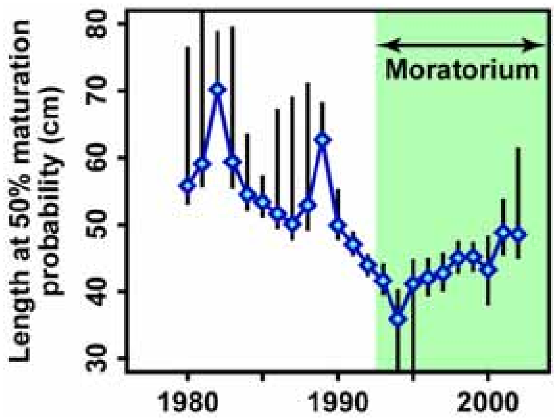


Figure 4.16. Length at 50% probability of maturing in five-year-old females of northern cod off the Canadian coast (NAFO division 2J). Redrawn from Olsen et a. 2004.

The probabilistic maturation reaction norm is not an appropriate indicator since it has infinite dimensions, as it involves specifying the probability of maturing for all relevant ages and sizes. Instead we suggest to concentrate on one of the probabilities that is part of the reaction norm: the length at which individuals have 50% probability of *maturing*, which is the so called mid-point of the PMRN (Heino et al. 2002). This can be obtained by fitting logistic regressions to the estimated probabilistic maturation reaction norms (Barot et al. 2004a).

Again, please note, that this indicator should not be confounded by parameters that are part of the maturity ogive, such as the L_{50} (or A_{50}), which is the length (or age) at which there is a 50% probability of *being mature*. A possible state indicator to be used to measure the "genetic effects" of fisheries (SGRN 2005) is the length of 50% probability of maturing calculated per age for the ages that are most impacted by fishing (which ages these are will have to be decided on a population level), for species that are targeted by commercial fisheries in the ecosystem. This indicator should not be applied for abundant species that are not affected by commercial fisheries, since these are the ecosystem components least likely to show any presumably genetic response in maturation to fishing. An example of how this indicator performs over time in exploited fish populations is given in figure 4.16, showing the length of 50% probability of maturing in one of the northern cod stocks off the Canadian coast, prior to its collapse and after the fishing moratorium (Olsen et al. 2004).

The data needed to calculate this indicator is fisheries-independent data with individual measurements of age, length, sex and whether the individual is juvenile (i.e. immature) or adult (i.e. mature) (Heino et al. 2001, Barot et al. 2004a). These measurements all need to be taken for each individual collected. If resting individuals (i.e. individuals that are mature but do not spawn in the sampled season) can be mistaken for juveniles, individuals need to be classified as either a juvenile, an adult that spawn(ed) within the season or a resting adult individual. Many existing schemes for classifying gonadal maturity status by macroscopic observation that are in use on RV surveys allow grouping of individuals into these categories (e.g., ICES 2000). A recent development of the original method allows the derivation of probabilistic maturation reaction norms without individual growth data (Barot et al. 2004a), and it has been shown that the method is robust to this simplification if there are individual measurements (age, length, sex and juvenile/spawning adult/resting adult) of at least 100 individuals per age and year-class (Barot et al. 2004a). These age groups need to contain both juvenile and adult individuals. That is, completely juvenile age groups or age groups in which all individuals are adults do not have to be sampled *if* they stay so during the whole monitoring period. The necessary sample size in a given catch year can thus be derived from the number of age groups in the population.

It is important to point out that change in PMRNs and, hence, also in length at 50% probability of maturing can be very rapid. For example, in the Northern cod length at which there was a 50% probability of maturing for five-year-old females in NAFO division 2J decreased by almost 25% in less than two generations, from about 60 cm in early 1980s to 45 cm before the closure of the fishery in 1992 (Figure 4.16; Olsen et al. 2004).

The general prediction based on life history theory is that selective fishing on large individuals will decrease the age and size at maturation (Law 1979, Michod 1979), and correspondingly also push the PMRN toward smaller ages and sizes (Ernande et al. 2004). This has also been observed in at least 11 of the 14 stocks to which this method has already been applied, including stocks of important commercial species such as cod (Heino et al. 2002, Barot et al. 2004b, Olsen et al. 2004, Olsen et al. 2005), plaice (Grift et al. 2003), and American plaice (Barot et al. 2005). Thus, both theory and empirical studies point to a general reference direction for PMRNs and the suggested indicator in response to size-selective exploitation: length at 50% probability of maturing per age will decrease in response to fishing of large individuals.

However, the extent of the decline in length at 50% probability of maturing will depend on the shape of the probabilistic maturation reaction norm (cf. Ernande et al. 2004). The shape of the PMRN depends on population specific characteristics, and hence any reference points or target levels of length at 50% probability of maturing per age will have to be determined not only on a regional basis, but at a population (or stock) level (and be specified per age).

A major challenge to the successful implementation of this indicator is species in which there are problems with the determination of age. This is the case for some of the major commercially targeted stocks such as the Atlantic hake and the Baltic cod (ICES 2006). Whether proxies for age determined from e.g. otolith weights or otolith shapes instead of growth rings on otoliths or scales, can be used as surrogates for traditionally assessed ages in the PMRN requires further research.

The length at 50% probability of maturing as derived using the probabilistic maturation reaction norm method is recommended as a specific indicator for “genetic effects” for immediate application. It must be noted, however, that this indicator is not a direct measure of genetic diversity, but it is based on phenotypic characteristics. That is, although the method accounts for phenotypic plasticity effects mediated through growth (in length) and survival the remaining changes in PMRN cannot be unequivocally attributed to genetic changes in maturation. Nevertheless, the midpoints of the PMRN (i.e. the length at 50% probability of maturing) can be considered a useful indicator of fisheries-induced genetic changes in maturation.

To obtain a more direct measure of “genetic erosion” genetic sampling has been proposed, e.g. genetic analyses on material obtained through fin-clipping on surveys (RCM North Sea report 2005). However, this is an issue that requires substantial research effort to be developed into an indicator, for several reasons. First, existing methods for genetic analyses of diversity such as microsatellite techniques or RAPDs are costly (especially for species for which primers have not been developed). Second, microsatellites and RAPDs are both based on non-coding genes, that is, genes that do not correspond to any traits of relevance. Thus, although microsatellite analyses would indicate a decrease in genetic variation it is not sure that there is a concurrent decrease in the genetic variation in relevant traits (although this is often assumed in studies applying microsatellite techniques). However, indicators based on direct genetic measures are needed to assess the genetic variation allowing local adaptations, and ensuring that fisheries do not compromise the future adaptive ability of species in exploited ecosystems, but any such indicators require substantial further research.

4.6.2 Fish community

All fish community indicators are based on RV monitoring programmes. One or more of these are conducted in each of the regions (see chapter 5 and Annexes in INDECO deliverable 5/6/7). Below examples will be provided for each of the regions.

Piet & Jennings (2005) show time-series based on historical data based on these two surveys (figure 4.17).

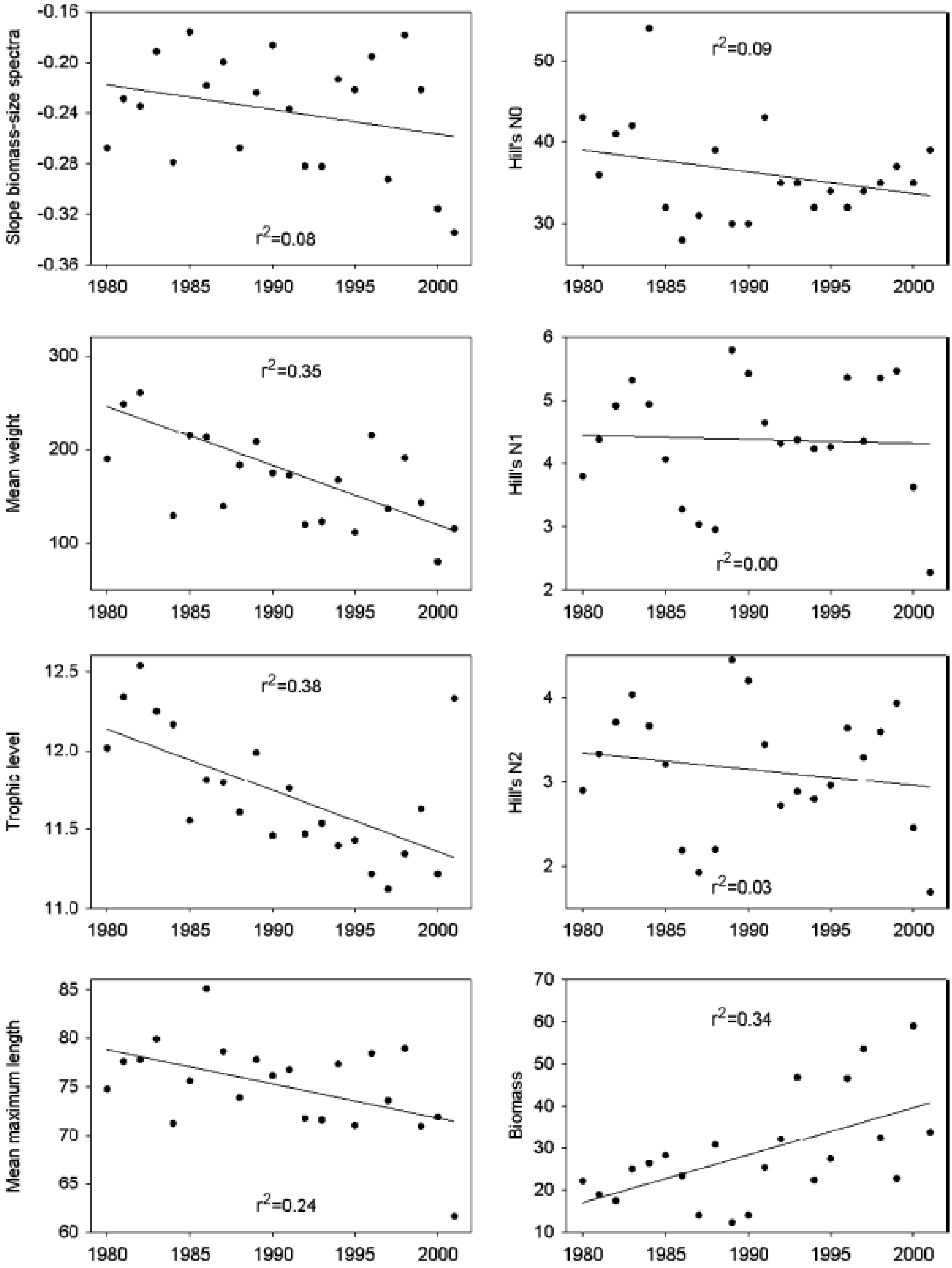


Figure 3. Time-series and trends of eight indicators calculated from the IBTS data.

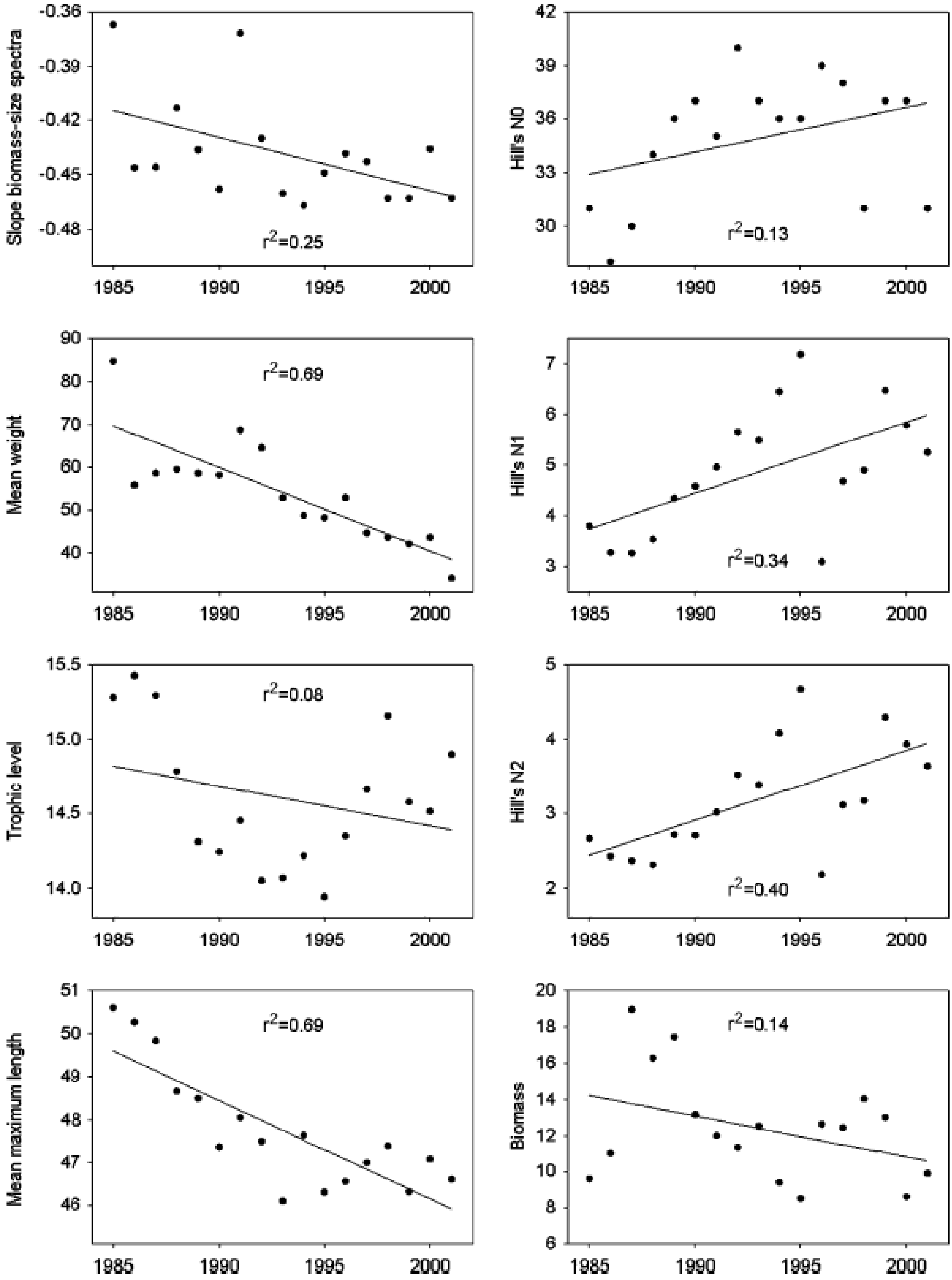


Figure 4. Time-series and trends of eight indicators calculated from the BTS data.

Baltic Sea - high fishing pressure

The demersal fish community in central Baltic Sea is dominated by cod, but also include flatfishes, salmonids, coregonids among other fish species. Data from BITS, collected by the Swedish R/V Argos, are used to demonstrate changes in mean weight of the fish community years 1987-2004 (Figure 4.18).

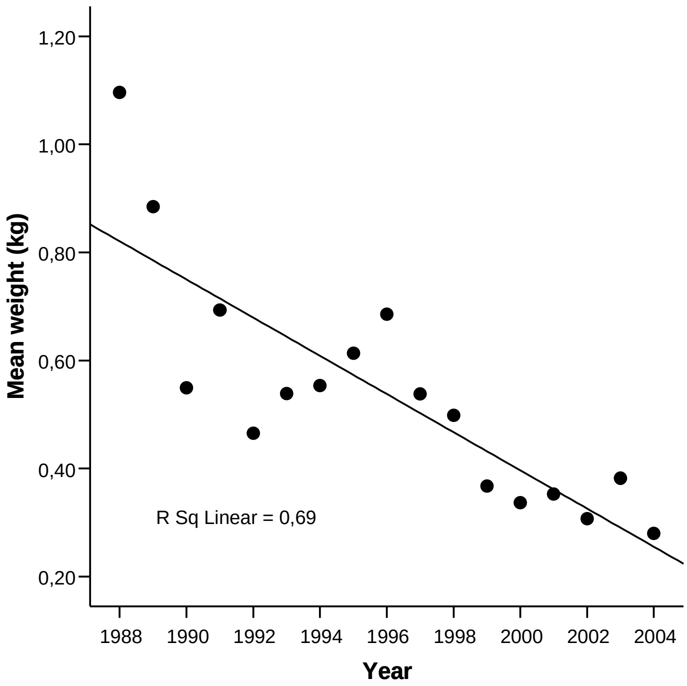


Figure 4.18. Mean weight of non-pelagic fish species in Baltic Sea (SD 25-28) years 1988 to 2004 based on BITS data (ICES 2005a).

It is reasonable to suggest that the observed reduction in mean weight of caught fish is related to a high fishing pressure directed towards demersal fish species, especially cod. However, when estimating maximum mean weight, both large sized salmon and sea trout will confound the results as these two species are less affected by the cod, sprat and herring fishery in the Baltic Sea (Figure 4.19).

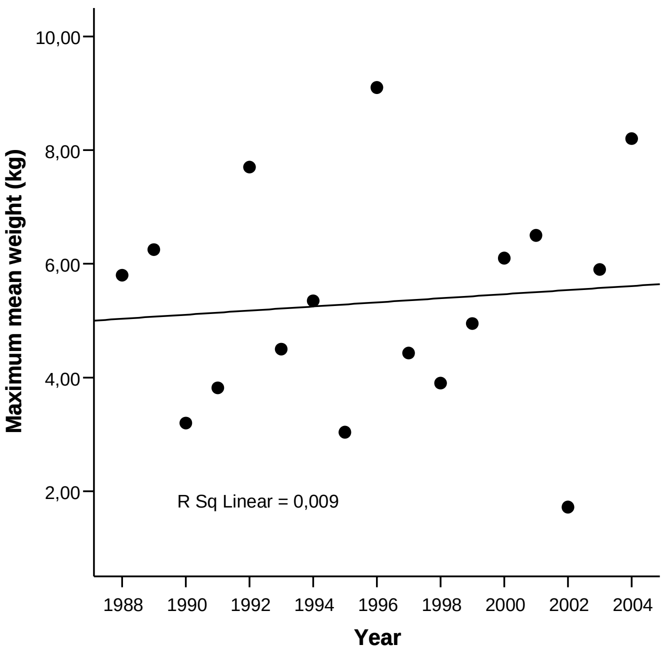


Figure 4.19. Maximum mean weight of non-pelagic fish species in Baltic Sea (SD 25-28) years 1988 to 2004 based on BITS data (ICES 2005a).

HELCOM data from the annual coastal monitoring in the western Estonian archipelago (the Sea of Straits) illustrate the effects of fishing on community size-spectra of coastal fish. During the investigation period (1993-2005), sampling was performed annually in August at 2–5 m water depths using multi-mesh gill net sets (Thoresson 1993). The smallest length group of caught fishes (<15 cm TL) is not fully selected. Therefore this size groups were excluded from further analysis in order to avoid problems caused by selectivity patterns.

Results expressed as slopes in ln(length) of all caught species show a negative trend over the studied period (Figure 4.20), where slope values decrease from around -3.4 to -6.9. The decrease corresponds to a substantial increase in fishing effort in early 1990s after the independence of Estonia (Vetemaa *et al.* 2005). Although the effort decreased in the late 1990s, the intense earlier fishing has been followed by tremendous changes in the fish community. The calculated slopes thus confirm other observations on less abundant large species and changes in biodiversity. The results should be used to indicate long-term trends rather than annual variations since annual perturbations in environmental or anthropogenic factors might confound the analysis. The important consideration is not the slope or intercept of a spectrum for a particular year, rather how the spectrum changes over a longer period of time (Rice& Gislason 1996).

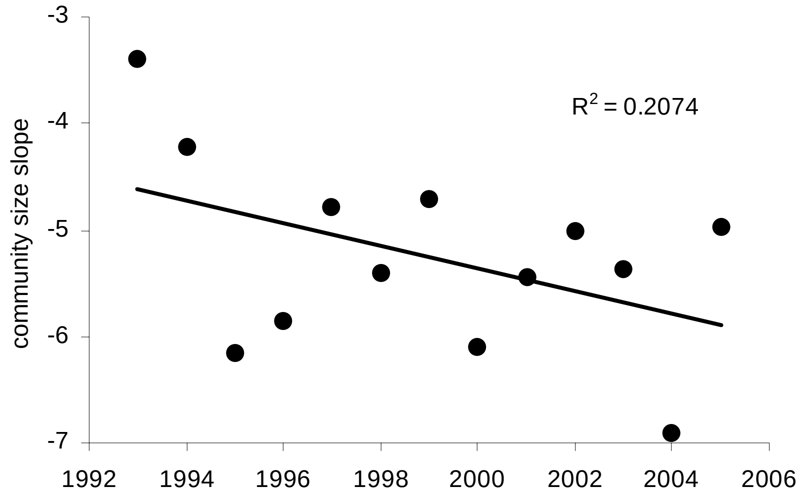


Figure 4.20. Size slopes of annual community size spectrum from West Estonian Archipelago 1993-2005 (fish lengths <15 cm excluded). Data from HELCOM COBRA database.

Baltic Sea - low fishing pressure

In contrast to the open Baltic Sea and coastal areas in eastern Baltic, coastal fish communities in western Baltic are less subjected to high fishing pressure. These communities, which generally are dominated by species with freshwater origin, show a different development. Using slope of size spectra as an indicator of community fish size, Figure 4.21 indicate an increase in size of coastal fish the last 20 years (r^2=0.21, p=0.05, linear regression).

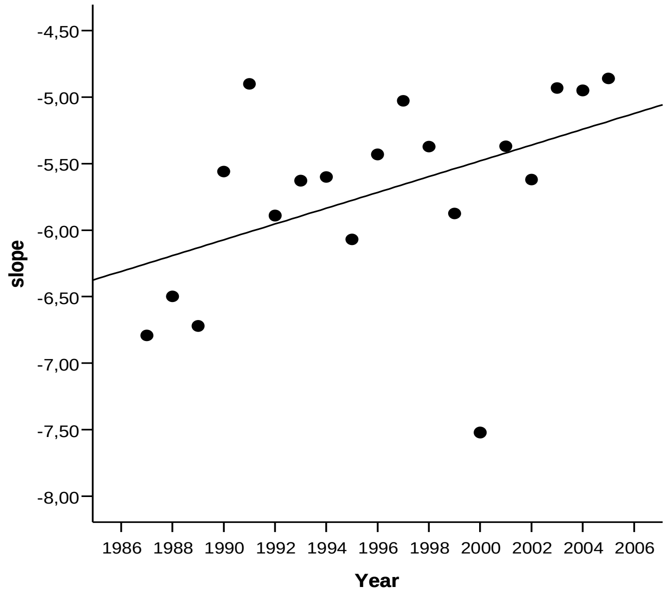


Figure 4.21. Size slopes of annual community size spectrum in a reference area in the archipelago in western Baltic Sea based on multi-mesh gillnet sampling. Data from Swedish Board Fisheries.

The observed increase in slope of the size spectra reflect increased abundance of two piscivorous fish species, zander and perch, and an increased abundance of the cyprinid species bream. These changes are also reflected as a significant increase in community mean trophic level (HELCOM 2006, Figure 4.22).

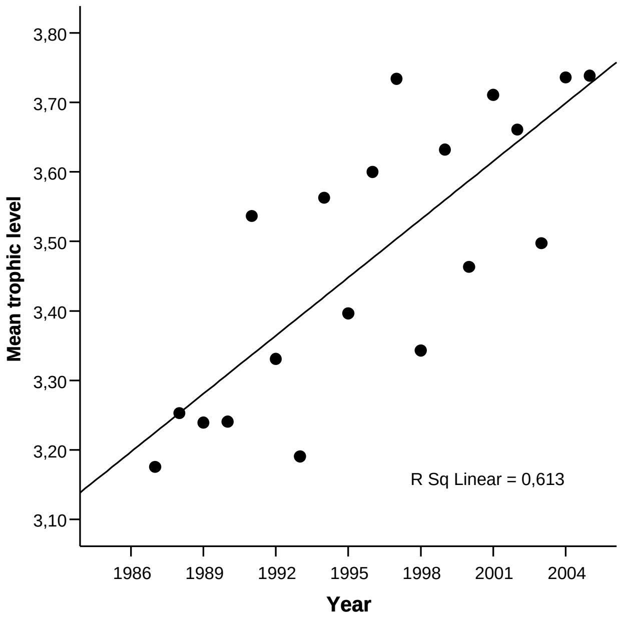


Figure 4.22. Changes in mean trophic level of the coastal fish community over time in a reference area in the archipelago of western Baltic Sea (HELCOM 2006).

Possibly environmental changes have had an impact on the development, as especially zander and bream have been gained by eutrophic conditions and high temperature during the1990s. This illustrate that other factors than fishing confounds the assumption on fishing being the driving force of population and community development in the Baltic Sea.

To be able to assess the outcome of suggested indicators we have to check the possible influence of other sources to the variation, especially the effects on environmental changes and perturbations. In many cases, as shown by the herring example, more complex interactions, such as indirect and cascading effects, have to be elucidated before selecting relevant indicators.

Ecosystem indicators at the community or population level holds promise as a tool for long-term surveillance and monitoring. However, in order to use them for ecosystem management reference points or reference directions need to be developed. Reference values can then be used to suggest management actions. Reference values can be derived from theoretical considerations (e.g. Jennings & Dulvy 2005). Studies indicate that such values can be used for the support of ecosystem management, but in general the power of the current research surveys to detect trends is poor. Therefore, specific targeted research is required to support monitoring and determine management options.

4.6.3 Benthos populations

There have been extensive critiques of proposals for the development of indicators for the benthic systems ((ICES, 2000, 2001b, 2002, 2003b). In answer to these critiques ACE established the Study Group on Ecological Quality Objectives for Sensitive and for Opportunistic Benthos Species (SGSOBS) in 2003.

SGSOBS used the following definitions:

- *Sensitive species* – 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.
- *Opportunistic species* 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 lifecycle (<1 year), small size, rapid growth, early sexual maturity, planktonic larvae through the year, and direct development.

These species proliferate after intense disturbance or pollution episodes. Surface or subsurface deposit-feeders dominate. In 2003 the ICES Working Group on Ecosystem effects of fishing (WGECO, ICES, 2003), based on the data for the North Sea soft sedimentary environments provided by the North Sea Benthos Survey database, recorded a total of 180 taxa as meeting the criteria for sensitive species, this includes biogenic structure-forming species as well as those with fragile morphological features, and 69 taxa as meeting the criteria for opportunists, this includes the opportunistic scavengers. WGECO considered this to be an initial and incomplete list. SGSOBS identified 242 sensitive species in genera beginning with the letter "A" alone and 54 taxa as 1st order opportunistic species and 119 as 2nd order opportunistic species (i.e., 173 opportunistic taxa). As previously stated by WGECO and SGSOBS, there remains a massive literature and incomplete knowledge of many species such that these estimates still remain conservative. However, they further serve to illustrate the problems of attempting to manage benthic systems using an indicator based on the density of individual sensitive and/or opportunistic taxa.

In the North Sea there are a few beam trawl surveys that sample part of the benthic invertebrate community (macro-epibenthos) on a regular basis. For one of these surveys, BTS conducted by RV Tridens and RV Isis, timeseries of a few selected species are shown in Figure 4.23.
Most of the species that are considered sensitive, however, are not regularly caught in these surveys thereby preventing the use of single-species indicators for “sensitive benthic species”.

