GENERAL FISHERIES COMMISSION FOR THE MEDITERRANEAN

COMMISSION GÉNÉRALE DES PÊCHES POUR LA MÉDITERRANÉE

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GENERAL REVIEW ON THE AVAILABLE METHODS FOR STOCK ASSESSMENT OF ELASMOBRANCHS, ESPECIALLY IN DATA SHORTAGE SITUATIONS
(by A. Abella)

SAC WORKSHOP ON STOCK ASSESSMENT OF SELECTED SPECIES OF ELASMOBRANCHS IN THE GFCM AREA

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1. Introduction

The main goal of fishery managers is to exploit resources maintaining healthy fish populations and a healthy fishing industry. As a consequence, the main purpose of stock assessment should be to provide advice on the best way aquatic living resources can be optimally exploited. Such goal is expected to be obtained through the definition of catch quotas, restrictions in size of the individuals caught, characteristics of gears used, limitations of fishing seasons, areas, etc. Decision makers have to determine which measures are more suitable and cost-effective in order to reach their objectives. Hilborn and Walters (1992) state that "Stock assessment involves the use of various statistical and mathematical calculations to make quantitative predictions about the reactions of fish populations to alternative management choices." There are two important issues explicitly included in such definition, namely the need of quantitative predictions and the provision of advice among possible choices.

In order to address such issues, some Reference Points have to be defined. Such *RP*s can be expressed as levels of total biomass, fishing mortality rate or other measurable characteristic of the population or the fishery. Reference points can be a target for management or a limit the fishery will not be allowed to go beyond. The more widely used Biological RPs are the Maximum Sustainable Yield (*MSY*) and the level of biomass *B* or fishing mortality *F* at which the population can produce the maximum sustainable yield (B_{MSY} and F_{MSY}). The definition of the values of these reference points require the availability of a minimal amount of information, that regards both biological and fisheries aspects. The more important information regards estimates of current stock status and reference values for biomass and/or exploitation rates.

While it is possible to target fishing activities and to fix catch, sizes or effort limits on a particular species by an appropriate choice of fishing ground, season and fishing gear, fishing activity most of the times removes individuals of several species in the same fishing trip. By-catches of not selective gears are in general important. In some cases, targeting of a number of species is the only way for making profitable a fishing trip.

The management issues described above also apply for elasmobranch species, which in the Mediterranean and Black Sea are most of the times susceptible to be captured as by-catch, especially in the fisheries using longlines, using nets targeting tuna and swordfish, in midwater trawling as well as in the mixed demersal fisheries.

Elasmobranchs are among the most vulnerable species to overfishing, due to their biological characteristics, especially because they show very slow recovery rates after depletion. In fact, elasmobranchs life cycles are characterised by a relatively low fecundity, slow growth to large body sizes, and dependence of self-renewal on survival rates of a limited number of well developed offspring. Sharks rays and skates have been in general classified as "*K*" life history strategists, with very low intrinsic rates of population increase. In such conditions, sustainable harvest is only possible at low fishing mortality rates. When they are taken in mixed fisheries together with other more productive species that constitute the target, some elasmobranchs may be extirpated while target species and the fishery remains viable. Although this life history traits apply for almost all the species, some variability between species can be observed. John Musick (1999) states that successful management of elasmobranches fisheries may occur, as the case of some small-sized species as *Mustelus antarcticus* that mature at relatively earlier age and producing a relatively large offspring.

2. Assessment of exploitation status of elasmobranchs

2.1. Data needs

Stock assessment needs of information on both the fish population and the fishery. In the case of elasmobranch species, most of the times such information is lacking or is partial. Due to their relatively low commercial interest and sometimes low number of specimens caught, the species-specific identification and recording of landings is considered of low priority and hence such information is often imprecise or incomplete. The attention that scientists pay to species belonging to this group is not too high, and hence the knowledge of the biological features of many species is poor or completely absent.

Statistics often report elasmobranch catches without distinction among species and without any mention on the amount of discards by species. Most of the catch of elasmobranch species is gathered in commercial categories or simply as by-catch without any mention in the official statistics.

The mentioned facts make difficult or impossible the assessment of the impact of the fishing activity on the stocks, and often overexploitation is perceived only after strong declines in abundance or in some extreme cases when local disappearance of some species has already occurred.

In recent times, the incidental catch of elasmobranchs by commercial fisheries is object of a major attention also considering that many of them are considered heavily exploited or endangered. Local extinction has been well documented for several elasmobranch species as *Raja batis* in the Irish Sea, *Pristis* and other elasmobranch species over the world.

In recent years, also in the Mediterranean sea scientists and managers have paid a major attention on the elasmobranchs stocks exploitation status and on its management. Thanks to several efforts of gathering of such information, the situation is improved in recent times. Only in the case of species which have some commercial value in the area in some countries as *Scyliorhinus canicula*, *Galeus melastomus*, , *Squalus acanthias*, *Mustelus spp*, *Raja miraletus*, *Raja asterias* and *Raja clavata,* information on landings is available. Moreover, abundance indices can be also available for the species that are vulnerable to bottom trawl nets and for whom catches are recorded during the conduction of scientific cruises. However, such time series are in general not long enough in order to allow detecting any clear trend.

The choice of the most suitable methods of stock assessment is not linked to a species or taxonomic group but rather depends on the availability of the data needed for performing any analysis. There are many methods for stock assessment, some of them more powerful and robust, but most of the times we are obliged to choose among those that are feasible with the available data. We have to keep in mind that as quality data decrease, assessments will be less reliable.

Fisheries managers should make more efforts to define cost-effective data collection systems. When data quality is not completely ensured and a big amount of uncertainty still remains, is advisable to use of all the possible approaches that can be applied with available data and successively to compare the results, searching for agreements in the conclusions, inconsistencies and patterns.

2.1.1. Biological information

Species identification

The reduced commercial importance of elasmobranches make them seldom a target species. This fact resulted in a low priority for data collection and research almost elsewhere in the Mediterranean sea. In consequence, shark fisheries are in general data deficient making stock assessments a difficult task. Despite the commercial importance of this group, elasmobranchs play an important role in the ecosystem and urgently require further scientific studies.

As a first step, it is necessary a better knowledge of the species that are present in the area, in terms of taxonomic lists and biological features. The production of guides for the filed identification is expected to improve the quality of the elasmobranchs catch reporting in the Mediterranean widespread and varied fisheries. Considering that most of the elasmobranchs have no high commercial importance and also that most of the species are caught in a fairly reduced number, the possibility of doing a fast and reliable identification of the species caught is essential.

A detailed catch information is vital for the successful management of these species.
The field guides are designed in order to assist commercial fishers in the identification of sharks and rays potentially encountered in the landings or on board during the fishing operations. Sharks and rays some times are not retained for commercial purposes and the work done of recording of discards on board in such circumstances is absolutely necessary. Catch and landings reporting require that any elasmobranch be correctly identified and recorded to the species or genus level. The guides must contain simple, user's friendly keys that highlight certain external distinguishing features for identification purposes. They should be supported by detailed species information and illustrations so that identification can be made with confidence. An example of field guides is the recent FAO product :

FIELD IDENTIFICATION GUIDE TO THE SHARKS AND RAYS
OF THE MEDITERRANEAN AND BLACK SEA
By F. Serena   Environmental Protection Agency – Tuscany Region   Livorno, Italy

Growth

A precise age information is needed for obtaining estimates of growth and other vital rates such as natural mortality and longevity which are of fundamental importance in fisheries research. The direct age determination in finfish and in particular on elasmobranchs is based on the examination of hard anatomical parts. Chondrychtians have no otoliths nor scales showing clear annuli that can be analysed for age reading and hence vertebrae or in some cases spines (modified dermal denticles) have been traditionally used for such goal. In some cases, however, interpretation of the annuli observed in vertebrae or spines is very difficult and hence, it is necessary to find alternative ways that allows the modelization of individual growth.

Length frequency analysis techniques, through the analysis of the progression of modes in the samples has been used in some cases. Unfortunately, in many cases, spawning season protracts for several months or more than one peak may be observed, and such characteristics make difficult the identification and getting track of the modal progression. Moreover, sexual dimorphism most of the times obliges to perform such analyses separately by sex.

Tagging fish is an alternative for studying growth, because each fish is measured when tagged and also when recaptured,. When tagging consist in injections of chemical markers, the number of growth rings that we can count in vertebrae or other hard structures between tagging and recapture may also permit ageing.

Growth assuming von Bertalanffy (1938) equation

When difficulties are so high and no method allows the estimation of growth parameters, some empirical relationships between von Bertalanffy's growth parameters allows defining a set of growth parameters by analogy that may be considered suitable for defining with some acceptable approximation the species' growth performance. For instance, it has been observed (i.e. Beverton &Holt, 1957, Pauly, 1988) that K declines when L_∞ increases, and vice versa . In general, fast growth (high K) is associated with small maximum size (low L_∞). This phenomenon, documented for many fishes allows the inference of a value of K even in the case only L_∞ is known. For each species, a parameter that related K and L_∞, th so- called Φ(phi prime) of Pauly (1988) can be defined through the following empirical equation:

$\Phi = \log K + 2*\log L_\infty$.

The maximum size reported for a species (L_{max}) can be used as a proxy for L_∞ and a first estimate of K can be obtained using the mean value of the parameter Φ in a given group of species that are the same or phylogenetically close to the species in question (Venema *et al.*, 1988; Pauly, 1998,2002).
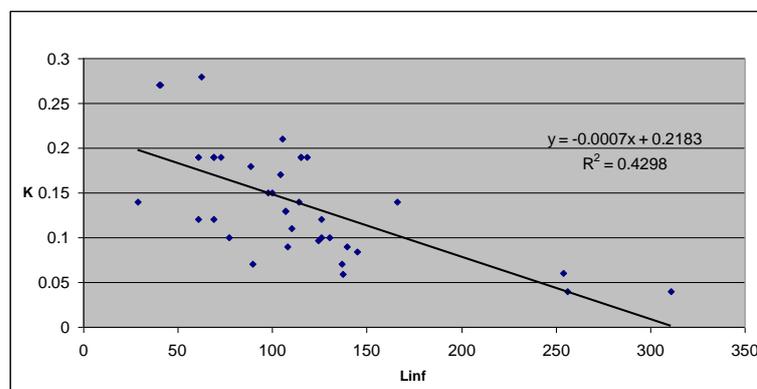
Relationship between L_∞ and K for 3 stocks of Rajidae species.
A mean value of Φ' of 3.12 has been obtained.

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An approximate value for L_∞ can be estimated using the available information of the total length of the bigger individuals ever observed.

K can also be estimated (if the value of L_∞ is known or assumed) inserting in the following equation the mean size L_A that the species reach in two successive relative ages (A1 and A2)

$$K = \frac{1}{A2 - A1} Ln \frac{L_\infty - L_{A1}}{L_\infty - L_{A2}}$$
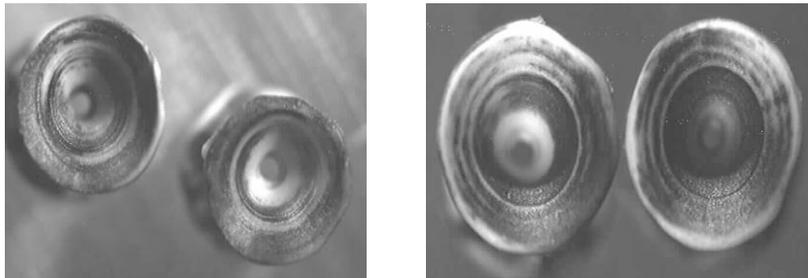
Annual rings in vertebrae of Raja asterias

Mortality
The quantification of mortality rates is of vital importance for understanding the dynamics of elasmobranch populations. There are however some difficulties for such estimation. The main reason is linked to the lacking of data. The traditional models as the catch curves and retrospective analyses as VPA are seldom utilizable.

Since assessment of status of the populations is an important issue, there were developed some indirect methods which are often used in the assessment of mortality rates in sharks and other elasmobranchs. The mentioned techniques utilize the observed relationships between some life history parameters and mortality (in particular natural mortality). These relationships have been described through empirical equations derived from the available information of well studies species. Such equations, however are most of the times based on observations regarding teleost fishes. In other cases, such relationships apply for several and different taxonomic groups.

Natural mortality
The methods have been developed to estimate natural mortality. Most of these indirect methods assumes that mortality is constant, independently of age, but there are other approaches that supply different values which are dependent to the age.

One of the more widely utilized empirical equation, has been proposed from Daniel Pauly (1981) using estimates of natural mortality of more than 100 stocks over the world.

The Pauly's well known equation is as follows:

$Log_{10} M = 0.0066 - 0.279 \log_{10} L_\infty + 0.6543 \log_{10} K + 0.4634 \log_{10} T$

where L_∞ and K are parameters of von Bertalanffy equation and T the average water temperature close to the bottom.

Jensen (1996) used the Beverton and Holt life history invariants (Charnov, 1993) for determining the relationships between life history parameters and natural mortality. He defined optimal trade-offs between reproduction and survival and obtained the following relationship :

M=1.65/Xm   where *xm* is the age at maturity.

Hoenig (1983), proposed a relationship based on available data on mollusks, teleosts and cetaceans:

ln Z = 1.44 - 0.982 ln $t$max

Gunderson & Dygert proposed an empirical equation that links mortality rates with gonad-somatic index assuming that the reproductive effort is a predictor of natural mortality rate.

$$M = 0.03 + 1.68 * IGS$$

Several authors have proposed methods for defining vectors of natural mortality rates by age. As examples, Caddy, 1991; Abella et al 1997) suggested a model for *M* which value is inverse with age. Chen and Watanabe (1989) proposed another method assuming natural mortality with age showing a decreasing shape for M at age for the former age classes followed by a dramatic increase of M in very old individuals.

Under certain circumstances, direct methods can be used for an estimate of total mortality, that coincides with M in the case the stock is not commercially exploited. An example is the use of the catch curve (Chapman and Robson, 1960). For the use of such approach, is necessary to assume constant natural mortality rates for all the age classes included in the analysis.

