## 1. INTRODUCTION

The workshop was held in the city of Corfu on Kerkyra Island, Greece, from 26 to 30 July 2000. Eighteen scientists in all originating from ten countries did attend the meeting at the invitation of CIESM.

In opening the meeting, the Director General Prof. Frederic Briand warmly welcomed all participants and expressed his gratitude to Prof. Kostas Stergiou, coordinator of the CIESM task force on fishery science, for his overall assistance, both conceptual and logistic, in the preparation. He stressed the exploratory nature of this event, where more emphasis ought to be placed on challenging current hypotheses and on identifying promising paths for future research, than on treading established and familiar concepts. From his own perspective as an early foodweb investigator, he was eagerly looking forward to learn whether current conceptual tools and methodologies could help assess the specificity, extent and gravity of trophic changes related to fisheries in the Mediterranean Sea and whether they could offer predictive insight on the future recovery or collapse of particular fish populations.

Kostas Stergiou, coordinator of the workshop, followed by recalling the genesis of the meeting. At an earlier CIESM workshop (1999), Mediterranean fisheries were recognised as a perfect case for testing precautionary approaches and identifying relevant reference points. The current workshop was developed on this ground, mostly aiming to:

(a) identify gaps in and assess the potential use of trophic levels for the identification and characterization of fisheries status,

(b) assess the usefulness of trophic levels as reference points, and

(c) establish directions for future ecological and fisheries research in relation to the above aspects for the Mediterranean Sea.

In particular recent field data and models of Mediterranean food webs would serve as a main focus for testing the “fishing down theory” in a Mediterranean context. Particular attention would be given in discussions to the precise definition of trophic reference points of potential use for a precautionary approach to Mediterranean fisheries management.

### 1.1. Background and objectives

Traditional assessment models, management strategies and reference points derived from them are largely inadequate to prevent overfishing for a variety of reasons. These include: uncertainty in estimates of sustainable yield; actual fishing mortality rates exceeding target rates because of unreported catches, discarding and ghost fishing; difficulty in enforcing traditional measures such as limits by numbers and minimum landing sizes; strong political influence over management decisions; and poor articulation of management objectives in fisheries management schemes (Masood, 1997; Spurgeon, 1997; Smith, 1998; Stokes *et al.*, 1999; Stergiou, 1999; Cochrane, 2000; Coelho, this volume).

The ecosystem effects of fishing can be generally classified into two categories: (1) effects at the community level and (2) effects at the level of individual species (*e.g.* Jennings and Kaiser, 1998; Stergiou, 1999; Gislason *et al.*, 2000). There is no doubt that the effects at these two levels, which can be either direct or indirect (*e.g.* Gislason *et al.*, 2000), strongly interact with each other in a complex and often unpredictable manner. The highly diversified and complex impacts of fisheries on ecosystems have led, in the late 1990s, to a growing interest in "ecosystem-based management" as a promising alternative strategy (*e.g.*, Jennings and Polunin, 1996; Mooney, 1998; Pauly, 1998; Gislason *et al.*, 2000; Cochrane, 2000; Pitcher, 2000a; Coelho, this volume). Within this framework, the adoption of a variety of "ecosystem" objectives, indicators and reference points, that trigger management actions (*e.g.* Gislason *et al.*, 2000), becomes a necessity. Yet, though the full implications of such a form of management remain elusive, there is a growing understanding that ecosystem-based management must consider the maintenance of large predators and trophic level balance in marine food webs (CIESM, 1999; Gislason *et al.*, 2000; Pauly and Palomares, this volume).

The species in a fishery typically most susceptible to depletion are those that are large and long-lived. As a fishery develops there will be therefore a shift to smaller species with faster turnover (Jennings *et al.*, 1998; Pauly *et al.*, 1998a,b, 2000a; Pauly and Palomares, this volume). The small species, which predominate in the later phases, are also those found lower in food webs (i.e., the prey species of larger fishes and marine mammals) and it is therefore expected that with increasing effort there will be a decline in the trophic level of the catch, especially if the discarded by-catch is taken into account. Indeed, the mean trophic level of fisheries landings, which can be used as an approximation of abundances of target species populations in the ecosystems, in the last 45 years has decreased steadily both at the global scale (total marine and freshwater landings) and in particular regions (*e.g.* the northern Atlantic and Mediterranean; Pauly *et al.*, 1998a; Pauly and Palomares, this volume; Stergiou and Koulouris, this volume). An implication is that this fishing down is ultimately unsustainable in that it increasingly leads to fisheries relying on organisms from lower trophic levels, such as small fishes and invertebrates.

The main objective of the workshop was to bring together experts from Mediterranean and other countries with experience in various fisheries science fields to: (a) identify gaps in and assess the potential use of trophic levels for the identification and characterization of fisheries status; (b) assess the usefulness of trophic-level balance as an objective for ecosystem management and thus the usefulness of trophic levels as reference points, and (c) establish directions for future ecological and fisheries research in relation to the above aspects for the Mediterranean Sea.

## 2. WHAT IS A TROPHIC LEVEL ?

The participants were of the opinion that it is essential to agree first on terminology.

### 2.1. Definition

Trophic levels (TLs) express the positions of organisms within the food webs that define a large part of aquatic ecosystems, namely the producer and consumer organisms, their feeding relationships and certain other components such as detritus. TLs are generally set at 1 in plants and detritus, 2 in herbivores (first-level consumers), 3 in second-level consumers, etc. However, real consumers, which may feed on organisms at more than one TL, do not usually have TLs with integer values (Odum and Heald, 1975). The definition of TL for any consumer species i is thus (Pauly and Christensen, 1995; Pauly and Christensen, 1998a):

$$TL = 1 + \sum_{j=1}^{G} DC_{ij} \quad TL_{j} \qquad (1),$$

where $TL_j$ is the fractional trophic level of prey j, and $DC_{ij}$ represents the fraction of j in the diet of i. Thus defined, the TLs of most fishes and other aquatic consumers tend to take values between 2.0 and 5.0, the latter being rare even in large fishes such as sharks (Cortés, 1999), and probably occurring only in specialized predators of marine mammals, such as killer whales or polar bears (Pauly *et al.*, 1998c). However, the possibility should not be discounted that the production of some fishes may be based on surprisingly long food chains (*e.g.* Badalamenti *et al.*, this volume).

Equation (1) defines the TL as a measurable entity rather than a "concept" (Rigler, 1975) and it can thus be validated using different methods, such as mass balance models of trophic fluxes in ecosystems (Christensen and Pauly, 1992; Pauly and Christensen, 1998b; Pauly *et al*., 2000b) and the relative abundance of the stable isotopes of nitrogen, namely $^{15}$N to $^{14}$N (Kline and Pauly, 1998; Polunin and Pinnegar, this volume).

### 2.2. Why measure trophic levels?

Managers face a number of possible objectives in management, in particular:

- profit maximisation,
- job maximisation, and
- good ecosystem health.

There are other very important issues influencing fisheries in the Mediterranean, including species invasions (i.e., many exotic species from the Atlantic ocean and the Red Sea: see CIESM Atlas of exotic species on www.ciesm.org and the Introduced Species Table in Fishbase on www.fishbase.org), natural environmental changes, and eutrophication, with the last being particularly important for the Mediterranean Sea (*e.g.* Pauly and Palomares, this volume; Caddy *et al.*, 1998; Caddy and Garibaldi, 2000).

The participants agreed that fractional TLs are a useful way of describing the state of fisheries because they:

- reflect complexity, including biological diversity, which is particularly relevant in a system such as the Mediterranean (for issues related specifically to the Mediterranean Sea, see recent reports of the FAO/GFCM and *CIESM Workshop Series* 5);
- largely relate to the sizes of target species (see equation 1 and Fig. 1 in Pauly and Palomares, this volume), which in turn are linked to their fecundity and thus to their scope for recruitment;
- tend to reflect other types of stress, such as pollution;
- allow the development of new approaches to the analysis of aquatic food webs, notably the estimation of the primary production required to maintain fisheries (Pauly and Christensen, 1995; Tudela, this volume); and
- allow the construction of a series of mean TL values of fish and invertebrates landed by fisheries as a means of evaluating their impacts on marine ecosystems (Pauly *et al.*, 1998a,b; Pauly *et al.* 2000a; Pauly and Palomares, this volume; Stergiou and Koulouris, this volume).

TLs therefore highlight the ecosystem basis of production and potentially underpin an ecosystem approach to management, for which a theoretical framework is presently lacking.

## 3. Estimating TLs

There are various methods for estimating fractional TLs, both direct and indirect, and these methods together with their assumptions, advantages and disadvantages were discussed during the workshop.

### 3.1. Estimates from gut contents data

TL estimates have traditionally been derived from gut contents data (with food items expressed in terms of weight, volume or caloric content, with all three measures being equivalent), principally using equation (1) (Palomares and Sa-a, 1998). With the exception perhaps of fish larvae, the food items of which are all uniformly small, frequency of occurrence is not a good index of a food item’s contribution to the diet of a given population and the various indices applied to frequency of occurrence data do not remedy this basic flaw (Palomares and Pasqualita, 1998). Estimates of TL obtained from diet composition will differ in space and time as well as with the size of the specimens of species i (*e.g.* Pauly *et al.*, 1998b; Pauly and Palomares, this volume).

It is worth noting that this method makes a number of assumptions, in particular that all major items in the diet are identified, that they are quantified correctly, and that their TLs are validated. In reality, there are major problems merely with identifying many types of organisms and other items, such as gelatinous zooplankton and detritus, which may nevertheless be important in diets. Quantification poses many problems also, including the fact that most gut-content data offer mere snapshots of diet, volume indicators are often inadequate, and carnivores may most of the time have empty guts. It is also the case that TLs are often poorly known; for example, many “piscivores” may feed extensively on invertebrates, and “herbivores” and “copepods” may be partly carnivorous. The latter shortcomings arise especially from aggregation of species into broad groups, but also highlight the paucity of actual data in practice, the latter being a reason for lumping into groups in the first place (*e.g.*, Polunin and Pinnegar, in press).

A stand-alone application for estimating the TLof any consumer given quantitative or qualitative information on the composition of its food, TrophLab, has been recently developed (Pauly *et al.*, 2000c; available for download at www.fishbase.org). TrophLab has two basic routines: (a) estimation of TL (with standard errors, SE) from quantitative diet composition data; and (b) estimation of approximate TL (with SE) from a list of items known to occur in the diet of an animal (i.e., from qualitative information). TrophLab also provides an estimate of omnivory (i.e. feeding on more than one trophic level: Pimm, 1982). Thus, an omnivory index (O.I.) can be derived from the variance of TL of a consumer’s food groups (Christensen and Pauly, 1992):

$$\mathrm{OI} = \sum_{j=1}^{G} \left(\mathrm{TL}_j - \mathrm{TL}_i\right)^2 \ \mathrm{DC}_{ij} \qquad (2).$$

The O.I. takes values of zero when all feeding occurs at the same TL, and increases with the variety of food items’ TL. Its square root is a SE (Christensen and Pauly, 1992; Pauly and Christensen, 1998a):

$$\mathrm{SE} = \sqrt{\mathrm{OI}}$$

### 3.2. Ecopath models

Ecosystems and their embedded fisheries can be modelled using the mass balance system of Ecopath (available for download at www.ecopath.org), which can simulate the energy flows between up to 50 “functional groups” of predators and prey (Christensen and Pauly 1992, 1993; Pauly and Christensen, 1998b; Okey and Pauly 1998; Pauly *et al.*, 2000c). For each functional group, estimates of all but one of the parameters of the Ecopath master equation, one of which is $\mathrm{DC}_{ij}$ of equation 1, must be obtained. Consequently, the Ecopath routines are used to solve the system of the linear equations corresponding to all functional groups for the entire system. Thus, the Ecopath approach can be used for providing additional constraints on TL estimates derived from various methods. There was some disagreement at the meeting about whether microbial loops should or should not be incorporated in Ecopath models.

### 3.3. Estimates of TL and omnivory based on isotopic data

Isotopes have been used for estimation of trophic level of fish and other consumers, but this relies on several assumptions, principally that there is a constant per trophic level fractionation of the N stable isotopes amounting to a $\delta^{15}$N increase (recently taken to be in the region of 3.4%), and that the base of the food web (or an organism of type of material representing it) can be identified and its trophic level is known. The following equation has been used (*e.g.* Post *et al.*, 2000):

$$\mathrm{TL}_{\mathrm{fish}} = \mathrm{TL}_{\mathrm{ref}} + \frac{\delta^{15}\mathrm{N}_{\mathrm{fish}} - \delta^{15}\mathrm{N}_{\mathrm{ref}}}{3.4} \qquad (3),$$

where $\mathrm{TL}_{\mathrm{fish}}$ = TL of the consumer in question, $\mathrm{TL}_{\mathrm{ref}}$ = TL of the organism or material used as reference for the base of the food web, and $\delta^{15}$N is the index of the relative abundance of the stable isotopes of nitrogen, calculated by the equation:

$$\delta^{15}\mathrm{N} = \left( \frac{\left(^{14}\mathrm{N}/^{15}\mathrm{N}\right)_{\mathrm{fish}}}{\left(^{14}\mathrm{N}/^{15}\mathrm{N}\right)_{\mathrm{standard}}} - 1 \right) \mathrm{x}\ 10^3 \qquad (4),$$

where “standard” refers to the standard used in the mass spectrometry, which is atmospheric $N_2$ in the case of the nitrogen isotopes. In contrast to equation 2, the assessment of omnivory with stable isotopes is in principle simpler, since it can be represented merely as the variability (*e.g.* SD or SE) of the $\delta^{15}$N or $\delta^{13}$C values for a particular consumer. Also, even if sampling is limited, the omnivory measure refers to periods of months in the case of most fishery target species.

The $\delta^{15}$N variability specifically reflects the TL omnivory, while variability in the $\delta^{13}$C data will indicate trophic source omnivory. TL omnivory is expected to increase with TL, a consumer at a higher TL feeding on a greater range of TLs than one lower in the food web (France, 1997).

Although the isotope method is being widely used to estimate trophic level (*e.g.*, Vander Zanden *et al.*, 1999), there are a number of shortcomings in the technique, including the fact that the per trophic level isotopic fractionation may be variable and deviate either way from the value of 3.4, and the material used isotopically as a basis for $TL_{ref}$ may not be a good reference and its TLmay not be precisely known. Basal materials such as phytoplankton present a particular problem because they are seasonally variable in isotopic data. There has been therefore a tendency to use primary consumers, thought to respond to seasonal changes of their size. Secondary consumers such as “zooplankton” (Vander Zanden *et al.*, 1999) or unionid mussels (Vander Zanden *et al.*, 1997) used as reference materials for isotopic studies of food webs are rather assumed to have integral trophic levels, yet this may not be the case, as implied previously.

### 3.4. Evaluation of isotope and gut-contents methods

Both gut-contents and isotopic methods for estimating TLs have strengths and weaknesses (Polunin and Pinnegar, this volume). The two sets of data have rarely been compared across food webs; one study indicates good agreement (Kline and Pauly, 1998), but other data suggest that TLs based on an Ecopath model may be higher than isotopic estimates at higher TL values (*e.g.*, Polunin and Pinnegar, this volume). One attraction of isotopically-based estimates of TL is that the data reflect long-term diet, whereas most gut-contents data poorly reflect substantial temporal changes in diet that may exist. Isotopic data will incorporate dietary items that may not be identified in gut contents. A case in point is detritus and accounting for microbial food webs. In Ecopath and food-web data, the microorganisms involved in microbial food-webs are subject to great uncertainty, because being microscopic they are scarcely quantified in gut contents, and for Ecopath, biomass and productivity data are often guessed at, or largely omitted. Although detritus can potentially come from a variety of sources (*e.g.*, faeces, mucus,) and from more than one trophic level (*e.g.*, decomposing phytoplankton, copepod exoskeletons), in Ecopath, it is assigned a definitional TL of unity (as initially proposed by the International Biological Programme). Yet much of the secondary production, for example in the littoral and bathyal of the Mediterranean must be based on detritus. In the case of isotopic data, no assumption is made about the TL of detritus, but without additional techniques, a single variable such as $\delta^{13}$C will scarcely distinguish between multiple potential sources of detritus, such as exist in the littoral (Polunin and Pinnegar, this volume). There is need for more cross-comparison of the two types of TL data, and ultimately for better validation of both.

### 3.5. Empirical equations in data-poor situations

Empirical equations relating TL and body size (*e.g.*, Pauly and Palomares, this volume) for a wide range of available fish species, as well as extrapolation of TL values from other areas, can also be used for estimating TL in data-poor situations. The first approach is now incorporated in the Key-facts Table of Fishbase 2000 online (www.fishbase.org), and generates TLvalues for fish species in the absence of dietary data.

## 4. HOW SHOULD TLs BE USED ?

The participants discussed how TL estimates should be used within the framework of the objectives of this workshop; in particular whether it is possible to define trophic reference points, i.e., how fisheries TLs and/or other related indices can be used to define reference points that will trigger management actions useful for the fisheries of the Mediterranean. The participants agreed that the potential development of “trophic level” reference points is an important issue for the Mediterranean Sea, where other “traditional” reference points are hard to establish, in the absence of routine fisheries data despite of the considerable increase in fishing intensity with time. Several potential uses of TLs were identified and are summarized below.

### 4.1. TL and management options

Firstly, it was recognized that TLs help to define the stages of fishery development (Fig. 1). In the initial stages, a small amount of fishing may quickly deplete those species of high TLthat are susceptible to the fishing practices concerned. With additional fishing effort, the TL probably declines more gradually, as the species with greater biomass and smaller body size are likely to have greater resilience to exploitation (rapid growth, low age at maturity, etc.). Where resource species, typically fishes and large edible invertebrates such as squids, are overexploited (fishing effort f"), the only option left then is aquaculture, which relies on what is left of the food web. A fishery maintained at a high TLmay be considered to have its options open, since it retains something of the wealth of species originally present. With fishing down of the food web, however, options are lost as species high in the food web are depleted. Below a fishing effort of f", the only option is aquaculture.

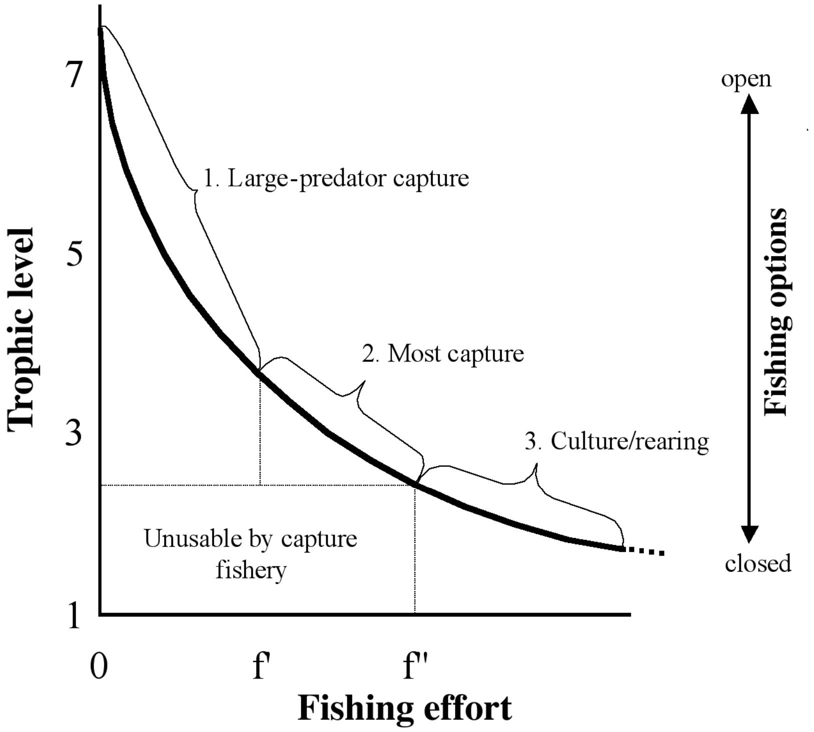

Fig. 1. Trophic level and stages of fishery development

### 4.2. Comparisons of ecological communities

TLs can be used in comparative community analyses, in which available community data can be re-expressed using TLvalues as a common currency, in other words known TL values are applied to the fish species in a community and the resulting mean TL can be derived (see Harmelin-Vivien, this volume). Whole communities can therefore be compared spatially or over time, and such analyses can also be extended to single species within them (Jennings *et al.*, 1997). Because in either case the TLis contributed to by a number of components and processes it can be expected to be a good integrative measure of ecological change.

The use of Lindeman pyramids can also be a powerful tool for comparing communities and this option has been incorporated into Fishbase 2000 (Froese and Pauly, 2000); it will also be soon incorporated into the online version of Fishbase (www.fishbase.org). However, at the level of single species some surprises were possible, as illustrated by the putatively high TL estimated for the red mullet, *Mullus barbatus* in Sicilian waters (Badalamenti *et al.*, this volume).

### 4.3. Ecological constraints on ecosystem exploitation

The combined use of two ecological indices such as the primary production required to sustain fishery (PPR; Pauly and Christensen, 1995; Tudela, this volume) and the mean TL, may be used for describing the state of a fishery with respect to the ecological constraints on ecosystem exploitation (Fig. 2). The PPR and TL values identify a space within which ecosystem exploitation may safely occur. A maximum sustainable PPR is given for each given mean TL value of a fishery and this could be a useful tool for establishing precautionary ecological reference points or regions (Fig. 2). This approach could be addressed by both gathering and comparing estimates of these ecological indices for well-known exploited ecosystems showing a similar level of annual primary productivity, as well as through ecosystem modelling and the subsequent dynamic simulation of alternative fishing scenarios operating at different TLs.

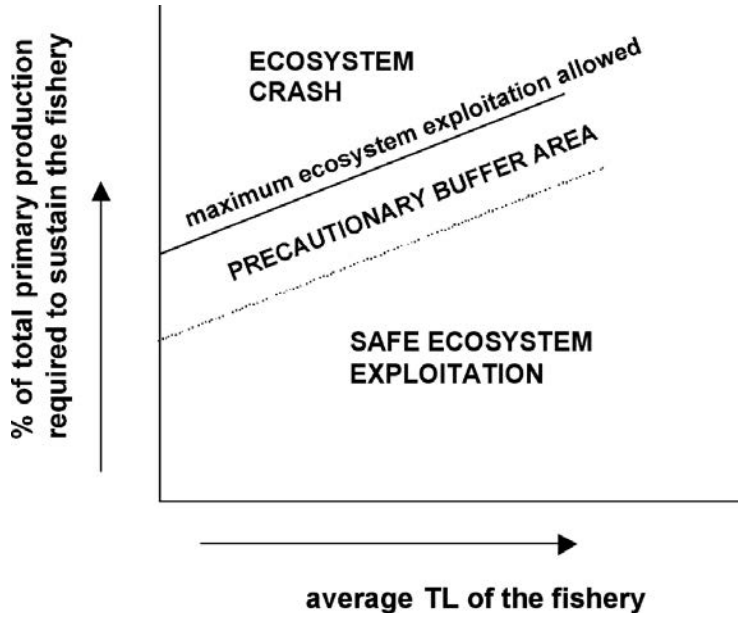

Fig.2. Scheme relating the PPR sustaining a fishery and its mean TL for describing the state of a fishery with respect to the ecosystem constraints on it.

### 4.4. Pauly's FIB index

The FIB index (Pauly *et al.*, 2000c) is defined as:

$$\text{FIB} = \log\left[\left(\Sigma\ \text{catch}_{iy}\ \ 10^{TLi}\right) / \left(\Sigma\ \text{catch}_{i0}\ \ 10^{TLi}\right)\right] \quad (5),$$

where i is a given species (or group of species) and y a given year, with year '0', i.e., the year at the start of a series, serving as anchor. The FIB index changes its value only when a decrease in TL is not matched by a corresponding increase in catch, and conversely for increasing TL (i.e., "corresponding" is defined as a 10-fold increase for a decline of one trophic level, as implied by a 10% transfer rate between trophic levels (Pauly and Christensen, 1995).

The FIB index can be an indicator of a "trophic level balance" objective in fishery management. A potential reference point can be a minimum value of the index, and especially values <0, which may be associated with the collapse of a fishery (*e.g.*, NWAtlantic fishery; Pauly and Palomares, this volume, Fig. 4).

### 4.5. Reconstruction of old time series

The participants discussed and agreed that in order for TL and other related indices to be useful within the framework of management, they must be compared against "a baseline" corresponding to the ecosystem state before the era of intense fishing activities. Nicholas Polunin illustrated one possible source of information for such an approach derived from stable isotope studies of old materials (*e.g.* seal bones, fish otoliths) stored in research centres and museums, indicating substantial decadal changes in the feeding of Atlantic seals.

A holistic approach to establishing the baseline conditions involves compiling scientific, archaeological, historical, archival and anecdotal data and evidence, and thereby reconstructing old time series that would represent the background against which current research results could be compared. This is the backbone of the so-called "Back to the Future" approach (Pauly *et al.* 1998d; Haggan, 1998; see also www.fisheries.ubc.ca), which combines the output of a number of packages, namely Ecopath, Ecosim, and Ecospace, and allows integration of the many types of information required to build an understanding of how ecosystems work. The information includes that from traditional ecological knowledge of fishing communities, from fishers and processors, and from historical, anecdotal, archaeological and sediment-core research to “re-construct” systems as they might have been prior to depletion by modern industrial fishing. Ecopath models simulate the energy flows, Ecosim (Walters *et al.* 1997) allows simulation of “what if” questions (i.e. what will the system look like in 5, 10, 20 up to 50 years if we continue to harvest at the current rate), and Ecospace (Walters, 1998) allows spatial modelling of ecosystems summarised by Ecopath. Various statistical and other techniques can then be used to compare the ecological, socio-economic and cultural costs and benefits of different management and harvest scenarios simulated with Ecosim (Pitcher, 2000b).