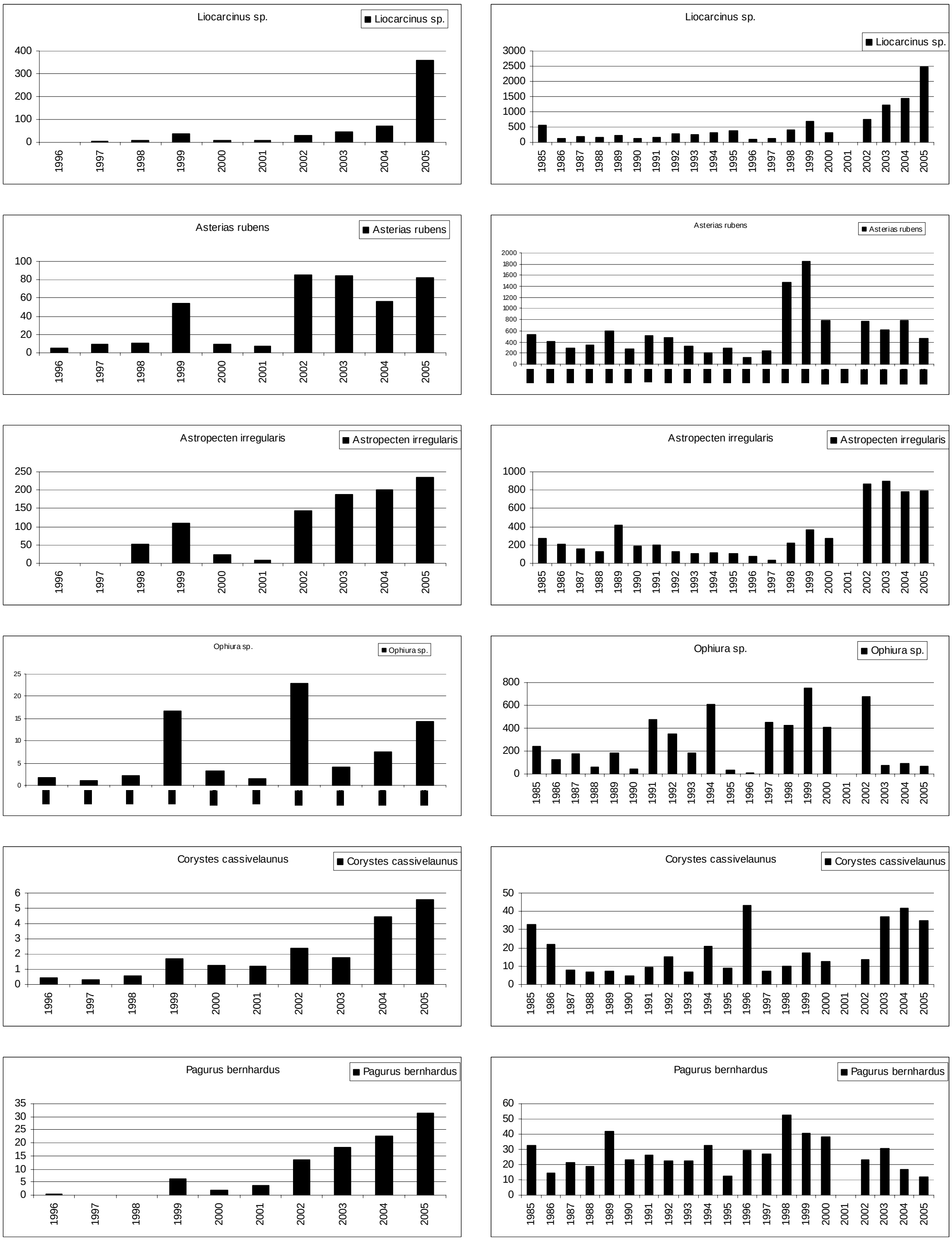


Figure 4.23. Timeseries of 6 epibenthic species based on beam trawl surveys conducted by RV Tridens (left) and RV Isis (right)

4.6.4 Benthos community

For indicators that describe benthic communities in the Bay of Biscay AMBI (Borja *et al.,* 2000) was chosen. This index has been developed as a human-induced indicator, and it has been tested with different impact sources. However, it has not been tested with fishing activities. In this particular case, probably AMBI does not respond in the right way, due to the physical disturbance produced by fishing, and it has been demonstrated that AMBI does not respond very accurately to this kind of impacts (Muxika *et al.,* 2005).

For the Bay of Biscay we do not have time series data of benthic communities that were impacted by fishing activities. In order to show the potential use of AMBI as indicator of pressures, here we studied two time-series obtained from two different locations of the Basque Country.

One of the stations (named Station E) is located at about 7 m water depth, in the inner part of the Nervión estuary (Bilbao, north of Spain), which has been monitored since 1989. The second sampling station (named Station M) is located at about 35 m water depth, in the coastal area near Deba (also, in northern Spain), and has been monitored since 1995. A study on the pressures over the entire Basque Country is available in Borja *et al.* (2006a), but none of the two sampling stations is affected by fishery pressures.

Station E was considered azoic until 1990 due to high levels of pollutants (heavy metals and organic matter) in sediments and low (even anoxic) dissolved oxygen concentration in bottom water layers (Borja *et al.,* 2006b). The estuary has been cleaned up in the past two decades, and an important recovery of the quality has been detected, especially in the inner part of the system (Borja *et al.,* 2006b).

Conversely, no important pressure is known in the surroundings of Station M and trends are not expected for the benthic communities.

For this exercise AMBI was selected as benthic community 'health' indicator (Borja *et al.,* 2000, 2003; Borja and Muxika, 2005; Muxika *et al.,* 2005). High values of AMBI (close to 7) indicate highly disturbed sediments; low values of AMBI (close to 0) correspond to undisturbed sediments. For power analysis, changes of 0.5 units•y^{-1} in AMBI were arbitrarily considered as ecologically meaningful.

Station E:

Selection of model:

Both the linear regression model (Figure 4.24) and the 2nd order polynomial regression model (Figure 2) between AMBI and the sampling year fitted quite the data quite well ($p<0.000$ for both of them, and $r^2= 0.768$ for the linear regression and $r^2= 0.80$ for the polynomial regression). However, a comparison of the likelihood between both models (χ^2) shows that the addition of one more parameters to the model does not improve it enough. Hence, the simplest one (linear regression) was chosen for this exercise.

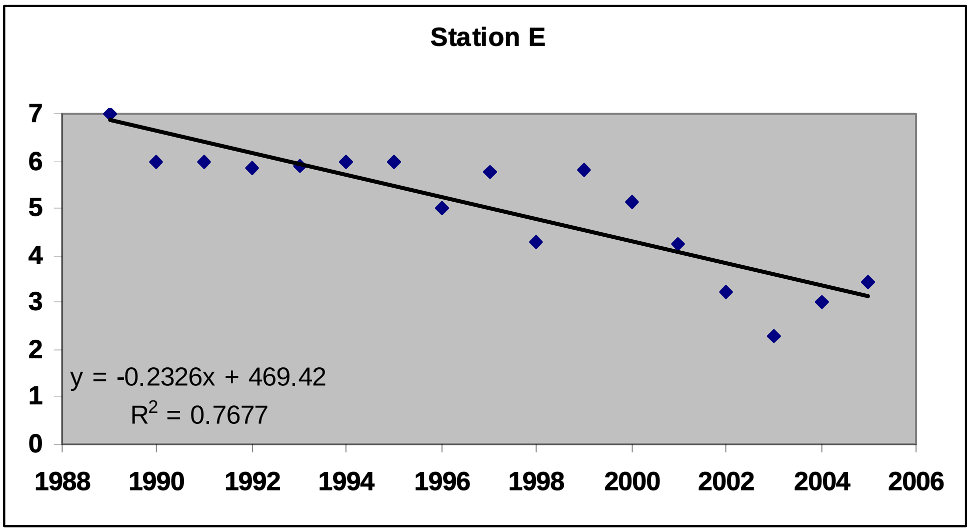


Figure 4.24: Linear regression between AMBI and sampling years for Station E.

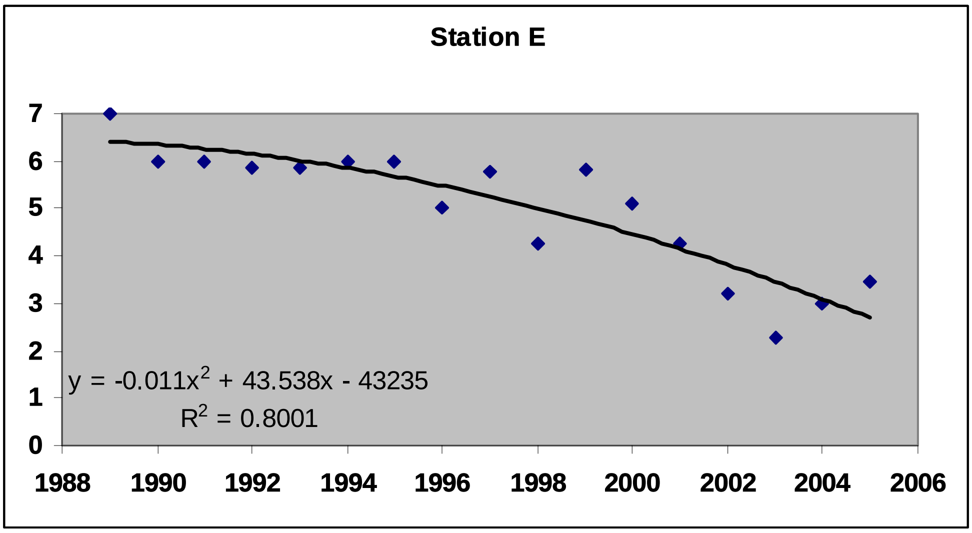


Figure 4.25: 2nd order polynomial regression between AMBI and sampling years for Station E.

At Station E, a recovery of the benthic community (negative trend of AMBI indicator) was detected due to the sewage plan, so a one tailed power analysis was carried out for the linear regression model. The improvement in 1990 (from azoic to extremely disturbed) (Figures 4.24 and 4.25) was due to the initial phase of physico-chemical clean-up (Borja *et al.*, 2006b). The slope of the curve is more pronounced after 1995, in which the most polluting company in the estuary closed (Borja *et al.*, 2006b). After 2000 there is an important improvement, after the commencement of the biological clean-up (Borja *et al.*, 2006b).

Power Analysis:

A power analysis was carried out for four different significance levels (α= 0.100, α= 0.050, α= 0.010 and α= 0.001) to study the effect of each significance level on power (Figure 4.26). In this way, it can be seen that in almost five years there is a power of 0.80 to detect a significant trend of 0.5 AMBI units for α= 0.100; almost six years are needed to detect the same trend for a significance level of α= 0.050; for α= 0.010, about eight years are necessary; and for α= 0.001, almost nine years.

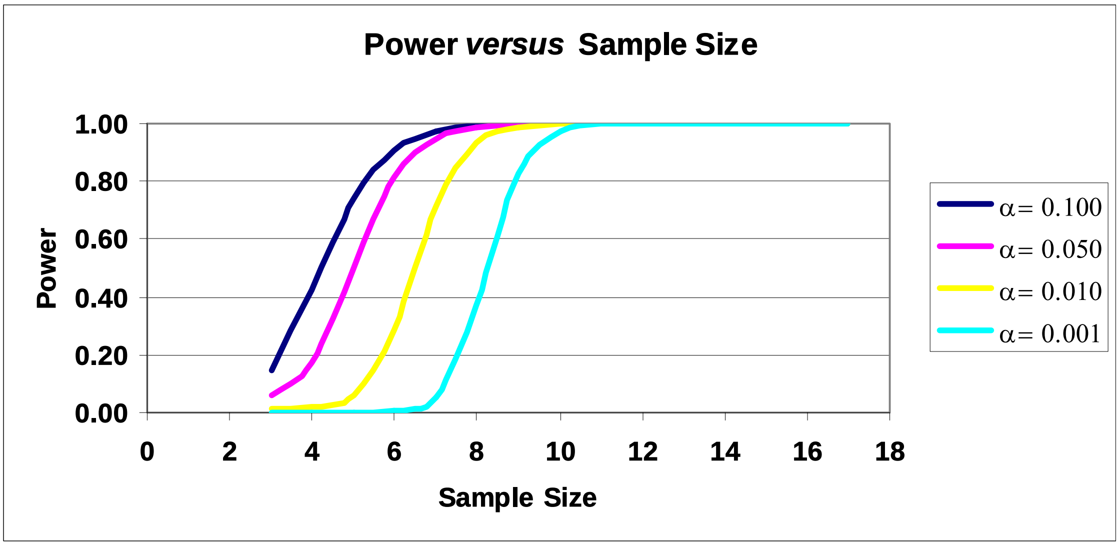


Figure 4.26: Power of the linear regression model for Station E versus sample size for different significance levels.

Station M:

Selection of model:

As expected from the absence of significant pressures in the area (Borja *et al.* (2006a), the regression between AMBI and sampling years was not significant for any of the regression models (p> 0.1). The linear regression is presented in Figure 4.27 as an example.

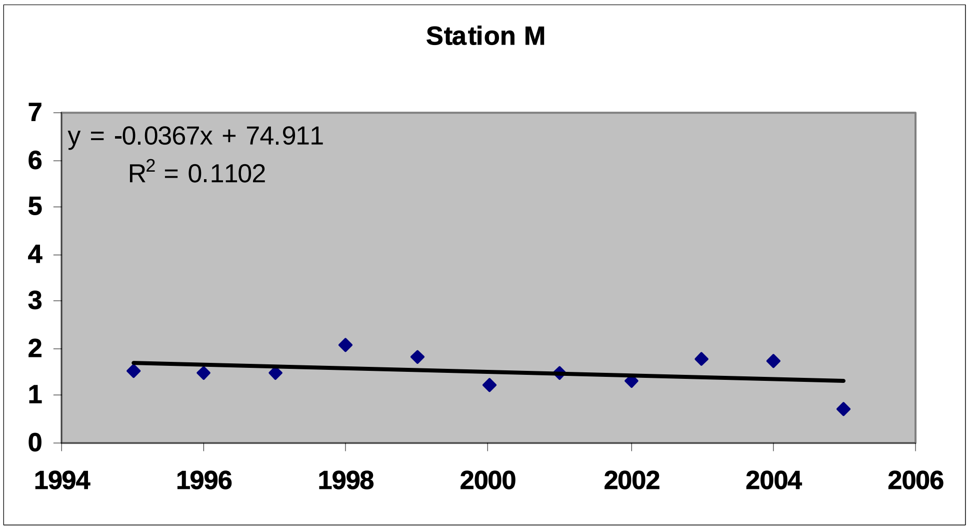


Figure 4.27: Linear regression between AMBI and sampling years for Station M.

Power Analysis:

At Station M, any trend in AMBI would be indicative of something: a positive trend would indicate an unknown pressure which has disappeared; conversely, a negative trend would indicate a new pressure on benthos.

So, it is interesting to analyze the power of the linear regression model with two tails. In Figure 4.28 the results for different significant levels are shown. For α= 0.100; a little more than four years would be enough to obtain a 0.80 power of detecting a trend of 0.5 AMBI units; almost five years are needed to detect the same trend for a significance level of α= 0.050; for α= 0.010, almost six years are needed; and for α= 0.001, about seven years.

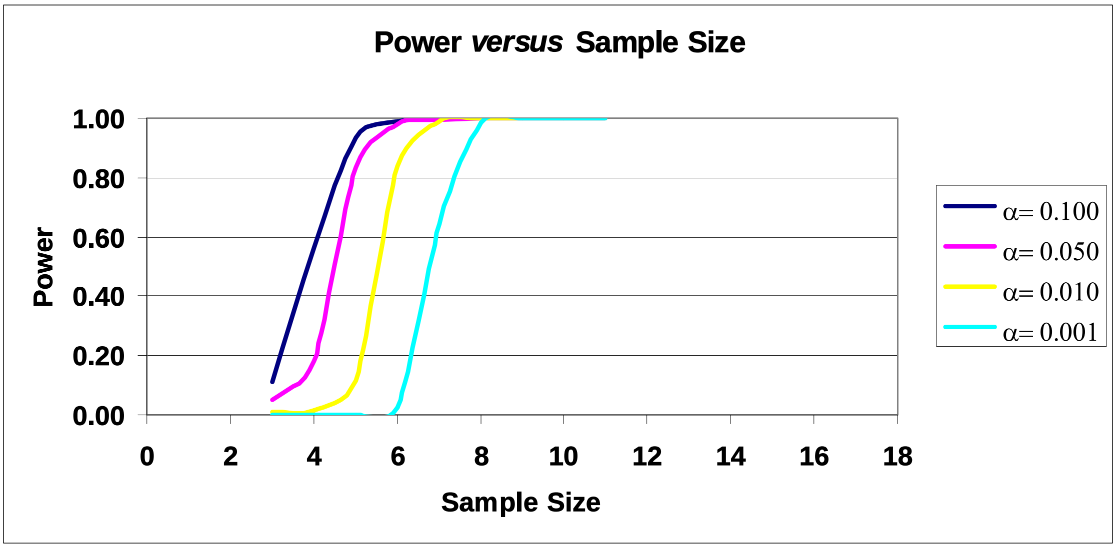


Figure 4.28: Power of the linear regression model for Station M versus sample size for different significance levels.

4.6.5 Seabirds

In various fora the suggestion is that there should be an indicator entitled "Abundance of vulnerable seabirds". This wording is consistent with a 'state' indicator for this faunal group. However, the subgroup on data collection (SGRN) of the Scientific, Technical and Economic Committee for Fisheries (STECF) considered that abundance was not a particularly suitable direct indicator of the effects of fishing on these species. Abundance may be affected by other causes of mortality, birth/recruitment rates and other factors that might cause changes in distribution. Abundance of these species is nevertheless a key factor in assessing the impact of a pressure (such as fishing) on a species – if this impact can be quantified (through e.g. by-catch monitoring schemes) then an assessment of effect on a population can be made.

A 'state' indicator of abundance (or relative abundance) could also be used to trigger further research to examine causes of change. On this basis, if an indicator showed a pre-agreed change, this could be used to trigger research to determine if the change was caused by fisheries or by another factor.

In contrast to indicators relating to fish, responsibilities for management, conservation status and health of mammals, birds and reptiles are shared between separate sections of the European Commission and member states. Consequently, some relevant information is gathered to meet other statutory or other requirements, not directly related to fisheries. The welfare and conservation status of these species is also of high public interest, so additional information needed for this indicator has been/is being already collected and collated by voluntary organisations and individuals.

As indicated above, the EU and its member states have, separately or collectively, already agreed to a number of commitments. One commitment relevant for seabirds is the Food and Agriculture Organisation (FAO) Code of Conduct for Responsible Fisheries. This Code, which was unanimously adopted on 31 October 1995 by the FAO Conference, provides a necessary framework for national and international efforts to ensure sustainable exploitation of aquatic living resources in harmony with the environment. Article 6 of the Code states that "*The right to fish carries with it the obligation to do so in a responsible manner so as to ensure effective conservation and management of the living aquatic resources*". It further states that fisheries management "*should not only ensure the conservation of target species but also of species belonging to the same ecosystem or associated with or dependent upon the target species*". Under the code, a number of voluntary International Plans of Action (IPOA) have been drawn up to make these Articles more specific. One of these IPOAs concerns by-catch of seabirds in longline fisheries. Under this IPOA, states with longline fisheries should conduct an assessment to determine if a problem exists with respect to incidental catch of seabirds. Although no figure is provided to define "problem", a technical note attached to the IPOA indicates that the statues of seabird populations, the total annual catch of seabirds and (the results of) monitoring of incidental catch of seabirds should be taken into account.

There are various sources of information for data on seabirds. Numbers of breeding seabirds are counted nationally in many, probably most, countries of Europe, but these are only rarely compiled internationally (see e.g. Tucker and Heath, 1994; ICES 2002). It would not be difficult to compile an update of these figures (and possibly identify gaps); resources would be needed for a compiler and negotiation of the submission of national datasets.

European seas, especially Atlantic seas, also support large numbers of migratory seabirds that breed elsewhere (e.g. the Arctic or southern hemisphere). Assessment of the abundance of these birds requires dedicated at-sea surveys. No comprehensive survey has been undertaken, but most existing data for the European Atlantic and North Sea has been compiled (voluntarily) into the European Seabirds at Sea (ESAS) database. Other data exist for the Baltic Sea but these are less accessible. Little or no at-sea data exists for the Mediterranean or Black Seas. Analyses can be undertaken of the ESAS database to indicate trends in relative abundance and geographic distribution.

There have been few studies of the scale of by-catch of seabirds in European waters, but northern fulmars appear to particularly susceptible to by-catch on longlines in northern European waters (Dunn and Steel 2001), auks, cormorants and seaducks in gill nets (e.g. in the Baltic, Kattegat and nearshore off Iberia) and Cory's shearwaters to longlines off southern Europe and in the Macronesian seas. In the Mediterranean, limited studies suggest that the Balearic shearwater (red-listed as Critically Endangered) is particularly susceptible to by-catch (Cooper *et al..* 2003).

For seabirds in the Mediterranean Sea, results from international surveys on the distribution on a basin scale, are available. The International Waterbird Census (IWC), provides an annual survey of midwinter numbers and distribution of waterbirds in the Western Palearctic and Southwest Asia since 1967, encompassing also the Mediterranean area. In Italy this activity began in 1993. Data have been collected as part of the Mediterranean Wetlands Initiative (MedWet), founded in 1991 to encourage international collaboration among Mediterranean countries, specialized wetland centers and international NGOs in protecting wetlands. Italy is not directly involved in this network. Finally, there are a lot of surveys carried out on local scale (a single wet area), or for a single species, such as the Audoni's gull, in the Ebro river Delta. This, however, is not part of any regular monitoring programmes and accessibility to these data is not always ensured.

Information on the bycatch of seabirds in the fisheries is not systematically collected.

4.6.6 Marine mammals

In various fora the suggestion is that there should be an indicator entitled “Abundance of vulnerable marine mammals”. This wording is consistent with a ‘state’ indicator for this faunal group. However, the subgroup on data collection (SGRN) of the Scientific, Technical and Economic Committee for Fisheries (STECF) considered that abundance was not a particularly suitable direct indicator of the effects of fishing on these species. Abundance may be affected by other causes of mortality, birth/recruitment rates and other factors that might cause changes in distribution. Abundance of these species is nevertheless a key factor in assessing the impact of a pressure (such as fishing) on a species – if this impact can be quantified (through e.g. by-catch monitoring schemes) then an assessment of effect on a population can be made.

A ‘state’ indicator of abundance (or relative abundance) could also be used to trigger further research to examine causes of change. On this basis, if an indicator showed a pre-agreed change, this could be used to trigger research to determine if the change was caused by fisheries or by another factor.

In contrast to indicators relating to fish, responsibilities for management, conservation status and health of mammals, birds and reptiles are shared between separate sections of the European Commission and member states. Consequently, some relevant information is gathered to meet other statutory or other requirements, not directly related to fisheries. The welfare and conservation status of these species is also of high public interest, so additional information needed for this indicator has been/is being already collected and collated by voluntary organisations and individuals.

As indicated above, the EU and its member states have, separately or collectively, already agreed to a number of commitments. Some of these commitments have been better implemented than others.

1. Habitats Directive (*Council Directive 92/43/EC on the Conservation of Natural Habitats and of Wild Fauna and Flora*)
Under Article 12(4) of the Habitats Directive, Member States must introduce a system to monitor the incidental capture and killing of all species listed on Annex IVa – this list includes all cetaceans and all turtles (that occur regularly in European waters). In light of the results of this monitoring, Member States are required to undertake further research or conservation measures to ensure that the incidental capture and killing “*does not have a significant negative impact on the species concerned*”. The deliberate capture, killing or disturbance of cetaceans is prohibited by Article 12(1). Member States have a duty under Article 2 to ensure that any measures taken under the Directive are designed to “*maintain or restore, at a favourable conservation status, natural habitats and species of wild fauna … of Community interest* (which includes all cetaceans and all turtles)*.”*

2. Council Regulation (EC) No 812/2004
This regulation came into force on the 1st July 2004. The regulation lays down measures aimed at mitigating incidental catches of cetaceans by fishing vessels operating in specific fisheries described in Annexes I and III. Under Annex I, Member States are required to assess the effects of acoustic deterrent devices over time, on vessels over 12m operating in the fisheries and areas concerned. Under Annex III, Member States are required to design and implement independent at-sea observer schemes to monitor cetacean by-catch on board vessels over 15m operating in the relevant fisheries. Additional monitoring is required on vessels less than 15m that operate in the same fisheries. Although this Regulation is very limited geographically and by fisheries, commitments exist on certain Member States.

3. Food and Agriculture Organisation (FAO) Code of Conduct for Responsible Fisheries.
This Code, which was unanimously adopted on 31 October 1995 by the FAO Conference, provides a necessary framework for national and international efforts to ensure sustainable exploitation of aquatic living resources in harmony with the environment. Article 6 of the Code states that "*The right to fish carries with it the obligation to do so in a responsible manner so as to ensure effective conservation and management of the living aquatic resources*". It further states that fisheries management "*should not only ensure the conservation of target species but also of species belonging to the same ecosystem or associated with or dependent upon the target species*".

Article 7 of the Code specifically deals with measures to reduce the by-catch of non-target species, which includes cetaceans. The Code says "*States should take appropriate measures to minimise …. catch of non-target species, both fish and non-fish species, and negative impacts on associated or dependent species, in particular endangered species*.

4. Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS).
ASCOBANS was set up under the auspices of the Convention on Migratory Species of Wild Animals (CMS) and came into force in March 1994. The Agreement was drawn up to co-ordinate and implement conservation measures for small cetaceans in the North and Baltic Seas. The Agreement has recently been extended to include Atlantic waters as far as 15°W and south to be contiguous with ACCOBAMS (see below) to the south of Portugal. Ten European countries are currently Parties to the Agreement, with a number of Range States considering whether to accede. The Agreement requires Member States to, amongst other commitments, to make efforts towards reducing by-catch in fishing nets. At the third Meeting of Parties to ASCOBANS a resolution was passed which called on competent fishery authorities to ensure that the total anthropogenic removal of marine mammals was reduced as soon as possible to below an unacceptable interaction. An unacceptable interaction was agreed as being above 1.7% of the best estimate of abundance. The resolution also underlined that the intermediate precautionary objective was to reduce by-catch to less than 1% of the best available population estimate. Note that this requirement means that a "best available population assessment" is available as well as a measure of by-catch levels.
OSPAR has adopted a very similar Ecological Quality Objective (EcoQO), with the implication that similar measurements are also required.

5. Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and contiguous Atlantic area (ACCOBAMS)
ACCOBAMS was concluded in Monaco in 1996 and entered into force in 2001. The Agreement presently has 18 Parties, with 7 of these being EU Members (and two further to join shortly). Under it, Parties have agreed, both through its Action Plan and through subsequent resolutions (e.g. Resolution 2.21 of the second Meeting of Parties) to "assess …impacts of interactions between cetaceans and fishing activities in the ACCOBAMS area."

There are various sources of information for data on marine mammals. The abundance of cetaceans has been assessed comprehensively in two surveys (both conducted under joint European and Member State funding) for waters off northern and western Europe. In 1994, this survey (SCANS I) covered the North Sea, the southern Baltic Sea and the Celtic shelf. In 2005 (SCANS II), all shelf waters from 62°N to southwest Portugal and the south-western Baltic were surveyed. It is worth noting that these surveys each took one month of one year – variations in abundance between years or within years (seasonal) have not been described. Other surveys have covered smaller parts of this wide area, other times of year and some parts of waters further west (summarised by ICES 2005). There has been no comprehensive abundance survey of cetaceans in the Mediterranean or Black Seas, but some smaller areas, especially the Ligurian Sea and the waters around the Balearic islands have been surveyed. Plans exist or are in preparation for full abundance surveys of deeper waters off western

Europe (CODA – project turned down for Life funding in 2006) and in the Mediterranean. Surveillance of cetaceans is required under the Habitats Directive, but there is at present no obvious European funding for this obviously multinational requirement, in contrast to funding that is provided from Europe to national programmes of protected areas.
In the Mediterranean Sea, 19 species of cetaceans can be encountered; 8 of them are considered common (Fin whale *Balaenoptera physalus*, Sperm whale *Physeter macrocephalus,* Striped dolphin *Stenella coeruleoalba*, Risso's dolphin *Grampus griseus*, long finned Pilot whale *Globicephala melas*, Bottlenose dolphin *Tursiops truncatus*, Common dolphin *Delphinus delphis,* Cuvier's beaked whale *Ziphius cavirostris*), while 4 are occasional (Minke whale *Balaenoptera acutorostrata,* Killer whale *Orcinus orca*, False killer whale *Pseudorca crassidens,* Rough toothed dolphin *Steno bredanesis*), and 6 accidental, alien to the Mediterranean, but occasionally sighted in the last 120 years (among them the Humpback whale *Megaptera novaeangliae*); moreover, we have to consider the presence of a small population of Harbour porpoise *Phocoena phocoena* in the Black Sea.
The Mediterranean Monk Seal (*Monachus monachus*) is the only pinniped to be found within the Mediterranean Sea. It is now very rare and listed as an endangered species. The only known colonies are in the Alboran Basin and in the Aegean Sea. It is very unlikely that any animals will be encountered around Sicily or Malta.

The numbers of seals in the Baltic are surveyed frequently and summarised by ICES for HELCOM on a bi-annual basis (see e.g. ICES 2005). Seal numbers in the North Sea (including the English Channel) have been summarised in OSPAR (2005). Seals in western Britain are counted annually or every five years (depending on species). Numbers around Ireland are not counted regularly. In the Mediterranean, numbers of the endangered monk seal are assessed regularly.

Assessment of cetacean by-catch assessment has been patchy in all European waters despite statutory requirements for such assessment to be undertaken. CEC (2002a, b) gathered existing information together and identified major gaps, but did not assess the effects of the sum of all fisheries by-catches on any one species. This would still be a difficult assessment to make. It may be possible to approximate this figure in some of the better studied areas – for example the by-catch of harbour porpoises in the North Sea.

One specific example is that of Grey seal in the Baltic Sea. Grey seal (*Halichoerus grypus*) is the dominant marine mammal in the Baltic Sea, the main distribution area covering the entire northern Baltic Sea (i.e. the areas of Sweden, Finland, Estonia and Russia). The present population size of grey seal is over 18 000 individuals. The other seals in the Baltic, ringed seal (*Phoca hispida botnica*) and harbour seal (*Phoca vitulina*) are less common, the present population sizes being 5 500 - 6000 individuals for the ringed seal and around 750 for harbour seal. Harbour porpoise (*Phocoena phocoena*) is distributed only in the southern parts of the Baltic Sea and most of the individuals are found in the Danish straits.
The population size estimates of grey seals are based on hunting statistics, and since the 1980s, on aerial and boating surveys. The surveys have been aimed to count all seals visible during the moulting season. The estimated number of grey seals in the Baltic Sea was almost 100 000 individuals in the beginning of the 1900s, and a sharp decline started around 1910-1920 (Figure 4.29).

