Chapter 16

# Research in stock assessment

## 16.1 From fish stock assessment to an ecosystem approach to fishery

*Spedicato M.T., Lembo G.*

Concern that human development can negatively impact natural renewable resources is remarkably growing in the last decades.

The need to balance present and future fishery yields, and to preserve the renewal capability of fishery resources so that future generations may enjoy the same benefits as their predecessors, has formed the basis of fisheries science since the 1950s and was explicitly examined in the decisive works by Schaefer, Beverton, Holt, Richer & Gordon. This need, originally expressed essentially in regard to target stocks, through the use of concepts such as maximum sustainable yield (MSY), maximum economic yield (MEY) and optimum yield (OY), is now extended to associated species and surrounding ecosystem. Achievement of a MSY for target species, is in turn merely a first step towards an ecosystem approach.

Article 31 of the Johannesburg Declaration on Sustainable Development (UN, 2002) indicated the need to maintain or restore stocks to a maximum sustainable yield (MSY) level, with the aim of reaching this target urgently for stocks unsustainably exploited , possibly no later than 2015.

The CFP reform objectives allocate the role of defining the limits within which the fishing industry can operate to public institutions. The European Marine Strategy Framework Directive (EMSFD, 2008) specifies that the CFP should take into account the environmental impact of fishery and its influence to the objective of a good environmental status (GES), which implies, among other things, ensuring that all fish populations are within safe biological limits. It is also essential to measure the progress made towards a GES using all available tools: conventional stock assessment methods, indicators, ecosystem models, etc.

Ecosystem-based fishery management (EBFM) is aimed at ensuring that activities are planned and developed in harmony with economic and social objectives, without jeopardising the options for future generations to benefit from the full range of goods and services provided by the marine ecosystem (FAO, 2003). Ecosystem-based fishery management requires managers to consider all the possible impacts of capture activities when defining their management objectives. To do this managers need the support of appropriate scientific analysis (Murawski, 2000; Browman & Stergiou, 2004). Integrating ecosystem considerations into the assessment processes a certain level of complexity that need to be addressed with various scientific approaches.

These can be divided into three main groups that should be considered as complementary rather than alternatives:

- an indicator-based approach;
- multi-species models and modelling of the effects of fishery on the ecosystem (trophodynamic models);
- modelling of the relationships between environmental variations and changes in the abundance and structure of populations and communities.

## Indicator approach

Indicators can contribute to decision-making processes and to orienting management in various ways (Garcia *et al.*, 2000; Rice, 2000; 2003; Rochet and Trenkel, 2003), particularly:

- by describing the state of the ecosystem, the pressure factors acting on the ecosystem and the effects of management on the ecosystem;
- by tracing and monitoring the evolution of the system towards the achievement of management objectives;
- by communicating complex impact trends and management aspects to a non-expert audience.

The ecosystem approach should therefore necessarily be based on indicators of state, indicators of the fishing impact on the ecosystem and indicators for the assessment of the economic and social consequences of management policies (Rice, 2003; Jennings, 2005; Piet *et al.*, 2008; Rochet and Trenkel, 2009). The importance of indicators for fisheries management has become increasingly clear, since they allow a holistic approach (e.g. FAO, 1999; Caddy, 2002; 2004; FAO, 2003; Garcia e Cochrane, 2005; Cotter *et al.*, 2009).

Various European reviews and projects focused on ecosystem indicators (e.g. INDECO, IMAGE), producing lists of indicators that vary in comprehensiveness, each providing specific information for answering targeted questions concerning ecological, social and economic sustainability. Numerous population, community and ecosystem indicators, which sometimes also include components on a spatial scale, were tested for sensitivity and responsiveness (Jennings *et al.*, 2001; Trenkel and Rochet 2003; Rochet and Trenkel 2003; Fulton *et al.*, 2005; Piet and Jennings; 2005; Shin *et al.*, 2005; Piet *et al.*, 2008).

Several methods were also proposed for classifying the various indicators and arranging them into logical frameworks that can identify cause-and-effect relationships. The most widely recognised and applied system, proposed by Garcia and Staples (2000) and by Garcia and collaborators (2000), is known as the Pressure-State-Response (PSR) framework.

The implications of the ecosystem approach are multidimensional, complex and not always sufficiently predictable, so that it is necessary to use several indicators while also avoiding redundancy, which could lead to contradictory signals that are difficult to interpret (Rice and Rochet, 2005). Another aspect concerns the specific time dimension for the various elements of the fisheries system, since this affects the period of validity (and therefore the reliability) of the indicator values. For example, the abundance of an anchovy stock will change more quickly than the behaviour and size of the fleet and should therefore be estimated every year. The various indicators should therefore be regularly updated at specific predetermined intervals. It is also important to take into account the significance of changes in the indicators and, in particular, to estimate whether they exceed the intrinsic uncertainty level.

The complexity of the indicator approach involves, specifically, the difficulty of distinguishing between anthropogenic effects, such as fishing, and other non-anthropogenic impacts, which can cause changes in the indicator (Trenkel & Rochet, 2003; Rochet & Trenkel, 2003; Jennings & Blanchard 2004) (figure 16.1).

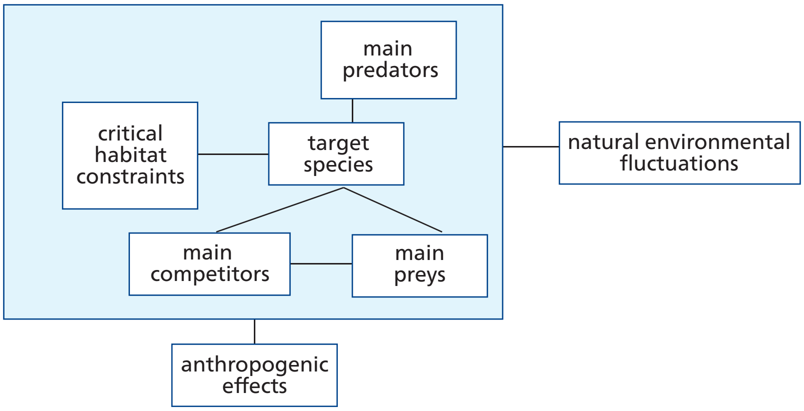


**Figure 16.1 - Key biological interaction factors (in the rectangle) that influence fishing and the stock management (intrinsic factors) and can be monitored with indicators, and some important external factors that can influence productivity (modified from Caddy, 2004).**

Furthermore, the definition of appropriate targets or limit reference points is the major obstacle in the conversion of indicators into decision criteria (Link, 2005).

In multi-species fisheries, an increase in fishing activity should produce a decline in the abundance of certain species, but production for each of these is assumed to be maximised at around half the biomass in the absence of fishing ($B_0$). The same behaviour is expected for the aggregated production of all exploitable resources. This behaviour could, however, be masked by phenomena which occur in the ecosystem, such as the replacement of species, a reduction in biodiversity and the consequent changes in the population structure and the mean trophic level. On the other hand, the total of all species in the community should be something more than the sum of the species in the community. The challenge is therefore to establish a level of fishing pressure, as in the case of maximum sustainable yield, that represents a compromise between maximisation of catches and an unacceptable reduction in biodiversity or trophic level. Each of the indicators can also show different levels of deterioration and therefore a method is required for establishing threshold values, for example, for diversity indices, slope of the size spectra, optimal trophic level, etc.
While there is not yet an operational framework of reference points for indicators, a wide range of studies have shown that the direction of trends in indicator time series may in fact accurately reflect the effects of fisheries (e.g. Jennings and Dulvy, 2005; Blanchard *et al.*, 2010).
In a recent work Rochet *et al.* (2010) attempted to combine the trends of various indicators using estimate methods based on the likelihood approach (Trenkel and Rochet, 2010), assuming that the exploited community was simplified into three levels, two upper levels (predators and omnivores) targeted by fishing fleets, and a third level represented by prey organisms subject

to environmental variations. The changes caused by variations in the fishing or the ecosystem, i.e. according to top-down or bottom-up perturbations, were forecasted through the use of Qualitative Modelling (figure 16.2).
The results of this work suggest the presence of coherent relationships between the most probable causes of change identified by independent information and the existence of compensatory mechanisms among the species within functional groups.

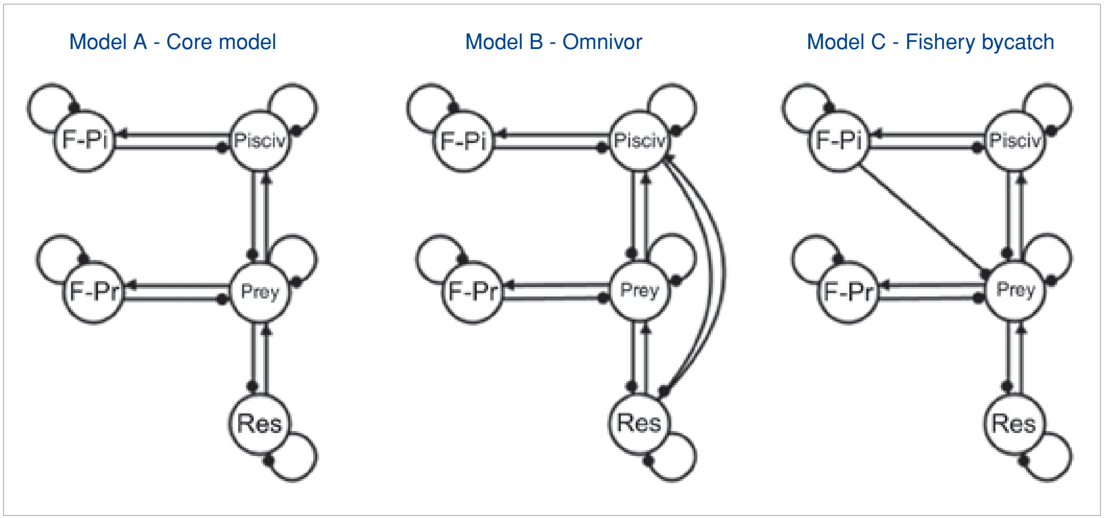


**Figure 16.2 - Examples of community models used to predict the effects of changes in fishing (of piscivores (F-Pi) or prey (F-Pr)) or in the productivity of the resources (Res) on the functional groups of prey and piscivores (the system variables are shown in the circles). The links represent direct effects; those ending with an arrow (or in full circles) represent a direct positive (or negative) effect, while links connecting a variable to itself indicate self-effects (modified from Rochet *et al.*, 2010).**

Another widely used approach that combines the various indicators in a single operational framework for assessing the state of resources and the ecosystem is the traffic light approach, introduced by Caddy and Surette (2005). This approach, also used in the Mediterranean Sea by Ceriola and collaborators (2008), provides a procedure for assigning the colours green, yellow and red within the cumulative distribution range of the time series of various indicators (figure 16.3).
This system also has certain limitations, due to the difficulty in identifying reference points that are independent from the time series and certain conceptual and operational problems for establishing causal relationships between impacts, the state of resources and the effects of the management measures. Regarding the formulation of a final diagnostic evaluation, various methodological options are possible, such as fuzzy logic approaches and expert systems or models based on the laws of Boolean algebra (Halliday *et al.*, 2001; Jarre *et al.*, 2008).