In the case a series of couples of data of fishing effort and total mortality are available, through a linear regression analysis, a rough estimate of *M* can be obtained as the value of the intercept with the Y axis at fishing effort =0. It is necessary to assume a fast (almost instantaneous) response of Z by changing the fishing effort, which is unlikely.

Tagging experiments can also allow a rough estimate of natural mortality in limited areas.

Sometimes it is necessary to have an idea of the mortality rates at early stages after hatching. Even though predation can be considered of lower importance, also in earlier stages, in elasmobranchs, due to the relatively big size of new born individuals or their development inside egg cases, evidences suggest much more vulnerability to predation than expected (mostly by gastropods). Particular attention on this phenomenon is paid in the Jerry Hoff's (2009) research on skate nursery areas.

Fecundity

Elasmobranchs have different reproductive strategies. While some species are oviparous (mostly rays, but also some sharks), in many cases they show aplacental or placental viviparity. The number of eggs or new born individuals that a female produce is relatively small. The annual average offspring can be from one to about 300 (Compagno, 1990).

For oviparous species, the embryos mostly develop inside egg cases while viviparous offspring at hatching is completely developed, most of the times after a relatively long gestation.

For the performance of stock assessments based on life history traits, the estimation of fecundity is very important. The number of eggs or embryos can be counted by inspection of mature females. However, the estimation is very difficult, especially due to the need to

account for the frequent failures during the gestation process. Moreover, the breeding period can be long and hence, this instantaneous counting of eggs or embryos will necessarily produce an underestimation of the species fecundity.

It is well known that fecundity increases with size and age of individuals. The relationship between fecundity and size is in general linear, while fecundity at age relationship follows an asymptotic curve.
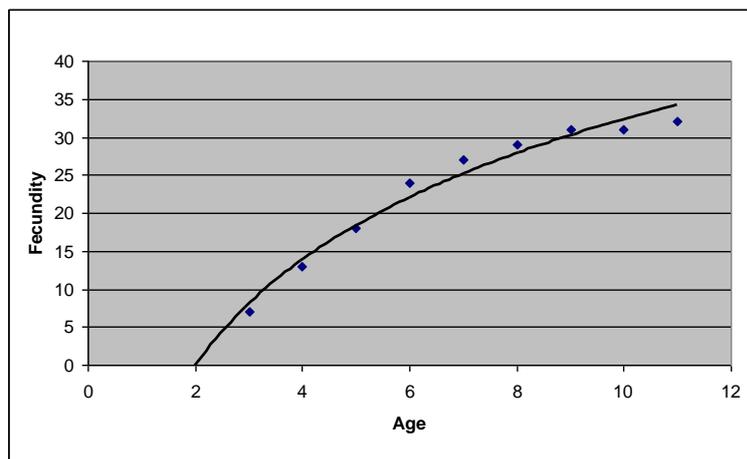
Example of relationship between number of pups per litter and age for female

Relationship Spawning Stock/Recruitment

The stock-recruitment (S/R) relationship is fundamental for the management of renewable resources. The nature of this relationship determine to what extent a population may be harvested by commercial fisheries.

Females produce a certain number of offspring in order to allow population to increase. The rate at which a stock may be harvested can be determined by the form of the S/R relationship.

The S/R relationship is presented graphically as a scatter plot with the number of the spawning stock in the abscissa and the number of recruits on the ordinate. The spawning stock is defined, as the number of organisms in the population of reproductive age and able to reproduce each year. The recruits are in general considered (and are quantified) estimating the juveniles that survive up to the size that are vulnerable by the fishery. The S/R relationship is normally dome-shaped. This means that the recruitment is expected to be zero or close to zero at very low spawning stock size (the relationship passes through the origin), there will be a maximum recruitment for certain size of the spawning stock followed by a decline or steady state situation for situations where there are too many mature adults. This last phenomenon can be explained by adults and immature fish competing for food or by the presence of cannibalism. There are several mathematical models used to describe the S/R relationship. The most popular is the Beverton and Holt model, which assumes a curve that at low levels of Spawning Stock recruitment increases, but with a successive reduction of the increase tending to an asymptotic value. The Ricker S/R model suggest a similar behaviour at low values of spawning stock, but with a decline at high abundance of spawners. Deriso and Schnute (Deriso, 1980, Schnute, 1985) have proposed a generalized model which become identical to the above described models depending on the value of one parameter. There have been seldom or never documented for elasmobranchs empirically derived stock-recruitment relationships. There are only few

attempts of definition of Stock/Recruitment relationships for elasmobranchs (i.e. Gedamke et al 2009). Bonfil (2005) state that the main difficulties are linked with the time delays caused by the longer reproductive cycles of sharks and their reproductive mode. Bonfil (op.cit) considers that such characteristics would cause a linear and direct stock-recruitment relationship.

In the case this relationship is realistic, it implies that shark populations will decline more sharply and with major difficulties for compensate harvesting compared with teleost fish.

Population growth

The changes in the size of a population is equal to the balance between births and deaths:

For computations, it is convenient to determine birth and death rates because this make possible to combine them and to produce a value that represent the net rate of population growth,

The product of such net rate of growth and the size of the population gives the value for the increase of the population per unit time.

The value used by population biologists to calculate the rate of increase of populations is the intrinsic rate of natural increase *r*. This rate can be defined as number of births minus the number of deaths per generation time.

Life tables are used for the study of the population growth. Some idea of its value can be derived from empirical equations that link this intrinsic rate of population increase with the adult body weight for each species. The population growth rates have been used to evaluate the effect that conservation efforts are expected to produce on protected or endangered species

Cortés (1998) proposed certain relationships that can be useful for the estimation by analogy or through the use of empirical equations of unknown parameters of some species.

The study was done by examining life history characteristics and correlations between traits related to body size, reproduction, age, and growth in sharks, that regard 230 stocks belonging to 164 species, 19 families, and 7 orders. Interspecifically, body size was found positively correlated with litter size and offspring size. Offspring size resulted negatively correlated with growth parameter K. The maximum size resulted negatively correlated with *K*. Parental size and size at maturity are positively correlated. The size at sexual maturity occurs in general at about 75% of the maximum size and females mature later and live longer than males. Age-at-maturity occurs at about half the maximum age in both sexes. Males are in general 10% smaller than females, but they reach its maximum length 34% faster than females on average. Maximum size and longevity are not significantly correlated in females, but are positively correlated in males.

There were also defined (Froese and Binohlam, 2000) other relationships, for example between age of maturity and lifespan for chondrichtyans:

$$Tm = 7.20 \times \ln(Tmax) - 12.68 \quad (n = 35, r2 = 0.57, p = 0.0001).$$

The same was done for the relationship between Natural Mortality rates and *K* of the vonBertalanffy's growth equation:

$$M = 0.42 \cdot k - 0.83 \quad (n = 30, r2 = 0.30, p = 0.002).$$

There is a software in excel file that can be free downloaded from FishBase web site: http://www.fishbase.org/download/

Stock units

The fisheries where elasmobranchs are involved must be monitored and regulated by international management bodies (i.e. GFCM), and the abundance of each one of the species and the response to exploitation have to be assessed at the level of management units within these areas, that not necessarily have to coincide with the Geographic Sub-Areas defined, which boundaries are most of the times arbitrary. For management purposes, a stock unit can be circumscribed by the geographical extent to which the effects of fishing pressure of a particular fishery are recognizable within the overall area where the species live. In theory, the individuals living in this sub-area should share same biological characteristics, especially related to growth rates, and probability of death due to both natural and fishing origins.

Many times, the priority goal in the management of fisheries where elasmobranchs are involved is population conservation, trying to ensure conservation of habitats, biodiversity, to prevent loss of genetic diversity. In order to address such goals, information should enable to identify and manage stocks separately.

2.1.2. Fisheries

In the Mediterranean and Black Sea, elasmobranchs represent about a small fraction of the total landings (Bradai, 2010)

Elasmobranchs seldom are the target of the fisheries. Some exceptions are represented by the case of some artisanal fisheries using gillnets in the Adriatic Sea and Gulf of Gabès, which target are different species of *Mustelus, Carcharhinus* and *Rhinobatus.*

As elasmobranchs have a modest commercial value, they are most of the times a component of the by-catch in fisheries targeting most commercially valuable species.

Many elasmobranch species due to their almost inexistent commercial interest are discarded. In other cases as *Scyliorhinus canicula*, *Raja spp* and *Galeus melastomus* only the bigger individuals are landed. In the pelagic longline fisheries that operate in Mediterranean waters, the elasmobranch by-catch can be of a relative importance. The main species caught are *Prionace glauca*, *Isurus oxyrinchus, Lamna nasus*, *Alopias spp.*.

Elasmobranchs are also a by-catch of trammel nets, gill nets, bottom longlines

As the fishing pressure is in general quite high, a clear decline of elasmobranch species populations, which are particularly sensate to fishing pressure has occurred. In a study in the Gulf of Lions, Ivonne Aldebert (Aldebert, 1997) has shown the dramatic reduction in abundance of elasmobranches from the 1960s to the 90s.

High declines in abundance were observed for some species of major economic interest as *Mustelus , Scyliorhinus stellaris* and *Squalus blainvillei*. The reduction in abundance of sensitive species (that drove to a proper extinction of stocks of sharks and rays in some areas) has been documented elsewhere (Stevens et al 2000).

In the Northern Tyrrhenian Sea, anecdotic information suggest that sharks and rays in the 1950s were an important fraction of the catches. Particularly dramatic examples are represented by the disappearance of some fisheries that were directed to elasmobranch species as *Squalus acanthias* and *Mustelus mustelus* (Serena and Abella, 1999). Similar phenomena were reported in the Alboran Sea and Maltese waters (Aldebert, 1997; Stevens et al., 2000).

It is however very difficult to gather quantitative information for the major part of the fisheries, because the total lacking of recording of landings and discards of elasmobranchs, as well as on the information of gears used, fishing areas, etc.

The lack of good information of elasmobranch catches and landings, fishing effort exerted, etc which is mainly due to the above mentioned causes, creates several problems when

we intend an assessment of the exploitation or conservation status or when management actions have to be implemented in order to address a recovery plan. In many fisheries, the evolution of the abundance of elasmobranch species, due to its special sensitivity to fishing has been used as indicator of pressure of the overall fisheries activity.

The condition of very low intrinsic rates of increase does not necessarily means that sustainable harvesting is unfeasible for the species of such group. In the case sufficient information on the demography is available and an effective harvesting strategy is enforced, a successful management can be obtained. Simpfendorfer (1999) reports on a sustainable fishery in Western Australia for *Carcharhinus obscurus*, focused on catch limits of recruits and protection of all the other age classes.

3. Models for stock assessment

3.1. Methods

3.1.1. Stock assessment methodology

Few formal stock assessments of shark resources have been conducted in the Mediterranean and elsewhere because neither catch and effort data series, nor age structure of catch data are sufficiently long, enough detailed and precise. The estimates of biomass and exploitation rates are in general linked with the construction of a formal stock assessment model. In the case of elasmobranch stocks, a complete information is not available, and hence, alternative approaches for giving indication on the stock's status and for management advice are needed. Age-structured models, dynamic pool models, surplus production models, etc require different information and in consequence the choice of an assessment approach may be driven by the available data.

Mike Holden (1977) was one of the first scientists to address the problem of stock assessment of shark fisheries. From its studies, he concluded that the classic production models were not suitable for being used to elasmobranchs. Surplus-production models have been consequently ignored for a long time for elasmobranchs stock assessment. At this time, the main problem of such models was the fitting assuming equilibrium status. Such choices conducted to wrong assessments and lack of prestige of the mentioned approach. The second problem was the quality of the used data.

More recently, several scientists as Ludwig and Walters (1989), Punt (1991), have recognised that often non-equilibrium surplus production models perform better than detailed age-structured approaches. Analytical approaches are data demanding and many times, some phenomena are not well known or are difficult to modelising (i.e. stock/recruitment relationship, life cycles, growth, fecundity, etc.)

The criteria for the choice of models and approaches for stock assessment can be different. In some cases the approaches are selected because they are widely used in the area, in some other cases, because they are suitable considering their limited request of data, considering the general shortage of available information. For some approaches, there is however a very limited experience in its use for Chondrichtyans. Among the most widely models utilized elsewhere we found the variants of VPA as XSA, ADAPT, etc, as well as the group of statistic catch-at-age models. Such approaches, however show objective difficulties for their use in the area and for this particular group. Considering that there is a very rich literature regarding their characteristics, type and quality of data needed, reliability of results, assumptions, parameters at which the models are more sensible, we consider that it is not necessary to pay special attention on them, considering

that seldom such models will be utilized in a short term. It was considered to dedicate a major attention to those approaches that are potentially useful at the present situation of knowledge of the fisheries (fleets, gears, catches, demographic structure, etc) where elasmobranchs are involved and with the limited knowledge of the biological characteristics of the species (growth, mortality rates, stock/recruitment relationship, etc).

Dynamic Pool approaches

Yield per recruit and other so-called dynamic pool models allows to assess the expected changes in the stock and on yields derived from changes in fishing pressure (*F*) and fishing pattern (*t*$_c$). Traditional versions assume equilibrium, and estimate the state of the stock and the long term yields. The more traditional version of these models (Beverton & Holt, 1957) assume that recruitment is constant and is a discrete phenomenon, gear selection and recruitment showing a knife-edge shape, natural and fishing mortality rates constant from the age of recruitment (t$_r$) to the older age in the fishery, and fish behaves as a dynamic pool, with a complete mixing of individuals within the stock. This model has been widely used in the Mediterranean and in some cases for elasmobranch species. The model allows to estimate the biomass per recruit and yield per recruit that can be expected at different levels of fishing pressure, by changing or keeping unchanged the age of entry to the exploited phase (tc). While the biomass curve shows a decreasing shape starting from *F*=0, there is a level of *F* that maximize the *Y/R* value along the curve. This value is called by several authors *F*$_{MSY}$ (*F* at maximum sustainable yield) and by other authors F$_{max}$. The term "sustainable" means, in this case, that although higher values can be obtained by increasing effort, these higher yields are only temporary, because the *F*$_{MSY}$ value is the maximum attainable at a certain fishing pressure kept unchanged by several years. In order to avoid misunderstandings is advisable to use *F*$_{max}$ intead of *F*$_{MSY}$ and this last reference value will be only used when a stock recruitment relationship is included in a *Y/R* analysis.

