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**SELECTED PAPERS PRESENTED AT THE
WORKSHOP ON BIOLOGICAL REFERENCE POINTS**

Rome, 20–21 April 2004

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GENERAL FISHERIES COMMISSION FOR THE MEDITERRANEAN

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PREPARATION OF THIS DOCUMENT

Following a request of the General Fisheries Commission for the Mediterranean (GFCM) to its Scientific Advisory Committee (SAC), the Sub-Committee on Stock assessment (SCSA) held in Rome (Italy), in April 2004, a Workshop on Biological Reference Points. The Workshop was organized with the support of the Fisheries Directorate of the Italian Ministry of Agriculture and Forestry Policies (MiPAF).

The Workshop favoured the "Traffic-light" approach and identified a list of performance indicators and related criteria. As a follow-up, the SAC recommended that, in addition to indicators and reference points for different single stocks, specific indicators be identified for each fishery and operational units (poly-indicator system). SAC acknowledged the use of multispecies and ecological indicators as well as indicators derived from composite models and surveys and that each reference points should undergo a robustness and/or sensitivity test.

Fourteen papers were presented at the Workshop. It was agreed that they would be published by the Journal *Biologia Marina Mediterranea*. The present document reproduces a selection of five of the papers which were presented at the Workshop.

ACKNOWLEDGEMENTS

ABSTRACT

The workshop stressed that the concept of reference points (RPs) needs to be closely connected to the management objective of concerned fisheries, taking into account various parameters such as maximum sustainable yield, economic and social benchmark and environment concern. Participants stressed the need to define different RPs for each dimension of management or to identify a single RP aimed at ensuring the sustainability of the system, thereby no longer limited to biological aspects. A review of possible approaches to setting RPs and indicators for Black Sea fisheries emphasized the dynamic nature of recent ecosystem changes. The traffic light approach was illustrated as a means of following dynamic changes and gaining a broad perspective on events at the ecosystem level. The workshop outlined the main RPs obtained by applying the bio-economic optimization models of effort and effort-catch as derived from the theory of population dynamics. For the Mediterranean Sea, it was recommended to adopt a multicriteria approach based on bio-economic optimization models which, by introducing a number of constraints, would allow the definition of the actual state of the system as well as specific management targets. The use of some indicators derived from bottom trawl surveys carried out in the Adriatic Sea highlighted the potential suitability of selected indicators of their estimators. In the case of large pelagic species, it was confirmed that time series of both longline catch per unit effort and recruitment indexes can be considered as RP for these species. The use of a biological RP based on exploitation rate threshold was appraised for the stocks of anchovy and sardine in the central and northern Adriatic Sea and the workshop pointed out that it could be used to prevent stock collapse along with the minimum biological acceptable level based on spawning stock biomass.

CONTENTS

THE POTENTIAL USE OF INDICATORS, REFERENCE POINTS AND THE TRAFFIC LIGHT CONVENTION FOR MANAGING BLACK SEA FISHERIES

by

J.F. Caddy[1]

This paper is dedicated to Kamen Prodanov, who passed away recently during the course of a recent meeting on Black Sea fisheries. A very competent scientist, Kamen provided a guiding light and the necessary coordination for Black Sea fisheries assessment. His energy and expertise will be sorely missed in the subregion.

ABSTRACT

The potential use of indicators and reference points in management of Black Sea fisheries is reviewed, and fisheries management rules using RPs and indicators are discussed, including their use in stock recovery plans. A review of possible approaches to setting RPs and indicators for Black Sea fisheries emphasizes the dynamic nature of recent ecosystem change. This means that models using steady state assumptions may not be appropriate, and an empirical approach to defining indicators is explored. Indicators of ecosystem instability and risk are also proposed based on rates of decline and extent of decline of commercial species characteristic of different habitats. The traffic light approach is illustrated as a means of following dynamic changes and gaining a broad perspective on events at the ecosystem level.

Keywords*: reference points, traffic light approach, fishery management, Black Sea.*

Introduction

This document discusses the use of indicators and reference points (RPs) as they are developing in the Black Sea region, but goes beyond the point at which most discussions on reference points end – by focussing on what reference points are used for, since in both the Black Sea and the Mediterranean, the first priority is to advise managers on how indicators and reference points may be used for management. If these can be placed in a multispecies context, so much the better, but the approach initiated in the Black Sea region places emphasis on using specific species as indicators of the health of particular environments. From Black Sea

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experience, the variety of models of multispecies interrelationships proposed so far have come to diverging conclusions, and modelling may not be an unambiguous first step to setting RPs for active management.

First, the term "reference point" as used in this paper represents a particular value of an indicator series commonly recognized to mark the state of the resource or environment. In ICES and many fisheries commissions, finfish reference points are commonly defined from the fitting of models. The author has suggested elsewhere (e.g. Seijo and Caddy, 2000) that this is not necessarily the only approach, and more empirical approaches are emerging where data are scarce or ecosystem changes are dynamic (see also Caddy, 1999 and Gilbert *et al.*, 2000). Multispecies fisheries inevitably raise the problem of dealing with multiple indicators and their reference points. This issue is not touched upon in classical approaches, but is discussed here with respect to the traffic light convention and the need to display a wide range of ecosystem variables prior to beginning an ecosystem modelling exercise in a dynamic environment.

Indicators and reference points can be used in three main ways:

1) *Passive monitoring:* as a means of monitoring a phenomenon (e.g. overfishing, environmental change or stock condition) where immediate management action is not necessarily tied to the value of the indicator. This may be referred to as a 'passive' management mode, for example, where quotas are not applied, or where year to year levels of effort cannot be regulated in real time.

2) *Active management:* involves using indicators as components of a 'management rule' such that when a limit reference point value is exceeded (Caddy and Mahon, 1995), this supposes some action will be taken to restore the fishery to a safer condition. Stress is placed on the fact that determining reference points is not a "stand-alone" scientific exercise – these points have little significance if not applied by management! An example of a management rule is the COMFIE-type rule suggested by ICES (1997), which defines two types of RPs, the so-called precautionary reference points B_{pa} and F_{pa}, and two limit reference points, e.g. B_{lim} and F_{lim}. These are generic reference points, in that they mark decision points of the rule, and can be derived either from models or based on well substantiated and accepted empirical values. There are problems in practice in applying a COMFIE style rule, discussed in Caddy and Agnew (in press), but the underlying concept is clear: indicators and reference points are needed to drive a management rule. What is more important, is that the fishing industry should understand the basis and utility of the reference points proposed.

3) *Stock recovery plans:* an extension of the use of a rule in routine management is its use in a stock recovery plan. Caddy and Agnew (in press) review a range of fisheries where recovery plans have been used. This application presupposes another class of reference points defining not only the fishing mortality and biomass levels at which recovery plan actions should be triggered, but also the target reference point expressed in terms of the spawning potential or biomass at which the population is considered recovered. Defining targets for recovery of depleted stocks is in itself a worthwhile activity.

In summary, a focus on reference points implies that the infrastructure and internationally-agreed regulations are in place allowing some form of management rule to be applied. This currently appears not to be the case for most Mediterranean and Black Sea fisheries. Quota control is inexistent here, and mechanisms used to maintain fishing mortality within reasonable

levels such as fleet capacity control, area and seasonal closures and technical measures, cannot be applied easily in real time. Other approaches to formulating management rules need to be urgently considered than the conventional approaches which are built around quota control.

Use of indicators and reference points in active management

While there have so far been few examples of the use of reference points in active management in the Mediterranean and Black Sea, the Black Sea Commission is currently exploring the possibilities of using an approach to formulating a management rule which takes into account both impacts of fishing and environmental/ecosystem change. A past analysis of previous stock assessments (Prodanov *et al*., 1997) provides material that allows us first to explore several different approaches to monitoring stock changes and defining reference point values, and potentially incorporating these data sources into active management.

Figure 1

THREE APPROACHES TO VISUALIZING FISHERY
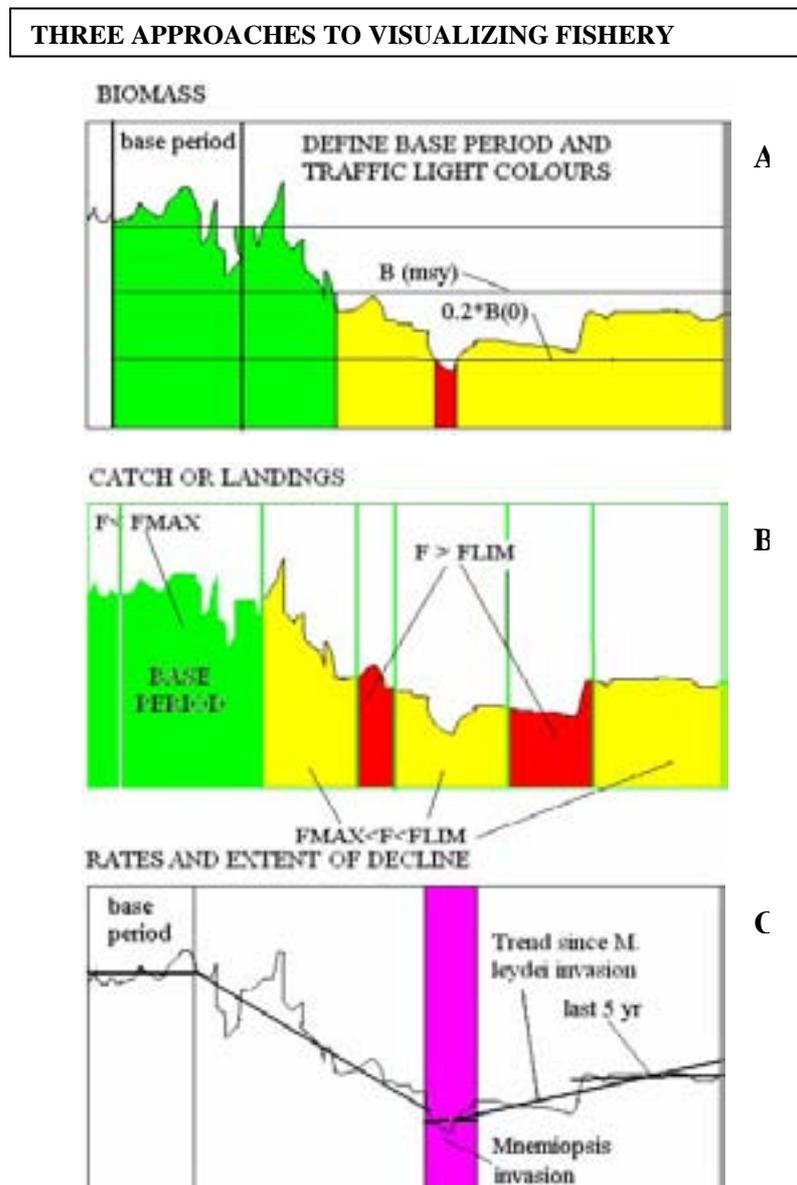
Use of trend analysis and the extent of decline in landings

Knowing that 1989-92 were the peak years of the *Mnemiopsis leydei* outbreak, one approach to defining indicators is to use this priori environmental information to segment data series using linear trends (Caddy *et al*., 2004) (Figures 1 and 2). Another is to fit long-term trends using polynomials (Fiorentini *et al*., 1997, and Figure 3). Rather than just focussing on changes relative to recent periods when the stock may already have been depleted, a FAO (2001) working group examined criteria for listing endangered fish stocks by CITES, and stressed the importance of examining the "rate of decline" over the short and long term, as well as the "extent of decline" from earlier historical periods. As a result, two types of indicators were examined, those for:

a) rate of decline (e.g. in biomass and catch);
b) extent of decline from a benchmark or "baseline period" (presumably when the ecosystem was in a "safe" condition). In both cases, critical values for extent and rate of decline could be used to establish reference points that trigger stock restoration.

Figure 2
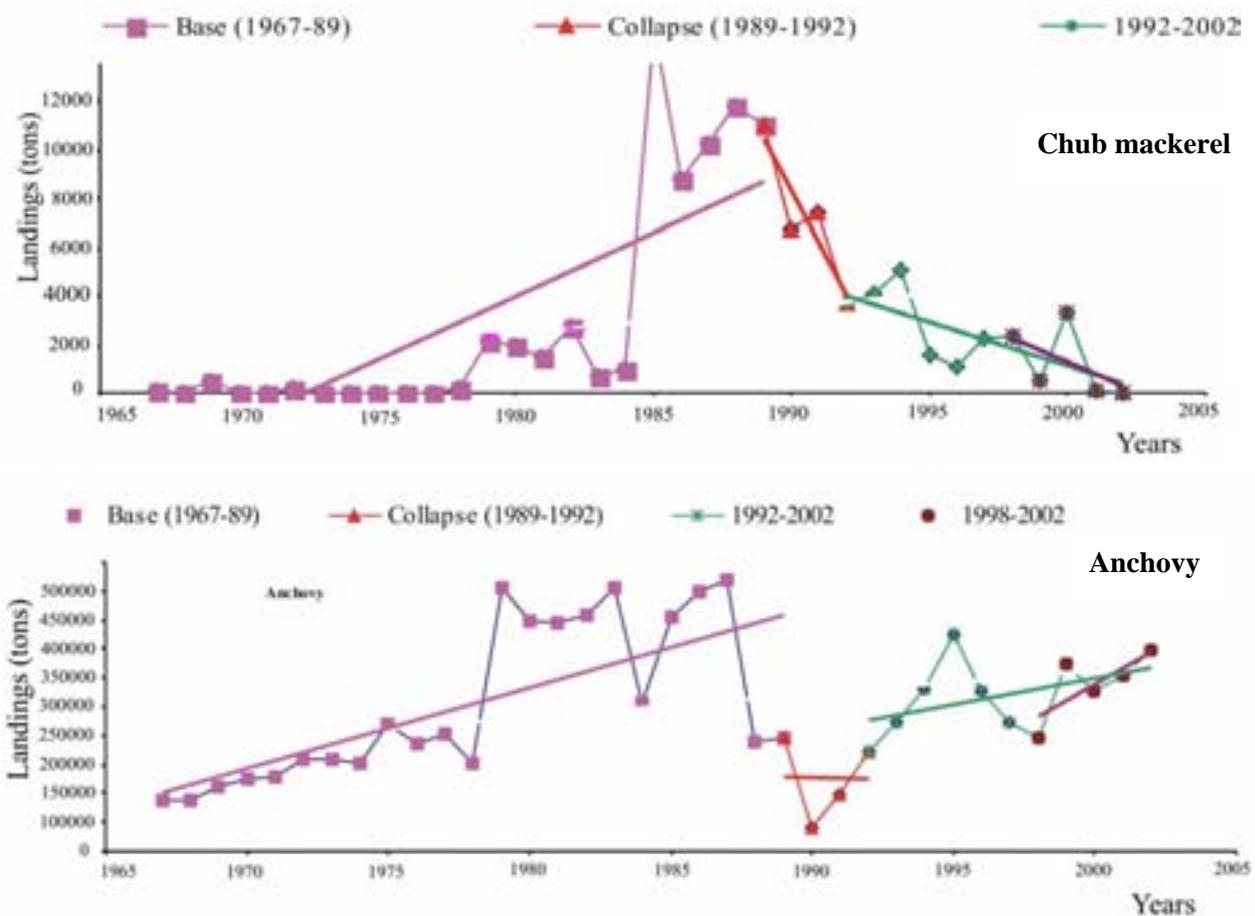
Evidently, chub mackerel and anchovy, with differing degrees and timing, both showed landing increases during an arbitrary "base period" at the start of the data series, and for both, a serious drop in landings occurred during the main *M. leydei* bloom, 1989–92. A subsequent recovery of anchovy landings has occurred over the last five years. In contrast, a continued long-term decline in landings occurred for chub mackerel (Figure 2) which dropped steadily to close to zero over the last five years on record. This comparison suggests that different ecological stresses applied to resident and migratory pelagic species. Table 1 extends this analysis to 14 important commercial taxa, and suggests that general ecosystem indicators could be the collective recent decline in landings (penultimate column), and the extent of decline (last column). Using these two criteria together, the mackerel (*Scomber scombrus*), bonito (*Sarda sarda*), Mugilidae and *Rapana* sp., emerge as priorities for management attention. Overall, the highest proportion of declining trends occurred in 1989-92, but a high ratio is also seen at the bottom of table 1 for the last five years on record, suggest that if "a recovery" did follow the *Mnemiopsis* outbreak it was short-lived.

Table 1

CATEGORY/ SPECIES	Baseline trend (before 1989)	Trend in 'Collapse years': 1989-92	Trend in 1992-2002	Trend over Last 5 yrs	Last 5 years (1998-2002) landings as % of baseline
RESIDENT PELAGIC SPECIES:					
Anchovy (*Engraulis* sp.)	+ +	-	+	+ +	136
Horse mackerel (T*rachurus* sp.)	+ +	- -	-	+	14
Sprat (*Sprattus* sp.)	+ +	- -	+ +	-	68
MIGRANTS :					
Mackerel (*S. scombrus*)	-	+	+	-	2
Chub mackerel (*S. japonicus*)	+ +	- -	-	-	42
Bluefish *(Pomatomus* sp.)	+ +	-	+	+ +	196
Bonito (*Sarda* sp.)	0	+	+	- -	5
DEMERSALS:					
Turbot (*Psetta* sp)	-	-	+	+ +	70
Whiting (*Merlangius* sp.)	+ +	- -	-	-	51
Spiny dogfish (*Squalus* sp.)	+	- -	-	+	32
Mullets (*Mullus* sp.)	-	+ +	0	- -	164
Mugilidae	0	++	-	- -	19
Gobies	+	- -	+	- -	139
Rapana	N/A	N/A	+	- -	13 (of '92-2002)
Ratio +ve/-ve	8/3 = **2.67**	4/9 = **0.44**	8/5 = **1.60**	5/9 = **0.55**	
KEY : (++): steep +ve slope (-): negative slope 0 : no trend					

Although the trend analysis in the above table is only indicative, for a significant proportion of commercial species, landing trends were generally upwards in the first period, dropped seriously in 1989-92 when the *M. leydei* "invasion" was at a peak (e.g. Mutlu *et al.*, 1994), and showed a "partial recovery" subsequently. However, when the last five years are considered separately, the apparent recovery looks less certain, except that landings of some species (anchovy, bluefish, gobies and Mullidae) seem to have staged a "comeback", while a benthic

species (*Rapana* sp.), and from the lagoon and coastal group, the Mugilidae, seem to be declining, as are immigrants such as the mackerels.

Although the single-species effects of fishing can be deduced from retrospective analysis of several Black Sea stocks (Prodanov *et al*., 1997), individual analyses do not help to integrate the whole ecosystem picture. Recent work within working groups of the Commission has tried to reconcile the effects of overfishing with environmental change. This includes nutrient runoff and the effects of the invasion by *M. leydei* on the pelagic biome. Such effects are complicated by the socio-economic consequences of fleet overcapitalization. Up to now, coordinated action by coastal states to manage shared fishery resources is at an early stage, but is now being addressed by the Black Sea Commission. As a first priority, the approach is to list existing indicator series and possible reference points, with a view to further exploring their use in fisheries management rules. Annex 1, though not referred to specifically in this communication, and table 1 above, underline how conventional assessments of specific faunal components can also provide indicators of the health of particular environments or habitats.

Multispecies indicators and reference points

The theme of this meeting is the use of reference points in multispecies situations. Given the categorization of indicator use given in the Introduction, the immediate practical application of reference points in the Mediterranean and Black Sea areas is likely to be within a passive management or monitoring category. The theoretical and practical complications of modelling complex ecosystems as a basis for a management rule raises serious problems for fisheries managers, notably the need to reduce a range of complex data series to a decision rule. It is almost axiomatic that size selective processes of fishing, by reducing the mean size of the surviving fish in the sea, will reduce the mean trophic level, but so will nutrient runoff by inflating the base of the food pyramid. This illustrates the problem facing fishery managers in trying to translate a change in the level of a multispecies indicator into specific action.

If truncation of top predators has occurred through overfishing apical predators (e.g. Pauly *et al*., 1998), the appropriate reference points for applying management action might be single-species reference points for top predators. For inland seas, ecosystem change is also likely to have a strong bottom up component (Caddy, 1993; De Leiva Moreno *et al*., 2000). If reduction of mean trophic level is due to enhancement of basic productivity, environmental controls on nutrient runoff might deserve priority. The issue of deciding what reaction is appropriate in response to changes in data-intensive multispecies indicators has to be dealt with early on. It has been evident since the 1970s that the dynamic nature of ecosystem change in the Black Sea makes "steady state" models not very useful. It is also important to add information series measuring environmental conditions, since if stock-recruit models are used to define reference points, care must be taken that environmental conditions are not the main factors influencing recruitment success.

Contrasting views have been expressed and supposedly supported by different mathematical models of the Black Sea system (Table 2). The wide range of these suggests that modellers are influenced by preconceptions, if only in choosing the information to incorporate in their models, and this can be misleading if a range of variables have influenced events. Taking a broad scale approach to monitoring a wide a range of indicators is recommended prior to attempting a specific modelling approach, and Figure 9 shows that several critical factors influenced events, and the dominant factor may have changed between the 1960s and the start of the millennium.

Table 2

| Author	Primary causes of ecosystem changes	Mechanism/resulting effects
Christensen and Caddy (1993)	Two static models of the Black Sea food web are presented: showing a) predicted effect of *M. leydei* on the pelagic ecosystem, and b) hypothesized effects of introducing *Beroe* sp. as a controlling predator on *M. leydei*.	After introduction of *M. leydei* in the early 1990s, it grossly dominated organic flows through the pelagic ecosystem, but given absence of predators, it short-circuited flows to the benthic bacterial loop. Introduction of *Beroe* was predicted to reduce flows of material to detritus.
Mutlu *et al.* (1994)	*M. leydei* has a shorter generation time than *Aurelia* and small pelagics.	As a result it reduces food availability for these competing species at the same trophic level.
Kideys (1994)	Increased nutrient inputs led to abnormal phytoplankton blooms. Introduction of *M. leydei* was a key event that radically changed the ecosystem.	Competition of jelly predators with small pelagics for zooplankton led to the collapse of small pelagic stocks.
Aubrey *et al.* (1996).	Anthropogenic effects on the N-W shelf near the Danube mouth are predominant due to nutrient runoff.	The interplay between high Danube nutrient loadings and Black Sea hydrological fronts provide opportunities for enhanced biological activity.
Daskalov (1999)	Recruitment of small pelagics is less dependent on parental stock size than environment variables (e.g. wind stress).	Recruitment of small pelagics in the Black Sea is predominantly influenced by environmental changes.
Berdnikov *et al.* (1999)	A specific Black Sea+Azov trophic model suggests that "bottom-up" effects are predominant lower in the trophic chain. "Top down" effects become only evident higher in the food pyramid.	Trophic competition between anchovies and *M. leydei*, rather than predation by the latter on anchovy larvae, was key factor for anchovy decline. Decomposition of unpredated *M. leydei* might destabilize bottom oxygen levels.
Rass (2001)	The reduction of cold spring flood outflow from Black Sea rivers due to damming rivers damaged water exchange.	This reduced the Rumelian stream in the western Black Sea formerly used by migratory species.
Gucu (2002)	A minimum role assigned to the *M. leydei* outbreak on the basis of a steady-state model.	Though eutrophication in the 1980s led to the outburst of jelly organisms, the decline in stocks was mainly due to overfishing.
Daskalov (2002)	Onset of industrial fishing and depletion of top predators (dolphins and migratory pelagics) in the early 1970s led to a trophic cascade affecting events for the next 30 years.	Deleterious events are explained mainly by top-down release of predator control on small pelagics: increased nutrification is supposed to only increase the biomass of all components in the model.

Empirical approaches to deciding on reference points

One school of thought (e.g. Gilbert *et al*., 2000; Seijo and Caddy, 2000; Caddy *et al*., in press) suggests that since reference points need to be implemented in a fisheries rule, they must have credibility and be understandable to the fishing industry. Although the classical approach has been to generate RPs from yield or SRR models, the assumptions underlying "generic" models may not fully apply, since they usually depend on an assumption of stability or equilibrium that is not tenable given the major ecosystem changes to the Black Sea that have

certainly occurred. The various models applied to Black Sea resources and environments mentioned in Table 2, differ dramatically in the prime causes assigned to the ecosystem/fishery changes observed. It is clear from this that the axioms and the data used by each model have differed. In these circumstances, establishing a series of indicators reflecting changes at different levels in the ecosystem and its physical environment, without supposing a specific mechanism, seems the logical first step prior to modelling. Expert judgement will be needed to establish boundaries corresponding to serious risk of overexploitation or depletion, but all likely driving forces need to be taken into account.