### 4.6. Marine protected areas (MPAs)

Marine protected areas (MPAs; or no-take zones) are considered to be the extreme case of the precautionary approach (Lauck *et al.*, 1998), and provide a refuge in space rather than a refuge in numbers, the latter being the aim of most traditional fisheries management measures. Among many other benefits, MPAs protect the biomass of species, maintain biodiversity including genetic diversity, decrease the trend for heavy evolutionary fishing selection for earlier maturity and reproduction and smaller adult fish size, and hedge against inevitable uncertainties, errors in estimations, and biases (*e.g.*, Roberts and Polunin, 1991; Agardy, 1994; Lauck *et al.*, 1998; Guénette *et al.*, 1998; Hall, 1998; Sumaila *et al.*, 2000; Polunin, 2001). Thus MPAs may simultaneously satisfy various objectives of ecosystem management, including maintenance of: ecosystem diversity, species diversity, within-species genetic variability, directly impacted species, ecologically dependent species and trophic level balance (Gislason *et al.*, 2000). However, in fisheries and any other large-scale ecological context, their ability to meet such objectives depends critically on a few things, including their degree of enforcement, but also their size and the biology of the target animals concerned. In some cases, MPAs may constitute in certain respects a form of “baseline” against which for example the TL of whole communities might be compared (Polunin, 2001; MacPherson, this volume; Harmelin-Vivien, this volume).

The participants however observed that the few MPAs in the Mediterranean Sea are small in size, and there are no effective large-sized MPAs despite natural exceptions, namely deep-water areas inaccessible to fishing which act as refuges for the adults of many target species (*e.g.* Caddy, 1999). Thus existing MPAs could only offer reference points for a few relative sedentary species, which migrate or otherwise move around at spatial scales much smaller than those of MPAs.

## 5. PRIORITIES FOR FUTURE RESEARCH

Having discussed the way TL can be used within the context of management, the participants of the workshop considered priority topics for future research. From this discussion it was evident that regular monitoring of the Mediterranean ecosystems and their embedded fisheries and the establishment of a network for data storage and exchange are both of high priority.

### 5.1. Monitoring

Discussions about particular fisheries and other phenomena such as blooms of nuisance organisms including jellyfish (see Benovic, this volume) highlighted the lack of detailed historical data. Participants agreed that the lack of information hampered understanding and that through routine monitoring this lack could begin to be addressed. Regular monitoring must include both the non-fisheries components of the ecosystem (*e.g.* zooplankton, jellyfish) as well as the fisheries communities (i.e., all organisms greater than a certain minimum size) including discards. The lack of discard data is particularly important in the Mediterranean Sea, where such data have only been gathered for the last 5-7 years, with their availability generally decreasing from west to east and from north to south. Questions of spatial and temporal scale of the magnitude and variability of the various events considered are also of primary importance for defining biological and ecological reference points (see Fromentin, this volume). The results of monitoring over a period of years will allow the identification and characterisation of trends and/or changes in trophic structure, among other things, information which is central to defining and developing TL signals and identifying TL concepts with respect to reference points.

## 5.2. Gathering, storage, exchange and synthesis of existing data

The participants recognised that there had been recent Mediterranean-wide research surveys such as that of MEDITS, the results of which had not yet become available to the scientific community, and that long-term datasets, such as those on tuna (Fromentin, this volume), exist and need to be fully evaluated. However, it was generally felt that there was an acute lack of general and historical data for the Mediterranean Sea. It was strongly recommended by the participants that an attempt be made to gather all available existing information (i.e. including that from fishers and processors, and historical, anecdotal and archaeological, sources and archives). This information could be shared across the research community and used in a variety of possible ways including development of simple Ecopath models and exploring “what if” scenarios using Ecosim.

The participants agreed that indicators related to TL, the identification of reference points and warning signals that will trigger management actions, will be greatly assisted through the use of modelling and simulation tools such as Ecopath, Ecosim and Ecospace (Christensen and Pauly, 1992; Walters *et al.*, 1997, Pauly *et al.*, 2000), which have been successfully applied in various areas (*e.g.* Christensen and Pauly, 1993; Shannon *et al.*, 2000). Within this context, the organization of Ecopath workshops in the Mediterranean area was also strongly recommended by the participants. Because of the potential importance of such models, it was also felt that every effort be made to validate major outputs of Ecopath, including TL data using isotopic techniques.

## 5.3. Specific concepts

(i) There was much discussion about sharks, their occurrence in the Mediterranean today and role in the marine ecosystems prior to intensive exploitation. Participants felt that sharks may still represent important components of Mediterranean food webs and concluded that there is a need to re-evaluate their trophic roles, using historical data, and tools such as Ecopath and stable isotopes to elucidate the food webs in which they participate. It was considered likely that there were old marine materials (*e.g.* teeth, bones, otoliths, scales) available from many areas for analysis, and that these could in principle provide information on historical TLs.

(ii) Participants recognised (*e.g.*, Figs 1, 2) that community or total-landings TL data could constitute an important proxy for other important attributes such as biodiversity, productivity and perhaps system stability. For example, it was recognised that the “folding back” of catch data as TL declined with exploitation (Pauly *et al.*, 1998a) could be better elucidated at smaller scales than those of the FAO fishery zones, for example by comparisons of fisheries sectors in Hellenic waters (Stergiou and Koulouris, this volume). It was therefore generally agreed that the nature and character of such relationships needed to be clarified in future research.

(iii) The current lack of consensus over the mechanisms and magnitudes of materials fluxes through microbial loops (see CIESM, 2000) indicated that research on these loops would reduce a major source of uncertainty in simple Ecopath models. Isotopic data indicate that in some food webs the ultimate sources of materials were unclear. It was agreed that a wider range of research techniques would be required to elucidate the role of detritus, the manner of its processing in food webs and its contribution to secondary production.

(iv) Participants recognised that there was little known about the spatial and temporal variability of TL data and other fundamental attributes of food-web structure, yet understanding of such variability was important to the use of TL data as reference points for management. It was noted that stable-isotope approaches constituted a highly accessible means of exploring spatial and temporal scales of variability in such data, and thus recommended that research be carried out to apply them for this purpose.

(v) Participants noted that there were phenomena in the plankton (*e.g.*, jellyfish blooms) and in the littoral (*e.g.* sea-urchin barrens) which respectively were, or might be, underpinned by fishery impacts. It was agreed therefore that a stronger basis for predicting such phenomena and linking them to fishing and other changes in the environment (*e.g.* introductions) through research and monitoring were sorely needed. In the littoral, some phenomena were amenable to spatial comparisons such as between MPAs and unprotected areas.

# Estimates of trophic level in the red mullet *Mullus barbatus* : comparison between gut-contents and stable-isotope data

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The red mullet *Mullus barbatus* is one of the most economically important fish species in the Mediterranean Sea (Relini *et al.*, 1999). Recent research has demonstrated that in many areas it is overfished and that a prolonged (Potoschi *et al.*, 1995; Pipitone *et al.*, 1996) or temporary ban (Cau *et al.*, 1993) on trawl fishing can markedly increase its biomass. Pipitone *et al.* (1996) reported that after a 4-year closure of the trawl fishing in the Gulf of Castellammare (Sicily), its average catch per unit of effort (cpue) increased by approximately 25 times. The success of this species following the trawling ban may be related to its ability to efficiently exploit the food resources available (Vassiliopoulou and Papacostantinou, 1993). The red mullet is a specialised feeder on soft sandy/muddy bottoms (Badalamenti *et al.*, 1993) and shows a marked selectivity for several polychaete species (Lipari *et al.*, 1998).

Despite the appreciation that red mullet responds markedly to protection in trawlable areas, studies on its feeding habits are scarce and there is little quantitative knowledge concerning its position in the Mediterranean soft-bottom food webs. The present study aims to provide new information on the feeding patterns and trophic level of the Mediterranean red mullet. Herein we (1) estimate its average trophic level using stomach-content data, (2) study the variation that exists in trophic level in relation to fish size, using stable-isotopes of nitrogen, and (3) compare the estimates of trophic level obtained using the two methods (stomach contents and stable isotope data).

## MATERIALS AND METHODS

All the specimens were caught using professional fishing vessels at depths between 20 and 200 m. The lengths and weights of freshly caught fish were measured and guts and white muscle samples were removed and stored in formalin or dried in the oven, respectively.

### Gut-contents data

A total of 277 specimens of *M. barbatus* were caught between June 1994 and March 1995, with an average total length (TL) of 15.0 cm ± 1.5 (SD). Prey items were identified from stomach contents, to the lowest taxonomic level possible, and trophic level was calculated according to Mearns *et al.* (1981).

### Stable-isotope data

A total of 16 red mullets were caught in December 1999, in order to compare the results obtained through the isotope analysis with those from the gut content analysis. The average total length of the specimens was 15.2 cm ± 0.7 (SD). In addition 18 specimens, with sizes ranging between 9 and 24 cm TL, were also collected and divided into 3 size classes (small, medium and large), in order to assess differences in average trophic level with size. White muscle tissue was sampled from near the dorsal fin of each fish (Pinnegar and Polunin, 1999) and mesozooplankton was used as a reference material, assuming that most herbivorous zooplankton (*e.g.* calanoid copepods) possess a trophic level of 2. It was also assumed that with each trophic interaction $\delta^{15}$N increased by 3.4‰ (DeNiro and Epstein, 1981; Minagawa and Wada, 1984).

## RESULTS

The isotopic analysis gave a mean trophic level of 4.05 ± 0.26 (SD), which is significantly higher ($P<0.001$) than the mean trophic level calculated from the gut-contents analysis, which was 3.46 ± 0.30 (SD). Average $\delta^{15}$N and trophic level of the three *M. barbatus* size classes ranged from 9.41 and 3.80 respectively for small specimens, to 11.38 and 4.38 for large ones (Table 1). The mean $\delta^{15}$N and trophic level differed significantly ($P<0.05$) with size class, being significantly higher for large and medium sizes in relation to the small ones (Table 2).

Table 1. Average $\delta^{15}$N(‰), trophic level values and standard deviation (SD) of different size classes of the red mullet *Mullus barbatus* from the gulf of Castellammare

| **Size class** | **n** | **$\delta^{15}$N(‰)** | **SD** | **Trophic level** | **SD** |
|---|---|---|---|---|---|
| small | 6 | 9.41 | 0.61 | 3.80 | 0.18 |
| medium | 6 | 10.95 | 0.29 | 4.26 | 0.09 |
| large | 6 | 11.38 | 0.36 | 4.38 | 0.11 |

Table 2. **a)** Results of ANOVA on $\delta^{15}$N(‰), for the 3 size classes of the red mullet *Mullus barbatus* from the gulf of Castellammare, ***p<0.001. **b)** SNK test of the factor Class size, >=p<0.05

**a) ANOVA**

| Source of variation | | **$\delta^{15}$N(‰)** | |
|---|---|---|---|
| | df | MS | F |
| class size | 2 | 6.43 | 32.44 *** |
| residuals | 15 | 0.20 | |
| Cochran's test | | C = 0.63, p> 0.005 | |

**b) SNK**

Large = Medium > Small

## DISCUSSION AND CONCLUSIONS

The average trophic levels calculated for the red mullet in the Gulf of Castellammare were higher than those for other benthic species of similar sizes collected in the same area. Using gut-content data, Badalamenti and D'Anna (unpublished report) found lower values for *Lithognathus mormyrus* and *Diplodus annularis*, *D. sargus* and *D. vulgaris*. Pinnegar (2000) reported average trophic levels for 10 Mediterranean shallow water fish species collected in the rocky littoral area of the Bay of Calvi (Corsica) and three sites in Mallorca (Spain). The average trophic level of all 10 species did not generally exceed 4.1, which is higher than the value we report here for the small size class of red mullet but lower than those for the medium and large size classes. Furthermore, Pinnegar (2000) reported that large-sized top predators, such as *Dentex dentex*, *Epinephelus marginatus*, *Seriola dumerili*, *Sphyraena sphyraena* and *Coryphaena hippurus*, exhibited average trophic levels between 4.0-4.4, whilst Herod *et al.* (unpublished data) estimated similar values for Mediterranean deep-sea macrourids and sharks. These values are similar to those of large-sized red mullet.

The high trophic position occupied by red mullet in the Mediterranean benthic food web may be explained by its diet, which consists largely of carnivorous polychaete species belonging to the Polynoidae and Sigalionidae families. Lipari *et al.* (1998) showed that adult red mullet exhibit a preference for only one species, the sigalionid *Sigalion squamatum*. Variation in the trophic levels calculated for the different-size classes of individuals may reflect differences in the diet of smaller specimens, which consume more crustaceans and bivalves (Vassiliopoulou and Papaconstantinou, 1993) and smaller-sized prey (Labropoulou and Eleftheriou, 1997).

The trophic levels obtained from isotope data analysis would place red mullet high in the marine food web. It remains to be understood whether or not individual food chains supporting benthic fish species are longer than previously supposed.

# Zooplankton biomass and fish production in the Adriatic

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Trophic relations in the pelagic ecosystem include phytoplankton as the producers of organic matter; herbivorous zooplankton as primary consumers; carnivorous and omnivorous zooplankton and larval nekton as secondary consumers. Tertiary consumers, fish and cephalopods, are most exclusively carnivorous. Since man, in general, uses the last link in the food chain, major attention is paid to the latter category in terms of evaluating the state of stocks and maximizing their potential yield. Accordingly it is calculated that biomass loss is 70 to 90% on individual trophic scales, and the average ecological efficiency for coastal seas is about 15% (Ryther, 1969).

Zooplankton biomass could be measured as volume, weight, chemical composition and caloric value (Innamorati *et al.*, 1990). Considering that no single method is clearly superior, it is important to select the appropriate method, taking into consideration the particular issued studied. Although each type of measurement could be used as an input parameter for the evaluation of ecological efficiency and further calculation of potential of pelagic fishes biomass, better results could be achieved from combinations of measurements derived from the same sample.

The Adriatic Sea, the northernmost part of the Mediterranean, is subdivided into a shallow northern zone and deep middle and southern zones. Based on nutrient distribution, Buljan (1964) identified four zones in terms of productivity: A- the middle and south offshore Adriatic, B- the northern Adriatic, C- the island region of the eastern coast, and D- lagoons and river mouths. Elsewhere we estimated total fish production to be 1,026,112 t/year, using primary production and fish production from zooplankton data (Benovic, 1983).

Table 1. Annual average zooplankton dry weight in the Adriatic Sea (from Benovic *et al.*, 1984).

| Zone | Surface (km$^2$) | Average depth (m) | Volume (km$^3$) | Dry weight (t/km$^3$) | Dry weight (t/zone) |
|---|---|---|---|---|---|
| Zone A | 79,000 | 350 | 27,650 | 8.3 | 229,496 |
| Zone B | 31,000 | 75 | 2,325 | 14.2 | 33,015 |
| Zone C | 25,000 | 60 | 1,500 | 6.8 | 10,200 |
| Zone D | 2,100 | 30 | 63 | 14.6 | 920 |
| TOTAL | | | | | 273,630 |

Table 2. Estimation of annual primary production and fish production in the Adriatic Sea (from Benovic, 1983).

| Zone | Zooplankton dry weight(t/zone) | Zooplankton production (t/zone) | Primary production (t/zone) | Fish production (t C/zone) |
|---|---|---|---|---|
| Zone A | 229,495 | 5,737,375 | 2.6x107 | 860606 |
| Zone B | 33,015 | 825,375 | 3.7x106 | 123,806 |
| Zone C | 10,200 | 255,000 | 1.1x106 | 40,500m |
| Zone D | 92000 | 23,000 | 1.0x105 | 2,700 |
| TOTAL | | | | 1,026,112 |

These results are a rough estimate of the Adriatic biological resources calculated using an efficiency of 15%. Although other estimates may differ in calculated quantities (*e.g.* Karlovac *et al.*, 1974), the general trends in different zones would be the same.

A different picture emerges if we take other parameters into consideration. Linear regression analyses of zooplankton dry weight and caloric values show that a higher dry weight corresponds to a higher caloric value only during the cold season. This was true for zones “A” and “C” but not for zones “B” and “D” and could be due to the qualitative plankton composition and domination of crustaceans, in particular copepods, in the cold season and in the “oligotrophic” areas of the Adriatic Sea.

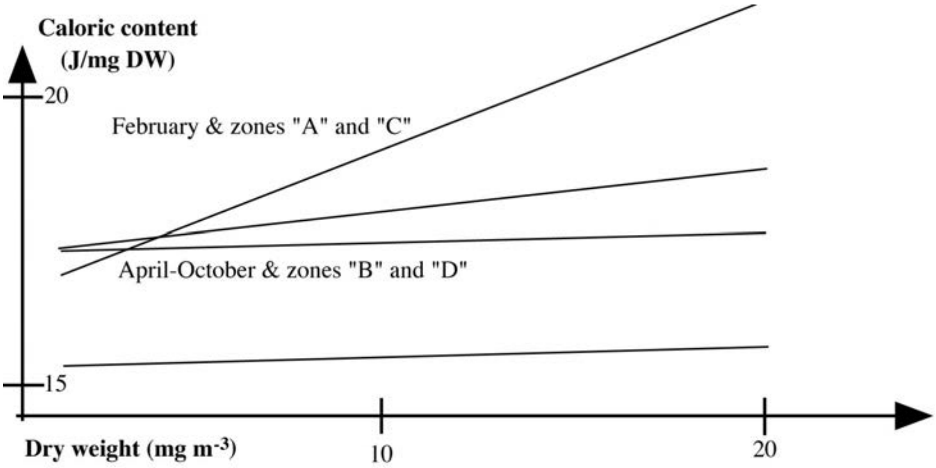

Fig.1. Zooplankton of the Adriatic Sea; relationships between caloric content and dry weight (data from Benovic & Bender, 1983).

Analyses of AFDW/AW ratio or caloric values could be useful as predictors of ecosystem perturbations. Detailed data on zooplankton quality in the North Adriatic would probably support this hypothesis. Analyses of particular plankton populations, such as medusae, might strengtten hypothesis of “signal” of future large perturbations in the marine ecosystems (Benovic *et al.*, 1987; Benovic *et al.*, in press).

# On fishery status, the precautionary approach and sustainability: does a role exist for “trophic level” reference points ?

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From an economic perspective, and with respect to the concepts of sustainability and precaution, the question posed in the title requires an investigation into whether trophic level indicators can contribute to the definition and implementation of efficient, long-term public policy. In reality, this question raises at least two others:

- how, operationally, can the environment be taken into account in the definition and implementation of economic policy (sectoral or global);
- how do such indicators contribute to the improvement sought by users or consumers of natural goods compared to the practice of market or non-market productive activities (conservation, protection, exploitation of certain natural goods or certain environmental qualities).

At the macro-economic level, attempts to take the environment into account in public policy decisions still run up against the evaluation of the expected benefits of environmental quality protection. There is rarely consensus on the definition and the quantification of impacts, even if there is not necessarily conflict. Water quality is not likely to be a consensus issue for coastal dwellers, swimmers, aquaculturists, fishers, windsurfers and so on. It also does not have a unique quantitative nature. But it may represent a federating objective for public policy. A phenomenon of eutrophication, even accidental, could also play such a role because its impacts will be controversial depending on the viewpoint taken.

Moreover, in practice, public decision-makers rarely have information on the costs associated with the deterioration or the protection of the environment. In such a context, legislating in the direction of sustainability remains a delicate exercise. The relative weight of different risks may however favour an intuitive precautionary policy. But the arguments in favour of such a policy need to be presented and documented using the best knowledge available. Probably in this first case, trophic level indicators are sufficiently integrative of biological and physical environmental data that they do add something. They may federate social actors in the debate over different possible competitive uses or different management options favouring different time scales (short, medium or long term)

More commonly, directly linking certain natural goods to a productive sector facilitates the evaluation of costs and benefits. Analysis proceeds by assessing impacts in terms of production

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This note presents a preliminary set of discussion points which require further development and must be completed with a set of bibliographic references. It may not be cited without the agreement of the authors.

technique (for example, métiers in fishing) and the natural functions of ecosystems. Frequently in this case the technical parameters of the métiers are related to production results classified according to certain biological parameters (resource type, stocks, age classes, size classes…). Simple methods can then be used to develop the relationship between these productive values and technical-economic conditions. By taking into account the characteristics of the exploited resources, the relation can be made dynamic and be projected into the medium to long term (population and economic parameter dynamics). This is done taking various uncertainties into account depending on the parameters being used, the state of development of the productive sectors and the access mechanisms to the desired resources. In this case, we remain in the conventional configuration of fishery management. Here, trophic level indicators could contribute by providing an improved synthesis of the set of indicators currently used and which we know can only be obtained with difficulty and even then only at the expense of costly and heavy monitoring protocols.

But this approach relies on a deliberately restrictive delimitation of the exploited system and on a segmented (or even sequential) analysis of the users thought to affect the natural resources. Activities and their impacts are spatialised. Predations or environmental actions are defined according to sequence or calendars (of activities) and the effects that they induce are then integrated. Trophic level monitoring which imposes its own spatio-temporal structuration could be an integrating factor for these sequential representations which comprise the reality of usage and which often impose a juxtaposition of specific indicators for the monitoring of each activity. In practice, interactions are much more complex and the total effect on the ecosystem is not simply the sum of each effect. The production of reference points could help to simplify this situation and thus facilitate choices.

In practice also, fishing is not the only factor that influences the ecosystem. In point of fact, this argument is put forward by the fishers themselves when they have to defend an increase in their fishing capacity which is considered excessive, or when having to deal with the publication of alarmist diagnoses concerning the resources that they exploit. It is also an argument that can be used where different exploitation methods compete for a common resource (for example, octopus in Mauritania, Senegal and Morocco and the conflicts between trawls and pots). Over and above these internal conflicts, this explains the social, political and economic pressures which often encourage the manager to deal with fishing in isolation, looking only at the relationship between fishing effort and catch*. The impact of fishing techniques on the environment, outside of the targeted and discarded species, is little considered. And all other factors which may affect the state of the resources are considered exogenous. Biological models used to define TACs and quotas remain relatively closed to such considerations. Indicators used to monitor fishing activities are thus often limited to parameters concerning effort and declared catch, together with at best voluntary declarations of discards and sampling of landings. But these two complementary elements contain numerous technical and social problems concerning the limits of involvement of fishers in the management system.

Issues of economic policy represent a legitimate element in that managers represent and define ecosystems. Although considered to be external, other uses do in fact exist and hence other impacts which also affect the state and the dynamics of the ecosystem under control. Ideally, it is the entire ecosystem that should be taken into account with respect to all factors that have an effect on the trophic structure**. Probably, this would allow access and control rules to be drawn up for each use reflecting social preferences expressed towards specific natural goods and certain ecosystem qualities (biodiversity for instance). In this case, could trophic level monitoring provide a better indicator for the definition of sustainable development policies and the precautionary approach? Can it simultaneously be linked to the preferences expressed by actors concerning the environment and have a link to the impacts produced by both present and future uses?

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* The difficulty in estimating discards means that in certain cases this relationship is restricted to landings and not to effective catch. This substantially reduces yet further the quality of indicators used to monitor this activity.

** The set of biophysical conditions favourable to the presence of such or such element at a particular trophic level due to the satisfaction of nutritional and/or reproductive needs (abundance and concentration indices having a major effect)

Supposing that the answer to the above questions is “Yes”, how can such indicators be put into place, how can their legitimacy be ensured (recognised by candidates for a particular use of ecosystems or for their protection) and how can they be monitored?

With respect to these questions, a number of problems remain:

- the specificity of the judicial status of environmental goods being dealt with here and the frequent sharing of responsibilities for the marine environment (from the international down to the local level) make it difficult to measure the value of natural assets. This is all the more true in that, although the resource may sometimes be the object of private rights (for example, community or individual quotas), the ecosystems within which it exists are statutorily only very rarely the object of similar rights.
- the development of these indicators supposes that we are able to establish the link between the causal factors in terms of degradation or enrichment of trophic levels and the specific needs of valuable species or resources. All this whilst checking that these factors do not contribute to diminish the welfare of consumers/users of ecosystems (for example, recreational users). Whilst taking into account external costs should allow economic compensation for measures taken to protect or rehabilitate threatened ecosystems, their evaluation is far from easy.

From a macro-economic viewpoint, no coherent information system currently exists which provides comprehensive coverage of the natural environment. However, integrated sub-systems (satellite accounts) compatible with national accounts have emerged in numerous countries. The Statistical Office of the United Nations has made some significant methodological progress. The System of Integrated Environmental and Economic Accounting (SEEA) starts from a qualitative assessment of the damage inflicted on the environment by economic activities and then measures in monetary terms the repercussions of the environmental deterioration on the economy and on welfare. This concerns in particular the exhaustion of natural assets. In theory, trophic level indicators could play a role in this measure. However, arriving at an effective monetary measure supposes that the inherent difficulties in evaluating non-market goods have been resolved. This concerns a large number of the external activities affecting commercial fishing. At the European level, the Environmental Protection Expenditure Account (EPEA) has been created, a satellite account which measures expenditure to protect the environment. In France, the French Institute for the Environment continues to produce resource balance-sheets.

Starting from trophic level indicators, a measure of natural capital could be developed and used for the definition of fishery management activities. This measure would allow the evaluation of production in monetary terms to be corrected for the costs of ecosystem protection, conservation or rehabilitation. This would require a good knowledge of the mechanisms which affect different trophic levels taking into account human impacts by usage and natural factors depending on society’s ability (scientific, operational, and economic) to prevent them, so as to be able to forecast the costs of compensating risks (principles of collective sharing and insurance) or the costs of reducing uncertainty. From a macro-economic viewpoint, a revision of the sectoral accounts and of the “marine and coastal GDP” would result, with account being taken of the use of natural assets. The legitimacy of protective policies and the precautionary approach would be enhanced.