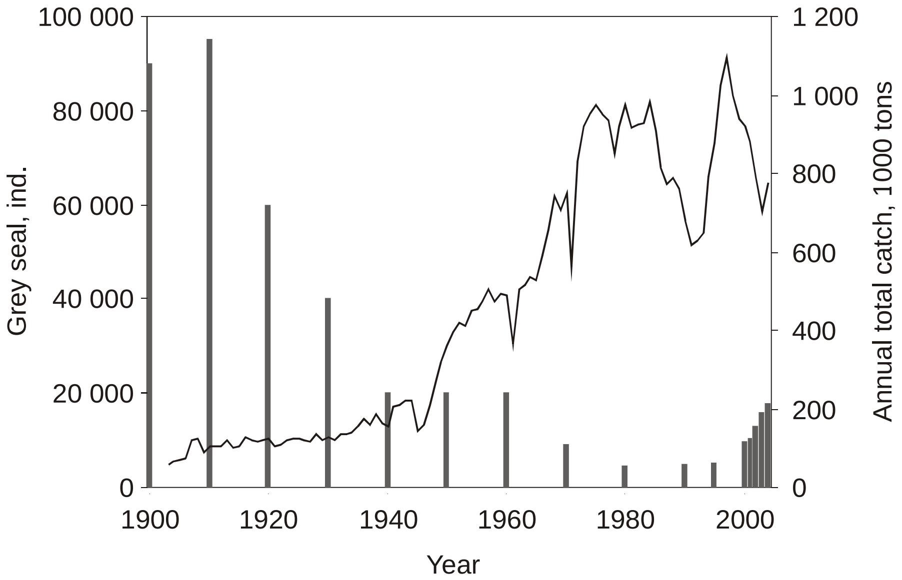


Figure 4.29. Estimated population size of grey seal in the Baltic Sea during 1900-2004 (bars, Data from Harding and Härkönen (1999) and FGFRI seal statistics) and annual total catch of Baltic Sea fisheries (curve, data 1903-1972 redrawn from Thurow (1997), later data from ICES).

Fishery may affect marine mammal population at least in two ways:

- Decreasing the amount of suitable food (fish) available for marine mammals. The time series of total catch in the Baltic Sea could be a potential pressure indicator for the first one. The total catch of The Baltic Sea fishery remained at relatively low level until the latter half of the 1940s. After that, the total fish catch increased tenfold to the 1980s (Figure 4.29), the bulk of the catch consisting of herring (*Clupea harengus*), sprat (*Sprattus sprattus*) and cod (*Gadus morhua*). The timing of the decline of grey sea and the increase in Baltic Sea fisheries do not coincide, suggesting that fishery is not the main cause for decline in grey seal population. It is, however, possible that decreased predation by seals has contributed to the increase of fishery catches. In the beginning of the 1900s, seals were abundant and the annual fish consumption of all seal species has been estimated at 300 000 tons, i.e. 30-40% of today’s fishery catches (Hansson and Rudstam 1990).
- Increasing direct mortality of marine mammals (as by-catch). Unfortunately, there have been only occasional surveys to estimate grey seal by-catch of fisheries in the Baltic Sea region. Rough estimates for average annual numbers of grey seals (killed) among by-catch in Finnish fishery during the latter half of the 1900s (table 4.4) demonstrates that the annual numbers of grey seal in by-catch have been relatively low (0.5 – 2%) of the entire standing population. It can be assumed that this proportion has been much lower during the first half of the 1900s, when the number of seals was high, fishing effort was lower that nowadays and the modern hard-wearing nylon was not available for gill nets and traps.

Table 4.4. Estimates of annual grey seal by-catch in Finnish commercial fishery in five time periods. The figures of the first three periods are from Helle (1979). Later estimates are unpublished estimates of FGFRI.

Period	Annual number of grey seals as by-catch
1956-1960	119
1961-1965	123
1971-1975	34
1986-1990	110
1997-1999	200

Grey seals, both adults and pups, are most commonly killed in trap net fishery for salmon and herring, and in drifting net fishery for salmon (*Salmo salar*) (source: Finnish seal management plan). Thus, the time series of effort of these fisheries could be a potential pressure indicator. Finland has a relatively long tradition to produce fishery statistics for the small-scale fisheries, too, and time series exist starting from 1980 onwards. The efforts of these fisheries have remained stable or may have decreased slightly, but no correlation between the efforts and the size of grey seal population in Finnish coastal areas can be visually detected (Figure 4.30).

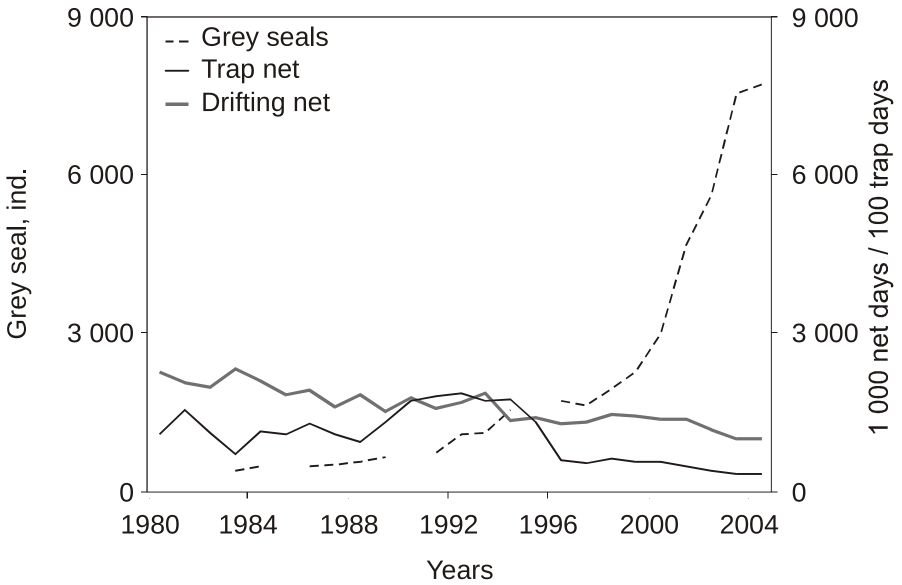


Figure 4.30. Grey seal population in Finnish territorial waters in 1983-2004 and simultaneous fishing efforts by trap net and drifting net of Finnish commercial fishing fleet (data from FGFRI).

Another important pressure other than fishing is hunting. The annual numbers of hunted grey seals in Finland have been relatively high in the first half of the 1900s (Figure 4.31). In 1935, for instance, the number of hunted seal in Finland was approximately 10% of the estimated total number of grey seals in the entire Baltic Sea. In addition to Finland, grey seal hunting was common also in Sweden and former Soviet Union, although it was not as extensive there as in Finland. People living in the archipelago were encouraged to hunt grey seals also by bounties, which were paid in Finland during 1900-1975. Extensive hunting has been the primary factor for the sharp decline in the Baltic grey seal population during the first half of the 1900s.

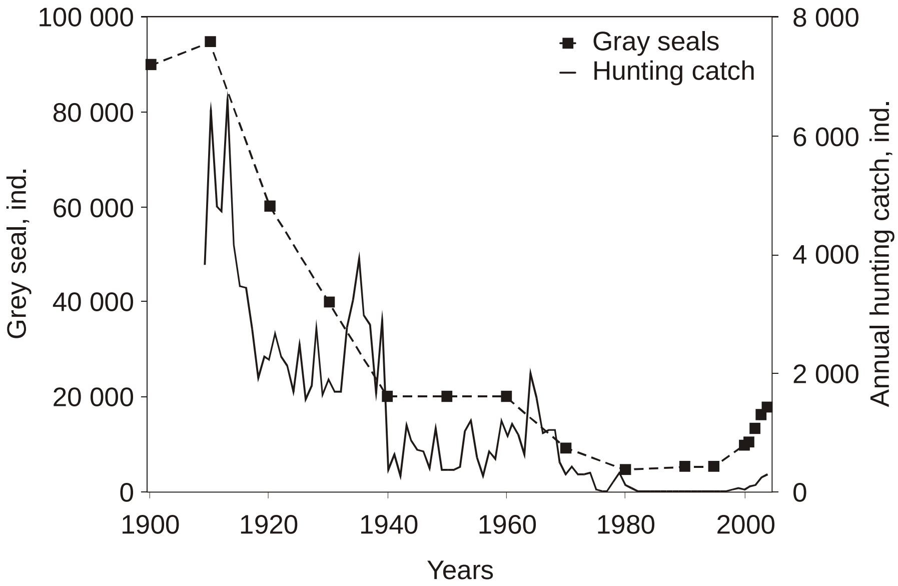


Figure 4.31. Estimated population size of grey seal in the Baltic Sea during 1900-2004 (data from Harding and Härkönen (1999) and FGFRI) and the Finnish seal hunting catch (data from FGFRI).

During the latter half of the 1900s, environmental toxins were another important factor contributing the decline of the grey seal populations. A considerable proportion of grey seal females were not able to reproduce during the 1970s and 1980s (Bäcklin et al. 2003). Unfortunately, good time series do not exist. According to Bergman (1999) the pregnancy rate of mature females was 9% in samples from 1977-86, and 60% in 1987-96. The sample sizes were, however, low. During the 2000s, the pregnancy rate has been around 80%, which is a 'normal' level. It is obvious that the lowered reproduction rate had some connection to PBC and DDT concentrations in the biota of the Baltic Sea. There are some time-series available of concentrations of these compounds in fish and seabirds, and these could possibly be used to explain (part of) the trends in grey seal reproduction problems.

Summarising, the Baltic Sea grey seal population has collapsed during the 1900s, and since the 1990s, the population has started to increase. However, any clear link between fishery and the historical state of grey seal populations in the Baltic Sea could not be detected. Thus, the state of grey seal population is not a suitable indicator of environmental performance of the common fishery policy in Baltic Sea.
Heavy hunting pressure, especially in the early 1900s and later also environmental toxins are supposed to be the main causes for the collapse of Baltic seals. The recent increase in grey seal population has caused troubles to coastal gill net and trap fishery in Finland and Sweden, and as a consequence of the strong claims of local fisherman organizations, a restricted hunting is again allowed in Finland and Sweden. The discussion of the appropriate grey seal population level has already started. It is evident that argumentation and conflicts between the fishery sector and conservationists will strengthen in the near future. HELCOM has requested ICES to give advice for grey seal conservation e.g. regarding possible target and reference points for grey seal population.

In contrast to the grey seal, the number of harbour porpoise (population size) in the Baltic Sea could be an interesting population level indicator for the environmental performance of the CFP, especially as the conservation of this species has been the main incentive for the total ban for drifting net fishery in the Baltic Sea (beginning in 2008). Here again, the semi enclosed and heavily polluted Baltic Sea has been a special case. Hard winters during the 1940s (entire sea had ice-cover) have been a major cause to the crash of the Baltic harbour porpoise population then. It was also suggested that environmental toxicants (e.g. PCBs and

DDT) have been another factor affecting harbour porpoise population in the Baltic Sea (Koschinski 2001), although the concentration of these compound have decreased during the latest decades. This means that the historical data is likely not very valuable for indicator purposes, but the future data collected by EU whale-observation system will probably offer good data for a population level state indicator.

4.6.7 Marine reptiles

For marine reptiles the same considerations and commitments apply as for the marine mammals (see 4.6.6).

Very little information exists to quantify marine turtle populations in Europe. In some places, an indication of numbers hauling out onto breeding beaches is available, with some attempts to assemble this information (e.g. Groombridge 1990). There appears to be no knowledge of at-sea distribution and abundance.

There have been studies of by-catch of turtles in European waters (e.g. Aguilar *et al..* 1992; Panou *et al..* 1992; Camiñas1997; Ferreira *et al..* 2001; Pierpoint 2000, CEC 2005), and in some countries this has been comprehensive, but there has been no overall assessment of total by-catch or of population effects.

4.6.8 Habitat

Potential indicators for habitat should address either the state of the habitat (e.g. area occupied) or the pressure of the fishery on this habitat (e.g. area of the habitat impacted). Often the indicator is intended specifically for "sensitive habitats".

A study report was finalised in August 2003 to review work on environmental indicators for fisheries in different scientific fora in order to build on existing knowledge and a proposal for a preliminary set of indicators (Jaako Pöyry Infra, 2003). This report was reviewed by the STECF (EC 2004) and based on this the Commission selected a reduced set of indicators among which two indicators for habitat:

1. Area coverage of highly sensitive habitats.
2. Mapping of effort distribution over the sensitive areas

The second indicator is effectively a combination the information necessary for the first indicator and information that is presented in section 4.6.12, where information on the spatial and temporal distribution of fishing effort is overlaid on maps of marine habitat extent. As this is only relevant when an impact of the fishing practices on the habitat is expected we used a matrix (table 4.5) to cross-reference them.

An example of the mapping of effort distribution over marine landscapes (not necessarily sensitive areas) is shown for the Irish Sea.

Table 4.5. Matrix of fishing gear/habitat type and fishing activity, X indicates a potential impact (after ICES, 2000; Gubbay, 2001).

	Sensitive Habitat Type (from Gubbay, 2001)						
Fishing Activity	Deep-water biogenic habitats[1]	Structural benthic epifauna[2]	Benthic infauna[3]	Mollusc beds[4]	Nearshore communities[5]	Intertidal mudflats	Maerl beds
Otter trawling	X	X	X	X	X		X
Beam trawling		X	X	X	X	X	X
Pelagic trawling							
Drift/gill netting	X						
Bottom long-lining	X?	X				X	
Pelagic long-lining							
Tangle netting	X?	X?			X	X	
Pot fisheries		X			X		
Dredging (Epibenthic)		X	X	X	X	X	X
Dredging (Hydraulic)		X	X	X	X	X	

Key to sensitive habitat types:
1 Deep-water biogenic habitats: Lophelia pertusa reefs, carbonate mounds, oceanic ridges with hydrothermal vents, seamounts and deep-water sponge communities.
2 Structural benthic epifauna: Sabellaria spinulosa reefs.
3 Benthic infauna: Seapens and burrowing megafauna communities.
4 Shellfish beds: Ampharete falcata sublittoral community, Ostrea edulis beds, Modiolus modiolus beds and intertidal mussel beds.
5 Nearshore communities: Zostera beds and littoral chalk communities.

At present there are insufficient biological data to define habitats. The Irish Sea Pilot project has classified 18 distinct marine landscapes based on geophysical and hydrographical information (Roff and Taylor 2000; Vincent et al. 2004). The main types of data used for landscape classification were depth, substratum type, current strength (sea bed stress), and topography (slope). The sensitivity of each landscape to trawling impact was assessed on the basis of whether the biotope complexes characteristic of the marine landscape would survive a one-off impact (Golding et al. 2004).

VMS fishing effort data were spatialised within the GIS package ESRI ArcGIS version 8 (Mills et al. 2004). The UK VMS database logs the geographical position, speed, bearing and identification number of European vessels over 24m in length within UK waters. In this study, VMS data were used from 1st January 2000 to 31st December 2004 for all UK and non-UK beam, dredge and otter trawling vessels over 24 m in the Irish Sea (ICES Division 7a). Approximately 98% of the vessels operating in the Irish Sea were British, French, Belgian and Irish, and these were used in all subsequent analyses. A small proportion of the gear type descriptors were missing from UK vessels, and on those occasions other national fisheries databases containing vessel registration and type were used to fill the gaps. A larger proportion (75%) of the non-UK vessel VMS returns were not attributed with gear type information. As detailed gear type databases for these international fleets were unavailable, a different approach was adopted. The gear types used by French, Belgian and Irish vessels were inferred by examining the location of the fleets by nation of origin, and applying expert knowledge of the local patterns of fisheries exploitation from throughout the Irish Sea. Using this method it was concluded that the Belgian fleet in the Irish Sea consisted almost entirely of beam trawlers while the French vessels were mainly otter trawlers. The Irish fleet, however, consisted of both beam trawlers and otter trawlers. Due to the known spatial distribution of the Irish fleets, which broadly separated the trawlers in the eastern Irish Sea from those exploiting the Nephrops fishery by otter trawlers in deeper muddy grounds, it was possible to distinguish the few remaining unclassified Irish vessels. Transmitted vessel speeds of UK vessels were used to differentiate among possible vessel behaviours: stationery, fishing, or steaming. The distribution of vessel speeds associated with active fishing was derived from UK vessels known to have been fishing based on logbook records. The frequency distribution of vessel speeds was then used to determine thresholds for fishing speeds of each of the three gear categories. The frequency distribution for beam trawlers suggested that fishing was likely to occur took place at speeds of between 2-8 knots inclusive. Data for dredgers and otter trawlers suggested that fishing occurred at speeds between 1-4 knots inclusive. We summarised fishing effort by calculating average daily counts of VMS fixes over the entire region and individual marine landscapes in each year.

In order to calculate the spatial distribution of fishing activity by each of the gear types, the VMS database was incorporated into a GIS. To estimate vessel density, we used a kernel density estimation (KDE) technique within ArcView Spatial Analyst. A 1 km grid was created covering the entire Irish Sea, and vessel positions were allocated to a search radius surrounding each cell. We used a radius of 14 km based on the least squares cross-validation technique that produced a surface that described local variation in fishing intensity with minimal over-smoothing (Silverman 1986).

The vessel density per 1 km^2 cell was calculated from the total number of records and the area of the search, with each vessel position weighted using an adapted Gaussian distribution so that recorded vessel positions close to the cells had a greater weighting. The distribution of VMS fishing effort over each marine landscapes was summarised by comparing the fishing effort density surface outputs with the marine landscape map beyond 6 nm from shore (Mills et al. 2004). The total number of VMS location records for active fishing vessels in each year was calculated for each marine landscape.

The most frequent fishing activities were otter trawling and scallop dredging with 20.2 ± 2.3 and 18.3 ± 3.6 VMS registrations day^{-1} $year^{-1}$ respectively, compared to beam trawling (8.1 ± 2.7 VMS registrations day^{-1} $year^{-1}$).

Otter trawling activity has been consistent over the past four years, with a slight reduction in 2001 (Figure 4.32a). Scallop dredging activity increased up to a peak in 2002 and remained stable for the next two years (Figure 4.32b). Beam trawl effort has remained consistently low until effort doubled in the last two months of 2004 (Figure 4.32c).

Otter, dredge and beam trawling occurs in just 12 of the 18 marine landscapes, and most fishing activity (98%) occurred in just six landscapes (Table 4.6). Most fishing activity occurred over low bed stress coarse sediment plains (49.7%), followed by high bed stress coarse sediment plains (13.9%), and fine sediment plains (13.8%), mud basins (deep = 8.6%; shallow = 5.7%) and sediment wave megaripple fields (6.4%).

There have been increases in fishing activity in three sensitive landscapes. Dredging activity has increased three-fold in low bed stress coastal sediment plains (Figure 4.33a). Otter trawling has almost doubled in shallow water mud basins (Figure 4.33b). Beam trawling has increased 7-fold in fine sediment plains (Figure 4.33c). Beam trawling has also increased in sediment wave megaripple fields (Figure 4.33e), which have relatively low sensitivity to the impacts of trawl gears.

This analysis demonstrates that historic data are available for 2000-2004 for otter, dredge and beam trawlers >24m in the Irish Sea. There is good scope to extend the fleet and habitat coverage of this indicator. All vessels >15m have been included in the VMS since 2004. The marine landscapes habitat classification is currently being extended, which will allow this indicator to be calculated for all UK waters to the median line.

This analysis shows that it is possible to generate an indicator to show how fishing activity varies over sensitive marine landscapes over time. Ideally the indicator should represent the impact on biological habitats and as currently formulated there are two problems with this index. First, it is difficult to relate this measure of fishing activity (VMS vessel activity 'pings' day^{-1} $year^{-1}$) to the mortality of benthic organisms or change in benthic secondary production, or some other direct measure of change in biodiversity or ecosystem function. It may be fruitful to develop further methods of swept area estimation from such low resolution VMS data (Eastwood et al. submitted; Mills et al. submitted). Second, the marine landscapes used are derived from physical features and thus are not a direct representation of biological value or sensitivity.

The continued development of this indicator is contingent upon the development of a Pan-European VMS database.

Table 4.6. Irish Sea marine landscapes subject to fishing activity. The size of each landscape is presented in areal extent (km^2) and expressed as a proportion of Irish Sea Pilot project study area. Sensitivity to trawling is scored categorically from 3 = highest to 1 = lowest sensitivity, NA indicates insufficient information on seabed habitats to assess sensitivity. Fishing activity is expressed as VMS registrations expressed as a percent of total.

Marine landscape	Landscape area (km^2)	Landscape area (%)	Sensitivity to trawling	Otter trawl activity	Dredge trawl activity	Beam trawl activity	Average trawl activity
Low bed-stress coarse sediment plain	15186	25.1	3	27.6	82.9	40.1	49.4
High bed-stress coarse sediment plain	11760	19.4	3	21.4	4.0	12.9	13.9
Fine sediment plain	13218	21.9	3	13.1	11.3	22.8	13.8
Deep-water mud basin	5024	8.3	2	16.8	0.2	0.9	8.6
Sediment wave/megaripple-field	6630	11	1	5.7	0.8	23.1	6.4
Shallow-water mud basin	980	1.6	2	11.1	0.4	<0.1	5.7
Deep-water channel	234	0.4	NA	3.4	0.0	0	1.7
Sand/ gravel-banks	540	0.9	3	<0.1	0.2	<0.1	0.1
(Irish) Sea Mounds	74	0.1	3	0.1	0.0	0	<0.1
Photic reef	278	0.5	3	0.1	0	<0.1	<0.1
Coastal sediment	3606	6	3	<0.1	<0.1	<0.1	<0.1

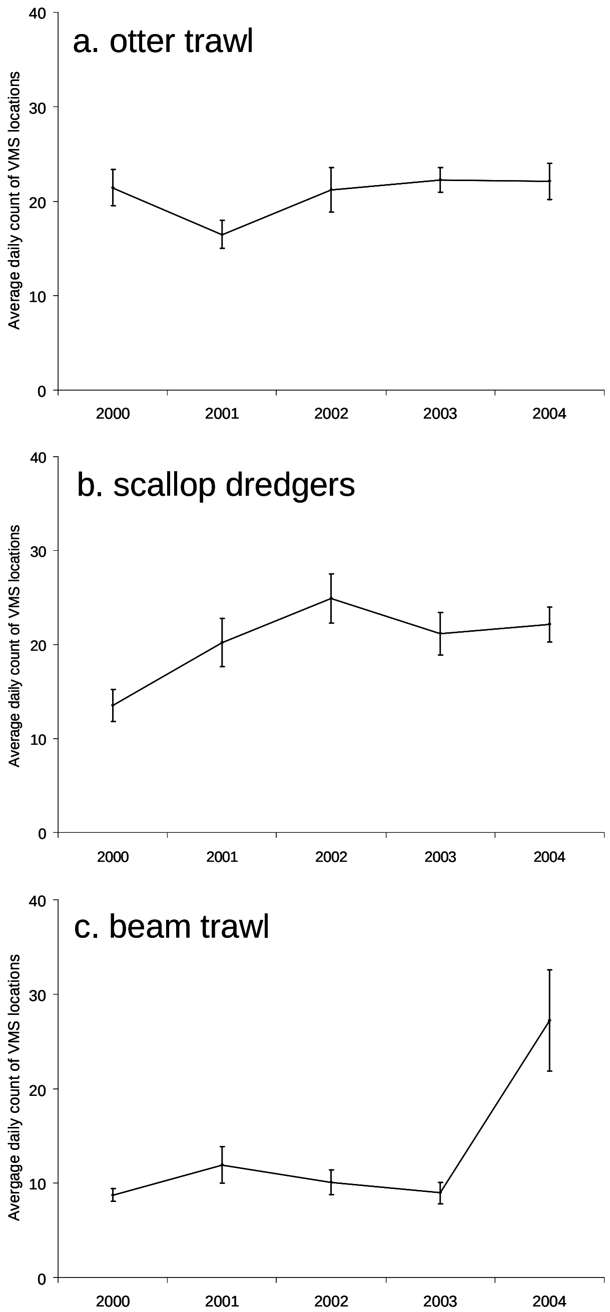


Figure 4.32. Annual variation in activity of (a) otter trawlers, (b) scallop dredgers, (c) beam trawlers and in the Irish Sea based on VMS locations. Error bars represent

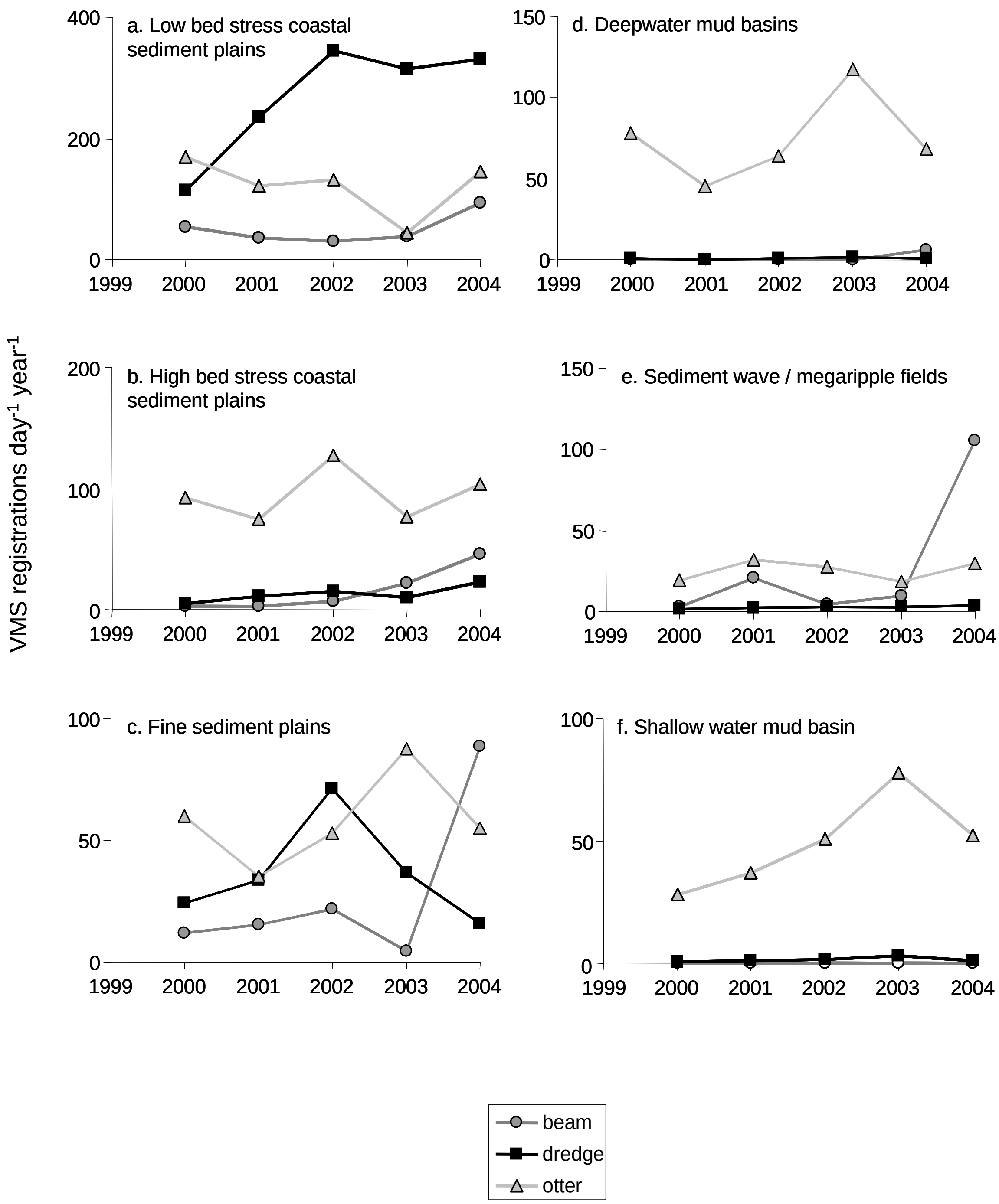


Figure 4.33. Annual variation in fishing activity of over six marine landscapes in the Irish Sea based on VMS locations.

The possibility to quantify this indicator suffers from the same problems as were already identified in sections 2.8 en 2.13, i.e. the lack of high-resolution data on the spatial distribution of sensitive habitats and the fishing activities that impact them. It is only these data that will allow us to advice on the spatial extent of fishing activity, and the potential impacts of these gears on demersal habitats.

4.6.9 Physical

Water temperature is probably the best studied physical characteristic of most of the European waters and extensive datasets exist. One example is shown here, the bottom water temperature at Viking bank, based on IBTS. It shows that in the North Sea bottom water temperature has been increasing by about 0.3°C and 0.6°C per decade since a cool period in the late 1970s.

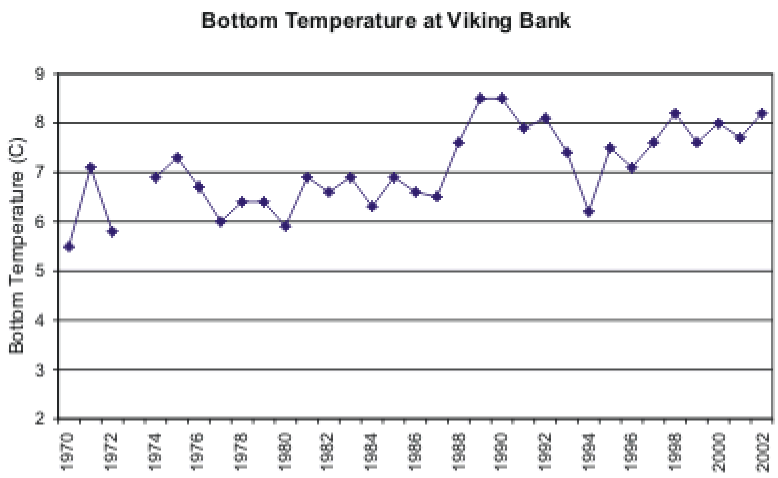


Figure 4.34. Bottom Temperature time series in the North Sea, 1970-2002 (Courtesy of the ICES Oceanographic Data Centre)

4.6.10 Chemical

Common chemical indicators are for those chemicals that are responsible for eutrophication: Nitrate and Dissolved Inorganic Phosphorous (DIP). A time-series for the Baltic is shown in Figure 4.35.