Trenkel and Rochet (2003) and Rochet and Trenkel (2003) note that most multispecies indicators to date have been based on theoretical considerations. "Empirical" population indicators such as mean length in the catch, the pelagic/demersal index (De Leiva Moreno *et al*., 2000 and Figure 3), the overall exploitation rate, the proportion of non-commercial species, and the proportion of piscivorous fish in the commercial catch (Caddy and Garibaldi, 2000) were found to be statistically more reliable than estimates of exploitation rate or indicators based on food web modelling. A similar conclusion on trophic modelling was reached by Jennings *et al*. (2002), from stable-isotope analysis of food web components, and by Patterson (1992) for small pelagics. Mean trophic level and the pelagic/demersal ratio (Caddy, 2000; De Leiva Moreno *et al*., 2000) have theoretical disadvantages as indicators in that they could be indicators of increased nutrient inputs as well as overfishing. An example of the use of trophic models for generating indicators is the huge effort put into developing the MSVPA model for the North Sea. This has been scientifically revealing, but is not been regularly used for fisheries management, but mainly to estimate natural mortality rates in single species assessments. This in part because diets are highly variable with season and age, intrinsic assumptions on ration size are suspect; and in part also because the extensive sampling required to fit multispecies models to a changing ecological situation in real time would be prohibitively expensive.
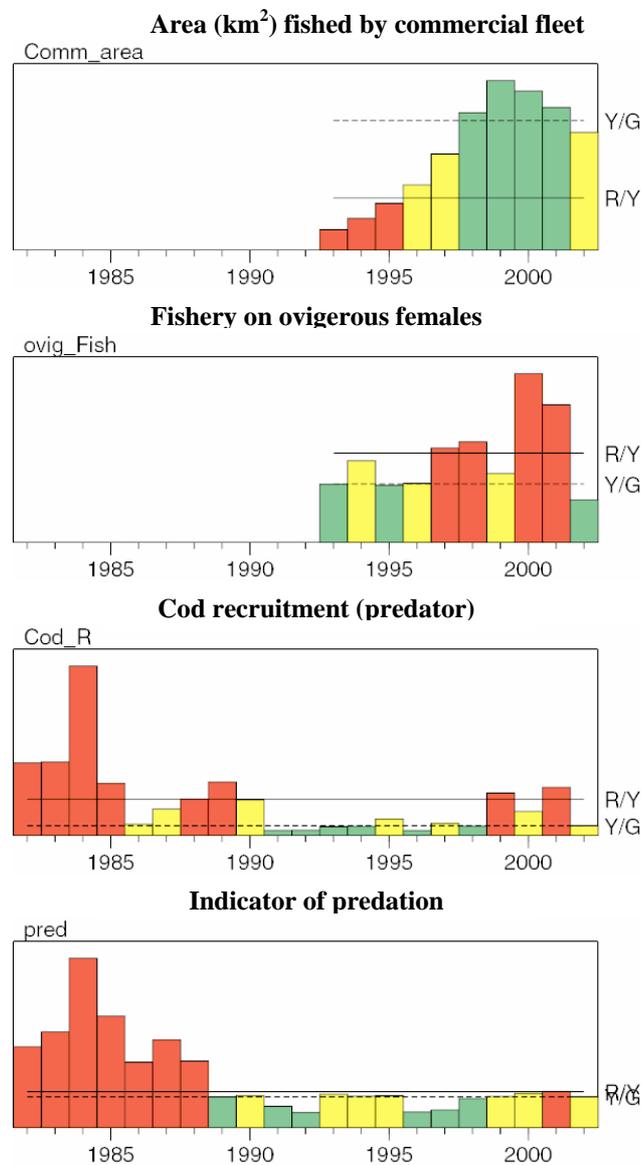
Figure 3
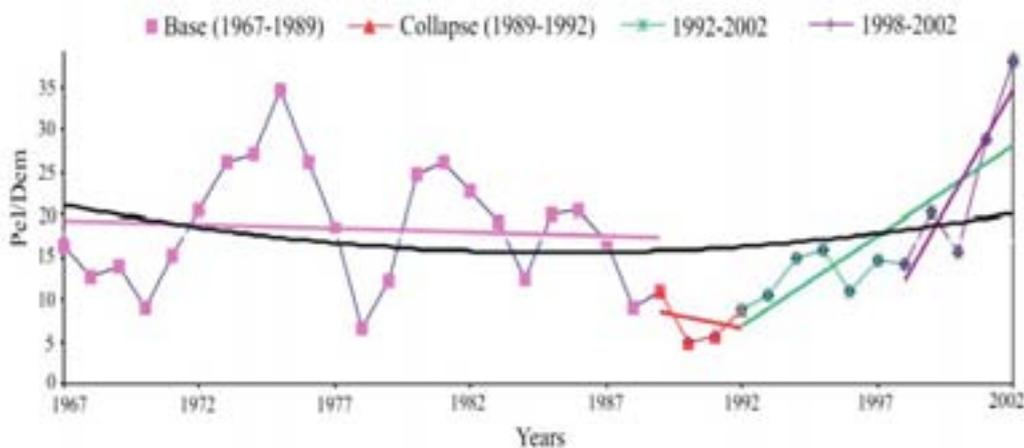
Displaying multiple indicators – the traffic light approach

The example in Figure 4 illustrates a procedure for dealing with multiple indicators, including predator recruitment and biomass (Koeller *et al*., 2000). Several indicators in a traffic light array may measure various population characteristic such as biomass or mortality directly or indirectly. Judgements made from a knowledge of life histories, or from previous events in the same fishery, may also be appropriate. Such an approach has been referred to as a traffic light monitoring methodology (Caddy, 1999; Halliday, Fanning and Mohn, 2001). The following example for a North Atlantic shrimp fishery shows the multiple indicators used in "shrimp monitoring", based mainly on surveys and catch analysis in an essentially "passive" management mode. Figure 4 shows for example, how expansion of the shrimp stock in the late 1990s coincided with a decline in cod (predator) biomass and recruitment (probably both due to a decline in ambient temperature favouring *Pandalus borealis,* but not cod)**.**

Figure 4

Area (km^2) fished by commercial fleet

Fishery on ovigerous females

Cod recruitment (predator)

Indicator of predation

Sequence of development of indicators and reference points in fisheries

The classical reference point approach is to use one or two generic model-based indicators and their reference points (ICES, 1997) as input to a fisheries management system (Figure 6), but these may not be easily estimated in absence of age-structured sampling. An alternate approach illustrated in Figure 5, is to develop a system of indicators and reference points for a range of population characteristics. This may also help to introduce biological realism into population monitoring. In developing indicators for Black Sea fisheries, one advantage is the lower species diversity than in the Mediterranean proper, but as shown by Figure 9, there are important driving functions operating in the Black Sea in addition to overfishing. Environmental changes associated with eutrophication and the impact of the introduction of exotic species have also affected the pelagic and demersal biomes (e.g. Zaitsev, 1993).

Figure 5

Sequence in development and use of reference points for fishery management and accompanying indicators (italics)
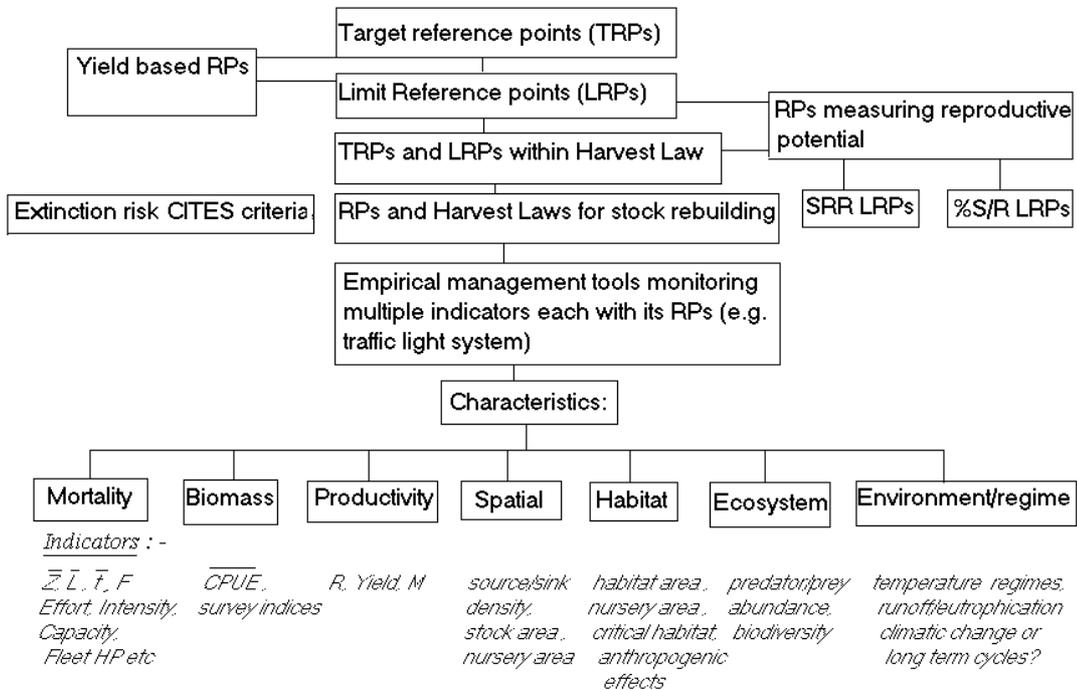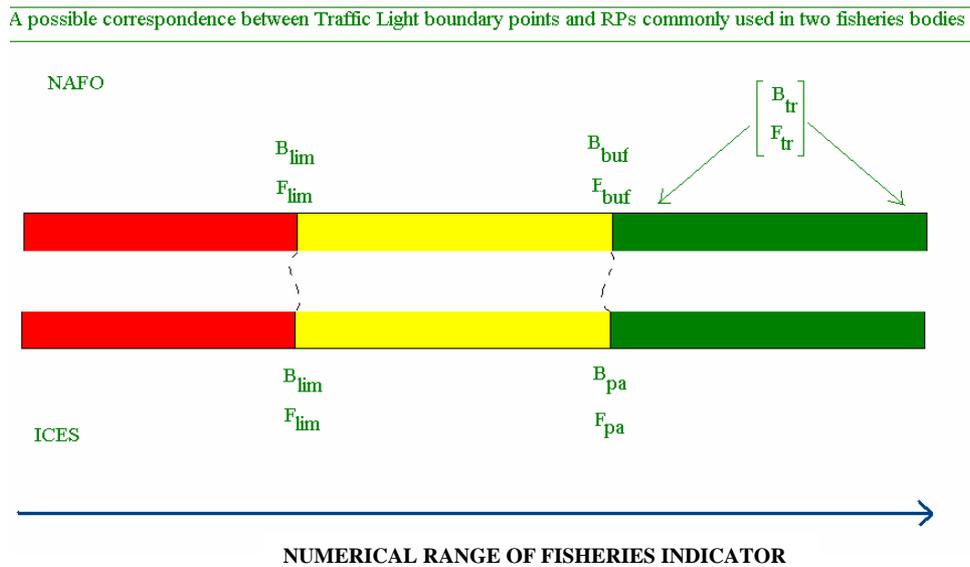
Figure 6

A possible correspondence between Traffic Light boundary points and RPs commonly used in two fisheries bodies

NUMERICAL RANGE OF FISHERIES INDICATOR

Choosing critical values for indicators

For an indicator to be useful, it must be possible from independent information or analysis to show that changes in indicator values correctly reflect changes in variables such as fishing pressure, biomass, or species composition which may not be easily measured directly, and the indicator must be easily understandable to non-technical audiences.

A form of graphical presentation was experimented with in Caddy (2002) and Caddy *et al*. (2004) which seems to provide insights into trends ongoing in the fishery. In the lower graph of Figure 7 for sprat, which was based on retrospective analysis of annual age composition data (Prodanov *et al*., 1997) traffic light colours have been assigned to annual catches depending on the ratio of the annual fishing rate to the natural mortality rate. This colour convention assumes that for small pelagics a fishing rate below that of the natural death rate is precautionary, given the high risk of death from natural causes that already applies (Patterson, 1992). For turbot, an alternative approach assigned colour ranges based on the ratio of F to an estimate of $F_{0.1}$ provided by Prodanov and Mikhailov (2003). Both cases illustrate that despite an apparent recent "recovery" of landings in these fisheries after 1992, the exploitation rate also increased towards the end of the time series of landings, adding to the high current probability of overexploitation. This type of graph which combines two indicators in one, offers an efficient summary of the situation, and can be convincing when discussing research results with industry and non-technical fishery managers.
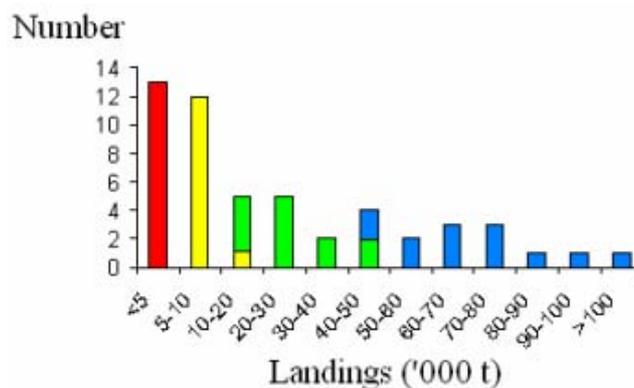
Figure 7
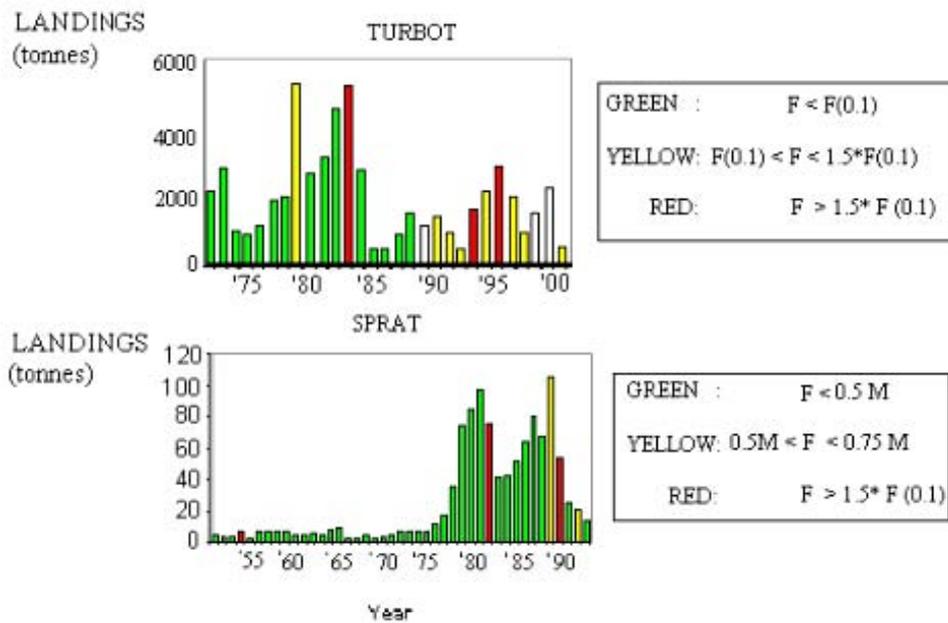
Empirical boundaries for indicators

Ideally, traffic light boundary points at the interface of green and yellow, and between yellow and red, should reflect a specific change in status, or segment the time series using values believed to represent important features of the population, or key events in the time series. However, in absence of data on specific boundary positions, the following empirical approach seems worthwhile for illustration purposes, and does not necessarily require such judgements. An empirical probability distribution function (pdf) is created from a time series of values, which are first ranked, and then as illustrated in Figure 8, the observed range for the variable during the time series is divided (for a 4-colour TL) into quarters; each segment containing as closely as possible the same number of years, and each assigned a different colour value using a small Excel macro. An alternative approach would be to segment the historical or feasible range of an indicator equally into colour segments, and this second approach was used by Caddy and Surette (in press) and Caddy *et al*. (in press) may be more compatible with traditional RP approaches, but is not used in the current paper.

Figure 8

The above empirical approach allows a large number of indicators to be displayed together. Such an approach to monitoring changes in a wide variety of Black Sea indicators is shown in Figure 9.

Although it would be desirable to divide the range of an indicator into zones corresponding to safe, uncertain, and dangerous conditions by boundary values tested by experiment or experience, this may not readily achieved. Dividing the observed range of an indicator into three or four zones, each containing close-to-equal numbers of years, seems one useful first step in making comparisons between indicators prior to formal ecosystem analysis or modelling; or alternatively as just mentioned, dividing the observed range of the indicator values into several colour segments which may not contain the same number of annual points. In Figure 9, for most indicators shown, the procedure in Figure 8 was followed, with the exceptions mentioned in the caption to Figure 9. Several relationships seem to emerge from the multiple comparisons within this figure that deserve further investigation:

1) Phytoplankton standing crop in the 1960s to early 70s was relatively low, but rose in the 1970s-80s presumably due to eutrophic effects associated with hypoxia of Black Sea shelves (Zaitsev, 1993). Landings of "resident" pelagics (anchovy, sprat and horse mackerel) increased in the 1980s as phytoplankton abundance peaked (red). Blue colouration of very high landings may therefore be an indication of what may be anomalously high productivity conditions for resident pelagics associated with eutrophication. Anchovy, and horse mackerel (and to a lesser extent, sprat) catches dropped drastically at the end of the 1980s, coinciding with the *M. leydei* outbreak, and zooplankton levels (not shown) also declined. Sprat and anchovy catches later recovered, but horse mackerel catches did not, nor did migratory pelagics until towards the end of the time series, and mackerel catches are still low. Blooms of jellyfish *(A. aurita)* and later and more drastically, the introduced ctenophore, *M. leydei*, led to competition with anchovy for food (Berdnikov *et al*., 1999). The impacts of this jelly predator bloom decreased after the mid-1990s, and a recovery of anchovy production occurred (perhaps due to the introduction of *Beroe ovata*, a predator on *M. leydei* into the Black Sea?). The effective fleet size fishing pelagics rose to still higher levels after anchovy landings had "recovered".

2) High phytoplankton abundance (red) also coincided with declines in catches of migratory pelagics (the early 1970s), (though catches outside the Black Sea also declined over the same period, suggesting that this decline may not have been specific to the Black Sea). (The "Out" indicators for migratory pelagics in the above figure consist of summed catches by all countries in the eastern and central Mediterranean except Turkey and Black Sea States, and is a proxy for stock abundance in the Aegean, Marmara Seas and eastern Mediterranean. Unlike the Black Sea fishery however, catches of bonito outside in the Mediterranean rose towards the end of the time series, perhaps suggesting that poor Black Sea catches after the 1980s were not due to a decline of the Mediterranean + Black Sea bonito resource? Timing of the rise in phytoplankton production seemed to coincide with a reduced entry of migratory pelagics into the northern Black Sea after 1980, suggesting the hypothesis that immigration of migratory pelagics through the Bosphorus to the Northern Black Sea may have declined after 1980; perhaps due to environmental deterioration and/or changes in current patterns, rather than due to a stock collapse *per se*.

Figure 9

3) Other ecological instabilities became evident in the early to mid-1980s with progressive disappearance of the *Phyllophora* (red algae) fields in the NW Black Sea. This was suggested during the GFCM meeting on Black Sea fisheries in 1993 as due to lower water transparency and bottom water hypoxia caused by organic debris from high planktonic production. Such a deterioration in benthic environments would have affected demersal resources, since decimation of macrophytes (*Phyllophora*) and reduction in shelf resources (turbot) tend to occur synchronously in Figure 9, also with the decline in migratory pelagics, suggesting that water quality rather than stock decline may have been a causative factor in both cases. The coincidence of increased jellyfish (*A. aurita*) abundance, and rises in landings of resident pelagics, presumably also resulted from increased system productivity. The general increases in fishing capacity of the Turkish pelagic fleet towards the end of the time series, which is currently much the largest in the Black Sea, presumably also imposed higher exploitation rates on the pelagic resources later in the series.

4) Although deductions based on Figure 9 cannot exclude any particular mechanism, it is difficult to see how the collapse of predatory migratory pelagics and demersals in 1973-77 led directly to the collapse of small resident pelagic populations in 1989-92, when both *Mnemiopsis* introduction and dramatic growth in capacity of the pelagic fleet occurred in the intervening years.

Figure 10
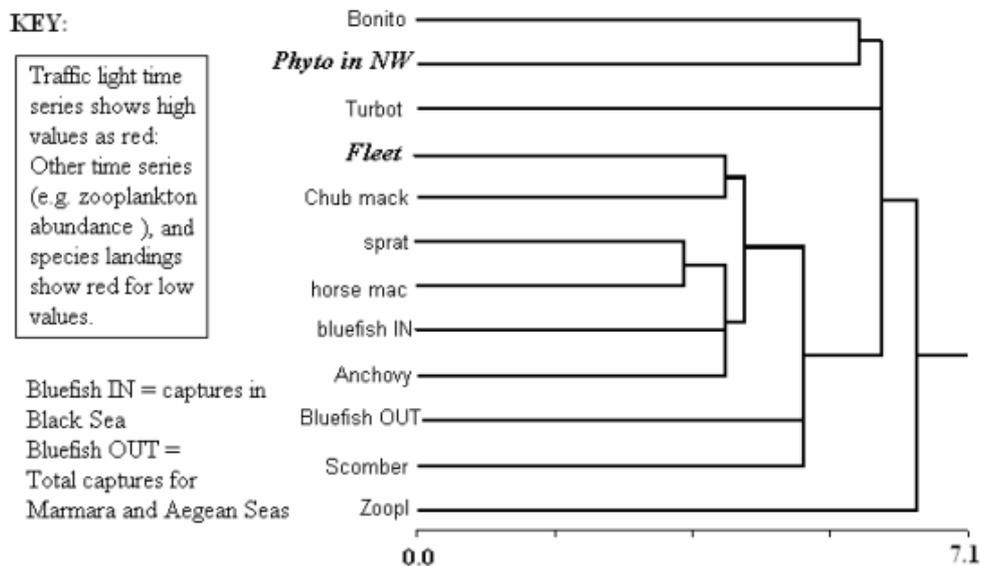
Cluster analysis was performed on the time series shown in Figure 9 for the period 1970-89 (StatistXL software: http://www.statistiXL.com), after converting colour values (r,y,g,b) -> numerics (1,2,3,4). Over this period the data set was continuous (Figure 10). This procedure allowed similarities between indicator series to be detected and the indicator series were then grouped into characteristics (though this was not done for the example shown). Cluster analysis

revealed other features of this data set, remembering that unlike the other traffic light series, high values for phytoplankton and fleet capacity have been coded as unfavourable (i.e. red):

- The three resident pelagic fish and one migratory (bluefish taken in the Black Sea) show similar trends, as do catches of two migratory pelagic fish (bluefish catches from outside the Black Sea, and *Scomber* sp. taken inside).

- The inverted colour series for fleet capacity showed a general similarity with the catch trends for small pelagics, (implying that increases in fleet capacity were inversely related to pelagic catch trends).

- At a lower level of similarity, the inverted time series for phytoplankton abundance in the NW Black Sea resembled that for bonito and turbot (implying that declines in catches of bonito and turbot may have been related to phytoplankton abundance).

This analysis can only be indicative and preliminary, but the above hypotheses are suggested directly by the original data without fitting a model.

A complex sequence of events with its hypothesized multiple effects and interactions can only be presented as hypotheses, but Figures 9 and 10 illustrate that the sequence of events cannot be easily explained as simply a trophic cascade effect resulting from a release in predatory pressure, (though some cascade effects undoubtedly applied earlier in the time period). The significant increase in human "predation" over the time series has probably more than made up for the decimation of natural predators. *Mnemiopsis* should be regarded as a competitor more than a predator for the resident small pelagics, since it occupies much the same trophic level. As suggested by some authors, this ctenophore was allowed to dominate the ecosystem niche in part due to overfishing, which released predatory control on zooplankton abundance. Further discussion of the possible hypotheses and their different consequences for management are discussed in GESAMP (1995). It seems that a graphical approach displaying all available indicators may be a useful first step to more intensive analysis and modelling of ecosystem indicators.

Using SRR data to establish indicators

Cohort analysis has rarely been applied in the Mediterranean, but more frequently so in the Black Sea. Another traffic light approach would be to segment SRR data into poor, average and good years for reproduction based on data on spawning stock biomass and recruitment from cohort analysis (Figure 11). Years were divided into good (green), medium (yellow) and poor (red) levels of recruitment, used an index, the log ratio of recruitment/spawning stock biomass using data from Prodanov *et al.* (1997 - table 48). The data points were divided into three segments containing equal numbers by lines through the origin. This procedure is similar to that of Sissenwine and Shepherd (1987) for obtaining the RPs, F(low) to F(high). Contrary to their assumption of stock equilibrium however, Figure 11 for Black Sea sprat shows that recruitment success is to a large extent independent of, (or even inversely related to) spawning stock size. It is interesting that sequences of poor and good years generally occur closely together, and the highest productivity years often correspond to low spawning stock biomasses (SBBs), though whether environmental conditions as well as stock biomass is the critical factor is not clear.

Figure 11
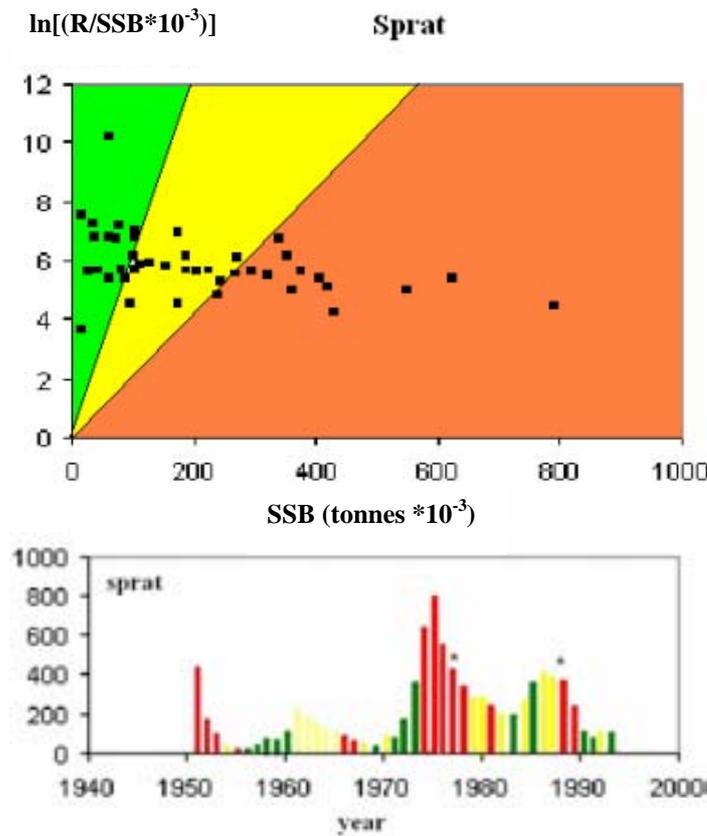
Use of a fisheries control rule employing indicators and reference points

Here, we discuss a simple management rule where there are two pairs of reference points marking a sharp jump from one "phase" of the fishery to another (Figure 12, and upper Figure 14). This is realistic, since for most fisheries data, the "noise" is too high to justify a management model dependent on precise estimates of indicator values.