By changing the age of entry, it is possible to obtain different curves with maxima at different values of *F*.

Growth overfishing occurs when the fishing pressure is so high and in consequence yields per recruit decrease, being yield a product of number of survivals at this level of effort and mean weight.

The traditional yield-per-recruit models do not consider the ability of the stock to sustain recruitment at any level of fishing pressure, so it is unable to detect if the current combination of *t*$_c$ and *F* will drive the stock to a situation of recruitment overfishing.

The model use the well known equation of exponential decay and the von Bertalanffy's growth model in weight:

$$Y/R = F * \exp(-M(tc-tr)) * W_\infty \left(\frac{1}{Z} - \frac{3(\exp(-K(tc-to)))}{Z+K} + \frac{(3(\exp(-K(tc-to))))^2}{Z+2K} - \frac{(\exp(-K(tc-to)))^3}{Z+3K} \right)$$

Biomass per recruit can be obtained with a similar equation by simply removing *F* from the initial part of the right term of the equation, because *Y/R=F(B/R)*.

Biomass dynamic models

Model description

The surplus production (or biomass dynamic) models assume that the current size of a population depends on some interacting forces (individual growth, recruitment, natural and fishing mortality). If we incorporate these phenomena into a single function, the net effect

of such contrasting forces at a particular stock size can be estimated. Stock size increases as a consequence of tissue growth as well as recruitment into the population of young fish which reach the fishing grounds and are large enough to be selected by the fishing gear. On the other hand, natural and fishing mortality produce an inverse effect, reducing the population size.

In the absence of fishing, the effects of recruitment and growth can be combined into a single term that is called production. If production is greater than loses due to natural mortality, the population will grow and in the opposite situation it will decline. Surplus production is the algebraic sum of recruitment, growth and natural mortality. The adjective "surplus" refers to the surplus of recruitment and growth over natural mortality (net production). It represents the increase of the population biomass in absence of fishing, or the amount of catch that can be taken while maintaining the biomass at a constant size.

Biomass dynamic models can be discrete or continuous (Schaefer, 1954; Pella and Tomlinson, 1969; Walters and Hilborn, 1976; Butterworth and Andrew, 1984) and traditionally they are based on a first-order differential equation governing the rate of change of stock biomass due to production. In absence of fishing, the changes in population rate of increase or decrease are only defined by the current populations size:

The differential form of the equation is:

$$dB_t/dt = rB_t - (r/K)B_t^2$$

where: B_t is Biomass at time t, r is the intrinsic rate of increase and K the maximum population size or carrying capacity.

If fishing mortality F_t is added, the model becomes:

$$dB_t/dt = (r-F_t)B_t - (r/K)B_t^2$$

The chages of production dB/dt as a function of stock biomass B, in the case of the Schaefer model utilized here, is assumed symmetrical around B_{MSY} (the biomass level that produces maximum sustainable yield). In fact, surplus production will be zero at a zero biomass and is maximized at a biomass of K/2.

While time series of catch at age data age are seldom available, size structured models are in general unfeasible. Biomass dynamic models are in such circumstances a valid alternative choice. Mainly due to data quality issues, production models are among the most frequently used approaches for the assessment of tuna fisheries. According to Prager, (2005) and Punt & Hilborn, (1996), Biomass dynamic models, often provide more precise and realistic assessments and their parameters are more easily understandable for management purposes that those derived from more "powerful" models.

The biomass dynamic models allows estimating the population growth parameters K and r, the catchability coefficient q as well as the reference points *MSY* (Maximum Sustainable Yield) and the associated effort or fishing mortality that generates *MSY* (F_{MSY} and f_{MSY}) given the respective biomass (B_{MSY}). The most important limits for its application regards the quality of data (specific fishing effort to the species in question) and enough contrasting data (catches and effort at sufficiently different levels of evolution of the fishery)

The ASPIC software (A Stock-Production model Incorporating Covariates) (Prager, 1994, 2005) implements a non-equilibrium, continuous-time, observation-error estimator for the production model (Schnute, 1977; Prager, 1994). The model is used to estimate r (the intrinsic rate of population growth), *MSY*, the ratios of current biomass or F to the biomass or F values at which *MSY* can be attained, and q (the catchability coefficient, the

proportion of total stock taken by one unit of fishing effort). The available software allows making yield forecasting and to derive precautionary target reference points facing the intrinsic uncertainty that characterises the analysed processes and the observation errors. New versions of Biomass dynamic models have been applied using Bayesian approaches. In this way, it is possible to incorporate experiences for instance on the structure of error into the modeling process.

Dynamic fitting using data from fisheries independent sources

There is a version of non-equilibrium surplus production model that can be constructed also using data of trawl surveys. It is very simple and provide estimates of two parameters (r and K of the logistic population growth model).

This model allows the definition of Z and F at the level of the Maximum Biological Production (MBP). MBP is the comprehensive maximum production derived from the harvesting and from the biomass losses due to natural mortality. Considering that we are not using Biomass but an index U, the approach will only allow calculating an index of the MBP. In any case, the Biological Production curve will have the same shape, although with proportional lower values, and its maximum will be localized at the same value of mortality rate.

The approach is based on the Caddy and Csirke (1983) variant of Surplus Production models that uses the instantaneous total mortality rate Z as a direct index of effort, and a catch rate as abundance index.

The used model is based on the modified Schaefer logistic equation (Walters and Hilborn,1976):

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

With the exclusive use of trawl surveys data, information on q and fishing effort f are not available, but this problem can be resolved by using the Baranov (1918) catch equation and modifying the above equation as follows:

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

The approach needs of an estimate of mean value of the instantaneous natural mortality rate M.

A computationally simple fitting procedure can be utilized for the estimation of the parameters using for example an Excel spreadsheet and the solver tool by minimizing the logs of the differences between observed and model-predicted quantities. The log likelihood value is minimized by changing the seed values of r and K.

With this variant of the Schaefer model, we need only to estimate K and r. Considering that through the minimization only one solution is possible, unreasonable results are unlikely. As soon as the values of K and r are available, it is possible to define F and Z corresponding to the Maximum Biological Production (F_{MBP} and Z_{MBP}). The approach also allows comparisons between the Biological Production at Z_{curr} with the BP at Z_{MBP}.

The confidence bounds for K and r can be estimated through the construction of a likelihood profile (Venzon & Moolgavkor, 1988; Schnute, 1989). In order to obtain the likelihood profile, the likelihood function is defined and the maximum likelihood estimated. Assuming log-normal error, the equation that has to be minimized is the following:

$$-LnL = (n-1)Ln\left[\sqrt{\frac{2\pi}{n}}\sum_{y=1}^{n}\left(LnI_y - Ln\hat{I}_y\right)^2\right] + \frac{n-1}{2}$$

Punt & Hilborn (1996) note that minimizing the following function:

$$n*Ln\left(\sqrt{X}\right) + \frac{n-1}{2}$$

is equivalent to minimize X alone, and in this case the value of the parameters r and K obtained through the maximum-likelihood estimates are identical to those estimated with a simple least squares method of the form:

$$SS = \sum_{y=1}^{n}\left(LnI_y - Ln\hat{I}_y\right)^2$$

The estimation of confidence bounds of the parameters is based on the observation that in this case, error shows a distribution that can be approximated by the $\chi2$ distribution with m degrees of freedom (Punt & Hilborn,1996). In consequence, within the confidence interval for K and r, (in this case at p=95% and 1 degree of freedom) there will be included all the values for which twice the difference between the negative of the log-likelihood and the negative of the log-likelihood corresponding to the maximum likelihood estimates is less than 3.841 (χ^2 1, 0.05).

The approach allows to estimate Z_{MBP} or F_{MSY} and an index of the *MBP* (Maximum Biological Production) or of *MSY*. The impossibility to estimate absolute values for *MBP or MSY,* as stated before, is due to the fact that the input values of abundance index derived from trawl surveys are in general expressed in kg/km^2 and are not related to catches.
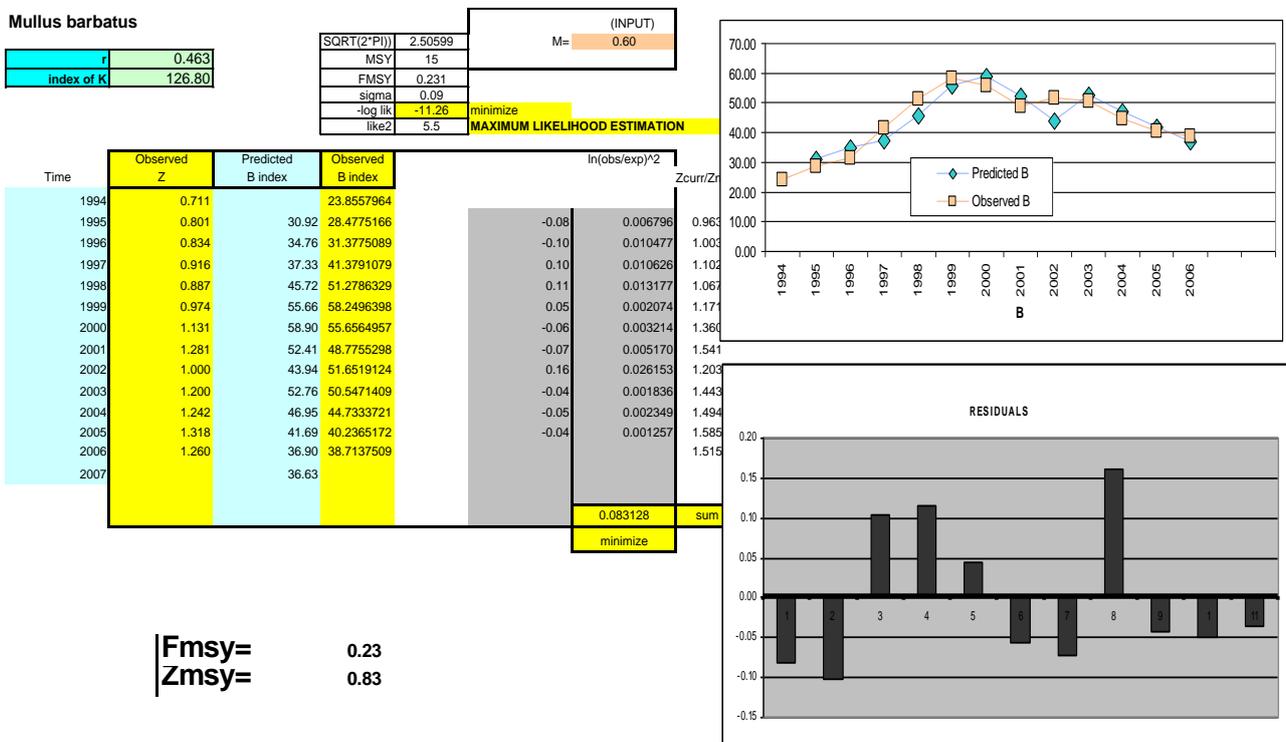
Example of output obtained with the Dynamic fitting using data from trawl surveys.

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Composite Model

Model description

Composite models (Munro, 1980) are surplus production models that use spatial information proceeding from different areas exploited at different rates, but for whom a similar productivity and evolution under different levels of fishing pressure is assumed. If the survey explores correctly the areas where the stock is present and allows reconstruct the demographic structure of the stock in the different areas, these data are suitable for performing the assessment with a Dynamic Biomass model.

The change from a time to space-based data set allows the utilization of production models even in the case long data series on catch and effort are not available. This is a particular merit of the approach and make it potentially suitable for its use in the case of elasmobranchs. Hilborn & Walters (1996) state that *the one type of equilibrium analysis that appears to be useful on occasion involves spatial contrast in fishing effort.* One of the advantages of such approach is that the results obtained are not affected by most of the problems that characterize the traditional versions of surplus production models as changes in catchability along time that can be due to an enhancement of the efficiency of fishing techniques.

The mentioned approach was used for the definition of a sustainable level of fishing pressure for some elasmobranchs in the Mediterranean and for a preliminary assessment of the current status of exploitation in grounds on which they were exploited with different rates. Data may proceed exclusively from trawl surveys and in consequence no information on fishing effort is needed. This is a further advantage, considering the difficulties of quantification of effort and commercial catch rates of such species. Considering that *Z* includes both, removals of fishing activity and deaths due to natural causes, there is not need of quantification of the actual fishing effort directed to a certain species, especially when this stock is not the target of a fishery. Total mortality rate is used here as a direct index of fishing effort, assuming *M* constant. The model allows the estimation of the Maximum Biological Production (*MBP*) (Caddy & Csirke, 1983).