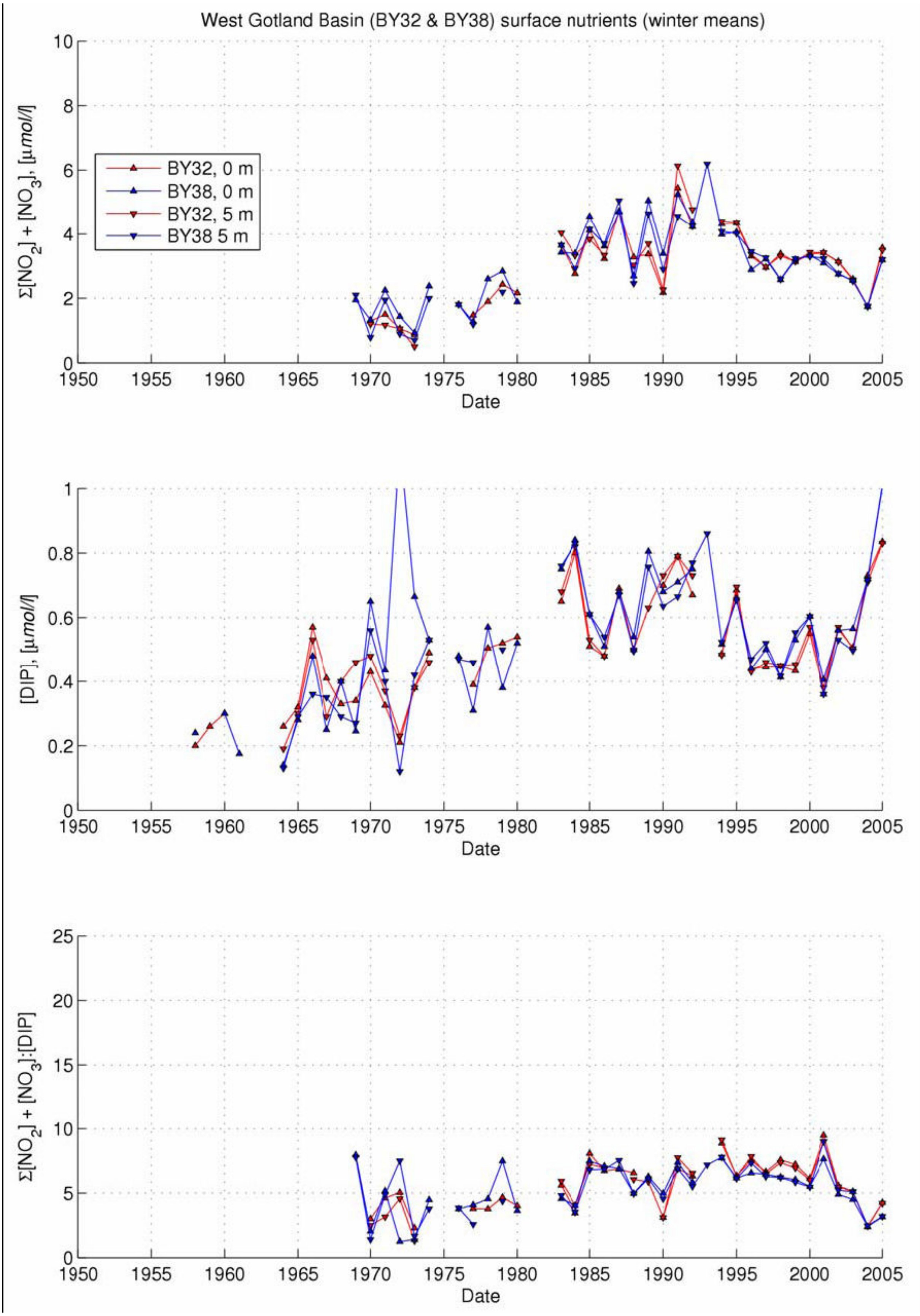


Figure 4.35. Time series of winter nutrients (Nitrate and Dissolved Inorganic Phosphorous, DIP) and their ratio in the western Gotland basin from surface and 5 meters depth.

4.6.11 Ecosystem functioning

Ecosystem indicators try to give an answer to the main question on how changes in the population/community structure can affect processes at the highest level of hierarchical complexity: that of ecosystem functioning. At present, when faced with indicators at an ecosystem level, the main challenges are data availability and the lack of reference points or reference trends. Moreover, many candidate indicators can be calculated only by using a model approach and hence require a model describing the ecosystem of interest.

In this context, a comparative analysis has been carried out, in order to assess the performance of indicators estimated from different data sources (model and field data), belonging, if possible, to the same geographical area (Mediterranean Sea). The different kinds of data used in relation to different geographic areas:

landings data	North Adriatic Sea (time series from 1946 to 2003)
survey data	Sicily Channel, MEDITS (1994-2004)
model outputs	North Sea (1974-1994)

Results for mean Trophic Level (mTL), Primary Production required (PPR), Fishing in Balance index (FiB), and Finn Cycling Index (FCI) are reported.

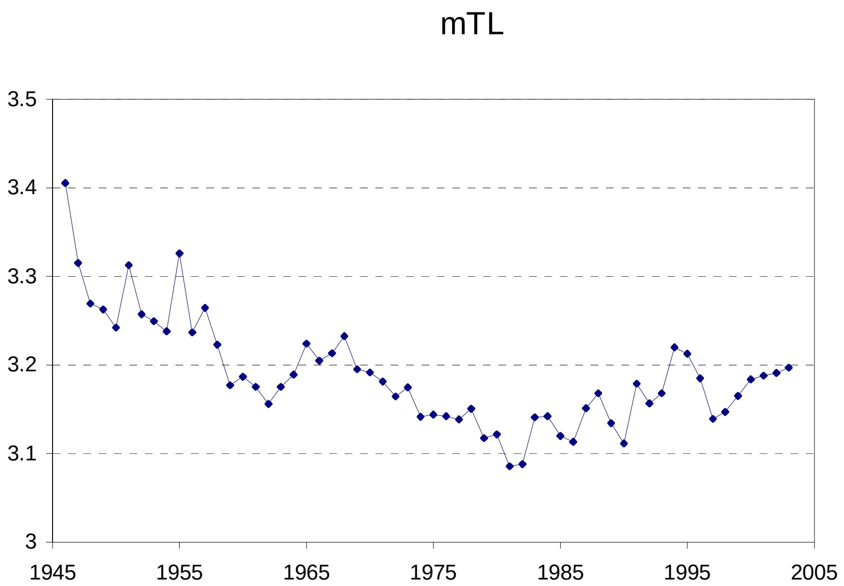
The mean trophic level of the catches is used as a proxy of the mean Trophic level at the community or ecosystem level and the decline of trophic level of catches was thus evidencing disrupting effects of fishing pressure on ecosystems: in fact, fishing activities target large individuals with high Trophic level thereby decreasing the mean trophic level resulting in an effect known as Fishing Down the Food Web (Pauly et al., 1998).

The 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 both from ecosystem models, through back calculation of the trophic flows that, from a considered species, go down to the primary producers through all possible pathways (Christensen et al., 2000), and from field data, taking into account landings of different species and their trophic level and combining them with the Transfer Efficiency of the system according to Pauly and Christensen (1995). 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 seems a straightforward indicator, 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 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 (Pauly et al., 2000; Pauly and Palomares, 2005). The FiB index accounts for landings quantity, for their mean trophic level and the efficiency of energy transfer in the trophic web (Pauly and Christensen, 1995) and can be estimated using both time series of landings and ecosystem models, using the first year of data/output as a reference (Pauly et al., 2000). However, since ecosystem models allow for a dynamic evaluation of transfer efficiency along time, models represent valuable tools for estimating the FiB index. The FiB index allows detecting both bottom-up effects of changes in primary production and top-down effects of fishing pressure. A positive trend in the time series of FiB 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 in accordance with the carrying capacity (Pauly et al., 2000).

The Finn Cycling Index, as defined in Finn (1976), allows the quantification of the cycling of energy in the form of organic matter within a given ecosystem and can be estimated easily from flows in the trophic web, as the fraction of total flows in the system that are cycled. Cycling is considered an important indicator of the ecosystem ability to maintain its structure through positive feedbacks (Ulanowicz 1986, Monaco and Ulanowicz 1997) and was used as an indicator of the maturity stage of the ecosystem (Odum, 1969; Christensen, 1995; Vasconcellos et al., 1997). Fishing is believed to disrupt the ecosystem interfering with normal cycling process and, although fishing impacts can increase or decrease the ability of the trophic web in cycling energy, it is not clear whether this can be positive or negative for the ecosystem. Likely the effects need to be evaluated for each specific ecosystem.

North Adriatic Sea
The time series of landings (1946-2003) was obtained by statistics data of the Chioggia fish-market, the most important one of the area. Collected data included only commercial species and catches were reported mainly at specific level. The Trophic Level (TL) of each species (or taxonomic group) was calculated according to the food items composition by using the Trophlab routine (www.fishbase.com); whereas, PPR was computed according to Pauly and Christensen (1995) and Pauli *et al.* (2000).
Trends of indicators during the time are reported in figure 4.36.



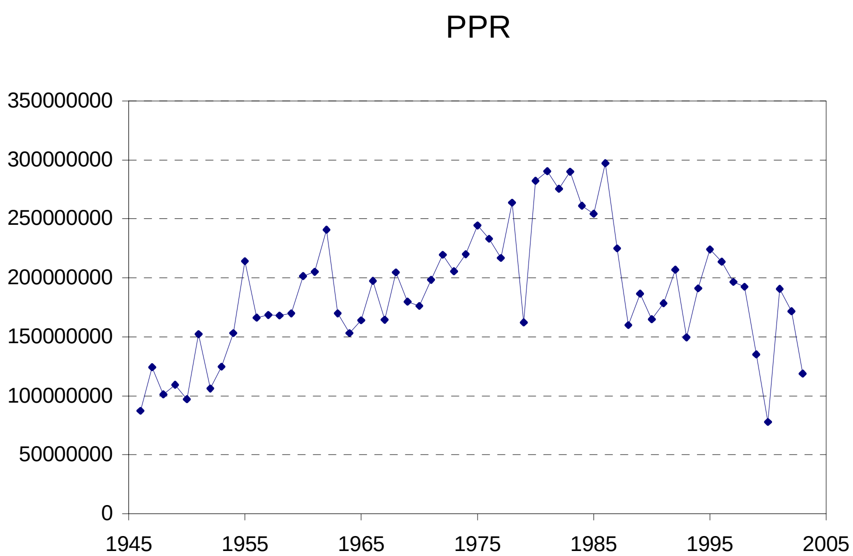
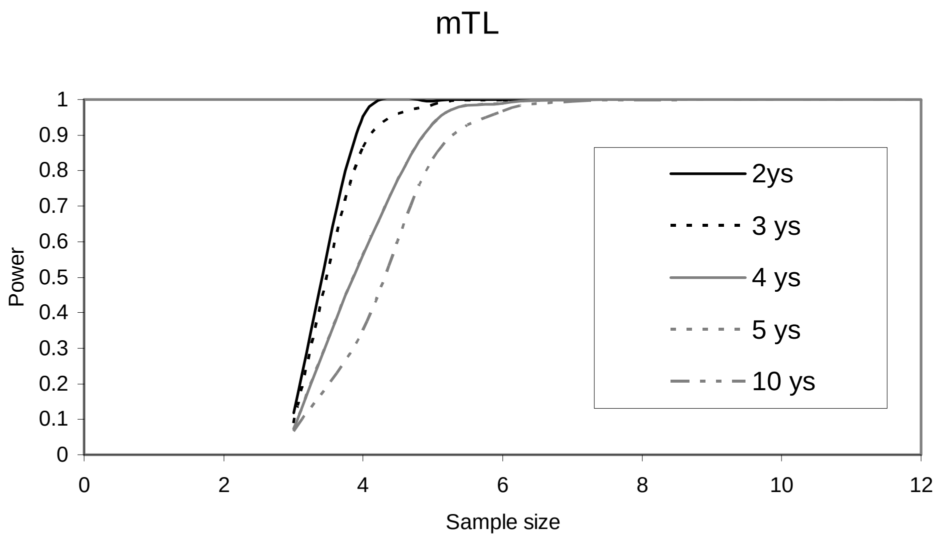


Figure 4.36. Time series for the selected indicators mean Trophic level (top) and Primary production Required (bottom) based on landings data in the North Adriatic Sea

In order to perform the time series analysis, several ARIMA models were fitted to each indicator and the best model was selected according to AIC values; when necessary the Likelihood Ratio Test was applied. The best models resulted to be ARIMA (0,1,0) and ARIMA (1,0,1) for mTL and PPR respectively.
Power analysis was applied to test the best ARIMA model of each indicator. In this framework, the effect size of interest was estimated “objectively” by calculating the slopes of each indicator along the whole time-series considering respectively 2, 3, 4, 5 and 10 years of time lag and selecting the maximum (positive) and minimum (negative) values.
Results shows that the mTL of the catches perform better than the PPR.



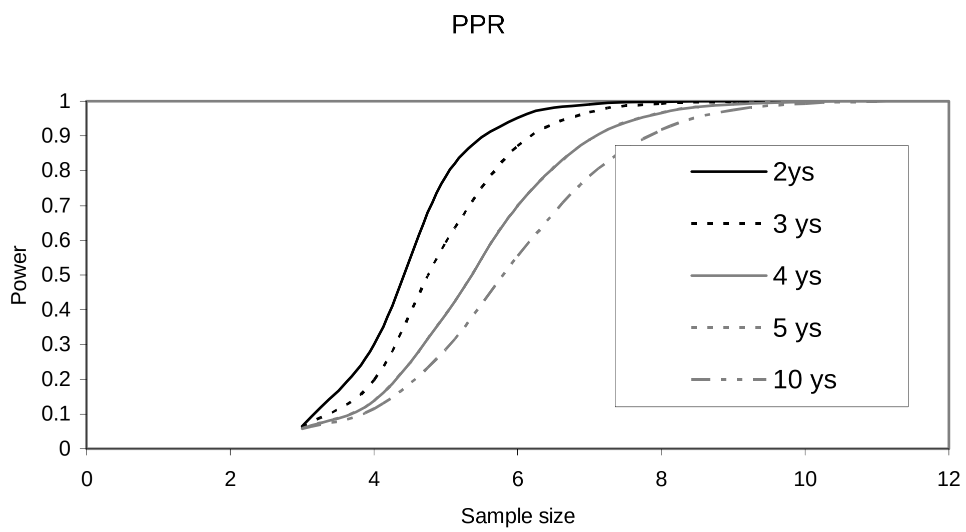
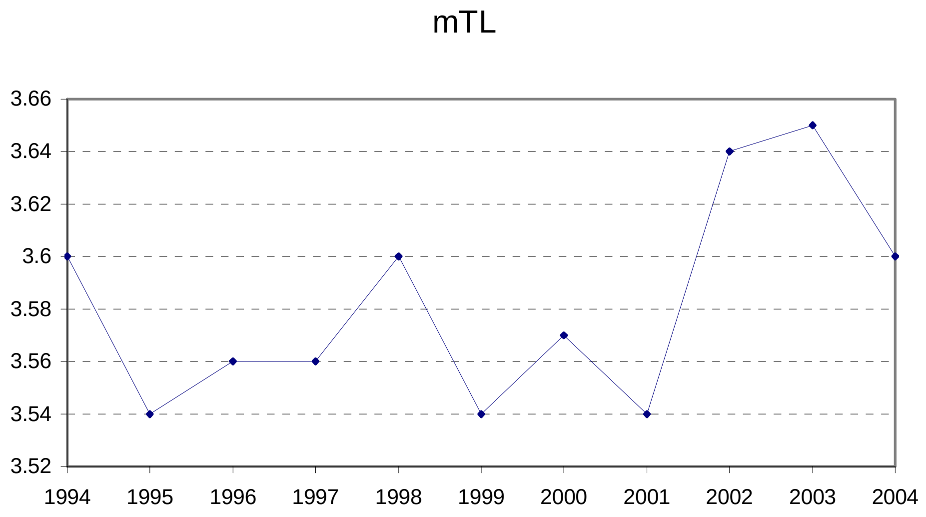


Figure 4.37 Results of Power Analysis for the selected indicators mean Trophic level (top) and Primary production Required (bottom) based on landings data in the North Adriatic Sea. Effects size: the maximum absolute value of slope for 2, 3, 4, 5, 10 years of time lag were considered.

<u>Sicily Channel</u>

The time series (1994-2004) was obtained by using trawl surveys data collected during spring in the framework of the program MEDITS, intended to produce basic information on benthic and demersal species as well as demographic structure, on the continental shelves and along the upper slopes at a global scale in the Mediterranean countries (Bertrand et al., 2002).



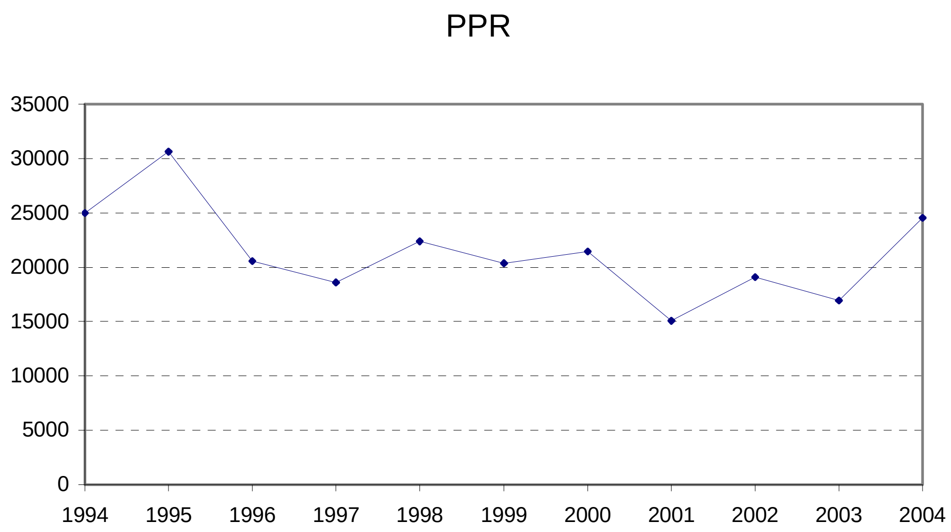


Figure 4.38. Time series for the selected indicators mean Trophic level (top) and Primary production Required (bottom) based on trawl survey data in the Sicily Channel

In order to perform the time series analysis, several ARIMA models were fitted to each indicator and the best model was selected according to AIC values; when necessary the Likelihood Ratio Test was applied. The best models resulted to be ARIMA (0,0,1) both for mTL and PPR.
The Power analysis was applied to test the best ARIMA model of each indicator. In this framework, the effect size of interest was estimated by using slope values of 0.0063 for mTL and -333.27 for PPR. Results shows that for trawl survey data the mTL of the catches have lower performance compared to the PPR (Figure 4.39).

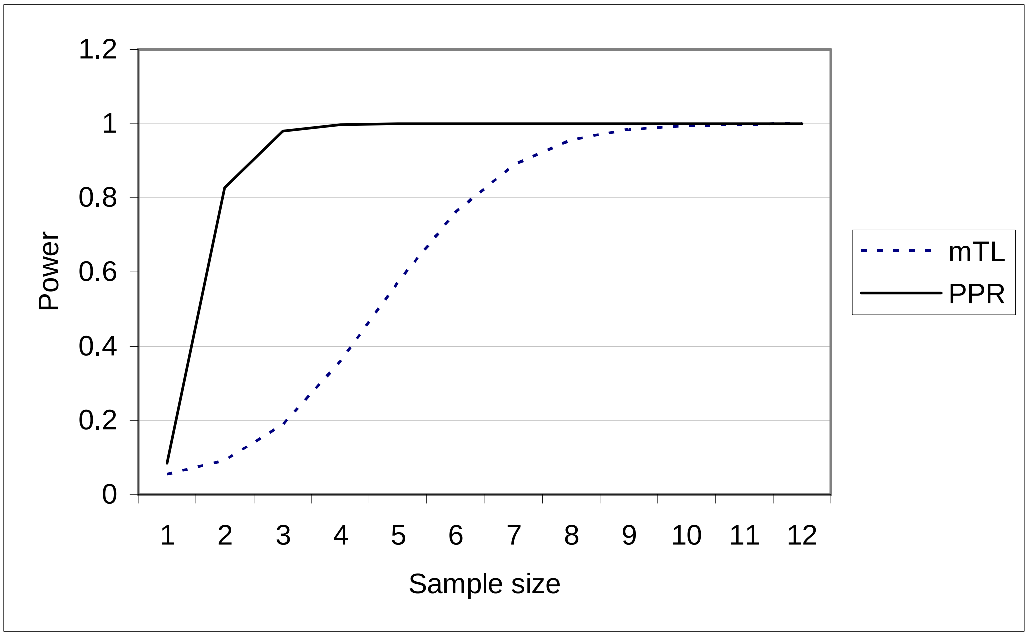


Figure 4.39 Results of Power Analysis performed on the selected indicators mean Trophic level and Primary production Required based on trawl survey data in the Sicily Channel.

North Sea

The North Sea time series (1974-1995) was obtained by using one of the few calibrated Ecopath with Ecosim models available (Christensen et al., 2002). Indeed, the lack of time series of experimental data long enough for the calibration, represents an important limit for the application of such model-based approaches to quantify ecosystem-level indicators. The model represents the trophic web of the North Sea through 32 functional groups (Christensen, 1995).

The dynamic model of the trophic web was fitted using biological data and fishing effort time series for the period 1974-1995. Data are from various sources, but in particular based on the MSVPA model for the North Sea, which gives estimates of relative biomasses by using catches for age classes as input. Since it estimates biomass through back-calculation, these model outcomes are least reliable towards the end of the time-series when the model has not converged (Sparre, 1991; Gislason, 1999). The North Sea model is calibrated using estimates of fishing mortality coming from MSVPA for most of the exploited functional group, and growth parameters for different age classes as in the MSVPA. Biomass indices obtained as output from MSVPA are used as fitting values for calibrating vulnerabilities. Figure 4.40 reports the best fit for biomasses after the calibration process, obtained from simulation of the years from 1974 to 1995, using fishing mortalities estimated from MSVPA as forcing. Analogous graphs can be obtained for total mortality (Z) and catch data. The interest of this analysis is focused on the methodology developed and there is no intention in investigating here the accuracy of the model.

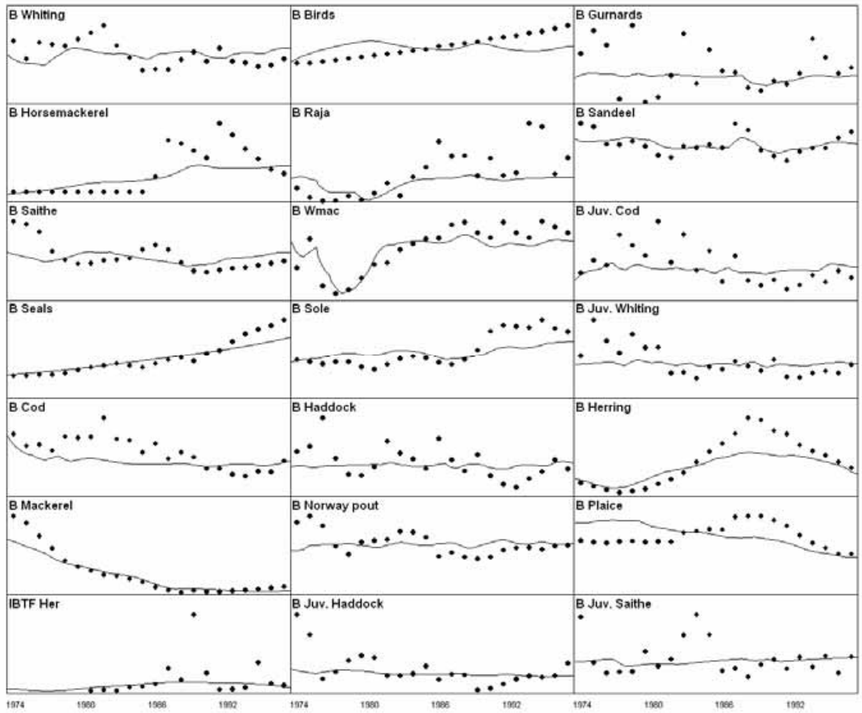
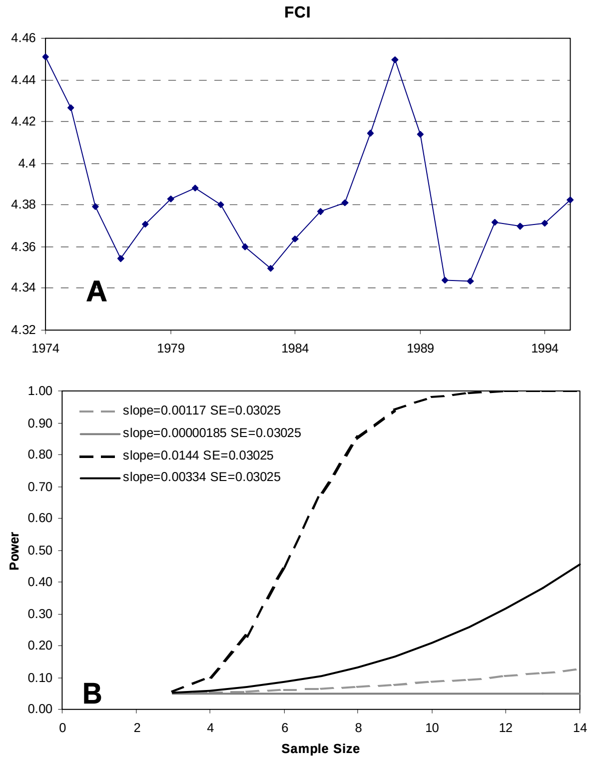


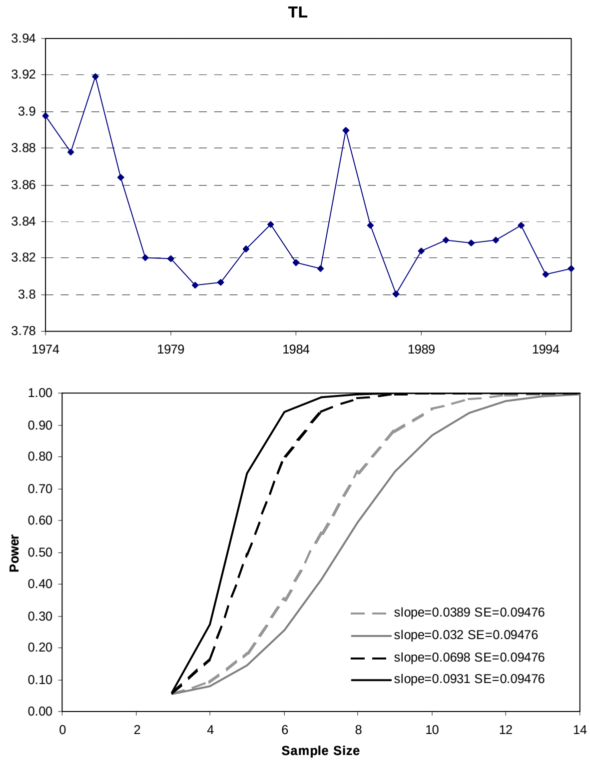
Figure 4.40: Best fit of estimated biomass data (dots) and model outputs (line) obtained from calibration of the North Sea Ecosim model.

The results presented in Figure 4.40 are based on the calibration scenario here referred to as the Reference Scenario. Using the set of vulnerabilities obtained from calibration and running the model for the reference scenario, ecosystem indicator time-series were estimated for the period 1974-1995 from Ecosim outputs.

Finn Cycling Index (FCI), trophic level of the catches (TL) and Fishing in Balance Index (FiB) were derived from the reference scenario for the North Sea Ecosim model. The time series of these indicators are presented in left panel of Figure 4.41.

Different relevant slopes (rate of change) were estimated in indicators time series, including estimates of the average and maximum slope in the time series. Power analysis was applied for all values of SE and slopes estimated, in a kind of sensitivity analysis of these input values on the power. This analysis was done for all input values of SE and slope estimated, but for brevity only the most significant results are reported in Figure 4.41. They refer to power estimations regarding the average and maximum slope estimated with linear and polynomial (sixth order) regression. The significance level used in tests is p=0.05.