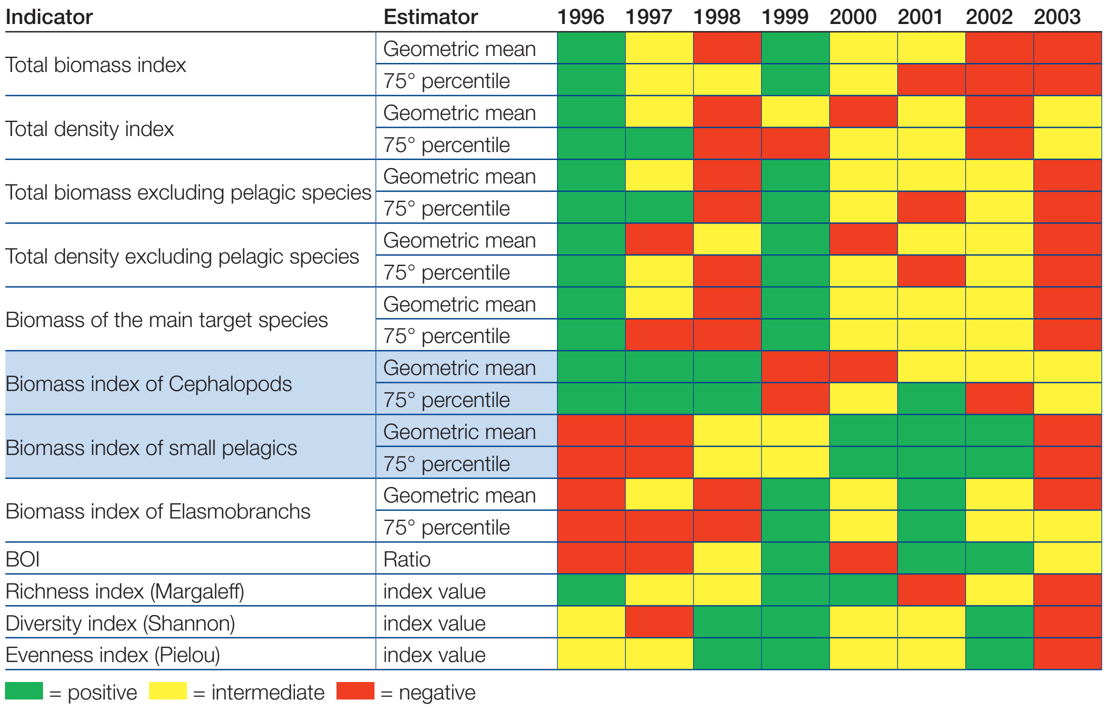


**Figure 16.3 - Illustrative diagram of a traffic-light table showing the response of community indicators in a times series (1996–2003) for the Southern Adriatic (modified from Ceriola *et al.*, 2008).**

A traffic light approach simplified for dissemination purposes is used in the Italian Marine Biology Society yearbook on the state of fishery resources (SIBM, 2010).
Given the multi-specific nature of the fisheries in the Mediterranean and Italian seas, the indicator approach has been in use for quite some time, thanks to the availability of time series from trawl surveys, such as the MEDITS campaign. This time series is now becoming long enough to allow more formal trend analysis on the community and ecosystem indicators..
The indicator approach is now an integral part of the Community framework for the collection, management and use of scientific data in order to implement the CFP (regulation (EC) 199/2008, and subsequent Commission Decisions Nos 949/2008 and 93/2010). A commented list of these indicators is given in table 16.1. Among these, for example, the proportion of large fish, or pLarge (indicator 2, table 16.1), would tend to decrease through the impact of fishing, due to a progressive reduction of apex predators caused by a "fishing down the food web" effect (Pauly *et al.*, 1998).

Table 16.1 - Data Collection Framework (DCF) indicators for measuring the effects of fisheries on the ecosystem (for details on the indicator assessment methods see COM(2008) 187 final-SEC/2008/0449 final).

| Indicator code | Indicator | Definition | Data needed | Indicator type |
|---|---|---|---|---|
| 1 | Conservation status of fish species | Indicator assessing and reporting trends in the biodiversity of vulnerable fish species | Species composition, demographic structure and abundance from fishery independent data (catches from scientific surveys) | State |
| 2 | Proportion of large fish | Indicator for the proportion of large fish in the assemblage by weight, reflecting the size structure and life history composition of the fish community | | State |
| 3 | Mean maximum length of fishes | Indicator for the life history composition of the fish community | | State |
| 4 | Maturation of exploited fish species | Indicator of the potential "genetic effects" of fishing on exploited populations | Individual ageing by length sex and maturity from fishery independent data (catches from scientific surveys) | State |
| 5 | Distribution of fishing activity | Indicator of the spatial extent of fishing activity. It would be reported in conjunction with the indicator for 'Aggregation of fishing activity' | Geographical position of fishing vessels based on Vessel Monitoring System (VMS) | Pressure |
| 6 | Aggregation of fishing activity | Indicator of the extent to which fishing activity is aggregated. It would be reported in conjunction with the indicator for 'Distribution of fishing activity' | | Pressure |
| 7 | Areas not impacted by mobile bottom gears | Indicator of the area of seabed that has not been impacted by mobile bottom fishing gears in the last year | | Pressure |
| 8 | Discarding rates of commercially exploited species | Indicator of the rate of discarding of commercially exploited species in relation to landings | Species composition and demographic structure of landings and discards | Pressure |
| 9 | Fuel efficiency of fish capture | Indicator of the relationship between fuel consumption and the value of landed catch | Value of the landings and fuel costs | Pressure |

# Multispecific models, trophodynamic models and habitat modelling

An extensive review of ecosystem models was made by Plagányi (2007), which should be referred to for more detailed information, while various European projects examined aspects concerning the development and use of multispecific models for an ecosystem approach, also including spatial components (e.g. GADGET, OSMOSE).

Ecosystem models should be conceptually viewed as complementary rather than as substitutes for single-species models, which continue to form the basis for assessment frameworks (Quinn & Collie, 2005).

If Lotka and Volterra can rightfully be considered as pioneers of the ecosystem approach, with the first formulation of a prey-predator model, the multispecies virtual population analysis (MSVPA; Pope, 1979) represented the first step towards ecosystem models in an organised scientific context such as the ICES.

Initially designed and formalised to overcome uncertainties in estimating natural mortality caused by the predation component, MSVPA became one of the first ecosystem models through which consideration of relationships within the food cycle entered implicitly or explicitly into the structure of the model (Magnússon, 1995). In fact, one result of MSVPA is the estimate of the quantities of the various species consumed by predators. MSVPA models need a high quantity of data on stomach contents, as well as a series of parameters also shared with normal Virtual Population Analysis. MSVPA models are a combination of a set of paired nonlinear equations, the Baranov catch equation and the equation for estimating natural mortality due to predation, to be resolved year by year (not by cohorts, as in standard VPA) in each step of the analysis. The basic assumption, not always corroborated, is that food ration and the prey preference are constant (i.e. independent from time for every species-age combination).

Also part of the multi-species approach, in 2006 Pope and co-authors proposed a simulation model based on size spectrum theory (the relationship between organisms size, measured as a logarithm of length, and their abundance, measured as a logarithm of the number of individuals of all species, by length class).

It was empirically observed that the slope of a size spectrum increases in response to an increase in fishing pressure. A dynamic version of this model was recently created by Andersen and Pedersen (2009) (figure 16.4) in order to include growth that is dependent on the availability of food. In the model, for example, the increase in the abundance of fish in a specific range can therefore be estimated as a consequence, not only of a reduced fishing pressure on a higher trophic level, but also of changes in the availability of food, which result in different growth in individuals.

The results of this model confirm that fishing can lead to an increase in both top-down and bottom-up "trophic cascade" effects, according to a combination of changes due to predation mortality or limitation of food. Furthermore, the difference in fishing patterns between ecosystems can affect the capacity to distinguish "trophic cascade" phenomena. For example, the more fishing acts on various trophic levels, the more difficult it is to identify its impacts in terms of "trophic cascade".

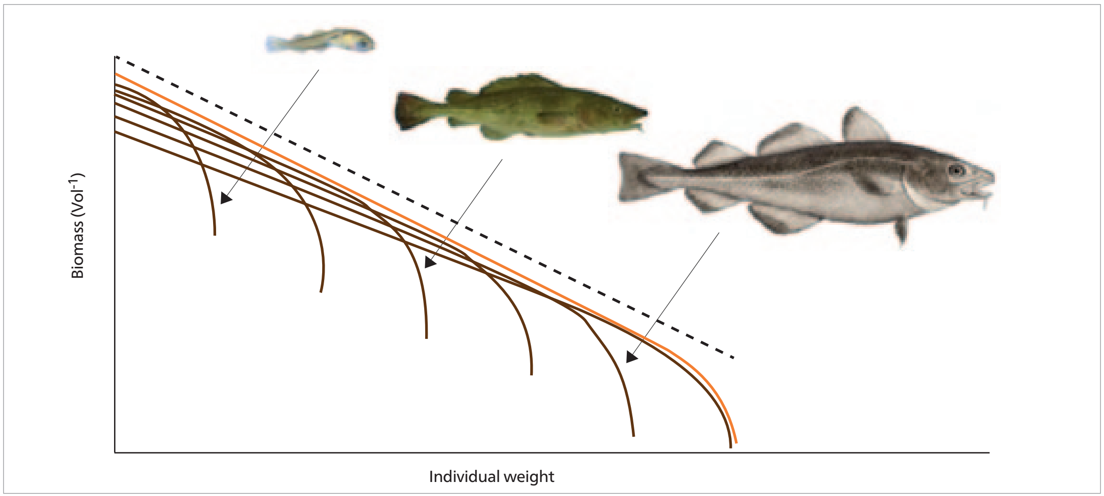


**Figure 16.4 - Representation of a dynamic model based on size spectra. The broken line represents the spectrum of carrying capacity, the brown lines are the spectra of the species groups with different asymptoric size and the orange line is the total size spectrum of the community.**

A multi-species approach that has recently become very popular is based on trophodynamic models, which are aimed at describing and predicting the functioning of ecosystems (Cury *et al.*, 2003). The trophic level (TL) has therefore become a key concept in many ecosystem models and is also a useful indicator for summarising the impact of fisheries (e.g. Pauly *et al.*, 1998). The most widely used model was the mass balance model (e.g. Polovina 1984; Pauly *et al.*, 2000), which is based on the allocation of biomass in discrete trophic groups. Ecopath/Ecosim (e.g. Christensen *et al.*, 2005) is also a software for building trophodynamic models (www.ecopath.org).

In the Ecopath/Ecosim approach, trophic interactions between functional groups of the ecosystem are described by a set of linear equations:

$$P_i = Y_i + B_i + M2_i + E_i + P_i \times (1\text{-}EE_i)$$

where $P_i$ is the total production of *i*; Y is the total catch of *i*; $B_i$ is the total biomass of group *i*; $E_i$ is the migration rate; $M2_i$ is the total predation rate for group *i*, and $EE_i$ is the ecotrophic efficiency of *i*, (the fraction of the production of *i* that is consumed within the system, exported or collected).

The model can also be expressed as:

$$B_i \times (P/B)_i \times EE_i - \Sigma B_j \times (Q/B)_j \times DC_{ji} - Y_i - E_i = 0$$

where $(P/B)_i$ is the relationship between production and biomass; $(Q/B)_j$ is the relationship between consumption and biomass; and $DC_{ji}$ is the fraction of prey *i* in the average diet of predator *j*. The dynamic part of the Ecosim model allows temporal analysis and fitting of the model to the time series.

Ecosystem overfishing is the condition that occurs when the composition per species and dominance are significantly altered by fishing, i.e. when the trophic level decreases (or when a reduction of large and long-lived species – predators – is noted, with a predominance of small, short-lived species that occupy lower trophic levels) (figure 16.5).

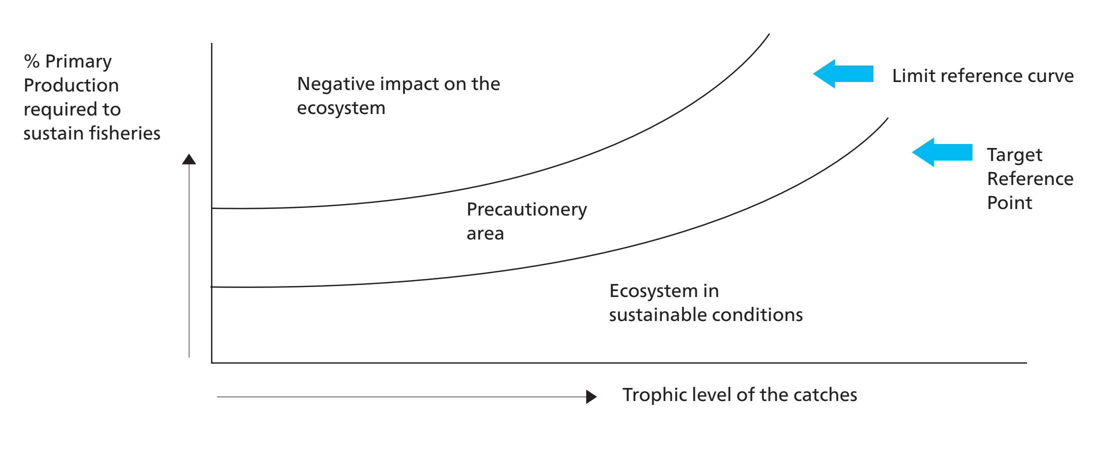


**Figure16.5 - Theoretic schema of the approach based on %PPR and TL indices to describe the state of overfishing at the ecosystem level (modified from Tudela *et al.*, 2005).**

According to Caddy (2004), the mean trophic level and the ratio of the biomass of pelagic species to that of demersal species both have conceptual disadvantages, since they can be indicators of a growth in input of nutrients as well as of overfishing at the same time.