# Relations trophiques et pêche en Algérie

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L'intérêt porté par les pouvoirs publics aux ressources marines revêt souvent un caractère essentiellement économique et financier. En Algérie, la valorisation du potentiel halieutique, après celui du sous-sol, a présenté un tel engouement qu'en 1999, les décideurs ont érigé au rang de ministère l'administration des pêches, rattachée auparavant à diverses structures d'importance relative, même s'il avait existé dans le passé un Office, puis un Secrétariat d'État, et même un Vice Ministère.

Fait paradoxal, alors que seuls les aspects liés au profit généré par l'exploitation halieutique retiennent l'attention des autorités, l'ensemble des textes législatifs concernant la ressource marine est basé sur des considérations écologiques et biologiques (zones interdites aux engins actifs durant l'été, profondeur minimale de chalutage, tailles de capture). En fait, et même dans l'éventualité où les règles auraient été respectées, l'application des mesures retenues ne répond pas aux objectifs assignés, en raison de:

- l'extrême variabilité des périodes de reproduction des différentes espèces;
- du caractère pluri-spécifique des espèces, la législation favorisant une espèce au détriment des autres;
- la dépendance des stocks vis-à-vis des paramètres environnementaux (température, salinité, apports terrigènes), avec pour conséquence des modifications importantes des périodes de ponte, surtout chez les pélagiques et les espèces côtières soumises aux influences des précipitations;
- l'ignorance de la fluctuation des maillons co-latéraux de la chaîne trophique ciblée.

Une estimation rapide de l'état de la ressource, et par voie de conséquence de l'ensemble de l'écosystème marin, consiste à formaliser les mécanismes de l'overfishing de manière à calculer un indice moyen défini comme le Taux d'Overfishing (TO), entre les maillons exploités, proportionnel au niveau où se situent les espèces cibles. Contrairement à la méthode d'estimation des niveaux trophiques (Pauly *et al.*, 1998a; Trites *et al.*, 1999), la valeur de l'indice augmente avec le niveau de surexploitation. La méthode consiste à affecter la valeur 1 au taux de capture des espèces bentho-démersales, appartenant souvent aux maillons supérieurs de la chaîne trophique, et une valeur 2 à celui des petits pélagiques, planctonophages pour la plupart. L'indice est d'autant plus élevé que les petits pélagiques sont bien représentés dans les prises.

Une première estimation fournit une valeur de TO = 1,7 (correspondant à 30% de bentho-démersaux et 70% de petits pélagiques) et l'interprétation de ce résultat reste néanmoins délicate. Ce n'est pas la valeur elle-même qui est intéressante du fait qu'une pêcherie traditionnelle orientée vers les petits pélagiques (exemple du Pérou) sera caractérisée par un TO élevé sans pour autant indiquer une augmentation de l'exploitation; par contre une variation dans le sens de l'élévation de cet indice sera liée à une exploitation accrue des maillons inférieurs de la chaîne alimentaire, correspondant au mécanisme de l'overfishing.

L'absence d'approche globale incluant l'étude d'espèces sans intérêt économique et situées en début de chaîne trophique (plancton, benthos), se traduit par une réglementation incomplète et même dangereuse. Les critères de gestion et de préservation doivent répondre à plusieurs exigences, en particulier:

- une description fine de chacune des espèces identifiées et de son environnement, ce qui revient à réaliser une typologie;
- l'estimation de paramètres synthétiques dynamiques et/ou écologiques.

L'analyse systémique préconisée présente l'avantage évident d'être synthétique, comparativement aux appréciations antérieures compartimentées. Mais une contrainte de taille persiste dans ce schéma, c'est l'impact du poids économique et social sur la préservation de la ressource biologique, qu'il s'agisse des aides à la profession, de l'aménagement littoral ou de la pression incontrôlable des marchés sur des espèces cibles. Le niveau d'investissement lourd et la relative rigidité du secteur, objectivement incapable de se redéployer rapidement lorsqu'une ressource montre des signes d'épuisement, ont abouti à une approche bio-économique intégrant divers aspects sociaux comme les risques de chômage en cas de désarmement d'une flottille, ou les modalités d'amortissement des investissements.

Plus fine, l'approche basée sur les niveaux trophiques (Caddy *et al.*, 1998; FAO, 1999; Kline and Pauly, 1998; Pauly *et al.*, 1998a) semble plus séduisante. Le nombre d'informations indispensables pour émettre un avis sur l'état de l'écosystème en limite cependant l'utilisation, d'autant plus que la méthode présente aussi l'inconvénient d'imputer toute variation d'abondance à la pêche, alors que d'autres causes auront des effets qu'il est impératif d'estimer, même s'ils restent encore mal connus, notamment:

- l'activité anthropique, surtout pour les espèces côtières;
- la destruction mécanique et chimique des frayères et nurseries, hypothéquant les recrutements ultérieurs;
- les fluctuations climato-météorologiques à moyenne échelle.

Dans tous les cas, l'absence de données de terrain en qualité et quantité suffisantes, onéreuses en termes d'acquisition, conduit à rechercher des indices synthétiques macroscopiques, faciles à obtenir, tels que:

- la taille moyenne des captures, aisément et fréquemment réalisable sur les débarquements;
- la combinaison de la longueur asymptotique l¥ et du coefficient de catabolisme K calculé à partir du modèle de croissance linéaire de von Bertalanffy;
- des coefficients de mortalité F.

Ce dernier paramètre peut cependant poser des problèmes supplémentaires; les espèces peuvent avoir un comportement spécifique selon leur âge, et n'être accessibles à une technique de capture que pour certaines classes d'âge dans des conditions particulières; les mortalités présenteraient par conséquent, un biais difficilement corrigible. C'est le cas des grands pélagiques comme le thon et l'espadon (Bard *et al.*, 1988) pour lesquels d'ailleurs l'étude menée depuis 1989 a permis d'aborder une approche classique (Beverton and Holt, 1957; Mesnil, 1988; Sparre and Venema, 1996), complétée par une approche trophique à partir des résultats du régime alimentaire, actuellement en cours de traitement.

Les estimations fournies par les différentes méthodologies convergent souvent pour aboutir à une vision synthétique sans laquelle il est difficile de dégager des mesures cohérentes (Grandperrin *et al.*, 1976). Les décisions de préservation de la ressource sont à la base du choix technologique. La diminution mondiale des captures a entraîné une réorientation de la technologie vers les pêches douces aux engins dormants qui présentent le double avantage d'être sélectifs et d'utiliser une main d'œuvre abondante, critère essentiel en cette période actuelle de tendance généralisée au chômage. Les limites propres de cette démarche sont liées au seuil de rentabilité des entreprises. La multiplicité des enjeux a conduit à la mise en place de groupes de réflexion pour la recherche d'un consensus visant à protéger le producteur confronté aux échéances financières, le consommateur limité par son pouvoir d'achat et le simple citoyen, soucieux de préserver son environnement social et surtout écologique, chaque rôle pouvant être dévolu à un même protagoniste.

Si les risques de dégradation de l'écosystème marin sont réels, surtout pour la mer Méditerranée très vulnérable, le potentiel halieutique reste mal connu et doit être valorisé. Mais un tel objectif ne peut se concevoir que sous certaines conditions qui ne sont pas encore réunies, avec en particulier:

- le renforcement des aspects scientifiques et de la recherche en tant qu'argument de négociation;
- la coopération entre les scientifiques des pays riverains;
- la multiplication des associations de protection de la mer.

# "Pandora's Box" in fisheries : is there a link between economy and ecology ?

**M. L. Coelho**

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There is currently an agreement among fisheries scientists that the objective of fisheries management has become increasingly difficult to meet, particularly in the context of sustainable exploitation using the precautionary approach. The present "global" crisis in fisheries and the environment is not only related to concerns in over-exploitation of resources, but also to the impacts of fishing on the ecosystem and in various socio-economic domains (Pauly *et al.*, 1998a; Coelho, 1999).

The formal aspects of fisheries management, translated by the economy of the fisheries, have corresponded to extremely limited representations of the "creative" processes involved in fisheries ecology. The dialogue between fisheries research (ecology) and fisheries management (economy) has produced a sort of dilemma. By definition, the two fields, eco-nomus and eco-logus, address two different views of the world (Vestergaard, 1998). While ecologists find the changes in the ecosystem difficult to interpret, as they relate to very complex marine ecological processes (a sort of endless, "Pandora Box", creation process), the economists question whether such changes are measurable, consequently, relevant to the sustainability of the fishery.

A link to these opposing views is the new paradigm of ecosystem management which poses a collaborative process (Papaconstantinou et Labropoulou, this volume) where the use of common knowledge and information can be used to identify indicators, which make a useful link between both economy and ecology.

In the past, fisheries research has mainly concentrated on simplification, considering only single species in classic prediction models. These methods have ignored the existing complexity involved in marine and fisheries ecology. In parallel, economy has not been able to clearly demonstrate (Catanzano, this volume) that an economic value exists in respects to present and future human activities, when the natural environment is exploited carefully. At present, the ecosystem approach to fisheries management proposes to identify ecological indicators such as the "fisheries trophic level" (Pauly *et al.*, 1998a) and this may be a valuable working link between the two views.

## AN APPLIED EXERCISE

Here, I shall summarise fisheries data for the Portuguese coastal area (Area IXa of ICES) on the basis of multi-species and trophic level considerations, as a preliminary approach to the available Portuguese statistical data using an ecosystem perspective.

In ecosystem terms, the Portuguese continental shelf area represents a complex of inter-related species coexisting in a variable environment. A typical Portuguese fishery catch consists of more than about 20 species. The fish biomass of the Portuguese coastal upwelling system is dominated by one species of sardine (*Sardina pilchardus*). The system appears to have a typical community configuration, with many species at the bottom, many at the top, but constricted to a very few dominant species at midlevel, similarly to the "wasp-waist" ecosystems (Bakun, 1996). Research on both pelagic and demersal species has mainly targeted species of economic importance (sardine, hake, horse mackerel, blue whiting, megrims, monkfish and Norway lobster (Cardador *et al.*, 1997).

Unfortunately, important information is missing on the species composition of the fisheries landings. The existence of several species of relatively (to demersals) lower trophic levels are known but are not recorded (for example, fish of the genera Macroramphosus and Capros, the crab Polybius, and the squid Alloteuthis). These species may occur in great abundance, as was observed during the 70s, 80s and 90s (pers. observ.). But, fisheries statistics do not take into account such occurrences as a result of their low economic importance. Research surveys have reported these species as low (?) "abundant" but occurring frequently.

Attempts to define subsystems in this complex were provided more than ten years ago (Gomes, 1987 and Serrão, 1989) based on demersal cruises (1985-1988), but the resulting information was not incorporated in current single species based fisheries management. In these studies, relatively homogenous faunistic components were identified, as Inshore North, Offshore North, Inshore South, Offshore South and Centre.

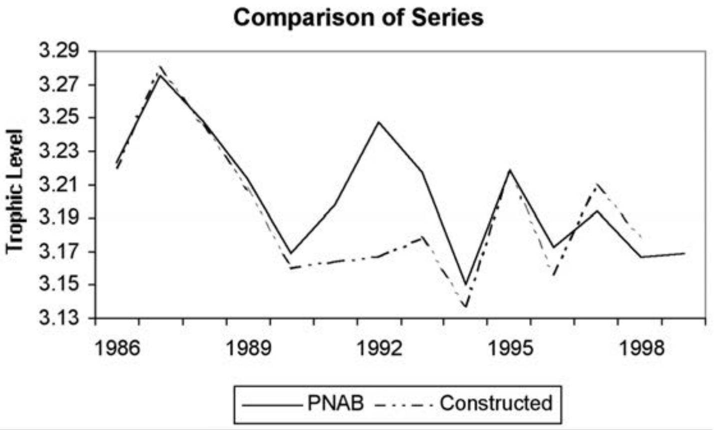

Fig.1**a**. Comparison of data from the two available time series. PNAB is considered to be of good quality, but ranging only from 1986 to 1999. The "Constructed" series was obtained from statistical publications and other sources, starting from 1927 to 1997.

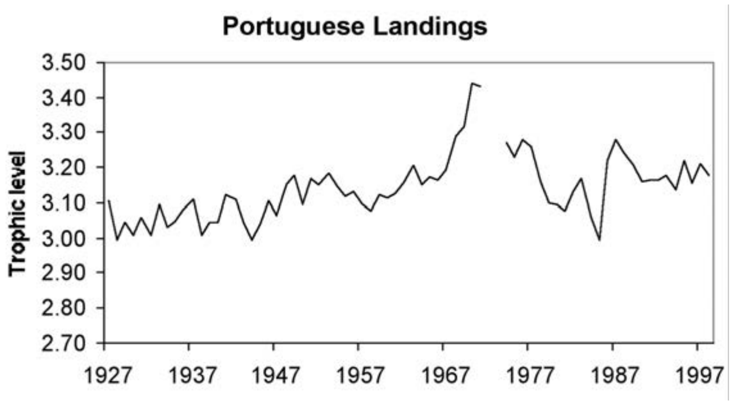

Fig.1**b**.Trophic level of Portugese landings. The trophic level of some 40 species was designated according to information obtained from Fishbase 99 and published ECOPATH models.

The present exercise is based on the officially reported Portuguese landings. Data were compiled and some 40 species were designated a trophic level, based on FishBase and published ECOPATH models. A short time series from 1986 to 1999 (PNAB, National Biological Sampling Profram of IPIMAR) is considered to be "better quality data". The other, the more extensive time series, from 1927 to 1997, should be considered cautiously. Also, a possible important impact of the Portuguese revolution (1975) on the quality of data is also to be considered.

As we can see in Figure 1, the Portuguese fisheries landings show only a very slight decrease in overall trophic level during the period 1986 to 1999, both in the case of the shorter PNAB series and the constructed series starting in 1927. However, the longer time series indicates an overall slight increase in trophic level (Fig. 1b), in parallel with great changes in the fisheries in the area. These changes could lead us to expect a decreasing trophic level and it is in fact an apparent contradiction which is difficult to explain. Data quality may be one cause, but we also had a great increase in the demersal fisheries after the decline of the pelagic fisheries, mainly after 1983.

A pronounced decrease of the traditional pelagic catches (sardine) took place after the early 60’s, and it was further followed by great changes in the demersal fisheries (Fig. 2). In the latter fisheries, a marked decrease was apparent after 1987 for the main species (the hake *Merluccius*). In parallel, increases of blue-whiting, great increases of octopus (Figure 2) and of various species of crustacean have been registered.

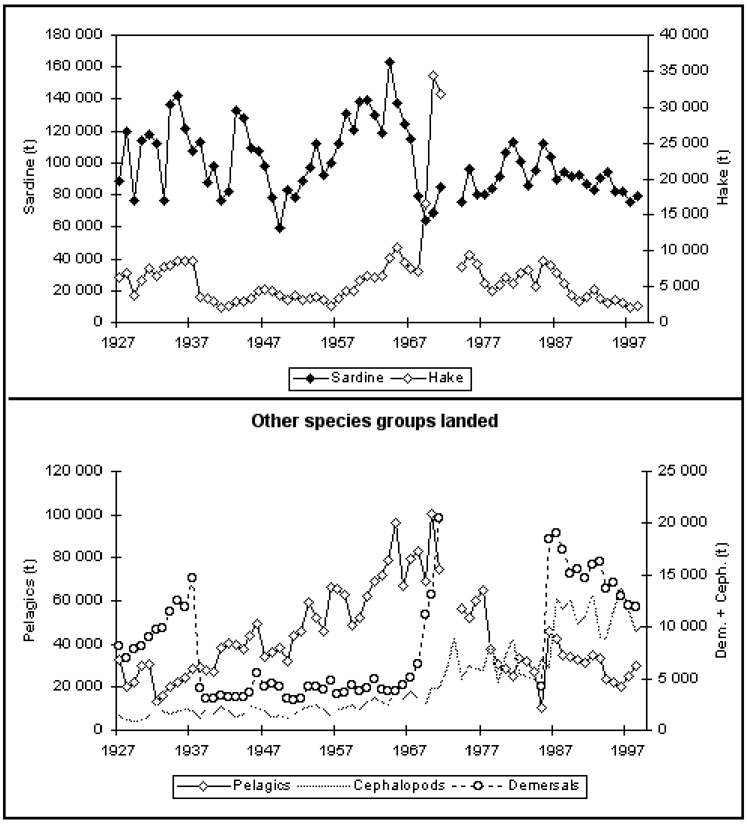

Fig. 2. Landings of the Portuguese fisheries. On top: landings of sardine and hake, which dominate the pelagic and demersal groups, respectively. Bottom: Landings of other species grouped according to pelagic, demersal and cephalopods.

Since 1983, altered conditions in the system, as those indicated by highly variable recruitments of sardine (Santos *et al.*, 1997) and modifications in the geographic distribution of spawning sardine (Straoudakis, 1999) have been detected. These alterations may have led to relatively rapid biological responses through the existence in the system of groups of apparently “responsive species” (including shorter life and opportunistic feeders species, such as cephalopods and some species of fish). For the same area and the same period, changes in the diet of hake have also been as a function of the available prey in the distributional area (Hill, 2000). This study has shown that the diet of hake appears to reflect the corresponding changes in the composition of catches.

## CONCLUDING REMARKS

This preliminary analysis indicates that the Portuguese fisheries have affected on the species composition in the system, leading to an impact on trophic level. However, due to problems with data quality it is very difficult to advance a definite ecological interpretation of this. We have to consider that statistics only take into account landed species and that research priorities have been placed on economically important species. The management of fisheries in an ecosystem context will continue to be very limited, if this situation does not change. Greater effort should be placed in giving a combined ecological/economical attribute to the fisheries systems and their components. In the future, trophic level attributes may be also very helpful to the follow-up of future measures for the rebuilding of these fisheries.

**Acknowledgements**. K. Stobberup and J. Pereira from IPIMAR have provided great collaboration with compiling statistics and analysis.

# Biological reference points and natural temporal variability in Mediterranean fish populations

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Defining biological reference points strongly depends on the management strategy (Caddy and Mahon, 1995; Rosenberg and Restrepo, 1995; Caddy, 1998). For instance, the maintenance of a given proportion of the population is the main objective for fisheries managed by quotas. In that situation, reference points based on production model, such as FMSY, or Yield/Recruit and SSB/Recruit analyses, such as F0.1 or F0.2, are more relevant than reference points based on stock-recruitment approaches, which would emphasise the subsistence of the spawning biomass and potential recruitment. However, most of the underlying models that allow to compute these reference points imply that the population reaches an equilibrium or a steady state (Gulland, 1977; Laurec and Le Guen, 1981). In other words, natural temporal variations in stock size, growth and reproduction are assumed to be low or negligible.

The stationarity assumption could be acceptable for several tropical populations, such as tropical tunas which do show stable recruitment and stable productivity over long periods (Fonteneau *et al.*, 1998). However, it is not for most of the temperate fishes, which tend to display significant variations in population size in relation to a higher variability of their environment (Longhurst, 1998). Natural long-term fluctuations in stock size (i.e., not related to fishing or human disturbances) is a well known feature which has been demonstrated for several fish populations, such as the Atlantic cod (Cushing, 1982; Dickson and Brander, 1993; Fromentin *et al.*, 1998), the Atlantic and Pacific herring, anchovy and sardine (Southward *et al.*, 1988; Baumgartner *et al.*, 1992; Alheit and Hagen, 1997) or the Pacific salmon (Beamish and Bouillon, 1993). These natural fluctuations in stock size are also likely to take place in several Mediterranean fish populations. Fromentin *et al.* (2000) reached that conclusion for the Eastern Atlantic and Mediterranean bluefin tuna from a preliminary analysis based on historical trap data. In this special case, long-term fluctuations in trap catches could be considered as a good proxy of long-term trends in true abundance because: (i) trap is a passive gear which was submitted to low modification until the early 20th century, (ii) traps were settled at the same locations for centuries, (iii) the main spawning areas are located in small coastal locations (i.e. the Balearic islands and the Messina strait) and bluefin tuna used to do the same genetic migration along the coasts to reach these points in May-June, and (iv) long-term variations in secular traps data of different countries (Portugal, Tunisia, Sicily and Sardinia) are significantly synchronous. The authors put forward that the long-term fluctuations in Mediterranean bluefin tuna could result from environmental forcing and/or biotic processes leading to resonant effects (see Bjørnstad *et al.*, 1999; a more extensive study is going on to validate these first results and to investigate the possible causes).

Temporal variability in stock size (or growth, reproduction and survival) actually refers to what is commonly named "objective uncertainties", i.e., uncertainties resulting from the underlying variability in stochastic processes. These could be opposed to the "subjective uncertainties", which mainly relate to an incomplete knowledge of the system or a mis-operating of the management. Although there is now an agreement to recognize that objective uncertainties play an important role within population dynamics, this type of uncertainty is still largely neglected (for various reasons) within stock assessment or various management. For instance, the Atlantic bluefin tuna TAC estimated by the BFT ICCAT working group is established on forecasts based on VPA estimates of SSB and recruitment, yield-per-recruit analysis and Beverton and Holt stock recruitment relationship. When computing stock assessment, establishing TAC or defining biological reference points, it appears important, for some species such as the bluefin tuna, to use more modern tools that allow to account for natural variations in stock size (*e.g.*, non-equilibrium production models), or to use the potential of the simulation modelling to test effects of long-term fluctuations on various estimates (see Kell *et al.*, 2000). What would be, for instance, the relevance of biological reference points based on steady state models for a population exhibiting large natural fluctuations, such as bluefin tuna? (Fig. 1).

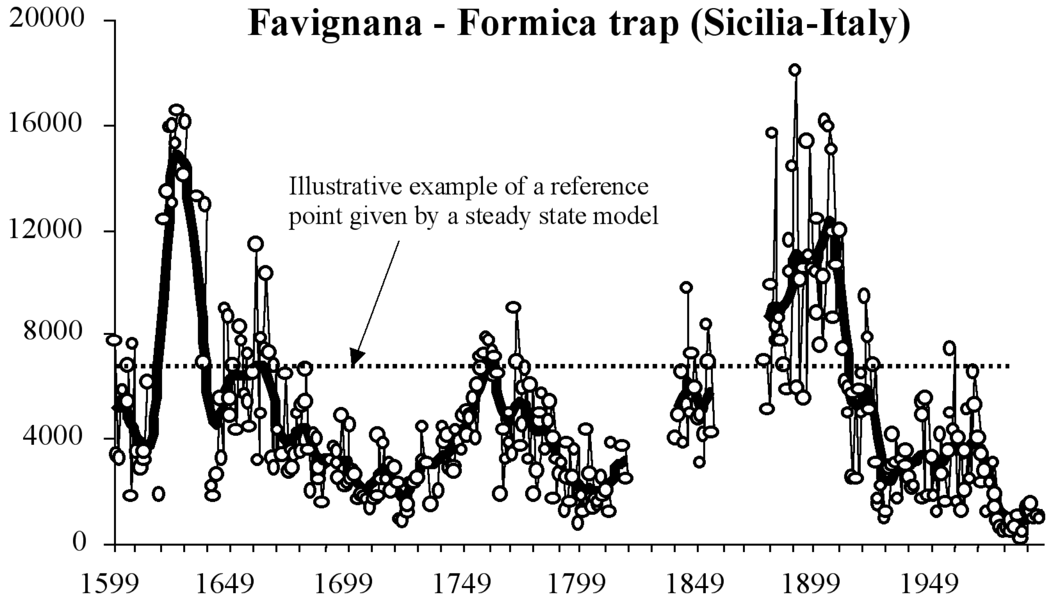

Fig. 1: Pseudo-cyclic long-term fluctuations in catches (number of fish) of a Sicilian trap from 1599 to 1996. Circles: raw data; bold line: Kernel smoothing

In comparison to Northern scientists working on cod or herring for instance, fish biologists working on the Mediterranean Sea have further to face a data poor situation and, generally, a lower knowledge of the basic biological and ecological features of fish populations. Even if we knew a Mediterranean fish population as well as the Baltic cod for instance (which would take at least 10 to 20 years of heavy work and effort), it is unlikely that we could manage efficiently that fish population, because the Mediterranean ecosystem is far more complex than the Northern ones. The Mediterranean displays rather high diversity, low dominance and complex trophic interactions, so that management procedures can rarely be stated at a single species level (possible exceptions being the marine mammals and the large pelagic fishes). The numerical tools that could define accurately biological reference points within a data poor and a multi-species situation do not exist for the moment but promising ways are emerging from current modelling approaches, such as the simulation modelling and the trophic models Ecopath-Ecosim-Ecospace (see the executive summary of the present volume, and Pauly *et al.*, 2000b). It is up to the scientists working on Mediterranean and tropical fisheries to challenge these issues and to bring original solutions.

# Influence of fishing on the trophic structure of fish assemblages in Mediterranean seagrass beds

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Mediterranean fisheries are characterised by the highly multispecific nature of the catches and the socio-economic importance of the "small-scale" fleets (Farrugio *et al.*, 1993). *Posidonia oceanica* seagrass beds represent one of the fishing grounds for artisanal fisheries and recreational fishing activities. This characteristic and widespread ecosystem in the Mediterranean extends from shallow waters down to 40 metres in oligotrophic areas (Boudouresque *et al.*, 1994). Seagrass beds constitute a spatially complex ecosytem which provides numerous habitats and food resources to a diversified fish fauna which exhibits strong spatial and temporal variations (Harmelin-Vivien, 1982). The most important fish families, according to the number of species or individuals, are the Labridae, Sparidae, Scorpaenidae, Serranidae, Centracanthidae and Pomacentridae.