Results of a composite model suing data of couples of Z and biomass index kg/km^2 from different GSAs.
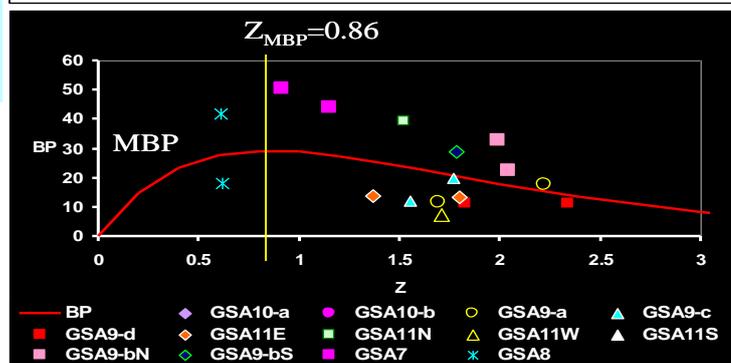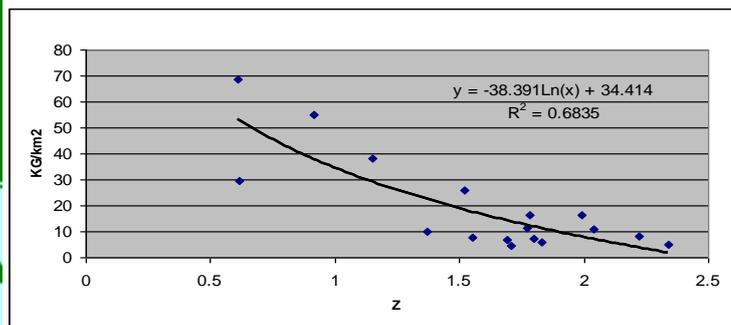
Bonfil (2005) states that surplus production models can and should be applied to elasmobranch fisheries because easy to implement, but their results should be considered as a preliminary assessment. It is advisable that a more complete and reliable assessment (i.e. age based) should be applied.

Delay-difference models

The delay-difference model of Deriso (1980) is a combination of production models with age structured models. The inclusion of some biological information is a clear enhancement of the reliability as regards the traditional surplus-production models where biological processes as recruitment and individual growth are only intrinsic, and that does not allow to modelising nor to predict the consequences of any change in exploitation pattern.

## Collie and Sissenwine model

The Collie-Sissenwine Analysis (CSA) model is a stage-based assessment model that estimates the abundance of two classes, defined as recruits and post-recruits. Recruits are assumed to become fully available for the fishery over the model's unit time step. The population consists of two stages: the recruits, and the fully recruited individuals. The time step is annual, with years defined on a calendar basis or as the time passed bye between regular surveys. The population dynamics is modelized by a discrete equation:

$$N_{t+1} = (N_t + R_t)e^{-M} - C_t e^{-M(1-\Upsilon)}$$

where N is the population in number at the start of year t; R_t the population in number of recruits at start of year t; C_t the catch in number during year t; M the instantaneous rate of natural mortality; Υ the fraction of the year when the catch occur, (i.e. $\Upsilon = 0$ when fishing season is positioned at the beginnings of year, or 0.5 when occur about the centre of the year or all along the year).

Estimating N_t and R_t for each year requires additional information on relative indices nt and rt of abundance for both stages, that can be derived from trawl surveys data, assumed proportional to actual population sizes.

The model requires indices of abundance for these two stages and estimates of the total number of animals removed via the fishery. The model provides estimates of both abundance and mortality rates. The user can flexibly specify alternative hypotheses regarding process error and observation error, including models that use process error only or observation error only. The underlying error structure for observation and process error can be specified as either normal and lognormal. If lognormal error is specified the user can choose whether or not to use bias correction methods. Bootstrapping can be based on either randomly resampling from the set of original error terms or by generating random errors from a parametric distribution with parameters derived from the original set of errors.

	Mortality Estimates		
Year	Total	Natural	Fishing
	Mortality	Mortality	Mortality
1982	0.267206	0.150000	0.117206
1983	0.267338	0.150000	0.117338
1984	0.296172	0.150000	0.146172
1985	0.286595	0.150000	0.136595
1986	0.295999	0.150000	0.145999
1987	0.280150	0.150000	0.130150
1988	0.295115	0.150000	0.145115
1989	0.302094	0.150000	0.152094
1990	0.397635	0.150000	0.247635
1991	0.345645	0.150000	0.195645
1992	0.306777	0.150000	0.156777
1993	0.321257	0.150000	0.171257
1994	0.392188	0.150000	0.242188
1995	0.353477	0.150000	0.203477
1996	0.365136	0.150000	0.215136
1997	0.358270	0.150000	0.208270

Some outputs of CSA

The Catch-Survey Analysis method of assessment (CSA) aims at estimating stock abundance from relative indices by filtering measurement error in the latter through a simple two-stage population dynamics model. The calculation engine is based on an adaptation of previous implementations of the model and by various specific versions developed by the Population Dynamics branch at NMFS Northeast Science Center, Woods Hole, MA.

A sensitivity analysis (Mesnil, 2003) confirms that CSA is capable of providing reliable information about general stock trends. The absolute estimates however, are quite sensitive to the parameter relating survey catchabilities, which needs to be estimated or corrected externally. In fact, in its basic version, CSA assumes that catchability in the survey or fleet providing the indices has been constant over the study period. Biased results may be obtained if observed changes in catchability are not corrected in data input. CSA is in any case a very valuable method for supporting management advice when age data are lacking or uncertain.

Ref: NOAA Fisheries Toolbox. An Index Method (AIM) Version 2.1.

An Index Method

An Index Method (AIM) allows the user to fit a relationship between time series of relative stock abundance indices and catch data. Underlying the methodology is a linear model of population growth, which characterizes the population response to varying levels of fishing mortality. If the underlying model is valid, AIM can be used to estimate the level of relative fishing mortality at which the population is likely to be stable. The index methodology can used to construct reference points based on relative abundance indices and catches and to perform deterministic or stochastic projections to achieve a target stock size. This index-based approach was developed for more fully utilization of the data sets from the surveys and historical landings. The method is very simple, but is based on linear population models, modern graphical methods, and robust statistical models. From data in abundance and fishing mortality, deducible from a time series of catch and survey indices, relative fishing mortality rate can be defined as the ratio of catch to survey index. The AIM calculates two derived quantities Replacement Ratio and Relative *F*. The replacement ratio is an analytical tool for examining the historical behaviour of a population and any potential influence of removals due to fishing activities. The method has been applied to a number of stocks in the NAFO area. The replacement ratio draws from the ideas underlying the Sissenwine-Shepherd model, delay- difference models, life-history theory, Collie-Sissenwine model, and statistical smoothing.

The replacement ratio is defined as:
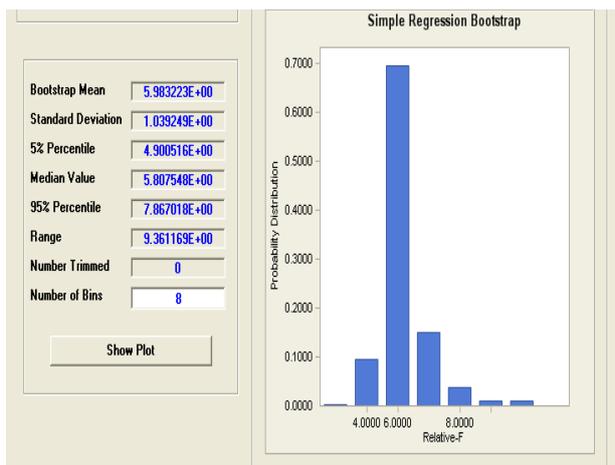
$$\Psi_{m,t} = \frac{I_{m,t}}{\sum_{j=1}^{J} I_{m,t-j} \Big/ J}$$

Relative F is defined as:

$$F_{REL\,m,t} = \frac{C_t}{\sum_{k=1}^{K} I_{m,t-k+1} \Big/ K}$$

In the above Equations:

$\Psi_{m,t}$ = Replacement Ratio for Index m at Time t

$F_{REL\,m,t}$ = Relative F for Index m at Time t

C_t = Catch at Time t

$I_{m,t}$ = Index of Abundance m at Time t

J = Number of years for smoothing Indices of Abundance

K = Number of Years for Smoothing Relative F

m = Index number
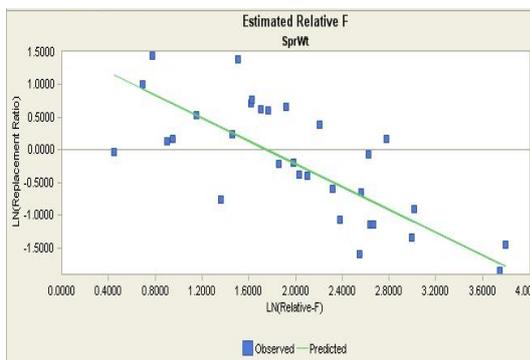
Some results of the NOAA toolbox AIM software

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The user must specify the following: Catch Time-Series; Index(es) of Abundance Time-Series; First and last Year in Data

The relationship of the replacement ratio (F_{rep}) as a proxy of F_{MSY} can be evaluated by comparing reported estimates of F_{rep} to estimates of FMSY from Non-equilibrium Production Models as ASPIC in the same units as the replacement ratio, catch/survey biomass. The theoretical basis of F_{rep} suggests that it may be a useful proxy for F_{MSY}, if the data used in its estimation come from a period when the stock was fluctuating around B_{MSY}. In a peer review of biological reference points for New England groundfish, Frep was proposed as a proxy for F_{MSY} for six stocks (NEFSC, 2002).

Ref: NOAA Fisheries Toolbox. An Index Method (AIM) Version 2.1.

3.1.2. The choice of the more suitable methods for elasmobranchs

The different surplus production models, those traditional that use commercial information on catch and effort and allows the definition of the MSY, the f_{MSY}, the F_{MSY}, or the other variants that allow the estimation of the value of Z that produces the Maximum Biological Production Z_{MBP} derived from trawl surveys are low data-demanding (only a time series of couples of Z and corresponding abundance indices are needed). When using the approaches for the estimation of F_{MBP}, they provide sustainable fishing mortality rates that are more precautionary than those derived through traditional models that defines a *MSY*.

Dynamic pool models, as Yield per recruit and spawning per recruit approaches, are another alternative potentially useful. They have been frequently used for a preliminary assessment of demersal stocks in the area, also in the case of elasmobranchs. They need of a modest amount of biological information (growth, natural mortality rates, age of maturity) which is available for some Chondrichtyan species in some areas of the Mediterranean basin. They allow to define some reference points easy to calculate as F_{max} and $F_{0.1}$, being the last one considered precautionary and a proxy of F_{MSY}.

As stated before, the yield per recruit approach is however less efficient for the analysis of the sustainability and without a stock/recruitment relationship it is not useful for the estimation of Maximum Sustainable Yield nor the level of F corresponding to *MSY*.

Approaches as the above described AIM and CSA can be good candidates because their relatively limited information required.

Through simulation it is possible to define a threshold level of F that reduces the spawning biomass to a certain percentage of its pristine biomass. They also allows to predict the % of the *MSY* fishing at this or at other rate. In the Mediterranean context, considering a general lacking of knowledge on the population dynamics and fisheries of many elasmobranch stocks, approaches like that, that are less information-demanding, have broad potential in fisheries such those of the mentioned area.

For the estimation of the surviving fraction of the spawning biomass at different levels of F as well as for many other purposes, we have to make however assumptions, many times not completely supported by scientific evidence, regarding the natural mortality. *Y/R* models are very sensible to changes in this input parameter and hence, results may be quite different when different values of M are introduced. In a comparative study, Caddy and Mahon (1995) found that *%SPR* is positively correlated with natural mortality and negatively correlated with some indices of size. In consequence, fast-growing short-living species should require higher percentages of the survival fraction of the spawning biomass to guarantee an adequate stock replacement.

3.2. Definition of the current exploitation status

3.2.1. Estimates of *F*

VPA and catch at age analyses
VPA variants can be used for the estimation of a vector of *F*-at- age, recruitment and numbers at sea by age using data on total annual catches by age and an estimate of natural mortality *M*. These age-structured models can be splitted into two groups: Virtual Population Analysis or VPA and the so-called statistical catch-at-age analysis.

When the time series are too short, at least the traditional versions of the mentioned methods are not feasible. In alternative, if some assumptions as equilibrium status is introduced, it is possible to use a variant that analyze the age structure of the catch in only one year assuming that the observed proportion of number by age has the same shape to the structure that should be observed by following the cohorts in time. The composition of this "pseudocohort" allows the analysis of the catch structure of a single year assuming it is representative of an equilibrium status. The length or age "pseudocohort" analysis provides in any case a vector of *F* and numbers at sea by age. The values of *F* have to be considered an approximation of the real ones, more close to the real ones as the stock is more close to an equilibrium status. It has been observed that in most of the cases the absolute values of *F* at age are in fairly close to those derived from a formal *VPA*. A version of cohort analysis using only one year of catch data was developed by Lleonart and Salat (1997). A version developed by IFREMER allows corrections in the variation in effort and recruitment (Laurec and Santarelli-Chaurand. 1986).

The more critical issue in the case of chondrichtyan stocks is the quality of data. Data have to represent the real catches at age/size of the stock and hence it is necessary to avoid any under-reporting or species misidentification that that may produce an unreliable reconstruction of the demographic structure of the catch and leading to wrong results.

Current *F* estimates from production models
Some available software for fitting Surplus Production models as ASPIC (NOAA, 2010) allows defining several parameters and variables, including values of Fishing Mortality rates and fishing effort per year as well as estimates of Biomass.

Results of ASPIC. Fishing mortality rate per year (left)
and evolution of B_{curr}/B_{MSY} and F_{curr}/F_{MSY} rates

Such values can be compared with the defined reference points F_{MSY}, f_{MSY}, B_{MSY} that are also produced as output of the software.

Catch curves

Catch curve analysis has been traditionally used to estimate the total mortality by observing the decline of individuals in a cohort. This decline is theoretically described with a exponential model. It is assumed that the population is closed to emigration and immigration. It is assumed that the population structure is only affected by the loss of individuals to mortality.

The surviving numbers for several age classes can be analysed contemporaneously with a catch curve using the natural logarithms of the numbers in each age class plotted against age. In this way a mean Z can be estimated for the age classes considered completely recruited to the fishery.