A criticism made of trophodynamic models is that, although they are very useful for studying the functioning of the ecosystem, they do not provide a general theory regarding the impact of fisheries on the ecosystem, and furthermore, perhaps due to the large number of parameters, they are not considered very useful for making forecasts, particularly as far as management measures are concerned.

Recently, Gascuel (2005) proposed a model based on trophic levels in which biomass is distributed along a continuum and divided into classes. As a consequence of predation and onthogenetic processes, biomass moves from one class to that above it, changing its own trophic level. The model is therefore based on the flow of biomass and analysis of the catch trophic spectrum to estimate exploitation rates and forecast the effects of management.

Finally, habitat modelling, generalized linear models (GLMs), generalized additive models (GAMs) and generalized mixed models (GMMs) are recent techniques that are widely used for combining the use of environmental variables, such as North Atlantic Oscillation (NAO), with historical series of indicators and to explain ecological responses to environmental changes.

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# 16.2 Conventional stock assessment methods

*Fiorentino F., Lembo G.*

The classic fisheries science approach views the effect of fishing on fish stocks as causing a reduction in abundance and changes to population structures. With an increase in fishing pressure, a decreasing number of individuals reaches an advanced age and the juvenile class becomes the predominant portion of the stock. Estimates of the abundance and population structure of stocks exploited by commercial fisheries and the study of their productivity under various exploitation scenarios is the main objective of the ecological discipline known as assessment of fishery resources.
The assessment of fishery resources is a complex process which requires the collection of data on population and fishing effort and subsequent processing with analysis techniques that rely on the statistical properties of the data series and/or mathematical population dynamic models.
The population features normally examined in the assessment of resources are abundance and demography. Although space is implicitly involved in stock assessment processes, only in recent times have aspects related to the spatial distribution of populations been explicitly considered in assessment processes.

Regarding the origin of the data, the methods for collecting useful information for assessment are divided into indirect and direct methods. The first involve data obtained from commercial fisheries (fishery-dependent data), which consists of statistics on catches and the corresponding effort or the population structure of catches. The second are based on data collected during scientific surveys and provide estimates, independent from commercial fisheries (fishery-independent data), on the abundance and demography of the resources in the sea.
The information collected with both methods is generally used in assessments made with mathematical models, which in turn are divided into three main groups: global or surplus production models, analytical or structural models, and recruitment models.
Other approaches, which are often viewed as complementary, as well as alternatives, to these methods and can be used as a basis for assessments, are those based on indicators. According to Cotter *et al.* (2009), these provide a complementary method of managing stocks, as well as an alternative to stock assessment methods. Underlying the development of these methods, which has been widely used in Italy in recent times (SAMED, 2002; SIBM, 2010), are uncertainties regarding data on catches and discards from commercial fisheries and the delay in the availability of this data compared to that obtained from scientific surveys (Mesnil *et al.*, 2009).
Another group of methods used in procedures for assessing the state of resources and management scenarios is represented by simulation models, a widely used technique that integrates monitoring and assessment tools (CNR, 1998).

## Monitoring the abundance and population structure of resources

### Indirect methods

Data on catches and effort forms the main contribution that the monitoring of commercial fisheries can provide to the assessment of the state of stocks. Catch per unit effort (CPUE), expressed as the biomass captured for each unit of effort applied to stock, is the most traditional index of fish stock abundance (Hilborn & Walters, 1992). The information on catches and effort is usually obtained by interviews with fishermen on landing or from fishing logbooks. The catch per unit effort can be obtained either by sampling the total catch, and the corresponding total effort, when the fleet vessels have landed, or through scientific observers on board fishing boats, who directly record the data on catches, discard and effort.
A classic presentation of the methodological aspects of recording catch and effort data is provided in Gulland (1983). Stamatopoulos (2002) has recently addressed sample-based surveys, reporting various combinations of census approaches with sampling organised in terms of space and time.
Although CPUEs are widely used in the assessment of fishery resources, their effectiveness in estimating true stock abundance can be invalidated by two different sources of error:

- fishing is concentrated where there is a greater density of resources and therefore CPUEs provide an optimistic estimate of abundances;
- the catchability coefficient of fishing boats varies over time, due to the effect of technological creep, altering the relationship between catches and abundance in the sea.

Standardisations have to be adopted in cases in which catch and effort data are taken from different fishing gear or from historical series in which the catchability coefficients of the gear have

not remained reasonably constant (Hilborn & Walters 1992; Bishop 2006) through the use, for example, of Generalised Linear Models (GLMs) or Generalised Additive Models (GAMs).
Despite these disadvantages, indirect methods nevertheless have the advantage of allowing a large quantity of information to be obtained on the abundance of the stocks fished throughout the year at relatively low costs.
Catch surveys are also carried out in order to examine the population structure (size, age and sex) and biological characteristics (maturity and fertility). In general, size structure is surveyed by sampling with a random stratified design, while age, sex, maturity and fertility are sampled using a two-stage design (Cadima *et al.*, 2005). A clear guide for estimating the length structure of commercial fleet landings, stratified by fleet, fishing area, port and period, is provided in Sparre (2000). To estimate the age of the organisms, sub-samples can be taken from the individuals sampled for length structure to estimate the proportion of individuals of different age groups for each length class. This procedure is used to prepare age-length keys (ALKs). Individuals are generally extracted in accordance with a stratified design for sizes, so that the sub-sample systematically covers all the sizes present in the sample.
ALKs are usually prepared each year in order to account for the interannual variability in recruitment numbers, growth rates and mortality.
A report on the methodological aspects connected with surveying the length and age structures of commercial landings in Italy is given in SIBM (2005).

## Direct methods

Direct methods allow information to be collected on the abundance, demography and spatial distribution of fishery resources through experimental surveys conducted from research vessels, specially equipped fishing vessels or by underwater visual surveys. These surveys can include sample collecting (removing methods) or taking field measurements that do not involve the capture of organisms (non-removing methods). The surveys are known as trawl and beam trawl surveys, echo surveys, ichthyoplankton surveys, hydraulic dredge surveys and underwater visual surveys, according to the type of resources and the survey equipment used.
Direct methods show a high consistency over time of the sampling tool, the sampling design and the data-processing protocol. These characteristics lead to the opinion that the variations observed in the abundance, population structure and distribution of resources correspond more accurately to actual changes in the population and are less influenced by effects connected with changes in the behaviour of fishermen, as in the case of indirect methods.
The reliability of the survey data obviously depends on the suitability of the method for the individual species or life stage investigated and on the statistical design used. Considering the different habitat preferences of marine organisms during their various life stages, it is difficult to pinpoint a technique that can provide information on the entire population surveyed and the pattern that emerges from the survey can be seen to be influenced by the method used. Details regarding the planning and execution of the scientific surveys and the collection, processing and elaboration of the data are provided in Gunderson (1993) and Sparre and Venema (1998).
The difficulties in obtaining reliable catch and effort data, due to the characteristics of Italian fishery operations (high fragmentation of landing sites along the coast, the use of a large variety of fishing equipment, the high number of species landed, often divided into commercial categories of a variety of species, such as “*zuppa di pesce*”, fish fry, etc.), have led to a great development in the use of direct methods in the Italian fishery research. The use of trawl surveys for assessing

demersal resources dates, in fact, to the early 1980s, whereas echo surveys began to be used in the mid-1970s.
The Italian national programme for the collection of fishery data, prepared in accordance with Regulation (EC) 1543/2000 and subsequent amendments and modifications, currently involves two scientific surveys: the MEDITS trawl survey, with the objective of gathering information on the state of the main demersal stocks, and the MEDIAS echo survey, to assess the abundance of small pelagic stocks.
In general, survey data has the great advantage of being geo-referenced. This allows fairly precise snapshots to be obtained regarding the spatial distribution of stocks.
The surveys are also widely used to provide information on the biological parameters of the stocks and they even allow an entire series of information to be gathered on non-commercial species and the environment, which is useful for introducing ecological aspects into the assessment, as part of an Ecosystem Approach to Fisheries Management (EAFM).

# Assessment of the state of stock exploitation

## Indicator approach

Use of the indicator approach has been increasing in fisheries management over the last decade, both in the Mediterranean area and elsewhere, probably due to the difficulty of carrying out conventional stock assessment over a wide number of species, even in areas with abundant data, such as that of ICES (Cotter *et al.*, 2009). Furthermore, awareness of the complexity of the assessment systems and the intrinsic uncertainty of the estimation processes has led to a reconsideration of a holistic approach that can combine ecological as well as impact, economic and social aspects in a single assessment framework (e.g. FAO, 1999). Numerous European reviews and projects (e.g. INDECO, IMAGE) have shown the usefulness of population indicators for assessment purposes (Trenkel *et al.*, 2007; Cotter, 2009).
Selection of the indicators should be guided by criteria of economy (avoiding redundancy), the capacity to effectively contain the desired information, correspondence to scientific evidence, comprehensibility and conciseness (Rice & Rochet, 2005).
The selected indicators are generally combined in an interpretative framework that evaluates, for example, changes in temporal tendencies (Trenkel *et al.*, 2007), or is analysed using a traffic light approach (Caddy, 2002), in order to make interpretation of the present condition, and the direction in which the indicator would move with the implementation of specific management measures, easily accessible.
The SAMED project (2002) was the first attempt, on a Mediterranean scale, to use the considerable information potential contained in trawl survey data, producing not only abundance indices but also population parameters (growth, maturity and mortality) for a large pool of demersal resources, through the use of common and standardised protocols. These estimates were then used for assessment purposes through the exploitation rate indicator (E=F/Z) or simple reference points based on total mortality levels compared to natural mortality rates. The approach was completed by an estimate of population parameters per cohort and by an initial simulation model, which operated in conditions of pseudo-equilibrium, based on the conceptual model of Thompson & Bell (1934), while the use of multispecies size spectra allowed certain elements of the ecosystem approach to be included in the analysis.

SURBA (*SURvey Based Assessment*; Beare *et al.*, 2005) is a method that integrates the indicator approach with models that analyse stock population structures obtained from scientific surveys. Through the use of abundance indices for age classes and population parameters (weight, maturity and natural mortality by age), it is possible to model cohort dynamics and estimate the trend of recruitment, spawning stock biomass (SSB) and mortality rates (Z) over time. This method has been widely used by Italian researchers within the European Commission SGMED-STECF working groups.

## Global or production models and their application to Italian resources

Global or production models, known in English literature as biomass dynamics models, are the simplest models available to fishery biologists for assessing the state of resources. They are used whenever it is not possible to know the age structure of catches or when the only available data is that for catches and effort.

These models treat the stock as a single biomass entity, regardless of the population structure, and estimate its production in relation to variations in fishing effort.

Although they do not explicitly contain recruitment, the entrance of new recruits into the stock is included, together with individual growth and natural mortality, in the instantaneous intrinsic growth rate of the population.

The theoretical formulation of these models has been revised by many authors, including Ricker (1975), Gulland (1983), Hillborn & Walters (1992) and Jennings *et al.* (2001).

The basic underlying concept of global models is the estimate of maximum sustainable yield (MSY), i.e. the highest catch obtainable from a given stock over a long period without compromising its renewal capacity.

Supposing an absence of emigration and immigration in the exploited stock, the increase in biomass due to recruitment and growth minus the loss due to natural mortality, known as “surplus production”, should correspond to catch quantities in order to maintain relatively constant biomass levels over time.

The best-known global model formulation is that of Schaefer (1954), who used the classic logistic model of population biomass dynamics, from which he removed the quota due to capture. The model can therefore be represented by the equation:

$$dB / dt = rB\,(1 - B / k) - C$$

where $B$ is the biomass of the stock, $r$ is the instantaneous rate of intrinsic growth, $k$ the carrying capacity of the population and $C$ capture in weight. $C$ is assumed to be proportional to the fishing effort ($f$) and to the biomass of the stock ($B$) through:

$$C = qfB$$

with $q$ being the catchability coefficient.

The term $rB\,(1 - B/k)$ represents surplus production: the relationship between surplus production and biomass is symmetrical, with a surplus production of zero both for low and high biomass values.