Figure 12 represents such a management rule, illustrating two legal criteria which have been used in the USA to manage stocks and define which stocks must be restored. Two triggers for action are specified: a stock can be "**overfished"** (i.e. the resource is depleted), and/or "**overfishing"** is currently occurring (i.e. the current level of F is too high, whatever the stock size). These two risk factors are treated independently along the two axes below, each with it appropriate RPs, such that routine management and stock rebuilding emerge as two components of the management procedure. A hypothetical time series of events during rebuilding is illustrated in Figure 12 (modified from Caddy and Agnew, in press). A specific type of management rule is illustrated in Figure 13 which is designed for rebuilding depleted stocks, in which a biomass-based reference point may be set as a rebuilding.
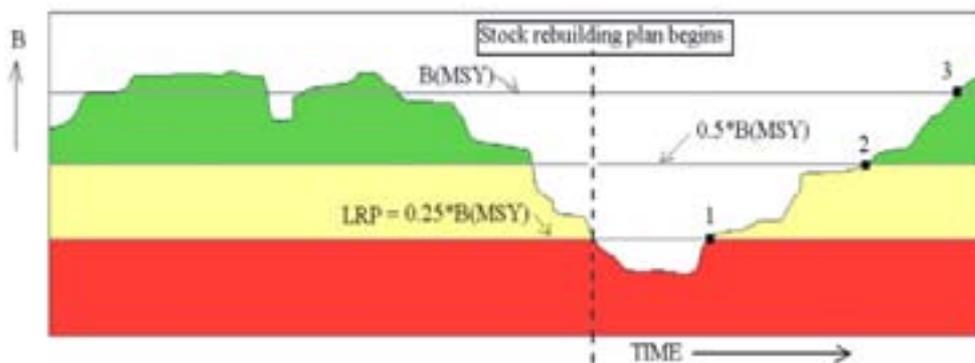
Figure 12
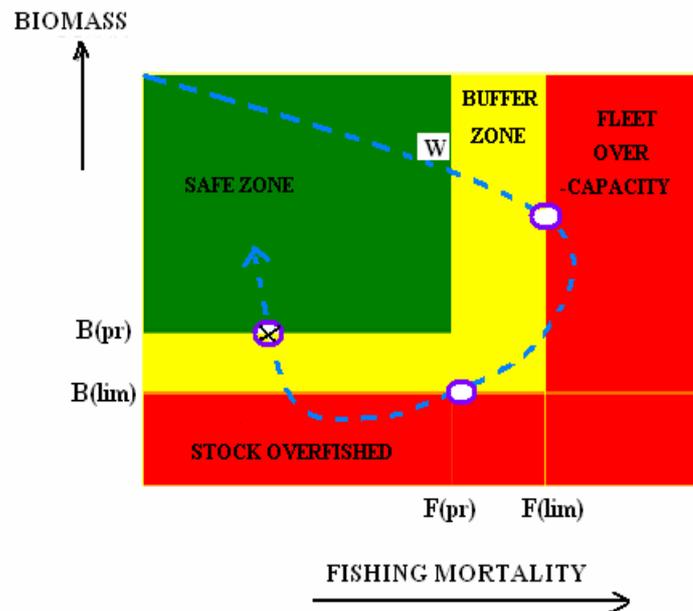
Figure 13

The conventional type of management rule presented in Figures 12 and upper Figure 14, uses only two indicator series to manage the stock: (spawning) biomass and fishing mortality rate. Because such a rule is based only on two indicators, it risks missing the ecosystem and environmental dimensions of the management problem. Another approach being considered for management of snow crab fisheries in the Gulf of St Lawrence (Caddy *et al*. in press) is potentially a biologically more precautionary type of control rule: in that case, if the percent of "soft-shell" unmarketable crabs exceeds a critical value, local fisheries are closed (but the same principle could be use with other biological data):

a) Control points may be, but do not necessarily have to be, derived from a model; they can incorporate arbitrary values for biomass and percentage harvest as long as there is industry consensus.

b) A supplementary biological rule can be incorporated for local fishing areas, such as the closure of a sub-zone of the fishery when over a fixed proportion juveniles are included in the weekly catch as judged by on-board observers. A rule which incorporates "redundancy" in this way, ensures that if the conventional control rule fails, there is a back-up based on an independent data series.

The potential application of this type of approach to the Mediterranean and Black Sea requires that capacity and license numbers be controlled, some measure of fishing mortality be estimated annually, and regular surveys of biomass be carried out. However, Figure 14 (lower) adds to this hypothetical F-Biomass rule a second management dimension, calling for an immediate reduction of fishing effort (e.g. a restricted number of days fished/week) if some other criterion is infringed. This second criterion under the management rule may apply to local grounds if observers on fishing vessels detect that rapid stock declines of critical life history stages are observed, where there are high incidental catches of protected species, or where damage to critical habitats is occurring.

Figure 14
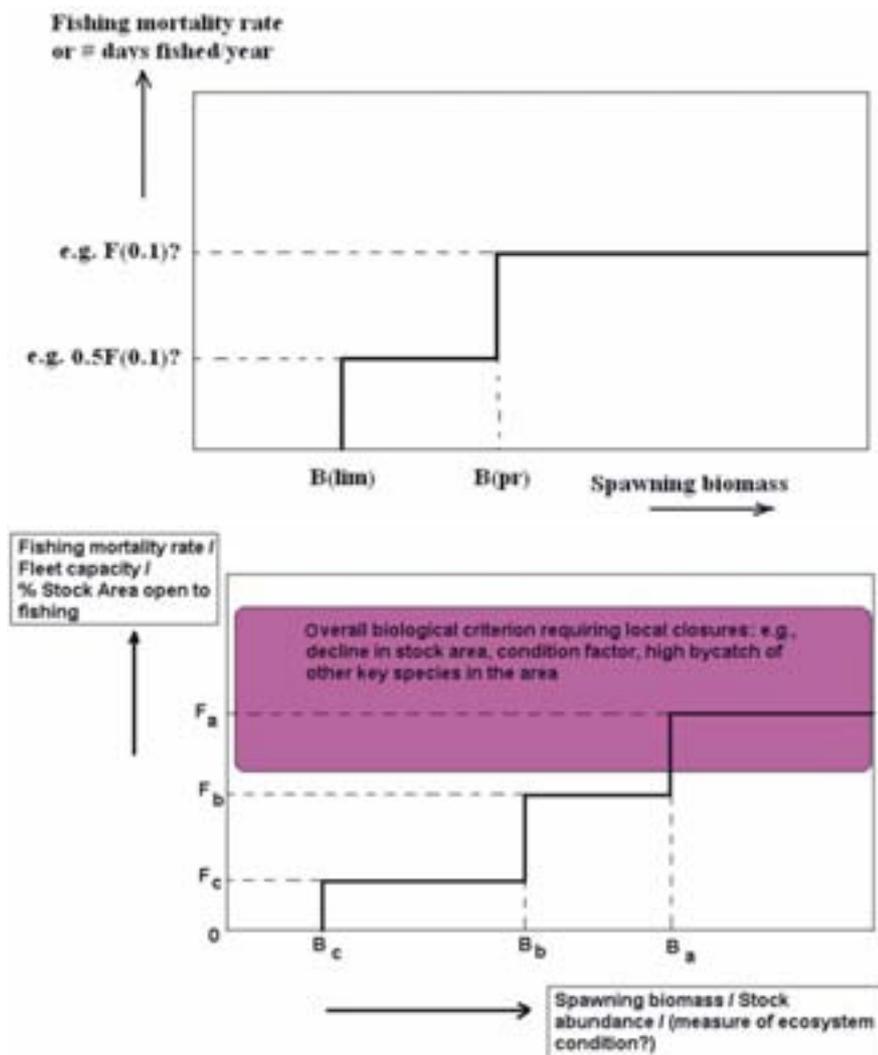
Alternative types of fisheries control rule are being considered by the Canadian "Fisheries Resource Conservation Council". The first considers how to improve the critical interface between "Science" and managers receiving research advice, where a critical lack of comprehension often occurs. Instead of suggesting a specific quota, in the approach shown in Figure 14, "Science" is requested annually to place each stock in a "box", based on an evaluation of its current productivity and stock size. For each of these boxes, the appropriate management action is pre-specified, thus discouraging discretionary action by managers under this type of management rule. A modification of the original approach which assumed quota control, is suggested in Figure 15 for fisheries where capacity and area fished are the two control variables used, as in the Mediterranean and Black Seas. The second approach is to design a rule which reacts progressively to restrict impacts on the stock as a function of the number of indicator series which are showing "red" (Caddy, 1999), or by the use of a more sophisticated rule using the same traffic light indicators, but based on fuzzy logic criteria (e.g. Halliday, Fanning and Mohn, 2001).

Figure 15

INDICATORS OF ENVIRON-MENT AND ECOSYSTEM ↓	STOCK CONDITION (based on annual surveys) →		
	HEALTHY STOCK CONDITION	BIOMASS BELOW Bpr	BIOMASS BELOW Blim
habitat/environ-mental conditions satisfactory	May increase capacity	maintain capacity constant	Close the fishery in this subarea
Evidence of deteriorating productivity	maintain capacity constant	Seek to reduce capacity or days fished	Close the fishery in this subarea
Habitat/ environ-mental conditions unsatisfactory	Seek to reduce capacity or days fished	Close the fishery in this subarea	Close the fishery in this subarea

Discussion

The need to determine reference points for a fishery is evident, but this paper notes that it is necessary to bear in mind that LRPs are primarily for use in fisheries control rules, and makes some suggestions as to the type of rules that might apply in the Mediterranean and Black Sea fisheries. Establishing indicators and reference points for Black Sea fisheries and ecosystems requires that the very dynamic changes of this system, and the multiple factors driving the ecosystem, be taken into account. Although landings data can provide some indications as to the dramatic events that have occurred over the last three decades, using fishing mortality estimates from retrospective analysis show that the increase in landings in the Black Sea after the early 1990s resulted from an increase in capacity as much as from stock recovery, though some modest recovery does seem to have occurred. In addition to overcapacity, the ecosystem

has responded to dramatic changes resulting from nutrient runoff, and the introduction of exotic species to the pelagic biome which have radically changed the ecosystem. An improvement in environmental conditions recently has been hypothesized; anecdotal information suggests that reductions in nutrient runoff may have been occurring, and a new species, *Beroe ovata*, which preys specifically on *Mnemiopsis*, has been introduced. Approaches to modelling following different hypotheses of causative factors underlying ecosystem change have inevitably led to differing conclusions as to the "prime" factor leading to ecosystem change. This makes a "model-based" approach to determining indicators and reference points controversial at this point. The approach that seems to be established by the Commission is to follow trends in productivity of those species that best represent the different biological communities and habitats of the Black Sea.

Plotting changes in a wide range of empirical indicators simultaneously after classifying their dynamic range using an empirical Traffic Light methodology, helps to formulate possible hypotheses, and illustrates that different factors have influenced the ecosystem through time. The key to immediate progress in upgrading the ecosystem will be agreement on fisheries management rules, but above all, their application! Several approaches to the use of indicators and reference points in such rules are suggested. Empirical reference points can be envisaged that seek to avoid those indicator values that applied during the dramatic ecosystem changes in 1989-92. When setting a realistic target for restoring this badly damaged ecosystem, it seems logical to aim as closely as possible, to restoring the baseline mesotrophic conditions that applied prior to 1989.

Acknowledgements

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Annex

Decisions made on indicators at the Workshop on Demersal Resources in the Black Sea and Azov Sea, of the Black Sea Commission; 15–17 April 2003, Sile, Istanbul, Turkey

At this technical meeting of scientists of member States of the Black Sea Commission, agreement was reached by national representatives on actions that need to be taken in developing indicators for selected commercial species and habitat/environmental indicators. Apart from reporting responsibilities on catches which apply to all harvested species, as an initial approach, it was decided that:

1) it would be better for the Commission to recommend a focus on collection of research indicators for a limited number of species and population characteristics, while allowing the possibility to revise the number of indicators subsequently if needed.

2) Some "keynote species" should be decided on as a focus for maximum effort of cooperative studies.

3) Country responsibilities for studies on keynote species are not confined to those countries currently taking the major catch – other countries should contribute data.

4) Suggesting a 'lead role' in studies for any country does not imply any special preference as to allocations, quotas etc. in any future management regime. The focus of research is simply a function of local availability of data or research resources for studies on the species and its research interest.

5) Some other activities (e.g. on environment, biodiversity, sturgeons) are being discussed in other components of the Black Sea Commission activities, and by CITES, hence coordination is implied by listing them in the following tables in which fisheries sector may not wish to take lead role.

REFERENCE POINTS: A BIO-ECONOMIC MODEL BASED APPROACH

by

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ABSTRACT

The concept of reference points (RPs) is closely connected to the management objective which, by also involving economic and social issues, may not be restricted to the sole biological maximization (maximum sustainable yield).

It is possible to define different RPs for each aspect of management or to identify a single RP aimed at ensuring the sustainability of the system, no longer limited to biology, but affecting environmental, economic and social features that involve the state of system as a whole. The latter is described as a multicriteria RP.

This paper briefly outlines the main RPs obtained by applying the bio-economic optimization models of effort and effort-catch, with specific reference to the Gordon-Schaefer model, which clearly derives from the theory of population dynamics.

The debate about the effectiveness and the efficiency of both the indicators and the RPs that are potentially useful in monospecific fishery is here conducted with respect to the multispecific character of Mediterranean fishery and the different selectivity of employed gears. Accordingly, as regards the Mediterranean context, we recommend the adoption of a multicriteria approach based on bio-economic optimization models which, by introducing a number of constraints, would allow the definition of the actual state of the system as well as specific management targets.

Keywords: *fishery management; fishing effort; mathematical models; multispecies fisheries; reference levels.*

Introduction

The UN Convention on the Law of the Sea of 1982 and the outcomes of the Rio de Janeiro Conference of 1992 allowed the adoption of the FAO *Code of Conduct for a Responsible Fishery* (FAO, 1995). The latter was a turning point in the States commitment to the preservation and the management of natural resources. In order to ensure the actual conservation and management of resources, this Code of Conduct establishes and recommends the adoption of international principles and models of behaviour focused on the respect for both natural ecosystems and biodiversity. Accordingly, the concept of sustainable development

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is described as *"a development that would meet the needs of the present without endangering the future generations' capacity to meet their own needs"* (WCED, 1987). This statement, which strongly evokes the principle of "sustainability", is worldwide regarded as the major guideline for policies concerning the management of marine resources. According to this principle, the Governments are to adopt an ecosystem-based approach. This involves the sustainable management of both commercial stocks and specific ecosystem factors that support the production such as the socio-economic quality of the sector. In other words, the ecosystem-based approach extends the management of the fishery from that of commercial stocks to that of the interaction between fishery, human and natural related systems.

In line with the above-mentioned principles, a sustainable development of the fishing sector requires that the actors involved in the preservation of the environment assume wider responsibilities. To this end, a clear and updated informative framework should support the socio-economic development oriented towards the preservation of the environment. The complexity, the uncertainty and the worldwide distribution of the issue call for the development of informative tools on which to base both a methodological organisation and an effective representation of reality.

A sustainable management policy is to be implemented by taking clear and specific steps, which can be summarized as follows:

a) to set the sub-goals to be achieved in order to ensure the exploitation of the marine resources belonging to the ecosystem;

b) to identify one or more indicators of sustainability for each goal;

c) to establish, for each indicator, one or more reference values through which a given situation may be considered as acceptable or undesirable; and

d) to define the measures to be implemented in order to achieve objectives determined on the basis of the specific conditions of the system, which is identified through the relevant indicators and reference values.

Once the objectives have been identified, the further step to be taken is the choice of the indicators. These are to play a considerable role as information and communication tools within the decision-making process. In a given moment, by adopting these indicators it is possible to draw an accurate picture of the sector in terms of exploitation of marine resources, environment conditions, and environmental and social impact. This information enables us to formulate hypothesis of intervention directed at improving the reference framework.

The position of the sustainability indicator associated with the reference values will describe the current state of the system and provide us with the relevant input to evaluate the situation and make management-oriented decisions.

In order to effectively interpret the information obtained through a specific indicator, some reference values are applied (RPs, reference points). These values can be associated with either a difficult or an optimal (or sub-optimal) situation. The former identifies a limit which is necessary to avoid (LRPs, limit reference points), while the latter represents a target to be attained by the system (TRPs, target reference points).

Reference points

The idea of reference points is strictly related to the management objectives to be attained. For instance, the maximum sustainable yield (MSY) is the reference value to be applied in order to reach the maximisation of production. However, management objectives usually include social and economic aspects, such as the profitability of fishing activities, the maintenance of employment levels and the safeguard of local economies strictly dependent on fishery. Thus, in order to ensure the environmental, economic and social sustainability, it is possible to define different reference points associated with each aspect of the sector or, to identify a single reference value that encapsulates all the aforesaid aspects. Furthermore, given that a single management aim would only meet the interest of a specific group of stakeholders, a management-oriented reference point should be agreed upon and shared by the stakeholders as a whole. As a consequence, the sharing of monitoring and control tools entails a multi-objective approach in which the different interests at stake are evenly represented.

As previously stated, reference points can be subdivided into two groups following their different use, such as: limit reference points (LRPs) and target reference points (TRPs) (Caddy and Mahon, 1995; Caddy, 1998). The LRPs consist in a definition of the thresholds that the resources or the sector indicators are not to overcome. Indeed, the overcoming of these thresholds would produce an irreversible long-term crisis affecting the resources and the sector as a whole. According to the reference indicator, LRPs can be associated to either minimum or maximum levels. The reaching of these limits should stimulate the implementation of comprehensive management measures directed at bringing the indicators back to levels considered as acceptable.

A quite different approach is required to define TRPs, which are usually related to indicators associated with the condition of stocks. These reference points represent the desirable level for some indicators of the exploitation status of stocks and, therefore, an objective to be pursued using appropriate management measures. TRPs may concern either the input or the output of the productive process. Accordingly, they might enable us to determine the optimal levels in terms of the size of catches, fishing effort or sector profitability.

The reference points within a single-species fishery

The most widely used management-oriented reference point (RP) is the maximum sustainable yield (MSY). As far as the fishing effort is kept constant until the system reaches an equilibrium, the MSY indicates the highest point of the curve that describes the functional relation between the annual fishing efforts made by the fleet and the resulting production. Apparently, this RP represents the natural aim of single-species fishery. Indeed, in the 60s and the 70s, it was widely applied by stakeholders. Nevertheless, further theories questioned the usefulness of the MSY with reference to the preservation of marine resources and suggested new approaches.

Like other RPs, the MSY is designed to meet a specific production model. Therefore, the reliability of the MSY depends on the consistency of this model with the reality. The most widespread models are the logistic or Schaefer model (Schaefer, 1957), the Fox model (1970) and the Pella and Tomlinson model (1969). All the parameters of these models are estimated by using the time series data of catches and effort: the relation between the two time series

allows setting the optimal value of catches at a desirable level of effort. Figure 1 shows the curve of sustainable profits obtained by using the Gordon-Schaefer model (Gordon, 1954). Obviously, the curve will change following the different models and the RP value.

The most common arguments against the adoption of the MSY as an RP concern the uncertainty of the estimation and the consequent risk of overcoming the optimal level of effort. The RP estimation depends on both the reliability of the data and the correct specification of the model, as well as on the degree of randomness that characterizes this phenomenon. Besides, the MSY estimation through time series data is conducted by assuming that the previous conditions will occur in the future with the same probability. Actually, any substantial change in the structure of the population along with the variations affecting the behaviour of the phenomenon, which may occur in variables not considered by the model, would turn the MSY and the effort estimation away from their true values. Furthermore, the adoption of the MSY as an RP will inevitably lead to a state of over- or under-exploitation, which would require repeated management interventions. Indeed, stocks react quite differently to the two previously mentioned situations that cannot be considered as symmetric. As evidenced by Beddington and May (1977), once the level of effort that corresponds to the MSY has been overcome, the stock fluctuations become wider and the time span needed to re-attain the equilibrium increases considerably.

The criticism about the adoption of the MSY as an RP led to a more cautious approach that takes into account the uncertainty of the system. A possible solution, maintained by Doubleday (1976), foresaw the setting of an optimal level of effort as equal to two-thirds (2/3) of the effort indicated by the MSY. (*2/3 Emsy*, see Figure 1). This effort level, which corresponds approximately to 89% of the MSY in terms of catches, by using the Schaefer model, entails a considerable decrease in the risks discussed above.

Figure 1: Gordon-Schaefer model. *Emsy* and *2/3Emsy* identification.
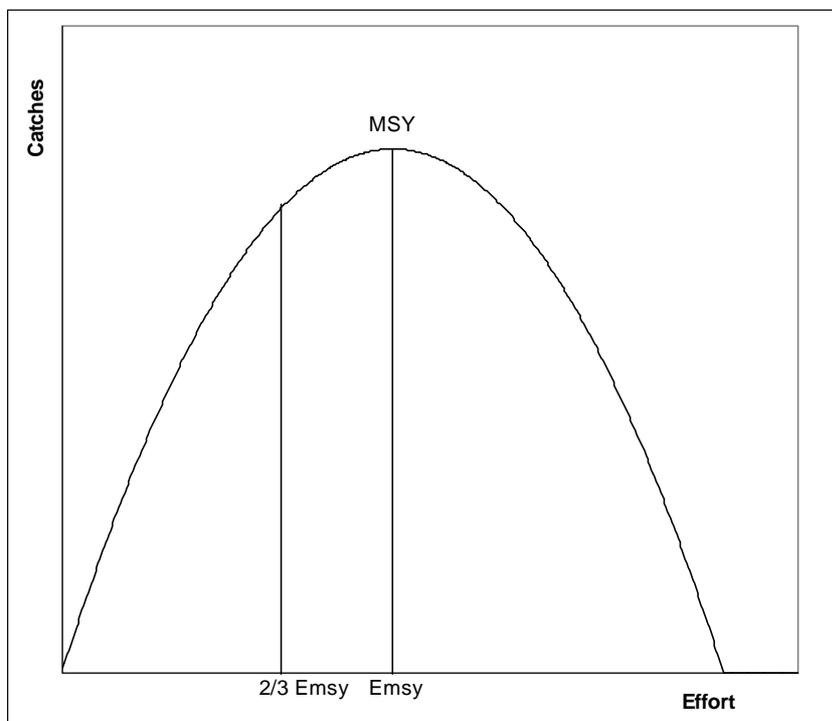
A different solution might be represented by an economic approach to the RPs designing. Several studies proposed the adoption of the Maximum Economic Yield (MEY) as an economic RP. This RP also originates from logistic models, as the Gordon-Schaefer equilibrium model. The MEY is equal to the level of effort in which the maximum profit is achieved. In other words, it corresponds to the highest difference between revenues and total costs. Given the cost function as linear, the MEY value is positioned before and below the MSY (that is, left of the MSY, see Figure 2). Since the effort that corresponds to the economic optimum is lower then that of the biological MSY, the adoption of the MEY as a RP will reduce the risk of overexploitation of resources. The level of effort associated with the MEY is likely to fluctuate as a consequence of the changes in the variables of the reference economic framework, such as the cost of fishing activities and the price of landings. When the price is a function of the catch quantity and, therefore, of the offer, low levels of catch may also correspond to higher profits. In these cases, the economic optimum will be positioned further left in the long-term equilibrium curve.

To sum up, the efficiency of the solution discussed above can be considered as equal or more effective than the Doubleday solution, which makes MEY more suitable (usable) than MSY.

Figure 2: Gordon-Schaefer model. *Emsy* and *Emey* identification.
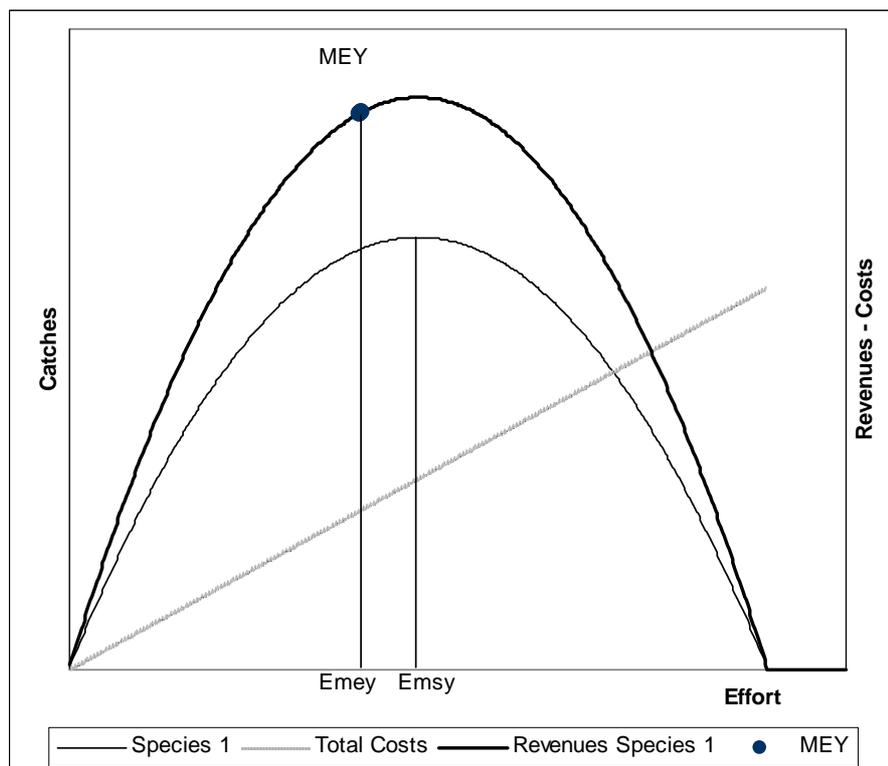
The problem of multispecies fishery

   The main hindrance to the adoption of the so far discussed RPs (MSY, MEY) is that they are based on models designed for single-species fisheries. In most cases, the fishing sector is characterized by a number of fishing systems which harvest a wide range of species. Particularly, within the Mediterranean, which hosts a number of animal and vegetal species and involves a fishery performed with low selective gears, the monospecific approach would prove to be completely ineffective. Given the large biological and technical interactions within the area, a management system based on single species RPs would be totally unfeasible. Indeed, the same fishing effort will be directed to harvesting different species, to which different MSYs and levels of optimal effort might be applied.

   Figure 3 provides a clear example of this situation. Within this system fishery targets three species, each species has its own MSY. If we consider the aggregate catches by summing up the curves of sustainable production for each single species we may obtain a MSY corresponding to the optimal effort *Emsy* through which we may determine the impoverishment of the less productive stocks. In the case under discussion, species no. 3 is doomed to become extinct if we adopt the point equal to *Emsy* as the level of long-term effort. Given these circumstances, wiser approaches, as the adoption of the two thirds (2/3) of the MSY effort as the optimal effort level, may no longer prove to be helpful. As regards stocks preservation, from a precautionary viewpoint, the only feasible solution would be to define an RP which takes into consideration the species most vulnerable by fishery (in Figure 3, species no. 3). Assuming that this choice might prevent marine resources from being overfished, nevertheless its huge socio-economic costs would make it totally unfeasible.