## TROPHIC STRUCTURE OF FISH ASSEMBLAGES IN SEAGRASS BEDS

The perception one can have of the trophic structure of seagrass bed fish assemblages depends on the active feeding period of the various species and the technique used to sample them (Harmelin-Vivien and Francour, 1992). All the data presented here are issued from trawlings performed by day and night with the same gear and following similar sampling strategies in various areas of France and Italy. Four main trophic categories are distinguished following Bell and Harmelin-Vivien (1983): (1) the planktivores (or microcarnivores) mainly represented by *Chromis chromis* (Pomacentridae), *Spicara maena* and *S. smaris* (Centracanthidae), *Boops boops* and *Oblada melanura* (Sparidae), and the Atherinidae, (2) the Type 1 mesocarnivores, entirely composed by the Labridae as they are the main mollusc feeders in this habitat, (3) the Type 2 mesocarnivores, represented by the other Sparidae, the Mullidae, Syngnathidae, Gobiidae, Blenniidae and the flatfishes, and (4) the macrocarnivores which include the Congridae, Muraenidae, Ophidiidae, Gadidae, Scorpaenidae and Serranidae. The herbivores, represented by the sparid *Sarpa salpa*, are not present in our samples.

By day, the trophic organisation of the sampled fish assemblage is dominated by Type 1 mesocarnivores (47% by weight) and macrocarnivores (38% by weight). By night, planktivores represent the most important category (43% by weight), followed by macrocarnivores (34% by weight). When averaging day and night samples, the mean importance in weight of the different trophic categories is as follows: macrocarnivores (36.5%), Type 1 mesocarnivores (31.8%), planktivores (25%) and Type 2 mesocarnivores (6.7%).

## INFLUENCE OF FISHING ON THE TROPHIC STRUCTURE

Data were collected in areas submitted to high fishing effort in France (Marseille, Carry-le-Rouet, La Ciotat) and Italy (Ischia), and in some French marine protected areas (Port-Cros, Corsica). The most obvious effect of fishing is a significant decrease (t-test, $p<0.01$) in the mean weight of individuals in fished areas compared to protected areas (Table 1). The density and biomass are also lower in fished seagrass beds, but differences are not significant due to the high variance of data.

Table 1. Differences in mean (± standard deviation) density, mean biomass and mean individual fish weight for seagrass bed fish assemblages in marine reserves and in areas submitted to fishing.

| | **Marine reserves** | **Fished areas** | **p** |
|---|---|---|---|
| Mean density (ind.100 $m^{-2}$) | 20.01 (20.51) | 14.84 (11.08) | n.s. |
| Mean biomass (g.100 $m^{-2}$) | 331.08 (330.60) | 180.71 (147.99) | n.s. |
| Mean individual fish weight | 17.58 (3.11) | 11.87 (3.83) | ** |

p = probability; ** = difference significant at $p<0.01$; n.s. = non significant difference.

The mean trophic structure (day and night data combined) of the fish assemblage significantly differs in fished seagrass beds compared to protected areas (Figure 1). The importance in weight of macrocarnivores significantly decreases in fished areas compared to protected ones (24.4% ± 13.3 vs 49.5% ± 16.5, $p<0.05$). The relative importance of Type 2 mesocarnivores significantly increases in fished areas but remains low (8.9% ± 4.4 vs 3.6% ± 2.7, $p<0.05$). Type 1 mesocarnivores quite double in weight percentage in fished areas, but this increase is not statistically significant although not far from the significant level (40.7% ± 25.4 vs 20.5% ± 13.9, $p=0.08$). The relative importance in weight of the planktivores does not exhibit any change according to the fishing pressure (25.9% ± 27.4 in fished areas vs 26.4% ± 23.7 in protected areas).

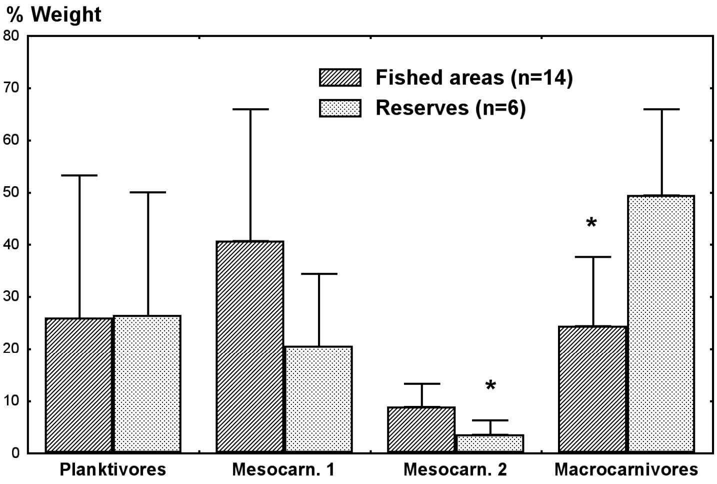

Fig. 1. Modification of the trophic structure of fish assemblages in Mediterranean seagrass beds due to fishing (* indicates a significant difference between marine reserves and fished areas for the trophic category concerned).

## INCIDENCE ON THE MEAN TROPHIC LEVEL OF FISH ASSEMBLAGES

A mean trophic level (TL) was ascribed to each fish species collected in seagrass beds according to their diets using data in FishBase (Froese and Pauly, 1998) or $\delta^{15}N$ data (Pinnegar and Polunin, 2000; Polunin and Pinnegar, this volume). A mean TL was then calculated for each of the four trophic categories differentiated. TL reaches 3.2 (±0.2) for planktivores (n=6), 3.3 (±0.1) for Type 1 mesocarnivores (n=10), 3.2 (±0.2) for Type 2 mesocarnivores (n=20) and 3.9 (±0.4) for macrocarnivores (n=9). Difference in weight percentages of the four trophic categories of fish inside and outside marine reserves results in a decrease of 0.16 of the mean TL of the fish assemblage from marine reserves (mean TLR=3.57) to areas submitted to fishing (mean TLF=3.41).

## DISCUSSION

The trophic structure of fish assemblages in Mediterranean seagrass beds is modified by fishing with a decrease in top predators (macrocarnivores), feeding mainly on fish and large crustaceans, and an increase in mesocarnivores that occupy a lower level in the food webs (Pinnegar and Polunin, 2000). Various studies have demonstrated that the removal of important predators can result in trophic cascades (Sala *et al.*, 1998) with various effects on marine ecosystems, including an increase in production at lower trophic level (Parsons, 1992). The increase in Type1 mesocarnivores (labrids) in fished areas may be due to a release in predation pressure by the macrocarnivores (scorpaenids and serranids) that are more susceptible to fishing. The decline in abundance of large predators as a response to high fishing activities is a general pattern, and Pauly *et al.* (1998a, 2000a) and Pauly and Palomares(this volume) demonstrated that the mean trophic level of fisheries landings have declined in the last decades. As evidenced here, the mean trophic level of fish assemblages in Mediterranean seagrass beds is lower in fished areas than in marine reserves. Thus, marine protected areas (MPAs) can be used as a functional tool for the recovery of higher trophic level in fish assemblages.

# Fishing effects on trophic structure of rocky littoral fish assemblages

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Fishing can have important effects on fish populations, reducing their abundances and potentially changing community structure. These changes are direct (*e.g.* decrease of fish biomass, shift to smaller individuals), and indirect (*e.g.* trophic cascades) (Sala *et al.*, 1998). Additionally, a clear consequence of the intense fishing activity can be a shift in the food web of the zone. Both direct and indirect effects are important, although the latter is usually quite difficult to evaluate, and few studies provide solid data on this topic.

The Mediterranean Sea is an oligotrophic system. Littoral communities are highly diverse, with a large proportion of small species, including fishes and other organisms. These features are nearer to a tropical sea than to a temperate water system. Littoral communities show a high level of conductance, with predators, in general, not too specialized, and originating a diffuse predation scheme. These characteristics can create much difficulty in the quantification of the feeding relationships between organisms. Therefore, the introduction of trophic parameters in multispecific models of the Mediterranean fisheries is not easy and with dubious results.

Obviously, fishing activities shift the ecosystem to smaller individuals and, in some cases, to small species (Macpherson *et al.*, 2000). However, an indirect way to identify gaps and to assess the fisheries status could be knowing “the initial point”, that is to say, how large was the total biomass, and how the trophic relationships affect the ecosystem before the intense fishing activities. Without a comparative scenario, multispecific models, and the uselfulness of trophic relationships will remain very speculative (Done and Reichelt, 1998). A good opportunity to establish a direction for future ecological and fisheries research is the use of protected areas or low fished areas as “the initial point” (Christensen and Pauly, 1998).

## INFLUENCE OF FISHING ON THE TROPHIC STRUCTURE

Data were obtained in one protected area (Medas Islands) and three littoral non-protected zones during 1997-1999. The Medas Island Marine Reserve (NW Mediterranean, Spain) was established in 1983. All fishing is prohibited around the islands, thereby affording complete protection to the rocky substratum and *Posidonia oceanica* beds of the surrounding bottoms. Both the length of time that has elapsed since the reserve’s inception and the size of the reserve (ca. 300 hectares) make this protected area an ideal site for studying fish populations in conditions that can be presumed to resemble the conditions that prevailed before the onset of intense fishing activity in recent decades. Three locations on rocky portions of the Costa Brava shoreline in the NW Mediterranean Sea were sampled: Port de la Selva, l’Estartit (opposite the Medas Islands), and Tossa, each locality being separated by a distance of roughly 50 km. The localities were chosen on the basis of morphological similarities between the bottoms and communities located there and those in the Medas Islands. Visual censuses were used to estimate the fish abundances of different species. The trophic classification by Bell and Harmelin-Vivien (1983) was adopted, including herbivores and considering the *Diplodus* group as omnivorous.

The age structure of many species is clearly modified by fishing (Figure 1) and also the general structure of the community (Figure 2). The icthyophagous (*e.g. Epinephelus marginatus*, *Dentex dentex*, *Dicentrarchus labrax*) predominated (40%) in protected areas, clearly decreasing in non-protected zones. On the contrary, the proportion of the biomass of mesocarnivores (*e.g.* Labridae) clearly increase in non-protected zones (ca. 60%), whereas in the marine reserve it is about 25% of the total biomass. Omnivorous do not show strong changes (ca. 20% in the protected area, versus 10% in the non-protected zone). Microphagous remain more or less constant (ca. 10%), whereas the low proportion of herbivores increases (2% to 10%).

These results indicate a clear change in the trophic structure of the littoral rocky fish community, and a decrease in the trophic web, in line with the findings of Pauly *et al.* (1998a)

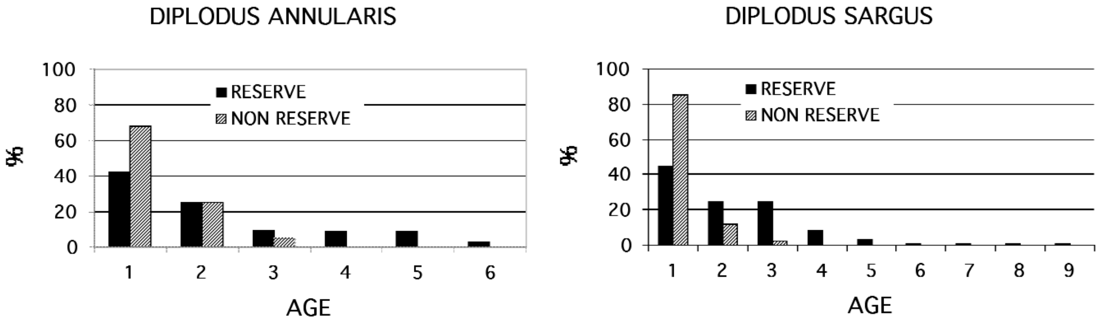

Fig. 1. Age structure of two common species (*Diplodus annularis* and *D. sargus*) taken in the protected (Medas Islands) and unprotected areas (Port de la Selva, l’Estartit and Tossa).

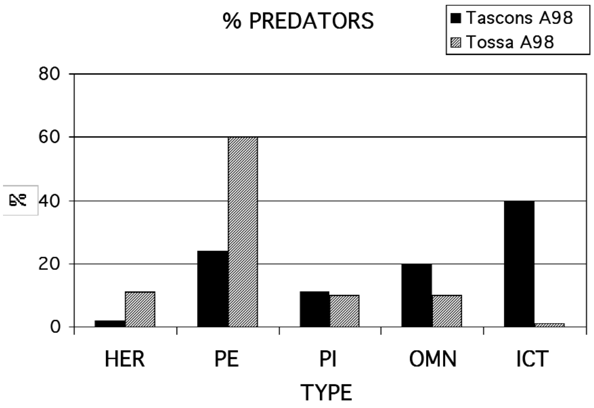

Fig. 2. Percentage contribution to the total biomass by the different trophic groupings in Mediterranean rocky zones, protected (Tascons, Medes Islands) and non protected (Tossa) areas. HER = herbivores, PE = mesocarnivores, PI = microcarnivores, OMN = omnivores, ICT = ictyophagous.

# Tuna purse seine fisheries and their offshore pelagic ecosystems

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In the pelagic offshore ecosystems, tunas and associated large pelagic species (such as billfishes and sharks) are high trophic level predators which are exploited by multiple fisheries and various gears (purse seine, longline, baitboat, drift nets). The tuna fisheries are increasingly active world-wide and catches are relatively well known since these fisheries are industrial and analysed by scientists and various tuna commissions (ICCAT, IATTC, IOTC, SPC, …). However, stock assessments are concentrated on several individual stocks such as yellowfin, bluefin or skipjack. Two main fishing modes are used by the purse seine fishery: the search and the catch of free swimming tuna schools (most often big size tunas in mono-specific unassociated schools), and sets under artificial floating objects (drifting Fish Aggregating Devices), set afloat by the purse seiners. FADs concentrate tunas (most often mixed concentrations of small size tunas dominated by skipjack), but also other pelagic species associated with the FADs (Arenas *et al.*, 1999 ; Ménard *et al.*, 2000). The purse seine fisheries are also killing unknown quantities of various untargeted by-catch species, dominated by small tunas (juveniles of commercial tunas and small species such as frigate tunas and little tunas), associated large pelagic species (such as billfishes, wahoo and sharks), other fish species (Balistidae, *Sphyranea barracuda*, dolphinfish, *Elagatis bipinnulata*, Kyphosidae), and emblematic species such as turtles. By-catches are often dumped dead at sea but they can be sold in local markets (Romagny *et al.*, in press) or commercially exploited.

The available data suggests that the yearly total discards by the world tuna purse seine fisheries could be estimated at less than 100.000 tons (Fonteneau *et al.*, communication at the ICES SCOR of Montpellier, March 1999), for about 1.6 millions tons of catches by the purse seiners. This is relatively minor compared with many other fisheries (Alverson *et al.*, 1994). By-catches were probably increased in recent years, because of increased catches with drifting FADs, under which the associated fauna is more abundant.

It is important to understand the impact that the industrial-scale removal of tunas will have on the balance of the trophic system from which they are removed (Polovina, 1984; Olson and Boggs, 1986; Kitchell *et al.*, 1999). There is a growing body of evidence that changes at the top of food webs are expressed at all trophic levels in a wide variety of aquatic ecosystems. The analysis by Pauly *et al.* (1998a) demonstrates a reduction in the average trophic level for marine fishes harvested over the past decades. To go further, a better understanding of the ecosystem functioning is needed, and a direct path to this goal is to elucidate the trophic interactions of the system.

## COMPONENTS OF THE OFFSHORE PELAGIC ECOSYSTEMS

The ecological components of the offshore pelagic ecosystems are relatively simple (Fig. 1) in comparison to coastal ecosystems. But biomass of many components is quantitatively poorly estimated in most offshore areas. This lack is partly due to the difficulty to sampling such large areas using scientific cruises, to the heterogeneity of these ecosystems, and to the difficulty to obtain precise fishery data including discards and by-catches.

### Predators

A quite low specific diversity is found: large-size tunas, billfish, swordfish, sharks, and mammals are the major components of the apex predators with few other species (trophic levels 4-5). Small tunas and juvenile tunas may be classified as predators or as preys (trophic levels 3-4). Most of the knowledge available on tunas and billfish is obtained from the fisheries, and the biomass of other predators such as sharks and mammals remains difficult to estimate in most areas. The densities of these predators are quite low but patches of high biomass are frequently observed: the world tuna biomass probably ranges between 4 and 12 millions tons, and is distributed in an area of about 60 millions nautical square miles. Furthermore most of the apex predators are able to do large migrations and to transfer energy between various ecosystems (including continental shelves) with varying trophic structures. Tunas and billfish are sometimes seasonally concentrated in the coastal areas, near the continental shelves, in order to better feed on small pelagic resources (*e.g.* Young *et al.*, 1997). It has frequently been stated that tunas and billfishes are very flexible and opportunistic in their feeding habits. Although they may be opportunistic in the short-term sense, they certainly are able to adjust their feeding behaviour to the available prey in each area. It is surprising to observe their ability to find appropriate food in relatively low productive ecosystems (subtropical and equatorial areas). Intraguild predation (and cannibalism) is a common feature of diets for these fishes: distinction between adult and juvenile stages for all apex predators must be taken into account, even if diet overlaps occur between juvenile and adult forms.

### Preys

The micronekton which groups together a large diversity of species (epipelagic and mesopelagic fish, cephalopods, planktonic crustaceans), is the only source of potential food in the tropical open ocean for numerous top predators such as tunas and associated large pelagic fish, marine birds, and mammals. But knowledge on both horizontal and vertical distributions of micronekton is often missing. For forage fish (epipelagic or mesopelagic species), schooling behaviour and the occurrence of dense aggregations are a salient characteristic, because such preys have to be concentrated in order to be available to tuna predation. Some prey items could be strongly dominant in certain circumstances, such as happens in an area of the Equatorial Atlantic that has become a major FAD fishing seasonal zone for tropical tunas (Ménard *et al.*, 2000). This area is subject to a peculiar type of top predation by tunas. Small tuna biomass is supported by a very small number of forage fish species, strongly dominated by a mesopelagic fish, *Vinciguerria nimbaria*, representing 63% of the daily meal of the small tunas caught on unassociated schools, and 49% for the small tunas under FADs. This fish is often considered as a typical mesopelagic species, diving to depths of 500 m or more during the day, and migrating in the 0-90 m layer at night. Marchal and Lebourges (1996) reported that the adult population of *V. nimbaria* have a peculiar diel behaviour in the SSA, concentrating in mono-specific dense schools in the upper layers during the day where they become available to tuna predation. At night, they concentrate at or below the thermocline mixed with other fish, squids and crustaceans. *V. nimbaria* shows a short life span (6-7 months) and a maximum standard length of 55 mm. The explanation of the peculiar behaviour of the adults may be found in their feeding activity (Lebourges *et al.*, 2000), but links between tuna and micronekton remain often not clear.

## ECOSYSTEM MODELLING

Ecosystem modelling will be a powerful tool to evaluate the dynamics in the upper layers of these pelagic food webs. Steady state modelling is a typical approach and has to take into account a variety of information: (i) identification of the components of the food webs and definition of

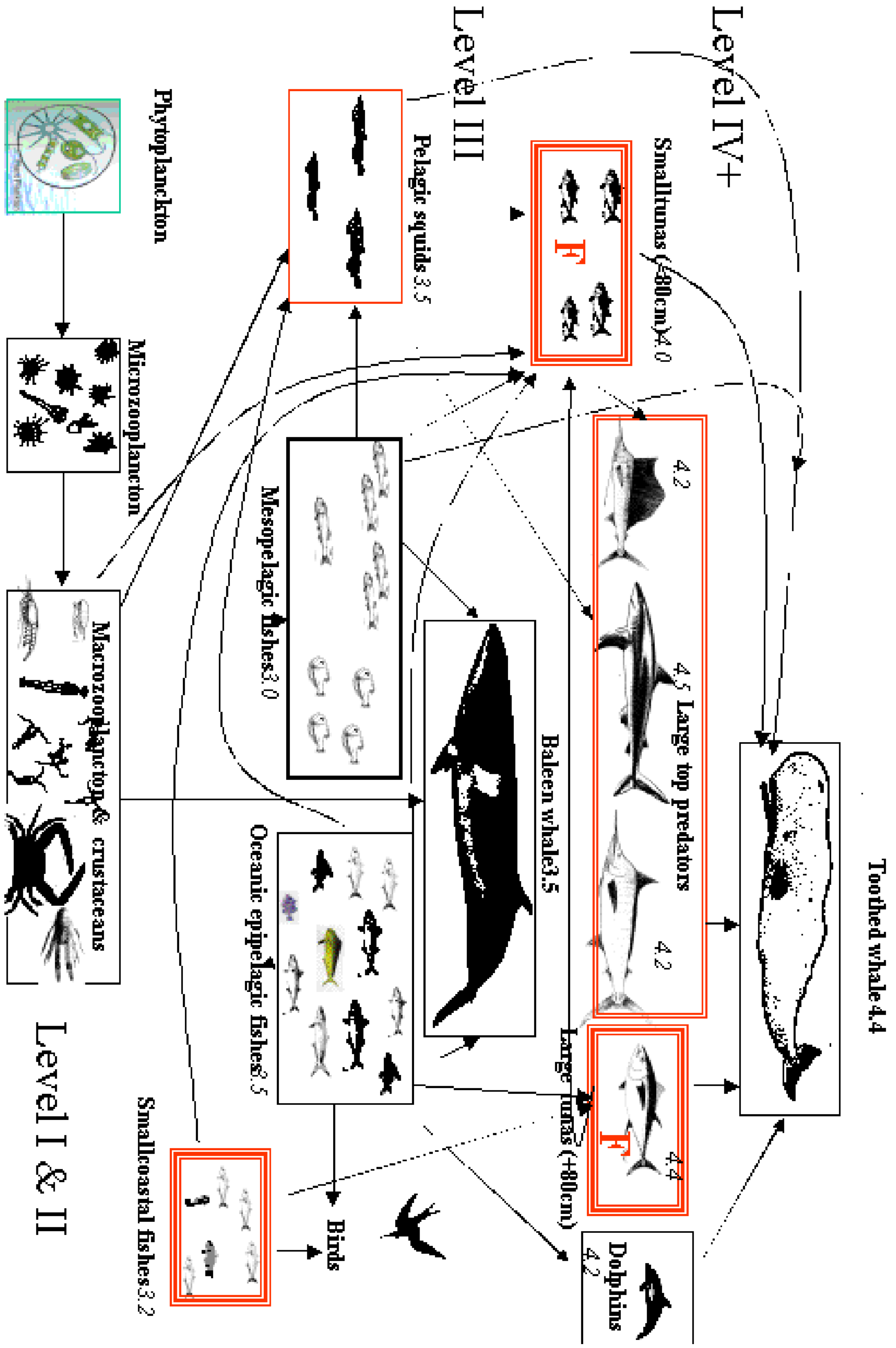

Fig. 1. Major components in the pelagic ecosystems and their relative trophic levels (F=Fishing mortality).

the functional groups; (ii) predator-prey links (diet composition); (iii) estimation of predator consumption rates or energy requirements; (iv) estimates of prey biomass and productivity; (v) other removals from the system (landings, discards, imports and exports). A mass balanced approach is developed in the software tool ECOPATH/ECOSIM (Polovina, 1984; Christensen and Pauly, 1992; Walters *et al.*, 1997). Kitchell *et al.* (1999) used it to evaluate the potential of keystone predator effects among the guild of sharks, tunas, and the billfishes at the apex of pelagic food webs in the central North Pacific ecosystem. They showed that substantial diet overlap and intraguild cannibalism (especially for sharks) appear to be major factors in such complex systems, but they did not find any clearly demonstrable keystone predator.

The pelagic ecosystem is not homogeneous, and the building of optimal frontiers is complex which requires to take the spatial dynamics of tuna populations into account (Maury, 1998). Dynamic modelling approaches should also incorporate other useful information: the variability of the environment, spatial heterogeneity of predators and preys, transition from small to large ontogenic groups of the same taxa, seasonal patterns of various parameters, recycling and diet switching, and the dynamics of the exploitation by fishing fleets. Thus, dynamic models involving coupling between environmental variability, spatial dynamics of tuna populations and exploitation by fishing fleets should be developed and extended, as in the approach of Lehodey *et al.* (1998), based on the coupling of a simple bio-geochemical model and a general circulation model allowing predictions of new primary production and biological transfer toward tuna forage.

## CONCLUSION

It is now necessary to take into account the potential effect of the fisheries on the ecological interactions between the various components of the pelagic offshore ecosystems. The definition of an ecological risk in the context of the precautionary principle is difficult due to our ability to measure objectively this risk when it concerns unexploited elements of the ecosystem (especially for emblematic species). Ecological models are tools for evaluating that complexity in an ecosystem context, and should provide a better understanding of food web dynamics, and of the potential trends of these exploited ecosystems.

A quadrennial tuna research program, THETIS (THons tropicaux: Environnement, sTratégies d’exploitation et Interactions biotiques dans les écoSystèmes hauturiers.) has been designed in IRD for the period 2001-2004. Covering both the Atlantic and Indian oceans, this program focuses on the mesoscale bio-physical processes for a more accurate appraisal of the dynamics of tuna populations at an ocean-wide dimension. Trophodynamics studies will be conducted in selected pelagic ecosystems that exhibit common features in the Atlantic and Indian oceans, in order to undertake a comparative analysis of the biological interactions between apex predators (tunas, billfishes, sharks), large cetaceans and their forage. Four types of ecosystems have been identified: (i) convergence zones where drifting FADs are found in great numbers, (ii) equatorial countercurrent ecosystems, which are spawning zones for yellowfin, (iii) areas where tuna and large marine mammals are associated, and (iv) oligotrophic ecosystems of tropical gyres, exploited mainly by longline fisheries. This research operation aims at a tentative modelling of the food web in these different ecosystems, with a subsequent comparison of the outputs in both oceans. Such models should contribute to a better assessment of the effect of tuna fishing on the pelagic biodiversity.