All the methods for fitting biomass dynamic models to data are based on the assumption that there is a relationship between the abundance of the resources and their indices (e.g. CPUEs). There are two fundamental approaches to estimating the parameters of global models. Methods that assume a steady state and those that work outside of assumptions of equilibrium. Clear and important reviews of the topic have been provided by Hilborn & Walters (1992) and Jennings *et al.* (2001).

## “Steady state” approaches

Steady state approaches were the first to be developed and are more widely used in the assessment of resources. They generally assume a linear relationship between fishing effort and CPUEs. In the equilibrium approach, it is assumed that the catch for each year is equal to the “surplus production” corresponding to the specific fishing effort exerted in that year (Hilborn and Walters 1992). This very stringent assumption is not very realistic, since CPUEs are rarely found to vary instantly and exclusively in relation to the fishing effort exerted.

An approach to estimating the parameters of the model which attempts to avoid this problem is that applied by Levi and Andreoli (1989) to the “aggregate” (all species together) of demersal resources in the Strait of Sicily, in which catches in a given year were related to the average effort exerted in the same year and in the two previous years. The input used in the work comprised paired data on the capture of demersal fish and the fishing capacity of the fleet from 1959 to 1983. The model provided a MSY estimate of around 70,000 tonnes, obtainable with an effort of 56,000 GRT (figure 16.6). The analysis also shows that production in the early 1980s was already beyond sustainable levels by about 10,000 tonnes, with an effort exceeding that compatible with the long-term renewal of resources by around 17%.

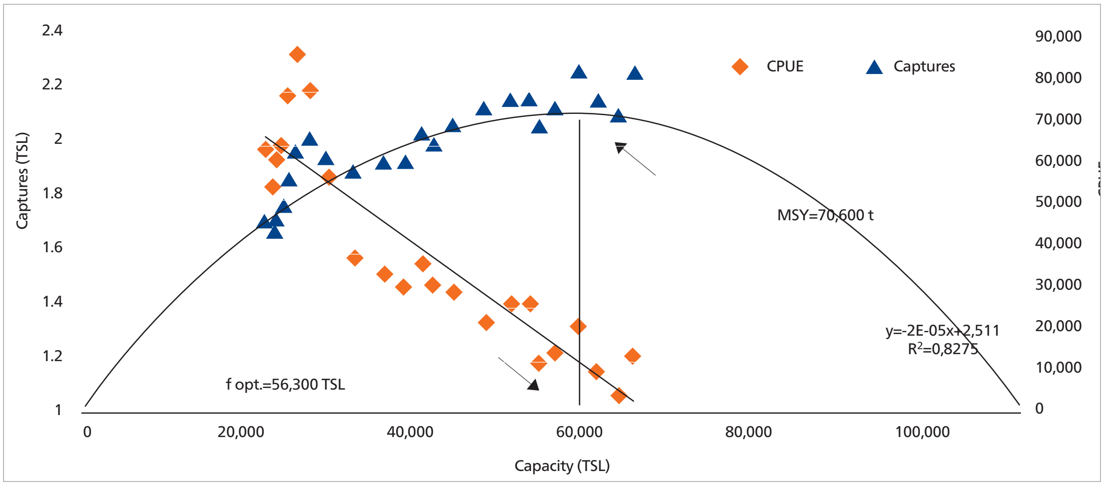


**Figure 16.6 - Surplus production model for the aggregate of demersal species in relation to the trawling fleet capacity in the Strait of Sicily for the years 1959-1983 (modified from Levi & Andreoli, 1989).**

## Out-of-steady-state approaches

There are various methods for estimating the parameters of global models outside of the steady state. The more effective ones include methods in which it is assumed that the relationship between production and effort and the difference between the model estimates and observed values are due exclusively to errors in abundance measurements in relation to actual stock sizes. The estimate of the parameters therefore begins with an estimate of the stock biomass at the start of the available time series using the model that estimates the biomass for the entire time interval in question. The catches or the observed stock biomasses are then compared with expected figures using statistical methods to minimise the difference between the expected and observed values (Hilborn & Walters, 1992; Quinn & Deriso; 1999).

With the Schaefer model the *r* and *k* parameters can be estimated from the time series of catch and fishing effort, making some assumptions on the catchability coefficient (Hilborn & Walters, 1992). The model can therefore be written as:

$$B_{t+1} = B_t + rB_t(1-(B_t / k)) - qfB_t$$

with the volumes and parameters already previously defined.

Abella *et al.* (2010) have recently modelled the trawl fishing production along the Tuscany coast using a multi-species surplus production model that accounts for the productivity of the aggregate of the eight main demersal species in relation to the fishing effort in trawling hours. The MSY of 1,330 tonnes per year for all the fished species is obtained for a fishing effort of around 49,000 hours, compared to a current production in 2008 of around 1,170 tonnes and a fishing effort of about 61,000 hours (figure 16.7). To move the exploitation on all the resources to conditions of higher sustainability (MSY), the fishing effort along the Viareggio coastline should be around 75% of that exerted in 2008.

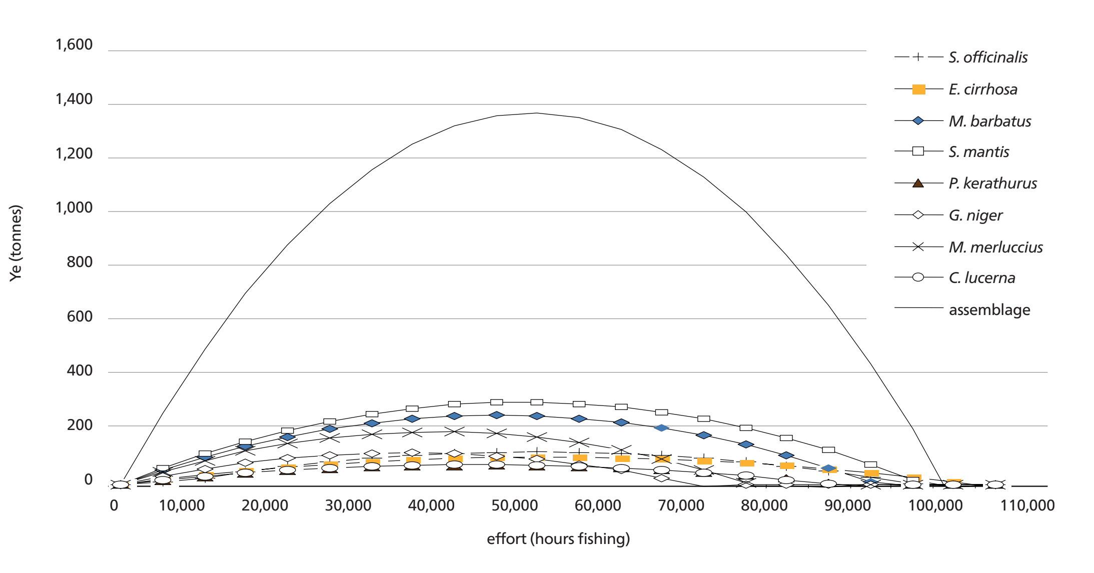


**Figure 16.7 - Surplus production model for aggregate production of the commercial species caught by trawling along the Viareggio coast. The MSY is obtained by a fishing effort of around 49,000 hours whereas, the effort during 2008 was around 61,000 hours (Source: Abella *et al.*, 2010).**

In the Italian situation, in which extensive historical series of biomass, catch and effort data are not available, but with time series of instantaneous total mortality rate (Z) estimates and biomass indices from scientific surveys, the following variation of the surplus production model, proposed by Abella (2007), may be used:

$$B_{t+1} = B_t + rB_t(1-(B_t / k)) – (F/Z)\ B_t(1-exp(-Z_t)\ )$$

in which the term *qfBt* = *C*, catch in weight (*Ct*) is replaced by Baranov's classic capture equation:

$$C=(F/Z)\ B(1-exp(-Z_t)$$

where Z can be directly estimated by analysing the population structures gathered with the scientific surveys and F, the instantaneous fishing mortality rate, which can be estimated by subtraction, as estimates of M, the instantaneous natural mortality rate, are available.

## Analytical or structural models and recruitment models

Analytical or structural models, known in English literature as dynamic pool models, and recruitment models are the most advanced tools in fisheries science and, although they vary greatly, are based on cohort dynamic analysis (Thompson & Bell, 1934; Beverton & Holt, 1957; Ricker, 1975). This involves describing the evolution in the number and the corresponding biomass of a group of individuals of the stock, known as a cohort, that are assumed to have been born at the same time (figure 16.8). This evolution is subject to the combined action of the numerical reduction starting from birth, due to natural mortality (M) and fishing (F), and the increase in weight of the survivors, due to growth.

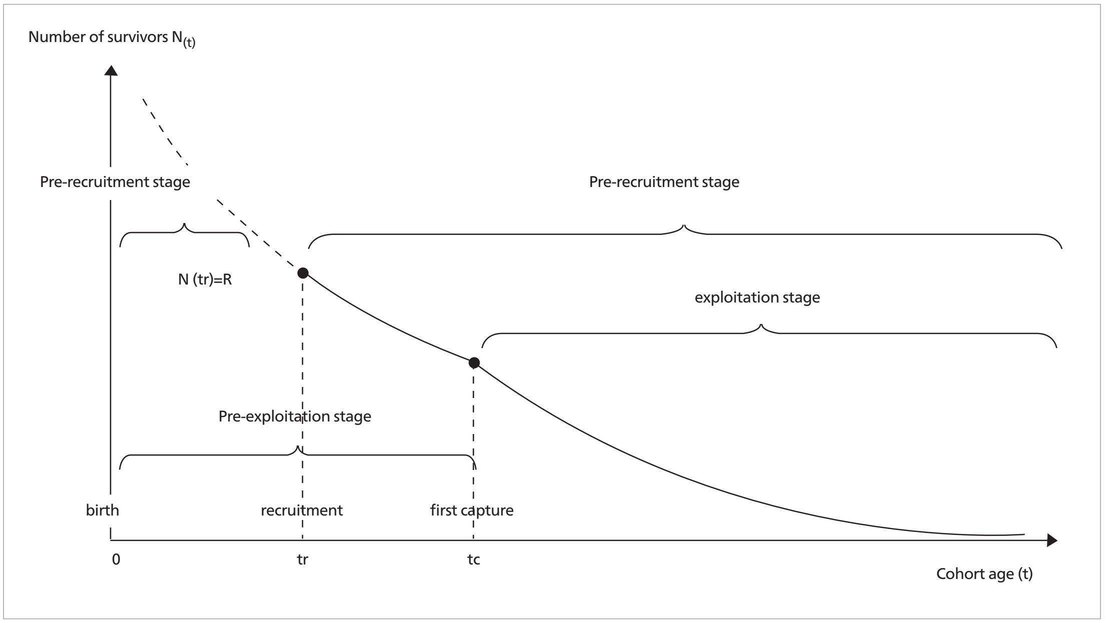


**Figure 16.8 - The main steps in the life of a cohort subject to fishing. The curves that illustrate the numerical decrease of the individuals in the cohort follow the classic exponential decay model.**

In order to avoid the uncertainties related to the variability of recruitment, these models classically express the yield (Y) and biomass (B) of the stock in terms of indices per recruit (Y/R; B/R) in relation to fishing mortality (F) and age/length at capture (tc/lc).

In Beverton and Holt's version (1957), which was widely used until the 1980s, production per recruit is obtained through the analytical solution of an integral in the interval of productive life of the cohort. Calculation of the Y/R and B/R values requires the Von Bertallanfy growth curve parameters, the length/weight relationship, the age or length at first capture, the constant instantaneous natural mortality rate and the age or length at first sexual maturity (figure 16.9).