Figure 3: Gordon-Schaefer model in a multispecies fishery. *Emsy* identification.
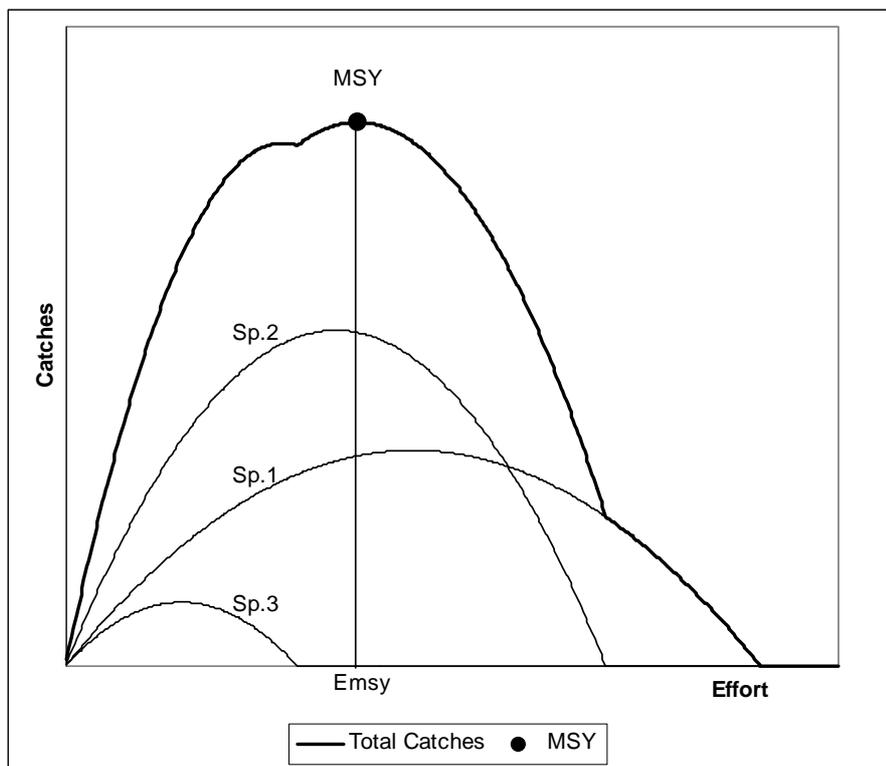
Within a multispecies context, the single-species economic approach (MEY) also follows different and much more complex guidelines. Compared to the MSY level, the level of effort corresponding to the MEY can no longer be considered as a prudential value. Indeed, since it depends on the ratios between the prices of the different species, it is likely to be positioned either on the left or on the right of the *Emsy* value. The latter case is expected to occur when consumers' choices determine a higher price of the most productive species. If species with higher *Emsy* also have a higher market value, the *Emey* value will be positioned to the right of the *Emsy* value. Figure 4 illustrates this case: compared to species no. 2 and 3, species no. 1 shows a significantly higher price. This determines a MEY effort (*Emey*) higher than the effort related to the MSY (*Emsy*). Conversely, when a higher price is associated with species whose intrinsic growth rate is lower, the MEY will be found on the left of the MSY. However, in this case the equivalent level of effort will not ensure the prevention of species belonging to the productive mix from being overfished**.**

In a multispecies fishery, such as the Mediterranean, the adoption of an RP derived exclusively from a biological or an economic model would be only partially applicable. In other words, it would not represent the optimal solution. On the contrary, we suggest the adoption of a multi-objective approach directed at maximizing the economic outcome that also takes into consideration the need for stock preservation. This scheme is based on the use of bio-economic models that summarize the long-term behaviour of the system in a context of optimal sustainability. Tailored to multispecies fishery, the approach based on the Gordon-Schaefer model will be discussed below as an example of a viable method. Nevertheless, it is also possible to adopt more complex and detailed models, which would take into account the biological features of the different species**.**

The problem of multigear fishery

The use of a surplus production model is based on the relation between catch and effort data defined for each species. In the Mediterranean, most of the species are exploited with different fishing gears, consequently it is very difficult to define the total amount of effort exercised on a single stock. It is not correct to simply sum the effort of each fleet segment since a unit of effort generally has a different productivity depending on the gear used. A possible solution is to define an equivalent effort independent from the different gears used to catch the species (Placenti *et al.*,1992).

Thus, for each species *i* in a specific area, the equivalent effort $E_{eq,i}$, is defined as the weighted sum of the effort E_{ij} of the N fishing systems competing for the same catch:

$$E_{eq,i} = \Sigma_{j=1,N} \; \alpha_{ij} \; E_{ij}$$

Generally, the effort E_{ij} for each fishing system *j* for each species *i* is unknown. It can be expressed as a fraction β_{ij} of the total effort (indicated by the subscript *t*) for the *j-th* fishing system in the area:

$$E_{ij} = \beta_{ij} \; E_{tj}$$

Therefore:

$$E_{eq,i} = \Sigma_{j=1,N} \; \alpha_{ij} \; E_{ij} = \Sigma_{j=1,N} \; \alpha_{ij} \; \beta_{ij} \; E_{tj} = \Sigma_{j=1,N} \; \chi_{ij} \; E_{tj}$$

where the χ_{ij} represent the product of the coefficients α_{ij} and β_{ij}. If terms χ_{ij} are assumed to be time independent, they can be estimated via non-linear regression based on the catch and effort data.

Assuming that:

$$\Sigma_{j=1,N} \chi_{ij} = 1$$

only N-1 independent coefficients are needed to define the equivalent effort for the N fishing systems competing on the same catch.

A multi-objective approach

Within a single fishing area, this model aims at determining the optimal level of fishing effort per each segment of the fleet. Therefore, the effort obtained using this method is to ensure the attainment of the maximum economic outcome (in terms of added value) consistent with the need for preservation of the species (biological restriction). Thus, the problem can be represented as follows:

$$\max_x VA(x) = R - C$$
$$\text{s. v. } VB(x) < VB_{max}$$

where R stands for the revenues obtained by summing up the values (per single species) of the catches (each species multiplied by its selling price). In addition, C describes the aggregate operative costs obtained per each segment of the fleet by multiplying the unitary cost of the fishing activity (not including labour cost) by the overall fishing effort. The average and the unitary prices of the fishing activity can be obtained from the time series data. Since 1985, a monitoring of the Italian fleet conducted by IREPA with a methodology of statistic sampling enables us to estimate the above-mentioned data. The vector x represents the variables to be optimized; that is, the level of effort pertaining to each segment of the fleet operating in the relevant area. The prices and the costs of the year of consideration, together with the time series of catch and effort are then used to establish the optimal levels of effort.

The maximization of the added value concerns segments of the fleet (identified on the basis of the fishing system used) rather than single productive units (vessels). Therefore, we indirectly assume that all fishing units have a similar behaviour and that their actions are cumulative. Thus, providing that the above-stated conditions are satisfied, it is possible to include these micro-economic features in the optimization algorithm. The added value obtained will then represent the returns of the productive sector, that is, the capital and the work, which are difficult to be distinguished within the Italian fishery (Placenti *et al.*, 1992).

As previously stated, the level of effort directed at achieving the Maximum Economic Yield (MEY) cannot be used as a RP within multi-species fishery since it does not ensure the preservation of the species as a whole. In order to avoid *overfishing* and consequently the significant impoverishment of the most vulnerable species, a biologic restriction (*VB*) has been included in the methodology. For each equation effort-catch, that is, for each species, a biological term VB_i has been defined. This stands for the excess of effort compared with what

we consider as an economically optimal effort. For species that are not overexploited, this is equal to zero:

$$VB_i = E_i - E_{m,i} \qquad se\ E_i > E_{m,i}$$
$$VB_i = 0 \qquad se\ E_i \le E_{m,i}$$

where $E_{m,i}$ represents the effort that, in a state of biological equilibrium (MSY) of the i-nth species, corresponds to the maximum catch. This value depends on the effort-catch model adopted. As for the Schaefer model, where the catches pertaining to a single species are a function of the fishing effort according to the parameter k_0 and k_1, we consider that:

$$C_t = k_0 E_t - k_1 E_t^2.$$

This value is obtained by

$$E_{m,i} = k_0/2k_1.$$

For each area, the biologic term VB is calculated as the ratio between the sum of the terms VB_i for each species and the total effort within the relevant area:

$$VB = \sum VB_i / \sum E_i.$$

This index can be read as the fraction of the aggregate effort that exceeds the condition of maximum sustainable catch (MSY) for each species. Figure 4 shows the excess of effort represented as segments AB and CD, which measure the excess of species no. 2 and no. 3 respectively. The sum of the two segments corresponds to the biological term VB. If a maximum limit VB_{max} is assigned to VB, the level of effort to be adopted as an RP will be positioned on the left of the MEY effort (*Emey*).

Figure 4: Gordon-Schaefer model in a multispecies fishery. *Emsy* and *Emey* identification.
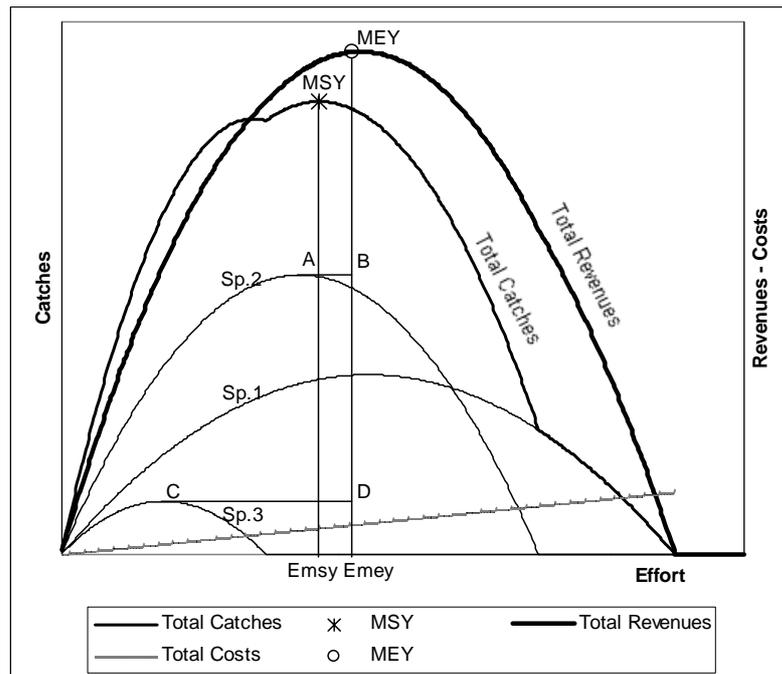
The bio-economic model of optimization is directed at determining the economic optimum that would allow a level of exploitation of the most vulnerable species not exceeding the pre-established VB_{max} value. The higher the value assigned to VB_{max}, the closer to the MEY is the optimal level of effort to be considered as a TRP. On the other hand, the lower this value is, the closer we may get to a TRP coinciding with the MSY of the species which are most likely to be negatively affected by fishing activities. The simulations conducted through this model allow assessing the economic and social consequences of the several values attributable to VB_{max}, and the different influence assigned to the environmental factor.

The model discussed above takes into account only environmental and economic issues. Nevertheless, other factors can also be considered. For instance, if the model is applied to a fishing activity performed in a situation of overexploitation, this will require a TRP lower than the current value, thus urging the operators to immediately reduce the effort. This decrease may concern any segment of the fleet operating within the relevant area. It is clear that a decrease in the capacity, i.e. a reduction of the number of vessels, may produce social damages which would vary according to the fleet segment examined. Thus, it is possible to introduce a further restriction directed at safeguarding the segments of the fleet that either ensure higher levels of employment or have lower impact on the ecosystem. A further issue is related to the importance of fishing activity within local economies strictly dependent on fishery or with high unemployment rates. In these cases, the approach to be adopted should take into account economic and social issues rather than the mere environmental ones.

The structure of the restrictions can be further modified and tailored to meet the specific requirements of the management tools, which we intend to adopt in conjunction with the prearranged objectives. Given that the re-arrangement of the fishing effort among different areas and segments of the fleet is the main management objective, once the optimal level per each area has been established, the best allocation of the exceeding effort can also be determined (Placenti *et al.*, 1995). In this case, to support the feasibility of the management measures and the relevant RPs adopted, the occurrence of further management costs should also be taken into account. To this end, it is possible to measure the differences between the proposed solutions and those that have actually been applied. Thus, an inertial term VI_i has been defined for each fishing system and area:

$$VI_i = \left| E_i - E_i^* \right| - \Delta E_i \qquad \text{se} \left| E_i - E_i^* \right| > \Delta E_i$$
$$VI_i = 0 \qquad \text{se} \left| E_i - E_i^* \right| \leq \Delta E_i$$

where ΔE_i stands for the maximum variation permitted around the reference value for the effort E_i^* obtained using the following formula: $\Delta E_i = \text{StDev}(E_i)F$. The ΔE_i value is then proportional to the standard deviation of the i-nth effort over the period of time considered within a given area and segment of the fleet. The F factor may vary according to the different scenarios and the management objectives considered by the optimization analysis. Within each area and fishing system, this procedure allows taking into account the *elasticity* shown by each fleet over the previous period.

For each area, a comprehensive term VI is then calculated as follows:

$$VI = \sum VI_i.$$

The inertia term allows limiting the variations of the current distribution of the effort while avoiding the implementation unfeasible solutions. Also it helps us to select the optimal distribution that would require lower effort and re-conversion costs while ensuring equivalent biological and economic outcomes. Given the application of this further restriction, the RPs will be identified by solving the following constrained optimization problem:

$$\max_x VA(x) = R - C$$
$$\text{s. v. } VB(x) < VB_{max}$$
$$VI(x) \leq 0$$

In this case, the model becomes more complex and permits to identify the TRPs that could best meet environmental, economic, social and management needs by assigning the appropriate values to VB_{max} and F.

Conclusions

Within the fishery sector, the multi-objective approach based on bio-economic models permits to take into account the various interests supported by stakeholders. Their objectives do not necessarily diverge. In fact, when considering long-term solutions, interests may coincide. However, management measures may also be implemented so as to produce the least inequality among the different groups of stakeholders. Furthermore, the objectives and the relevant variables included in the model may evidence specific synergies and relationships. The knowledge of the system and its rules allows finding solutions that would meet the interests of certain groups while causing the least possible disadvantages to those of others. This knowledge is represented by the equations included in the bio-economic models, which, by taking into account all the interests at stake, enable us to find the optimal solutions. Obtained through the adoption of constrained optimization techniques, these solutions require an object-oriented function. In the case examined in the present paper, the latter is represented by the maximum economic profit. Additional restrictions direct the search for the optimum towards a set of solutions considered as desirable. These are aimed at safeguarding the marine resources and avoiding their impoverishment. Also, they should preserve employment levels, support local economies and, above all, rather than exclusively consider the welfare of fisheries, take into account the interests of the society at large.

As a consequence, the multi-objective function enables us to shift from RPs merely based on population dynamics to TRPs that represent the operators' consent to take full responsibility for the social aspects involved in their activities. Accordingly, in a multi-objective perspective, bio-economic models allow searching for the optimal levels of effort in which the importance of each aspect determines the weight of the resulting restriction within the optimization model. All stakeholders shall then contribute to define these weights. Thus, by simulating the outcomes of those choices on the system as a whole, the model may prove to be extremely useful in this phase.

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ON THE SUITABILITY OF SOME INDICATORS FROM TRAWL SURVEYS DATA. MEDITERRANEAN GEOGRAPHICAL SUB-AREA No. 18

by

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ABSTRACT

The suitability of some indicators from bottom trawl surveys carried out in the southern Adriatic Sea (GFCM – Mediterranean geographical sub-area 18) was investigated using data available from the Medits programme (years 1996–2003). The trajectories of some population indicators such us abundance and density indices were analysed for two main target fishery species, the European hake and the deepwater rose shrimp, utilizing statistical estimators as arithmetic and geometric mean, median, and 75th percentile. Temporal variations of mean length values have been also considered. Furthermore, the indicators concerned and the BOI index (bottom-dwelling fish and overall fish biomass ratio) were applied to the pool of Medits target species. The results highlight the potential suitability of some of the indicators considered and of their estimators, at least for the area investigated and the time period.

Keywords: *Bottom trawls, demersal fisheries, indicators, fishery management, Mediterranean Sea*.

Introduction

The concept of sustainable development was formally acknowledged as an internationally relevant issue by the World Commission on Environment and Development in 1987 (WCED, 1987) and subsequently ranked as a priority at the United Nations Conference on Environment and Development (UNCED) in 1992. The present state of every system is usually measured by progress indicators (Eldredge, 2002). The use of indicators of sustainable development is recommended in Chapter 40 of the Agenda 21 adopted at the UNCED and subsequently reaffirmed at the World Summit on Sustainable Development in 2002.

The identification and use of suitable indicators and their reference points for sustainable fisheries development and management is nowadays a central issue worldwide. Work is in

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progress to define relevant indicators and how they can be best applied (Des Clers and Nauen, 2002; Halliday, Fanning and Mohn, 2001; Laë, Ecoutin and Kantoussan, 2004; Laloë, 2004; Le Gallic, 2002; OECD, 2000).

In this work the suitability of some indicators from bottom trawl surveys is considered using data available from the EC Mediterranean International Trawl Survey (Medits) programme carried out in the southern Adriatic Sea (GFCM Geographical sub-area, GSA, 18) from 1996 to 2003.

The aim was to identify and test a first set of simple indicators and their respective statistical estimators based on fishery-independent data, and in coherence with some of the following well established criteria and desirable properties (FAO Fishery Resources Division, 1999; Hall and Mainprize, 2004; MOFI/ALMVRV/SEAFDEC/FAO, 2001; Segnestam, 2002):

- scientific validity in the sense they should be indicative of the objective they intend to reflect;

- easy compilation and processing procedures;

- reliable performance with respect to interactions between fishery, environment and resources;

- applicability to different scenarios and capability to show response to management measures;

- feasibility and cost-effectiveness in terms of data collection requirement;

- comprehensibility and acceptability for all stakeholders;

- easy integration and comparison to each other and with indicators from other sources (e.g. indirect methods, fishery-dependant information, economic and environmental indicators, etc.).

The scientific background and some proposed indicators presented are mostly based on the work of Bellail *et al.* (2003), Caddy (1999, 2002), Fiorentino *et al.* (2003), Hall and Mainprize (2004), Hutchings (1996), Koeller *et al.* (2002).

Material and methods

Raw data were obtained from experimental trawl hauls carried out in the GFCM GSA 18 during the period 1996-2003 (Figure 1). The trawl surveys were carried out during the spring-summer season in the framework of the Medits Programme, according to a specified protocol (Bertrand *et al.*, 2002).

The catch data were standardized by using the swept-area method (Sparre and Venema, 1998) in order to obtain biomass (kg km^{-2}) and abundance (n° km^{-2}) indices.

Selected indicators were applied both to single species populations and to pooled demersal fishery species by using different statistical estimators. The European hake, *Merluccius merluccius* (L.), a relatively long-lived species, and the deepwater rose shrimp, *Parapenaeus*

longirostris (Lucas), a short-lived species, were selected for the first case, all the fish species and the Medits target species ("short list" as reported by Bertrand *et al.*, 2002) for the second.

The indicators were tested as follows:

- Arithmetic mean of catch indices (abundance and density) of European hake, Deepwater rose shrimp and the pool of Medits target species from the short list, weighted by stratum area in order to reduce the variance (Cochran, 1977 in Souplet, 1995).

- Geometric mean of catch indices (abundance and density), weighted by stratum area (Cochran, 1977 in Souplet, 1995). The logarithmic transformation (Log X+1) of original data was applied to the European hake, the Deepwater rose shrimp and the pool of Medits target species (short list).

- Median value and 75th percentile of catch indices (abundance and density) of European hake, Deepwater rose shrimp and the pool of Medits target species (short list).

- Mean length of *M. merluccius* (Total Length, TL) and *P. longirostris* (Carapace Length, CL) from the survey catches.

- BOI, bottom-dwelling fish/overall fish biomass ratio index (mean value) (Fiorentino *et al.*, 2003). The bottom-dwelling species were chosen according to some morphological characteristics (Fiorentino *et al.*, 2003). All the fish species collected were included in the computation, excluding pelagic species belonging to the families Clupeidae, Carangidae, Engraulididae and Scombridae.

Moreover, the temporal pattern of spatial distribution (mapping by GIS; exponential model kriging) of the European hake and Deepwater rose shrimp was obtained to have additional information available for the discussion of results.

Results

The results related to both the single species, *M. merluccius* and *P. longirostris* and the pool of Medits species highlighted the following:

The chosen indicators showed a decreasing trend of most of the relative biomass and abundance estimators in the case of the European hake. The decrease proved to be more marked by the trend of biomass indices with respect to the abundance ones. The best fit of the linear trend was from the use of the 75th percentile estimator applied to the abundance index (Figures 2 and 3). No trend was evident for the population mean length indicator (Figure 3).

The time series of the indicators for the Deepwater rose shrimp showed a positive trend for all the chosen estimators. The increase seems to be better indicated by the trend of biomass indices with respect to the abundance ones, as was reported for the European hake. In the case of *P. longirostris* the best fit of the linear trend also resulted from the use of the 75th percentile estimator of the abundance index (Figures 2 and 3). As for the hake, no trend could be estimated for the population mean length indicator (Figure 3).

The relative biomass index of the pool of Medits species indicated a stable pattern over the time period considered (Figure 4).

The BOI mean index for the whole of GSA 18 seemed to increase over the investigated time period; nevertheless, the increase was related mostly to the shelf bottom area while a fluctuating trend was observed for slope data (Figure 5). With regard to the species' spatial distribution pattern the yearly maps of biomass and abundance of *M. merluccius* and *P. longirostris* would confirm the prevailing trends of indicators for both the species. Particularly the European hake appeared both to be undergoing an overall decrease and a reduction in its area of occurrence (an increased in "blank" zones) (Figures 6 and 7). Moreover, the distribution of individual mean weight at six-year intervals showed a decrease in the higher values (the number of larger specimens in the trawl catches declined) (Figure 8). Unlike the European hake, the yearly maps of the Deepwater rose shrimp indicated an increase in this resource in GSA 18 and particularly the evident expansion over the whole basin during the last reported years (Figures 9 and 10).

Discussion

The indicators considered are among the simplest and most readily available biological indicators which may be included in a fishery management framework, provided that their performance and suitability are known. A series of stock abundance and density indices were analysed using arithmetic and geometric mean, median and 75th percentile as estimators.

The arithmetic mean estimator showed an appreciable trend only in the case of European hake abundance. However, it is widely reported (Conquest *et al.*, 1996; Hutchings, 1996; McConnaughey and Conquest, 1992; Pennington, 1996) that the arithmetic mean values can be strongly influenced by skewed distributions (i.e. few tows with large catch). The precision estimators (i.e. variation coefficient) seem to be mostly influenced by sampling density, masking the supposed increase of variance expected as a consequence of increasingly higher exploitation rates (Blanchard and Boucher, 2001).

The geometric mean of abundance and density indices highlighted significant linear trends for both *M. merluccius* and *P. longirostris*. In the study cases the log transformation was useful to normalise the skewed distributions.

The median value of abundance and density indices indicated significant linear trends for both the analysed populations as was observed for the geometric mean estimator. It could be particularly meaningful for those species with large occurrence in the samples (Ungaro, unpublished).

In the case of skewed distributions, a useful summary measure of the observed variable can be obtained using percentiles (Costanza, Galobardes and Moraiba, 2002). The 75th percentile estimator highlighted significant linear trends for both *M. merluccius* and *P. longirostris* populations. Most of R^2 values obtained from the use of the 75th percentile resulted higher than geometric mean and median ones. Moreover, this estimator also accounts for the fraction of the medium-high catches for the species, which is very important for the sustainability of fishery exploitation (Hutchings, 1996). The 75th percentile can constitute a useful estimator for the relative biomass indicator of individual species.

The population mean length value does not highlight any appreciable trend. It is probably strongly influenced by fishing gear selectivity and recruitment strength (Bellail *et al.*, 2003).

The BOI index does not seem to perform well in the investigated scenario where semi-industrial fishery has been a well-developed activity for some time. The index can probably be applied to longer time series, or for the comparison between areas characterised by marked differences in fishery exploitation levels.

The auxiliary information on spatial distribution could be a powerful tool to integrate the discussion on the obtained results. In fact, it has been hypothesised that temporal modification of spatial distribution patterns can be considered as an index of resource condition for groundfish (Hutchings, 1996) and small pelagic species (MacCall, 1990). Moreover, if the bio-ecology of the species is known, the spatio-temporal distribution of individual mean weight values could provide additional information on the consistency of the spawner and juvenile fraction.

In general, the numerical abundance index (n/km^2) performs poorly; as Mediterranean bottom trawl fisheries mostly exploit juveniles, the recruitment strength and intensity probably limit the significance of this indicator. The same would apply when using the mean length as an indicator. Nevertheless, the abundance indicator could be taken into consideration as secondary information.

Some of the evident trends observed raise questions on the possible cause. The decreasing trend of index of relative biomass of the European hake appears mostly related to the adult fraction of the population. Some possible causes of such a decrease could be linked to the fishing mortality exerted on large individuals by bottom long-liners (De Zio *et al.*, 1998) and/or the increase of demersal fishing effort in the eastern Adriatic sector (Mannini, Massa and Milone, 2004). In any case, deeper investigation is needed to explain the results obtained.

The increasing trend of biomass indices of Deepwater rose shrimp is observed together with the expansion of the range of its geographical occurrence in GSA 18, as indicated by the GIS representation. Some possible causes of such an increase could be linked to the effects of environmental conditions (i.e. increase of bottom temperature; Ungaro and Gramolini, unpublished) and/or to the variation of inter-specific ratios (e.g. predator-prey relationships).

The supposed stability of the species pool considered could be linked to inter-specific compensation (single species variation higher than total species variability), as well as to the influence (variation) of the main environmental parameters and the modifications of predator-prey relationships.

Conclusions

World fisheries are believed in many cases to be approaching their sustainable limits (Buckworth, 1998; FAO 2002, Grainger and Garcia, 1996) and the need to rethink fisheries management is widely perceived (Pitcher, Hart and Pauly, 1998). Fisheries management is in a sort of deadlock because of the perception that fisheries models alone (model-driven approach) are not a sufficient tool through which management advice can be formulated. An increasing awareness has arisen that multidisciplinary-based appraisal and monitoring of fisheries should be further developed and implemented to address fishery management issues.

Indicators for fishery management and sustainable development are increasingly being defined and developed in world fisheries, however their practical application is still limited (Garcia, 2004). The necessity to select and test indicators (and their associated reference values) is becoming a priority for many fisheries.