# The deep-sea fishery of the Balearic Islands and the trophic level of the exploited resources

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Since the 1940s, the fishing grounds off the Balearic islands have expanded towards the continental slope (Oliver, 1993), reaching a maximum depth of 800 m. The target species of the deep-sea trawling fleet are the hake *Merluccius merluccius*, red shrimp *Aristeus antennatus*, and Norway lobster *Nephrops norvegicus.* The historical data series of landings of these species show marked fluctuations (Fig.1). These might be due to changes in effort, which may be directed towards one or another target species during certain periods, depending on the level of catches obtained at that moment, and on the respective strength of recruitment (Oliver, 1993; Merella *et al.*, 1998; Carbonell *et al.*, 1999).

The deep-sea fleet exploits the upper slope species assemblages (Moranta *et al.*, 1998), with fishes representing the major proportion of the catch (73%), followed by the crustaceans which represent 16%. On average, 42% of the total catch is discarded (Moranta *et al.*, 2000). The discards decrease with depth and also show seasonal variations in their relative abundance and composition (Moranta *et al.* 2000). The fish discard is composed of undersized marketable species or small unmarketable fish, while crustaceans discard is small.

The stable carbon and nitrogen isotope composition of the components of the slope communities were analysed between 200 and 1800 m depth with material collected from two research cruises (Q1 and Q2). The main results showed some variability along the slope (Table 1).

Table 1. Mean δN data on the target species of the deep-sea fishery in the three communities on the slope and for the research cruise. nd = not determined.

| | Mean delta N data<br>Lower | <br>Middle | <br>Upper |
|---|---|---|---|
| Calanoid copepods Q2 | 6.5 | 5.5 | nd |
| Gelatinous plankton Q2 | 6.9 | 6.9 | 5.2 |
| *Merluccius* Q1 | nd | 13.9 | 9.9 |
| *Merluccius* Q2 | nd | nd | 10.8 |
| *Aristeus* Q1 | 10.9 | nd | 10.9 |
| *Aristeus* Q2 | nd | 11.2 | 11.3 |
| *Nephrops* Q1 | nd | nd | 9.2 |
| *Nephrops* Q2 | nd | nd | nd |

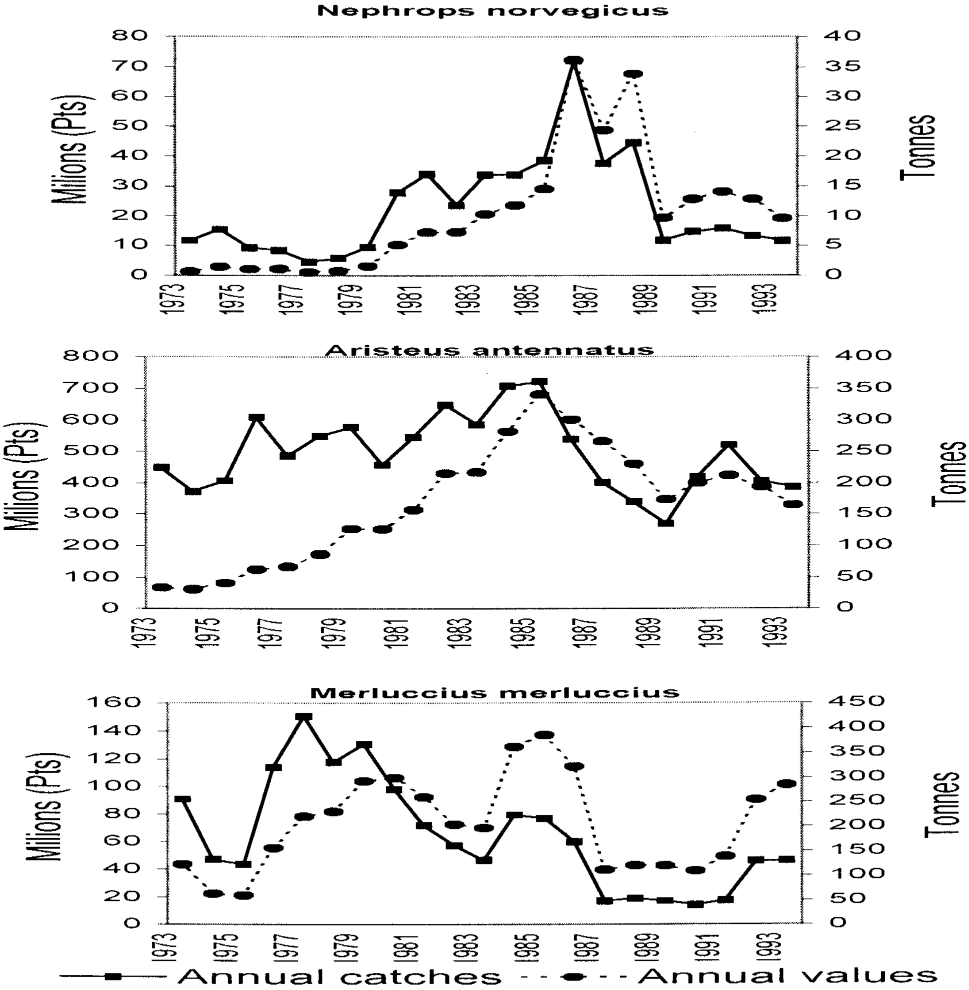

Fig.1. Historical data series of prices and landings for the target species of the deep-sea Balearic fishery.

Assuming that a) Copepods and gelatinous plankton have a trophic level (TL) equal to 2, and b) that a δN difference of 3.4 per TL, the TL values of the target species of the deep-sea trawl fishery are around 3.3-4.4 for hake, 3.2-3.7 for red shrimp and 3.1 for Norway lobster. In the calculations the reference values used are from Calanoid copepods and gelatinous plankton. The results suggest that the target species form part of short food chains.

*Aristeus antennatus* and *Nephrops norvegicus* are euryphagous predators mainly feeding on a wide variety of benthic and benthopelagic invertebrates, ranging from polychaetes to crustaceans of different groups (amphipods, isopods, mysids, euphausiids) and also molluscs or echinoderms. *N. norvegicus* also feeds on fish remains and seasonally even on gelatinous plankton, while *A. antennatus* shows a major preference for infaunal species (polychaetes or the burrowing decapod *Calocaris macandreae*). *A. antennatus* shows a specially high trophic diversity. Seasonal changes in the diet have been reported along the slope for both species (see Cartes, 1994; Cristo and Cartes, 1998; own unpubl. data), with lower seasonal changes with increasing depth in the case of the rose shrimp (Cartes, 1994). Due to the bathymetric substitution of both species around 400-500 m, their similar high sizes, and the fact that both species exploit basically similar food resources, some competition for food between these two can not be excluded, despite the distinct physical habitat occupied for both decapods.

European hake diet is comprised primarily of fish and crustaceans, but there are size-related changes in the trophic spectrum. The youngest hakes feed mainly on small crustaceans (euphasids and mysids) and small benthic fishes (gobids). As the hake grows the importance of fish in the diet increases; larger fish prey on decapod crustaceans and mainly in active natatory fish (Bozzano *et al.*, 1997).

These diet data support the isotope findings: the three target species are operating at different levels, with hake operating at a higher level than both crustaceans. Also, the crustaceans detritivory appears very limited. A future more complete analysis of the deep-sea fishery should include the important fraction of the discards on the assessment of the exploitation level of the deep-sea ecosystem.

# Mediterranean resources in relation to their sustainable management within the “Precautionary Approach to Fisheries”. Recent initiatives and proposals to fill the gaps

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The Mediterranean is an area where the implementation of the Precautionary Approach to Fisheries (PA2F) should be considered as a basis for sustainable development of fisheries (CIESM, 1999). FAO has developed guidelines for the implementation of the precautionary approach (FAO, 1996) as well as for the development and use of sustainability indicators (FAO, 1999). These indicators which present use tend to be limited to biological components of the fishery system – i.e. stock biomass (B) and fishing mortality (F) – provide an operational tool for providing advice for fisheries management. Changes in indicators over time, however, cannot be meaningfully interpreted in relation to sustainable development without considering them in relation to a reference value (Fig. 1) corresponding to the sectoral or societal objectives (or target) and ecosystem constraints (or limits). In fisheries, these reference values are conventionally called target reference points (TRPs), limit reference points (LRPs), or threshold reference points (ThRPs) and, presently, mainly concern the target stock. PA2F requires also the adoption and use of indicators and reference values to determine the areas and degrees of risk created by the various sources of uncertainty (FAO, 1996). The evolution of the implementation of the two concepts (PA2F and sustainable indicators) in fisheries has been recently, extensively reviewed (Garcia *et al.*, 1999; Garcia, 2000a, 2000b, in press; Garcia and de Leiva Moreno, 2000).

The Code of Conduct for Responsible Fisheries establishes that conservation and management measures should be based on the best scientific evidence available. In order to provide advice, the Scientific Advisory Committee (SAC) of the General Fisheries Commission for the Mediterranean (GFCM) analysed all the scientific information available in the region, produced during the period 1985-1999, concerning the eight most important demersal and small pelagic species within those management units where shared resources existed (hake, red mullet, striped red mullet, blue whiting, red shrimp, anchovy, sardine and sardinella). More than 100 evaluations were identified and analysed. Three species of large pelagics (bluefin tuna, swordfish and albacore) were also taken into consideration (ICCAT, 1999). The result of this exercise is summarised in the Table 1.

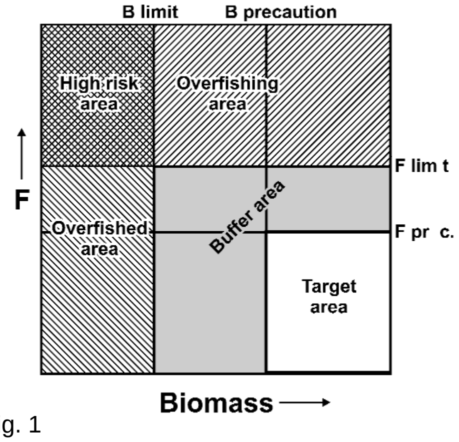

Fig. 1

Table 1

| Species | Number of assessments[1] | State of resources | | | Comments |
|---|---|---|---|---|---|
| | | Over-fished | Fully-fished | Under-fished | |
| *Merluccius merluccius* | 36 | 28 | 7 | 1 | general growth overfishing |
| *Aristeus antennatus* | 10 | 6 | 3 | 1 | |
| *Mullus barbatus* | 32 | 18 | 14 | | |
| *Mullus surmuletus* | 8 | 2 | 4 | 2 | |
| *Micromesistius poutassou* | 4 | 3 | 1 | | |
| *Engraulis encrasicolus* | 14 | 2 | 8 | | risk of recruitment overfishing |
| *Sardina pilchardus* | 7 | 2 | | 2 | three without results |
| *Sardinella aurita* | Never evaluated | | | | Unknown |
| *Thunnus thynnus* | See ICCAT 1999 | | | | Over-fished |
| *Thunnus alalunga* | Never evaluated | | | | Unknown |
| *Xiphias gladius* | See ICCAT 1999 | | | | Unknown |

[1] *The number refers to separate stocks or separate assessments of the same stocks.*

It should be noted that most of the evaluations correspond to stocks of the northern and western parts of the region. In general the assessments are based on the application of Virtual Population Analysis (VPA) or Length Cohort Analysis (LCA) together with a Yield per Recruit Analyses (Y/R) and sometimes "surplus production methods", based on short series of data and on the results of scientific surveys. Furthermore, it seems that important non-published information, relevant to stock assessment and already existing in some countries was not available at the meeting.

The analysis has evidenced a lack of fishery databases with enough coverage and reliability to allow for a correct assessment of resources. Statistically valid series exist only in a few areas and time periods corresponding to projects in which research teams have concentrated on compiling databases for the assessment of specific stocks. Sadly, when these projects terminated, the database usually met with problems of continuity and the time series were interrupted. Nonetheless, SAC detected "*a clear growth over-fishing in some selected demersal species and the risk of recruitment over-fishing of anchovy*" and it recommended "*to develop and apply management measures in order to correct these problems*". The SAC proposed to "*temporary adopt of a Harvest Control Rule (HCR), based on the Exploitation Rate (E=F/F+M) and current Spawning Stock Biomass (SSBc) preferably, or Standing Stock Biomass (Bc), both expressed as percentage of the estimated unexploited condition (SSB0, B0)*". SAC suggested also that "*recent literature reviews indicate plausible ranges for preliminary proxies of target reference points: 0.4 to 0.5 for the Exploitation Rate (E target), 20 to 30% of Virgin Spawning Stock Biomass (SSB target) and 30 to 40% of Virgin Standing Stock Biomass (B target). Both indirect (i.e. LCA and VPA) and direct methods (i.e. scientific surveys) can be used as source of information to estimate and improve the above described reference points as well as to analyse their performance*". However the use of these methods implies a high level of uncertainty or even ignorance (in the sense that relevant factors and/or elements cannot be fully assessed); this must be pointed out.

A substantial effort is urgently needed in the Mediterranean to improve the scientific advice provided to managers. The current level of stock assessment and, more generally, of fishery research for assessment purposes is not enough or not appropriate for the elaboration of proper and efficient advice to fishery managers and industry. In order to encourage the efforts to produce the needed scientific advice, the SAC, at its last meeting (Madrid, May 2000; GFCM, in press), recommended *inter alia*, to update and improve the quality and coverage of fishery data and statistics, to increase the number of assessments in the southern and eastern areas and to ensure that all the assessments will be carried out on a regular basis. SAC also drew attention on the importance of the role of the relationship between the environment and resources. Additionally the SAC indicated that the definition of geographical management units to report the indicators is essential and that homogenous socio-economic indicators in each of these management units had to be developed.

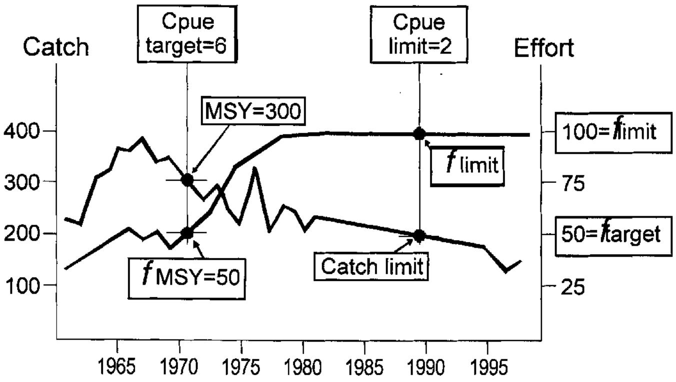

Fig. 2

While solutions are sought to correct deficiencies in the scientific advice, and in line with the PA2F, a provisional Recovery Control Rule (RCR), based on the data available and specifying the objective, planned recovery trajectories and time frame could be adopted for stocks which are obviously overfished. For that purpose, a simple set of indicators reflecting abundance and fishing capacity (such as cpue or even yields and fishing effort) could be used. The Maximum Sustainable Yield (MSY), together with the corresponding effort level could be taken as Target RP for resource recovery (in line with 1982 UNCLOS and 1995 UN Fish Stock Agreement). In Figure 2, the evolution of the deep shrimp trawl fishery of the Balearic Islands in the last 40 years, which can be considered as a typical example of one of the most common current situations in the Mediterranean fisheries, is presented to illustrate how the suggested RCR could be elaborated and used. The stock has been evaluated several times by means of surplus production models (Oliver, 1983; Carbonell and Laurونce, 2000). The results are estimations of MSY (close to 300 mt) which correspond to an optimal fishing effort of 5000 kW, since the current effort is over 10,000 kW and the current annual catch level is under 200 mt. The MSYparameters on the equilibrium curve (5,000 kW and a cpue of 60 Kg per kW) can be used to establish: 1) the TRPs at which the fishery should be placed, 2) the current values for the same parameters (10,000 kW and 20 Kg per kW) as LRPs that should not be exceeded (Figure 3)

Certainly, it is necessary to acknowledge the high risk inherent of this proposal, due not only to the weakness of the basic data used, but also to the uncertainty of the degree of reversibility of the present (overfishing) situation. The economic impact (i.e. the costs of the process) also has to be taken into account. Figure 4 provides a theoretical view of the trajectory to be followed to reduce the fishing effort. The theoretical evolution of yields in case the effort were increased is also indicated.

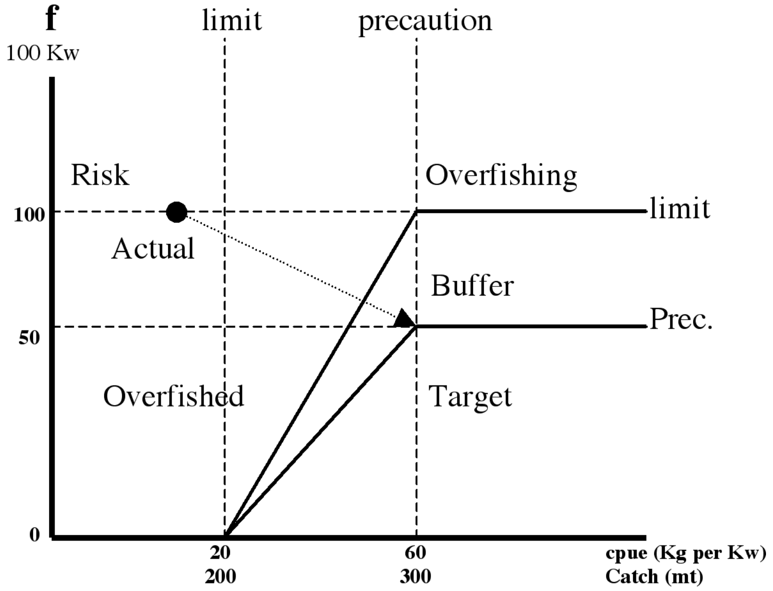

Fig. 3

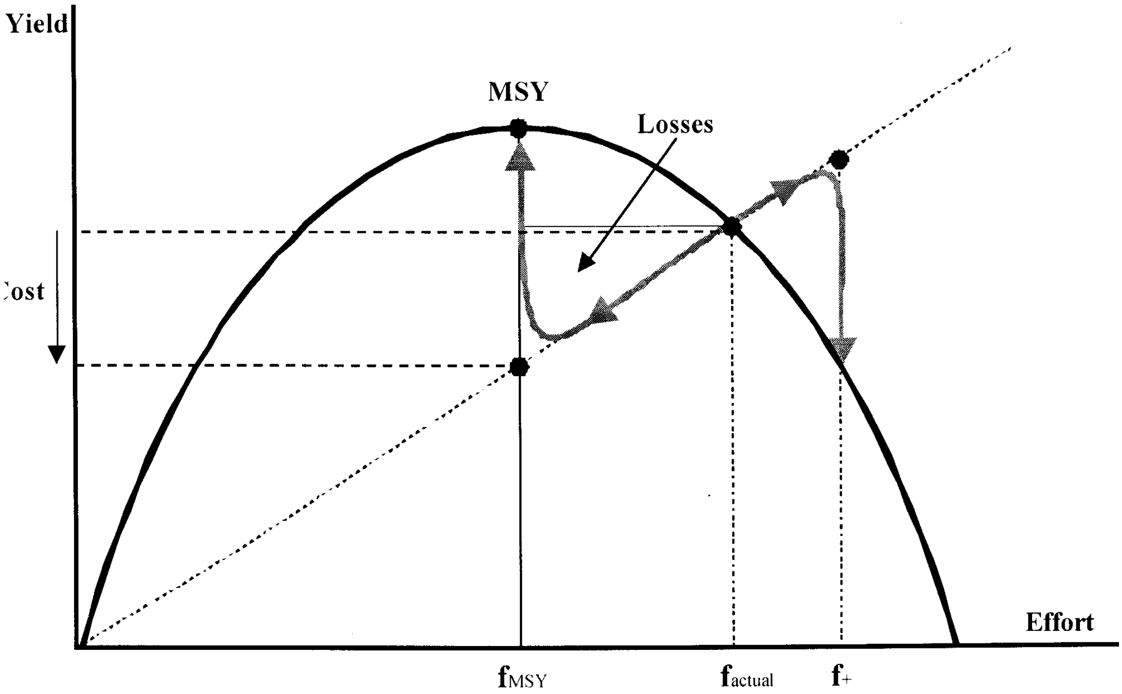

Figure 4

In any case, considering the scientific data available in the region and the low probability to improve its quality in the medium term, the suggested RCR can represent an option to the application of the PA2F in one of the regions where it seems particularly urgent to do so. Naturally, the approach would have to be fishery-specific, based on a careful analysis of the current situations. Furthermore additional elements like the fact that technological progress increases F without changing nominal effort should be considered as well as the possibility of combining a direct reduction of F trough fishing capacity – reduction measures combined with other technical measures such as No Take Zones (NTZ) to accelerate, where appropriate, the recovery of the fishery.

# Ecosystem management and sustainable fisheries : implication on marine food webs

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## INTRODUCTION

Ecosystem management has been offered as a new paradigm for resource management in response to the perception that the old paradigms are not adequate to deal with changing human values and complexity of today's resource issues. Ecosystem management is a collaborative process. Optimum sustainable yield generally replaced maximum sustainable yield as a goal for fisheries managers in the 1970s (Roedel, 1975). Optimum sustainable yield differed from its predecessor by recognizing the need for sustained ecological, social, and economic values of fish stocks. The definitions of optimum sustainable yield and ecosystem management are strikingly similar. Despite the shift to optimum sustainable yield, which some assert replaced defensible quantitative management goals of maximum sustainable yield with subjective management goals, many fish stocks continue to be overfished, some to the point where restoration is doubtful. Continued overfishing of marine stocks, growing lists of imperiled and extinct species, and failure of law makers to recognise that wetlands and clean waters are necessary for more than just cryptic aquatic creatures are testimony to the complexity and magnitude of resource management issues.

Concerning the fisheries management, the days of managing fish and only fish are gone. Fisheries managers are learning to use economic value to solicit support for fisheries conservation programs, and they have developed strong allies in the fisheries industry. However, responding to the desires of fishermen and the perceived need to develop more and better-equipped gears has challenged managers with ecological realities such as carrying capacity, tropic status, food web, etc.

Fisheries ecologists have long debated the importance of trophic interactions in determining distributions and abundances of organisms. Those who agree that trophic interactions are important still debate whether the primary control is by resources (bottom-up forces) or predators (top-down forces). According to the bottom-up view, organisms on each trophic level are food limited. The top-down view holds that organisms at the top of food chains are food limited, and at successive lower levels, they are alternatively predator, then food limited.

The relative efficacy of top-down *vs.* bottom-up forces in food webs will depend in part on the efficiency with which consumers can exploit their prey. Interactions among consumers, between consumers and resources and between nonadjacent trophic levels can affect consumer efficiency and thereby modify top-down forces in food webs. A recent theoretical controversy has set into relief different assumptions in multi-trophic level models about behavioural interactions among predators and about time scales of prey attack *vs*. predator population response.

## EFFECTS OF FISHING ON THE ECOSYSTEM

Fishing is one of many natural and anthropogenic disturbances that occur in oceans and on essential fish habitats. Pickett and White (1985) define disturbance as any discrete event in time that disrupts an ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment. Disturbance include natural phenomena such as waves, currents, predation, and such as human activities as dredging of harbours, runoff from alteration or land use patterns, and fishing.

Fishing disturbs the benthic community, directly affecting survival of young fish, but it may have an ecosystem level impact as well, *e.g.*, at levels significantly removed from the immediate ecological process supporting the production of fish. Therefore, the linkage to fish production is not readily apparent but may be equally important when considered at the appropriate scale. Nutrient regeneration and primary production have been shown to be affected by fishing, and such work highlights the need to understand these effects at the scale of the fishing ground.

Recovery of the populations of benthic primary producers following a disturbance, like the recovery of the benthic and megabenthic animals, is not well studied. On the continental shelf, trawling has been shown to change sediment grain size characteristics and the suspended particulate load, and to alter sediment transport processes (Churchill, 1989; Pilskaln *et al.*, 1998). Researchers speculate that this could shift the primary production and allied food web to favour harvested species, but it also may stimulate harmful algal blooms.

### Interaction between trawling and benthic communities

The enhancement of a particular trophic level's food supply has been documented for macrobenthic species as a direct result of current fishing practices. Animals damaged but not landed by trawling, as well as the discarding of non-commercial species, all contribute to the food web. There are numerous reports of fish and invertebrates scavenging in trawl tracks, and anecdotal reports suggest that fishers routinely re-fish areas to enhance their landings (Caddy, 1973; Evans *et al.*, 1996). The passage of a trawl across the seabed leads to the direct mortality, or indirect mortality through subsequent predation, of some benthic species. Bottom trawling leads to the death, injury or exposure of benthic fauna, thus creating a potential source of food for predators and scavengers. The speed with which scavengers show up in trawled areas is a function of the mobility of the animals and their behaviour. Fish often are the first animals on the scene, followed by slower-moving invertebrates such as starfish and whelks. Studies based on stomach content analysis further show, that scavengers will eat more than their conspecifics collected from untrawled control sites. They also may feed on animals that were not damaged but only displaced during trawling or animals that moved into an area to prey on trawl-damaged individuals. Increased consumption may put the animals at an energetic advantage, resulting in faster growth or higher reproductive potential. The benthic community is not the only beneficiary from trawling. The bird population has clearly been showed to depend on fish discards and offal from processing fish at sea. However, scavenging behaviour can not be easily distinguished and counted in real food webs, suggesting that trophic levels may as abstractions have sufficient correspondence to reality to be useful dependent variables in comparisons of trophic structure along productivity gradients.