The instantaneous total mortality parameter Z is defined by the following equation

$$N_t = N_0\, e^{-Zt}$$

where N_t is the population size at time t and N_0 is the initial population size. The total mortality rate Z, can also be estimated from a length-converted catch curve (Pauly 1990; Gayanilo and Pauly 1997). Each size is transformed in relative age using the von Bertalanffy equation. The use of the catch curve has some limitations. In fact, for the use of this approach, is necessary to assume that both, natural and fishing mortality are constant over all the ages/sizes included in the analysis. This is a very strong assumption that is unlikely to apply for most of the exploited stocks, first at all because neither species nor fleets are distributed randomly in space and time and hence, it is likely that some ages will be more exposed to fishing pressure. Moreover, natural mortality is in general size-dependent and is likely that small-sized (younger) individuals are more exposed to higher M rates . In the case of elasmobranches species, that have a relative large size when they hatch and become vulnerable to the gear, it is likely that the assumption of constant M for all the age classes is a reasonable assumption and will have a limited influence in the results. Widely used software that allow estimating Z with different methods and manuals are those of Hoggarth D *et al*, 2006 and Sparre and Venema, 1986.

SURBA

SURBA (Needle 2003; Beare et al., 2005) is a VPA based model that assumes that F is separable into an age (s) and a year effect (f). SURBA perform such analysis by minimizing the squared sum of differences between observed and survey derived abundances at age correcting for catchability assuming a vector at age for *q*. Data used by SURBA derived from surveys are catch at size, maturity, weight. A natural mortality vector that can be a constant value or variable by age has to be used as input. Since abundance estimates of surveys furnish indices and not absolute values, the model only allows the estimation of relative population numbers. SURBA has been used in many Working Groups of Stock Assessment of ICES and in recent years also in Mediterranean Working groups .
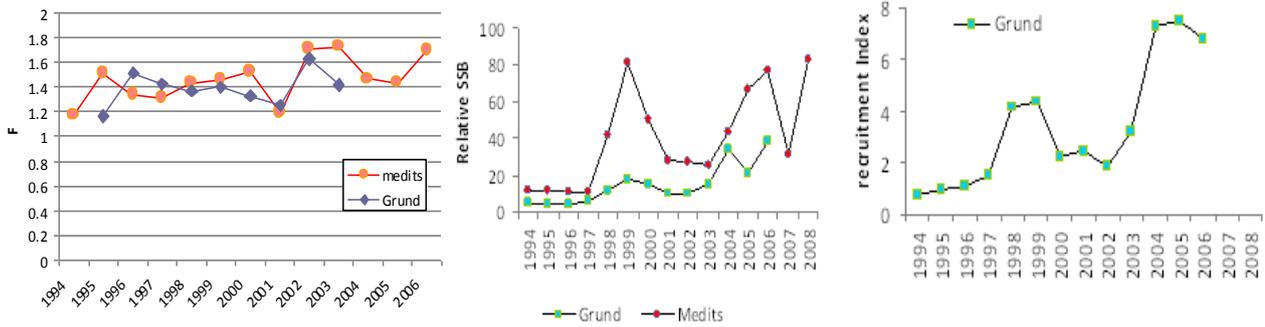
Some output of SURBA (F for each year, relative Spawning Stock Biomass, recruitment index).

Beverton & Holt equation

The Beverton & Holt mortality estimator have been in the past widely used, especially in situations of data shortage, because only requires the von Bertalanffy growth parameters (*K* and *L'*), an estimate of the length of full capture *L'* and a mean length of fish caught which are larger than *L'*.
The traditional equation is :

$$Z = K \frac{(L\infty - \bar{L})}{(L\infty - L')}$$

The Beverton and Holt mortality estimator however, is based on the equilibrium condition assumption that is hardly to hypothesize.

Variant of Beverton & Holt equation by Gedamke and Hoenig

Gedamke and Hoenig (2006) have proposed a variant of the approach, implemented in the SEINE software (Survival Estimation in non-equilibrium situations) which is included in the NOAA Stock Assessment Toolbox. It is a new procedure that allows a series of mortality rates to be estimated from mean length data representing non-equilibrium conditions in multiple years. The estimation of *Z* is done using weighted information of mean size of catch, size of full capture and growth parameters. With this approach, the transitional behaviour of the mean length statistic is derived for use in non-equilibrium conditions.
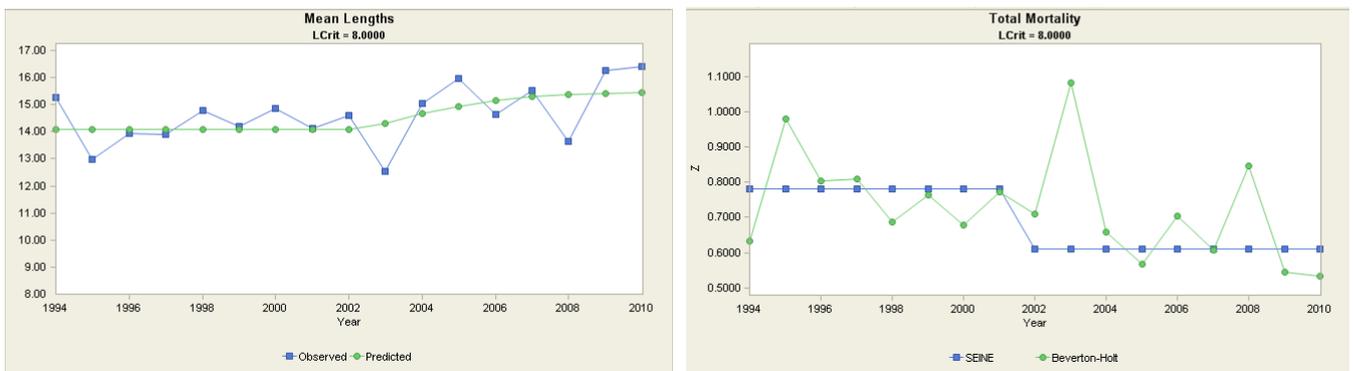
Fitting of mean lengths (left) and results of Z estimates (right) obtained with the traditional Beverton & Holt approach assuming equilibrium (green) and with SEINE (blue).

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The estimation of total mortality rates *Z* in non-equilibrium situations can be accomplished by selecting a year (or years) of change and the values of *Z1, Z2*, etc that cause predicted mean lengths from equation to best match a time series of estimated mean lengths. The method of maximum likelihood estimation is used.

Evolution of the approaches

A method that incorporates information on life-history, stock/recruitment considerations, and an estimate of depletion was developed by Brooks et al. (2010). The authors proposed some analytical solutions for biological reference points in terms of maximum lifetime reproductive rate. This rate can be derived from biological information on maturity, fecundity, and natural mortality. The analysis allows a re-parameterization of the stock–recruit relationships in terms of spawning potential ratio. The possibility of calculating reference points from biological data, or a meta-analysis, without a full assessment model nor the fitting a stock assessment model makes the method a suitable option for assessment in fisheries where data is poor.

The method estimates the Spawning Potential Ratio (*SPR*) at the point of Maximum Excess Recruitment (*MER*) which is analytically derived from the maximum lifetime reproductive rate (the product of the number of spawners produced by a recruit over its lifetime in the absence of fishing, known as the net reproductive rate in life tables or matrix population models) and the slope at the origin of the stock/recruitment curve (typically a Beverton & Holt curve), which in sharks is simply the first-year survival. The determination of the stock status (overfished or not) needs of an estimate of the current depletion. Such status can be estimated by dividing the current abundance index by the estimated abundance of the unexploited stock. The Reference Point is the *MER* depletion

4. Precautionary management

The application of precautionary approaches for fisheries is related to the needs of conservation, management and exploitation of the living resources and to the preservation of the aquatic environment (FAO, 1995). For the implementation of the precautionary approach, it is needed to improve decision-making for resources conservation and management. These decisions should be based on sound scientific evidence allowing the definition of stock specific reference points, as well as on actions to be taken if they are exceed.

Restrepo *et al* (1999) state that "In fisheries, the precautionary approach is about applying judicious and responsible fisheries management practices, based on sound scientific research and analysis, proactively (to avoid or reverse overexploitation) rather than reactively (once all doubt has been removed and the resource is severely overexploited), to ensure the sustainability of fishery resources and associated ecosystems for the benefit of future as well as current generations."

They also state that the design of research in support of precautionary management should provide accurate and complete data and analyses, to develop operational targets and constraints, to evaluate the likely consequences of management actions incorporating uncertainty into assessments and management. The elements of the precautionary approach that need of more attention are the definition of overfishing, the choice of the management actions to prevent overfishing and to promote stock rebuilding and the use of risk-averse approaches to calculate reference points, to impose certain levels of fishing pressure and for stock rebuilding.

A management procedure is defined by Kell *et al.*(2006) as a set of rules used to determine management actions in which the data, assessment methods, or rules used for

decision-making, and the harvest control rules for implementing management action are pre-specified.

Harvest control rules are protocols of fishing strategies useful for the preservation of the stock at a desired status, to be defined in accordance with the limit and target reference points.

When there is a great uncertainty regarding the maintenance of productivity of a stock it is necessary to be more cautious when setting catch or effort limits. This principle is seldom applied in fisheries in the world, and is almost inexistent in the Mediterranean. Under a precautionary point of view, for instance if we have defined a value for the Limit Reference Point Maximum Sustainable Yield, a Precautionary Reference Point value below *MSY* has to be defined, that will be lower as greater the uncertainty.

Many international agreements deal with the precautionary approach and the definition of minimum standards for defining management limits. Some of the more important international agreements are The Rio Declaration on Environment and Development 1992 stating that the precautionary approach shall be widely applied by States according to their capability. Lack of scientific evidence shall not be used as a reason for postponing measures to prevent environmental degradation. The 1995 FAO (FAO, 1995) for Responsible Fisheries says that States should determine, based on available scientific evidence, stock-specific reference points as well as the actions to be taken in the case they are exceeded

4.1. Reference points for Management of elasmobranch stocks

The United Nations Fisheries Agreement (1982)**,** regards the Conservation and Management of Straddling Fish Stocks and High Migratory Fish Stocks. It provided one of the first attempts of defining limit reference points as boundaries which are intended to constrain harvesting within safe biological limits within but trying to maximize sustainable yields. Precaution is clearly quoted in the following sentence "…management strategies shall ensure that the risk of exceeding limit reference points is very low".

Reference points are conventional values of the state of a fishery or a population that are considered the desirable objective to be reached (target reference points) or an undesirable state of the fishery which needs to be avoided (threshold or limit reference points). The definition of such limits is aimed at safeguarding the long-term productivity of a stock

Moreover, it is here stated that the fishing mortality rate *F* or Biomass corresponding to the Maximum Sustainable Yield (*MSY*) has to be regarded as "a minimum standard for limit reference points" and management strategies have to guarantee that *F* does not exceed F_{MSY} and that biomass "does not fall below a predefined threshold". In order to follow the single-stock-based objectives defined by the Johannesburg plan of implementation, fishing on all species should be done at rates equal or lower than the F_{MSY}. The modelling approaches that provide a basis for deriving reference points for preventing overfishing and for maximization of yields and revenues may use production models, dynamic pool and spawner-recruit models.

As defined before, Surplus production models allows deriving reference points f_{MSY} or F_{MSY} or their correspondents in Biomass.

The approaches as yield-per-recruit analyses (Beverton & Holt, 1957; Thompson and Bell, 1934) analyse the long term changes in yield as a function of fishing mortality and age of entry to the fishery. The levels of *F* associated to different levels of Spawning per recruit

(i.e F30%SSB) can be used to characterize recruitment overfishing thresholds. $F_{0.1}$ is one of the most popular RP based on Y/R approaches and represents the value of F at which the slope of the curve of yield-per-recruit is 10 % of its slope in the origin. Fishing at this rate is assumed (even though without any biological evidence) to produce lightly lower yields than the MSY and it considered by several authors as a proxy of F_{MSY}.

The third type of models is based on the relationships between spawning stock biomass and recruitment. The more widely used reference points derived from such relationship are certain levels of fishing mortality where the recruitment contribution to the stock balances the losses due to mortality. It is likely that fishing at that levels, there is a good chance for the stock to be sustained. Moreover, other threshold values as F_{low} and F_{high} are other levels of F at which recruitment is considered sufficient or insufficient to balance mortality. F_{crash} is defined as the fishing mortality corresponding to the slope at the origin of the stock-recruit model. Mace (2001, 2003) and Myers et al (1994) include as indicators of recruitment overfishing the stock size corresponding to the 50% of the maximum estimated average recruitment, the stock size corresponding to 20% of estimates of virgin stock size, values of fishing mortality rates ($F_{10\text{-}30\%}$) from Spawning per Recruit analyses and the concepts of $F_{extintion}$ or F_T used in different areas that are almost identical to F_{crash}.

Certain values of the exploitation rate ($E=F/Z$) have been proposed as candidates of reference points. Precautionary values of E=0.4 have been defined for sardines and other small pelagics (Patterson 1992, Patterson & Melvin, 1996). Such rates derive from the analysis of observed responses of several stocks when exploited at different rates and is likely that mostly apply for small pelagic bonefish. There are however some other problems that may preclude a wider utilization of such type of reference values. The first problem is that this value is species-specific, and may also be affected not only by the dynamics of each species, that may have faster or slower turnovers and resilience to fishing pressure, but also by local ecological and environmental characteristics that should make the same value adequate or not for the same species in different areas. There is a further reason, that probably is more important than the previous one, and is related to considerations on the exploitation pattern of the species. Without the specification of the characteristics of this pattern, is almost impossible to define an optimal E value for all the areas and fisheries where the species is caught even if an intrinsic equal resilience may be hypothesized for the species elsewhere. It is reasonable to assume that the same E will not have the same impact if recruitment to the fishery start at younger age classes or only regards exploitation of older individuals.

Some authors have proposed the use of some life-history invariants for the estimation of potential yield and stock status (Beddington & Cooke, 1983; Beddington & Kirkwood, 2005). It is possible for instance to estimate Maximum Sustainable Yield or the fishing mortality that produces MSY from estimates of growth parameters, size of first capture and information about the steepness of the stock/recruitment relationship.

4.2. Indicators

An indicator is a value that defines the state, response or developement of important aspects of a species or group of species.