FCI
A
B
slope=0.00117 SE=0.03025
slope=0.00000185 SE=0.03025
slope=0.0144 SE=0.03025
slope=0.00334 SE=0.03025
Power
Sample Size


TL
slope=0.0389 SE=0.09476
slope=0.032 SE=0.09476
slope=0.0698 SE=0.09476
slope=0.0931 SE=0.09476
Power
Sample Size

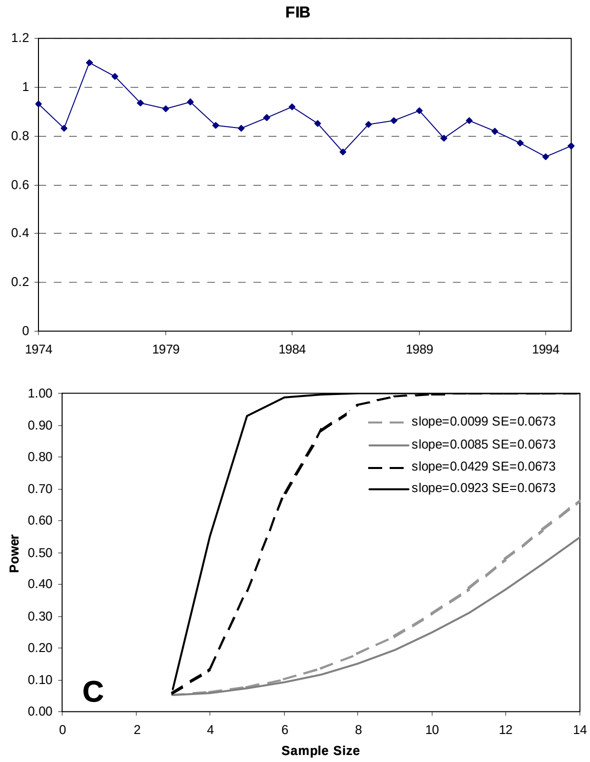


Figure 4.41: Indicators estimated from outputs of North Sea ecosystem model run under the reference scenario. Finn Cycling Index (FCI, panel A), trophic level (TL, panel B), and Fishing in Balance index (FiB, panel C) are estimated for the period 1974-1995 using outputs of Ecosim run under the reference scenario. One tailed power analysis results for average slopes estimated (grey continuous and shaded lines) and maximum slopes (black lines) and for SE estimated from linear regressions.

4.6.12 Pressure indicators

Based on the work of Piet et al. (in press) we will introduce the most common units in which fishing pressure can be expressed and a framework that ranks these units according to their accuracy in quantifying the actual degree of fishing pressure. We will show that the level of accuracy increases with the level of information content because less assumptions need to be made, ultimately leading to indicators that describe the actual ecological disturbance caused by fishing, i.e. the amount of habitat degraded, or the level of mortality inflicted on particular ecosystem components. We chose the Dutch beam trawl fishery in the southern North Sea as a case study. Beam trawling accounts for a high proportion of all fishing activity, particularly in the southern North Sea (Jennings et al 1999), and the Dutch beam trawl fleet is responsible for more than 70% of total beam trawl effort. Furthermore, this fishery has been intensively studied in recent years, resulting in a high level of knowledge regarding the precise operation of the fleet (Rijnsdorp et al., 1998; Rijnsdorp et al. 2000a, Rijnsdorp et al. 2000b; Piet et al. 2000). Despite its potential to cause collateral damage to other components of the marine ecosystem, and although more detailed data are available, beam trawling in the North Sea is still reported using one of the least informative measures of effort (i.e. days-at-sea) at a spatial scale of ICES rectangles (approximately 30x30 Nm). Such measures take little account of species-specific encounter mortality rates, or how these are influenced by the micro-scale (sub ICES rectangle) spatial distribution of fishing effort. We introduce a framework that clearly identifies the information required to quantify different levels of pressure indicator (Figure 4.42), and how each level may be related to different management measures. We show that these pressure indicators become increasingly better descriptors of the actual ecological disturbance caused by fishing as the information content required parameterizing them increases, ultimately producing a measure of fishing-induced mortality. At this highest level, pressure indicators can be directly linked to several of the existing state indicators and perhaps allowing the setting of more realistic reference levels and target points. We investigate the impact of the fishery on a “virtual population” to illustrate the performance of each pressure indicator level as a measure of the actual impact this type of fishing has on key ecosystem components. The consequences of these findings for the collection of data needed to support an EAFM are discussed.

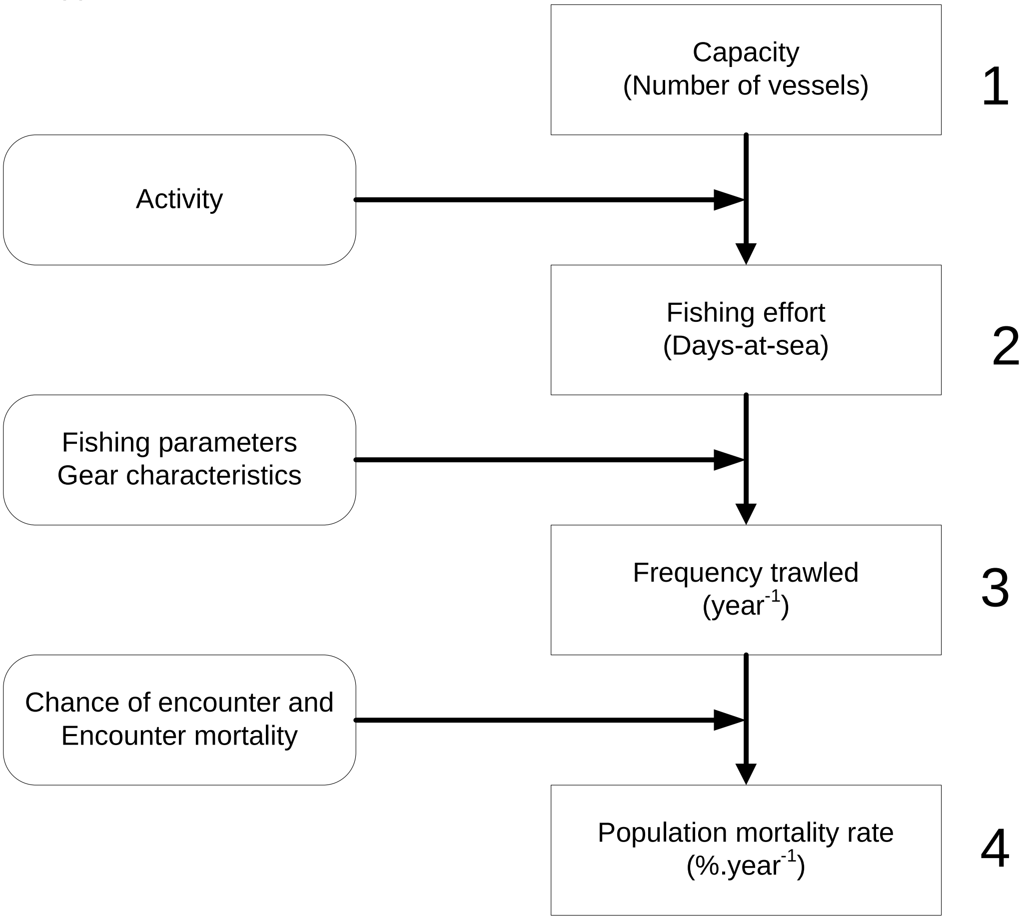


Figure 4.42 Schematic representation of Fishing Pressure indicators at different levels of information content. Activity indicates the number of fishing hours or days at sea per vessel. The fishing parameters and gear characteristics determine how much area is covered by a unit of fishing effort which translates into the frequency with which a specific area is trawled. The chance of encounter determined by the spatial distribution of the population relative to that of the fishery gear and the encounter mortality, expressed as the proportion mortality of individuals in the path of the gear determine the extent to which a population is actually affected by the fishery (i.e. the population mortality rate).

Because beam trawl fishing has such high potential to cause collateral damage to other components of marine ecosystems, including fish and benthic invertebrate communities as well as the seabed habitat, it has long been the focus of considerable attention from fisheries scientists. This has lead to the collection of a diverse range of data, including information on the capacity, spatial coverage and behaviour of different types of vessel within the fleet, and temporal and spatial variation in these at several different scales (see Rijnsdorp et al., 1998; Piet et al., 2000). Two databases that differ in their spatial resolution were analyzed in this study:

1. Low spatial resolution. The VIRIS database, which contains information on fishing activities of the entire Dutch fleet at a spatial resolution of ICES rectangles

(approximately 30x30 Nm) stored in individual fishers' EC-logbooks. Data were extracted for the years 1994 to 2004. The database distinguishes different segments of the fleet based on their engine-power, contains information on the time of the start and end of the fishing trip, the gear used, the ICES rectangles fished and the landings by fish species. The database is designed for quota management purposes but available for research purposes and similar databases are available for other EC countries.

2. High spatial resolution. The APR/VMS database consists of Automated Position Registration (APR) and Vessel Monitoring through Satellite (VMS) data at a resolution of 1 minute latitude x 2 minute longitude squares (approximately 1x1 Nm). APR data are based on a sample of about 10% of the Dutch beam trawl fleet that was equipped with APR equipment for the period 1993-2000 during which the position of the vessels was recorded every 6 minutes (see (Rijnsdorp et al. 1998). The VMS data became available from 2000 onwards when positions of all EU vessels >24 m were recorded for enforcement purposes. From September 2003 onwards this was extended to vessels >18 m and subsequently from the 1st of January 2005 to vessels >15 m. Positions are recorded approximately every 2 hours. Although these data are collected by all EC countries for enforcement purposes, not all countries have access to VMS data for research purposes. For the Dutch beam trawl fleet VMS data from only a subset of the vessels are available for research purposes. In addition to detailed data on track positions, some of the vessels provided data on a haul-by-haul (HBH) basis of the catch of the target species, the trawling speed and the times of shooting and hauling of the gear.

The ecological disturbance of fishing by the Dutch beam trawl fleet was described by pressure indicators at four levels of increasing information content (Figure 4.42). Level one quantifies fleet capacity, i.e. the number of vessels in a fishery, where different fishing métiers (defined by the target species, the fishing gear used and the area visited, (Laurec et al. 1991) may be defined as necessary. Level two is a measure of fishing effort, calculated as fleet capacity (usually in number of vessels but this may also take account of vessel tonnage or engine-power) multiplied by their activity (e.g. number of fishing hours or days at sea). At level three, pressure is described by the trawling frequency and includes information on fishing practice and gear characteristics, enabling for example, the total area of seabed swept by the gear, or the volume of water filtered, in a given period of time to be calculated. At this level it becomes relevant if information on the spatial distribution of effort exists and when this information is available, at what spatial resolution. We evaluated this by distinguishing between: (1) No spatial information available, (2) Low spatial resolution, or (3) High spatial resolution.

Ideally, the effect of a fishery on any ecosystem component should be expressed as a population mortality rate (i.e. ratio of number of deaths per unit of time to population abundance). This can be calculated from landings and discard data. However, landings data are only collected for a limited number of commercial species and are usually not available for international fleets. Discard data are even more difficult to obtain and because of their high cost often suffer from lack of representivity. Therefore we chose to calculate this based on information on the species-specific mortality rate per contact with the gear, combined with information regarding the overlap in spatial distribution of populations of different organisms of concern with fishing activity, gives rise to the level four pressure indicator: population mortality rate.

Level 1: Fleet capacity

Two principal fishing métiers were identified within the Dutch beam trawl fleet; "Large vessels" with horsepower of 300Hp or more that are not allowed to fish inside the 12 nm zone or the “Plaice box” and “Eurocutters”, vessels of less than 300Hp. The number of registered Dutch beam trawl vessels belonging to each métier was determined for each year from the VIRIS database.

According to the VIRIS database the total number of registered beam-trawl vessels declined over the last decade from 378 in 1995 to 224 in 2004 (Figure 4.43). The reduction in the number of “Large vessels” was much smaller in relative terms than the reduction in the number of “Eurocutters”, consequently the proportion of “Large vessels” within the fleet increased from 55% in 1995 to 63% in 2004 (Figure 4.43).

Level 2: Fishing effort

Total annual fishing effort, in terms of the number of days-at-sea was determined for each métier within the Dutch beam-trawl fleet based on the VIRIS database.

Based on the VIRIS database the activity per vessel varied considerably within and between métiers. 87% of the “Large vessels” spent 150-250 days-at-sea per year, with an average of 170 days-at-sea. For “Eurocutters”, mean activity was much less with an average of only 67 days-at-sea per year, but the distribution was skewed because 25% of “Eurocutters” registered less than 10 days-at-sea per year, with many registering only 1 day-at-sea. For both métiers, mean activity per vessel per year decreased by about 1.5 days per year. Total Dutch beam-trawl effort decreased from 49765 days-at-sea in 1995 to 26034 days-at-sea in 2004. Over the same period the proportion of total Dutch beam trawl fishing effort that was undertaken by “Large vessels” increased from 76% to 82% (Figure 4.43).

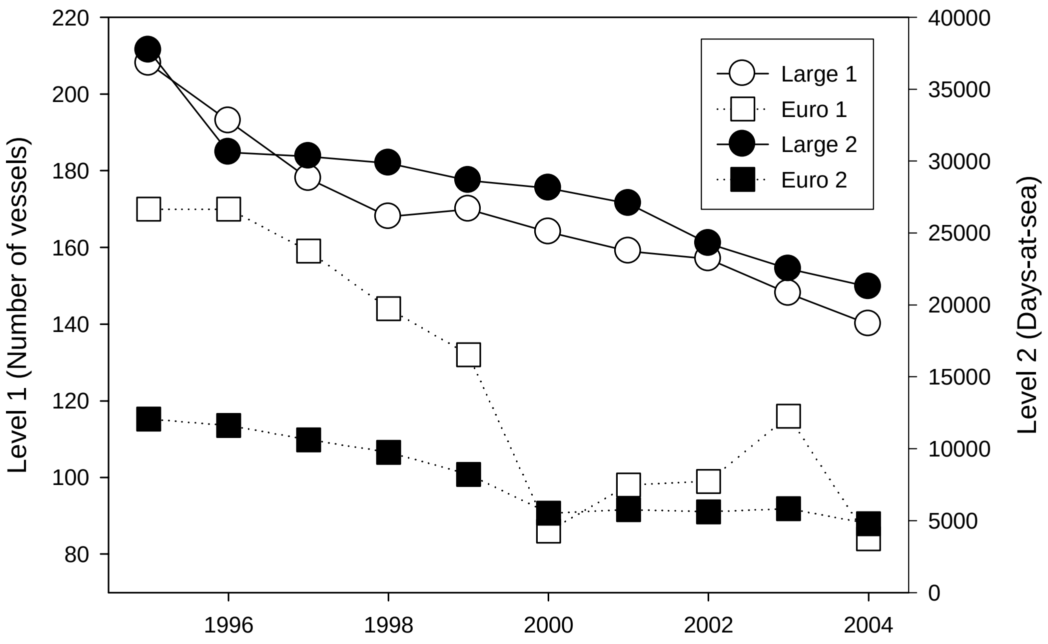


Figure 4.43. Time-series of the pressure indicator at Level 1 (fleet capacity - open symbols) and Level 2 (fishing effort - shaded symbols) for two métiers within the Dutch beam trawl fleet. (Large 1: Level 1 indicator for “Large vessels”; Euro 1: Level 1 indicator for ”Eurocutters”; Large 2: Level 2 indicator for “Large vessels”; Euro 2: Level 2 indicator for “Eurocutters”).

Level 3: Frequency trawled

The total area of seabed swept by the Dutch beam-trawl fleet in any given year (SA in $m^2.y^{-1}$) was estimated by:

$$SA = E * HF * S * 1852 * 2W \qquad 1.$$

where E is the measure of effort, i.e. the number of days recorded at sea by the entire fleet ($d.y^{-1}$); HF is the mean number of hours fished in a day ($h.d^{-1}$); S is the mean trawling speed (knots, converted to $m.h^{-1}$ by multiplying by 1852); and W is the width of the beam (m) with two beam-trawls towed by each vessel.

The APR/VMS database holds registration data recorded at different time intervals. In order to combine these data sets the number of hours fishing per year was calculated for each set as follows:

$$E * HF = \frac{FR * TI}{PD} \qquad 2.$$

where FR is the number of fishing registrations, TI is equal to 0.1 hour (6 minutes) for APR and approximately 2 hours for VMS and PD is the proportion of the fleet in the APR or VMS sample (i.e. for which data are recorded in the database). Note that the left hand side of this equation can be substituted directly into equation 1. The mean trawling frequency (TF) within the area of Dutch beam-trawling operations was calculated as:

$$TF = \frac{\sum_{i=1}^{I} \left(SA_i / A_i \right)}{I} \qquad 3.$$

where SA_i is the area of seabed swept by the Dutch beam-trawl fleet in (sub)area A_i. The area A_i was calculated using GIS (projection UTM-1983, zone 31). Whether or not spatial information on fishing activities was available and if so at what resolution determined I. If no spatial information was available I=1 and A was defined as ICES area IV minus the area deeper than 200m. If based on the VIRIS database or APR/VMS database I equalled respectively the number of ICES statistical rectangles or 1x2 minute squares in which Dutch beam trawl activity had been recorded in that year.

The two métiers within the Dutch beam trawl fleet differ markedly in fishing practice and gear characteristics. Typically “Eurocutters” deploy two beam trawls each of 4 m width, fish with a speed of 4.2 knots and spend about 80% of their time fishing. “Large vessels” deploy two beam trawls each of 12 m width, fish with a speed of 6.7 knots and spend about 75% of their time fishing. This results in “Eurocutters” sweeping an area of 1.2 km^2 on average each day, while “Large vessels” sweep an area of 5.3 km^2. Multiplying these values by the total number of days at sea recorded for each métier in the VIRIS database completes the calculation of equation 1, giving estimates of the total area swept by each métier within the Dutch beam-trawl fleet. The area swept by the total Dutch beam-trawl fleet (SA) is the sum of these two values.

The VIRIS database provided information that identified all the ICES rectangles in which Dutch beam-trawlers were recorded fishing in each year. Likewise the APR/VMS database identified the fished squares. Knowing the area of each rectangle or square, and summing over all rectangles or squares in which fishing occurred, allowed the total area of Dutch beam-trawling operations in each year (A) to be estimated depending on the spatial resolution of the data (Figure 4.44a). The low resolution data indicated that just over one hundred ICES rectangles, amounting to about 58% of the trawlable area of the North Sea, was fished at the start of the time series, declining to about 50% at the end. In contrast, the high resolution data indicated that approximately 20% of the trawlable area (about 26.000 squares) was fished at the start of the period declining to 14%. Figure 4.44b illustrates the final result of calculating equation 2 to determine the level three indicator value for each year, the frequency fished (*TF*).

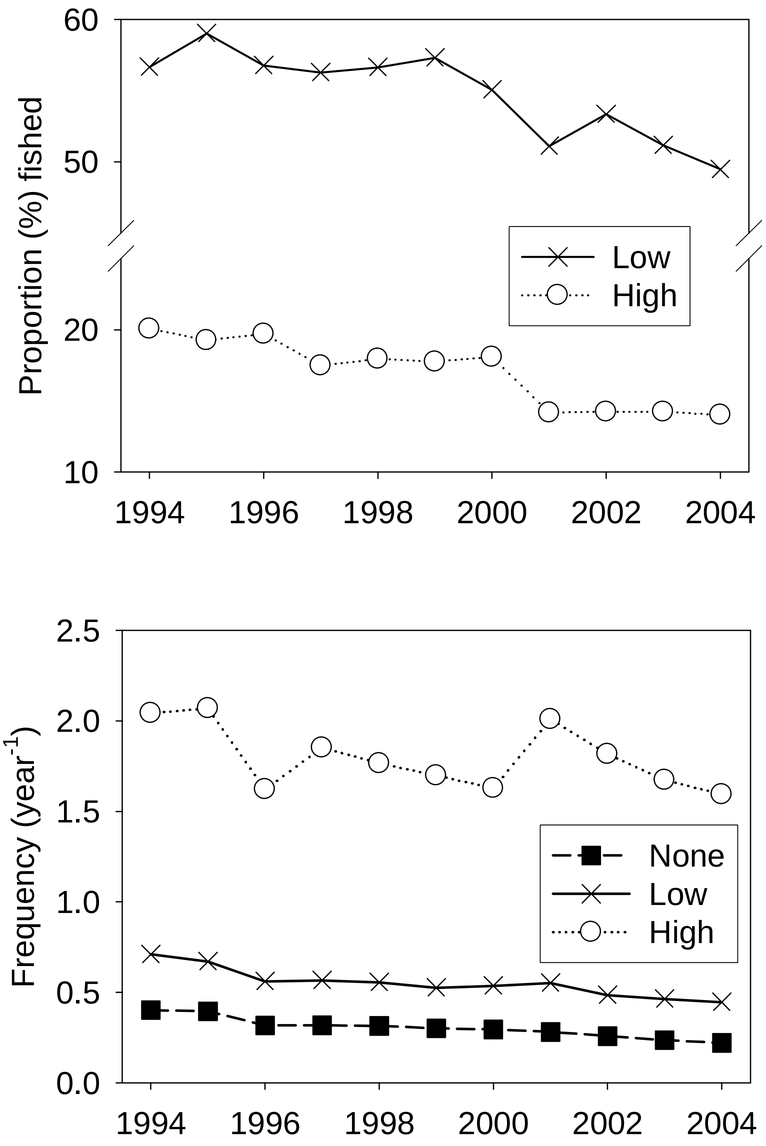


Figure 4.44. Time-series of the pressure indicator at Level 3. The figures show the variation in the proportion of the total area of Dutch beam trawl operations fished in each year (upper panel) and the mean frequency that the "fishable" area was trawled each year (lower panel) depending on whether or not information on the spatial distribution is available and at what resolution (High or Low), If no spatial information was available the proportion of the area fished is by definition 100%.

Level 4: Population mortality rate

The annual population mortality (%) of plaice and sole, inflicted by the beam trawl fishery, is estimated as part of the stock assessment process (ICES 2005) (figure 4.45). The time-series show that over the last decades annual population mortality has increased and contrary to what was observed for the lower level indicators the time-series of both species do not show a decrease in annual population mortality over the last decade but rather a slight (but not significant) increase of 0.04% for sole and 0.55% for plaice (table 4.7).

For non-target species there are no stock assessments and hence population mortalities were estimated following the swept-area method. This resulted in time-series of population level mortality for different virtual populations, with various scenarios depending on the spatial resolution of the fishing data and how the spatial resolution of the virtual population being impacted relates to that of the fishery (Figure 4.46). The time-series of annual population mortality differed considerably between scenarios both in terms of the absolute value as well as the relative change (Figure 4.46 and Table 4.7).

Overall, annual population mortality increases with increasing overlap of the population with the fishery, encounter mortality and spatial resolution and varies between 0.7% and 80.1%. Without spatial information of the fishery (NS) the scenarios that differ in the spatial distribution of the population (low/even/high overlap) show the same annual population mortality for each of the encounter mortalities (20% or 80%). The relative change of the annual population mortality over time also varied between scenarios from a 1.8% decrease to an increase of 1.8%.

Figure 4.45. Time-series of the pressure indicator at Level 4 expressed as population mortality rates of the two target species of the Dutch beam trawl fishery: plaice and sole. The values are for age-classes 2-6 and are based on the 2005 ICES stock assessments.

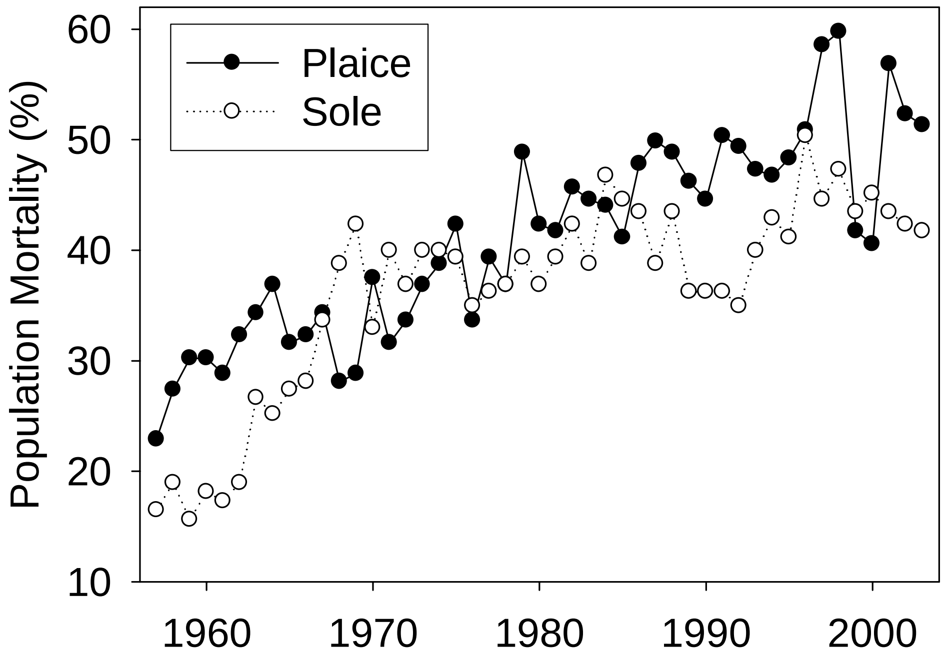
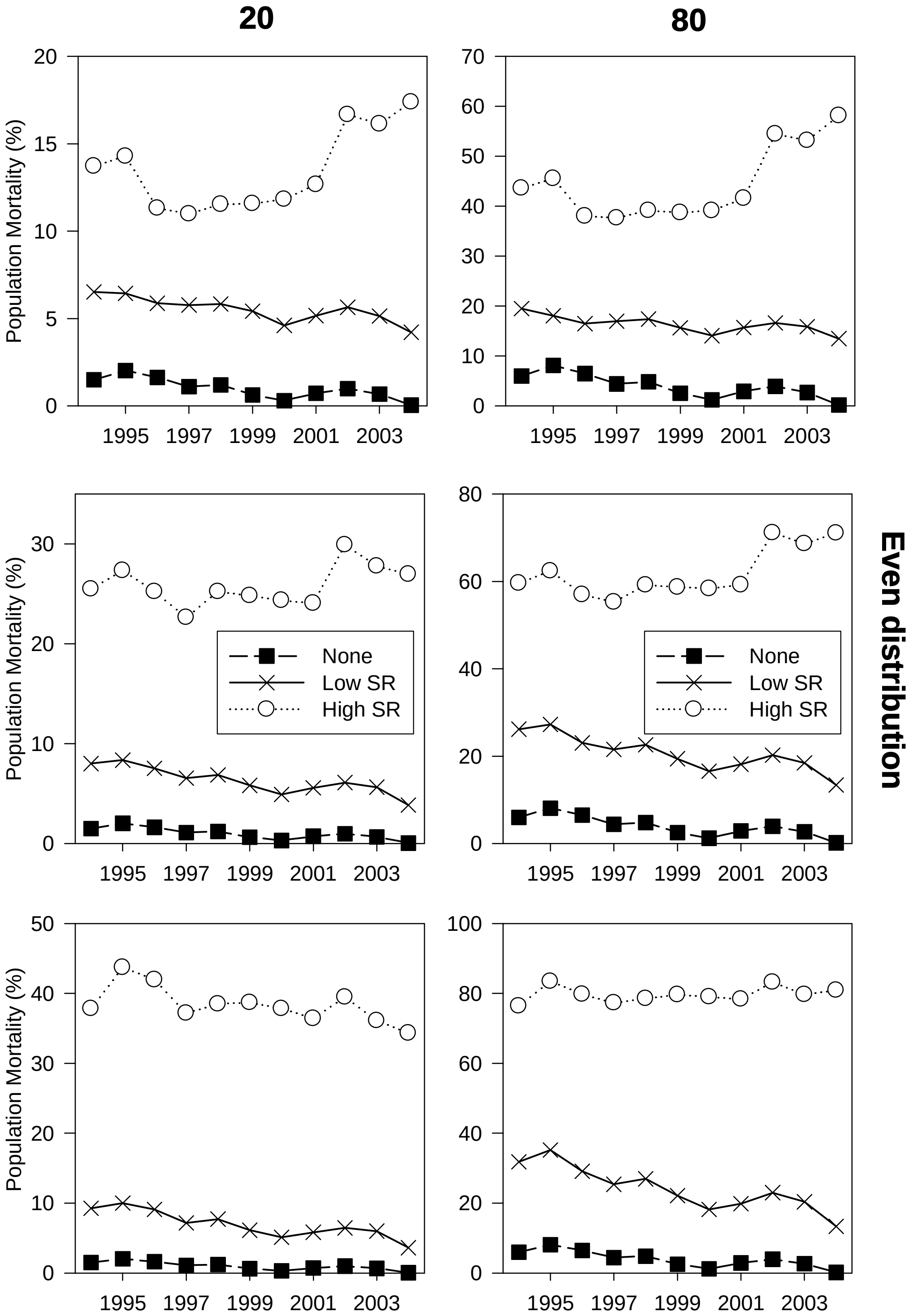


Table 4.7. Summary of absolute values and trends of the pressure indicators at different levels of information content and for populations that differ in vulnerability to that fishery. Level 1 is the fleet capacity, Level 2 the effort in days-at-sea, Level 3 the frequency trawled ($year^{-1}$) and Level 4 the annual population mortality (%). For the spatial distribution of the fleet the distinction is based on the information content of the input, e.g. no information, or spatial information at high or low resolution, for the spatial distribution of the population the distinction is based on the distribution in relation to that of the fleet.