### Interaction between fishing and predators

An indirect effect of fishing on food webs is created by the commercial harvest of keystone predators. Removal of herbivorous fish from coral reefs, for example, can shift the system from coral-invertebrate to algal dominated communities. Similarly, differences in the abundance of cancrid crabs for the inshore *versus* offshore regions of the Gulf of Maine have been attributed to intense harvest of Atlantic cod, a regionally important keystone predator and its prey are not always clearly examples. Experimental studies of a variety of aquatic systems indicate that even complex, highly interconnected webs can respond to perturbations of higher trophic levels with chain-like dynamics. Chains of strong trophic links in communities are revealed when removing top predators triggers trophic cascade. Communities that exhibits trophic cascades have at least one species or guild per trophic level with sufficiently strong potential effects on their resources in the next lower trophic level to produce chain-like, rather than indeterminate web-like responses following perturbations of higher trophic levels (Power, 1992).

Other processes such a climate change also can bring about food web alterations. Understanding these cause-and-effect relationships is equally as important to understanding the immediate biological interactions resulting from human-induced disturbance in the benthic communities. From a fisheries management perspective, food web shifts from whatever cause are serious and potentially irreversible, so precautionary management approaches have to be implemented in light of our incomplete knowledge of benthic community dynamics.

### Interaction between ecosystems and by-catch or discard

Food web and ecosystem effects of by catch or discard are even more subtle; by-catch or discard can alter the relative abundance of species and so change trophic interactions. If low-valued discard species taken along with fishery targets survive, such a scenario may contribute to a change in species dominance within heavily exploited ecosystems. Highly species-selective culls of the catch, combined with discard survival, may damage the productivity potential of ecosystems more than moderate exploitation of species communities, landing all species caught.

Although the effects of by-catch or discard on food webs have not yet been evaluated, there is a good evidence that directed exploitation of fishes or other “strong interactors” can lead to major food-web changes in marine systems (Roberts, 1995; Fogarty and Murawski, 1998). In the Northeastern United States, both Atlantic mackerel (*Scomber scombrus*) and Atlantic herring (*Clupea harengus*) declined due to overfishing, followed by increases in less-valuable sand eel (Ammodytes spp.) populations (Fogarty and Murawski, 1998). This situation has been reversed, but only after two decades of very low fishing on mackerel and herring stocks (Sherman *et al.*, 1994; Fogarty and Murawski, 1998). Heavy harvests of whales in the Antarctic led to reduced competition for krill (*Euphausia superba*), which allowed population increases in other krill feeders, including crabeater seals (*Lobodon carcinophagus*) and seabirds (May *et al.*, 1979); it also allowed the development of a krill fishery. Removal of large sharks off of South Africa to protect human swimmers led to increases in their prey (small sharks) and to reductions in teleost fishes (the prey of small sharks), reducing fishery yields (van der Elst, 1979).

Community impacts of dead discards depend greatly on the magnitude of such mortalities, their distributions in time and space, and the ability of the ecosystem to assimilate them. Direct predation of fishery discards is often observed. To the extent that such discards represent food resources otherwise unavailable to predator or scavenger populations, such mortalities represent a prey subsidy. If such subsidies are a significant component of predator diets, they may result in higher predator-scavenger populations, and/or alterations in predator distribution patterns and vital rates. Fisheries professionals have often speculated that bird populations, in particular, may be stimulated by the fishery discards and the discharge of offal. Identifying discards in the diets of fish or shellfish populations is more problematic, owing to the inability to distinguish such fish from live captures in the gut contents of predators and scavengers. *In situ* studies have documented the responses of benthic scavenger populations (*e.g.*, crabs and starfish) to the presence of fish and invertebrate mortalities from fishing activities. Such mortalities are generally used quickly. However, if the discharge of dead discards is sufficiently high in a localized area, it may overwhelm scavenger populations, resulting in decaying remains and attendant environmental problems. There also may be higher biological oxygen demand and released nutrients, possibly leading to local hypoxia depending on specific conditions at the sea bottom. In the extreme, such conditions may result in yet other mortalities of sessile animals in the vicinity of such events. Ultimately, nutrients released from dead discards will be recycled within the ecosystem, but the pathways and impacts of such nutrients are not well understood for any ecosystem. A significant proportion of the primary productivity of some ecosystems is required for the support of fishery populations (i.e., averaging 8%, but ranging up to 35% for temperate shelf ecosystems), of which discards may comprise a significant fraction (Pauly and Christensen, 1995). Therefore, in some cases the recycling of such discarded materials could significantly contribute to the energy budgets of some systems.

Because both the target and by-catch or discard species removed by fisheries often constitute the largest predators or herbivores in a system, the potential for propagation of food web effects is large. However, detecting these effects in marine ecosystems is difficult due to their openness, large size, rich species diversity, and potential for large numbers of (potentially compensating) direct and indirect effects

# Approaches for dealing with three sources of bias when studying the fishing down marine food web phenomenon

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In the Mediterranean as elsewhere, fisheries assessments are performed on a single-species basis, although the fisheries that are to be regulated tend to be multispecies in nature, and to have broad impacts on the ecosystems in which these species are embedded.

These impacts of fisheries have led, in the late 1990s, to a growing interest in “ecosystem-based management” (see *e.g.* NRC, 1999). Moreover, though the full implications of such form of management continue to remain elusive, there is a growing understanding that ecosystem-based management must consider and maintain the structuring role of large predators in marine food webs. The demonstration by Pauly *et al.* (1998a) that the mean trophic level of fisheries landings has been declining in most parts of the world thus confirmed the widespread view that most of the world fisheries are not sustainable in that they increasingly rely on organisms with low trophic levels, such as small fishes and invertebrates, i.e., on the prey species of larger fishes and marine mammals.

The approach of Pauly *et al.* (1998a) having been criticized on methodological grounds, with implicit reference to the Mediterranean (Caddy *et al.*, 1998), this contribution addresses three of the four sources of bias alleged to have affected its results:

1- ontogenic effects, due to changes in trophic levels with age, i.e., with size;

2- taxonomic effect, due to over-aggregation of landing statistics;

3- bottom-up effects, i.e., trophic level changes due to changes in primary productivity.

A fourth potential source of bias, i.e., using the abundance of species groups in landings as proxy for their relative abundances in the ecosystems, was addressed in a first response by Pauly *et al.* (1998b).

Before we can address items (1) to (3), however, we must briefly review the entities called “trophic levels”, crucial to the considerations below.

Trophic levels (TL) express the position of organisms within the food webs that largely define aquatic ecosystems. Their values are set as 1 in plants and detritus, 2 in herbivores (first-level consumers), 3 in second-level consumers, etc. However, real consumers, which tend to have catholic diets, do not usually have TL with integer values (Odum and Heald, 1975). The definition of TL for any consumer species i is thus:

$$TL_i = \Sigma_j\, TL_j \quad DC_{ij} \qquad 1)$$

where $TL_j$ is the fractional trophic levels of the preys j, and $DC_{ij}$ represents the fraction of j in the diet of i.

Thus defined, the TLof most fish and other aquatic consumers can take any value between 2.0 and 5.0, the latter being rare even in large fish (Cortés, 1999), occurring only in specialized predators of marine mammals, such as killer whales or polar bears (Pauly *et al.*, 1998c).

Equation (1) defines TL as measurable entities rather than “concepts” (Rigler, 1975); as such, they can be cross-validated using different methods, notably mass balance models of trophic fluxes in ecosystems (Christensen and Pauly, 1992; Pauly *et al.*, 2000b), and the ratios of $^{15}N$ to $^{14}N$ (Kline and Pauly, 1998; Polunin and Pinnegar, this volume). Fractional TL also allow the development of new approaches to the analysis of aquatic food webs, notably the accurate estimation of the “Primary Production Required” to maintain fisheries (Pauly and Christensen, 1995; Tudela, this volume) and the construction of a series of mean TL values of fish and invertebrates landed by fisheries as a means of evaluating their impacts on marine ecosystems (Christensen, 1998; Pauly *et al.*, 2000b; Stergiou and Koulouris, this volume) thus complementing more traditional approaches (Parsons, 1996). Herein, mean trophic levels (mTL) are computed, for any year y from:

$$mTL_y = \Sigma_i\, (TL_i \quad Catch_{iy}) \,/\, \Sigma_i\, Catch_{iy} \qquad 2)$$

where i refers to the species (groups) included in fisheries statistics (i.e., landings, or preferably catches consisting of landings and discards). This contribution shows how the accuracy of such analyses can be improved.

## MATERIAL AND METHODS

### Overall approach

Of the three sources of bias mentioned above, the first can be tackled by quantifying the relationship between body size and TL in a wide range of fishes. This can be achieved using data in FishBase (Fig. 1), and using a model of the form:

$$TL_i = 3 + b_i \quad \log_{10}(\bar{L}_i) \qquad 3)$$

where $TL_i$ is the trophic level of species i, $\bar{L}_i$ the corresponding mean length (in cm), 3 is the intercept [for 1 cm larvae, using log(1) = 0, and assuming the larvae feed on herbivorous zooplankton], and $b_i$ is the (mean) slope linking the available $TL_i/\log(\bar{L}_i)$ data pair(s) to the intercept.

This model can then be incorporated into either of two newly-derived equations for estimation of population-weighted estimates of mean TL, presented and illustrated below through an application to *Merluccius merluccius.* The growth parameters $L_\infty = 65$ cm (total length); K = 0.10 $year^{-1}$; $t_0 = -0.17$ year were used for this, based on data from the Northern Aegean Sea for the period 1991-1993 (Stergiou *et al.*, 1997). Also, we used a value of M = 0.15 $year^{-1}$, referring to the Korinthiakos Gulf and the period 1983-1985 (from Papaconstantinou *et al.*, 1985).

The effect of taxonomic over-aggregation (item 2 above) was documented by deriving a series of mean trophic levels using disaggregated data, then comparing the resulting series with one based on the same landing statistics, but in which related taxa had been aggregated before computation of mean trophic level.

Finally, we used the FIB index (Pauly *et al.*, 2000b) to identify bottom up effects (item 3 above), i.e., changes in absolute catches from lower trophic levels. The FIB-index takes the form:

$$FIB = \log\,[(\Sigma\, catch_{iy} \quad 10^{TLi}) \,/\, (\Sigma\, catch_{i0} \quad 10^{TLi})] \qquad 4)$$

where i is a given species (group) and y a given year, with year “0”, i.e., the year at the start of a series, which serves as anchor. The FIB index changes its value only when a decrease in TL is not matched by a corresponding increase in catch, and conversely for increasing TL. Here, “corresponding” is defined as a 10-fold increase for a decline of one trophic level, as implied by a 10% transfer rates between trophic levels (Pauly and Christensen, 1995).

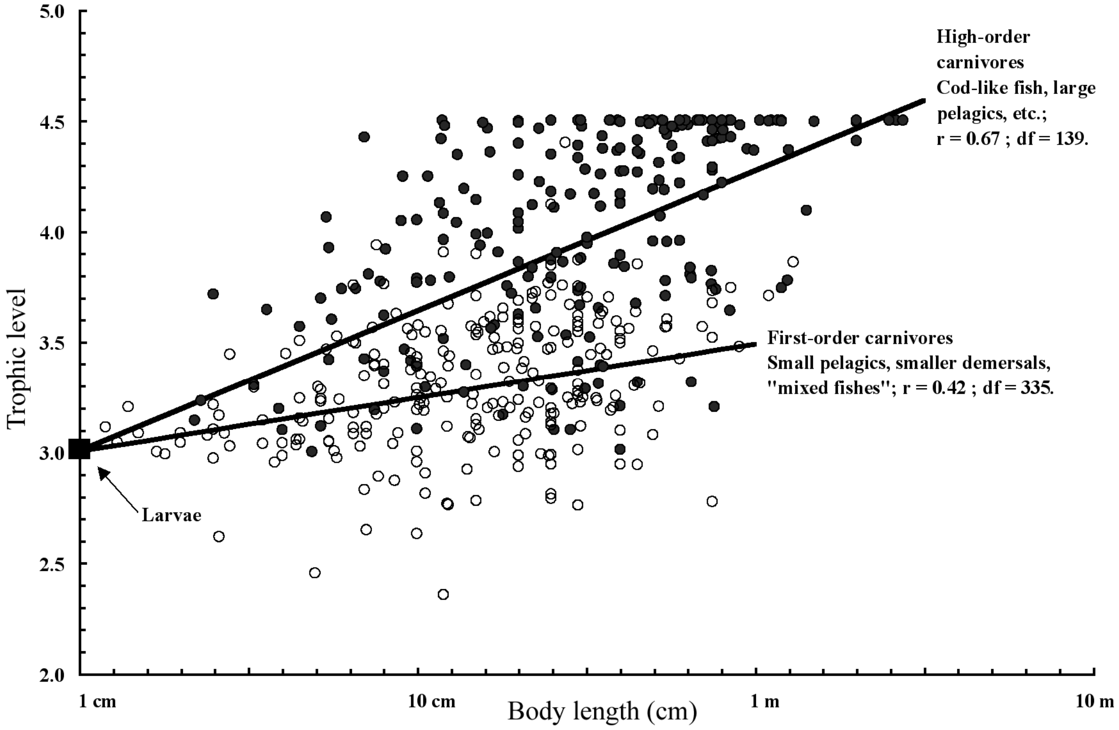

Fig. 1. Plot of estimates of trophic levels (based on diet composition data) *vs*. body size in fishes, based on data in FishBase (www.fishbase.org). Note steeper slope for high-order carnivores (which include *Merluccius merluccius*), implying a stronger effect of fishing on the mean (population) trophic level (see text).

## Length-based relationship between fishing mortality and trophic level

Total mortality (Z), in fish population whose individuals grow according to the von Bertalanffy Growth Function (VBGF), can be expressed by:

$$Z = [K \quad (L_\infty - \bar{L})] / (\bar{L} - L_c) \qquad 5)$$

where $L_\infty$ is the asymptotic length, i.e., the mean size the individuals in the population would reach if they were to live and grow indefinitely, K the rate at which $L_\infty$ is approached, and $\bar{L}$ is the mean length in the population, computed from $L_c$ upward (Beverton and Holt, 1956). Here, $L_c$ represents the length at entry into the fishery, assuming knife-edge selection. From this, one can derive:

$$\bar{L} = [L_\infty + (L_c \quad Z/K)] / [(Z/K) + 1] \qquad 6)$$

Defining exploitation rate (E) as E = F/Z, and E = 1 – (M/K) / (Z/K), we also have:

$$\bar{L} = \{ L_\infty + ( [(M/K) / (1\text{-}E)] \quad L_c) \} / \{ [(M/K) / (1\text{-}E)] + 1 \} \qquad 7)$$

for $0 \leq E < 1$. Thus, given Eq. 3, the change in TL ($\Delta$TL) in a fish population where body sizes have been reduced by fishing can be estimated from:

$$\Delta TL = b \quad \log_{10} \left[ \frac{ \left[ L\infty + \left( \frac{M/K}{1-E} \quad L_c \right) / \left( \frac{M/K}{1-E} + 1 \right) \right] }{ \left( L\infty \left( M/K \quad L_c \right) / \left( M/K + 1 \right) \right) } \right] \qquad 8)$$

where all parameters are as defined above.

The estimates of $\Delta$TL thus obtained are approximate because the TL values corresponding to the mean length obtained from Eq. 8 can only approximate the mean TL value obtained by integrating TL within an age-structured population, our next topic.

### Age-based relationship between fishing mortality and trophic level

The mean TL of an age-structured population in which TL changes linearly with (log) length can also be estimated from:

$$TL = \frac{\left(\int_{tr}^{tc} N1_t \;\; A\, dt + \int_{tc}^{tmax} N2_t \;\; N3_t \;\; A\, dt\right)}{\left(\int_{tr}^{tc} N1_t dt + \int_{tc}^{tmax} N2_t \;\; N3_t\, dt\right)} \qquad 9)$$

where $N1_t = e^{-M\,(t\text{-}tr)}$; $N2_t = e^{-M(tc\text{-}tr)}$; $N3_t = e^{-F+M(t\text{-}tc)}$; $A = 3 + b \;\; \log_{10} L_t$ (see Eq. 3); $L_t = L_\infty \;\; (1 - e^{-K\,(t\text{-}t_0)})$, i.e., the VBGF for growth in length with parameters $L_\infty$ and K defined as above, and $t_0$ setting the origin of the VBGF; $t_r$ = the recruitment age, i.e., the age from which on Eq. 9 applies; $t_c$ = the age at entry into the exploited stock, assuming knife-edge selection (i.e., corresponding to $L_c$; see above); and $t_{max}$ = the maximum age in the population.

## RESULTS AND DISCUSSION

The nomogram in Fig. 2 illustrates three points: (a) that the absolute value of the correction (ΔTL) due to size changes can be substantial, especially when fishing mortality is high; (b) that the sign of this correction is negative, and hence that not considering it underestimates the strength of the fishing down marine food web effects (hence invalidating that part of the critique by Caddy *et al.*, 1998); and (c) that the length-based model (Eq. 8) slightly underestimates the required correction (ΔTL) implied by the age-structured model (Eq. 9).

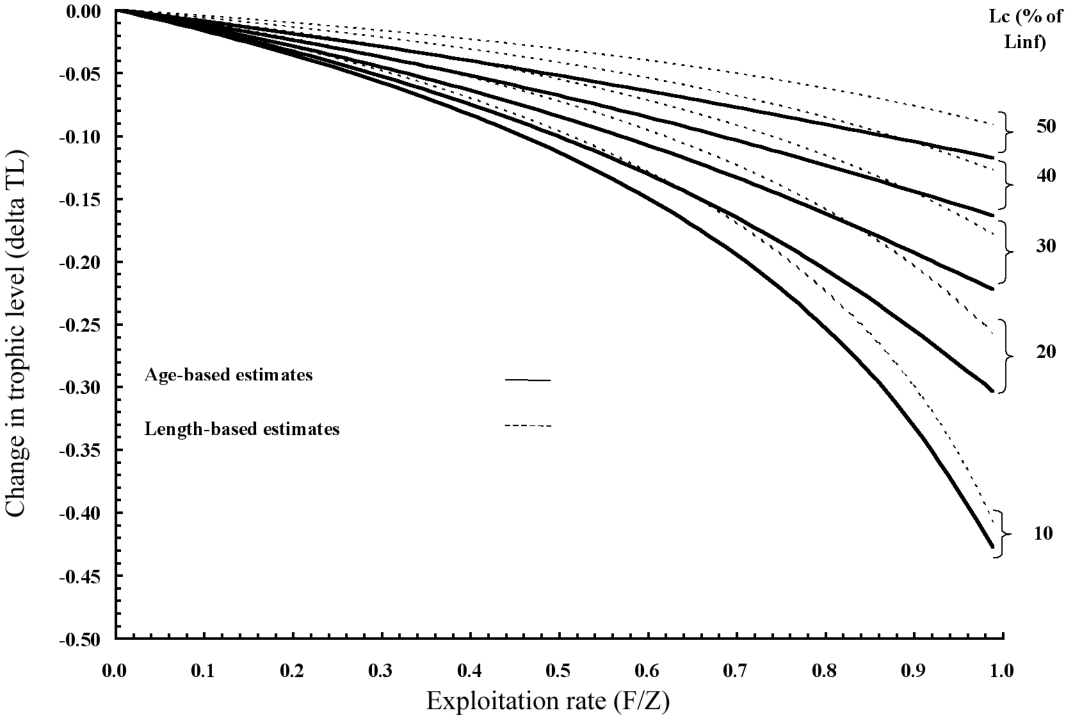

Fig. 2. Nomogram summarizing the impact of fishing (expressed by the exploitation rate F/Z) on the trophic level of high-level predators (as defined in Fig. 1). Note that Eqs. 8 and 9 give similar results, with the length–based model (Eq. 8) slightly underestimating the strength of this effect (see text).

Given these, we propose a generalized model for calculating mTL from multispecies catch (or landings) data, resulting from combining Eq. 2 and Eq. 3, viz:

$$mTL_y = \Sigma_i\, (3 + b_i \;\; \log_{10}(\overline{L}_{iy}) \;\; Catch_{iy}) \,/\, \Sigma_i\, Catch_{iy} \qquad 10)$$

where the $TL_{iy}$ for a species i and year y are obtained via a constant, species-specific previous estimate of $b_i$ (as can be derived from pairs of $TL_i/\overline{L}_i$, and from year-specific mean lengths, obtained either from field data, or via Eq. 7 (but preferably not a mixture of the two length types in the same species).

As such, Eq. 10 will allow time series of mTL to account simultaneously for both between-species effects (via changes in the species mix, as expressed by the catches) and within-species effects (via the impact of mean length changes on the TL estimates).

Moreover, Eq. 10 should also be useful for analyses such as that of Harmelin-Vivien (this volume), where relative abundances in the ecosystems (*e.g.*, in comparisons of fished and unfished areas) are used instead of catches or landings. Also note that species-specific estimates of $b_i$, required to convert mean length into TLestimates, can be derived from data in FishBase for most of the commercial species of the Mediterranean (Froese and Pauly, 1999), with more species to be included in forthcoming updates (see: www.fishbase.org).

Fig. 3 documents the effect of over-aggregating fisheries statistics on the estimation of mTL. As might be seen, the downward trend of mTLestimated from detailed catch statistics (over 80 % of catch reported at species level) disappears when these same data are regrouped into coarser entities, here the functional groups of FAO’s International Standard Classification of Aquatic Animals and Plants (ISCCAAP). As only 50 % of the world’s, and about 65 % of Mediterranean landings are reported on a per-species basis, the “fishing down” effect reported by Pauly *et al.* (1998a) for the world as a whole, and for the Mediterranean (see Fig. 1 in Executive Summary, this volume) will underestimate the decline of mTL values. Note that this effect is added to the ontogenic effects discussed above, thus further invalidating the critique of Caddy *et al.* (1998).

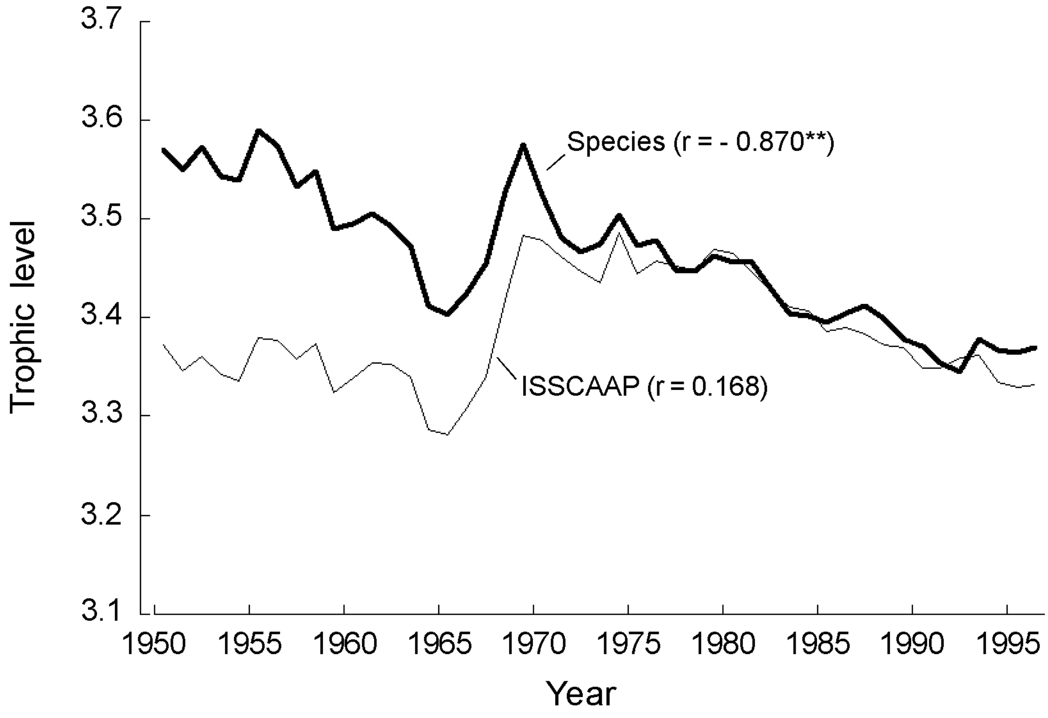

Fig. 3. Trends of mean trophic level in the Eastern North Atlantic (FAO Area 27), without (thick line) and after (thin line) aggregating the catch data into the broader ISSCAAP categories. As may be seen, the resulting loss of taxonomic information leads to a disappearance of the previously significant downward trend of mean trophic level (see text).

Fig. 4 finally contrasts time series of the FIB index for two FAO areas with strongly diverging features: the Northwest Atlantic, where TL and catches increased from the 1950s to the 1970s, following a relative increase of effort on previously under-exploited deep demersal stocks, which eventually collapsed (as in many other parts of the world), and the Mediterranean, where catches increased without a strong geographic expansion of the fisheries, suggesting a bottom-up effect (eutrophication leading to increased catches of herbivores, *e.g.*, bivalves and small pelagics; see Caddy, 1993).

This may justify the notion that the Mediterranean is somehow "different", though its food webs may also end up being "fished down."