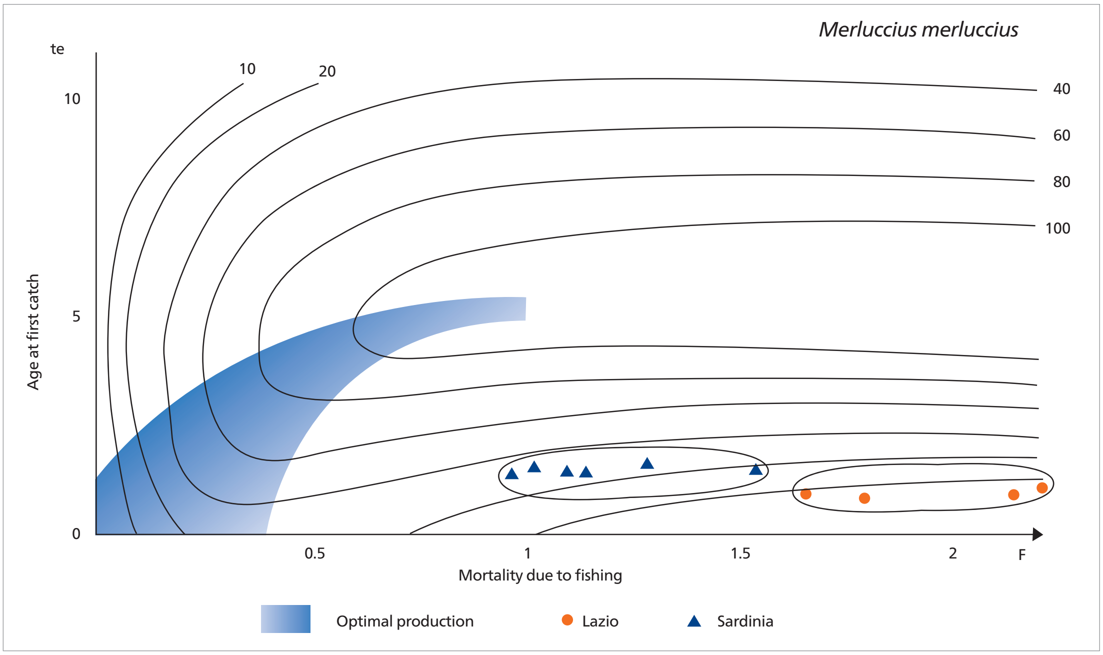


**Figure 16.9 - Assessment of the state of hake exploitation in the Central Tyrrhenian using the Beverton & Holt model (Source: Ardizzone & Cau, 1990)**

One of the main limits to the use of the classic version of analytical models are the stringent assumptions on recruitment, which is considered constant and continuous over time, and on mortality rates, both natural and from fishing, considered constant over time and throughout the various age/length classes.

The great development in numerical calculation, due to the availability of increasingly fast and powerful calculators, has led to a return to prominence, since the early 1990s, of the classic analytical approach of Thompson & Bell (1934), which had fallen into disuse because the solution of the production per recruit equation required numerous and repetitive calculations.

According to the Thompson and Bell approach, the life of the cohort is divided into intervals $\Delta t$ in which population parameters can be reasonably considered as constant. The following are calculated for each $\Delta t$ interval:

- The number of effectives in the cohort at the start of $\Delta t$ as N $(t+\Delta t\ t) = Nt\ exp\ (-Z\Delta t)$;
- The number of "total" deaths in $\Delta t$ as $Nt - N\ (t+\Delta t)$;
- The number of captured in $\Delta t$ as $C = (Nt - N\ (t+\Delta t)\ )\ F/Zt$;
- The yield in $\Delta t$ as $Yt = Ct\ Wt$;
- The biomass in $\Delta t$ as $Bt = Yt/Ft$;
- The renevue in $\Delta t$ as $It = Yt\ vt$;
- The yield, the biomass and the revenue of the entire life of the cohort as the sum of the values obtained in the individual $\Delta ts$.

The approach by time intervals, which can be easily implemented on electronic spreadsheets, has the great advantage of allowing the assumption of constancy of the population rates during the life of the cohort to be overcome, as this represents a limit for all those resources that are

already fished from an early age. The number of recruits considered at the start of the calculation procedure can be nominal or taken from the evaluations of stock abundance at sea from surveys or Sequential Population Analyses (VPAs and connected methods).
In general, analytical models allow the simulation of biomass trends, both for the entire stock (B/R) and for the portion of spawners (SSB/R) and of yield per recruit (Y/R) in relation to variation in fishing mortality and at the age/length at first capture, providing a useful medium to long-term management tool for the fishery activity.
While biomass per recruit always decreases when fishing mortality increases, the yield per recruit curves can show peaks or tend towards a more or less constant value with the increase in fishing mortality.
The point that corresponds to the peak of the Y/R curve is indicated as $F_{max}$ and provides an important biological reference point (BRP), being used as a biological limit to indicate situations of overfishing (Limit Reference Point). In order to avoid uncertainties in the identification of overfishing, in the case of "flat" Y/R curves, Gulland & Borema (1973) proposed a further BRP indicator with $F_{0.1}$ and defined as the point at which the value of the tangent to the production per recruit curve equals 10% of that at the origin (marginal yield).
It was then demonstrated that this BRP, which came from essentially economic considerations, corresponds to the point at which the SSB is around 20% of the virgin one in those resources for which age/length at first capture tends to coincide with that of first reproduction.
If the analytical models described above allow assessment of indices of productivity and size of stocks with variations of exploitation, one of the key points in the assessment process is to identify, how fishing mortality is distributed among the various age/length classes (exploitation pattern) and how it stands in relation to the reproductive potential of the stock.
There are numerous methods for estimating mortality rates related to the fishing activity and the selectivity and/or exploitation patterns used for a stock.
There are two different approaches for estimating F: equilibrium methods, which estimate an F value that represents an average for various years and for various age/size groups, and methods capable of estimating different F values for various years and age groups. A recent review of the problem is provided in Hoggart *et al.* (2006).
In situations where assessment is limited to the current fishing pressure on adults in a given stock compared to optimal long-term pressure, the first group of methods is capable of providing management indications for the medium to long-term period. Among these should be mentioned Baranov's classic age-structured catch curve and the length-converted catch curve proposed by Pauly (1984).
The second group of methods, which allows the assessment of fishing mortality and the exploitation pattern for each year, is more useful in management contexts in which annual capture quotas need to be estimated. The best known of these is virtual population analysis (VPA), the most famous of an entire series of methods, also known as "sequential population analyses". Among these should be mentioned extended survivor analysis (XSA), integrated catch analysis (ICA) and catch-age analysis (CAGEAN).

The development of VPA has a long history and an in-depth review of the methods based on analysis of the age structure of catches is provided in Megrey (1989). In regard to Italian seas, the virtual population analysis approach has been applied to small pelagics fish stocks in the Adriatic Sea (Santojanni *et al.*, 2005). An approach which lies halfway between steady state methods and out-of-steady state methods is length-based cohort analysis or length VPA. This involves applying algorithms of the VPA or its approximation, known as cohort analyses, to length structures converted into age structures by means of age-length relationships (VBGF). It is assumed that the population composition found in a given year respects the evolution of a cohort during its lifetime (pseudo-cohort).

This approach, developed on VIT software (Lleonart & Salat, 1992), allows estimates of abundances according to length or age classes, estimates of the corresponding fishing mortality vector, simulation of variations in productivity and in stock abundance with variations in exploitation patterns and, finally, calculation of BRPs ($F_{max}$ e $F_{0.1}$) to assess the state of exploitation. Because it does not require long series of data, the VIT package has enjoyed great success in the Mediterranean and it is still used in the GFCM SAC workgroups and those of the EC STECF (SGMED).

Yield and biomass curves per recruit nevertheless have the great disadvantage of not including recruitment dynamics in the analysis, which is one of the key factors in the renewability of stocks and the sustainability of fisheries.

The study of the relationship between the abundance of spawners and the success of recruitment is another classic topic of fisheries science. There are numerous population dynamic models that describe "spawning stock-recruitment" relationships (SSR-R). The most prominent ones include that of Ricker (1954) and that of Beverton and Holt (1957).

The general properties of SSR-Rs can be summarised as follows:

- the curves pass through the origin, therefore in the absence of spawners there will be no recruits;
- the recruitment rate (R/A) decreases as the density of spawners increases;
- the curve never falls on the x-axis for high densities of spawners, therefore reproduction is not completely eliminated at high densities.

The study of the relationships between adult stock and recruits is one of the most complex points in the dynamics of exploited resources, as there are numerous factors, related to both the nature of the phenomena and to the data collection, which can obscure the existence of these relationships. Despite these difficulties, knowledge of these relationships is of major importance and allows the theme of the sustainability of exploitation patterns in terms of stock renewal capacities to be introduced.

The first example of the relationship between adults and recruits in Italian seas was produced by Zamboni *et al.* (2000), who studied the renewal capacity of red mullet stock in the Ligurian Sea.

Although the abundance of the parent stock is known to have a relevant role in stock renewal processes, recruitment success is also attributed to environmental factors and to the population characteristics (age/size) of spawners (Chambers & Trippel, 1997).

On the basis of these considerations, Levi *et al.* (2003) studied the SSR-R of red mullet in the Strait of Sicily, taking into account the effects of surface temperature anomalies. The results have shown that with an equal abundance of spawners, the number of recruits is greater in those years in which the temperature of the surface waters is higher than average.

SSR-Rs can be combined with yield and SSB per recruit curves to estimate the sustainable production of a stock, using the classic replacement lines procedure (Sissenwine & Shepherd, 1987; Quinn & Deriso, 1999). With spawner abundance estimates for various fishing mortality values (F), it is in fact possible to associate a recruitment value (R) with each F value and, knowing Y/R and B/R, to simulate the corresponding yield and biomass value for each R.

Together with approaches based on formal models, there are also empirical approaches, which begin with an analysis of times series of paired R and SSB indices, identifying threshold values to be used as BRPs. Given that the inverted recruitment rate (R/A, being R the recruits and A the adults) is equal to the survival rate (A/R), it is possible to order the A/R values and identify some recruitment rate threshold values (10th percentile, 50th percentile and 90th percentile) to which total mortality threshold values were associated (Z=-ln A/R). The procedure, developed by Zamboni *et al.* (2000) similar to what was proposed within the ICES with F values, has been recovered by Abella *et al.* (2005).

The use of information from the study of the empirical distribution of the historical data series A vs. R, or the "Spawning Stock–Recruitment" relationship, to assess the status of fished stocks is summarised in figure 16.10. Sector 1 identifies conditions of significant risk of stock collapse, whereas sector 4 indicates conditions of low productivity. Exploitation conditions that are compatible with the long term renewability of the stocks are associated to total mortality values close to target values of Z.

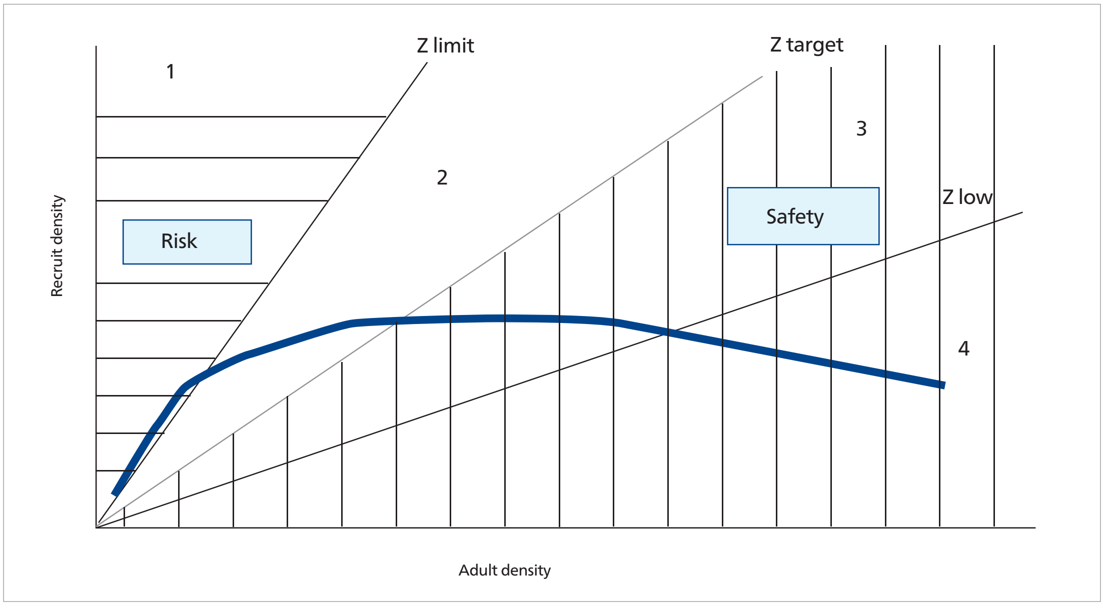


**Figure 16.10 - The use of values for Adults (A) and Recruits (R) and of corresponding total mortality values (Z), for the assessment and management of stocks. The Z reference values (replacement lines) divide the plane into 4 sectors characterised by: 1) low density of reproducers and low recruitment; 2) low density of reproducers and high recruitment; 3) high density of reproducers and high recruitment; 4) high density of reproducers and low recruitment.**

## Simulation methods

In stock assessment procedures, the integration of structural models with simulation models has assumed an increasingly important role in the assessment of management strategies with multiple objectives. In this context, operating models (OM) and management procedures (MP), which represent the central components of the evaluation systems (Butterworth, 2007), are arranged into assessment models (Methot, 2000) and simulation platforms (cf. Fisheries Library in R FLR; Kell *et al.*, 2007), at times with a focus on bioeconomic models (Ulrich *et al.*, 2007).