In subregional contexts where shared stocks occur, as it is the case for the Adriatic Sea (GFCM, 2001), the use of internationally concurred indicators assumes critical importance to support cooperative management by the countries concerned. The use of fishery-independent information is maximized if scientific survey protocols (same sampling design, haul density, period, etc.) are standardized thus allowing for more a reliable comparison and integration of the indicators used (Bellail *et al.*, 2003).

The reported results show the potential suitability of some of the indicators considered, and their estimators, at least for the area investigated and the time period. Further appraisal of the performance of these indicators should be carried out applying them to a larger set of species. The most suitable indicators, identified on the basis of the results attained, and their metrics (i.e. geometric mean, median and 75th percentile) should be considered for inclusion in a multidisciplinary poly-indicator suite upon which agreed management procedures could be based.

The indicators tested in this work are among those, such as relative abundance of target species, commonly considered for the ecological dimension of fisheries sustainable development (FAO, 2000). Moreover, they can meet most of the basic properties required and their time series, although relatively short, are available for several areas of the Mediterranean Sea. Among the indicators and their metrics tested, the geometric mean, median and 75th percentile estimators of biomass (abundance) index performed better, and considering that indicators of relative biomass are of priority relevance, as they are believed to be a meaningful proxy of stock size at sea, they should be regularly utilized in fishery management frameworks, also considering their simplicity and comprehensibility.

Figures

Figure 1: Area Investigated, Mediterranean GSA 18 (south Adriatic Sea)
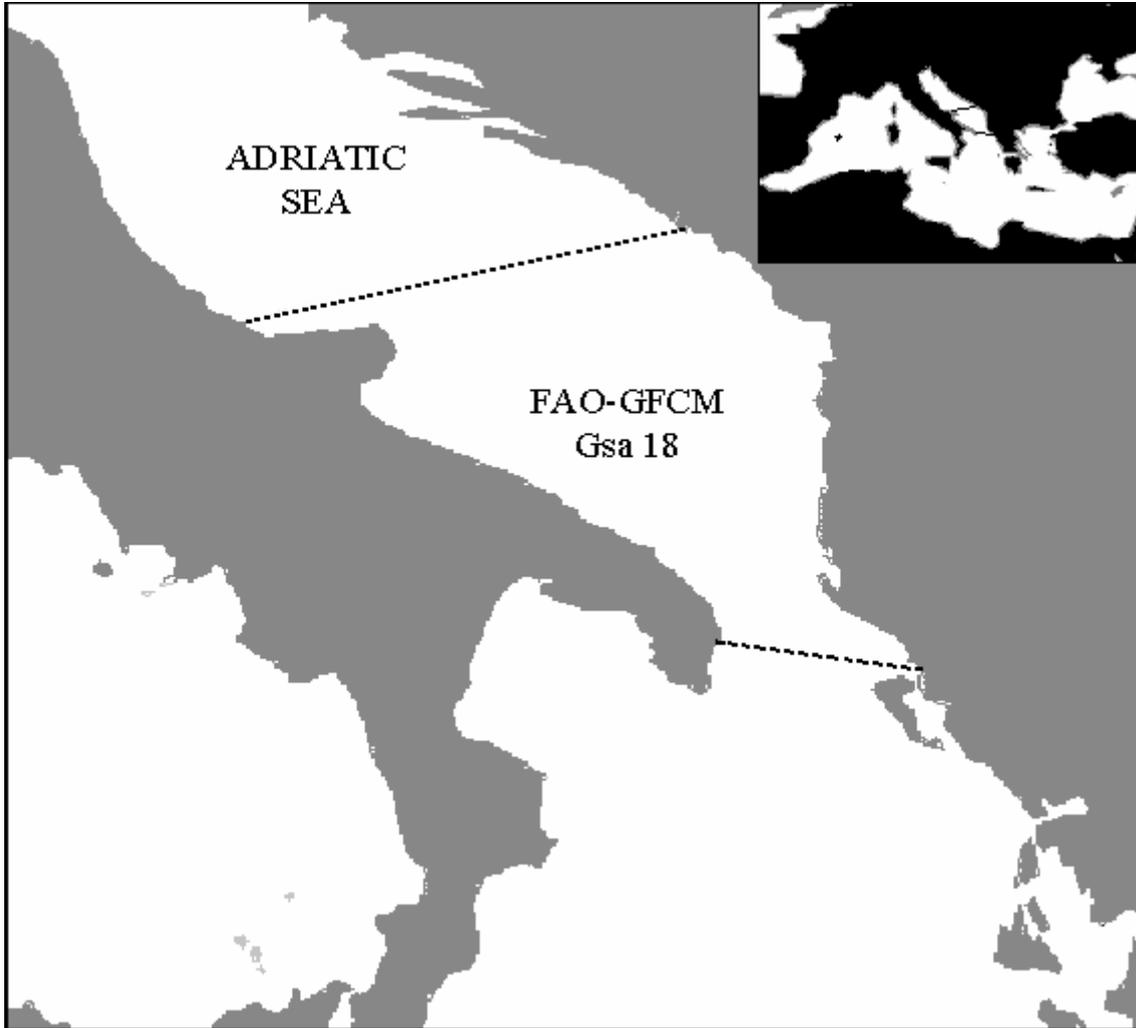
Figure 2: Series of selected indicators (biomass and abundance index) and estimators (arithmetic mean and geometric weighted mean) for the European hake (left) and the deepwater rose shrimp (right). The bars indicate the +/- standard deviation.
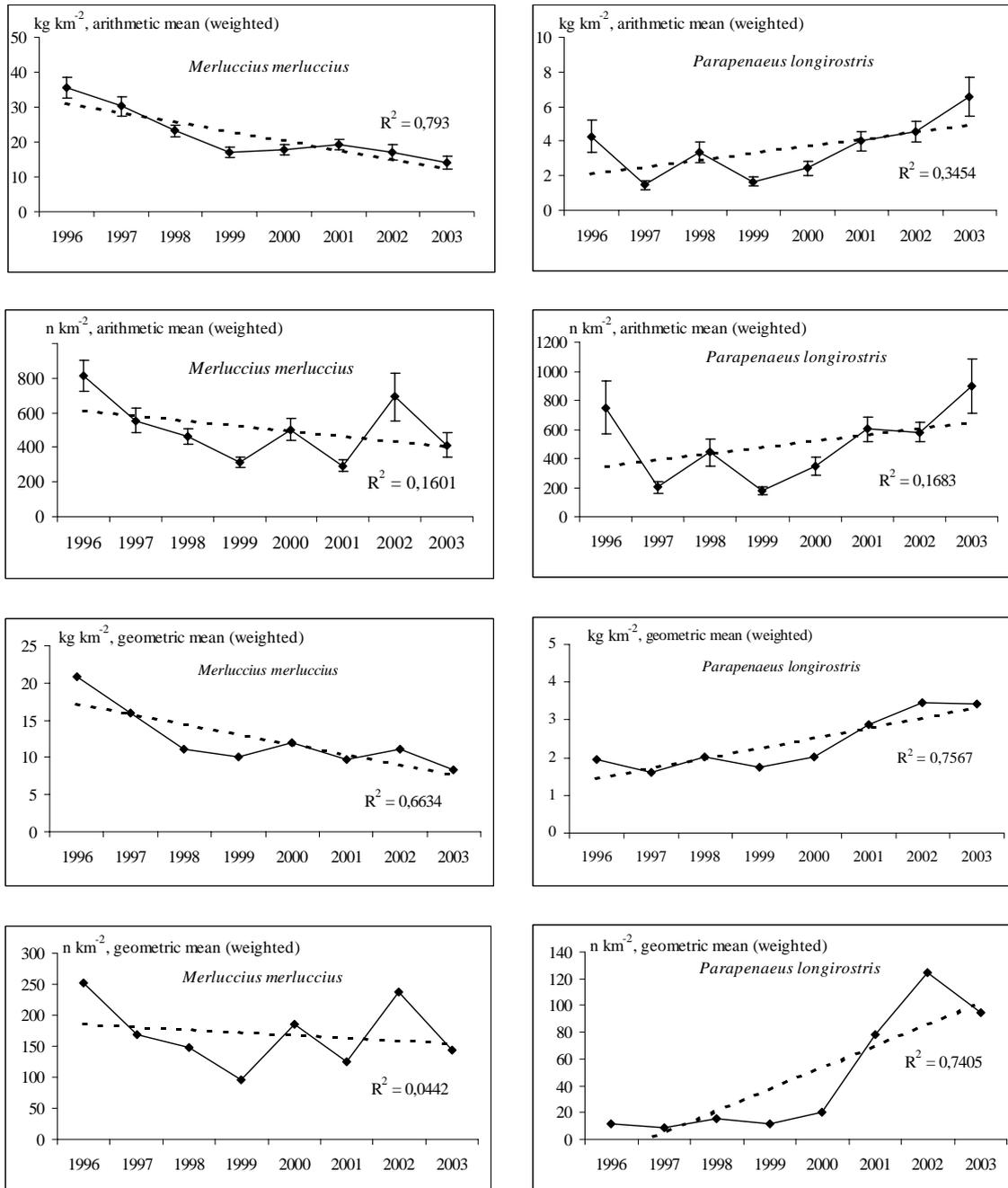
Figure 3: Series of selected indicators (biomass, abundance index and length size) and estimators (median, 75th percentile, mean individual length) for the European hake (left) and the deepwater rose shrimp (right). The bars indicate the +/- standard deviation.
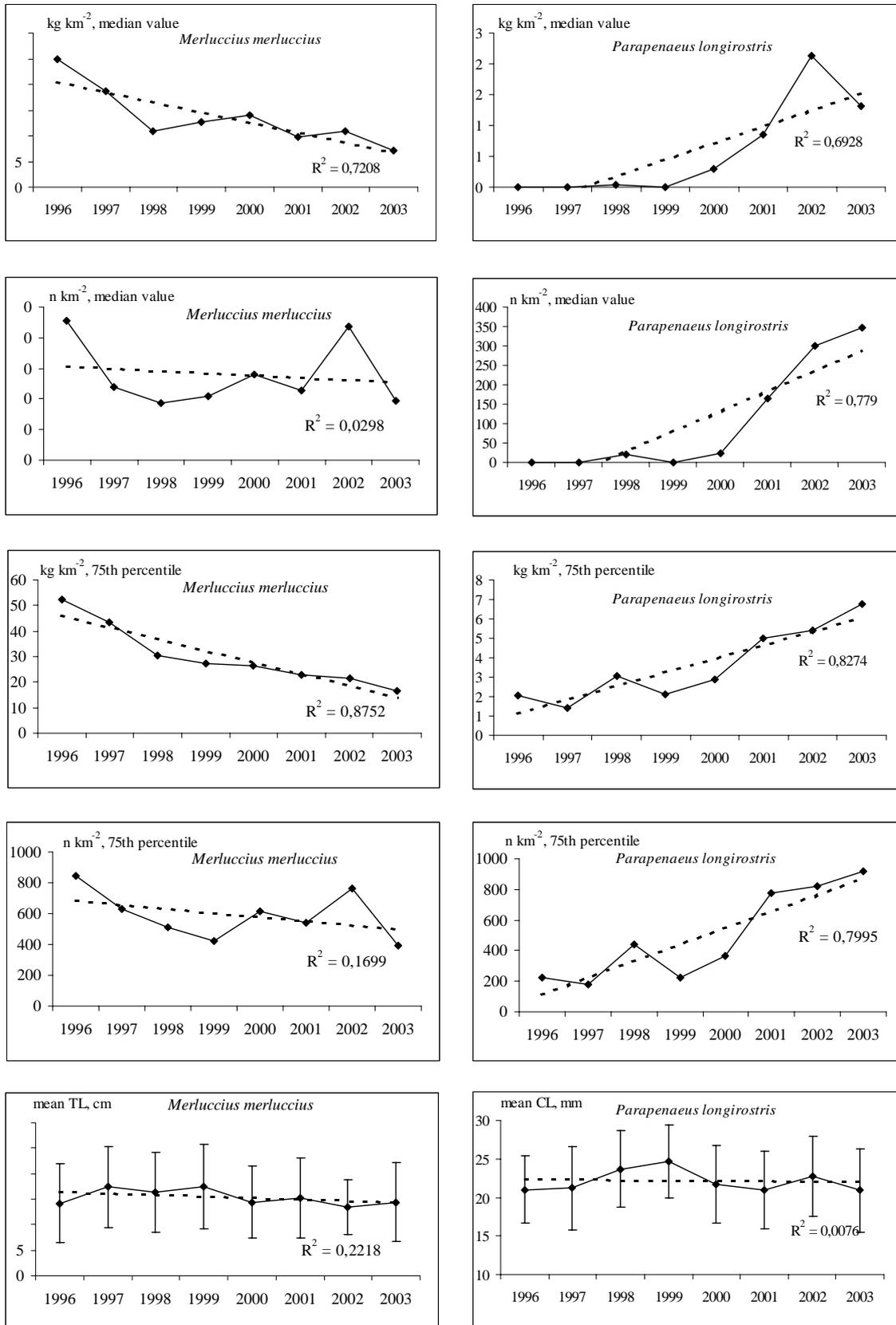
Figure 4: Series of indicators (biomass and abundance index) and estimators (arithmetic mean, geometric weighted mean, median, and 75th percentile) for the pool of Medits species.
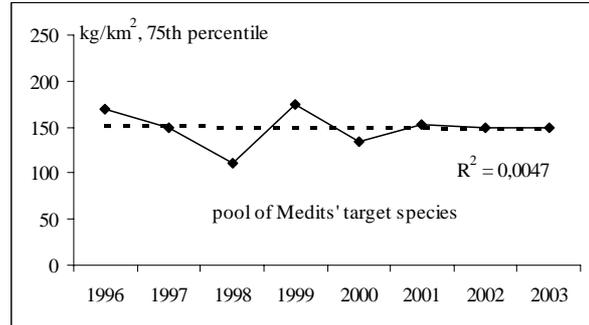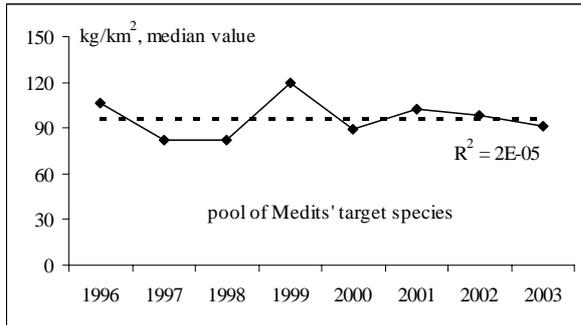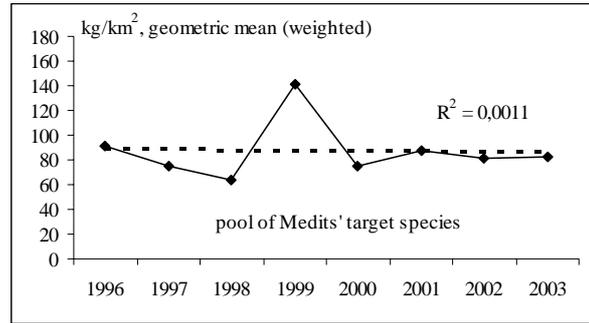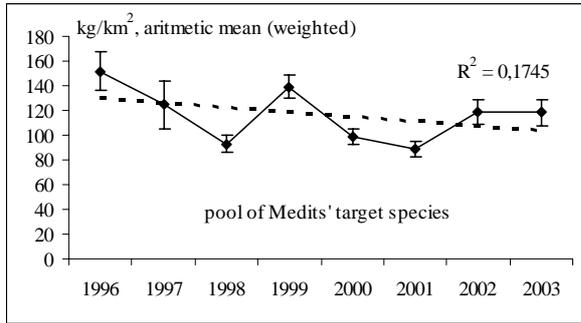
Figure 5: Mean BOI index (1996-2003).

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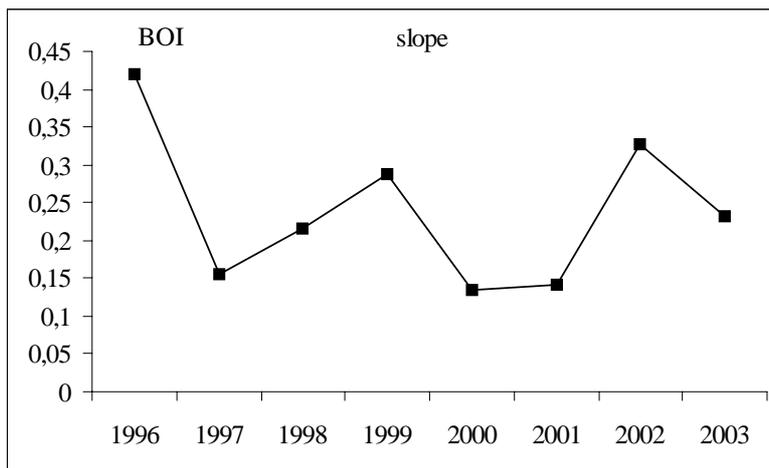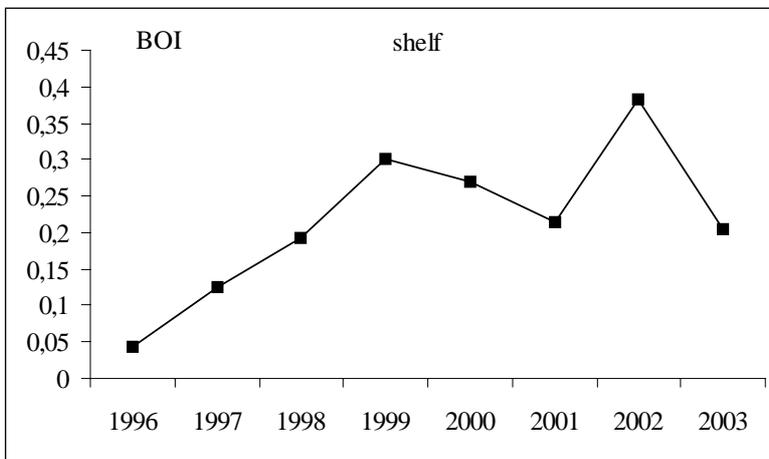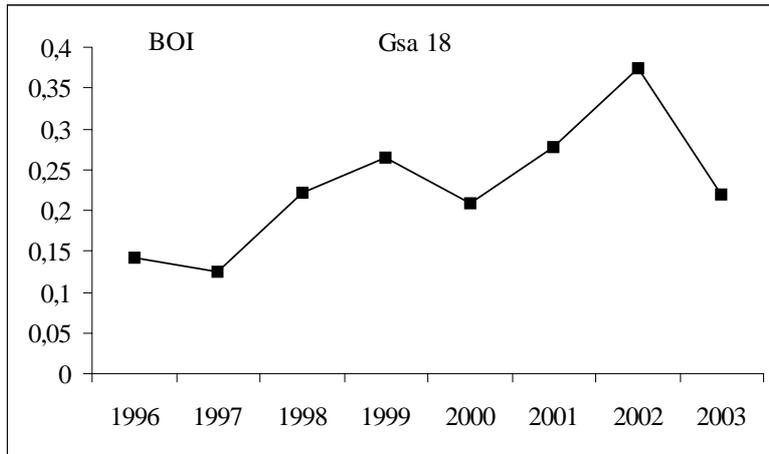
Figure 6: Spatial distribution of biomass index (kg km^{-2}) for Mediterranean hake (from 1996 to 2003).
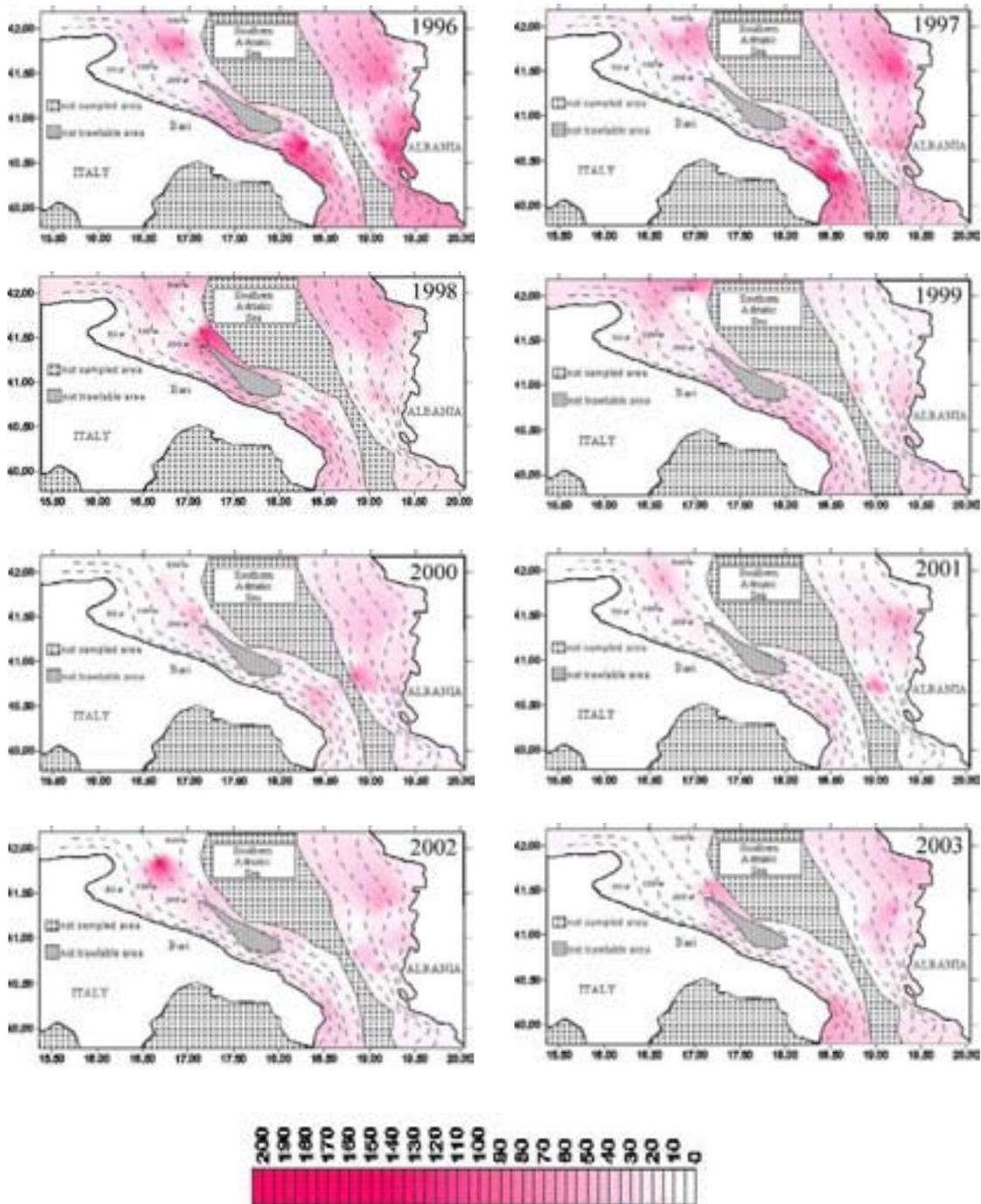
Figure 7: Spatial distribution of abundance index (n km^{-2}) for Mediterranean hake (from 1996 to 2003).
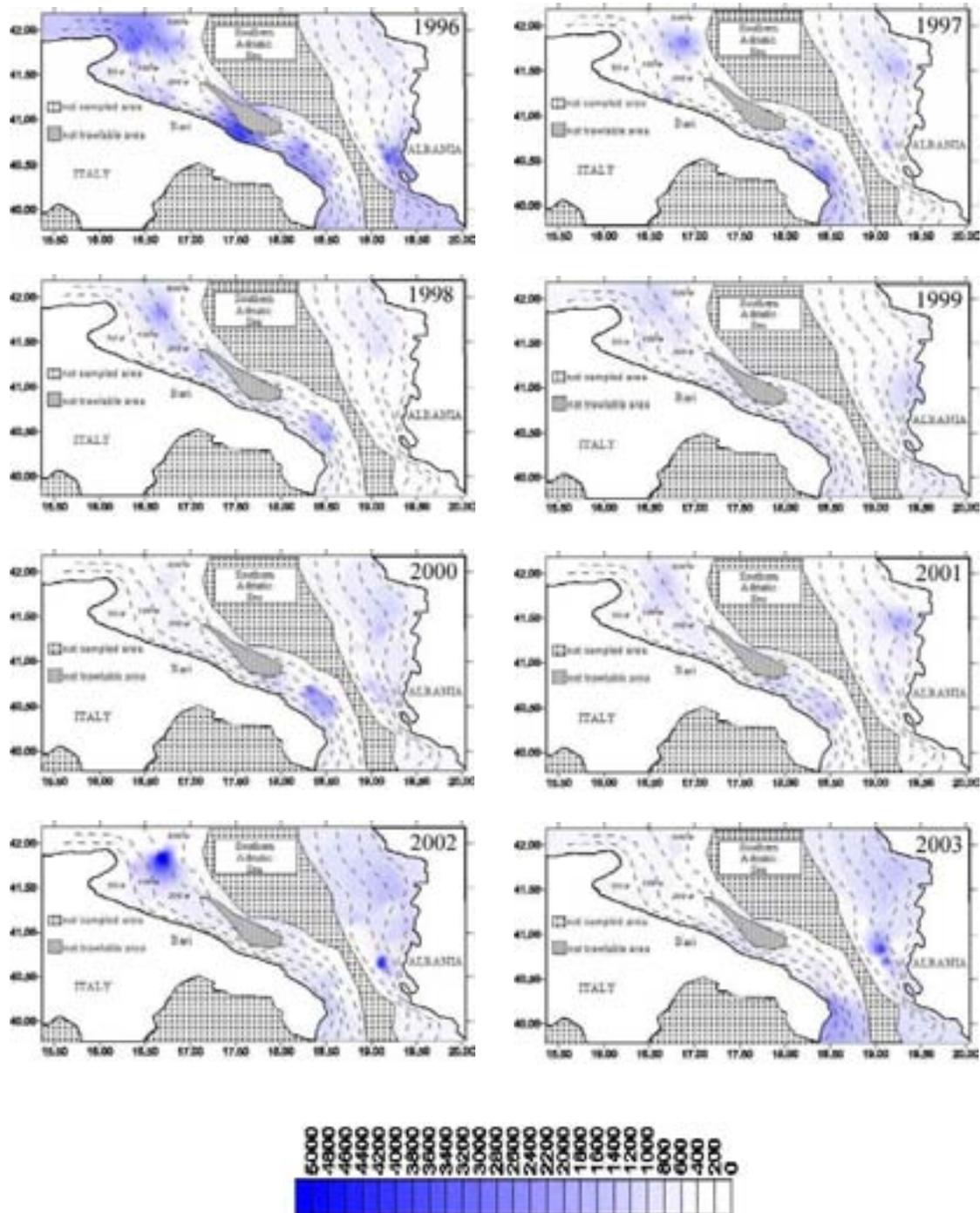
Figure 8: Spatial distribution of the Mediterranean hake mean individual weight (g ind.$^{-1}$).

