# Elasmobranch Fisheries:

# Status, Assessment and Management

by

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Abstract.

An overview of the situation of elasmobranch fisheries around the world and problems for their assessment and management are presented. Four different studies are carried out, each attacking a particular problem under this general topic. The first, is an in-depth review of recent trends in elasmobranch exploitation and management on a worldwide basis aimed at closing the gap in baseline information about these fisheries on a global scale. In the second study, a deterministic age-structured simulation model is developed to analyse density-dependent changes in fecundity as a response to increased fishing mortality in a hypothetical shark population. The use of the model as an aid in management decision-making is exemplified with a case from a tropical shark fishery. Monte Carlo analysis is used in the third study, to evaluate the Schaefer and Fox surplus production models and the delay-difference model of Deriso-Schnute for the estimation of assessment and management parameters of elasmobranch fisheries. The fishery models are evaluated by comparing their estimates of stock assessment and management parameters against the known values of a full age-structured stochastic simulation model of a shark population. Different scenarios of stock recruitment relationship, fishable stock size, spatial behaviour of the sharks, and data quality are used for testing robustness. None of the fishery models performs satisfactorily under situations of density-dependent catchability. When catchability remains constant, the Deriso-Schnute model outperforms the Schaefer or Fox models, both for biomass and management parameter estimation. In the final study, the multispecies shark fishery of Yucatan, Mexico, is used as an example of the problems for elasmobranch stock assessment in the real world. The fishery is analysed by fitting the Schaefer model to catch and cpue data. The results highlight severe deficiencies in the data available for assessment which are characterised by a lack of contrast in the cpue data. Some alternative management recommendations aimed at improving the data for assessment are given.

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***To my wife, Ying,***
***for her unconditional support and love,***
***which made this work possible.***

# Preface.

This dissertation comprises four main chapters which are essentially separate studies. Although these studies are linked by the common subject of fisheries for sharks and rays, they might be regarded as addressing a different aspect of that subject. In particular, Chapter 2 is a comprehensive review of world's fisheries for elasmobranchs, which should be considered as a reference source for the rest of the dissertation.

Chapter 2 was published by the Food and Agriculture Organization of the United Nations during 1994. Permission has been obtained from FAO to include this work in the present dissertation. The full citation of the document is:

Bonfil, R. 1994. Overview of World Elasmobranch Fisheries. FAO Fisheries Technical Paper 341. 119p. FAO, Rome.

The remaining chapters address more quantitative aspects of population dynamics and fisheries assessment. Readers interested in population dynamics should refer to chapter 3. Those interested in the theoretical aspects of assessment methods are advised to go straight to Chapter 4. Finally, readers looking for a practical example of assessment of a real fishery for sharks should go directly to Chapter 5.

A list of the scientific and common names of the sharks and rays mentioned throughout this thesis is included in appendix 1.

# CHAPTER 1

# ELASMOBRANCHS: A SPECIFIC PROBLEM OF FISHERIES ASSESSMENT AND MANAGEMENT.

*The most chivalrous fish of the ocean,*
*To ladies forbearing and mild,*
*Though his record be dark*
*Is the man-eating shark*
*Who will eat neither woman nor child...*

*Wallace Irwin*

## 1.1 Introduction.

Elasmobranch fisheries are coming of age, and with them, a need for the responsible management of these resources. While sharks and rays have usually been shunned by most western cultures as unpalatable or undesirable, they have been the focus of important fisheries in other parts of the world for a long time. According to catch statistics, elasmobranchs play only a secondary role in the world fisheries arena. The global recorded chondrichthyan commercial catch totalled 704,000 t in 1991, an equivalent to 0.7% of the world fisheries during that year (see section 2.2.1.1); even considering an under-reporting level of 50% of the recorded catch, sharks and rays comprise only about 1% of the world fisheries production. Despite this modest role, elasmobranchs can be of prime importance in some regions of the world where they presently sustain significant fisheries. In countries like Sri-Lanka, Pakistan, and Australia, elasmobranchs represent between 5% and 9% of the total fisheries production. In these countries, and in other cases of locally important elasmobranch fisheries (i.e. USA Atlantic shark fishery), the adequate assessment and management of the resources should be a major concern.

Over the last few decades the acceptance of elasmobranchs as food and their importance as fishery resources have grown worldwide, although for reasons discussed below, this is not always properly reflected in the fishery statistics. Two main factors have converged in the creation of new markets for elasmobranchs as food around the globe. Notably, enhanced levels of income in many countries are apparently responsible for an enormous expansion of markets for shark-fin soup. This commodity has superseded its role as a purely traditional Chinese dish to become a trendy gastronomic icon of wealth and power. Parallel to this, there are pressing needs to increase the catches of non-traditional species in order to replace many established fisheries that are currently overexploited (Garcia and Newton 1995). As a net result, elasmobranch fisheries continue to develop in many parts of the globe and the total catches of sharks and rays keep growing without any apparent levelling off.

The problem with unchecked fishery exploitation or inadequate fisheries assessment and management is that they almost inevitably lead to stock collapse and ensuing socio-economic hardship. The NW Atlantic cod fishery is a recent bitter example of this reality. In the case of elasmobranchs, there are more causes for concern about their overexploitation than is usual with most fishery resources. Elasmobranchs and their fisheries possess particular characteristics that warrant their treatment as a specific problem of fisheries science deserving a careful and close attention. This thesis is conceived precisely with this in mind.