Simulation models are also used to verify the quality of population parameters and the efficiency of the evaluation models used, to project, under particular assumptions, the state of the stock in the future, to estimate the performance of the indicators in relation to various fishing scenarios, and to evaluate the consequences of various stock exploitation patterns.

ALADYM (Age-Length Based Dynamic Model) is a simulation model that belongs to the dynamic pool model group. It was developed as part of the European FISBOAT project (Fisheries Independent Survey Based Operational Assessment Tools; Petitgas *et al.*, 2009) and applied to the analysis of various stocks, both in the Mediterranean and elsewhere. ALADYM has been used in particular to forecast the effects of various management measures in the Italian fishery Management Plans prepared by MiPAAF in 2008. The model simulates the population dynamics of an individual stock, tracing at the same time the evolution of the various cohorts on a fine time scale (monthly) and accounting for uncertainty in growth, maturity and recruitment. The model can simulate various scenarios in terms of exploitation patterns and management measures. Uncertainty can be incorporated through a Monte Carlo-type approach (Lembo *et al.*, 2009; Spedicato *et al.*, 2010).

Another tool to support short and long-term social and economic assessments through a simulation approach is the BIRDMOD model, developed to allow for the multi-species and multi-gear characteristics of Italian fisheries (Accadia & Spagnolo, 2006) and organised into four main modules: management, biological, economic and variations in state.

## Conclusions

The review of established resource assessment methods presented here is merely intended to provide a reasonable idea of the enormous work of development and refinement of methods carried out by Italian research as part of the work promoted initially by Law 41/1982 and then by the entry into force of the EC regulations connected with the Common Fisheries Policy.

The traditional distinction between direct and indirect methods, related to the nature of the data, and that between an analytical and global approach, connected with the types of models used, from the North European and North American school, has been fairly rapidly abandoned, in line with what is seen in the research developed in the Mediterranean and in tropical seas (Caddy, 2009). The consistent use of data collected during scientific surveys has been one of the distinctive features of the resource assessments conducted out by Italian fisheries research since the 1980s (Abella *et al.*, 1999; Relini, 2000; Zamboni *et al.*, 2000; Abella, 2007; Lembo *et al.*, 2009).

Another aspect regards the tendency, given the complexity of the topic and its particular context of uncertainty, to consider the various data collection and analysis methods, which are often independent, in a complimentary manner. This approach, which goes beyond the traditional use of abundance indices obtained from the monitoring of commercial fisheries, has had the advantage, in comparison with traditional methods, of considering ecological aspects and multidisciplinary experiences in the evaluation procedures.

The availability of data on the structure of catches by commercial fisheries since 2005, nevertheless, now allows the complex problem of assessment to be more adequately addressed, moving towards a new synthesis that integrates sources of independent data, gathered as part of the National Programme for Collection of Fisheries Data, and also includes the spatial dimension of information, for an increasingly effective management of Italian fisheries.

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# 16.3 Sensitive habitats, essential habitats and their fragility

*Ardizzone G. D.*

According to the classical definitions of ecology, a habitat is the space in which a species lives. Apparently this definition would seem to be unequivocal, but often things are more complex, as habitats are not always uniform, nor do species live continuously in a single space. For example, most fish go through a pelagic larval phase and therefore live in open water; they then move to the seabed, changing location in relation to their size. At the same time, seabed complexity creates a diversity of microhabitats that offer shelter from predators and food resources for many species. However over and beyond definitions, the key point is that habitats in their complexity and integrity are fundamental for the survival of species and there is increasing concern about how their deterioration can affect the survival of exploited fish species. Indeed the continuous growth in fishing effort in all seas throughout the world has caused the deterioration of the habitats in which fish species live, eat and reproduce (Jennings & Kaiser, 1998).

## Sensitive habitats

Sensitive habitats can be defined as being important areas for several species that are of interest to fisheries, and are often also important from a naturalistic point of view. One example of this type, recognised at international level for the Mediterranean, are the meadows of *Posidonia oceanica*. The main benthic communities, described as ecologically significant and at the same time fragile and easily affected by the anthropic impact of fishing, are hereafter indicated. Generally, these habitats are well known, even in the more typically naturalistic environmental protection contexts. Their spatial distribution in the various seas is, however, not clearly known. The most important information regarding these sensitive habitats will be summarised hereafter, listing firstly the ones present on the continental shelf and then the ones present in deep waters, referring to the nomenclature indicated in Relini and Giaccone (2009).

### Sensitive habitats on the continental shelf

#### Coastal lagoons and brackish water

Lagoons are traditional coastal area habitats, semi-closed and not very deep (figure 16.11). In these conditions sea water enters into the inner areas, via channels or mouths, giving rise to areas characterised by a mixture of both fresh and sea water. Tides, currents, waves as well as sedimentation from inland waters have an effect on the salinity of these areas which can vary from 0.05 to over 4 per cent.

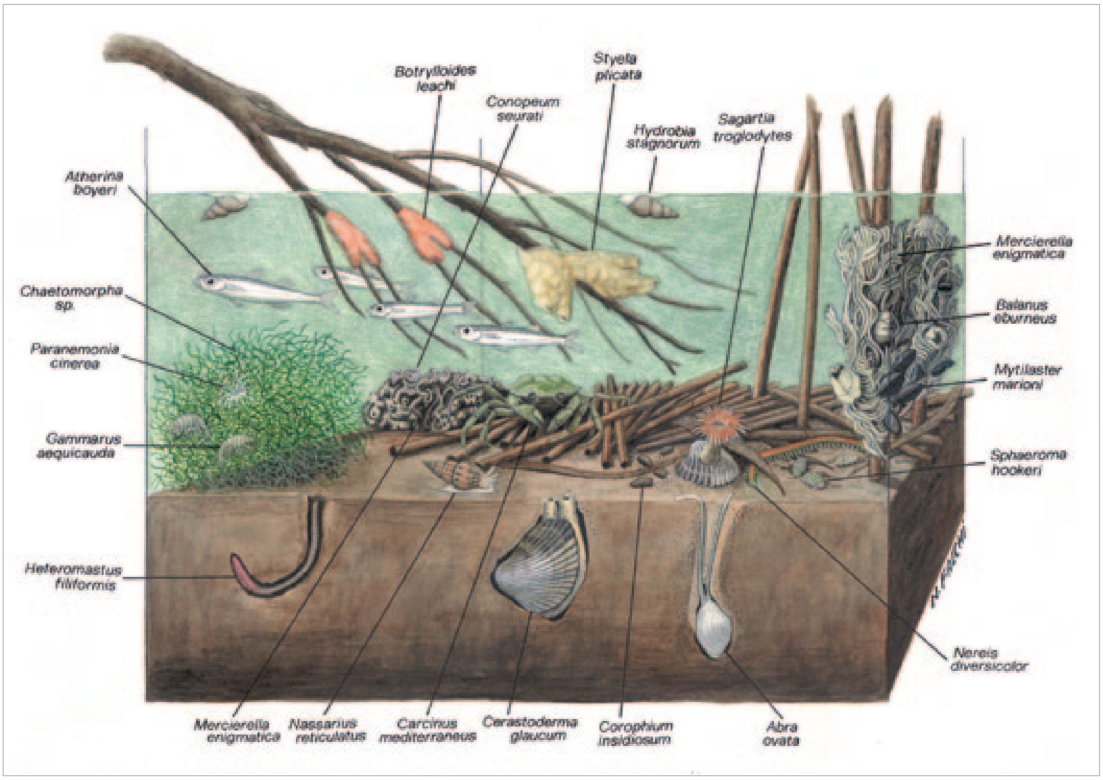


**Figure 16.11 - Schematic diagram of a coastal lagoon bed showing the characteristic species (drawing by N. Falchi).**

Inflows from inland waterways are the main sources of nutrient supplies. Coastal lagoons and brackish waters are highly productive and their primary output can be compared to the most important land plant ecosystems. Furthermore the abundance of plant organic substances and of detritus favours the concentration of juveniles of many marine fish species. The recruits enter the lagoons from the sea, attracted both by the abundance of food and by the reduced risk of predation. At a later stage, the adults go back to sea to reproduce. The main fish species in a lagoon environment are eels, *Anguilla anguilla*, European sea bass, *Dicentrarchus labrax*, gilthead sea bream, *Sparus aurata* and five species of grey mullets: *Mugil cephalus*, *Liza ramada*, *L. saliens*, *L. aurata* and *Chelon labrosus*.
The main risks for the areas are tied to pollution from mainland waters and to eutrophication, which can cause severe dystrophic crises with important fish mortalities. The concentration of pollutants in the sediments of these basins can over time create conditions from which recovery is difficult.
Since coastal lagoons are priority habitats, they are protected according to the Habitat Directive (Attachment I). Over the last thirty years, there has been a progressive impoverishment of the fish stocks naturally present in Italian coastal lagoons, due to the progressive decreasing of juveniles from the sea, caused by the marine stock overexploitation. The phenomenon is more complex and tied to the drastic reduction in spawning stock or to the poor quality of the brackish water or to habitat loss. Eels are a particular case, with recruitment having decreased dramatically over the last few years, and in some areas, such as in the North Adriatic lagoons, having almost completely disappeared.

### *Posidonia oceanica* meadows

*Posidonia oceanica* is the most important endemic plant in the Mediterranean, forming extensive meadows up to 40-50 metres deep (figure 16.12). These underwater meadows play a key role in the marine ecosystem, both because they are amongst the most important producers of oxygen and organic biomass, and because they are essential for the life cycle of many organisms, since they offer the latter the ideal environment for feeding and reproduction. Furthermore, *Posidonia meadows* play an essential role in coastline dynamics, since they stabilise the substrate, reducing the effects of erosion.

The Posidonia beds host a large number of species that characterise both sandy and rocky beds. Amongst the fish species that are most significant for fisheries, there are the Labridae or wrasses (*Labrus viridis*, *Labrus merula*), the Sparidae or sea breams (*Diplodus annularis*, *Diplodus vulgaris*, *Sarpa salpa*, *Boops boops*, *Pagellus acarne*), the striped red mullet (*Mullus surmuletus*), the Mediterranean moray (*Muraena helena*), the European conger (*Conger conger*); amongst crustaceans, the Mediterranean lobster (*Palinurus elephas*); and amongst cephalopods, the common octopus (*Octopus vulgaris*). The meadows are also recruitment areas for diverse species, such as *Trachurus* spp., *Pagrus pagrus*, *Serranus cabrilla* and *Chromis chromis*. In the Mediterranean Sea there is an ongoing progressive reduction in the extension of Posidonia meadows, which has been taking place over several decades, and it is particularly accentuated in the vicinity of the most heavily populated coasts. This regression has created destabilising effects for the entire coastal marine ecosystem.

There is excellent knowledge about the distribution and relative cartography of *Posidonia oceanica* along the Italian coasts.