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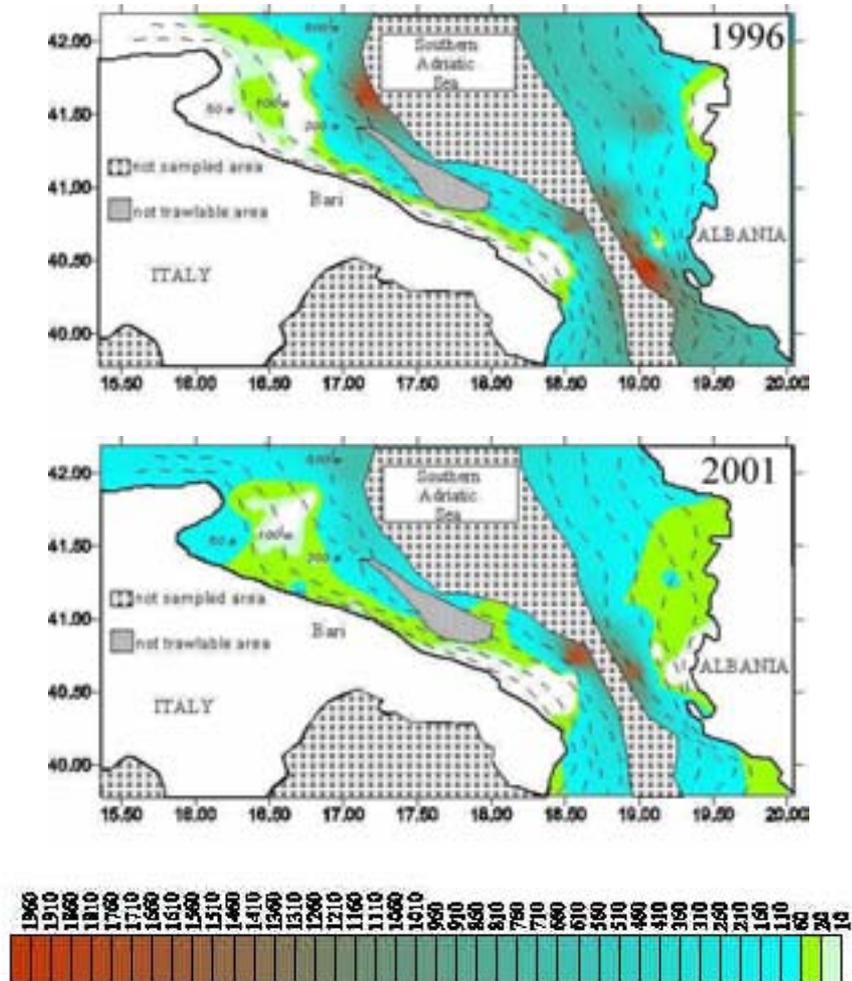
Figure 9: Spatial distribution of biomass index (kg km^{-2}) for deepwater rose shrimp (from 1996 to 2003).
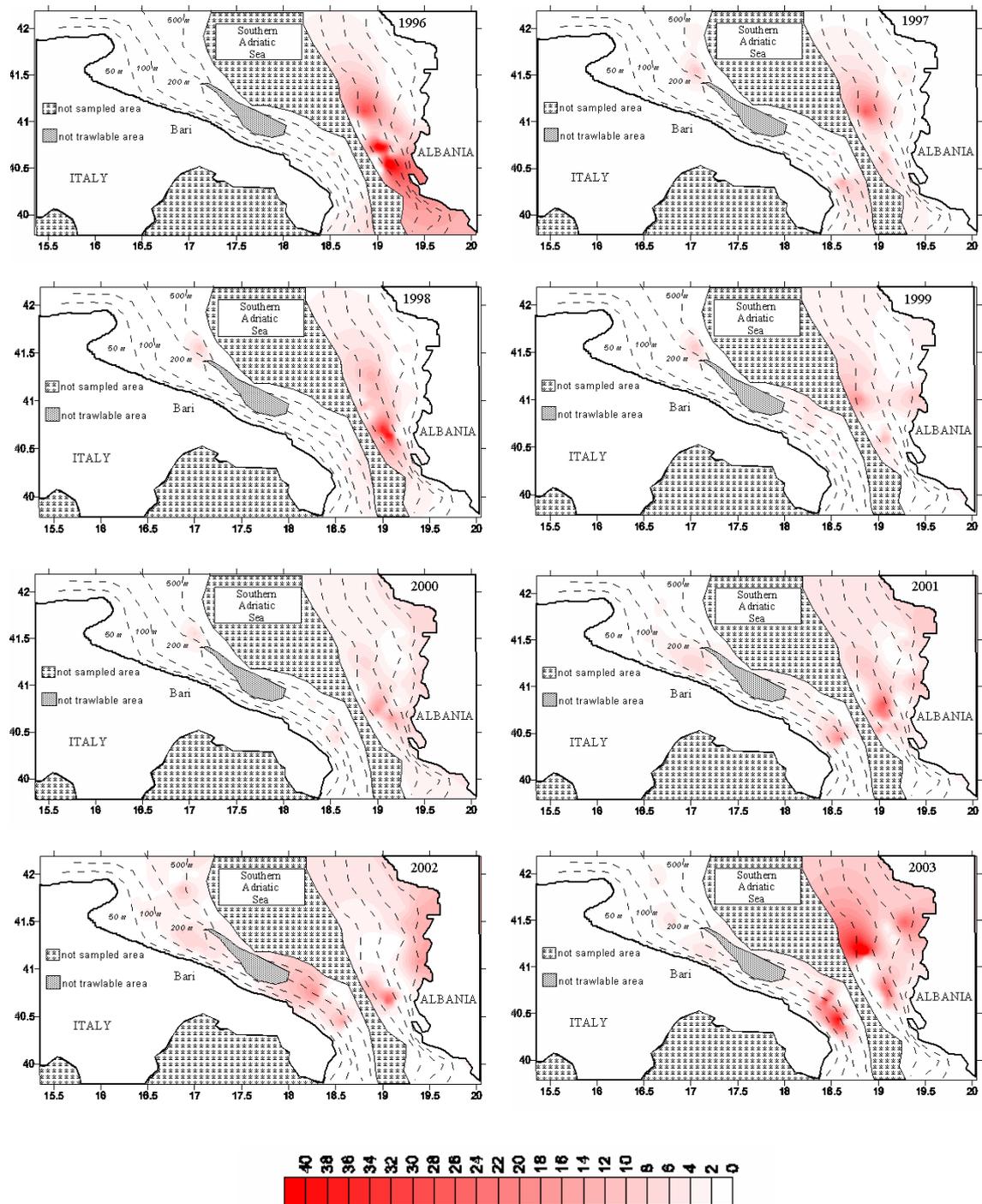
Figure 10: Spatial distribution of abundance index (n km^{-2}) for deepwater rose shrimp (from 1996 to 2003).
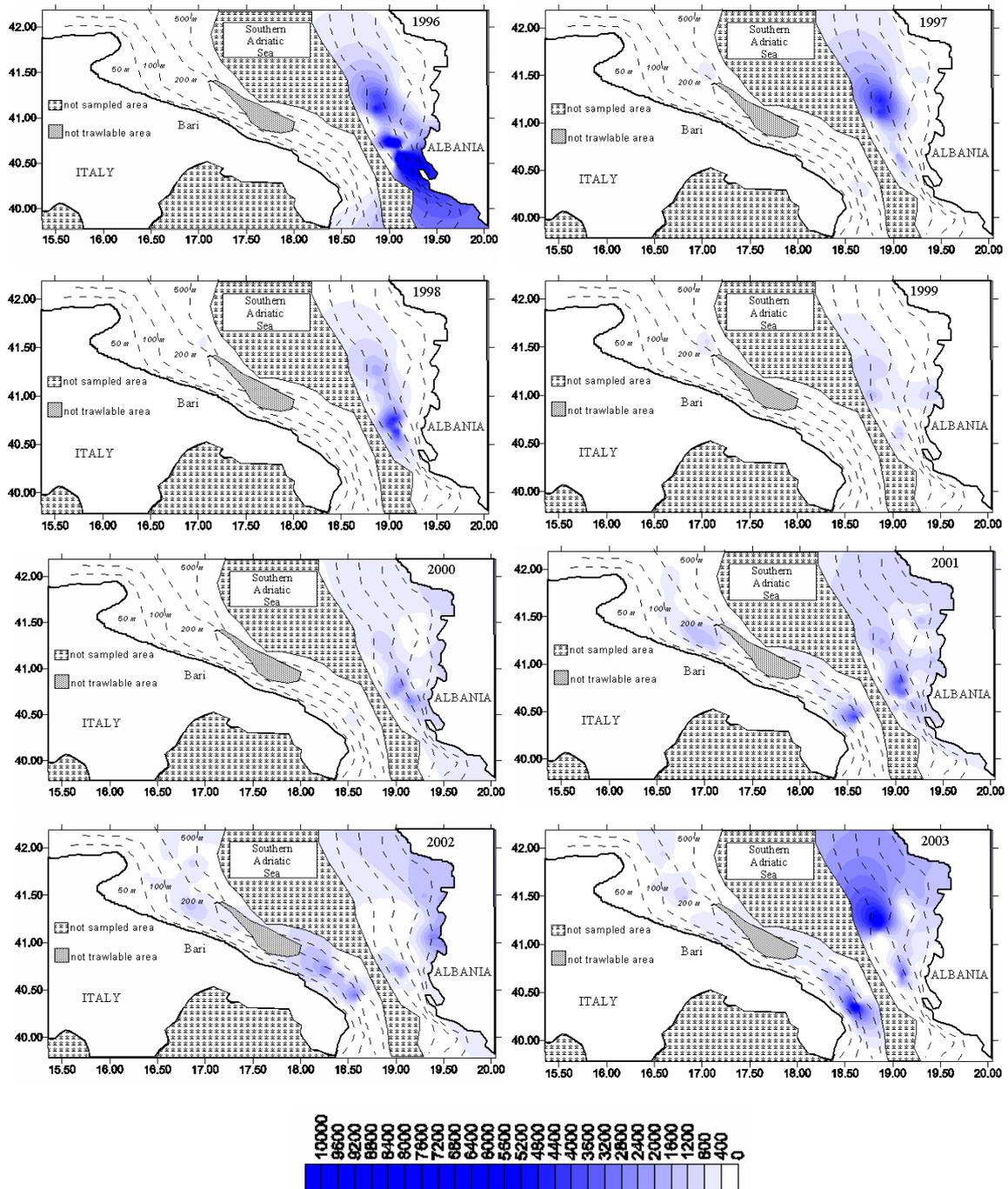
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LOOKING FOR REFERENCE POINTS IN THE MEDITERRANEAN SWORDFISH FISHERY: THE CASE STUDY OF THE LIGURIAN SEA

by

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ABSTRACT

Several teams studying the biology and fishery of swordfish in the Italian Seas, have been recently involved in the first Mediterranean attempts of measuring recruitment processes in this species.

The interest of this kind of research was suggested by ICCAT to the Italian MiPAF. In ICCAT studies, recruitment indexes of large pelagic fish, such as swordfish or bluefin tuna, are presented in the framework of VPA assessment, when the structure of the fished stock is studied in details in the time series. They are also related to large scale climatic or oceanographic factors. In the present note we recall the main biological parameters of the swordfish in our study area, the Ligurian Sea, presenting time series of both longline CPUE (kg per 1000 hooks) and recruitment indexes. Possible relationships between recruitment levels and abundance of spawners are also discussed. Results show that the CPUE of swordfish longlines (kg/1000 hooks) since 1992 are increasing. The recruitment index, in terms of number of fish aged 1 per 1000 hooks, doesn't show a positive trend but is oscillating around an average of 1.72, a figure which appears higher than in oceanic longline fisheries. The adult fish abundance is growing. These positive aspects of the Ligurian fishery appear to be related to the limitation of fishing activities, which were introduced in the study area in the framework of the establishment of an offshore Cetacean Sanctuary.

Keywords: *Ligurian Sea, reference points, longline, swordfish, recruitment index, CPUE, trends.*

Introduction

The Ligurian Sea probably represents the only Mediterranean area in which limitations to fishing activities in offshore waters have been enforced. In fact the fishing grounds of the

swordfish (*Xiphias gladius*, L. 1758) in Western Ligurian Sea (Figure 1) are now included in a "Cetacean Sanctuary" recently established on the basis of international agreements (1999–2001). Moreover, since 1990 the Italian government introduced a fishing ban for swordfish driftnets ("spadare") in order to protect pelagic life in this area. The ban mainly succeeded in stopping foreign vessels which used to arrive in late summer in the Ligurian Sea to complete their fishing season started in southern Tyrrhenian and Sicilian waters. Since 1992 the ban was completely enforced in the protected area, i.e. included also the small resident fleet.

Studying the biology and fishery of swordfish in the Ligurian Sea in the framework of national programmes of assessment, we have elaborated both CPUE and recruitment indexes from 1990 onward. These two data sets are usually (e.g. in ICCAT assessment, Mejuto, 1999; 2000; 2001; 2003a,b) assumed to be important to identifying suitable reference points for this fishery. Aim of this note is to present the trends of such indexes in our study area, the Ligurian Sea, and analyse possible relationships between them.

Materials and methods

Swordfish fishery activities have been monitored yearly (since 1990) in the harbours of Sanremo and Imperia, which produce about 90% of the landings of the Ligurian coast. The fishing activity was monitored daily in Sanremo and weekly in Imperia. The fishing season goes from June to December and the most important gear commonly used in the study area is swordfish longline.

During each day of observation at landings, all the information about the various fishing boat were collected, i.e. total number of fish, total weight of the catches, total number of used hooks, etc., to obtain CPUE in number and weight per 1000 hooks of swordfish longline. Moreover, all the caught specimens were measured (lower jaw fork length, LJFL) and weighted (gutted weight, GW) in order to obtain length (L/F) and weight-frequency distributions.

According to ICCAT SCRS indications, the recruitment index of the swordfish in the longline fishing is calculated as number of 1-year specimens, caught by 1000 hooks (Mejuto, 1999; 2000; 2001; 2003a,b).

To calculate the recruitment index on the Ligurian data, only fish caught during the period July-September, which represents the main part of the fishing season, were considered. This is because the recruitment occurs in autumn, thus the first modal group in summer length frequency distributions is represented by the fish aged 1. However, in some years (e.g. 2003, Figure 4) also a small group of 0-aged fish appeared, this fish was overlooked.

On the basis of Battacharya's method (Gayanilo, Sparre and Pauly, 1994), both the first and the second age groups were isolated in L/F distributions of each year, from 1990 onward, and related to the fishing effort. The identification of such age groups was made on the basis of a growth function previously elaborated for the study area (Orsi Relini *et al.*, 1999a). We calculated the following CPUEs:

CPUE (N):	number of specimens/1000 hooks
CPUE (W):	weight (kg) of specimens/1000 hooks
CPUE (age 1):	number of specimens aged 1/1000 hooks=recruitment index
CPUE (age 2):	number of specimens aged 2/1000 hooks

Results

Total CPUE

The evolution of CPUE in weight showed a significant (α=0.01) temporal trend in the studied period (Figure 2).

In detail, it is possible to observe a decrease of the CPUE between 1990 and 1992; after 1992 the values regularly increased till 2003, albeit with large fluctuations especially from 1998 to 2002. The overall trend represents an important amelioration of the catches.

It is interesting to verify if the increased catches are due to more numerous or to larger swordfish. The CPUEs in number of individuals (N) (Figure 3; significant trend, α=0.01) show that the fish caught by 1000 hooks increased in number from about 3.5 to more than 7, while the average size was slightly changed (see also the weight increase in Figure2). Thus, if we assume a previous state of overfishing (from the negative trend before 1992), to which the protection of the area represented a remediation, it was more probably a recruitment overfishing than a growth overfishing, because the banned driftnets caught mainly adults individuals.

Recruitment index

In the period 1990–2003 the CPUE values (age 1) ranged between a minimum of 0.4 and a maximum of 4.5 specimens/1000 hooks. Their temporal evolution is shown in Figure 4.

The recruitment index does not appear to show any temporal trend, but there are fluctuations around an average value (mean 1.72; SD 1.11), becoming annual since 1997. It is interesting to verify these features in the length/frequency distributions (Figure 5): a relevance of the first cohort, i.e. age 1 specimens, with modal length of about 95 cm LJFL, occurs alternately with that of larger specimens (modal length 115 cm LJFL or more). In other words, each recruitment pulse can be traced in the following year by means of the abundance of specimens aged 2; in fact the two indexes of the same cohort of fish one year apart are significantly correlated (Figure 6; α=0.01).

To have a more detailed view of the temporal evolution, the recruitment can be described by a polynomial function (Figure 7), in which two periods can be separated: a first one, when the recruitment is scarce and a second one in which it is more abundant.

Positive effects in the fishery are delayed mainly to the second period. These facts suggest that, after recruitment, some years are necessary to obtain significant biomasses. In the present series of data, if a delay of 3 years is introduced between CPUE (W) and the recruitment index, the two temporal data sets appear positively related (Figure 8; significant R^2 for α=0.05).

Discussion

In the Ligurian Sea, the establishment of a Cetacean Sanctuary by international agreement is recent (November 1999), but since 1990 unilateral actions of the Italian Government were aimed at reducing the impact of drifnets (spadare) on the large pelagic fauna. Such measures were promoted on the legal basis of the "Fishery Act 1965", which mentioned the possibility to

establish "zone di tutela biologica" (i.e. a protected areas for the recovery of a depleted resource). On request of the Ministero della Marina Mercantile (30 March 1990), we had a part in promoting a protected area where cetaceans (very important for their ecological value) and the swordfish (a high valuable commercial fishery resource) were associated, in an *ante litteram* ecosystem approach to the management. A zone of this kind was established on 18 July 1990 (Orsi Relini *et al.*, 1992). However, at that time, we were at the beginning of our study of large pelagic nekton of the Sanctuary and our statement were largely theoretical.

The swordfish has a long life and females are characterized by a late sexual maturity. In the Ligurian Sea, known to be also a spawning area (Garibaldi, Palandri and Orsi Relini, 2004), the maturity size, L_{50}, resulted 149 cm LJFL, which corresponds to an age of 4 years (Orsi Relini, Palandri and Garibaldi, 2003); ages up to 11 years were observed for females (Rollandi *et al.*, 2004), but are extremely rare. A useful combination of number per age occurs in relatively short time. Tagging experiences carried out in the area using traditional "spaghetti" tags, suggested that the population could be partially resident (Garibaldi, Palandri and Orsi Relini, 1999). In this way, protection measures addressed to cetaceans were "*de facto*" very important for fishery resources also.

The present time series shows that, after a lag of few years, the CPUE values obtained by longlines (which are very selective in respect of their target, more than driftnet - Orsi Relini *et al.*, 1999b), increased year after year, reaching during 2001 more than 155 kg, a figure rare in the Mediterranean. In our opinion, we face here a first interesting point: for swordfish, population recovery needed a lag of about 3 years. The catches composition, in terms of age, explains such effect: in fact the commercial product is mainly formed by fish aged 1, 2 and 3.

In the last period of the studied series (1998–2003), CPUEs showed fluctuations, which however were around a general positive trend. In very recent times, writing the present notes (2004–2005) however, we observe that the swordfish catches are decreasing, reaching in 2005 one of the lowest CPUE of the series.

Not differently from the first years of the nineties, the present decreasing of the CPUE seems related to overfishing, due to a combination of different situations.

Recalling a traditional fishery on tuna which occurred on coastal waters of Marseille, both by mean of a dozen of small tuna trap and also with pelagic gear (Gourret, 1894), French fishermen began a new fishing activity in offshore waters addressed to large pelagic fish and based on a net called "thonaille". This is a drifting gear targeting the bluefin tuna, the albacore but also the swordfish, whose quantities are growing accordingly to the increasing of the mesh size. Started in 1998, in 2000 this activity was carried out by 46 vessels with the aim to reach one hundred in the following years. At the present, at least one third of the French fleet reaches the Sanctuary with an effect on the resident swordfish population similar to those of the above mentioned "spadare".

In the meantime, the illegal driftnet fishery in EU countries and the relatively new driftnet fishery developed by the southern Mediterranean countries, such as Morocco (Tudela *et al.*, 2003; ICCAT, 2004), increased the total fishing effort in the Western Mediterranean basin.

In the Ligurian Sea, the decline of catches has been predicted in the trend of recruitment index. In the figure 7, a decreasing phase appears in the last years and the 2003 figure is the

lowest of the series. We have found a significant relationship between recruitment index and catches, introducing a time lag of 3 years; thus, in our study area, the predictive value of such reference point is confirmed.

Only with the continuation of similar studies, these first attempts will gain accuracy and possibly their predictive value might be consolidated.