Garcia and Staples (2000) define as indicator of a fishery status a variable or pointer which fluctuations reveal changes in some key attributes (Reference points or others) linked to the sustainability of the species involved or of the whole ecosystem. An indicator should provide a representative picture of conditions, pressures on the population responses. It is necessary that they be simple, easy to interpret and useful for looking for

trends over time. Indicators has to respond to changes in the population. It is likely that for each indicator a threshold or reference value can be defined in order to compare its evolution with this defined value. The indicators should be theoretically sound, adequately documented and justified in scientific terms and data needed should be got with a reasonable economic and human cost.The indicators can be single-species based or may regard the whole ecosystem. In many cases, the suitability of certain indicators has been tested (Piet and Jennings; 2005; Trenkel & Rochet 2003). The theoretical basis of some indicators used for the detection of impact of fisheries have been analysed (Jennings et al. 2001; Shin et al., 2005). Several methods have also been proposed for classifying environmental and management indicators, including the pressure state response (PSR) framework proposed by Garcia and Staples (2000). As population dynamics, ecosystems and fishery dimensions are so complex and unpredictable, it is desirable that a series of indicators should be used in order to get an adequate picture of the state of a resource. For example, indicators that relate directly to stock abundance (biomass, recruitment, age structure) are more closely linked to stock status than indicators such as habitat or condition factor. Furthermore it is important to account for significance of indicator changes, because only changes greater than the inherent level of uncertainty in a variable can be considered good descriptor of the phenomenon. Besides, in case of signals of indicators have to be synthesized or to be aggregated into a unique value, weighting is necessarily based on the relative importance of each one defined by experts and/or based on management priorities.

Size based indicators are traditionally considered useful for tracking direct effects of fishing. This idea is based on reasonable statements related to the higher value of larger individuals in fisheries and the concentration of effort on them, that the fishing gears are size selective gears with a major probability to catch bigger individuals, that by increasing effort the older individuals become fewer. The problem is to know if these general considerations do apply for the Mediterranean fisheries and resources. When a fishery targets mostly juvenile individuals, it is difficult to notice any change in the mean size of the catch. The idea that the mean size is inversely related to fishing pressure can be assumed as correct, but it is necessary to take under consideration how much this indicator is sensible to year to year recruitment strength and sampling period.
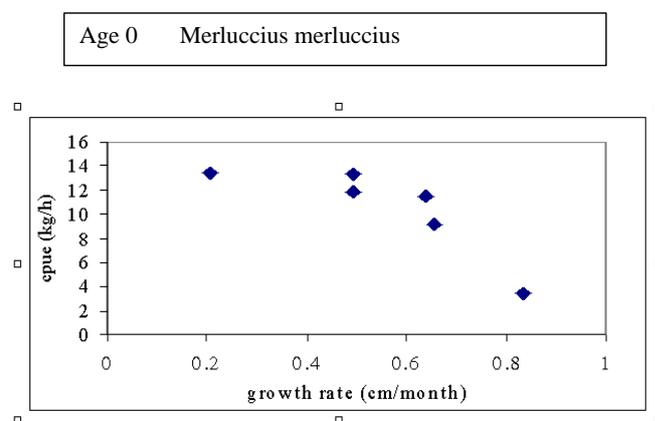
Changes in growth rate depending on density in hake nursery areas in Ligurian Sea.

Caddy (2004) states that: 'in addition to errors and biases in measuring variables contributing to the indicator, the choice and "message" derived from the indicators may be subject to serious errors of interpretation', this because the value of the indicator may be

conditioned by other independent causes. In theory, the mean size should decrease when exploitation increases and size indicators could be thus considered an index of impact of the fishing activity, but recruitment effects and density-dependence growth may mask these expected consequences.

Changes in growth rates depending on density for juveniles of *Merluccius merluccius* in nursery areas. A frequent problem encountered when dealing with indicators regards the difficulty in distinguishing between anthropogenic effects (such as fishing) on indicators and other non- anthropogenic impacts as changes in temperature, availability of food, etc. (Trenkel and Rochet, 2003; Rochet and Trenkel, 2003; Jennings and Blanchard 2004).

The Catch/Biomass rate is an indicator of pressure frequently used. As the biomass seldom is expressed in absolute values, this indicator is only useful for the analysis of time series and a reference value is in general not defined.

Biomass changes useful for a diagnosis of the stock or ecosystem health. The direct observation of stock size is considered a powerful state indicator. Observed declines can be considered as a worsening of the stocks condition. Such observations are not directly available from fisheries, but it is possible to use indirect observations of stock abundance as catch per unit effort or density estimates proceeding from fisheries independent sources (i.e. trawl surveys). Total landings are potentially useful, but it is necessary to check that the target, operation area, seasonality, etc of the fleet has not been changed along the analysed time period. Indices of abundance, however, are not in themselves sufficient to identify reference points that can be converted in clear management advice.

In some cases, however, an observed stability of the overall biomass can result in a misleading perception of the real status. When fishing mortality falls and abundance rises, inter-annual variance in abundance is likely to fall, because effects of annual variations in recruitment will be buffered by the increasingly extended age structure of the population, possibly augmented with effects of density dependence.

The analysis of trends is very useful and often allows to predict with certain level of probability the future evolution based on the past and the knowledge of future management actions. In the models can be included some explanatory variables, derived from the analysis of the time series, that we know will have influence on the future abundance. Available software allows discrimination in the time series of seasonal variations, cycles and trends.
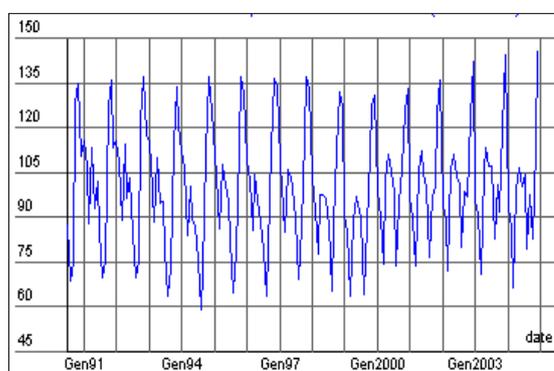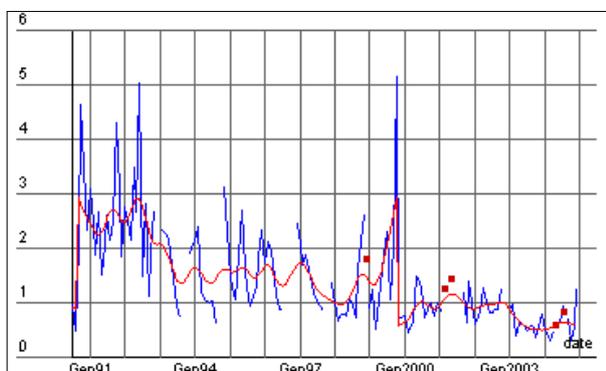
Analysis of trends using TRAMO-SEATS (time series Regression with Arima noise, Missing observations, and Outliers - Signal Extraction in Arima Time Series) implemented in DEMETRA. Trend including residuals (left) and seasonality (right).

The Coefficient of variation of total biomass.

The stability of a species biomass or of the ecosystem is measured as the coefficient of variation (CV) or its inverse over the last 10 years. It indicates levels of "biomass stability", ecosystem stability and resistance to perturbations. Is calculated using the mean(total *B* for the last 10 years) /sd(total *B* for the last 10 years)(Shannon et al, 2010, Blanchard and Boucher, 2001)

The fishing effort exerted on a stock or on a fishery is a simple indicator of pressure. When time series are used for finding trends, it is likely that it will be necessary to proceed with a standardisation between fleets and over time in order to make data consistent and comparable.

Essential fish habitats

Essential fish habitats are habitats which are a very important for the development of a certain life phase of a specific species. They may consist on suitable substrates for spawning or juvenile feeding or refuge areas. The size and quality of these essential habitats can be used as a state indicator. The analysis of the suitability of the environment and the actions aimed at enhancing the environmental conditions compatible with the presence of critical life phases and with the self-renewal of elasmobranchs allow defining how susceptible are regarding fishing pressure and other sources that negatively impact the elasmobranch populations.

Other indicators

Other indicators regards trends in amplitude of the geographic distribution of the fishery (indirect indicator of trends of abundance assuming that the species expand their habitats as abundance increase, mean size or changes in age at maturity, condition factors, changes in natural mortality rates, changes in characteristics and distribution of water masses, salinity, temperature, dissolved oxygen, etc. can be of potential use as indicators of the status of the stocks.

Froese (2004) suggested some indicators which are simple to estimate and can be used for the management of fisheries resources in order to rebuild and maintain healthy spawning stocks.

Indicator 1: 'let them spawn'. This refers to the percentage (i.e., 100%) of mature specimens in the catch and aims at letting fish spawn at least once before they are caught.

Indicator 2: 'let them grow'. This refers to the percentage of fish caught at optimum length, Lopt, i.e., the length at which the number of fish in a given unfished year-class multiplied with their mean individual weight is maximum and where the maximum yield and revenue can be obtained. Lopt is typically a bit larger than Lm and can be estimated from growth and mortality parameters (Beverton 1992):

$$L_{opt} = L_\infty \cdot (3/(3+M/K))$$

It can also be estimated from the empirical equation of Froese and Binohlan (2000), which was based on data from FishBase. The aim here is to catch all fish (100%) within, e.g., *L*opt±10%.

Indicator 3: 'let the megaspawners live'. This refers to the percentage of old, large-sized fish in the catch, i.e., fish of a size > Lopt+10%. The aim here is to implement a fishing strategy for which no (0%) mega-spawners are caught. If such a strategy does not exist, and thus the catch reflects the age and size structure of the stock, values of 30–40% megaspawners in the stock represent a healthy age structure, whereas values of <20% should be alarming.

In the absence of detailed information on catch, abundance and fishing effort, time series of catches in the databases of the Food and Agriculture Organization of the United Nations (FAO 2003) can be analysed retrospectively, using a method inspired by Grainger and Garcia 1996) and modified by Froese and Pauly, 2003). With this method, different 'stages' of a fishery can be distinguished, based on annual catch levels, standardized with the maximum catch in the series. In this way it is possible to define for instance successive phases of development, full exploitation and overexploitation of the resource. In the case of elasmobranches, FAO statistical information is limited, and most of the times, in the catch statistics several species are gathered.

The Depletion Corrected Average Catch Model (DCAC) (Alec MacCall, NMFS/SWFSC/FED) allows the estimation of a level of yield assumed to be sustainable. The goal of the approach is the identification of a relatively high and sustainable yield that guarantees a low probability that fishing at such level the Maximum Sustainable Yield (MSY) can be exceeded.
The evidence of sustainable yield may proceed from the observation of catches in long periods without detecting any sign of reduction of abundance at sea. Such long term yield can be estimated as the mean catch along this time range in which no trend has been observed.

In the case a decrease of the resource is observed, this means that the current catch rate is not sustainable.

The approach is very simple and suitable in situations of data shortage and only need of a reliable and fairly long catch data series. If the abundance declines this means that a fraction of the catch exceeds the potential future yield that can be supported by sustainable production. Such portion is removed in the averaging procedure in order to estimate a sustainable yield.

The method is however relatively new and only a limited number of results derived from its use do exist. The performed tests using both real and simulated data suggest that the approach is robust.

4.3. Assessment of Risk

The risk of overexploitation can be assessed through several approaches. We can quote here the Ecological Risk Assessments and the method based on the inflection point of the population growth curve (which is a proxy for B_{MSY}) The mentioned approaches are not a substitute for stock assessment, but only a method helpful for giving advice using in the best way the limited available data.

Considering the high vulnerability and intense fishing pressure associated with elasmobranches, it is necessary to use in the best manner all the available data, that most

of the times consist in information on life history and characteristics of the fisheries where these stocks are involved.

The Lenfest Ocean Program organized an Expert Working Group for the assessment of alternative approaches for stock assessment of shark species in data-limited situations (Simpfendorfer & al. 2008).

The results for certain elasmobranch stocks obtained through Ecological Risk Assessments as the PSA (Productivity and Susceptibility Analysis) (Braccini et al. 2006, Hobday et al. 2007, Rosenberg et al.2007) and from the analysis based on the position of the inflection point of the population growth curve (Cortés, 2008) were compared by the Working Group with the results obtained for the same stocks using formal population assessments. The integration of the several methods has provided interesting results and encourage further efforts for developing such kind of simple semi-quantitative approaches

Productivity-Susceptibility Analysis PSA

Jennings et al, (1999) state that the more vulnerable species decrease faster in abundance and this is linked with the fact that they mature later, in general at a greater size, grow more slowly towards a greater maximum size and has lower rates of potential population increase. Phylogenetic analyses demonstrated that trends in the community structure can be predicted considering the responses of the better known commercial species. By analogy, based on the existing knowledge of phylogenetic relationships and life histories of the better known species, also trends in abundance of non target spp can also be predicted. Some easily measured parameter as length at maturity or maximum size can be useful for predicting the level of vulnerability to overexploitation of these non-target species.

The method called PSA (Productivity and Susceptibility Analysis) (Stobutzki et al,2001) can be potentially used for assessing the vulnerability of elasmobranch species in a fishery. This semi-quantitative approach allows examining the vulnerability of stocks based on quantitative and qualitative data. It has been tested on many fisheries, especially in Australia and the USA (Hobday et al, 2007; Rosenberg et al, 2009) and recommended as a reasonable approach for determining vulnerability and risk of driving stock to unsustainable levels. Two are the fundamental elements for determining vulnerability. The first one is the stock productivity that mainly depends on the life-history characteristics. It is related with the capacity of the stock to recover when is at low values of abundance. The second is the stock susceptibility, that is the potential of a stock to be negatively impacted by a fishery. The vulnerability of a stock to becoming overfished is linked to the potential for the productivity of the stock to be reduced by fishing pressure. Vulnerability may differ among stocks due to their different life history features and also by their levels of susceptibility to the fishery. The knowledge of the vulnerability of the single species is very important in the management of multispecies fisheries because in such circumstances it is very important to define priorities for the stocks that will be included and monitored under a management plan.