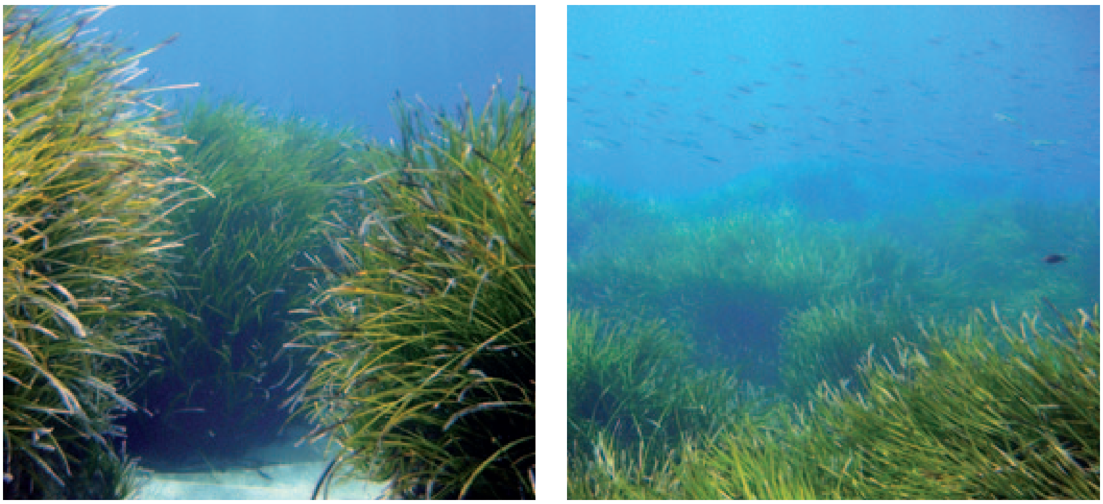

**Figure 16.12 - *Posidonia oceanica* meadows (Photos by A. Belluscio).**

Considering the slow growth of Posidonia on the short- to medium-term time scales, the reduction of meadows must be considered as being an irreversible process for wide areas of seabed.

The main regression factors for underwater meadows are associated with coastal area mismanagement, with changes in sedimentation rates, increase in water turbidity, coastal bottom trawling and vessel anchorages (Boudouresque *et al*., 2009).

Figure 16.13 - The effects of bottom trawling on a meadow of *Posidonia* (Photo by G. D. Ardizzone).

*Posidonia* meadows are protected by specific European and national regulations. *Posidonia* meadows are considered one of the main coastal marine habitats and are protected as a SCI (Site of Community Importance) by Habitat Directive 43/92. Community Regulation (EC) 1967/2006 prohibits fishing with trawl nets on all beds with spermatophytes in EU waters.
Bottom trawling carried out illegally at the edges of the meadows (at depths lower than 50 metres), to catch the more valued species that seek refuge and nourishment in the meadows, has contributed to the regression of deeper meadows (Ardizzone *et al.,* 2000, 2006).

## *Maërl* and rhodolith beds

*Maërl* beds are characterised by dense populations of calcareous red algae, which move on the seabed when there are strong currents. Typically, *maërl* can be found on beds with laminar irregular currents present between 20 and 90 metres deep in west Mediterranean and between 90 and 120 metres deep in south and east Mediterranean. Characteristic species of *maërl* are *Lithothamnion coralloides* and *Phymatolithon calcareum* (Bressan & Babbini, 2003). Rhodolith beds are similar to these, although the characteristic species are *Peyssonnnelia rosa-marina* and *Lithophyllum racemus*.

The elliptical, spherical or articulate shapes of rhodoliths are related to the type and intensity of hydrodynamics. The composition of various species of rhodoliths is often a vertical stratification of several coralline species. In favourable conditions *maërl* is capable of covering large seabed surfaces and it is for this reason that it is considered, alongside *Posidonia oceanica*, as one of the largest benthic communities dominated by plants (Ballestreros, 2006). Maërl produces a sort of microscopic forest which houses a very diversified algal and animal community: more than 300 plant species and 700 animal species were surveyed in these environments for the Mediterranean. *Maërl* has a very slow growth and renewal rate (50-75 years) and is an important concentration area for species that are targeted by professional fishing, such as *Scorapena notata*, *S. scrofa*, *Trigloporus lastoviza*, *Trigla lucerna* and *Pagellus erythrinus*.

This habitat is subject to important stresses tied mainly to bottom trawling, which is capable of altering and fragmenting the community structure, scattering the rhodoliths and modifying the associated fauna (Barbara *et al*., 2002).

The main plant species that constitute a *maërl* community are included in Annex V of Habitat Directive 92/43, which concerns "animal and plant species of a community interest the naturalistic interest of which is recognised and the exploitation of which must be carefully managed". It is a habitat that is also protected by regulation (EC) 1967/2006.

Knowledge about the presence of *maërl* beds along the Italian coasts is still somewhat fragmentary. Indications can be found in the following: research for establishing Marine Protected Areas; studies carried out for Environmental Impact Assessment (in particular investigations for excavating relict sands or for pipe laying or drilling facilities); and fishing surveys with bottom trawling nets. Habitats are identified and accurately mapped in the cartography for some fifteen locations spread over Liguria, Tuscany (Gorgona & Capraia Islands), Lazio (Pontine Islands), Campania, Calabria, Apulia, Sardinia and Sicily.

Lack of scientific knowledge and the difficulty in studying this habitat makes it difficult to evaluate the state of *maërl* bed conservation. As this habitat is particularly sensitive to silting and to bottom trawling, it can be supposed wherever areas are not altered by these two factors, environment is still in good conditions.

## Coralligenous beds

The distribution of the coralligenous community is tied to a combination of abiotic and biotic factors: the presence of a hard substrate (either original or formed from concretions of organic origin), reduced luminosity, relatively constant low temperatures, reduced sedimentation rate and dominance of plants. This biocenosis is situated between 10 and 60 metres deep on coastal beds in the presence of turbid waters, but can extend down to 120-140 metres in extremely clear waters. Amongst algae species, the calcareous Rhodophyceae *Lithophyllum stictaeforme*, *Neogoniolithon brassica-florida*, and *Mesophyllum lichenoides*; amongst the soft algae the Phaeophyceae Cystoseira opuntioides and *Cystoseira spinosa* are frequent, as well as other Rhodophyceae such as *Osmundaria volubilis*, and the green algae Halimeda tuna and *Flabellia petiolata*, can be found.

**Figure 16.14 - A detail of a coral-covered hard bed (Photo by A. Belluscio).**

This biocenosis can have different morphologies that can be narrowed down to two main types: the first is cliff-face coral, present on a generally highly inclined hard substrate, on underwater reefs, at the entrance to caves or under large rocks. Along with algae, these substrates tend to be colonised by large erect invertebrates such as: the Gorgonacea *Paramuricea clavata*, *Eunicella singularis*, *Eunicella cavolinii* and *Leptogorgia sarmentosa*; the Alcyonacea *Alcyonum acaule* and *Alcyonum coralloides*; the sponge *Axinella polypoides*, and the Bryozoa *Smittina cervicornis*, *Porella concinna*, *Pentapora fascialis* and *Myriapora truncata*. The second is an encrusting or so-called platform coralline algae which covers more or less horizontal surfaces and elastic sea beds, due to the concretion of various calcareous algae species such as the Corallinaceae and the Peyssonneliaceae.
Coralline reefs are frequented by fish species with a high commercial value, such as those of the genera *Diplodus*, *Epinephelus* and *Serranus*, and they are inhabited by crustaceans such as the lobster and by Anthozoa such as red coral.

The main risks for this biocenosis are the increase in the rate of sedimentation and of some fishing activities. Coralline reefs are indeed sensitive to sedimentary imbalances, and are adversely affected by silting and by excessive water turbidity. They are also specifically damaged by fishing with bottom set nets carried out on rocky beds and by bottom trawling and uncontrolled underwater diving.
Habitat directive 43/92 EEC lists coralline reefs amongst the priority environments. As reefs of an organogenic nature, they are also protected by regulation (EC) 1967/2006. Furthermore, particular Coralline reefs, known as “tegnue” or patch reefs, are protected as Fisheries Restricted Areas in the Adriatic Sea.
Knowledge about the distribution of coralligenous reefs in Italian seas is somewhat limited and only in the last 10-15 years has additional information been obtained through surveys carried out to establish Protected Marine Areas.
The conservation state of coralligenous reefs along the Italian coasts is still not well known.

*Leptometra phalangium* beds
This *facies* of the shelf-edge detrictic bottom biocenosis is not listed along with the priority habitats of the Barcelona convention, although it is a fragile and extremely important environment for many fish species of commercial interest. It develops at the edge of the continental shelf, at the slope break points and in the presence of detritic deposits. The crinoid *Leptometra phalangium* is a characteristic species, distributed between 120 and 180 metres depth, where it reaches high density and biomass levels. The meeting of rising bathyal waters and laminar shelf currents creates a turbulence which produces a suspension of sediment particles, favouring colonisation of filtering organisms such as the above-mentioned crinoids which feed on these particles and on numerous other benthic organisms. A complex community associated with *Leptometra* therefore develops and this supports a large number of demersal fish species. Amongst the main species, there are *Merluccius merluccius*, *Trisopterus minutus capelanus*, *Mullus barbatus*, and *Argentina sphyraena*, whereas *Illex coindetii* and *Parapenaeus longirostris* can be listed respectively amongst cephalopods and decapod crustaceans. It is worth mentioning that this facies is the most important concentration area for young hake in the recruitment phase and therefore has become a particularly critical seabed to ensure they are protected (Colloca *et al*., 2004).
There are many risks for this habitat, since the mobile substrate and crinoid fragility make it extremely sensitive to bottom trawling which is normally carried out in these areas without any limitation.
No regulation is currently in place to protect this habitat.


Figure 16.15 - Effects of trawling on the detritus beds (a) and (b) and on *Leptometra phalangium* (c) (Photos by G. D. Ardizzone).

## Sensitive habitats in deep waters

In the Mediterranean Sea, the transition between the circalittoral and the bathyal zones is set at around 180-200 metres deep and corresponds with the limit of the continental shelf. The deep water or aphytal system starts with the bathyal zone. From the 1950s, bottom trawling has moved offshore, increasingly impacting the particularly fragile bathyal biocenosis.
A precautionary proposal was made as from February 2005 to prohibit bottom trawling beyond 1,000 metres deep (GFCM, 2005). Council Regulation (EC) 1967/2006 made this ban executive for EU countries, even though this type of fishing is no longer carried out by any Mediterranean country. The intention was to avoid future developments concerning exploitation of beds that are as yet little known.

### *Funiculina quadrangularis* beds

The *Funiculina quadrangularis, facies* of biocenosis of bathyal muds, is found in beds occupied by fluid soft mud film and it is comprised between 170 and 800 metres deep. *F. quadrangularis* is a Pennatulaceo or sea pens, which can exceed one metre in height (figure 16.16) and densely populates bathyal mud bottoms. As a filtering species the ecological role of Funiculina is not clear in an environment with a low level of hydro-dynamism. Nevertheless, the fluid consistency of the mud allows suspended detritus particles to pass through and this is an useful food source for the species.
These beds are particularly rich in crustaceans with a high commercial value, amongst which are the pink shrimp *Parapenaeus longirostris* and the Norway lobster *Nephrops norvegicus*. Cephalopods such as *Eledone cirrhosa*, *Illex coindetii* and *Todaropsis eblanae* are also common.
The conservation state of this habitat is strongly compromised due to the intensity of bottom trawling, carried out regularly along the entire Italian coastline, as these are highly productive beds particularly for the above species of crustaceans.
No regulation is currently in place to protect this habitat.
The *Funiculina beds*, as well as the *Leptometra* and *Isidella* ones, are widely distributed throughout the deep soft beds of Italian seas. Even if they are not the subject of specific investigations, the characteristic species of this habitat are mentioned by all operating units participating in the Demersal Resource Evaluation Projects. As these beds are particularly sensitive to the action of bottom trawling nets, capable of completely destroying the habitat, its best conservation conditions can be observed only in areas with a low fishing effort or near to rocky outcrops, capable of offering natural protection.