Figures

Figure 1: The Cetacean Sanctuary of the Ligurian Sea and the main swordfish fishing areas of the local fleet (shaded areas).
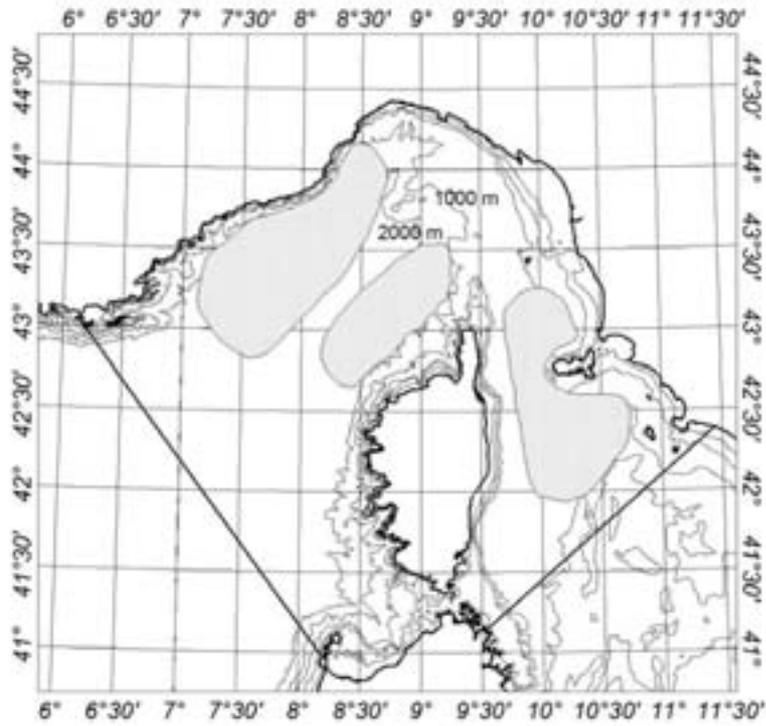
Figure 2: Trend of CPUE (kg per 1000 hooks) (mean annual value) of the swordfish caught by longlines in the Ligurian sea.
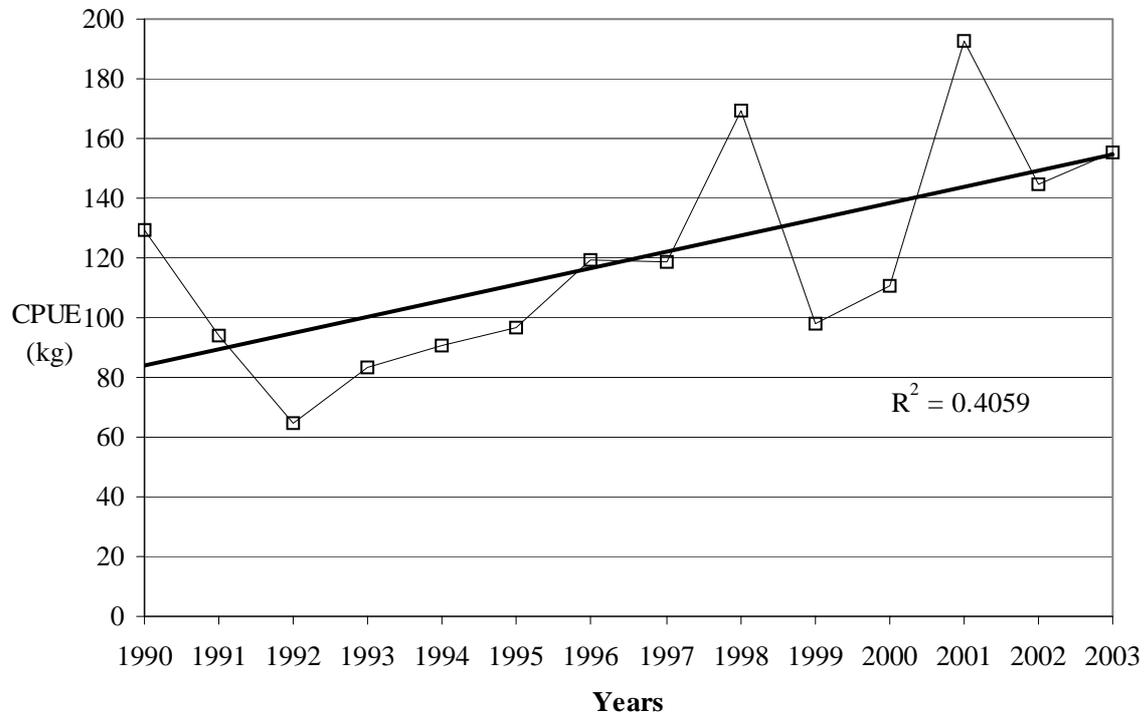
Figure 3: Trend of CPUE (N) of the swordfish caught by longlines in the Ligurian sea.
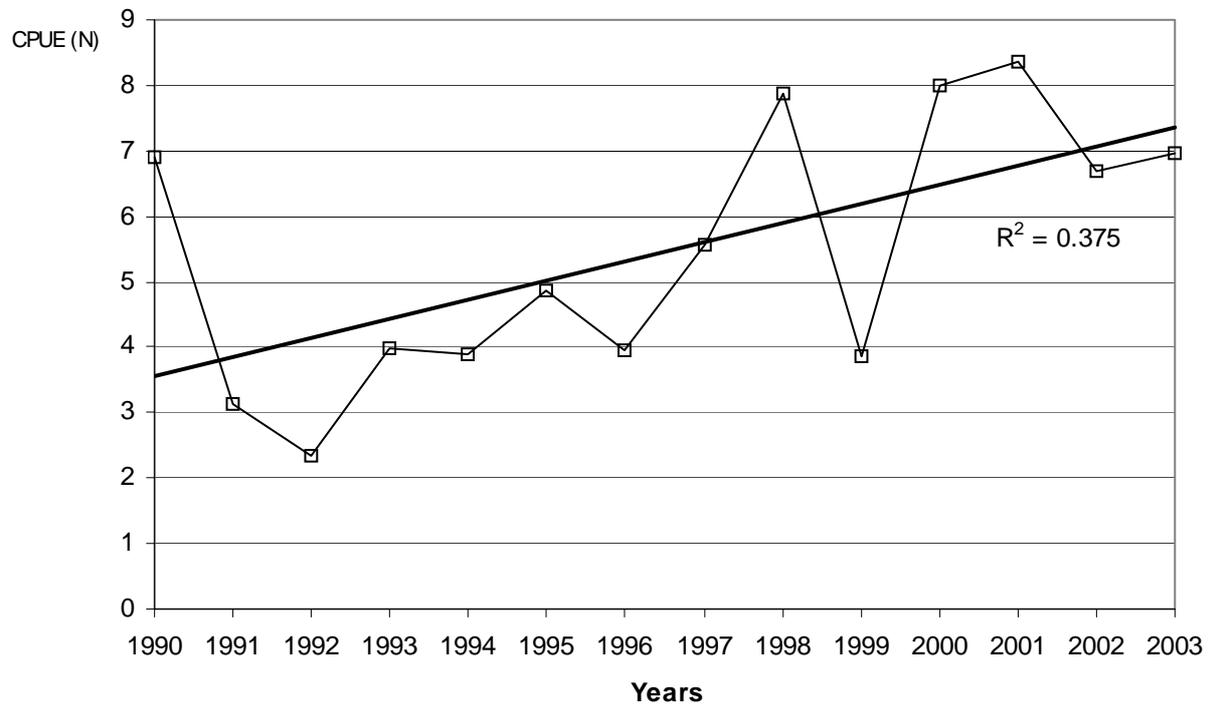
Figure 4: Values of the recruitment index (CPUE of specimens of age 1) of the swordfish in the studied period. It is not possible to indicate a linear trend.
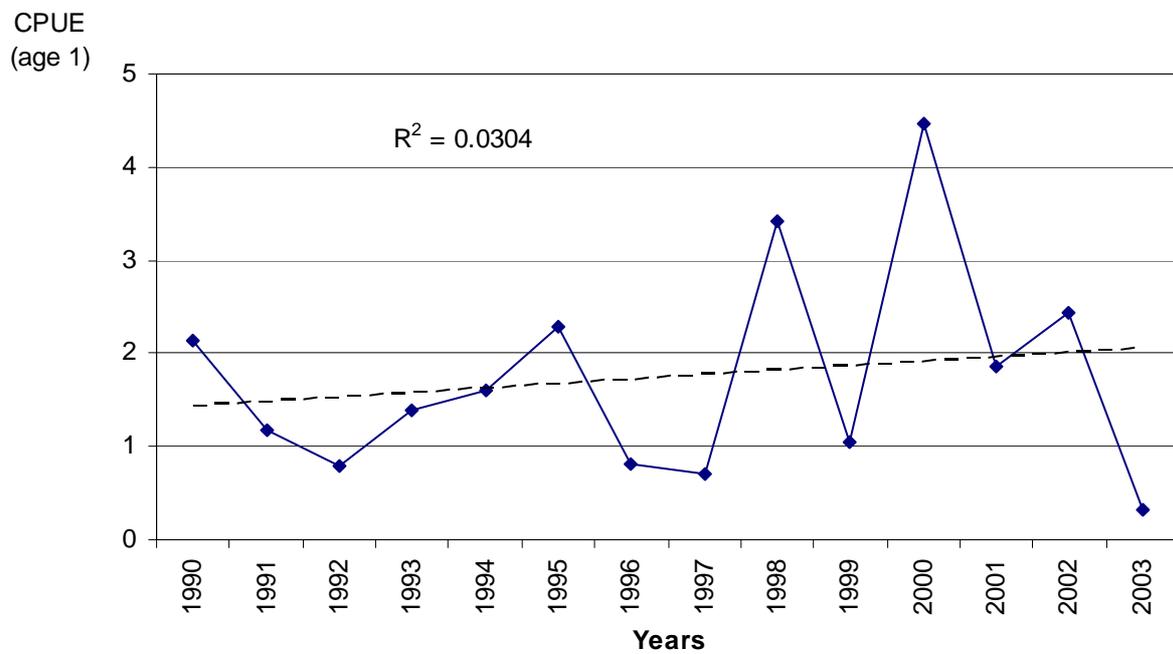
Figure 5: Length frequency distributions of swordfish summer catches from 1998 to 2003.
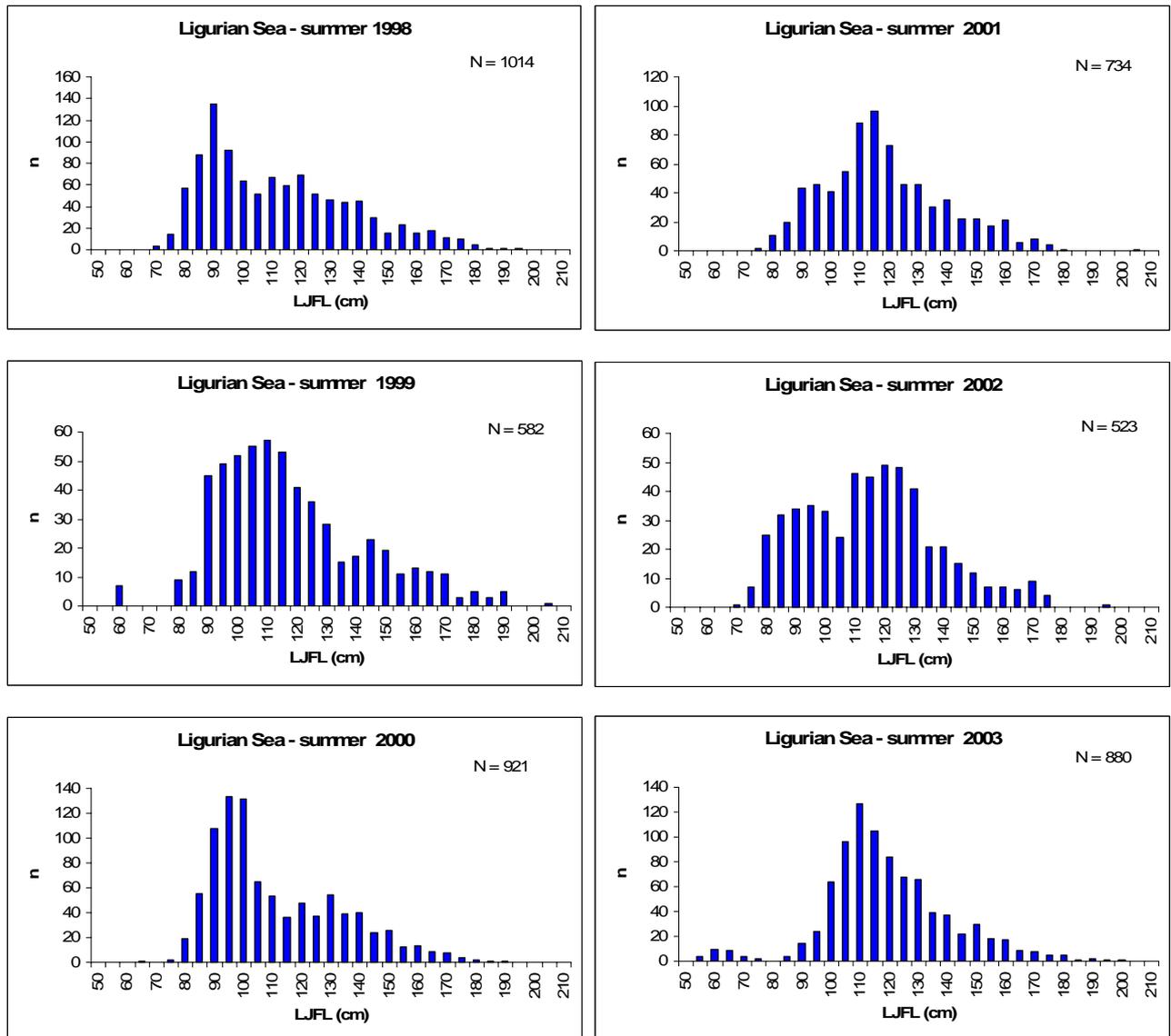
Figure 6: Correlation between the CPUE (age 1) and CPUE (age 2) of the following year, in the longline catches of the Ligurian Sea.
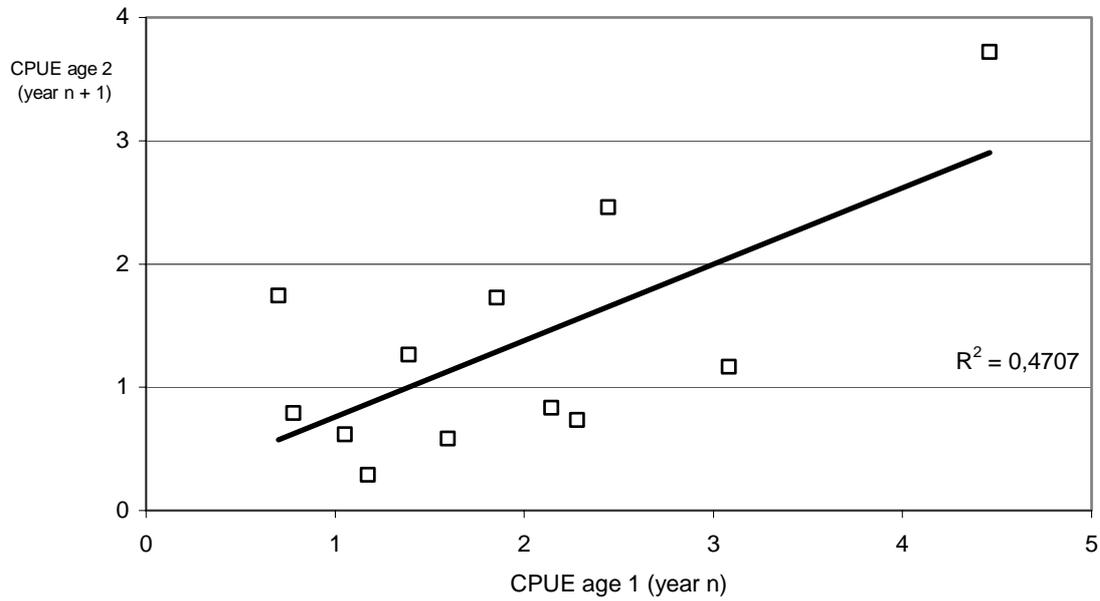
Figure 7: Recruitment index (CPUEn of fishes of age 1) described by a polynomial function.
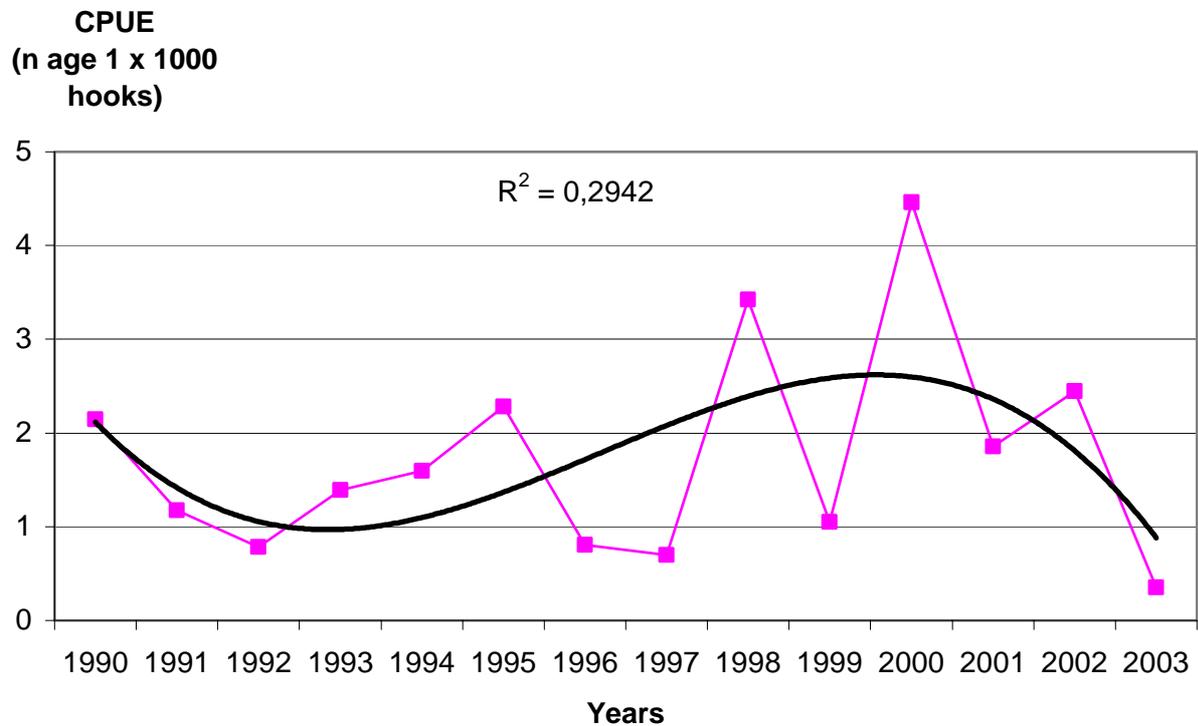
Figure 8: Correlation between the CPUE (age 1) and CPUE (W) recorded 3 years after, in the longline catches of the Ligurian Sea.
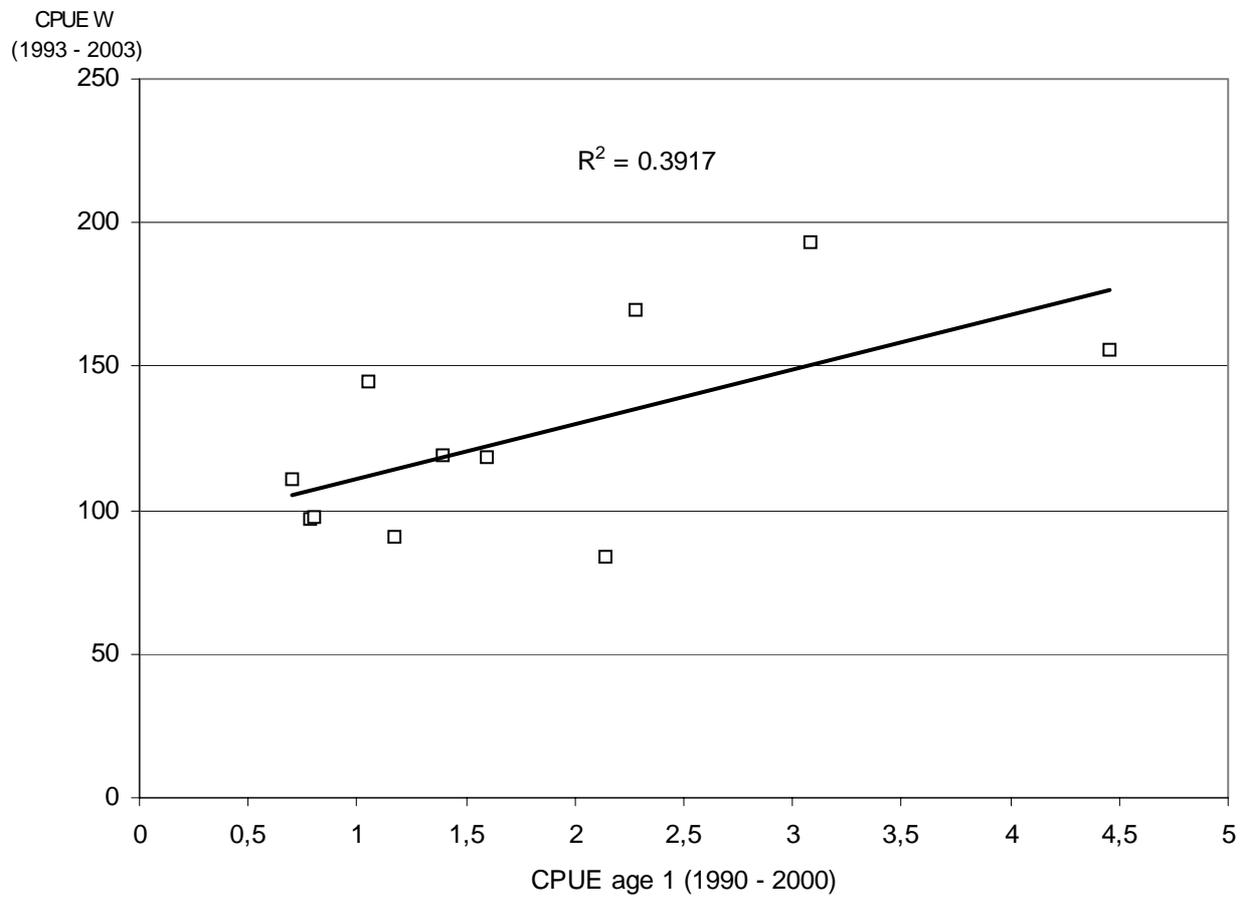
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USE OF AN EXPLOITATION RATE THRESHOLD IN THE MANAGEMENT OF ANCHOVY AND SARDINE STOCKS IN THE ADRIATIC SEA

by

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ABSTRACT

The use of a biological reference point based on exploitation rate threshold was evaluated for the stocks of anchovy and sardine in the central and northern Adriatic Sea. This threshold was suggested for small pelagics by K. Patterson in 1992. The values of the fishing mortality rates F in the exploitation rates F/(M+F), with M being the rate of natural mortality, were derived from VPA carried out for the time interval 1975-2001. The results were encouraging so that this threshold could be used to prevent stock collapse along with the Minimum Biological Acceptable Level (MBAL) based on spawning stock biomass, currently implemented for small pelagic fish and other species. The same exploitation rate threshold could even substitute MBAL, when not sufficiently long time series of stock-recruitment data are available to obtain reliable estimates of MBAL.

Key-words: *Adriatic Sea,* Engraulis encrasicolus, Sardina pilchardus, *stock assessment, fishery management*.

Introduction

Anchovy (*Engraulis encrasicolus*, L.) and sardine (*Sardina pilchardus*, Walb.) are among the most important commercial species both in the Mediterranean and Adriatic Sea. Since 1975, the ISMAR Marine Fisheries Section of Ancona has been conducting research on the biology and stock assessment of both species in the central and northern Adriatic, by means of population dynamics methods (Cingolani, Giannetti and Arneri, 1996; Cingolani *et al.*, 1998 and 2000; Santojanni *et al.*, 2001; Cingolani *et al.*, 2002a,b; Santojanni *et al.*, 2002, 2003).

In the management perspective of these stocks, as for other exploited species, the choice of appropriate biological reference points is an important matter. These indicators are utilized to avoid an excessive fishing mortality and, thus, depletion of resources (Hilborn and Walters, 1992; Patterson, 1992; Myers *et al.*, 1994; Haddon, 2001; Murawski, Rago and Trippel, 2001; STECF, 2002). In the present paper, the use of a critical threshold for the exploitation rate is discussed. This application is relative to recent assessments of anchovy and sardine stocks in the ambit of a project funded by Ministero Italiano per le Politiche Agricole e Forestali (Cingolani *et al.*, 2002b). These assessment were carried out by means of Virtual Population Analysis (VPA) on the two corresponding time series from 1975 to 2001 (the data collection for subsequent years is still ongoing).

The exploitation rate is given by the ratio between the two instantaneous rates of fishing mortality, F, and total mortality, Z, which is equal to F+M, with M being the natural mortality rate. In the present work, both F and M are always meant on annual basis. The critical threshold for F/Z was taken to be equal 0.4. Such a value was suggested by Patterson (1992) as a possible biological reference point for the management of small pelagics. In order to find it, Patterson used data derived from many literature sources, represented from values of biomass and corresponding fishing mortality rates estimated by VPA, along with the same M values employed by VPA, referring to assessments of 28 stocks distributed in all the world and relative to 11 species of small pelagics: *Sardina pilchardus*, *Sprattus sprattus*, *Clupea harengus*, etc. The ratio F/Z was used instead of F because the former one allows to obtain more comparable indices, which account for different natural mortality in different species and even stocks of the same species. The rates of variation of biomass, plotted as a function of F/Z, were calculated by the author using time intervals ranging from 5 to 10 years. In the case of those stocks with long historical series, more than one time interval was used, so that the data points fitted by means of regression techniques were 53, instead of 28, i.e. the number of examined stocks. The statistical analysis performed by Patterson suggested that, over the value F/Z = 0.4, stocks show high probabilities of decline. Such probabilities become particularly high over F/Z = 0.5, which is obtained when F = M: this outcome is consistent with the conclusions derived by other authors on the basis of different methods (Patterson, 1992). On the contrary, a value of F/Z under 0.3 seems to be compatible with increasing stock abundance and associated to a relatively low risk of decline.

Finally, in the same work, Patterson remembered the disadvantages shown by different biological reference points applied in the management of small pelagic stocks. In particular, he emphasized that the minimum biological acceptable level (MBAL), currently implemented (STECF, 2002), requires long time series relative to the abundance of both spawners and recruits. In fact, only long series allow to "capture" the threshold of spawning stock biomass to be maintained at sea, under which, otherwise, a strong decline is likely to occur.

Materials and methods

Three kinds of data were collected, related to catch, fishing effort, biometric features. Landing data were collected on census basis for both west (Italy) and east (Slovenia, Croatia, ex Yugoslavia) Adriatic ports (Cingolani, Giannetti and Arneri, 1996; Santojanni *et al.*, 2003). The geographical distribution of the ports involved in the data collection has Vieste as southern limit (Figure 1), so that the whole GSA 17 of GFCM is investigated. Anchovy landings can be considered a reliable estimate of catches because the Adriatic anchovy is mainly fished by the Italian fleet and, in this country, discarding of anchovy is thought to be negligible (Cingolani,

Giannetti and Arneri, 1996; Cingolani *et al.*, 2000). For sardine, a more relevant practice of discarding at sea occurred in some Italian ports since the end of the 1980s onwards, and estimates of discards – ranging from around 900 to 4 000 tonnes per year – were obtained (Cingolani *et al.*, 2000, 2002a).

Figure 1: Geographic extent of the small pelagic data collection: the port of Vieste represents the southern limit (see text).
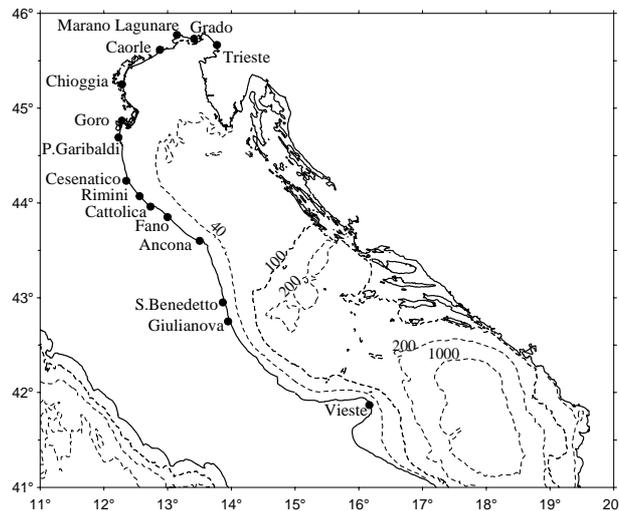
All these landings are relative to mid-water trawlers (in Italy *volanti*) and purse-seiners (in Italy *lampare*) using fish attraction by light (Cingolani, Giannetti and Arneri, 1996), which usually target schools formed by individuals larger than about 9 cm in length.

Biological samples were collected in the most important Italian ports on the basis of landed catches, in order to obtain length, weight, age data of both species. Age of fish was estimated by reading otoliths. The use of calendar year data in fishery stock assessments implies that the conventional birthday (*i.e.* the day on which a cohort grows one year older) is on the first day of January. This is the case for Adriatic sardine but not for anchovy. Since the reproduction of the Adriatic anchovy is particularly relevant in spring-summer (Regner, 1996), a conventional birthday on the first of June is more sensible: the assessment for this species was thus carried taking into account such a birthday date. The birthday effect is expected to be not negligible more likely in assessments based on catch-at-age data, just like VPA. Consequently, all data originally recorded according to calendar year were then modified in order to calculate split year ones, so that data relative to one year were referred to the time interval ranging from the first of June of the year before up to the 31st day of May of that year (Santojanni *et al.*, 2003).

The VPA is a method used for many species, including small pelagics. Some examples from literature, relative just to small pelagics, can be found for stocks distributed in all the world (Patterson, 1992; Schwartzlose *et al.*, 1999; Barange, 2001). The VPA yields estimates of the stock abundance at sea on the basis of catch-at-age data over time and, to do that, it assumes the instantaneous natural mortality rate to be constant over time and age classes (Hilborn and Walters, 1992; Haddon, 2001). The values $M = 0.6$ yr^{-1} and $M = 0.5$ yr^{-1} were used for anchovy and sardine, respectively, taking into account both the age composition of catches and estimates reported in literature (Cingolani *et al.*, 2002b; Santojanni *et al.*, 2003). In

particular, the value M = 0.5 was obtained for Adriatic sardine by Sinovcic (1986). It is also of some interest to remember that values of M ranging from 0.29 to 0.62 were obtained for a well studied sardine stock (Catalan Sea) in the Mediterranean (Pertierra and Perrotta, 1993).

Table 1 and 2 show examples of age-length key for anchovy and sardine, respectively. It should be noted for sardine that individuals older than 6 up to 12 years are recorded, suggesting relatively high longevity and, hence, probably relatively low natural mortality. In other words, that can justify the use of M = 0.5, because higher values of M could not explain the occurrence of the old individuals mentioned. Further, this is derived from catches: without the fishing pressure (*i.e.* Z given by M only), these older individuals would probably be better represented in the population at sea and, thus, more frequently encountered in sampling.