Level	Distribution Population	Encounter mortality (%)	Spatial Resolution	Value in 2000	Relative change (%)
1				276	-6.5
2				34829	-6.8
3			NS	0.3	-5.7
			Low	0.4	-4.1
			High	1.8	-1.6
4		Plaice		50.9	0.5
		Sole		43.9	0.0
	Low overlap	20	NS	0.7	-0.2
			Low	5.1	-0.2
			High	14.2	0.5
		80	NS	2.8	-0.6
			Low	15.4	-0.4
			High	47.2	1.8
	Even distribution	20	NS	0.7	-0.2
			Low	5.5	-0.4
			High	26.1	0.2
		80	NS	2.8	-0.6
			Low	18.4	-1.1
			High	64.2	1.4
	High Overlap	20	NS	0.7	-0.2
			Low	5.9	-0.5
			High	37.2	-0.8
		80	NS	2.8	-0.6
			Low	20.7	-1.8
			High	80.1	0.1

Figure 4.46. Time-series of the pressure indicator at Level 4 expressed as population mortality rates depending on the encounter mortality: 20% (left) versus 80% (right) and the distribution of the population in relation to the fishery: Low overlap (Top), Even distribution (Middle) and High overlap (Bottom). In each graph the time-series were calculated without spatial information on the fishery (None) or a spatial resolution of ICES rectangles (Low SR) or a spatial resolution of 1x2 minute squares (High SR).

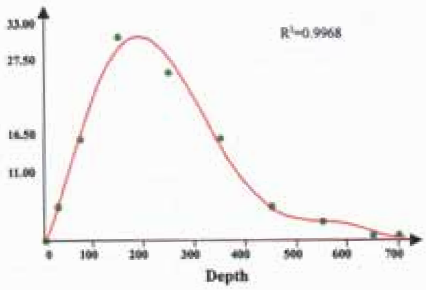


Determining the spatial distribution of fishing without VMS

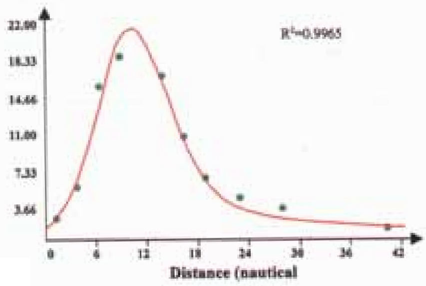
Corsi et al. (2001) analysed the fishing effort distribution in the GSA 9 using a deductive approach. They correlated the spatial distribution of fishing effort with catch per unit of effort (cpue) distribution to assess the status of exploitation of hake. Due to the lack of a direct means of assessing effort distribution, this variable is modelled through a deductive approach that correlates fishing intensity with two variables: distance from the port and depth (Figure 4.47). The range of these two variables was divided into classes and proportion of effort per class was obtained from interviews with local fishing experts. The values obtained were used to fit a continuous function (Figure 4.47). The function was used to assign a score for each 1 km^2 cell of the study area. The overall fishing effort of each port was then partitioned assigning to each cell an estimate of fishing effort that was proportional to its score.

. Use scores for distance from the port and depth as defined by the expert.

Distance (nautical miles)	Use score	Depth (metres)	Use score
2.5	2	10	0
5	5	50	5
7.5	15	100	15
10	18	200	30
12.5	20	300	25
15	16	400	15
17.5	10	500	5
20	6	600	3
25	4	700	1
30	3	>700	1
>30	1		



. Least squares fit function for the depth use scores identified by the expert.



. Least squares fit function for the distance use scores identified by the expert.

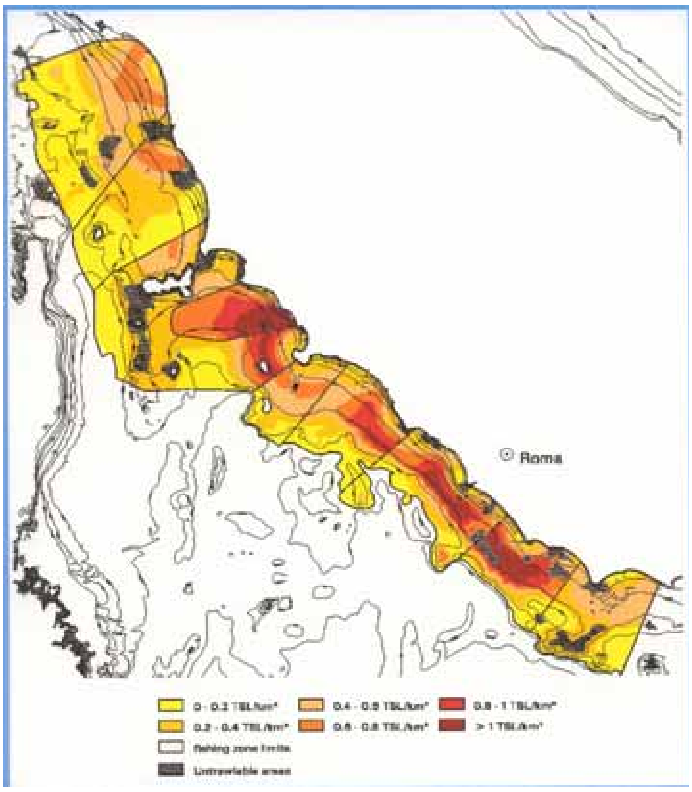
Fig. a. *Scores for variables and least squares fit functions for variables (from Corsi et al., 2001).*

The fishing effort of a generic point p was calculated using the following formula:

$$effort_p = \sum_{i=1}^{P} \left(effortPort_i * \frac{WDist * ScoreDist_{pi} + WDep * ScoreDep_p}{WDist * \sum_{k=1}^{N_i} ScoreDist_{ki} + WDep * \sum_{k=1}^{N_i} ScoreDep_k} \right)$$

$Effort_p$ = Effort of the generic point p
$EffortPort_i$ = Total effort of the i[th] port
$ScoreDist_{ki}$ = Score of the k[th] point according to distance from the i[th] port
$ScoreDep_k$ = Score of the k[th] point according to depth
$WDist$ = Distance weighting factor
$WDep$ = Depth weighting factor
P = Total number of ports in the study area
N_i = Total number of pixels in the fishing ground of the i[th] port

The map below shows the spatial distribution of fishing effort obtained using the above function.



4.7 Sensitivity

Here we will present several studies that examined the sensitivity of fish community indicators to fishing: two studies in the North Sea and one in the Mediterranean. The two North Sea differed in their use of data-sources resulting in Piet & Jennings (2005) studying a larger area but shorter time-span whereas Greenstreet & Rogers (2006) studied indicators in a smaller area but with a longer time-span.

Piet & Jennings (2005) studied the sensitivity of several fish community indicators to fishing used by using the known spatial differences in effort to explore the relationship between various indicators and fishing effort. For this analysis they used the IBTS data to examine differences in the trends in indicator values in areas subject to different intensities of fishing. Since depth explains much of the spatial variation in the composition of North Sea fish communities they distinguished two study areas based on depth: a southern area with depth <60 m and a deeper (>60m), mostly northern area. Within both areas three suites of rectangles or "treatments" were distinguished based on the distribution of international otter- and beam trawling effort by ICES rectangle in 1998 (Jennings *et al*., 1999b; Callaway *et al*., 2002): low effort, intermediate effort, and high effort (Table 4.8 and Figure 4.48). The boundaries were chosen so as to distribute effort categories evenly among rectangles. The effort ranges differed in the two depth categories because notably beam trawl effort in the southern areas was markedly higher. This approach assumes that the spatial distribution of effort over time is relatively constant, in accordance with the observations by Jennings *et al*. (1999b) and Callaway *et al*. (2002).

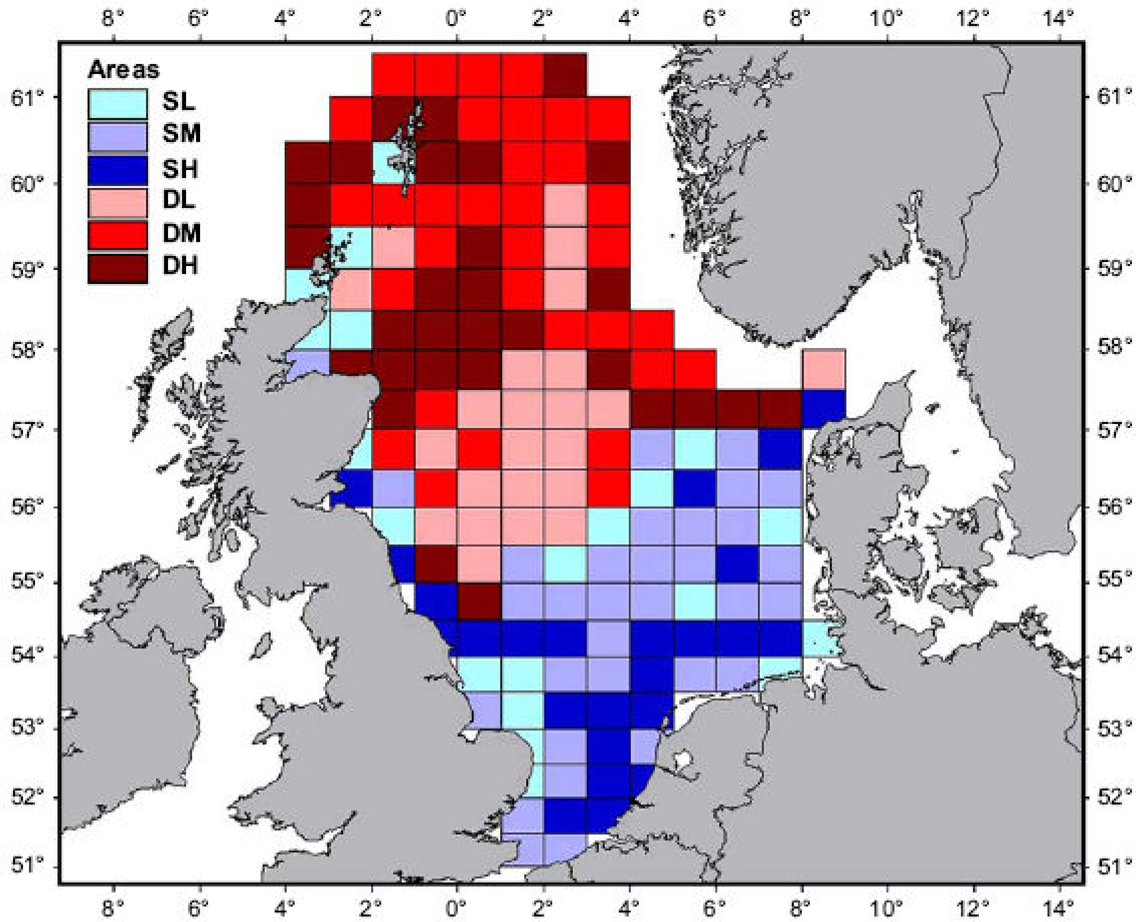


Figure 4.48. Location of six suites of ICES rectangles that are distinguished based on depth and fishing effort. For selection criteria see Table 4.8.

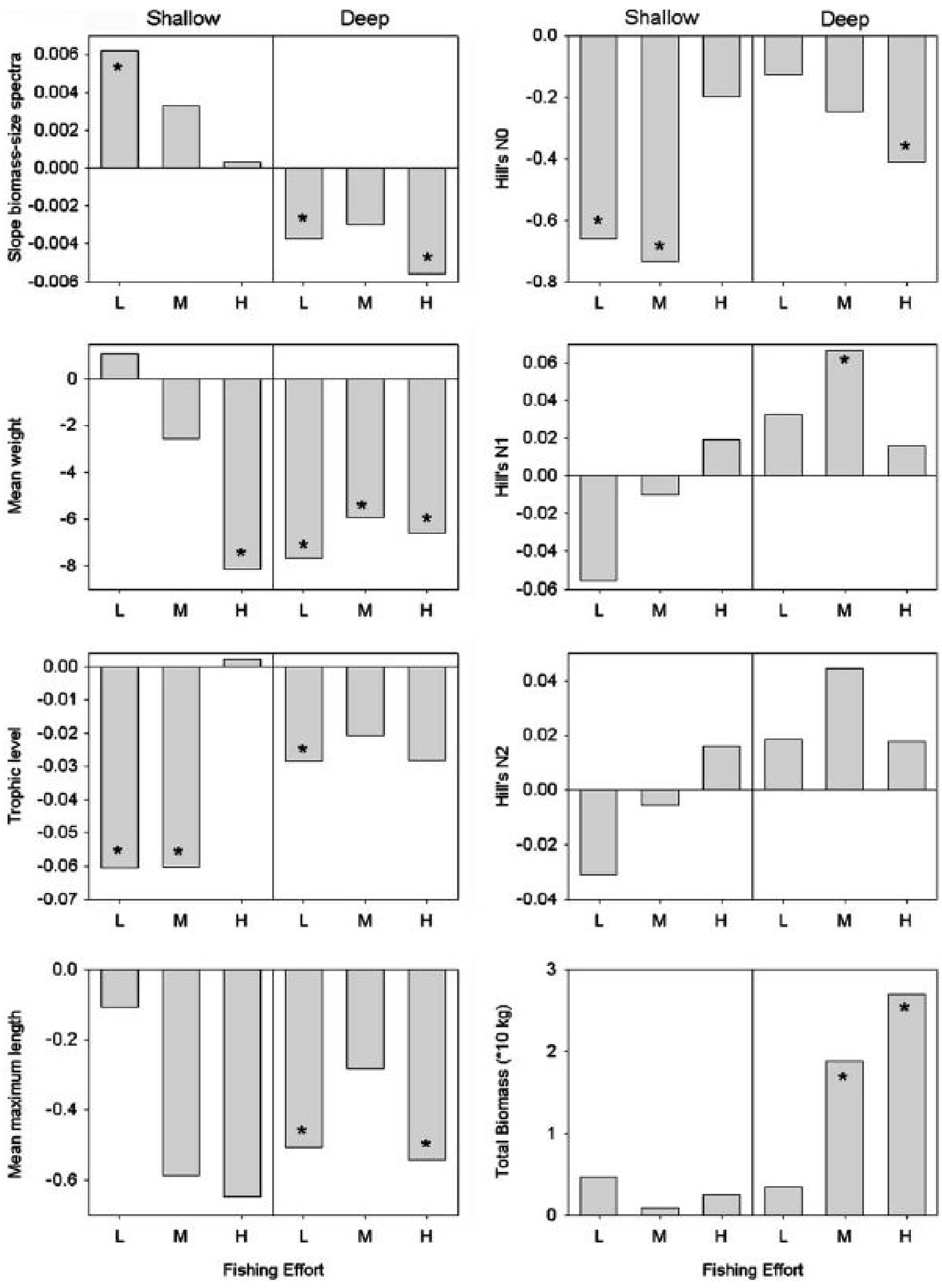
The sensitivity of the indicators to different levels of fishing effort was studied in two areas (see figure 4.48) using the IBTS data. In the shallow areas a distinct relationship between several of these indicators and fishing effort was observed which often was not apparent in the deep areas. Most indicators showed in the shallow areas a gradual change in trend as effort increased. This was not apparent in the deeper areas. For total biomass a gradual change in the deeper areas was observed which was not apparent in the shallow areas. Hill's N0 showed inverse patterns in the deep and shallow areas.
If these correlation analysis reflect a causal relationship then a reduction in fishing effort will result in a slower decrease or even increase of the slope of the biomass-size spectra, mean weight and mean maximum length while the trophic level and Hill's diversity indices N1 and N2 show the opposite effect.

For the analyses aimed at identifying the relationship between fishing effort and fish community indicators, the assumption of an integrated response of the assemblage implies that those indicators that are expected to decrease as a result of fishing will show the strongest decline in the slope of the trend in the high effort areas. In the shallow area this is observed for

most of the indicators (slope of the biomass-size spectrum, mean weight, mean maximum length and the biodiversity indices). However, in the deep area they did not observe a relationship with fishing effort, as there was no gradual change of slope with a change in effort. Also remarkable was that for several indicators the slope in the deep area resembled that in the high effort shallow areas, despite the fact that most of the 'high' effort in the deep area is equivalent to the 'low' effort in the shallow area (Table 4.8). This shows that, in general, there is no straightforward linear relationship between fishing effort and the indicators.
For the biodiversity indices even the cause-effect relationship, as well as the direction of the effect of fishing on these metrics is unclear, consistent with the expectations of Rice (2000) or Bianchi et al. (2000). Fishers may select areas with relatively high abundance of few commercial species and through exploitation of these stocks increase evenness and thus biodiversity. Or, in case of a community with high evenness, the least abundant species may be depleted to the point where they do not occur in the surveys catches which results in a decrease of species richness and thus biodiversity. In previous studies, both positive and negative responses of community diversity to fishing have been found (Greenstreet *et al*., 1999; Rogers and Ellis, 2000). These contradictions are also observed in this study where Hills diversity indices for different surveys, areas and analyses respond inconsistently to fishing. Community biomass did not appear to be a useful indicator of fishing. This is perhaps not surprising when biomass falls most rapidly when fishing is initiated and the greatest reductions in the overall biomass of the North Sea fish community probably took place well before the start of the surveys used in this analysis (Greenstreet & Hall 1996). Moreover, there has been compensation in the North Sea food web due to exploitation and, since the late 1970s, the biomass of larger individuals and species has fallen while the biomass of smaller individuals and species has increased (Daan et al 2003). This compensation would limit the apparent effects of fishing on total biomass, but would lead to decreases in size-based indicators such as mean size, mean maximum size and slope of the size spectrum because the decrease in large fishes and the increase in small fishes act additively to reduce the value of the indicator.

Table 4.8. Characteristics of the six IBTS areas used to assess the impacts of fishing on indicators for the North Sea fish community.

	SL	SM	SH	DL	DM	DH
Area	Shallow	Shallow	Shallow	Deep	Deep	Deep
Effort	Low	Medium	High	Low	Medium	High
Depth range	10-60	10-60	10-60	>60	>60	>60
Effort range (*1000)	<10	10-25	≥25	<5	5-12.5	≥12.5
Number of hauls	123	294	216	277	456	311
Average effort (hours/rectangle)	5178	17275	34356	2787	8831	19531
Beam trawl (%)	47	53	63	14	4	7
Otter trawl (%)	53	47	37	86	96	93
Mean depth (m)	38	34	35	93	134	112
Number of rectangles	20	32	24	23	33	30



Spatial analyses of the responses of fish communities to exploitation such as the above are easily biased if the fish communities routinely move and migrate on larger scales than the scale of analysis (Dingle, 1996). This is because the effects of fishing in one area will be manifest in another. It is certain that many North Sea species routinely move on scales greater than the ICES rectangle (Hunter *et al*., 2004), but there is also clear spatial structure in fish communities that is persistent over time (Callaway *et al*., 2002). The movements of fish to areas where fishing intensity is lower or higher than in the rectangle for which a metric is calculated will undoubtedly modify the relationship between the trend in the indicator and fishing intensity. Thus at progressively smaller scales, and notwithstanding the other influences on fish communities that they describe, trends in the indicator are expected to provide less information on the impacts of fishing.

Greenstreet & Rogers (2006) applied a similar approach to a suite of twelve indicators to Scottish August Groundfish Data collected in the northern North Sea over the period 1925 to 1997. These include indicators of size-structure, life-history character composition, species diversity and trophic structure within the community.
In this study data were analysed for 75 ICES statistical rectangles (0.5° latitude by 1° longitude) divided into three "treatments", of low, medium and high "current" fishing effort. The area considered includes nearly half the surface area of the North Sea. Twelve potential indicators of different characteristics of the ground-fish assemblage were examined to determine the extent to which each was affected by fishing. The hypothesis that each metric was "most affected" in the rectangles of highest fishing disturbance, and least affected in the rectangles of lowest disturbance was tested. Even should the data support these initial hypotheses, this does not necessarily confirm that fishing has been responsible for any observed differences. An alternative interpretation, however unlikely, that fishing activity may have been attracted to areas where the ground-fish assemblage may have displayed particular characteristics, such as low average weight, or low species diversity, cannot at this point be discounted. To rule out this alternative interpretation, long-term time-series trends for each of the groups of rectangles were examined. If fishing is responsible for the change in the community characteristics, then predictable temporal trends should be apparent. Little or no long-term trend should be apparent in rectangles where fishing disturbance is low, whereas in rectangles affected by fishing, temporal trends in a predictable direction should be detected. The greater the impact from fishing, the steeper the gradient should be (Figure 4.49). An assumption underlying this analytical design is that prior to any apparent fishing effects, the community characteristics in the different treatment areas had the same start point, and that each was subjected to approximately the same low level of fishing disturbance.

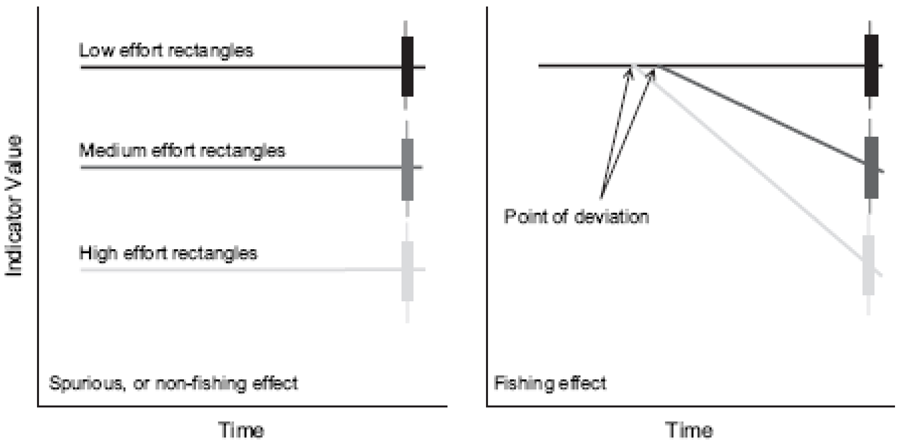


Figure 4.49. Illustration of the analytical design to test specific hypotheses. Box and Whisker plots to the right of each panel indicate community indicator values in each fishing effort treatment in the contemporary period. The lines show the long-term temporal trajectories of indicator values to reach this point under circumstances where fishing activity explains the difference between the treatments and under circumstance where the difference is not a fishing effect.

Long-term trends for the twelve community metrics were determined for each of the three fishing effort treatments. Regression analysis statistics for these trends are provided in Table 4.9. Our hypotheses impart the following predictions regarding these regression statistics. Firstly, gradients of the slope in the low effort treatment rectangles should be indistinguishable from a slope of zero. Secondly, gradients of the slopes in medium and high effort treatment rectangles should be statistically different from zero in a predicted direction. Thirdly, the gradients of slopes in a predicted direction should increase as one progresses from low, through medium, to high effort treatment rectangles. Table 6 summarises the performance of each metric against the three predictions. In some instances, statistically significant outliers from the regression fits were detected. In all cases where this occurred, the regression analysis was repeated excluding these outlying data. In each instance the effect of doing this was to reduce the gradients of the regression lines in High and Medium effort treatments, or to reduce the residual variance in the Low effort treatments, thereby making it easier to obtain a significant trend in these latter cases. With regard to our hypotheses therefore, excluding the outlying data was a conservative action, making it more difficult to support the hypothesis concerned.

Table 4.9. Long-term temporal trends regression analysis results for each community metric in three fishing effort treatments.

Metric	Effort	Constant	Slope	P	Outlier excluded	N
Percent of large fish	High	13.266	-0.1005	0.001	Yes	17
	Medium	9.892	-0.0380	0.231	Yes	17
	Low	10.229	-0.0114	0.819	Yes	17
Average fish weight	High	166.791	-0.9606	0.000	Yes	17
	Medium	152.467	-0.7084	0.009	Yes	17
	Low	156.236	-0.4754	0.219	No	18
von Bertalanffy $Length_{infinity}$	High	55.217	-0.1594	0.000	No	19
	Medium	56.211	-0.1873	0.000	No	19
	Low	52.955	-0.0547	0.238	No	19
Age at maturity	High	2.465	-0.0037	0.001	Yes	18
	Medium	2.398	-0.0019	0.023	Yes	18
	Low	2.373	-0.0007	0.364	No	19
Length at maturity	High	27.307	-0.0510	0.000	Yes	18
	Medium	28.253	-0.0610	0.000	No	19
	Low	26.669	-0.0154	0.460	No	19
von Bertalanffy growth	High	0.214	+0.0018	0.000	Yes	18
	Medium	0.227	+0.0017	0.000	No	19
	Low	0.230	+0.0008	0.000	Yes	18
Number of species	High	45.268	-0.2042	0.000	No	19
	Medium	37.389	-0.0396	0.265	Yes	18
	Low	28.687	+0.0319	0.310	Yes	18
Margaleff's species richness	High	4.380	-0.0215	0.000	No	19
	Medium	3.699	-0.0102	0.005	Yes	18
	Low	3.210	-0.0066	0.046	Yes	18
Pielou's species evenness	High	0.572	-0.0018	0.000	Yes	18
	Medium	0.540	-0.0018	0.000	Yes	18
	Low	0.489	-0.0006	0.299	No	19
Hills N1 species diversity	High	8.143	-0.0476	0.000	No	19
	Medium	6.739	-0.0331	0.000	Yes	18
	Low	5.129	-0.0072	0.393	No	19
Hill's N2 species diversity	High	5.809	-0.0300	0.000	No	19
	Medium	4.607	-0.0182	0.004	Yes	18
	Low	3.428	+0.0008	0.916	No	19
Nitrogen stable isotope ratio	High	14.181	+0.0051	0.193	No	18
	Medium	14.217	+0.0047	0.078	No	18
	Low	14.517	-0.0008	0.676	No	18

Table 4.10. Summary of the performance of the long-term regression analyses performed on each community metric (Table 4.9) against hypothesis predictions. Departures from the predictions are shown in bold italic font. Asterisked entries in final column indicate situations where slopes in both high and low effort treatment rectangles were almost equal and both steeper than the slope in the low effort treatment rectangles.

	Fishing effort treatment			
	Low	Medium	High	
Prediction	Slope in predicted direction significantly different from zero			Steepness of slope gradients in predicted direction
	(No)	(Yes)	(Yes)	High>Medium>Low
Metric				
Percent of large fish	No	***No***	Yes	Yes
Average fish weight	No	Yes	Yes	Yes
von Bertalanffy $Length_{infinity}$	No	Yes	Yes	***No****
Age at maturity	No	Yes	Yes	Yes
Length at maturity	No	Yes	Yes	***No***
von Bertalanffy growth	***Yes***	Yes	Yes	***No****
Number of species	No	***No***	Yes	Yes
Margaleff's species richness	***Yes***	Yes	Yes	Yes
Pielou's species evenness	No	Yes	Yes	***No****
Hills N1 species diversity	No	Yes	Yes	Yes
Hill's N2 species diversity	No	Yes	Yes	Yes
Nitrogen stable isotope ratio	No	***No***	***No***	***No***

For two of the metrics, significant long-term trends were observed in rectangles with low fishing effort. Thus, even in rectangles where fishing activity was low, average growth rates in the fish community increased over the period 1925 to 1996, while Margaleff's index indicated a decline in species richness. Although these significant trends in the Low effort treatment rectangles were not anticipated by our hypotheses, as predicted, the gradients of the trend lines for both metrics were steeper in the medium and high fishing effort treatments. Long-term trends in the Nitrogen Ratio were not significant in any of the three fishing effort treatments. For all other metrics, statistically significant trends in the directions predicted by the hypotheses were observed in the high fishing effort treatments, and in all cases, again as predicted, the gradients of the slopes were steeper than the trend-line gradients fitted to the low effort treatments. Trends in the percentage of fish >30cm and species richness (count of species) were both not statistically significant in the medium fishing effort treatment. However, for all remaining metrics, and in line with the hypotheses' predictions, significant trends were detected on the medium effort treatments, all with steeper trend-line gradients that those fitted to the low effort treatments.

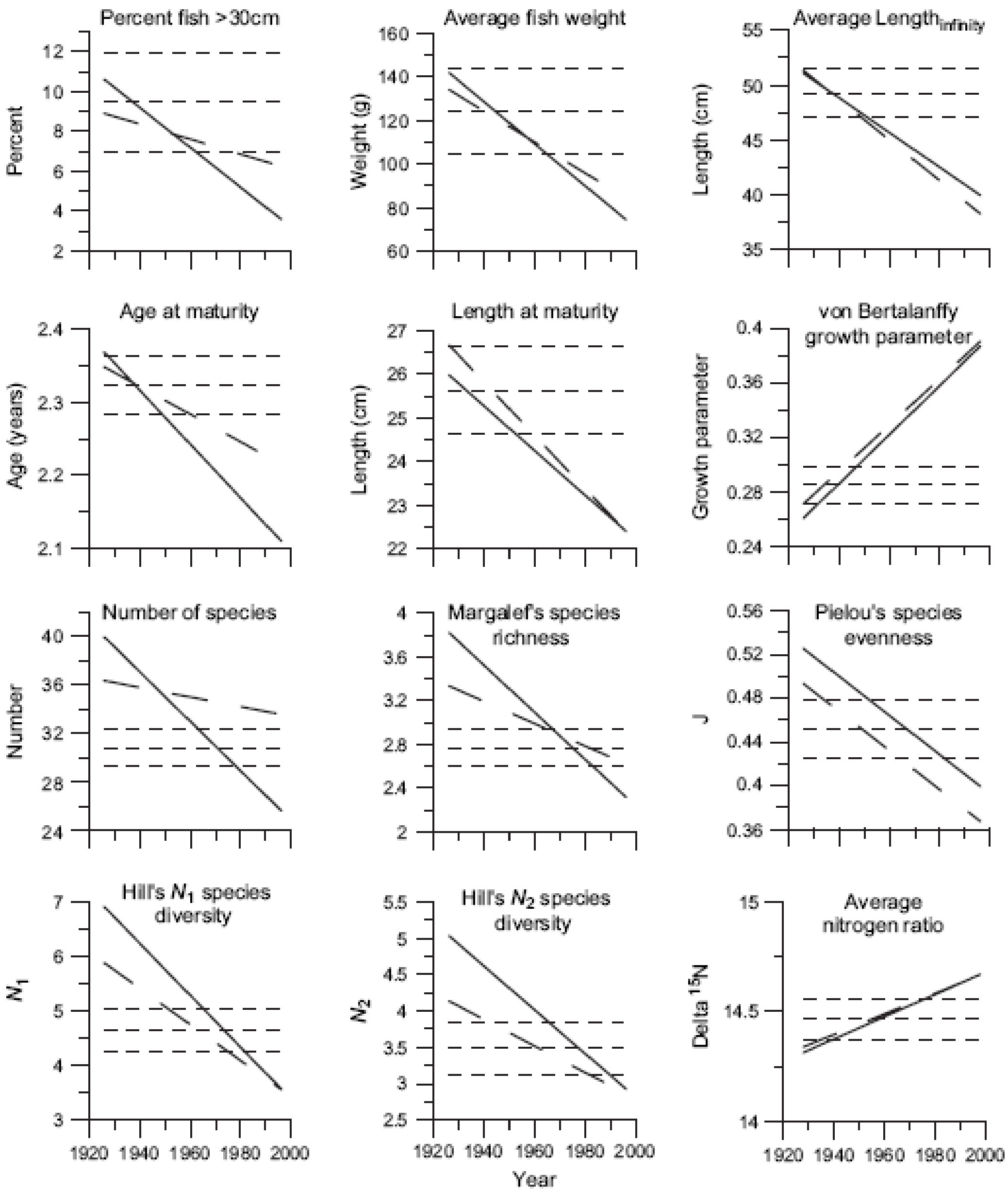


Figure 4.50. Long-term temporal trend-lines for the twelve fish metrics observed in the high (heavy solid line) and medium (heavy large dash line) effort treatments compared with references levels determined from the mean (heavy small dash line) and 95% confidence limits (light small dash lines) of the data in the low effort treatment.