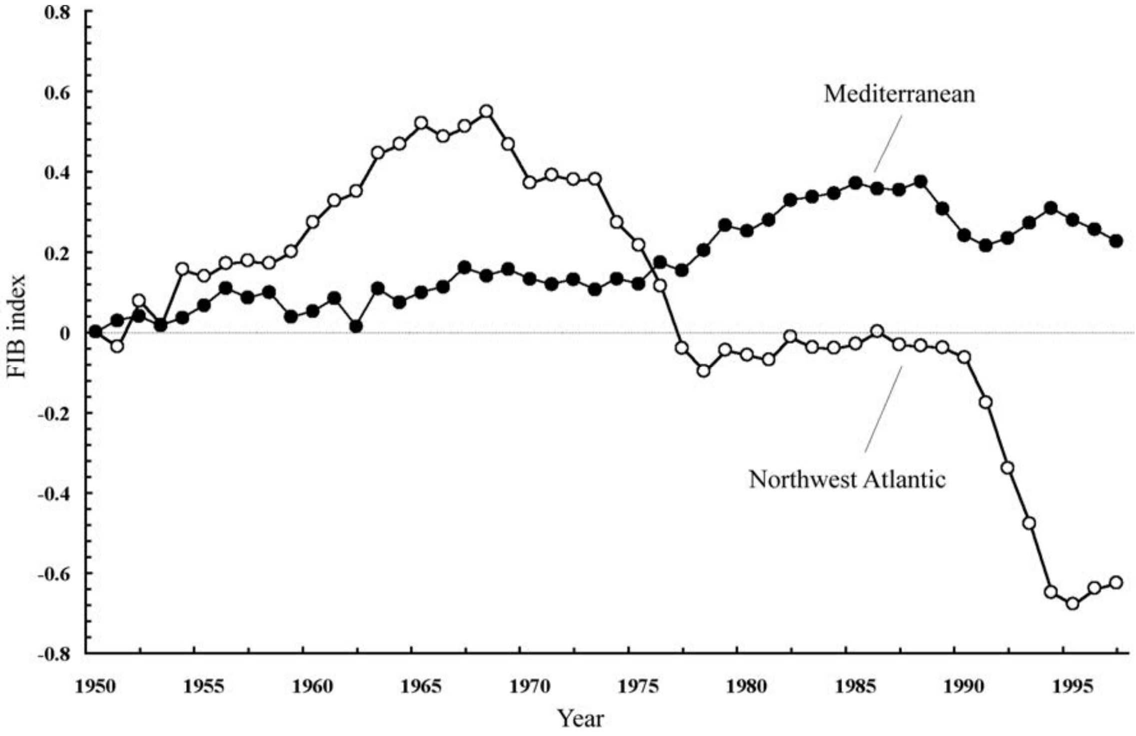

Fig. 4. Simultaneously considering catch variations and changes in trophic levels, as allowed by the FIB-index, shows that in the Mediterranean, catches increased as trophic levels declined, suggesting a bottom-up effect (i.e., eutrophication possibly leading to increased populations and catches of bivalves and small pelagics). The Northwest Atlantic, in contrast, saw "extensions" (new areas and new species fished), from 1950 to the mid 1960s, then a stepwise decline, culminating in the 1990s in a collapse of large demersal fishes with high trophic levels (especially Atlantic cod), but without compensating effects from the bottom-up.

# Classification of the stocks according to their exploitation status. The case of the data-poor stocks

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The Mediterranean fisheries constitute a perfect case where the Precautionary Approach should be considered (CIESM, 1999). The fishing activity has been increasing along time and the scientific information about it is scarce, (Lleonart, 1999). In other areas, assessed by international commissions the precautionary approach has been implemented and also, biological reference points have been set for several stocks or stock units.

In its last meeting the SCSA recommended the development and adoption of Biological Reference Points (BRP) specific for Mediterranean resources. Considering the possibility of fixing limits RP that should be exceeded without dangerously affecting the sustainability of the resources, the proposal is to adopt target RP based on biomass indices (spawning and standing stock biomass, expressed as percentage of virgin biomass) and mortality rates (expressed by exploitation rate, E=F/Z), (SCSA, 2000).

During its last meeting, the STECF recognised that about half of the stocks assessed by ICES, more than 80% of the stocks assessed by NAFO, and all stocks assessed by ICCAT are considered to be data moderate or data poor: age-based assessments cannot to be successfully applied or indirect ageing methods are used.

STECF proposes that for such stocks, the availability of information will determine what can be inferred about the state of the stock or its state of exploitation. Accordingly for these stocks, it may not be possible to provide firm guidelines for classification in the frame of the Precautionary Approach.

Therefore, STECF has attempted to describe three states of data availability that would indicate the exploitation status of a stock.

## No data

STECF suggests that even in a situation where there are no fishery or stock-related data, a qualitative indication of over-exploitation may be obtained through opinion surveys. This may not always be possible or reliable but this approach should not be totally ruled out. Similarly an indication of overexploitation may be obtained if there are conflicts or competition between different fishery sectors for the same resource. If no such conflicts exist this may indicate that the stock is not over-exploited.

## Data poor

An estimate of total landings or total catch from a fishery and preferably an estimate of the total numbers of units exploiting the resource, either vessels or participants, could indicate whether a stock is over-exploited. An indication of a decline in catches with increasing effort over time would suggest that the stock is being overexploited.

**Data Moderate**

STECF suggests that data-moderate include the data available for the data-poor state, plus additional information on all or some of the following variables:

- total catch by different fleet sectors;
- effort trends by fleet sector;
- basic biology of the resource especially growth and distribution;
- age/size composition of the catch;
- trends in abundance, density or biomass from fishery independent surveys.

The development of precautionary approach reference points for data-poor and data-moderate stocks is actively being pursued by advisory bodies. When these are eventually proposed, an evaluation of the state of exploitation of data-poor and data-moderate stocks in relation to the precautionary reference points will be possible.

The classification of stocks with respect to their risk of depletion or of recruitment over-fishing would normally require information on time trends in spawning stock and recruitment. In data-poor or data-moderate situations, such information, by definition, will not be available, hence a more generalised form of classification is required. In this case the use of the category “overexploited” is proposed. It would be straightforward to determine whether or not a stock was overexploited using most of the simpler assessment methods, which would be most appropriate in such cases. Furthermore, even if the data were not sufficient to permit even the use of one of these simpler methods, it may still be possible to assess whether a stock is overexploited using one of the approaches outlined.

In the Mediterranean area the majority of the stocks which have commercial interest could be included in the second category of knowledge. A great effort should be made to reach a level of data and information similar to those stocks in which regular analytical assessment is made. This is a major challenge for the Scientific Advisory Committee of the GFCM.

The development of “trophic level” reference points is an important issue for fisheries research in the Mediterranean and could be an alternative to the traditional reference points establishment.

# Trophic-level dynamics inferred from stable isotopes of carbon and nitrogen

**Nicholas V.C. Polunin and John K. Pinnegar**

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According to FAO fisheries statistics and gut-contents data, the mean trophic level (TL) of Mediterranean catches has declined significantly and quite steadily since the late 1950s (Pauly *et al.*, 1998a). This conclusion has helped to highlight not only the plight of many fisheries world wide, but also the paucity of data on the marine food webs concerned. As a means of describing the structure of food webs, gut contents analysis has several shortcomings (Table 1) and it is important to broaden the base for understanding the TL dynamics underpinning fisheries. Drawing on examples of its application to various Mediterranean systems, we will highlight some of the ways in which stable isotopes are helping to elucidate patterns of change in food webs and the TL dynamics of species.

Table 1. Comparison of gut-contents and stable isotope data in elucidating food web structure (Polunin and Pinnegar, in press)

| **Information** | **Gut contents** | **Stable isotopes** |
|---|---|---|
| Resolution of trophic pathways in food webs | Can be good for identifiable sources (*e.g.*, hard parts) | Can be good if pathways well distinguished by $\delta^{13}C$ of basal materials, poor if >2 pathways |
| Connectance (proportion of linkages realised) | Good for sources that are individually identifiable | Poor because only broad categories distinguishable as a rule |
| Nutritional roles of different dietary items | Poor because diet, not actual absorption, quantified | Can be good: isotopes are in materials that are assimilated, not merely ingested |
| Short-term differences in diet of large predators | Potentially good because data are only short term | Poor because turnover of most tissues rather slow |
| Diet of top predators | Poor because of intermittent feeding and regurgitation of food upon capture | Good because isotopes in tissues are integrated from prey over a long period of time. |
| Spatial differences in diet of large predators | Good where major items identifiable | Good where shifts occur in items with distinct $\delta^{13}C$ and/or in trophic level |
| Trophic level and food-chain length | Often inaccurate: diet incompletely described | Good if basal materials identified, and change in $\delta^{15}N$ per trophic level validated |

### Estimation of trophic level

Estimation of TL using $\delta^{15}$N data and herbivorous zooplankton as reference materials largely corroborates TL data derived from an ECOPATH model of the littoral; this accords with work on Arctic-marine and Canadian-lake systems (Vander Zanden *et al.*, 1997; Kline and Pauly, 1998). There is however an indication that at higher values the TL is over-estimated by ECOPATH, under-estimated by the $\delta^{15}$N data, or both (Fig.1).

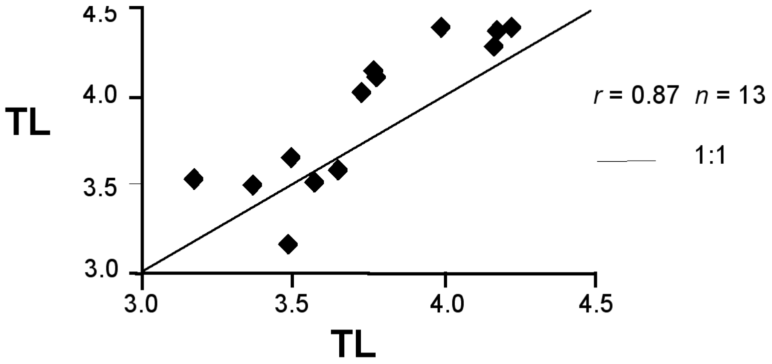

Fig. 1. Relationship between estimates of trophic level derived from $\delta^{15}$N data (TLN) and ECOPATH modelling/gut contents data (TLE), for 13 species of Mediterranean rocky littoral fish from 4 sites in Corsica and Mallorca.

### Differential focusing of food-web source materials

Plots of $\delta^{15}$N and $\delta^{13}$C data for major macroconsumers and source materials constitute a sort of finger-print of food webs, and this may be useful in highlighting fundamental characteristics of the system concerned. Thus greater variability about the regression line between $\delta^{15}$N and $\delta^{13}$C data of a littoral system (Pinnegar and Polunin, 2000) than about that between the isotope variables from a deep-sea system (Fig. 2) suggests that there is a greater range of source materials and thus distinct trophic pathways in the former. In the littoral case, primary producers span from phytoplankton to a variety of benthic algal types.

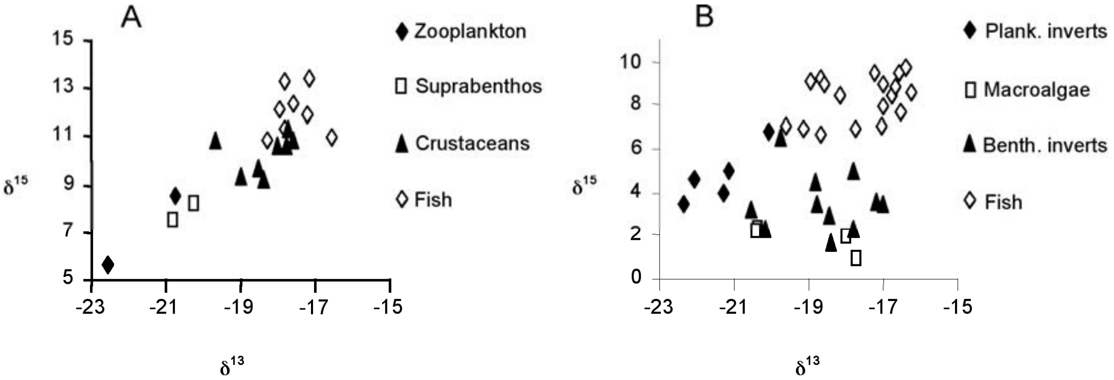

Fig. 2. Relationship between $\delta^{15}$N and $\delta^{13}$C of consumers in two Mediterranean ecosystems: **(A)** middle slope (800-1400m depth) assemblage, SW Balearic Islands; **(B)** Rocky sublittoral (0-9m depth) assemblage, Bay of Calvi, Corsica

### Estimation of food-chain length

There has been some debate about how long food chains are. Hall and Raffaelli (1993) concluded that most food chains are short, and this is corroborated by TL estimates based on $\delta$15N data for a variety of fishes from three Mediterranean systems. Using herbivorous zooplankton as a reference, the mean TL of large consumers is 3-4 for littoral and deep-sea systems in the Western Mediterranean, and the maximum is no greater than 4.4 (Fig. 3).

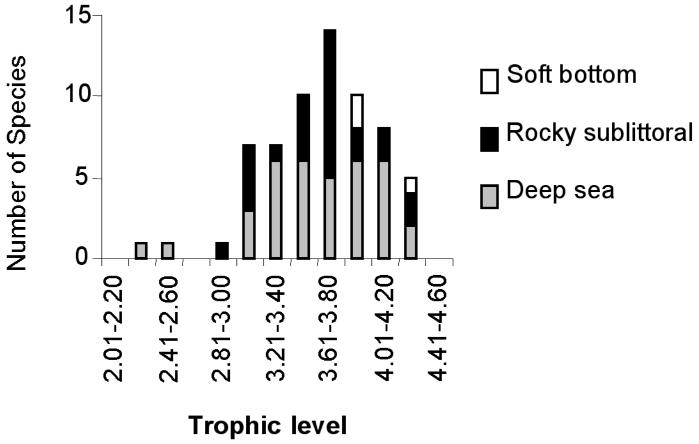

Fig. 3. Mean trophic level of 64 Mediterranean fish species, derived from a database of $\delta^{15}$N values.

### Spatial scale of variation in food-web structure

Raw $\delta^{15}$N data for fishes may be quite variable spatially because the fishes are more or less omnivorous or feed at different TLs (Jennings *et al.*, 1997), the basal materials such as plankton are variable in TL, or all of these. Comparing a site in Corsica with three in Mallorca, we found differences among the Mallorcan sites and between Mallorca and Corsica based on the fish $\delta^{15}$N data alone, but calculating TL with reference to a baseline of herbivorous zooplankton (assuming TL = 2) rendered the Corsican site similar to one of those in Mallorca (Fig. 4).

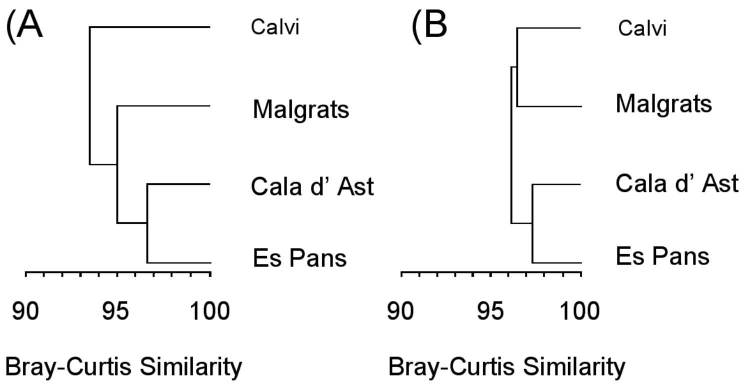

Fig. 4. Similarity of 3 Mallorcan (Malgrats, Cala d'Ast, Es Pans) and one Corsican reef site (Calvi), based on **(A)** $\delta^{15}$N of fish species, and **(B)** trophic level (TLN) of the same fish species

### Size-related changes in trophic level

$\delta^{15}$N data from the Bay of Castellammare (NW Sicily) indicate that mean TL increases with size in fishery species such as hake, mullet and monkfish (*e.g.*, Fig. 5), a result found elsewhere in some other species (*e.g.*, Lindsay *et al.*, 1998). It follows that fishing will tend to reduce the mean TL of such target species and conversely that TL should increase in areas where fishing is greatly reduced in intensity, as when a MPA is established. We are investigating potential size-dependent and size-independent changes in TL in three target species in the Castellammare no-trawl area established over ten years ago.

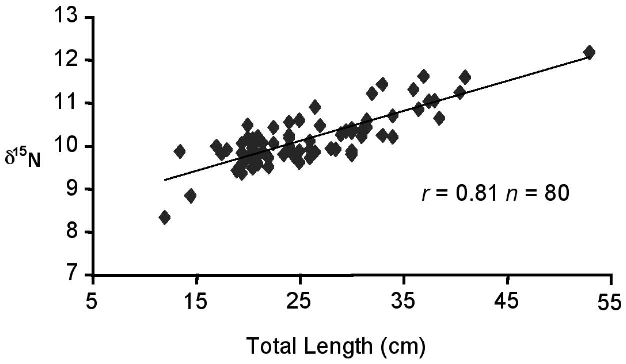

Fig. 5. Change in the mean δ15N of hake (*Merluccius merluccius*) with increasing fish length in the Gulf of Castellammare (50-200m depth), Sicily.

## Conclusion

Like any method applied to anything as complex as a food web, stable isotope analysis has weaknesses (Table 1), but these case histories indicate that it has valuable roles to play in elucidating food webs of the Mediterranean.

**Acknowledgements.** Fisheries Society of the British Isles and the Foundation for Environmental Conservation for funding, and our collaborators Fabio Badalamenti and Beatriz Morales-Nin for permission to use data.

# Fishing down the marine food webs in the Hellenic seas

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## INTRODUCTION

Fishing has important effects on marine ecosystems both at the level of the life-history of the individual species (*e.g.* decreasing mean body size and mean size/age at maturity, changing sex ratios, and decreasing population reproductive potential: *e.g.* Jennings *et al.* 1998; Stergiou, 1999) and at the ecosystem level (*e.g.* altering the structure and heterogeneity of benthic habitat, changing species composition and diversity, decreasing stock abundance, changing predation and competition rates, changing trophic structures and energy flows: Pauly and Christensen 1995; Pauly *et al.* 1998a, 2000a; Stergiou 1999; Tudela, this volume; Harmelin-Vivien, this volume).

Among the latter effects, those concerning the trophic structure and energy flow of marine ecosystems are very important. Indeed fisheries landings, including by-catch, require about 24-35% of the global marine primary production over the continental shelf and major upwelling areas of the world ocean, a fraction that is much higher than previously thought (Pauly and Christensen, 1995). Such a large-scale removal is not without any effects on the structure of food webs. Thus Pauly *et al.* (1998a, 2000a) clearly showed that the mean trophic level of the fisheries landings in the last 45 years has decreased steadily both at the global scale (total marine and freshwater landings) and at the regional, ocean-specific, scale (i.e. Atlantic, Pacific, Mediterranean), with few exceptions. Pauly *et al.* (2000a) maintained further that such a decline might be even faster than documented, because of arguments that relate ontogenic changes in feeding of individual species with trophic levels. The approach of Pauly *et al.* (1998a), currently known as the “fishing down the food web” theory (FDFW), seems to be a promising tool for identifying whether the fishery in a particular region is sustainable or not. Although a debate was raised (Caddy *et al.* 1998) on four sources of bias that may have affected Pauly’s *et al.* (1998a) results, all these methodological issues have been recently addressed (i.e., using the abundance of species groups in landings as proxy for their relative abundance in the ecosystems: Pauly *et al.*, 1998b; ontogenic, taxonomic and bottom-up effects: Pauly 2000a, Pauly and Palomares, this volume). The FDFW hypothesis triggered fisheries scientists to start searching for similar patterns at the local scale. In this work, we test the hypothesis that the mean trophic level of the landings in the Hellenic seas (i.e. Ionian and Aegean) has declined during the last 34 years, using the available landing data for the period 1964-1997.

## SOURCES OF DATA

The fishing fleet operating in the Hellenic seas includes (Stergiou *et al.*, 1997): (a) trawlers operating in open-sea waters, (b) purse seiners operating in open-sea and coastal waters, (c) “mixed vessels” operating in waters both as trawlers, mainly in the wintertime, and purse seiners, mainly in the summertime, (d) beach seiners operating along the coasts, and (e) "other coastal boats" (including small ring netters, drifters, liners, etc.) operating along the coasts. The National Statistical Service of Hellas records the Hellenic landings since January 1964 (Stergiou *et al.*, 1997). For a better evaluation of the available data, Hellenic waters have been divided into statistical fishing sub-areas (Fig. 1).

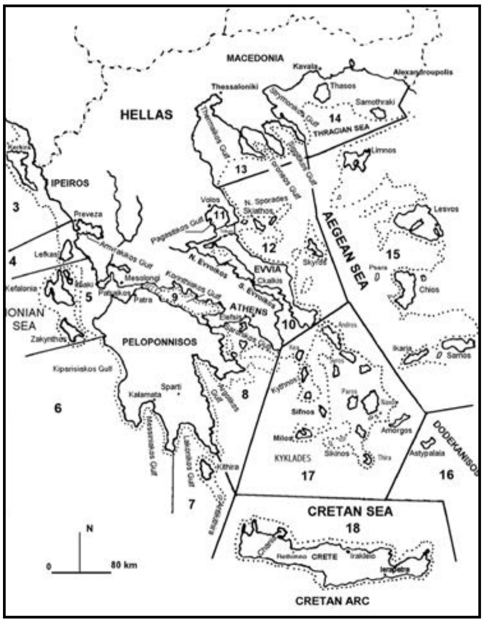

Fig. 1. Map showing 16 Hellenic fishing sub-areas. The dotted line represents the 200 m isobath.

Catch data are collected directly from a sample of fishing vessels that are surveyed by local customs authorities (stratified random sampling). For each vessel surveyed, a statistical questionnaire is completed showing the quantities of each major species caught during the previous month for the actual periods of activity. It is noteworthy that since 1969 the local customs authorities do not record the catches of the smaller inshore ring netters, drifters and liners (i.e. "other coastal boats" with engine horsepower of less than 20 HP). For the period 1964-1981, separate catch statistics are available for 23 species only (or group of species; henceforth called species) whereas for the years following 1981 separate catch statistics have been available for 66 species of commercially important fishes, cephalopods and crustaceans. Although the NSSH data suffer from various biases, which are higher for inshore fisheries and the degree of bias can not be easily estimated, they are the best figures available with respect to: (a) length of time, (b) spatial and temporal resolution, (c) consistency and degree of subjectivity in data collection, and (d) statistical design of data collection (Stergiou *et al.*, 1997).

The trophic level values (TL; for definition of TL as measurable entity, see Pauly and Palomares, this volume) of all species recorded by NSSH were taken from Pauly *et al.* (1998a; available on the Internet at www.fisheries.com). For each sub-area, the mean mTLi for each year i was estimated by multiplying the landings (Li) by the trophic levels of the individual species j, then taking a weighted mean: $mTL_i = (\Sigma_{ij} TL_{ij} L_{ij})/(\Sigma L_{ij})$ (Pauly *et al.* 1998a; Pauly and Palomares, this volume).

## RESULTS AND DISCUSSION

For most sub-areas (i.e. 3, 4, 6, 7, 9 10, 13-15, 17 and 18), total landings steadily increased from 1964 up to the early-mid 1990s and then declined sharply (figures not shown here). The same was true of sub-area 16, except for an earlier peak in the 1980s. For sub-areas 5, 8 and 12, total landings steadily increased from 1964 to 1997 and those of sub-area 11 exhibited peaks in the mid 1970s. It is noteworthy that the engine-horsepower/boat and tonnage/boat ratios also sharply increased from 1964 to 1997 for all component fisheries, with the exception of the "other coastal boats" fishery for which the rate of increase was much smoother (Stergiou *et al.*, 1997; Stergiou, unpubl. data). This clearly indicates the modernization of the Hellenic fishing fleet, with older boats being replaced by new ones of higher capacity (larger tonnage and engine HP), mainly because of EU subsidies.

The mTL of the landings during 1964-1997 for the 16 fishing sub-areas is shown in Figure 2. In general, mTL fluctuated greatly with time. In addition, mTL was significantly (t-test, $P<0.05$) lower in 1964-1981 when compared to 1982-1997 for the majority of fishing sub-areas, a fact that is most probably attributed to the higher taxonomic resolution of the catches for the latter period (Pauly and Palomares, this volume).

The mTL of the landings declined significantly with time [i.e. the slope of the mTL-time regressions being significantly ($P<0.05$) different from zero and negative] for both study periods only for sub-areas 16 to 18 (southern Aegean Sea) and the decline was of the order of 0.1 TLunits (Fig. 2). It also declined significantly ($P<0.05$) during 1964-1981 for sub-area 7 (by about 0.1 TL unit) and during 1982-1997 for sub-areas 4 and 10 (by about 0.3 TL units). All remaining sub-areas showed no clear trends in the mTL changes over time. Thus, some sub-areas exhibited an increasing trend in mTL either during 1982-1997 (*e.g.* sub-areas 7, 12 and 15) or during both periods (*e.g.* sub-area 9) (Fig. 2) whereas others exhibited no trend (*e.g.* sub-areas 5, 8, 14). Yet, during the last decade, mTL sharply declined by 0.1 to 0.4 TL units in all sub-areas with the exception of sub-areas 7, 11, 12 and 15.