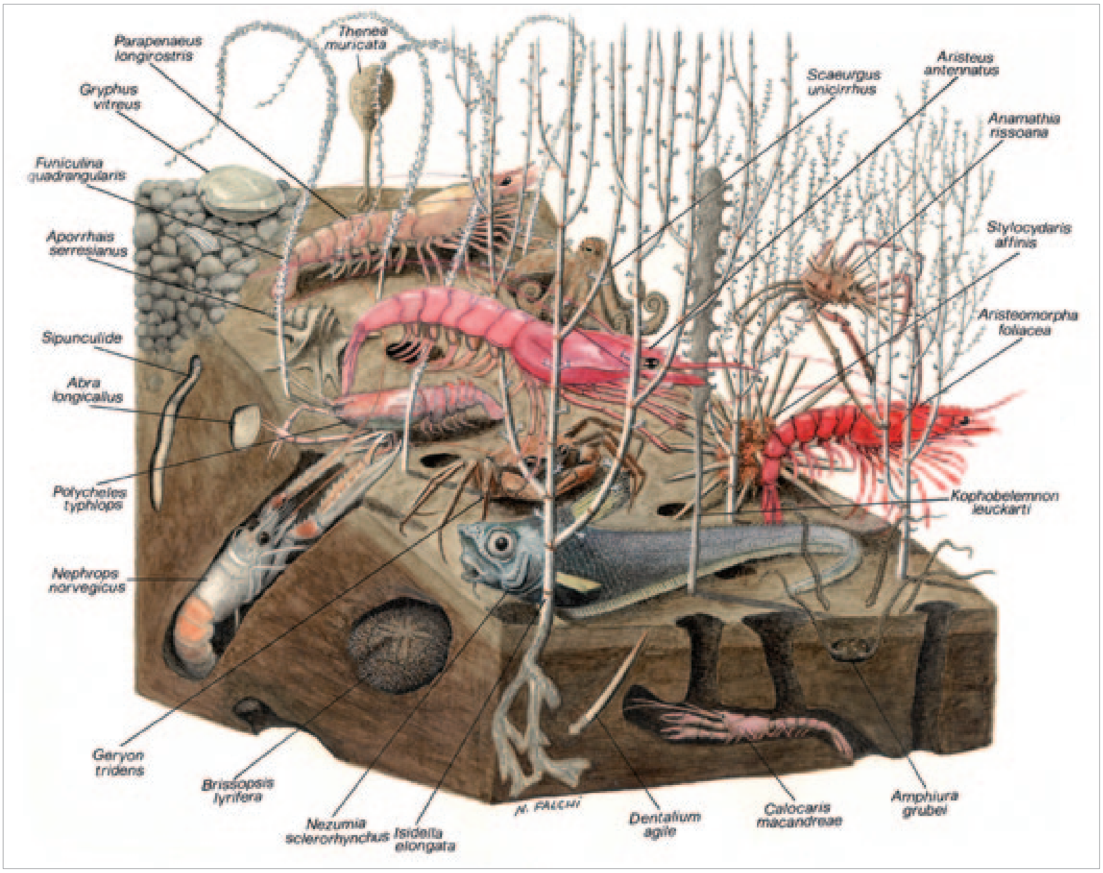


**Figure 16.16 - Schematic diagram of the main benthic and demersal species of the *Funiculina quadrangularis* and *Isidella elongata* beds (drawing by N. Falchi).**

### *Isidella elongata* beds

*Isidella elongata* is a Cnidaria that colonizes reduced inclination bathyal beds in their deepest position, between 500 and 800 metres, where the sediment is made up of compact mud with a thin layer of liquid surface mud. In these conditions, this species forms the characteristic *Isidella facies* (figure 16.16).

This environment is particularly suitable for the large red shrimps *Aristeus antennatus* and *Aristaeomorpha foliacea*, but also for numerous cephalopods, such as *Rossia macrosoma*, *Sepietta oweniana*, *Bathypolypus sponsalis* and *Pteroctopus tetracirrhus*.

The *Isidella facies* is highly jeopardised in all the Italian seas due to intense bottom trawling carried out on these beds to catches valuable red shrimps. No regulation currently protects this habitat in Italy, despite it having been included in Habitat Directive 43/92.

### Underwater canyons

Underwater canyons are special geological structures present in the bathyal shelf and are a physical interruption of the continental shelf. Active canyons are an important route for transporting inland sediments towards the abyssal zones. A high concentration of macro- and meiofauna lives

near these canyons and the fishermen that work in these areas know that they are characterised by an abundance of species with a high commercial value, such as red shrimps.
The hard materials that make up the canyon walls are a special portion of bathyal stage beds, and are host to, as yet, little known communities which should be protected against the potential damage arising from bottom trawling.
No regulation is currently in place to protect this habitat.

### White coral beds

Deep water or white coral is a rare hard base biocenosis of the bathyal zone. It is a coral barrier produced by Madreporaria which, unlike their tropical water relatives, need cold and dark waters in which to grow. This biocenosis is named from the colonising species *Lophelia pertusa* and *Madrepora oculata*, which is also normally associated with the isolated corallite species *Desmophyllum cristagalli* (Taviani *et al*., 2005).
In Italian seas, these beds are to be found from 300 to 1,000 metres deeep, but their distribution in the various geographical areas is little known. Areas with dead or sub-fossil white coral deposits have also been indicated, and these are probably remains from the last Ice Age, when they were very common throughout the Mediterranean Sea (Corselli, 2001).
White coral has a very slow growth rate (1-2.5 cm per annum) and an extremely long life span (the 10-30 metre high and 330x120 metre wide colonies along the Atlantic coasts of UK has been estimated between 1,700 and 6,250 years old).
The diversity of organisms associated with this biocenosis is very high and these deep coral reefs can be considered as being biodiversity hotspots (Danovaro *et al*., 2010).
They are, furthermore, a preferred habitat for many commercially important fish species. Bottom trawling is the main source of disturbance for this habitat. The main effect is mechanical destruction caused by fishing gear. This destruction does not just concern the complex animal community, but is capable of even modifying the hydrodynamic and sedimentation processes in the area. Even bottom trawling carried out on the soft bottoms surrounding white coral reefs is capable of altering sedimentation and therefore of suffocating coral.
No regulation currently protects this habitat in Italy, despite it having been included in Habitat Directive 43/92. Forms of protection have been suggested for the reefs in Apulia by the GFCM.
The Santa Maria di Leuca coral bank is probably the site on the Italian coast that is in the best condition, albeit the lack of available knowledge about other sites and the slow growth of coral make it important that urgent protection measures be put in place to preserve a habitat which, as yet, is little known, but at high risk of regression.

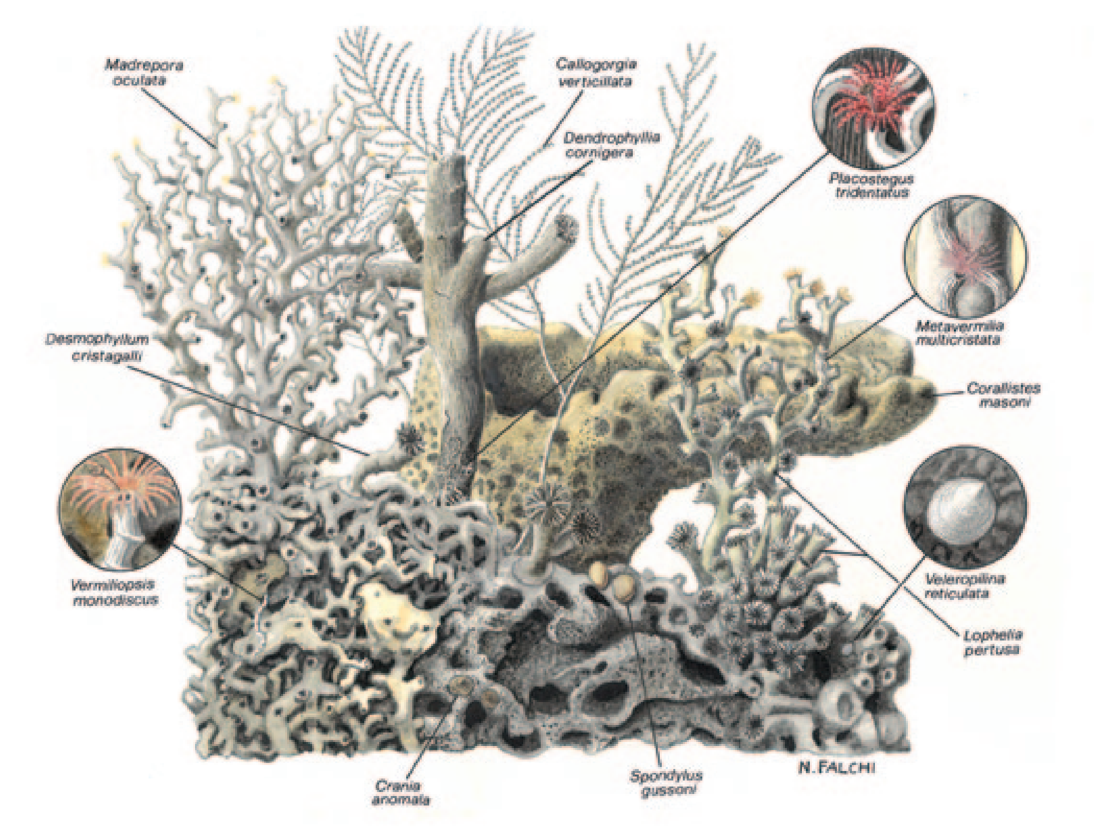


**Figure 16.17 - Schematic diagram of the main benthic species for the white coral biocenosis (drawing by N. Falchi).**

# Essential habitats

Essential habitats can be defined as being the more fragile and critical parts of each habitat in relation to the biological and ecological needs of each single species targeted by fishing. If these parts of the habitat are under some kind of impact, they must be protected to allow the species to continue a sustainable production.

Unfortunately, we are unable yet to list the essential habitats of the most important species in Italian fisheries. Up to now, little has been done to biologically and geographically describe and define them. Despite much research work being available for the main species, little wide-scale work has been done and above all little methodological work has been carried out at a regional level. The problem to be tackled is what needs to be done in order to identify essential habitats and how to select the species requiring action. If we choose to operate at a Geographical Sub-Area (GSA) level, then we must identify the most significant species from the point of view of both abundance and commercial value per geographical unit, and once these have been identified, we have to decide what species require measures to be taken as well as the priority for this action. At this point, it is necessary to widen our knowledge about their relative feeding, reproduction and recruitment habits, so that we can assess the critical points of the biological cycle for the species under consideration compared to the different areas in which each animal lives.

For example, in GSA 9, the main species is *Merluccius merluccius* and it is known that recruitment occurs on beds of *Leptometra phalangium*, where the concentration of young juveniles reaches extremely high values in the waters of Tuscany and Latium. A good way to protect the essential habitat for this species would be to spatially identify the recruitment areas and protect them, prohibiting any fishing that destroys its habitat and captures juveniles with an extremely low commercial value. Similar work should be carried out for the most important species in all GSAs, in order to build a defence plan for essential habitats.

## How to protect habitats

Identifying essential or sensitive habitats is not a general strategy for protecting all marine habitats and communities that are suffering from the adverse effects of modifications or damage. Nor is it a way to consider the impact of fishing activities on all vulnerable species (such as turtles, the Mediterranean monk seal, cetaceans, etc.). On the contrary, the objective is to try to identify ecological and biological phases taking place in particular habitats for important species subject to fishing, in such a way as to reduce the risk of damage due to fishing activities. In many European nations, experimental sampling surveys are carried out on a regular basis, as is the case for the International Bottom Trawl Survey in the Mediterranean (MEDITS) (Mertrand *et al*. 2000). In several cases, counting both national and European statistics, over 25 years of historical dataset series are available. These data were mainly used in numerous publications, but even more could be used in order to build up a reference framework that would be of use in creating a protection system for critical habitats. For example, GIS (Geographic Information System) techniques could be used to integrate the different available components, such as bio-ecological information, socioeconomic, geographical and morphological data, and this would be very advantageous for obtaining a spatial description of the complex interactions existing in various geographical areas, thereby enabling management to be correctly planned. If one or more species is overfished in a certain area, and there is an evident need to reduce the fishing effort, then a reduction approach based on biological and ecological criteria could have better effects causing less disturbance for fishing. Indeed, closing sensitive habitats, and therefore the ones that are essential to fishing, can be more effective, compared to a general closure (as is the case for the current temporary fishing moratoria) and can occupy less space. To avoid ghost measures, which may seem to be real protection measures, but are so only on paper, it is fundamental to also define protection dimensions, namely, for example, what percentage of a trawl seabed has to be protected and what impact this protection can have on fishing effort reduction. In this sense, modelling was carried out to assess the size of the area to be protected (Colloca *et al.*, 2009) and this technique should be extended to all areas, where habitat protection measures are to be implemented. The protection of sensitive and essential habitats imposes a new vision as regards protected marine areas, including the ones governed by Council Regulations on Mediterranean Fisheries Management. The current procedure for establishing “Protected Fishing Areas” is at present lacking an overall methodology. A selection policy in protected fishing areas within each Geographical Sub-Area (GSA), could be better directed if criteria for Member States to identify habitats were to be introduced, with a list of priority GSA commercial species together with an identification of their relative essential habitats. At the same time, general measures could be taken for sensitive habitats, recognised at a GSA or basin level, including them amongst protected habitats, as it was done for *Posidonia oceanica* meadows.

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