With two exceptions therefore, the potential fish community indicators examined here have behaved in the way predicted by the initial hypotheses. Gradients of the long-term trend-lines in the low effort rectangles have either not differed significantly from a slope of zero, or where significant trends were detected, the gradients were shallower than those in the high and medium effort treatments. These two exceptions are considered further in the discussion, but for simplification, in the next step of the analysis the two metrics involved, Margaleff's species richness index and the von Bertalanffy growth parameter, are treated as if their long-term trends were zero. These results suggest that the low effort treatments provide an indication of the situation "where the anthropogenic influence on the ecological system is minimal". Under these circumstances, the mean and the 95% confidence limits of the low effort treatment data might be considered as possible "reference levels" against which EcoQOs might be set.

Figure 4.50 shows these "reference levels" for all twelve community metrics, and against these, the trend lines for the medium and high effort treatments have been plotted. These plots suggest that the SAGFS data set is sufficiently long-lived as to have "captured" the point in time where fishing activities in the north-western North Sea started to affect the characteristics of the fish community examined. Figure 4.51 shows the same "reference levels" as Figure 4.50, but here the actual time series data for the high fishing effort rectangles are shown. These plots suggest that with respect to the fish size metrics, since 1970 the fish community in the most heavily fished parts of the area was outside the "reference level" lower 95% confidence limit for over 95% of the time. An almost identical situation is apparent for the three life-history metrics. At first glance the situation does not seem to be so bad with respect to fish community species richness and species diversity. Since 1970, only around half the data points for the five metrics fall below the "reference level" lower 95% confidence limit. However, the trend-lines shown in Figure 4.49, and the data in Figure 4.50 suggest a problem with the original assumptions illustrated in Figure 4.49. It is clear that species richness and diversity in the high and medium effort rectangles at the start of the SAGFS period were actually higher than those in the low fishing effort rectangles. Thus the assumption that the three different fishing effort level treatment trend-lines have deviated away from a similar start point is violated. It would seem that levels of fishing activity have been greatest in areas of high fish species richness and diversity, and that fishing activity in these areas has reduced fish species richness and diversity to the same, or even lower, levels found in rectangles of low fishing effort. Under these circumstances, the mean metric values in the low effort treatment do not adequately represent the non-anthropogenically influenced situation.

Mediterranean case study

The results presented for the Mediteranean are based on FAO geographical-subarea 9, Tyrrhenian – Ligurian sea (western Italian coasts). The datasets used are the MEDITS trawl survey data (1994-2004) for the biota and the official fleet data by port for the effort (total engine power, gross tonnage, number of vessels) for the effort.

Analysis of data:

- Fishing pressure: the ratio between total engine power of the fleet and the total surface of the shelf was adopted as indicator of mean fishing pressure. Such index was calculated for three different geographical sectors (central Tyrrhenian Sea, North Tyrrhenian Sea, South Ligurian Sea). In each of these sectors, trawlers perform daily trips, starting early morning and coming back in the afternoon.
- Community indicators: two different indicators have been adopted: mean abundance (n hour^{-1}) and the mean individual weight calculated for each group of organisms (teleosteans, selaceans, crustacean decapods and stomatopods, cephalopods) on the continental shelf (10-200 m depth). These indicators have been compared between the three areas to evaluate the effect of fishing on the shelf community.

Results

Fishing effort appeared to be not homogenously distributed between the three sectors. The central Tyrrhenian Sea is under a much higher fishing pressure (7 Hp km^{-2}), than the north Tyrrhenian sea (3.2 Hp km^{-2}) and the south Ligurian Sea (2.8 Hp km^{-2}) (figure 4.51). Such differences are mirrored in the abundance and composition of the demersal fauna in the three sectors.

In the less exploited sector (south Ligurian Sea) the total CPUEs of demersal fauna were twofold or threefold higher than those obtained in the heavily exploited sectors (central and north Tyrrhenian Sea) (Figure 4.52). Such between-sectors differences were mainly related to an increased abundance of teleosteans, selaceans, and in a lesser extent of cephalopods, in the south Ligurian sea compared with the other two areas. Geographical differences in crustacean abundance are not clear, while the two Tyrrhenian sectors showed similar temporal patterns, in the Ligurian sea abundance showed wide fluctuations from year to year.

Very similar results have been obtained for the CPUEs of piscivorous fishes (Figure 4.53). Data have been pooled for the most abundant top predators of the continental shelf community, hake (*Merluccius merluccius*) over 20 cm TL, John Dory (*Zeus faber*) and the two anglerfish species (*Lophius budegassa* and *L. piscatorius*). A very low abundance of these fishes was observed in the central Tyrrhenian Sea.

Concerning the mean weight, significant between sectors differences have been observed. The mean weight of teleosteans, was generally higher in the south Ligurian Sea than in the two Tyrrhenian areas. Differences from year to year are related to fluctuations in seasonal recruitment of very abundant species, such as hake. No clear differences in mean weight have been obtained for selaceans, crustaceans and cephalopods.

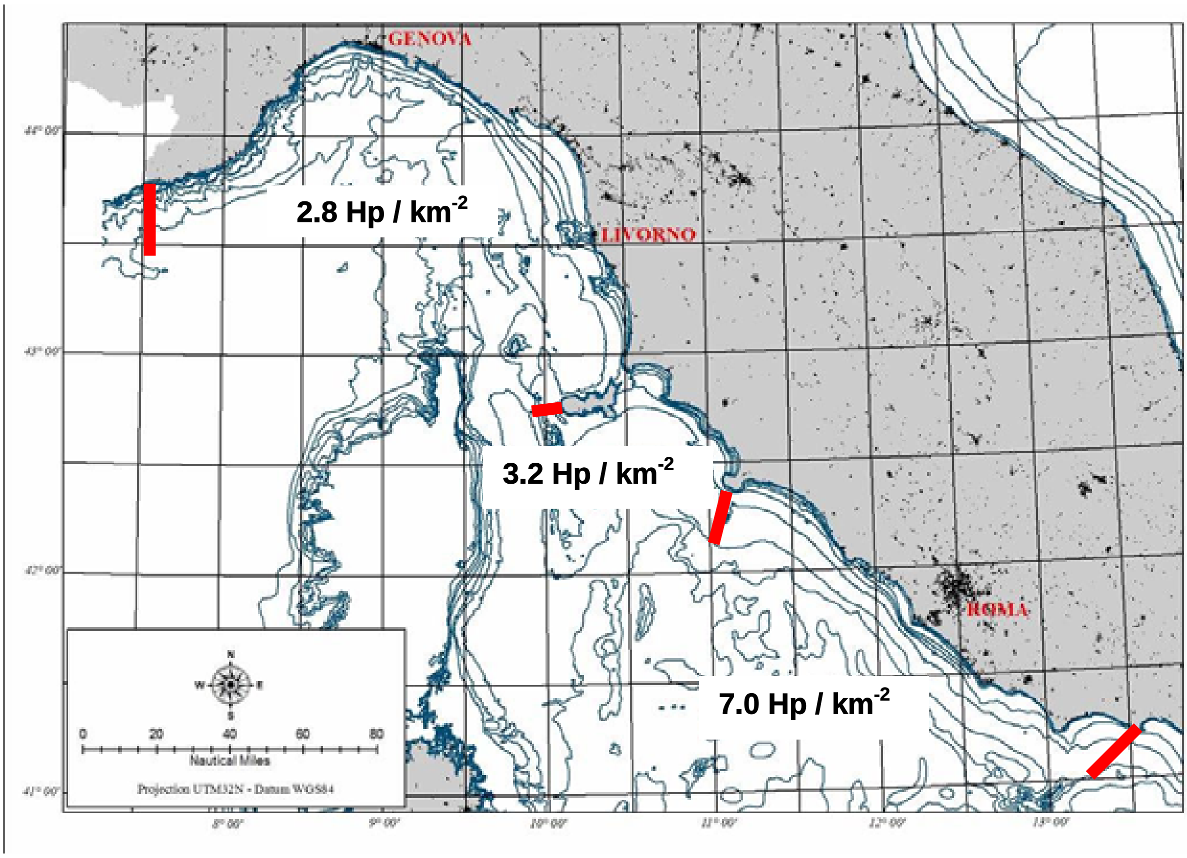


Fig. 4.51. Map of the effort distribution in the three sectors in the FAO geographical-subarea 9, central Tyrrhenian – Ligurian sea.

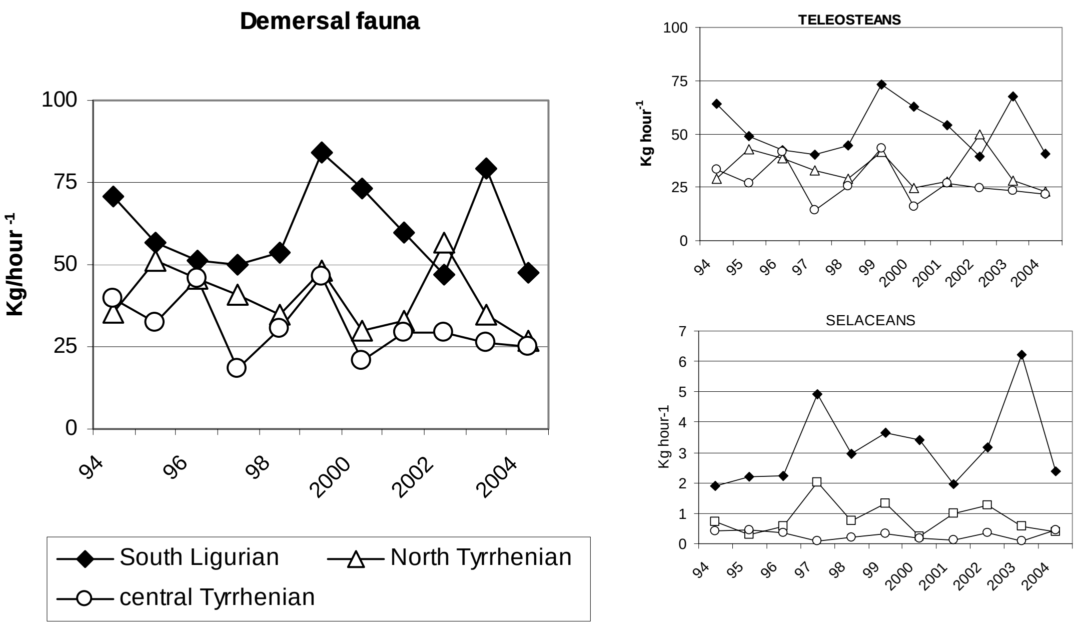


Figure 4.52. Abundance of the whole demersal fish fauna (sx), teleosteans and selaceens (dx), in the three sectors of GSA 9.

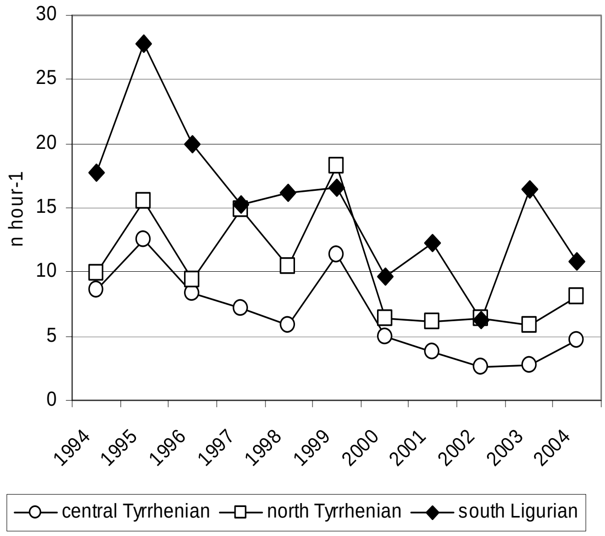


Figure 4.53. Abundance of the piscivorous fishes in the three sectors of GSA 9

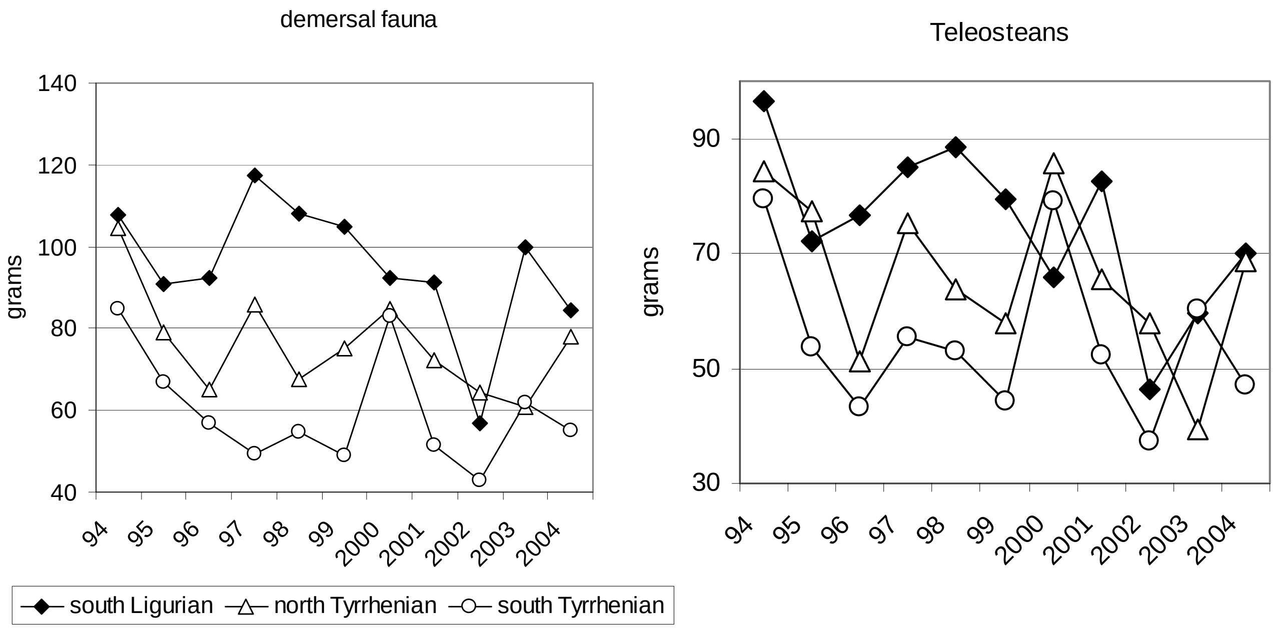


Figure 4.54. Mean weight of the demersal fish fauna (sx) and teleosteans (dx) in the three sectors of GSA 9.

The observed geographical differences in abundance of demersal taxa are mirrored in the structure of communities. In the less exploited area the importance of cephalopods in the community is reduced if compared with the highly exploited areas, while there is an increased importance of cartilagineous fishes.

<u>Conclusions</u>: preliminary results of the analysis showed that the comparison of indicators on a spatial scale can be an appropriate approach to evaluate the effect of fishing pressure on demersal communities, particularly in Mediterranean areas where time series data are relatively short.
The main results showed a strong impact of trawl fishing pressure on the whole community both in term of abundance and mean weight of demersal organisms. Fishes, both teleosteans and selaceans are particularly affected by the fishing pressure, but also cephalopods seem to suffer the current exploitation pattern. Concerning the community structure the main evidence of the effect of fishing

are related to the abundance of predators at the highest trophic level, which showed both differences in abundance on a spatial scale and a negative temporal trend.

4.8 Responsiveness

Piet & Jennings (2005) studied the responsiveness of several fish community indicators to fishing in the SE North Sea after the imposition of a (semi)closed area: the plaice box. The plaice box was established to reduce the discarding of plaice in the nursery grounds along the continental coast of the North Sea, an area between 53^o N and 57^o N (Figure 4.55). It was closed to trawlers with engine power >300 hp in the 2nd and 3rd quarter in 1989, in the 2nd to 4th quarters in 1994 and for the whole year since 1995. These measures resulted in a marked reduction in fishing effort in the box (figure 4.56) and a shift to offshore areas (Pastoors *et al*., 2000). We used the BTS data to compare indicators in two areas: inside and outside the plaice box and in three periods: (1985-1989) high effort inside the box, (1990-1994) medium effort and (1995-2001) low effort. For each metric the mean value and mean rate of change over time was determined per area/period. Spearman's rank order correlation was used to test if the trend was significant.

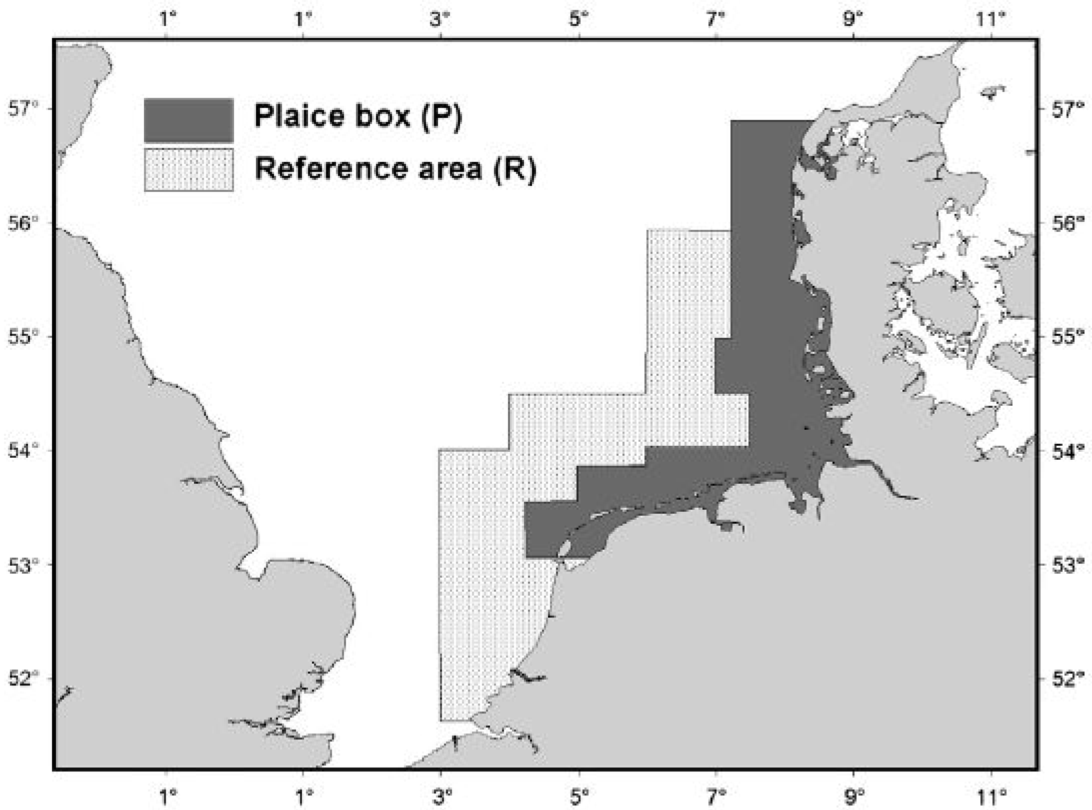


Figure 4.55. Location of the Plaice box (B) and Reference Area (R)

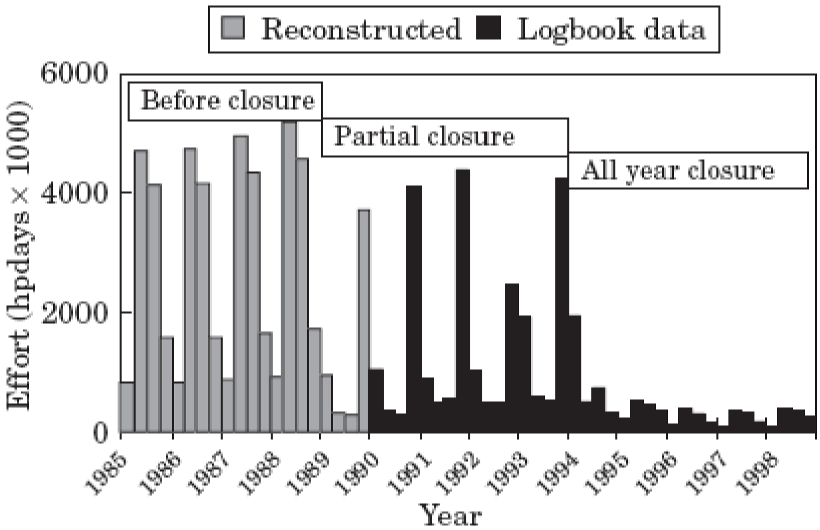


Figure 4.56. Fishing effort (HP days-at-sea) of the Dutch beam trawl fleet fishing in the plaice box by quarter, 1985–1998. Data for years before 1990 were reconstructed; thereafter logbook data are available.

The responsiveness of the indicators to spatio-temporal changes in fishing effort could be studied using the BTS catches in the south-eastern North Sea covering an area where a significant displacement of effort took place after (partial) closure of the plaice box.
For most indicators similar slopes in the trend were observed in the box and reference areas during the subsequent periods (Figure 4.57). Several indicators (i.e. slope of the biomass-size spectra, mean weight, trophic level, mean maximum weight) show a downward trend (negative slope) in the box area during the first period that becomes less steep in the subsequent (lower effort) periods. However, a similar pattern is observed in the reference area in spite of the assumed increase of effort in that area. The difference between the box and reference area, however, does show interpretable patterns (i.e. a gradual change in the slope in the subsequent periods) for indicators such as the slope of the biomass-size spectra, Hill's N2 or the biomass per haul.
Our assessment of the effects of spatial management measures resulted in a response of one indicator (slope in the biomass-size spectrum) that was completely in accordance with theory: a decrease in effort in the box area results in the downward trend becoming less negative while the trend in the reference area becomes more negative as effort from the box area is reallocated to this area. For other indicators the difference between the two areas does reveal a pattern in accordance with theory (Hill's N2 and to a lesser extent mean weight and Hill's N1) but this is not the case in each of the areas separately. The remaining indicators do not show interpretable patterns (trophic level and mean maximum length) or even patterns that indicate an inverse relationship with fishing effort (Hill's N0 and biomass). Thus, the majority of community indicators do not appear to be suitable for assessing the effects of spatial management measures. Whether this is a result of the size of the closed area or other factors, remains to be assessed.

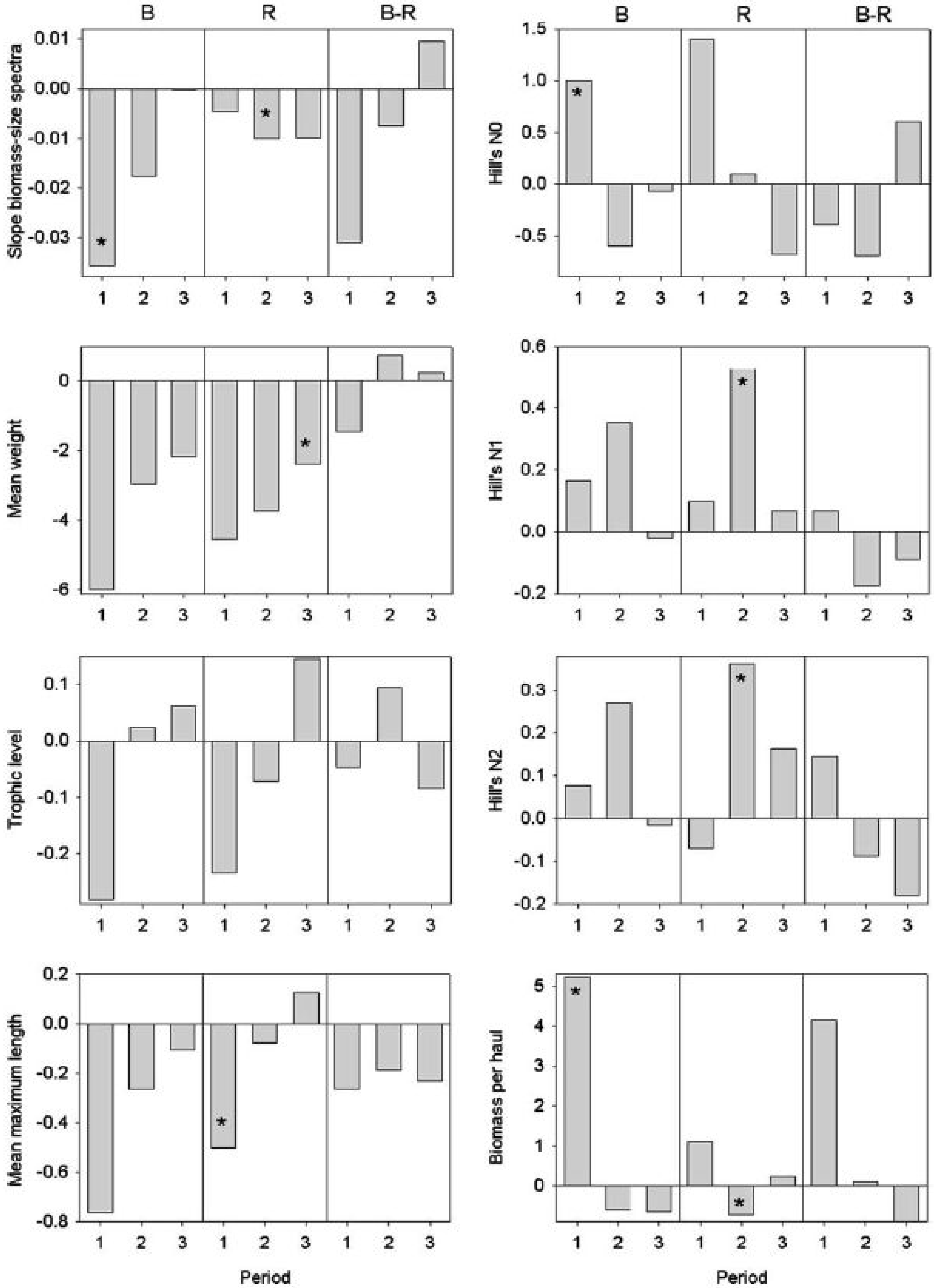


Figure 4.57. Slope of the trend of eight metrics in six area/period subsets based on the BTS. Codes: B=Box (closed) area, R=reference (open) area; 1 (Box open, High effort in Box)=1985-1989, 2 (Partial closure, medium effort in Box)=1990-1994 and 3 (Year-round closure, Low effort in Box)=1995-2001). B-R indicates the difference in trend between the box area (B) and the reference area (R) (= $Slope_B$- $Slope_R$). Asterisks indicate that the calculated slopes were significantly different from zero (Spearman $p<0.05$).

In the case of the plaice box, Pastoors *et al*. (2000) have shown that the measure resulted in a reduction of over 90% of the effort in the box (figure 4.56), and part of this effort was probably reallocated into the reference area thereby increasing fishing effort. Although this provides a clear example of the responsiveness of the indicator of fishing pressure (i.e. fishing effort in days-at-sea), the observed changes in the indicators may not be determined by this effect alone. Pastoors *et al*.

(2000) suggested that increased predation or changes in the distribution of juveniles, factors not necessarily directly related to fishing effort, may have impacted the stock developments in plaice and thus the wider fish community. In addition, temperature measurements during the survey indicate that (at least in the 3rd quarter) there has been a marked increase in the temperature in the south-eastern North Sea that coincided with the establishment of the plaice box and this may have confounded the effects of the changes in fishing effort.

4.9 Specificity

Ideally an indicator is only affected by fishing and should therefore have a high specificity. Belox examples are shown that show how some population and community level indicators may also be affected by factors other than fishing. All these examples come from the Baltic Sea which is known to be heavily affected by environmental factors.