It is noteworthy that the mTL of the landings of the three most important sub-areas of the Hellenic Seas, namely sub-area 8 (Saronikos Gulf) and sub-areas 13 and 14 (North Aegean Sea), which make up more than 45% of the total Hellenic landings (Stergiou *et al.*, 1997), also showed no evidence of the FDFW effect (Fig. 2). However, anchovy and sardine generally dominate the landings of these sub-areas, making up more than 30-40% of the total, depending on the sub-area

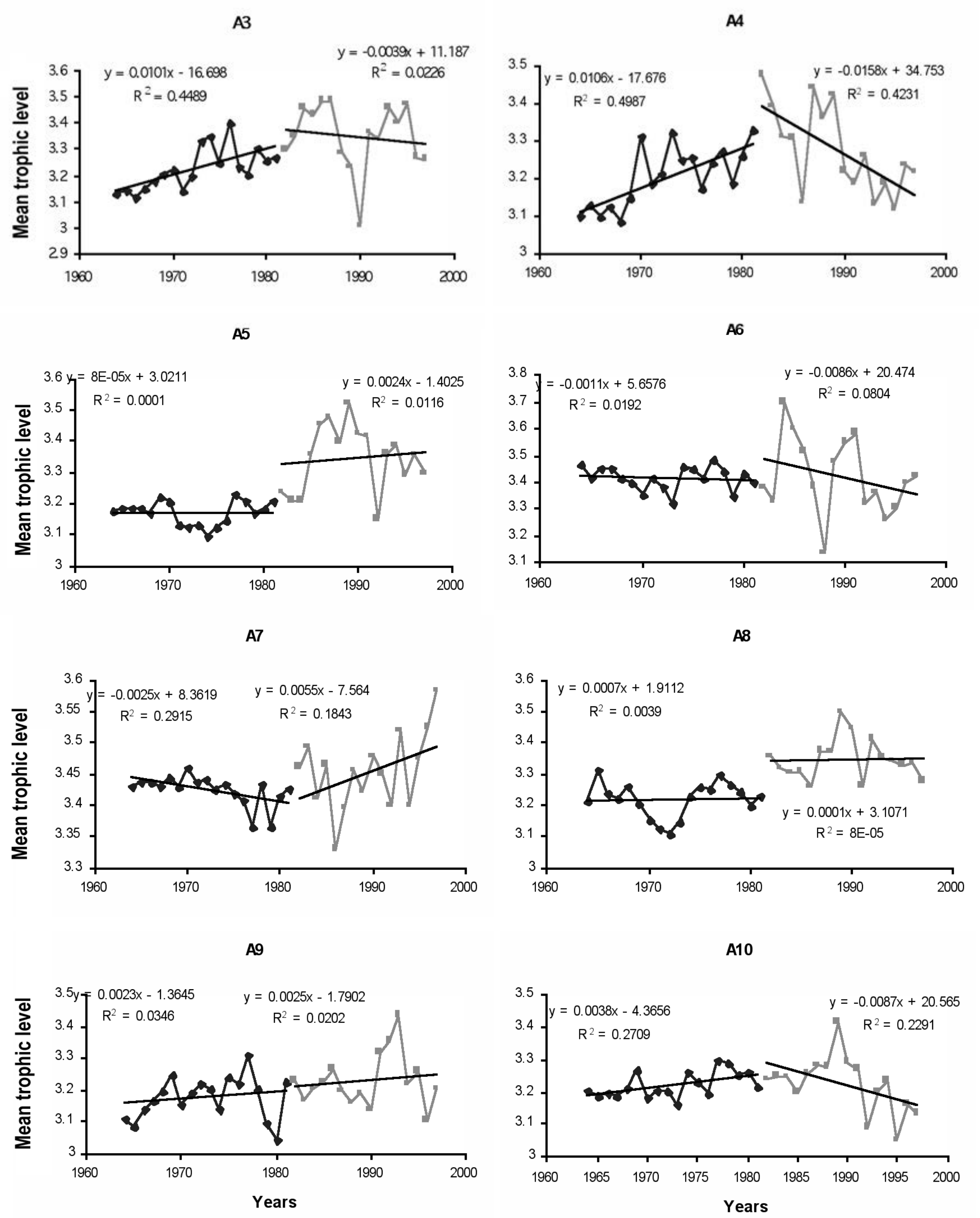

Fig. 2 **a.** Mean trophic level of the landings in 16 fishing sub-areaS (A3 to A18) of the Hellenic seas during 1964-1997 (first part).

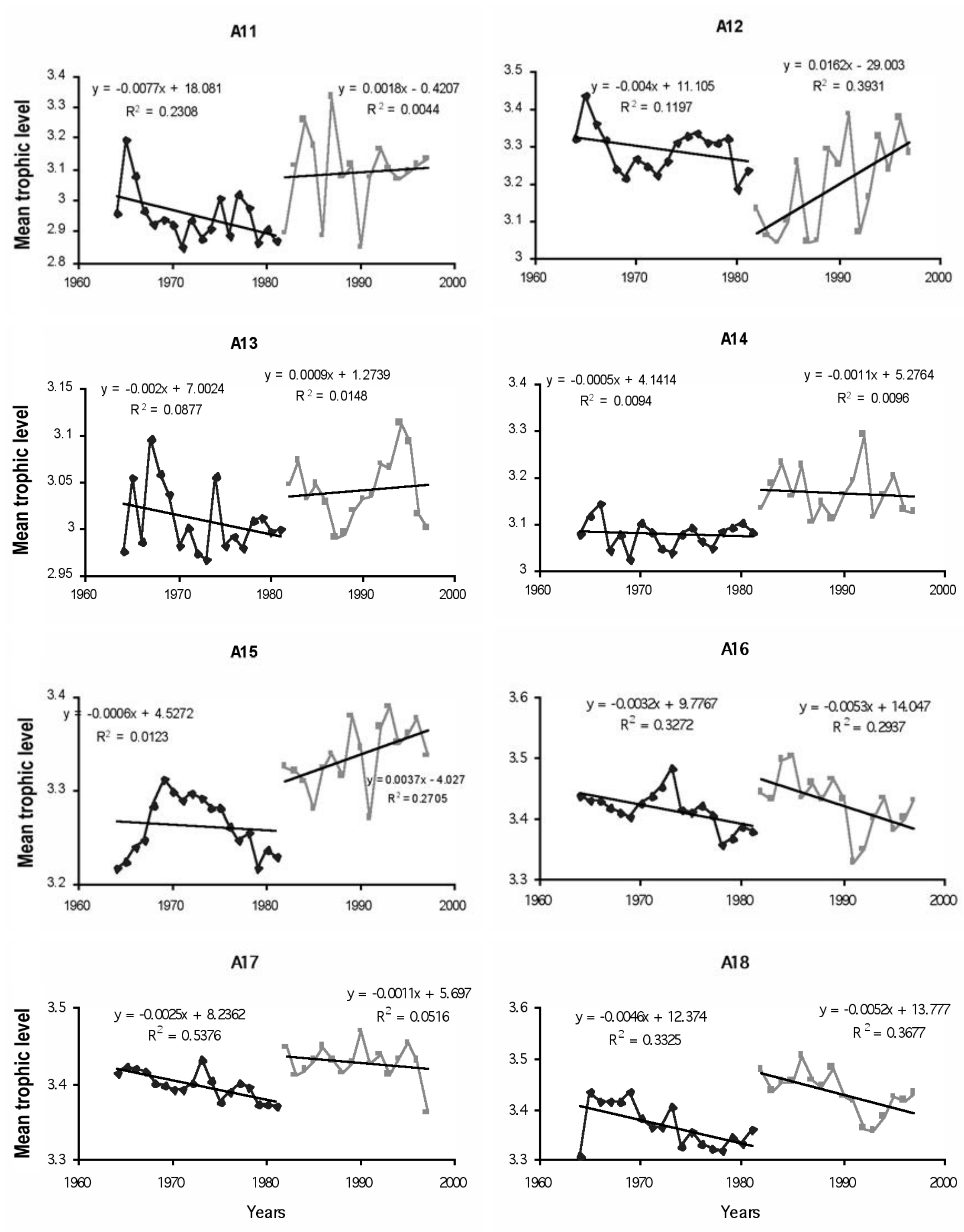

Fig. 2 **b**. second part.

(Stergiou *et al.*, 1997). These two small pelagic species generally undergo large fluctuations in the landings because of climatic and/or economic factors (*e.g.* Stergiou and Christou, 1996; Stergiou and Lascaratos, 1997; Stergiou *et al.*, 1997), a fact leading to large year-to-year fluctuations in mTL, which may mask any other long-term trends. Indeed when the anchovy and sardine landings were not taken into account for the estimations of mTL (figures not shown here), mTL in all three sub-areas declined significantly with time ($P<0.05$) during both study periods by about 0.1 TL unit, except for sub-area 8 in 1964-1981. The same was also true of sub-area 11, for which mTL declined by almost 0.2 units during 1982-1997, as well as for sub-areas 12 and 15 during 1982-1997, although in the latter two sub-areas the trend was not significant ($P>0.5$).

During the last years the mean body size of anchovy and sardine has decreased in the North Aegean Sea (Stergiou, unpubl. data; Voulgaridou and Stergiou, 2000). In addition, the majority of the demersal fishes in the Hellenic seas suffer from growth overfishing, with commercial landings consisting mainly of young, immature individuals (Stergiou *et al.*, 1997). Such changes, which cannot be captured by the present analysis, would strengthen the observed trends (or even reveal unobserved ones), because smaller individuals generally feed at lower trophic levels (Pauly *et al.* 1998a, 2000a; Pauly and Palomares, this volume).

The results show that in the main part of the Aegean Sea (i.e. sub-areas 8, 13, 14 and 15-18) the mTL of the landings has decreased during the last years. This trend, which seems to be more pronounced during the last years (Fig. 2), indicates that the Aegean fisheries resources are not sustainable, a fact found in agreement with the results of both field and catch-effort studies (Stergiou *et al.*, 1997). In contrast, the FDFW effect was less pronounced in the Ionian Sea (i.e. sub-areas 3 to 7), although in sub-areas 4 to 6 the mTL of the landings declined during the last years. Such a difference between the two seas may be attributed to: (a) the low level of the Ionian landings, about 10-13% of the grand total; (b) the smaller accuracy of the Ionian when compared to the Aegean data because of the predominance of the artisanal fishery (mainly in sub-areas 6, 7 and 9, which are almost exclusively fished by small-scale fishing gears); (c) the fact that Hellenic landings might not be representative of the Ionian situation because the Hellenic part of the Ionian Sea comprises part of a larger area in which the existing stocks are fished by a number of other nations (*e.g.* Albania, Italy), with the Hellenic landings representing about 7% of the total Ionian ones (Stergiou and Pollard, 1994; Stergiou *et al.*, 1997); and (d) a potential expansion of the Ionian fishery into new fishing grounds. We are still working on combining the two time series of landings (i.e. 1964-1981, 1982-1997) and interpreting the combined mTL-catch plots for all 16 sub-areas (an example is presented in Fig. 3) as well as on estimating Pauly's FIB index (Pauly *et al.*, 2000b) for further evaluating the FDFW effects in Hellenic waters.

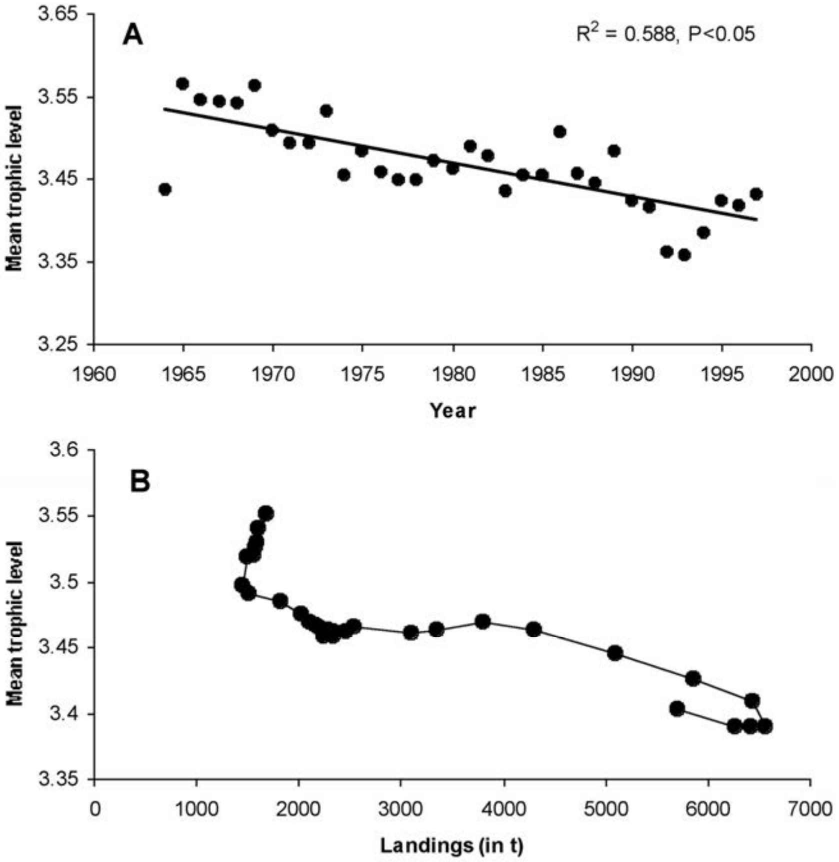

Fig.3. **(A)** Mean trophic level, mTL, of landings in fishing subarea 18, reconstructed so that to take into account the higher taxonomic resolution of landings in 1982-1997 when compared to 1964-1981; reconstruction was done by adding the difference in the mean mTL of landings for the two periods to the mTL of the landings during 1964-1981); and **(B)** relationship between the 5-year running means of mTL of landings and of landings for fishing subarea 18.

Fisheries management in Hellenic waters is practiced through static, technical measures in a non-adaptive, and certainly far from being precautionary, way. The inadequacy of the technical regulations currently in force is justified by the fact that, despite the reinforcement of such measures, demersal and inshore fisheries resources are currently overfished. Thus, Stergiou *et al.* (1997) have suggested that such measures should be either replaced or complemented by Marine Protected Area (MPA) which represent the extreme case of the precautionary approach. MPAs protect marine biodiversity, decrease the trend for heavy evolutionary fishing selection for earlier maturity and reproduction and smaller adult fish size, and hedge against uncertainties, errors in estimations and biases (see Lauck *et al.*, 1998, and references therein), thus maintaining the trophic structure of marine ecosystems.

# Assessment of the ecological footprint of fishing in the Catalan central coast (NE Spain)

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The study of the amount of primary production that is “appropriated” by Man through the fishing activity has proven to be a powerful tool for assessing of the impact of fishing on the whole marine ecosystem (Pauly and Christensen, 1995). The resulting ecological indicator, the percentage of the primary production required to support a given fishery (%PPR), appears to be easy to obtain provided that a limited amount of basic information is available. Besides that, it is able to provide a rather holistic view of the fishery, for it highlights its ecosystemic role in channeling marine trophic flows towards human uses.

The Catalan coast has supported a traditional fishing exploitation since ancient times. Commercial landings suffered a dramatic increase during the 60’s and the 70’s as a consequence of fleet developmental policies, reaching in the 80’s a peak at about 70000 t, from a mere 15000 t in the early 60’s. During the last 20 years the amount of landings has remained centered around a value of 55000 t, with frequent oscillations owing to fluctuations in small pelagic populations. The last few years, however, landing values have drastically dropped from 66000 t in 1994 to only 43000 in 1999, the lowest record in more than 20 years.

In order to obtain useful lessons from the study of the ecosystem impact of fishing its peak, we estimated the %PPR for 1996, a year representative of the period of highest harvests. Official estimations of landings in Catalonia for this period are at about 53000 t. The area considered for the study was the exploited grounds off the Central Catalan coast, a coastal area of 7487 $km^2$ ranging from the port of Vilanova in the South to the port of Palamos in the North, covering the entire area ranging from the coastline to the isobath of 1000 meters (Fig. 1). This depth can be regarded as both an offshore limit for coastal fishing and an ecological barrier between coastal and oceanic waters because of the existence there of a marked, well-known shelf-slope hydrological density front (Font *et al.*, 1988). Also, fishing features of this area, which encompasses a total of eight fishing ports harbouring important trawler and purse seine fleets, together with other varied, mainly small scale fleet, make it straightforward to know the volume of catches originating in the area (Table 1).

Catches were estimated from the best possible official statistics: daily information from bill notes providing disaggregated information per port, boat and species. Based on reliable information obtained through direct enquiries to key persons involved in the commercialization process we estimated the occurrence of misreporting at about 30% of total landings, affecting both demersal and pelagic fish. Subsequently, official statistics were increased accordingly in order to prevent underestimations. Finally, in order to estimate real catches we increased the estimated

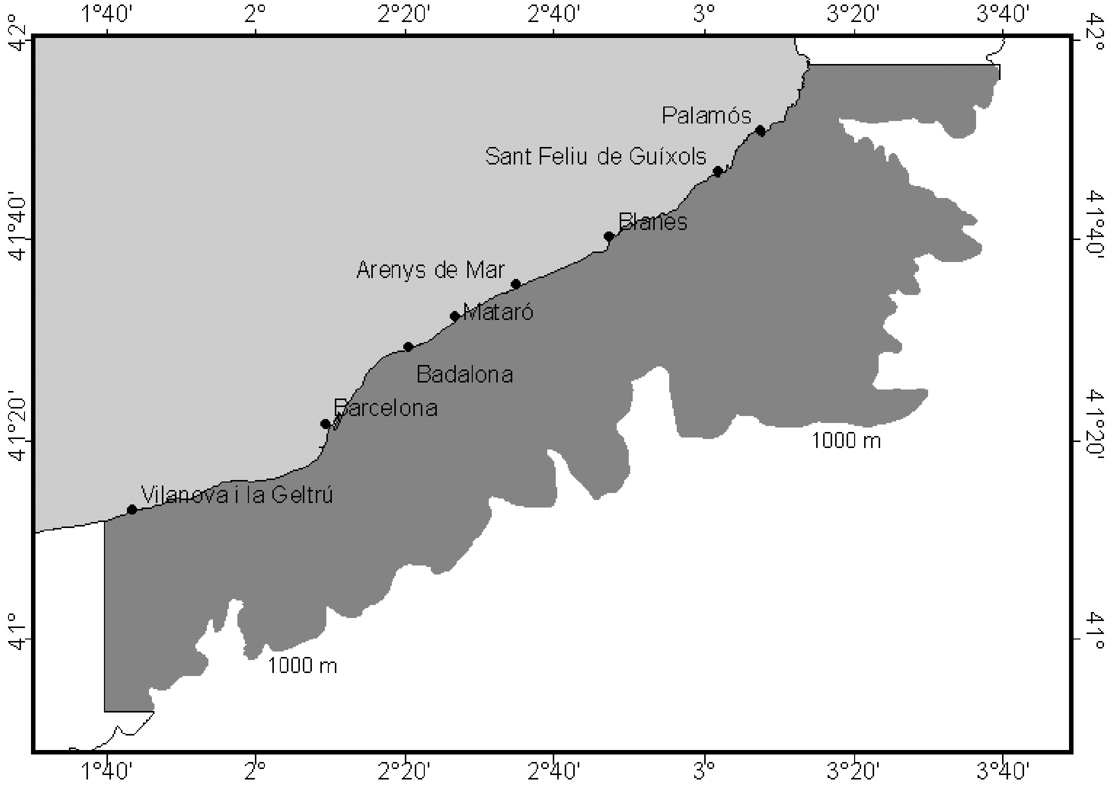

Fig.1. Map of the Central Catalan coast. The area of study (ranging from 0 to 1000 m) is shown (dark grey). The location of fishing ports is also specified.

landings of all demersal categories by considering a reliable figure of discards to be about 30% of actual catches, 60% of them being fish and the remainder 40% crustaceans. These estimations were based on extensive work on discards by the trawling fleet operating at the port of Vilanova (Carbonell, 1997). Catches for certain species like mackerel and bogue, harvested by both purse seines and trawlers were splitted according to the appropriate rates. Since no data on discards are available for the purse seine fleet, only raw total landings estimates were considered. All estimated catches from the eight ports analyzed were assumed to have been originated in the area considered with two exceptions. One refers to large pelagic catches (bluefin tuna and swordfish); given its highly migratory behaviour we considered that only half of its production should be assigned to the coastal ecosystem studied. The other involves small pelagic catches by purse seine fleets landing at the most extreme ports of the area, namely Vilanova in the South and Palamos and Sant Feliu in the North. Purse seine fleets in quest of small pelagics are highly

Table 1. Structure of the total fleet and relative weight of the different fleet segments based at the 8 ports located within the study area.

| | **N (> 9 m)** | **N (< 9 m)** | **Total N** | **GRT (t)** | **GRT (%)** |
|---|---|---|---|---|---|
| Bottom trawlers | 127 >12 m | 52 <12 m | 179 | 6643,4 | 55.4 |
| Beam trawlers | 7 | 31 | 38 | 166 | 1.4 |
| Purse seiners* | 125 | 3 | 128 | 3762,3 | 31.4 |
| Bottom longliners | 2 | 28 | 30 | 100,5 | 0.8 |
| Surface longliners | 1 | - | 1 | 10.8 | 0.09 |
| other | 45 | 364 | 409 | 1298.2 | 10.8 |
| **Total** | 307 | 478 | 785 | 11981.2 | |

** purse seiners are not obliged to land at the same ports where they are based. Small pelagic landings reflect, thus, approximate catches within the study area by boats of whatever origin.*

mobile and despite landing catches at the ports considered, a fraction of them could not have originated within the specified area. A correction factor of 0.5 was applied to landing estimations from these fleets.

The total amount of primary production required to sustain the estimated catches was computed as in Pauly and Christensen (1995), by applying the equation PPR = (catches/9) x $10^{(TL-1)}$, where TL is the specific trophic level, to each specific category of catch. This relationship is based on a 9:1 ratio for the conversion of wet weight to carbon and a 10% transfer efficiency per trophic level. Calculations were based on the set of specific trophic level estimates compiled by Pauly *et al.* (1998a), available at the Fishbase website (http//www.fishbase.org), duly adapted to our needs. A reliable estimate of 232 gC/m$^2$y for annual primary productivity in the study area was obtained from a map referred to the period 1997-1998 elaborated by the Institute of Marine and Coastal Sciences of the State University of New Jersey, Ocean Primary Productivity Team (http//marine.rutgers.edu/opp).

The resulting final estimate of the primary production required to sustain catches for the area analyzed, referred to 1996, is at about 40% of total primary production (39.85%; Table 2). Weighted mean TL of catches points to a value of 3.04. Our results for the Central Catalan Coast appear to be in good agreement with the magnitude of the mean %PPR calculated by Pauly and Christensen (1995) for world non-tropical shelves, 35.3%. These authors found mean world %PPR ranging from 1.8% in open ocean ecosystems to the above-referred value for non-tropical shelves. They are also higher than some scattered values available in the literature, i.e. 5% in the South-eastern Brazilian shelf, 24% in the Southern Brazilian shelf or 6.1% in the Eastern Bering Sea during 1980's, when the fishery was centered on harvesting finfish (Trites *et al.*, 1999; Vasconcellos and Gasalla, 1998). Estimated %PPR in this latter region during the 1950's, when human exploitation of the ecosystem was focused on whales, is however somewhat higher than ours, 47% of net primary production.

Table 2. Summary of PPR results.

| | |
|---|---|
| **Surface area (km$^2$)** | 7.487,221 |
| **Official catches (t)** | 24.080,9 |
| **Actual estimated yields* (t)** | 33.943,8 |
| **TL of catches (weighted average)** | 3,043 |
| **Primary Production (kg C)** | 1.739.955.300 |

| | **kg C** | **% PPR** |
|---|---|---|
| **PPR from catches** | 602.704.876 | 34,6 |
| **PPR from discards** | 90.748.004 | 5,2 |
| **Total PPR** | 693.452.881 | 39,8 |

** Including discards*

Regarding marine ecosystems, a fishing activity relying on as much as 40% of total primary production seems at first sight to be potentially very impacting (and certainly, it is). Nevertheless, this ecological index alone should be accompanied by additional information in order to achieve a more complete picture on the potential effects of fishing on the structure and functioning of the exploited ecosystems. In this sense, knowledge of mean TL of catches is extremely useful. Mean TL of 3.04 for the area of study suggests a mixed demersal/small pelagic fishery, heavily dependant on the rather low TL's of the latter target species. In effect, a remarkable specificity of many Mediterranean fisheries is the high pelagic/demersals ratio (Grainger and Garcia, 1996). Such features became reinforced in NE Iberian fisheries since the late 1960's, when a new fishery on anchovy rapidly developed and added to that traditionally existing on sardine. In Catalonia, landings of the three main small pelagic species combined (namely sardine, anchovy and horse mackerel) account for an average of 67% of total fish landings for the last 5 years. In this context, the occurrence of both a very high %PPR together with a low average TL of catches points to a dramatically high intensity of human exploitation of the ecosystem production in terms of both energy and biomass. This way, as far as it concerns Catalan fisheries during the past two decades, the conjunction of 1) high %PPR, 2) low mean TL of catches and 3) maximum historical records of catches, shows an emerging pattern of full human exploitation of the ecosystem. Most recent trends of the fishery, as highlighted by the severe decline of small pelagic catches and the stagnation of demersal catches in spite of the continuous increase in fishing capacity though technological improvements, seem to suggest that this huge level of exploitation is hardly sustainable. A very high %PPR on low trophic levels, specially in a mesotrophic coastal area, could lead to the highest possible impact on the structure and functioning of the ecosystem. For fishing, this could result in the interruption of major energy pathways to higher trophic levels and to the subsequent decrease of yields, as Vasconcellos and Gasalla (1998) state regarding a general fishing down marine food webs effect.

The %PPR presented here is a realistic assessment based on the best information available. Of course, there are factors not dealt with in this study pointing to both the overestimation and the underestimation of our results. Among the first it could be argued that the assignation of TL's to the demersal species did not take into account the ontogenetic differences in feeding, specially since overfishing catches for most of these species are based on juvenile, small individuals. Concerning possible causes of underestimation, it should be noted that not all the net primary production is susceptible to be fully suitable for higher trophic levels, as shown by Micheli (1999) in the case of production enhanced by anthropic run-off. This is of special importance for our results giving the growing claims attributing a human origin to the reported increase in primary production in the Mediterranean in recent years (Caddy, 1997). On another hand, in the absence of reliable information, no discards have been considered for the purse seine fishery. Also, the ecological role of discards remains to be adequately addressed since this by-product of fishing is both a fraction of human catches and a source of food for other marine species. The latter is especially true as far as sea-gull populations are concerned: in some parts of the Catalan Sea large populations of the globally threatened Audouin's gull are heavily dependent on them (Oro *et al.*, 1997).

In any case, the results presented here could be considered as a preliminary account of the higher ecological limits to the exploitation of coastal (shelf-slope) Mediterranean ecosystems by means of mixed demersal/pelagic fisheries. Further studies are needed to ascertain to what extent this ecosystem overexploitation is related to a potential parallel decrease in the resilience of the marine ecosystem. Taking into account the continuous perturbation caused by human activities on the marine systems (from pollution to global change), this information would be of paramount importance in the near future.