PSA assumes that the species in a fishery will be at risk if they are characterised by low productivity, that implies long recovery times and/or if they are very susceptible (high exposure) to the fishing activity. A separate PSA is undertaken for each species. Several units of analysis, that include indicators linked to productivity and susceptibility are used and a score is determined for each one. The scores are calculated for each factor and they

are displayed on a scatter plot. The graph is divided into regions corresponding to high, medium and low risk.

A tool called PSA 1.4, developed in 2010 by NOAA and included in the Stock Assessment toolbox is available (NOAA, 2010).
The stocks characterised by low productivity score and high susceptibility can be considered the least sustainable and high vulnerable.

The Productivity attributes used in the NOAA PSA software are the following:

The intrinsic rate of population growth r is the maximum population growth that would occur in the absence of fishing.
Maximum age is linked to the natural mortality rate M, because M is negatively correlated with maximum age.
Maximum size is correlated with productivity, because large fish have in general lower levels of productivity.
von Bertalanffy growth coefficient K measures the time a fish needs to reach its maximum size. In general, long-lived species have low *K* and also are characterised by a lower productivity
Natural mortality M. Natural mortality rate defines the population productivity because the stocks with high *M* need of higher levels of production for maintain adequate levels of abundance.
Fecundity is the number of eggs that a female produce in certain period. Musik (1999) suggested fecundity be always measured at the age of first maturity.
Breeding strategy indicates the level of mortality during the first life phases than is related to the way eggs and larvae are placed in the water column, the time (if any) of parental protection of eggs or larvae, the time of gestation.
Recruitment pattern is related to the frequency of recruitment success.
Age at maturity is in general related with the maximum age and long-lived low-productive species in general show a older age of maturity.
Mean trophic level can be a useful information for inferring stock productivity. Major productivity is in general observed for the species that are at lower trophic levels in the community.

The Susceptibility attributes used in the PSA software are the following:

Management strategy. The susceptibility of a stock to be hardly fished will depend on the existence of control rules. Stocks managed by using some effort or catch limitation is expected will have a low susceptibility to overfishing.
Area overlapping measures the level of spatial overlap between the distribution of the stock and the distribution of the fishing effort. A major overlap makes the stock more susceptible to be impacted by fishing activity.
Spatial concentration supplies information on the geographic distribution pattern of a stock. A highly aggregated stock is more susceptible than other with a very scattered spatial distribution.
Vertical overlapping measures the degree of overlapping by comparing the vertical distribution of the stock along the water column with the depth in which the fishing gear operates.
Fishing mortality rate (in relation to M). When estimates of fishing and natural mortality rates are available, it is possible to define which portion of the production is removed by

each one of them. Here a threshold value for F/M=1 is used as a conservative reference value.

Surviving fraction of spawners biomass. The rate between the current stock biomass and the expected level for the unfished stock is used as an indicator of susceptibility to fishing activity. It is expected that the value of this rate will decrease as fishing effort increase.

Seasonal migrations. Movements of exit or entry from the fishery area affect the level of overlapping between the stock and the operational area of the fishery and the "encounterability".

Schooling aggregation addresses behavioural features that may affect catchability, but also changes in the area of distribution due to changes in the population size.

Morphology affecting capture. The efficiency of capture of a gear may be conditioned by the morphological characteristics of a species and the individual size. The inclusion of this attribute considers the portion of the demographic composition of the population that is vulnerable to the fishing gear in use.

Survival after release. The survival of released individuals may vary by species, utilised fishing gear, depth, affecting the susceptibility of the stock.

Desirability or value of the fishery It is assumed that the more valuable species are more prone to becoming overfished due to a major fishing effort where or when they are more concentrated.

Fishery impact on habitats. This indicator is related with the potential indirect effect of a fishery on a species through impacts on the habitat.

As a precautionary measure for risk assessment, in PSA the highest-level risk score is used for missing data but this choice, in the case of a data-poor stocks may result in a high-risk evaluation. PSA can consider missing data within the context of data quality and report the overall quality of data as a separate value. The data-quality index represents the quality of the information on individual vulnerability scores with a ranking ranging from best data to no data that means a little belief in the assigned score. These data-quality scores are computed for the productivity and susceptibility as a weighted average.
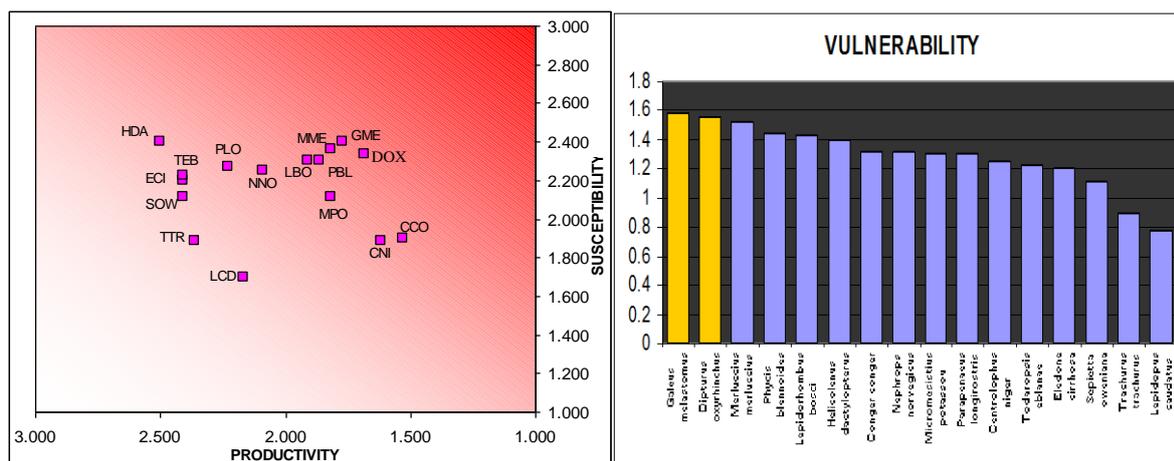
Example of some results derived from PSA in a Nephrops fishery in Southern Ligurian Sea

IUCN Red List status

IUCN, the International Union for the Conservation of Nature, regularly meet specialist groups to assess the conservation status of species on regional and global scales. In these meetings, the conservation status of each species is analysed and are identified

which are under threat and need of conservation actions. Experts determine species relative risk using a set of qualitative and quantitative criteria.

There is however a huge number of species or populations for which data is not enough for producing a reliable assessment.

Position of the inflection point of the population growth curve (R)

An Expert Working Group Report ICCATT SCRS/2008/140 has developed an integrated approach to determining the risk of overexploitation for data-poor pelagic Atlantic sharks. The value of the inflexion point of the population growth curve R is assumed to provide a measure of the level (relative to virgin biomass) at which B_{MSY} may be achieved. This is not an exact measure of B_{MSY}, but rather an indicator of where the level lies relative to other species and as such it should be interpreted cautiously (Cortes 2008). Values of R were calculated using the formula of Fowler (1988):

R= 0.633 - 0.187 ln rT

where rT is the rate of population increase per generation. Median values of intrinsic rate of population increase (r) and generation time (T) can be taken from the matrix population models. The values of B_{MSY} indicated a level of risk relative to the level of depletion in the stock, with higher values indicating greater risk of overexploitation.

The inflection points of the population growth curves ($R \sim B_{MSY}$) can occur at different levels, and so, it is possible to make groupings of species based on the results: (1) those with values around 0.5, that means that they will probably achieve MSY at levels of virgin biomass at or above 0.5, and those that probably achieve *MSY* at levels of depletion at virgin biomass levels much greater than 0.5.

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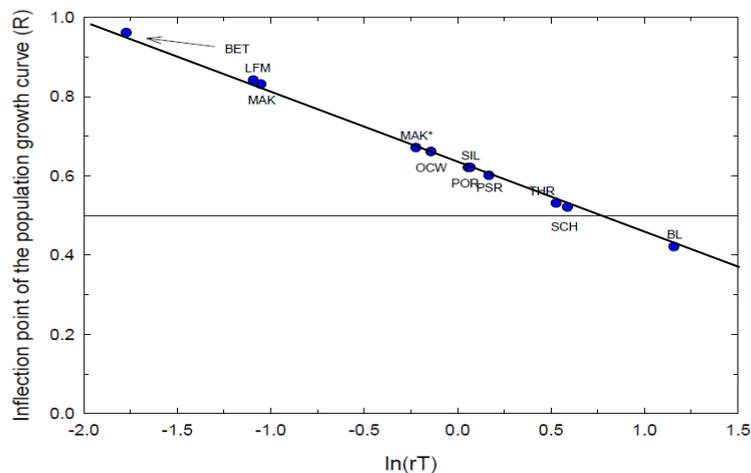
Estimates of the inflection point of population growth curves (proxy of B_{MSY}) for Atlantic pelagic shark. Species. BET, bigeye thresher; BL, blue shark; LFM, longfin mako; MAK, shortfin mako; MAK*, shortfin mako; OCW, oceanic whitetip shark; POR, porbeagle; PSR, pelagic stingray; SCH, scalloped hammerhead; SIL, silky shark; THR, common thresher. The horizontal line, at R=0.5 indicates the position at which K is reached. (From Cortés, 2008, modified).

The intrinsic rates of increase and generation lengths are the best descriptors of the life history strategy of a population.

The rate of increase per generation (Fowler 1988) represents a further step for integrating these related aspects of life history strategy. The rate of increase per generation is linked to the concepts of *r* and *K* selection life strategies (Fowler 1988).

Elasmobranchs may show noticeable variability in these rates. While some studied long-lived, large sharks are more "*K*-selected" reaching *MSY* very close to the level of carrying capacity *K*, some others appears more productive and reach *MSY* at approximately 50% of *K*.

4.4. Demographic models

The approaches based on life history traits as fecundity and survival rates are widely used for modelling population dynamics in elasmobranchs (Hoenig and Gruber, 1990; Cailliet, 1992; Cortés and Parsons, 1996; Cortés, 1999, 2002; Smith et al, 1998; Simpfendorfer, 1999; Mollet and Cailliet, 2002).

These models allow to foresee the responses as changes in the structure of the stock due to mortality (both natural and by fishing). They are useful for deriving management decisions in the case of species in overfishing conditions when not too much information on catches, effort, biomass trends is available. It mainly utilizes information on life history traits. For these species, most of the times data on catch, effort, age composition in the catch are not available and hence surplus production or age structured models are not suitable for stock assessment.

Although for many species life history is well documented, this information has not been up to now used for the analysis of the stock status in the Mediterranean.

The method is potentially useful and has been utilized in several geographic areas as a reference point. It is a good alternative, especially when the age structure of the commercial fisheries is unavailable or when age related parameters are poorly known.

Life history tables (Krebs, 1985) summarize age specific mortality and fecundity rates that characterize a steady state population. It allows the estimation of some of the more important parameters useful for the modelization of the population dynamics of a species.

Life tables and Leslie matrices are based on the Euler-Lotka equation, and needs of the definition of the survival at age, fecundity at age (female pups per female), age at maturity, and maximum reproductive age. They keeps track of the age-specific mortality and reproductive rates, and allows estimating r that measures the potential for growth in a population.

The main information needed regards lifespan, age at maturity, fecundity by age/size and sex ratio.

The Euler-Lotka equation is the following:

$$\sum_{t=0}^{w} l_t \times m_t \times e^{-rt} = 1$$

where l_t is the proportion of animals surviving to the beginning of a given age class, m_t is the age-specific natality, w is the maximum reproductive age, t=age and r is the intrinsic population growth rate.

It is possible to estimate the finite or annual rate of change (e^r) from the estimated values of *r*.

The net reproductive rate R_0 is the offspring number of an individual female belonging to a certain age $t \geq \alpha$ and can be calculated as

$$R_0 = \sum_{t=\alpha}^{w} l_t \times m_t$$

while the mean generation length (G) is the average time that occur between the birth of a parent and the birth of their offspring:

$$G = \frac{\sum_{t=\alpha}^{w} l_t \times m_t \times t}{R_0}$$

The doubling time of population size can be calculated as:

$$T_{\times 2} = \frac{\ln(2)}{r}$$

and the stable proportion of each age P_t in the total demographic distribution of the population can be estimated as follows:

$$P_t = \frac{(e^r)^t l_t}{\sum_{t=o}^{w} (e^r)^t l_t}$$

It is also possible to estimate the intrinsic capacity for increase despite of any particular condition of the environment r_m following the well-known Verlhust-Pearl population growth function as:

$$r_m = \frac{\ln(R_0)}{G}$$

The method assumes that the population is in a steady state situation. It does not include the likely compensatory effects to fishing pressure as reduction in natural mortality, increase in reproductive rates, earlier age of first maturity, etc.

Age specific survival values, that in the former studies using life tables were based only on natural mortality can be easily modified by including fishing effects by using total mortality rates Z for the estimation of survival.

The method however does not include compensatory effects to fishing pressure as reduction in natural mortality, increase in reproductive rates, earlier age of first maturity, etc.

In the former studies using life tables, age specific survival values were only based on natural mortality. Such formulation can be easily modified by including fishing effects by using total mortality rates Z for the estimation of survival.

In alternative, but obtaining almost identical results, the age-structured matrix models, in particular the Leslie matrices, are now frequently used in the assessment of shark populations. (i.e. Hoenig and Gruber,1990). The use of a Leslie Matrix allows estimating the so-called dominant eigenvalue λ The value of λ is found by searching this dominant eigenvalue of the matrix using matrix algebra.

Age-structured matrix models were used in the assessment of sharks and rays (Hoenig and Gruber, 1990; Cortés 1999; Heppell et al. 1999; Brewster-Geisz and Miller 2000).

The theory that is behind the production of estimates of the finite rate of population growth ($\lambda = e^r$) is, but now there are some available software that allows the matrix analyses. A

free shared software called Poptools is a versatile add-in for PC versions of Microsoft Excel (97, 2000 or XP) that facilitates the analysis of matrix population models and simulation of stochastic processes (http://www.cse.csiro.au/poptools/).