The open sea pelagic fish community in central Baltic Sea constitute few marine fish species. Herring and sprat make up the principal prey items of the top piscivores fish cod and salmon. Both clupeid species feed on zooplankton and herring also consume nektobenthos. Thus, they comprise an intermediate link between the higher and lower trophic levels in the sea. Baltic Sea is subjected to a high fishing pressure, mainly directed towards cod, herring and sprat. Due to natural variation and anthropogenic pressures to the fish community, the sea has undergone severe changes during the last decades. Cod biomass has been reduced from above 1,000,000 tons in early 1980s, to less than 200,000 tons in early 1990s, and the stock has not recovered since then (Figure 4.58, Casini 2006). Also herring biomass has decreased since mid 1970s, from 3,000,000 tons to below 300,000 tons in late 1990s. In contrast, sprat biomass increased in late 1980s and peaked in mid 1990s.

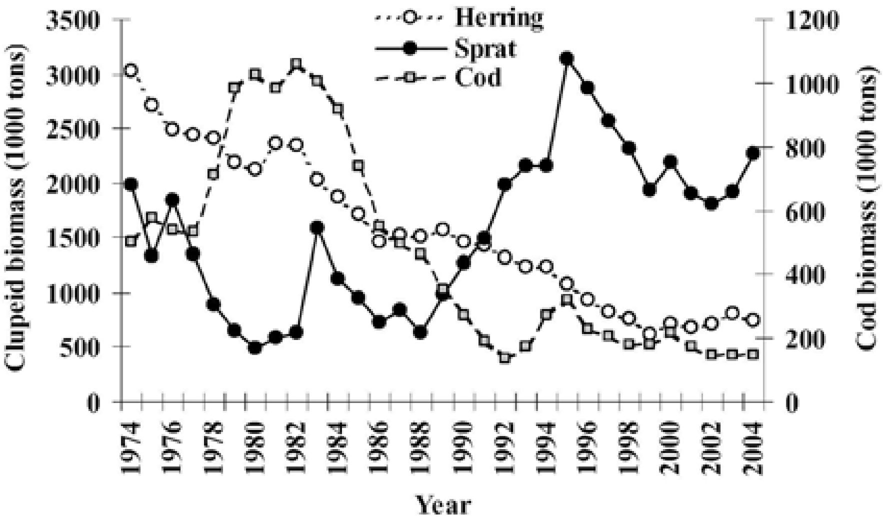


Figure 4.58. Biomass of cod, herring and sprat in central and western Baltic Sea 1974 to 2004. Cod refers to ICES SD 25-32, sprat SD 22-32 and herring SD 25-29 and 32, Gulf of Riga excluded. (From Casini 2006).

Causes to the substantial changes in fish biomass over time possibly relates to several, partly confounding, factors. Climatic variations resulting in an unusual warm period in the 1990s, low inflow of marine water from the North Sea, as well as an ongoing eutrophication possibly play an important role. However, fishing is regarded to be one of the major causes. For cod, fishing mortality (F) decreased in the 1970s, followed by an increase in the 1980s and a dramatic drop in 1992 (Figure 4.59). Fishing mortality of herring increased significantly until late 1990s, where after it has decreased to low levels again during the last six years.

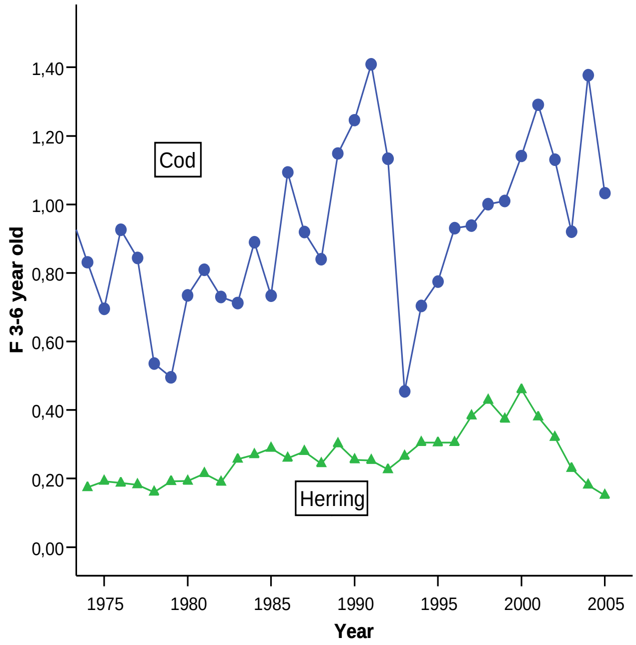


Figure 4.59. Fishing mortality (F) in 3-6 y old cod (in SD25-32) and herring in SD25-29 (excl. Gulf of Riga) years 1974-2005. Data from ICES 2006a.

Cod in the Baltic Sea have been monitored annually since 1982 through bottom trawl surveys (Baltic International Trawl Survey; BITS) that are carried out by research institutes in most Baltic countries. The national research vessels each surveyed part of the area with some overlap in coverage but applied at that time different gears, sampling seasons and sampling designs. In 1985 ICES established formal study groups in order to standardize the surveys. A common standard trawl (the TV3 trawl) and standard sampling procedures were implemented in year 2000. A survey manual has been agreed (ICES 2000) that specifies the standard gear design, fishing methods, biological sampling and formats for data exchange. The manual is regularly updated.

The standard TV3 trawl come in two sizes for different sizes of research vessels, one 520 meshes in circumference and one 930 meshes. The small trawl is used for vessels up to around 800 HP and the larger trawl for vessels with higher engine power. The design and construction of the standard trawls are given in the BITS manual.

BITS is conducted as a depth-stratified random survey during May and December each year. The surveys cover only the southern half of the Baltic Sea (Baltic Sub-divisions 22-28; Figure 4.60). The strata are based on Sub-divisions and depth layers. Each year the necessary stations are randomly selected before the beginning of the international trawl surveys from a list of clear haul data. The standard haul is a 30 minute haul with a towing speed of 3 knots over ground and trawling is restricted to daylight conditions.

Data from the survey results are checked nationally but stored in the ICES international database DATRAS (http://www.ices.dk/datacentre/datras/datras.asp)

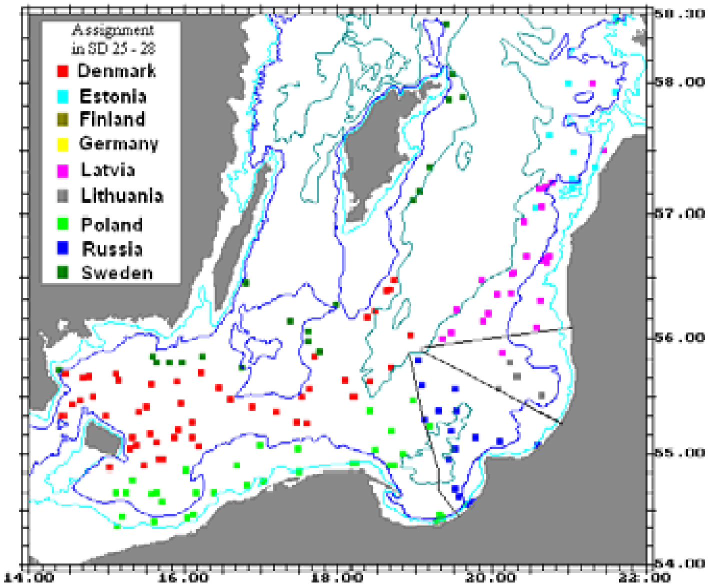


Figure 4.60. Example of the BITS coverage: Positions of planned stations in the Baltic Sub-divisions 25 to 28 in November 2005 by country (ICES 2006b).

In addition to sampling the open sea fish community in the Baltic, sampling of coastal fish have been performed since 1980s using multi-mesh gillnets (Thoresson 1993). Sampling of these coastal reference areas is performed annually in August at depths above 20 m as part of the HELCOM coastal fish monitoring programme (Ådjers *et al.* 2006). The selected areas are generally subjected to low fishing pressure and minor local anthropogenic impact. However, several of the HELCOM sampling sites are exposed to coastal fishing and may thus be tested as ecosystem indicator of fishing impacts.

Size related metrics

There are several metrics describing size related changes over time. For herring, normalised mean weight of two-year-old individuals was found to decrease over time in SD25-28 (Figure 4.61).

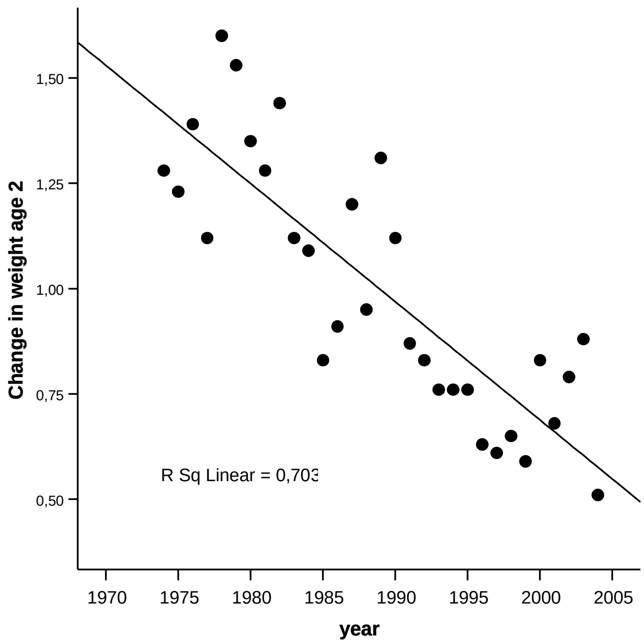


Figure 4.61. Normalised mean individual weight of two-year-old Baltic herring in SD25-28. Data from ICES 2005b.

To what extent this decrease in mean weight is related to fishery is not clear, and several contrasting explanations have been put forward (Sparholt 1996). Among them, it has been suggested that biomass of the zooplankton *Pseudocalanus elongatus* have declined due to reduced inflow of saline North Sea water to the Baltic Sea. Being a main prey item for herring, this decline is suggested to cause reduced condition of herring (Möllman *et al*. 2003, Figure 4.62).

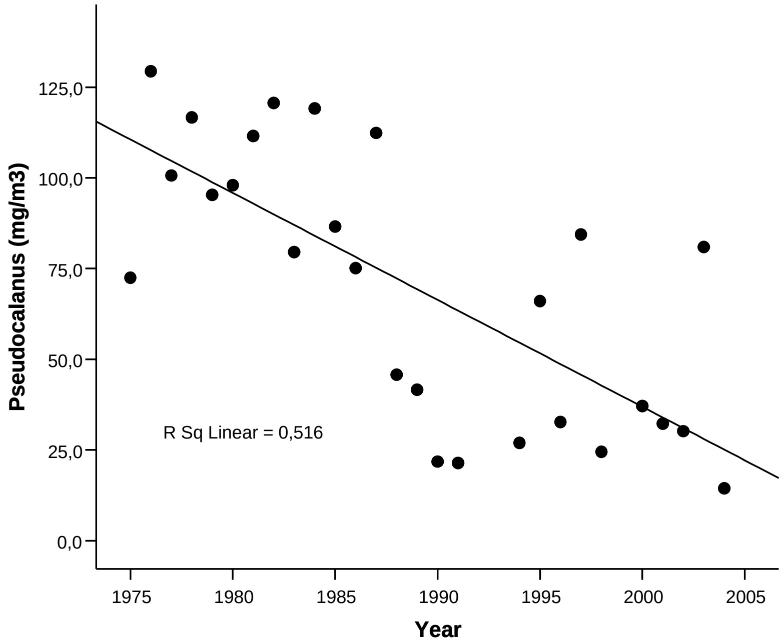


Figure 4.62. Summer biomass of Pseudocalanus (mg/m^3). Data from Christian Möllman, DIFRES.

However, recent studies suggest that a decline in herring condition could be related to a density dependent top-down regulation on a common food resource by the clupeid species (sprat and herring) in the sea (Casini *et al*. 2006). Thus, the observed change in individual mean weight of two year old herring may be a cascading effect of the substantial reduction of cod biomass, relaxing predation pressure on sprat biomass, thereby increasing inter-specific interaction between sprat and herring. Both hypotheses are to some extent supported by the positive relation between *Pseudocalanus* volume during summer and individual mean weight of two-year-old herring (Figure 4.63).

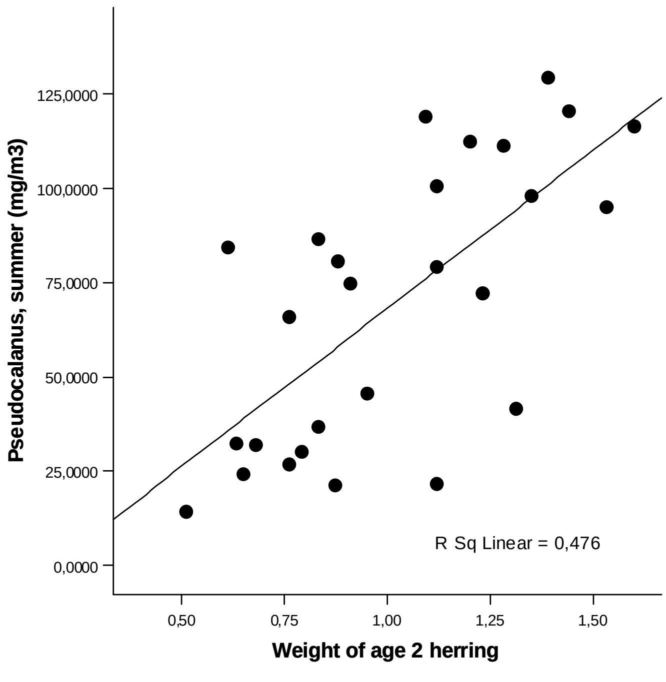


Figure 4.63. Relation between Pseudocalanus biomass (mg/m3) in central Baltic and normalised weight of two- yr-old herring (data from Christian Möllman, DIFRES, and ICES 2005b)

Another example of a size related metrics is the reduction in cod mean size over time in the Baltic Sea. Maximum mean weight of cod was estimated for the period 1988 to 2004 in the areas SD25-28 using fishery independent data from BITS (ICES 2005a, Figure 4.64)

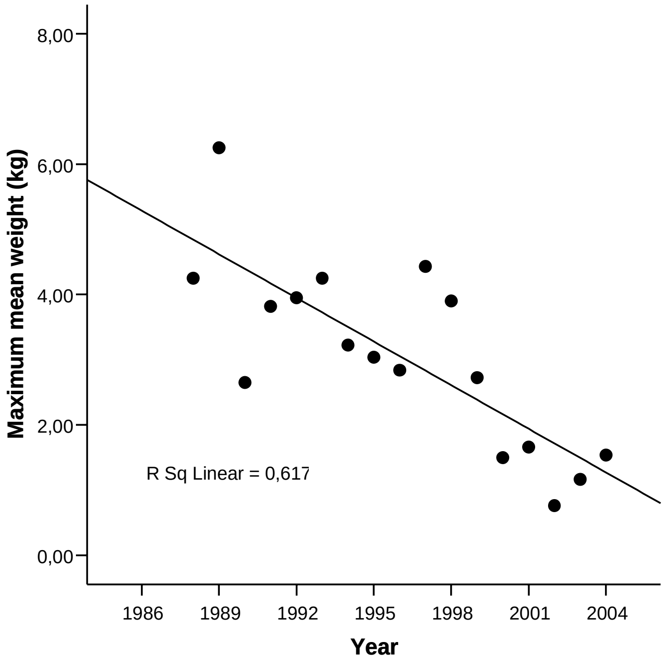


Figure 4.64. Maximum mean weight of cod in the Baltic Sea, years 1988-2004. Data from Swedish R/V Argos surveys in BITS, (ICES 2005a).

As fishing mortality of cod has been high and sprat biomass, which is the major prey item for cod, has increased over the time period, it seems plausible that the intense fishing pressure on cod is the main cause to the reduction in cod size over time.

The above examples point to the problem in finding straightforward relations between pressure and state indicators. Changes in size related population metrics may not only be the direct result of fishing pressure (as in the cod example), but also reflect indirect effects caused by environmental changes and/or more complex cascading effects.

Size related metrics - high fishing pressure
The demersal fish community in central Baltic Sea is dominated by cod, but also include flatfishes, salmonids, coregonids among other fish species. Data from BITS, collected by the Swedish R/V Argos, are used to demonstrate changes in mean weight of the fish community years 1987-2004 (Figure 4.65).

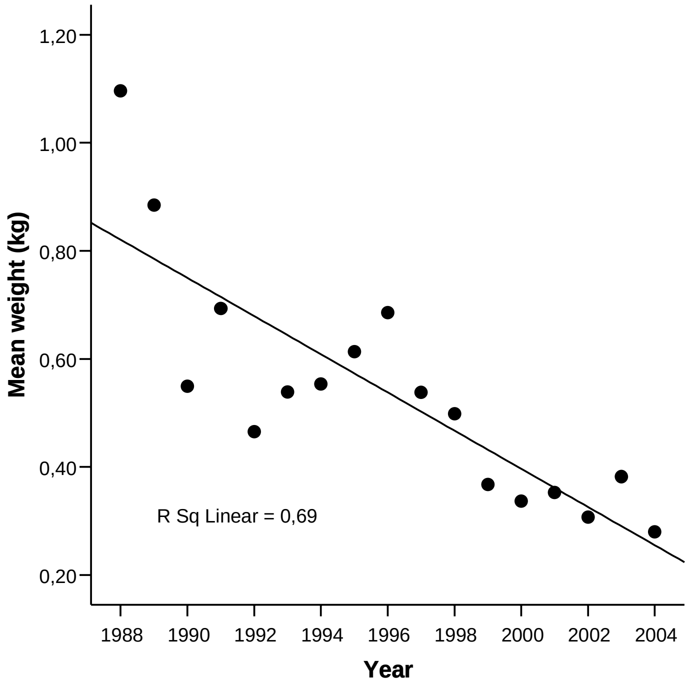


Figure 4.65. Mean weight of non-pelagic fish species in Baltic Sea (SD 25-28) years 1988 to 2004 based on BITS data (ICES 2005a).

It is reasonable to suggest that the observed reduction in mean weight of caught fish is related to a high fishing pressure directed towards demersal fish species, especially cod. However, when estimating maximum mean weight, both large sized salmon and sea trout will confound the results as these two species are less affected by the cod, sprat and herring fishery in the Baltic Sea (Figure 4.66).

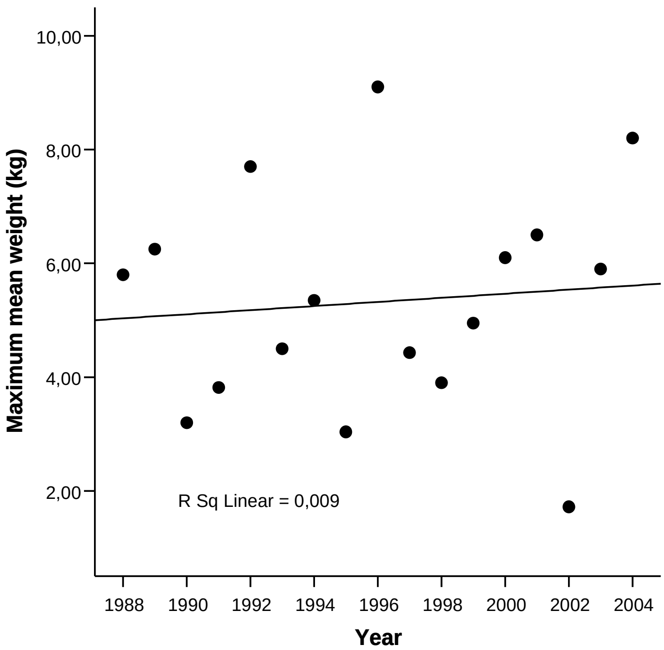


Figure 4.66. Maximum mean weight of non-pelagic fish species in Baltic Sea (SD 25-28) years 1988 to 2004 based on BITS data (ICES 2005a).

HELCOM data from the annual coastal monitoring in the western Estonian archipelago (the Sea of Straits) illustrate the effects of fishing on community size-spectra of coastal fish. During the investigation period (1993-2005), sampling was performed annually in August at 2–5 m water depths using multi-mesh gill net sets (Thoresson 1993). The smallest length group of caught fishes (<15 cm

TL) is not fully selected. Therefore this size groups were excluded from further analysis in order to avoid problems caused by selectivity patterns.

Results expressed as slopes in ln(length) of all caught species show a negative trend over the studied period (Figure 4.67), where slope values decrease from around -3.4 to -6.9. The decrease corresponds to a substantial increase in fishing effort in early 1990s after the independence of Estonia (Vetemaa *et al.* 2005). Although the effort decreased in the late 1990s, the intense earlier fishing has been followed by tremendous changes in the fish community. The calculated slopes thus confirm other observations on less abundant large species and changes in biodiversity. The results should be used to indicate long-term trends rather than annual variations since annual perturbations in environmental or anthropogenic factors might confound the analysis. The important consideration is not the slope or intercept of a spectrum for a particular year, rather how the spectrum changes over a longer period of time (Rice& Gislason 1996).

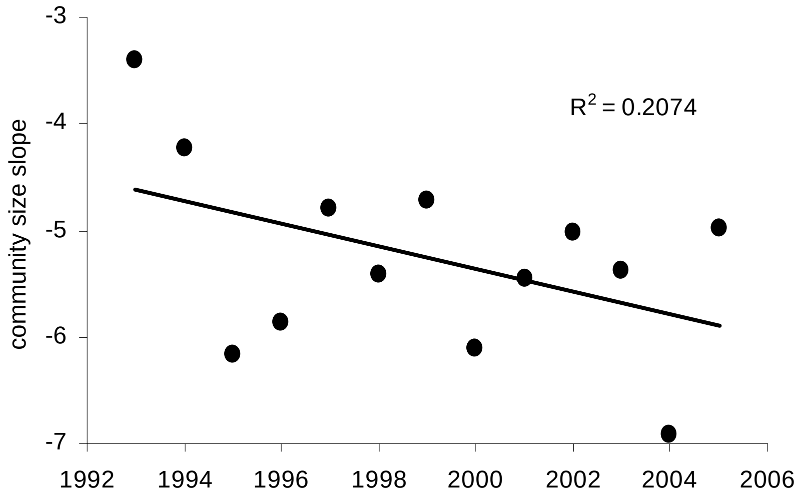


Figure 4.67. Size slopes of annual community size spectrum from West Estonian Archipelago 1993-2005 (fish lengths <15 cm excluded). Data from HELCOM COBRA database.

Size related metrics - low fishing pressure

In contrast to the open Baltic Sea and coastal areas in eastern Baltic, coastal fish communities in western Baltic are less subjected to high fishing pressure. These communities, which generally are dominated by species with freshwater origin, show a different development. Using slope of size spectra as an indicator of community fish size, Figure 4.68 indicate an increase in size of coastal fish the last 20 years (r^2=0.21, p=0.05, linear regression).

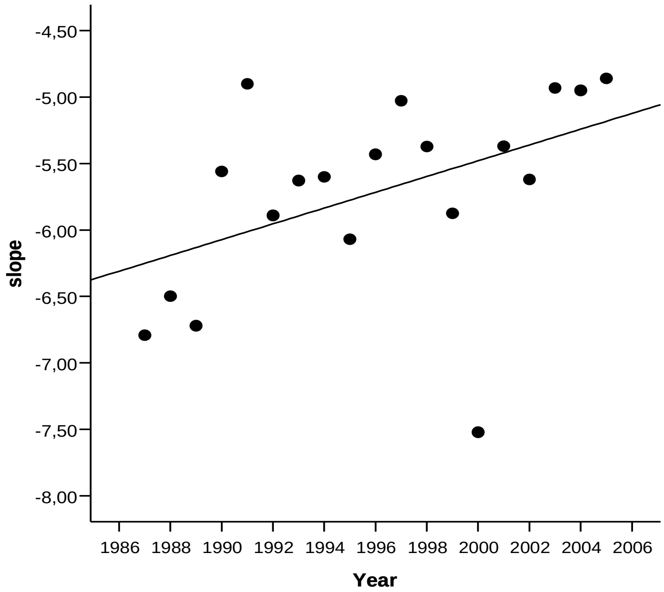


Figure 4.68. Size slopes of annual community size spectrum in a reference area in the archipelago in western Baltic Sea based on multi-mesh gillnet sampling. Data from Swedish Board Fisheries.

The observed increase in slope of the size spectra reflect increased abundance of two piscivorous fish species, zander and perch, and an increased abundance of the cyprinid species bream. These changes are also reflected as a significant increase in community mean trophic level (HELCOM 2006, Figure 4.69).

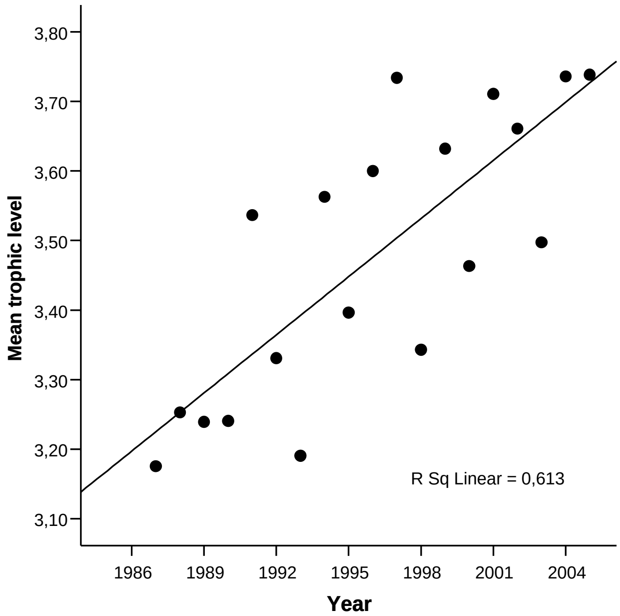


Figure 4.69. Changes in mean trophic level of the coastal fish community over time in a reference area in the archipelago of western Baltic Sea (HELCOM 2006).

Possibly environmental changes have had an impact on the development, as especially zander and bream have been gained by eutrophic conditions and high temperature during the1990s. This illustrate that other factors than fishing confounds the assumption on fishing being the driving force of population and community development in the Baltic Sea.

To be able to assess the outcome of suggested indicators we have to check the possible influence of other sources to the variation, especially the effects on environmental changes and perturbations. In many cases, as shown by the herring example, more complex interactions, such as indirect and cascading effects, have to be elucidated before selecting relevant indicators.

5 Potential suite of indicators

Based on current knowledge which is partly represented in the examples in section 4 and comprehension of the different ecosystems a suite of potential indicators is put forward that will be formally tested against the criteria described in section 1.1 to determine their suitability as part of an Ecosystem Approach to Fisheries Management.
The aim was to cover all relevant ecosystem features for the State indicators and matching Pressure indicators including less informative proxies in case the required information is not available. For this we introduce a hierarchy from very broad and general features (e.g. physical/chemical, fish or other ecosystem components) to more specific features (e.g. physical environment or abundance of commercial stocks) to the actual indicator (e.g. "Proportion of commercial stocks that are within safe biological limits"). In cases where no specific indicator has been developed for a particular ecosystem feature we used a more general phrasing (e.g. Abundance index of selected marine mammal species).

Physical/chemical

- Physical environment
    - Temperature
    - NAO
- Chemical environment
    - Salinity
    - Oxygen levels
    - N and P levels (Eutrophication)

Plankton

- Phytoplankton
    - Primary production
    - Water transparency
    - Chlorophyll a level
- Zooplankton
    - CPR derived plankton indicators
    - Zooplankton biomass

Fish

- Status of commercial stocks
    - Proportion of commercial stocks that are within safe biological limits
- Abundance of populations that are not regularly assessed
    - Abundance (numbers) index of selected species (e.g. elasmobranchs)
    - Biomass index of selected species (e.g. elasmobranchs)
    - Decline (threat) indicator
- Size/Age Structure of a fish species
    - Average length of selected species
    - Average weight of selected species
    - Average age of selected species
- Genetic composition of a fish species
    - Maturation norm
- Size structure of the fish community
    - Mean weight
    - Mean length
    - Proportion of large fish
- Species composition including biodiversity of the fish community
    - Mean maximum length
    - Biodiversity indicator (Hill's N0)

- Biodiversity indicator (Hill's N1)
- Biodiversity indicator (Hill's N2)
- Proportion of target species

- Abundance of the fish community
    - Total numbers
    - Total biomass

Other ecosystem components

- Status of marine mammals
    - Abundance index of selected marine mammal species
- Status of Seabirds
    - Abundance index of selected seabird species
- Status of marine reptiles
    - Abundance index of selected marine reptile species
- Status of benthos
    - Abundance index of sensitive benthic species
    - Epibenthos community indicator
    - Infauna community indicator
- Status of sensitive habitat
    - Area coverage of highly sensitive habitats

Ecosystem

- Ecosystem functioning including trophic level
    - Primary production required
    - Catch ratios
    - Mean transfer efficiency
    - Trophic level
    - Fishing in Balance index
    - Finn Cycling Index

Fishing pressure

- Fleet capacity per métier
- Fishing effort per métier and its spatial and temporal distribution
    - Days-at-sea or hours fished per spatial unit (e.g. ICES rectangle) per time (e.g. year or month)
- Fishing impact including catch, by-catch and habitat destruction
    - Fishing-induced proportion mortality of commercial fish species
    - Fishing-induced proportion mortality of non-assessed fish species
    - Fishing-induced proportion mortality of benthic species
    - Fishing-induced proportion mortality of marine mammals
    - Fishing-induced proportion mortality of vulnerable and/or protected species
    - Proportion of catch discarded
    - Proportion of area or (sensitive) habitat impacted

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