Table 1:   Anchovy: age (year) - length (cm) key calculated for the year 1995. Proportions represent age frequency distributions by length class

Length	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6
9	0	0	0	0	0	0	0
9.5	0.98	0.02	0	0	0	0	0
10	0.86	0.14	0	0	0	0	0
10.5	0.92	0.08	0	0	0	0	0
11	0.75	0.25	0	0	0	0	0
11.5	0.50	0.50	0	0	0	0	0
12	0.26	0.74	0	0	0	0	0
12.5	0.16	0.83	0.01	0	0	0	0
13	0.06	0.81	0.14	0	0	0	0
13.5	0.01	0.45	0.52	0.02	0	0	0
14	0	0.17	0.45	0.37	0.02	0	0
14.5	0	0.11	0.22	0.57	0.11	0	0
15	0	0.02	0.16	0.36	0.40	0.07	0
15.5	0	0.01	0.21	0.24	0.39	0.14	0.01
16	0	0	0.07	0.30	0.29	0.28	0.06
16.5	0	0	0.06	0.23	0.19	0.32	0.19
17	0	0	0	0.42	0.19	0.15	0.23
17.5	0	0	0	0.33	0.50	0.17	0
18	0	0	0	0.67	0.33	0	0
18.5	0	0	0	0	0	0	0

Table 2: Sardine: age (year) - length (cm) key calculated for the year 1998. Proportions represent age frequency distributions by length class.

Length	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9	Age 10	Age 11	Age 12
13	1	0	0	0	0	0	0	0	0	0	0	0	0
13.5	0.72	0.28	0	0	0	0	0	0	0	0	0	0	0
14	0.33	0.67	0	0	0	0	0	0	0	0	0	0	0
14.5	0.26	0.69	0.05	0	0	0	0	0	0	0	0	0	0
15	0.08	0.64	0.25	0.03	0	0	0	0	0	0	0	0	0
15.5	0.02	0.38	0.53	0.07	0	0	0	0	0	0	0	0	0
16	0	0.03	0.56	0.39	0.02	0	0	0	0	0	0	0	0
16.5	0	0.01	0.31	0.49	0.17	0.02	0	0	0	0	0	0	0
17	0	0	0.06	0.42	0.42	0.09	0	0.01	0	0	0	0	0
17.5	0	0	0	0.15	0.47	0.30	0.06	0.01	0	0	0.01	0	0
18	0	0	0	0.03	0.29	0.54	0.12	0.02	0	0	0	0	0
18.5	0	0	0	0	0.07	0.38	0.35	0.14	0.03	0.02	0.01	0	0
19	0	0	0	0	0	0.04	0.39	0.37	0.14	0.04	0.02	0	0
19.5	0	0	0	0	0	0	0	0.50	0.15	0.15	0.15	0.05	0
20	0	0	0	0	0	0	0	0	0.25	0.50	0	0	0.25
20.5	0	0	0	0	0	0	0	1	0	0	0	0	0

The longevity suggested for anchovy by the age-length key is lower than for sardine, with the oldest individuals being in the age class 6. The age classes 5 and 6 resulted to be in other annual age-length keys, except for the years immediately after the collapse of the stock (Santojanni *et al*., 2003; see also below). Anyway, the value 0.6 is near the lowest end limit of the interval of values reported in literature and, here, was taken also according to a precautionary approach (Santojanni *et al*., 2003). Therefore, VPA was also performed using $M = 0.8$ and results are reported below when the disadvantages of the critical threshold of F/Z are discussed. Again, a reference to the well studied case of Catalan Sea is worth: for the anchovy stock, Pertierra and Lleonart (1996) mentioned $M = 0.54$ and $M = 0.81$, which are very similar to the values employed here for Adriatic anchovy. Finally, the criterion to select M on the basis of the inverse relationship with longevity is adopted for small pelagics by other authors; as reported in a recent GLOBEC report (Barange, 2001), Pacific sardine (*Sardinops sagax*) is usually assumed to have a relatively low annual natural mortality rate, $M = 0.4$, and a lifespan of about 10 years, whereas for northern anchovy (*Engraulis mordax*) $M = 0.8$ is associated to a lifespan of about 4 years.

The VPA allows to calculate annual values of stock abundance at sea and fishing mortality rates by age and year. The rates relative to the oldest age groups and most recent (final) year were calculated using the procedure defined by Cingolani *et al*. (2002b) and Santojanni *et al*. (2003); this, in particular, involved Laurec-Shepherd tuning on CPUE-at-age data obtained for Porto Garibaldi (Santojanni *et al*., 2002), whose fleet accounts over years for 20-25% of catches of anchovy as well as sardine. The software package employed for the VPA runs was MAFF-VPA, developed by Darby and Flatman (1994).

The annual exploitation rates, F/Z, were calculated using the same values of M utilized by VPA and the annual values of F_{0-3} and F_{0-5} for anchovy and sardine, respectively. These values of F are the averages of the F-at-age values yielded by VPA for the age intervals 0-3 and 0-5. They were calculated as arithmetic unweighted means or FBAR (Darby and Flatman, 1994). As stressed by Patterson (1992), the same F values, weighted on the basis of the abundance at sea by age class, would have been preferable. However, Patterson performed statistical analysis on a data set mainly formed by unweighted averages. Hence, the threshold and/or grid for F/Z pointed by this analysis have to be thought like an "instrument tuned" on unweighted averages. In fact, unweighted averages are expected to be higher than weighted ones: in the latter case, the arithmetic weight of younger age classes increases because of the higher abundance, and lower values of F are usually estimated for these classes. Anyway, short accounts about the use of so weighted mean F values were reported.

Results and discussion

The annual values, on split year basis, of mid-year stock biomass at sea of anchovy are shown in Figure 2, along with the corresponding biomass of spawners and annual total catches. These represent, on the average, the 24% of mid-year stock biomass. The trends of biomass at sea show higher values in the second half of the 1970s and, then, a decline till 1987, when the minimum level of the series is observed. In consequence of that, in the calendar 1987, a strong crisis of this fishery occurred in the Adriatic, as pointed by the fall of catches in the same figure. After the collapse, a partial recovery of both abundance and catches is observed. The increase in 2001 is likely an overestimate yielded by VPA, due the fact that catch-at-age data processed by VPA cannot account for the complete history of the most recent cohorts.

Figure 2: Anchovy: total annual catch, stock biomass and spawning biomass calculated in the middle of the year by VPA, from 1976 to 2001 (split year data, see text).
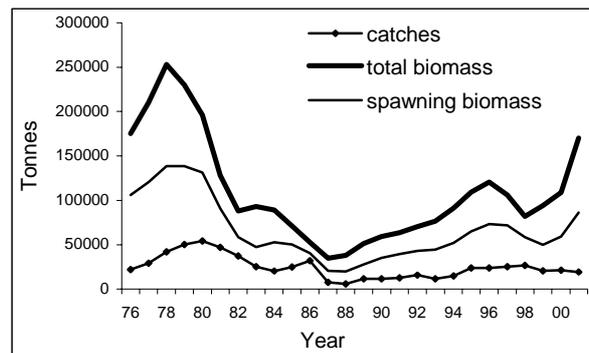
In Figure 3, the annual values of stock biomass estimated by VPA are compared with the corresponding series of biomass density obtained by echo-surveys approximately in the same Adriatic area (Azzali *et al.*, 2002a). The two trends are not very different. Both methods perceive the peak in the second half of the 1970s, strong decline and partial recovery. The contrast between the first period mentioned and the end of the 1980s resulted to be also in the assessment based on ichtyoplanktonic method carried out by Regner (1996), as shown by Santojanni *et al.* (2003). Coming back to acoustics, the low level of abundance in 1976 strongly increased in the subsequent year and the decrease at the beginning of the 1990s are not

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seen by VPA: it is difficult to understand if these fluctuations obtained by the echo-surveys are due to changes in the natural mortality, not taken into account by VPA, or to different sources of error for both methods. The series shown here for acoustics is updated to 1998: data from 1999 to 2001, shown by Azzali *et al.* (2002b) in an unpublished report, are still consistent with the VPA trend.

Figure 3: Anchovy: mid-year stock biomass estimated by split year (see text) VPA and biomass density obtained by echo-surveys, are compared over years.
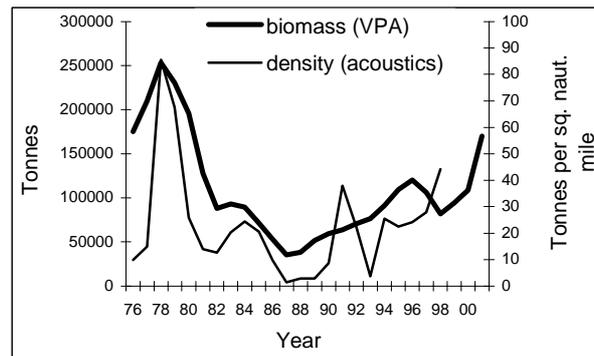
Figure 4 shows the estimated annual values of F_{0-3} obtained for anchovy: they increase progressively since 1976 and a quite high value is observed just in 1986, $F_{0-3} = 0.67$. Likely, a high fishing pressure could have contributed to the collapse. However, the levels of recruitment in 1986 and 1987 were particularly low in relation to the spawning biomass of the previous years 1985 and 1986: similar levels of spawners yielded higher levels of recruits in other years, e.g. 1995 (see below). This suggests that different factors, environmental ones and/or interactions with other species, could have contributed to the collapse (Cingolani *et al.,* 2002b; Santojanni *et al.*, 2003). The importance of multiple factors in determining the population dynamics of small pelagics – such as other species – is widely discussed in literature (Smith, 1983; Lasker, 1985; Beverton, 1990; Hilborn and Walters, 1992; Schwartzlose *et al*., 1999; Barange, 2001).

Figure 4: Anchovy: fishing mortality rate, F, for the age class interval 0-3, estimated by VPA from 1976 to 2001 (split year data, see text). The average for the whole period is also reported.
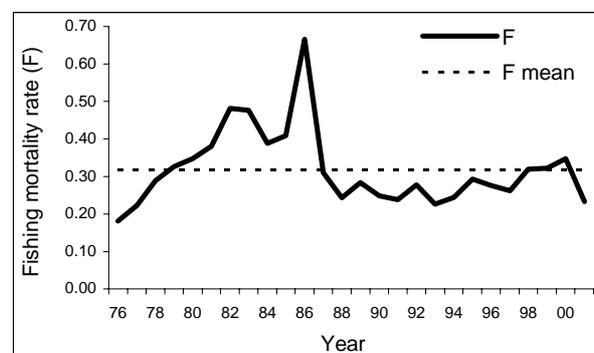
The annual values of the anchovy exploitation rate F/Z are shown in Figure 5, along with the corresponding average for the whole period 1976-2001. This is estimated to be equal to 0.34, which is under the threshold 0.4. However, some annual values are close (in 1984 and 1985) or higher (in 1982, 1983, 1986) than the value 0.4. In 1986, F/Z is even higher than the "very dangerous" limit 0.5; like for fishing mortality rates, this occurs just immediately before the collapse. As said in Introduction, the analysis performed by Patterson (1992) was based on variations of stock biomass within periods between 5 and 10 years. That is not necessarily detrimental to the hypothesized negative influence on the Adriatic anchovy due to fishing pressure in those years. In fact, as stressed by the same author, a high level of exploitation, even if occurring within a shorter time interval, can contribute to the decline of stocks. In the present case, warning should be detectable at least three times (= annual estimates of F/Z) before collapse. After this event, instead, the values are always lower than 0.4: that is consistent with the contemporaneous stock recovery. Finally, the exploitation rates calculated using weighted values of F_{0-3} showed an average, on the whole period, equal to 0.24, against 0.34 obtained when using the unweighted values of F_{0-3}. In the former case, the values decreased because of the increased weight of the younger age classes which displayed lower rates of fishing mortality. Anyway, the estimate of F/Z obtained for 1986 remained essentially unchanged: 0.51 against 0.53 obtained without weighting. In conclusion, the weighting procedure increased the difference between F/Z in 1986 – as well as in 1982 with 0.39 against 0.44 – and F/Z in all the other years.

Figure 5: Anchovy: exploitation rate, F/(F+M), from 1976 to 2001 (split year data, see text). The average for the whole period is reported along with the threshold, 0.4, suggested by Patterson (1992).
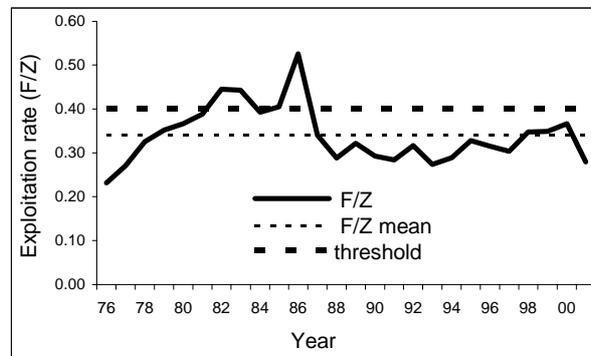
The annual values of mid-year stock biomass at sea of sardine are shown in Figure 6, along with the corresponding biomass of spawners and annual total catches. These represent, on the average, the 14% of mid-year stock biomass. The trends of biomass at sea increase since 1975 up to 1984, then a quasi continuous decline is observed till 2000, in correspondence with the lowest abundance levels of the series. The increase shown by the stock biomass in 2001 – not sustained by an increased spawning biomass as well – is likely an overestimate yielded by VPA due the same reasons mentioned for anchovy.

Figure 6: Sardine: total annual catch, stock biomass and spawning biomass calculated in the middle of the year by VPA, from 1975 to 2001.
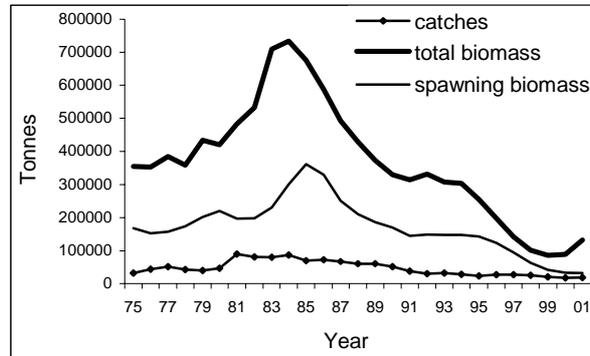
The comparison between abundance trends from VPA and the echo-surveys carried out by Azzali *et al.* (2002 a), shown above for anchovy, was also done for sardine (Figure 7). Analogous considerations can be made: an overall agreement is observed but the increase at the beginning of the 1980s and the decrease in the middle of the same decade are clearly smoothed in the VPA trend. It is worth mentioning that the biomass density in the series updated to 2001, reported by Azzali *et al.* (2002 b), is still declining just like in the VPA estimates.

Figure 7: Sardine: mid-year stock biomass estimated by VPA and biomass density obtained by echo-surveys, are compared over years.
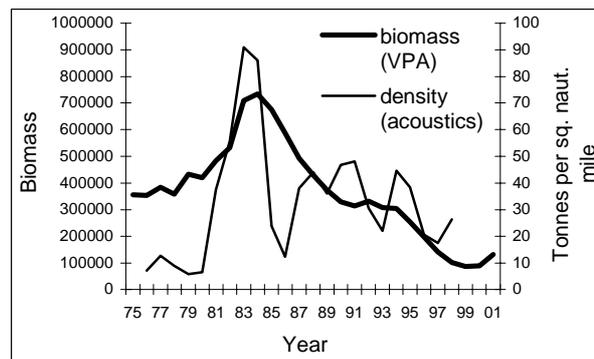
Figure 8 shows the estimated annual values of F_{0-5} obtained for sardine. It is evident that the relatively high values observed for anchovy in some years are not recorded for sardine.

Figure 8: Sardine: fishing mortality rate, F, for the age class interval 0-5, estimated by VPA from 1975 to 2001. The average for the whole period is also reported.
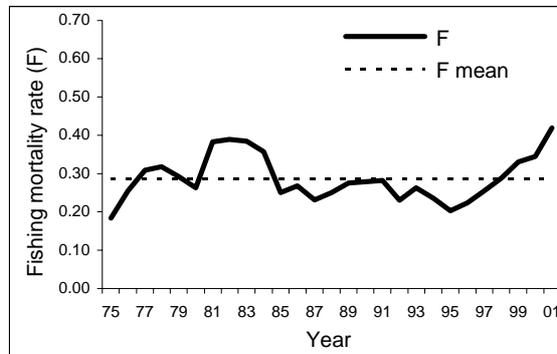
The annual values of the sardine exploitation rate F/Z are shown in Figure 9, along with the corresponding average for the whole period 1975-2001. This is estimated to be equal to 0.36, again under the threshold 0.4 as in the case of anchovy. Some annual values of F/Z higher than 0.4 are also observed for sardine as, for example, in the period 1981-1984 and, after the mentioned quasi continuous decline of the stock, in 2000-2001. Such a decline begins just in the middle of the 1980s. However, it is so pronounced and long over time to be unlikely attributable only to few values of F/Z slightly higher than the threshold 0.4. In fact, the level of estimated biomass is high in comparison with the catches both before and after the decline, with the exception of the most recent years. Likely, as (if not more than) in the case of anchovy, other factors than fishery influenced the population dynamic of sardine in this period. Anyway, it should not be ignored that, in 1981-1984, the catches were particularly high (around 80 000 and 90 000 tonnes) and relatively not far from the level of spawning biomass, i.e. around 40%. This accounts for a high value of F in the higher age classes and, thus, of unweighted F/Z in these years. The warning, on the contrary, seems to be more reliable for the year 2000 and, even if with more caution as it is the final year in the VPA run, for 2001. The overcoming of the threshold is here associated to the lowest estimated biomass, observed after a long time decline, which also appears in the echo-survey assessment. Moreover, some difficulties in obtaining economically satisfactory catches by fishermen were perceived since the spring of 2001 up to current months of the beginning of 2004. That occurred particularly in Croatia (G. Sinovcic, pers. comm.), where sardine is more requested by the market than in Italy, but also in some Italian ports such as Chioggia and Cesenatico. Furthermore, the exploitation rates calculated using weighted values of F_{0-5} showed an average, on the whole period, equal to 0.18, against 0.36 obtained when using the unweighted values of F_{0-5}. The highest values in the series obtained by weighting ranged from 0.24 to 0.29 and corresponded to the period 1997-2000: that further stresses a more reliable warning for the most recent years than the first half of the 1980s.

Figure 9: Sardine: exploitation rate, F/(F+M), from 1975 to 2001. The average for the whole period is reported along with the threshold, 0.4, suggested by Patterson (1992).
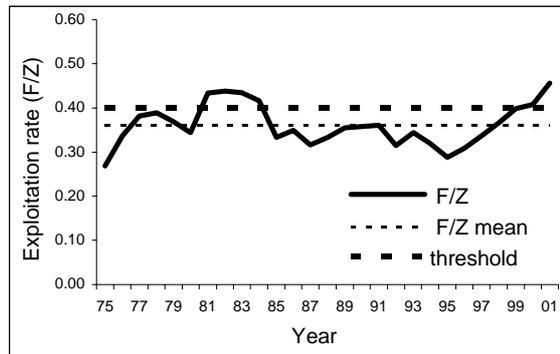
The use of F/Z as a biological reference point has a disadvantage: it requires a value of M, a parameter difficult to estimate (Hilborn and Walters, 1992; Haddon, 2001). The higher the value of M the lower the exploitation rate. However, the changes of M do not necessarily imply drastically different scenarios for this potential reference point. For example, in the case of anchovy stock, also according to a precautionary approach, M = 0.6 was used. When the VPA calculations were repeated with the alternative value M = 0.8, the exploitation rates resulted to be equal to 0.41 in 1986 and ranging from 0.3 to 0.4 in the four years before. Therefore, F/Z values around the critical threshold 0.4 were obtained, suggesting again, even if in a less strong way, a situation of risk.

The reason that could make the critical threshold of F/Z preferable to a critical value of spawning biomass at sea, is due to the fact that it is not always possible to work on enough long time series to estimate the threshold under which collapse is probable. An example of empirical stock-recruitment relationship, obtained for the Adriatic anchovy on the basis of the same VPA run which yielded F/Z, is reported in Figure 10 (the analogous graphic for sardine is shown in Figure 11). Here, the number of recruits in the year n+1 is plotted as a function of the spawning biomass in the previous year n. The abundance of spawners in the period 1985-1986, associated to particularly low recruitment in 1986-1987, shows values lower than 60 000 tonnes. It is not a case that a value of Minimum Biological Acceptable Level (MBAL) equal to 60 000 tonnes was pointed for this stock in a previous assessment discussed in a scientific report to the Commission of European Communities (Cingolani *et al.*, 1998). To be close or under this critical threshold does not necessarily imply collapse: the spawning biomass relative to the year 1994 was similar to that recorded in 1985, but yielded a quite higher number of recruits in 1995. Anyway, on the basis of this kind of plot and the experienced crisis really occurred, this threshold was thought to be a warning of risk. A value of MBAL around 60 000 tonnes was also obtained independently of the knowledge gained from the experienced crisis. Such a value was the estimated spawning biomass associated to the half of theoretical maximum recruitment level, which was derived by fitting the Beverton-Holt model to the empirical distribution (Myers *et al.*, 1994). Now, an interesting exercise should be done, looking at Figure 10 excluding the data points in the down left-hand corner, or in the top right-hand corner as well: without the experienced collapse, in order to estimate MBAL, methods like that based on the half of theoretical maximum recruitment should be applied. That would yield different estimates of MBAL in comparison with all the data points being used. In the former case, the threshold of spawning biomass would be either too much or not enough conservative than the latter one. Independently from collapse occurred, plots with missing

information are expected in situations with relatively short time series. It is easy to understand the usefulness of F/Z in such situations, as it does not require long time series.

Figure 10: Anchovy: empirical relationship between the abundance at sea of spawners in the year n and recruits in the year n+1 (the latter being reported near the data points), on the basis of split year (see text) VPA estimates.
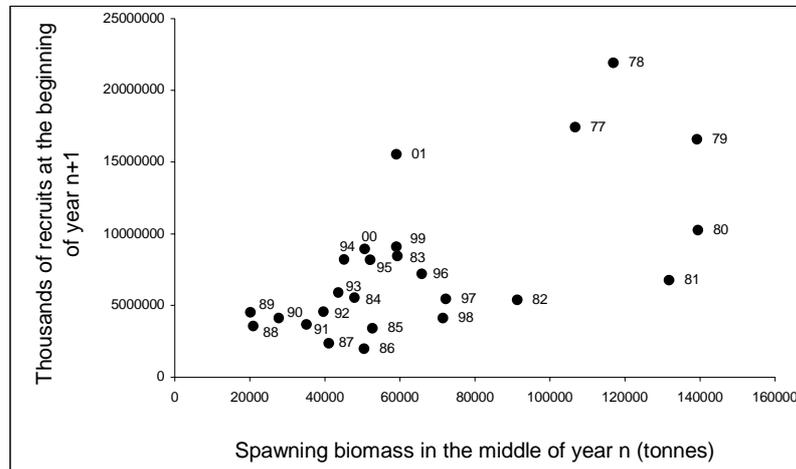
Figure 11: Sardine: empirical relationship between the abundance at sea of spawners in the year n and recruits in the year n+1 (the latter being reported near the data points), on the basis of VPA estimates.
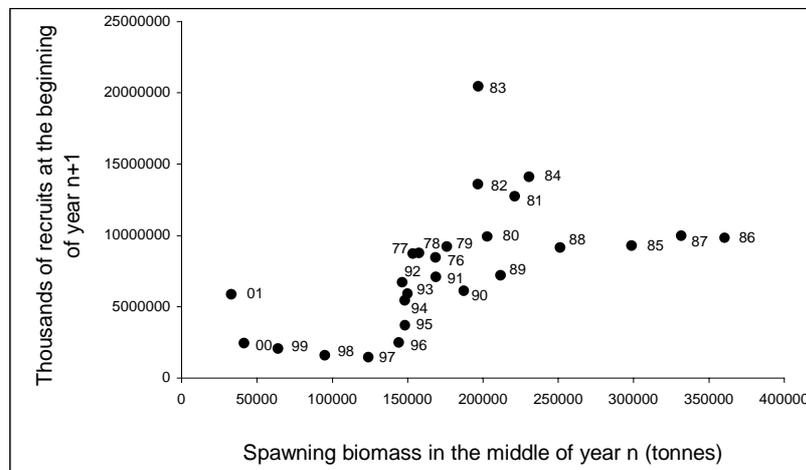
Conclusions

On the whole, the results of the analysis of exploitation rates F/Z seemed to be consistent with the conclusions derived by Patterson (1992). The use of the critical threshold F/Z = 0.4 in the management of anchovy and sardine stocks is therefore encouraged. Two other potential reference values are 0.3 and 0.5, so that a grid of values of F/Z could also be employed. This type of biological reference point could be added to the estimated spawning stock biomass to be maintained at sea. In particular, it could even substitute MBAL when not sufficiently long time series of stock-recruitment data are available to obtain reliable estimates of MBAL.

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The workshop stressed that the concept of reference points (RPs) needs to be closely connected to the management objective of concerned fisheries, taking into account various parameters such as maximum sustainable yield, economic and social benchmarks and environment concern. Participants stressed the need to define different RPs for each dimension of management or to identify a single RP aimed at ensuring the sustainability of the system, thereby no longer limited to biological aspects. A review of possible approaches to setting RPs and indicators for Black Sea fisheries emphasized the dynamic nature of recent ecosystem changes. The traffic light approach was illustrated as a means of following dynamic changes and gaining a broad perspective on events at the ecosystem level. The workshop outlined the main RPs obtained by applying the bio-economic optimization models of effort and effort-catch as derived from the theory of population dynamics. For the Mediterranean Sea, it was recommended to adopt a multicriteria approach based on bio-economic optimization models which, by introducing a number of constraints, would allow the definition of the actual state of the system as well as specific management targets. The use of some indicators derived from bottom trawl surveys carried out in the Adriatic Sea highlighted the potential suitability of selected indicators of their estimators. In the case of large pelagic species, it was confirmed that time series of both longline catch per unit effort and recruitment indexes can be considered as RPs for these species. The use of a biological RP based on exploitation rate threshold was appraised for the stocks of anchovy and sardine in the central and northern Adriatic Sea and the workshop pointed out that it could be used to prevent stock collapse along with the minimum biological acceptable level based on spawning stock biomass.