The Leslie Matrix can be adapted in order to include information on fishing mortality at specific ages, or changes in the reproductive schedule. Poptools displays a list of all the *eigenvalues* of the projection matrix. The dominant *eigenvalue* give the stable age structure and reproductive values. They are "normalised" to one (100%). It also lists the rate of increase and the generation time.

The software allows the performance of sensitivity analyses that enable an assessment of the influence of changes in the used estimates of the vital rates fecundity at age and mortality rates on the estimated population growth rate. The software allows to perform sensitivity analysis expressed as elasticity, which is simply the proportional (relative) change of sensitivity. This choice facilitates the comparisons related to the consequences (impact on the estimates of population growth rate) of small changes in fecundity and on the mortality rates, which are obviously expressed in different absolute scales. Elasticity is calculated from the elements of the transition matrix, the population growth rate (*r*) and the elements of the right and left eigenvectors.

With the Elasticity analysis it is possible to identify for instance the ages at which smallest changes in vital rates can produce biggest changes in the population growth rate. Elasticity analysis allows defining the management choices likely to produce more benefits to the stock, by estimating how much vulnerable is the species to changes in the survival of the juveniles (or for the adults) depending on the characteristics of the species in question (small or large, slow or fast-growing, long or short-lived species).

Input parameters

The input parameters are natural mortality rate M that can be expressed as survival fractions, fecundity at age, age of maturity, as defined above.

		0	0	0	16	26	34	40	46	50	54
survival	1	0.631284	0	0	0	0	0	0	0	0	0
	2	0	0.398519	0	0	0	0	0	0	0	0
	3	0	0	0.251579	0	0	0	0	0	0	0
	4	0	0	0	0.158817	0	0	0	0	0	0
	5	0	0	0	0	0.100259	0	0	0	0	0
	6	0	0	0	0	0	0.063292	0	0	0	0
	7	0	0	0	0	0	0	0.039955	0	0	0
	8	0	0	0	0	0	0	0	0.025223	0	0
	9	0	0	0	0	0	0	0	0	0.015923	0
	10	0	0	0	0	0	0	0	0	0	0.010052

A classical matrix used for the analysis with Poptools.

The demographic analysis can suggest whether the current combination of exploitation pattern and level of *F* is able to guarantee sustainability for the stock. In the case of elasmobranchs, characterized by low fecundity and relatively late age of sexual maturity, a modest fishing mortality rate is in general the limit in order to guarantee the self-renewal. It is clear that this value of *F* may change if the exploitation pattern changes (i.e. by increasing the age/size of first capture).

```
Eigenvalues                        Eigenvectors (R&L)
Real              Imaginary        Age/stage struct              Reprod val
       1.19258562            0                                    42.7%       0.2%
       0.458223607   -1.005010095                                29.3%       0.3%
       0.458223607    1.005010095                                16.5%       0.6%
       0.028186075   -0.086890538                                 7.6%       1.4%
       0.028186075    0.086890538                                 2.9%       3.6%
      -0.005975042   -0.15739413                                  0.9%       5.1%
      -0.005975042    0.15739413                                  0.2%       6.7%
      -0.028112363    0.270238358                                 0.0%       7.9%
      -0.028112363   -0.270238358                                 0.0%       8.7%
      -0.074649539   -0.106387218                                 0.0%       9.2%
      -0.074649539    0.106387218                                 0.0%       9.8%
      -0.138219898            0                                   0.0%       9.6%
      -0.162856646    0.089466371                                 0.0%       9.4%
      -0.162856646   -0.089466371                                 0.0%       9.3%
      -0.741998952    0.616611554                                 0.0%       9.2%
      -0.741998952   -0.616611554                                 0.0%       8.9%
          r     0.17612374 (rate of increase)
          Ro    2.726377308 (expected number of replacements)
          T     5.694710629 (generation time - time for increase of Ro)
          mu1   5.767271949 (mean age of parents of offspring of a cohort
N (fundamental matrix)
```

Poptools allows calculating a value related to r (the so-called innate capacity for increase under particular conditions that is mostly noted as r_m (Krebs,1985).

It provides other estimates as the expected number of replaced individuals at this mortality rate, the generation time T, the mean age of parents of the offspring in a cohort, etc..
The Leslie matrix can be modified for allowing the inclusion of fishing mortality rates and simulate changes in age of first capture. This enables the estimation of the values of the rate of population growth r_m obtained with different combinations of age first capture t_c and fishing mortality rate F. In the following figure, the green area represents combinations that define a positive value for r_m. The values used for the construction of the isoplets are included in the table below.
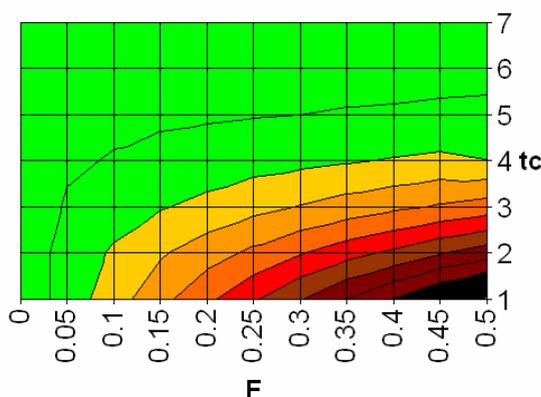

Contour plot of rate of population increase r_m as a function
of fishing mortality (F) and age at first capture (t_c). The green
area represents combinations that define a positive value for r_m.

	age first capture						
F	1	2	3	4	5	6	7
0.00	0.176	0.176	0.176	0.176	0.176	0.176	0.176
0.05	0.058	0.059	0.090	0.114	0.136	0.133	0.135
0.10	-0.055	-0.014	0.049	0.094	0.120	0.130	0.134
0.15	-0.167	-0.086	0.008	0.075	0.115	0.129	0.133
0.20	-0.277	-0.156	-0.030	0.060	0.110	0.128	0.133
0.25	-0.386	-0.220	-0.070	0.041	0.105	0.125	0.133
0.30	-0.494	-0.290	-0.105	0.024	0.100	0.124	0.133
0.35	-0.600	-0.360	-0.140	0.009	0.096	0.123	0.133
0.40	-0.705	-0.420	-0.177	-0.006	0.093	0.123	0.133
0.45	-0.810	-0.490	-0.210	-0.020	0.088	0.122	0.132
0.50	-0.910	-0.550	-0.245	-0.003	0.085	0.120	0.132

Values of the intrinsic rate of population growth r_m obtained with
different combinations of age of first capture L_c and fishing mortality rate F.

The net reproductive rate (R_0) is the total number of female offspring produced per individual in a single cohort:

Expected number of replacements
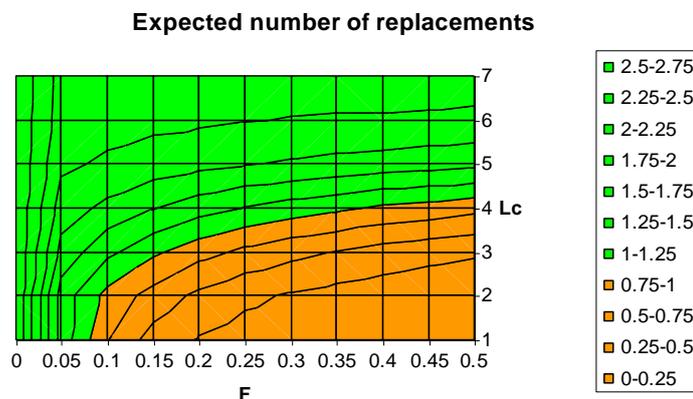
Expected number of replacements per female at different combinations of F and tc.

The mortality rate that is considered as a limit in order to avoid a reduction of the populations (F_c) is derived from the demographic analysis for the current exploitation pattern.

General considerations and findings related with assessment and management derived from life tables analysis

In his study, Frisk at al (2001) give more evidences for the hypothesis that species that maturing later are more vulnerable to exploitation. In general, larger elasmobranchs can be associated with delayed/larger maturity. The authors state that the relation between age of maturity and life span is logarithmic in elasmobranchs and hence long-lived species should spend a major part of their life span as mature adults. This observation leads to the hypothesis that a major duration of the reproductive years produce by a major adult survival is of greater importance in some ray species.

Also Heppell et al. (1999) analysed if the population growth is more sensitive to late/large juvenile and adult stages than to early stage survival and fecundity. A sensitivity analysis can be used to improve the ability to determine the efficacy of conservation plans aimed at the recovery of certain marine species. A single reserve is not able to protect all life stages and habitats. Moreover, the implementation of a reserve designed to reduce mortality of a particular life stage, will benefit some life history strategies more than others.

As regards to a proper use of demographic sensitivity analysis in marine species conservation planning, from analyses of both mortality elasticity and absolute change in *r* derived from a decrease in adult mortality, Gerber and Heppell (2004) found that they resulted highly correlated with adult survival rate and maximum fecundity, and also have shown a significant relationship with age at sexual maturity, longevity and juvenile survival. The mentioned authors state that standardized demographic analysis may be the first step for comparing alternative conservation goals for marine reserve design for species with distinct life histories.

The use of the *elasticity analysis* makes possible to estimate how much vulnerable to changes in the survival of the juveniles (or the adults) depending on the characteristics of the species in question (small or large, slow or fast-growing, long or short-lived species. The results, that allow a comparison among the consequences (sensitivity) to small changes in fecundity and on the mortality rates, derive from standardized information,

considering that data are originally expressed in different absolute scales. The higher elasticity values are highlighted in the next figure as a scale of red.

The higher values are found in the survival at age cells, suggesting that is in the first ages where smallest changes in vital rates are likely to produce biggest changes in the population growth rate. For the species in the example, the analysis suggests that management measures aimed at protecting juveniles (e.g., mesh size increase, nursery area closures) should provide greater benefits to the population than a strategy aimed at protecting adults.

The current F is compared with F_c a drastic reduction of F should be necessary in order to drive the stock to a more productive and sustainable status. The demographic analyses (growth rates at different combinations of F and t_c and elasticity) have shown that the achievement of such goals might be facilitated by an increase of the age of entry to the fishery of the species.

	1	2	3	4	5	6	7	8	9	10	11	12
	0	0	0	0	0.1004	0.0522	0.0192	0.005	0.001	0.0001	2E-05	2E-06
	0.1779	0	0	0	0	0	0	0	0	0	0	0
	0	0.1779	0	0	0	0	0	0	0	0	0	0
SURVIVAL	0	0	0.1779	0	0	0	0	0	0	0	0	0
	0	0	0	0.1779	0	0	0	0	0	0	0	0
	0	0	0	0	0.0775	0	0	0	0	0	0	0
	0	0	0	0	0	0.0254	0	0	0	0	0	0
	0	0	0	0	0	0	0.0062	0	0	0	0	0
	0	0	0	0	0	0	0	0.0011	0	0	0	0
	0	0	0	0	0	0	0	0	0.0002	0	0	0
	0	0	0	0	0	0	0	0	0	2E-05	0	0
	0	0	0	0	0	0	0	0	0	0	2E-06	0
	0	0	0	0	0	0	0	0	0	0	0	1E-07
	0	0	0	0	0	0	0	0	0	0	0	0
	0	0	0	0	0	0	0	0	0	0	0	0
	0	0	0	0	0	0	0	0	0	0	0	0

Results of an elasticity analysis comparing the consequences (sensitivity) of small changes in fecundity and on the mortality rates, standardized considering that they are originally expressed in different absolute scales. The higher elasticity values are highlighted in a scale of red. In this case, the higher values are found in the survival at age cells, suggesting that is in the first ages where smallest changes in vital rates can produce biggest changes in the population growth rate.

Otway et al, (2004) have found that Juvenile survival is the major contributing factor to population growth in many different long-lived shark species. In consequence, management aimed at reducing mortality during the earlier stages of these sharks life history are very important for the long-term conservation of these long-lived species. Au and Smith (1997) also recognize that the greatest opportunity for compensation occurs during the juvenile stages using a modified demographic model

Elasticity analysis is a powerful tool for finding where management or conservation action might produce the greatest benefits to the population. Cortés (2002) state that elasticity analyses of sharks suggest that some long-lived species are more vulnerable to changes in the survival of the juveniles species juvenile survival while for others, the same author applied simple sensitivity analyses of life to ther species and found that other vital rates, as age at maturity and reproduction can have the greater effect on r (e.g. Cortés, 1995, 2002).

Elasticity analysis performed on some skates Frisk et al, (2002), indicates that reducing first year and juvenile mortality should produce greater positive results in terms of

increasing growth rate. Size limits and mesh size regulations can be efficient measures for reducing juvenile mortality.

Stage-based models

Leslie matrix is applied to populations with annual breeding cycle. A similar model, which is known as stage based or Lefkovitch matrix, replaces age classes with life stages. In this case, individuals can both remain in the same stage class or move on to the next one.

The life history of a species is divided into discrete segments or stages as neonate, juvenile, breeding adult, non-breeding adult, etc. This type of model is especially indicated where there is only a limited age information regarding the studied species or the time the species last during each stages is quite variable. In long-lived species, stage-based models is useful for simplifying the computations. The formulation of the stage-based matrix is very similar to that of a age-based Leslie matrix, with columns representing stages instead of ages, and the survival values recalculated using the probabilities of an individual to survive and move from one stage to the next There are several approaches to calculating these parameters.

Cortés (1999) applied a method for the sandbar shark using the duration of each stage G_i and the stage-specific survival probabilities P_i.

Also in the case of using a stage-based matrix model it is possible to examine management options. The effect of fishing mortality on r can be also estimated and in this way to define the rate of fishing mortality r will equal 0.

The use of stage-based models is still not frequent for elasmobranches probably because the concept of stage-structure is more easily applicable to plants or for other animals which have more distinct life-history. Brewster-Geisz and Miller (2000) were the authors that first used a stage-based model for the sandbar shark, *Carcharhinus plumbeus*. According to the mentioned authors, the stage-based models provided a more realistic view of the dynamics of the species. They suggest that for long-lived marine species small errors in parameter estimates can be magnified using age-based models.

stages than do elasmobranchs.

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