## 1.2 A note on taxonomy.

The elasmobranchs are part of the Chondrichthyes. The Class Chondrichthyes comprises a diverse group of fishes whose most obvious common feature is the possession of a cartilaginous skeleton, as opposed to the bony skeleton of the Osteichthyes or bony fishes. The cartilaginous fishes form an ancient successful group dating back to the Devonian, in which basic models remain largely unchanged since their last large flourish during the Cretaceous. Despite their ancient origin, they possess some of the most acute and remarkable senses found in the animal kingdom, allowing them to coexist successfully with the more modern teleost designs.

The chondrichthyans are grouped into two main subclasses: Holocephalii (Chimaeras or ratfishes and elephant fishes) with 3 families and approximately 37 species inhabiting cool and deep waters; and the Elasmobranchii which is a large and diverse group (including sharks and rays) with representatives in all types of environments, from fresh waters to the depths of marine trenches and from polar regions to warm tropical waters. The great majority of the commercially important species of chondrichthyans are elasmobranchs. The latter receive their name from their plated gills, which communicate to the exterior by means of 5-7 openings.

The classification of elasmobranchs is a subject of continuous debate but they are generally divided into 3 groups of sharks (i.e. squalomorphs, galeomorphs and squatinomorphs) which include 30 families and approximately 368 species, and a group known as the batoids, comprising the rays, skates, torpedoes and sawfishes, and embracing a total of 14-21 families and about 470 species (Compagno 1977, 1984; Springer and Gold 1989).

For the practical purposes of this thesis, all the Chondrichthyes (sharks, skates, rays and chimaeras) will often be treated together under the name "elasmobranchs" or "sharks and rays". Although this is an inaccurate term if taken strictly, it simplifies writing and reading by avoiding uncommon or lengthy terminology such as "chondrichthyans" or "sharks, skates, rays and chimaeras" every time I need to make reference to the group.

## 1.3 Problems for the Assessment and Management of Elasmobranch Fisheries.

The elasmobranchs present an array of problems for fisheries assessment and conservation. These problems can be loosely classified as those associated with the particular biology and ecology of these fishes, perceived problems with fisheries theory, problems caused by informational constraints, and finally problems that depend on economic factors.

### 1.3.1 Biology and Ecology.

One of the chief problems faced when dealing with elasmobranch fisheries is that their biological and ecological profiles makes them highly prone to overexploitation. Most shark and many ray species can be classified as strong **K** strategists (Hoenig and Gruber 1990): they are long-lived and this, together with their typical slow growth, results in a late age of first sexual maturation, which commonly ranges between 3 and 25 years depending on the species. Most elasmobranchs have very low fecundity when compared with bony fishes or marine invertebrates, the range of young produced by each female is between 2 and 125 per litter (see Pratt and Casey 1990 for a summary of key life-history characteristics of sharks). The combination of the above factors translates into a low reproductive potential and means that the productivity and resilience of elasmobranch stocks is comparatively low.

At the community level, the top predator niche occupied by many sharks raises the question of their importance as regulators of other species' densities. What are the implications of their removal/depletion from the ecosystem? Although it could be desirable to control shark populations in very specific situations (i.e. because they can affect the economy of important beach resort areas like Natal or Hawaii), it is likely that their removal could bring undesirable ecological and economical consequences, as documented in the coast of Natal by van der Elst (1979). It is very difficult however, to assess the effects of shark depletion in the ecosystem or to know which stocks of elasmobranchs are actually endangered, when there is insufficient information about their basic biology and ecology, the size and state of their stocks, and the real magnitude of their exploitation.

### 1.3.2 Fisheries theory.

It is common belief that another constraint for the assessment and management of sharks and rays is poor development of theory (Anderson 1990, Anderson and Teshima 1990). Fisheries research on elasmobranchs has been scanty if not inadequate and to date there is no specific methodological framework for assessing their fisheries. For a start, surplus production models have been traditionally disregarded for the assessment of shark and ray fisheries for several reasons, without authors actually examining the suitability of the range of these models to specific elasmobranch fisheries. Some authors (Anderson 1990, Anderson & Teshima 1990, Silva 1993) believe that production models are of limited use mainly because of their lack of biological reality (no age structure, no explicit account of growth and reproductive modes, only a very crude incorporation of compensatory mechanisms, etc.). Others, like Holden (1977) and Wood et al. (1979), dismiss surplus production models arguing that elasmobranch biology violates some of the assumptions of these models (see Chapter 4).

The multispecific and multigear nature of most shark and ray fisheries further complicates their assessment. Take the fisheries for sharks and rays in the tropics as an example (which incidentally account for more than 50% of the reported world elasmobranch catches). These fisheries include a mixed catch of several species of sharks and rays; furthermore, these catches are often obtained with a great variety of gears and from several types of vessels. Multispecies fisheries present serious methodological problems because of their complex biological and technological interactions. As a consequence, the theoretical development of multispecies assessment and management is still lagging behind the rest of fisheries science (Hilborn and Walters 1992). In addition to this, the usage of multiple gears and fleets for the exploitation of any fish resource introduces another level of complexity for their assessment and management (e.g. standardisation of effort, the complications of allocating quotas to the various types of gears and vessels).

### 1.3.3 Informational constraints.

There are several kinds of information deficiencies that make the assessment of elasmobranch fisheries very difficult. Probably the most widespread and pressing of them is the lack of adequate fishery statistics. The latter are not well maintained for elasmobranchs around the world, partly because of problems in species identification (specially for tropical species), partly for economic reasons. Most statistical records aggregate all skates and rays in a single group without further species identification, while shark catches are commonly split into two categories, large and small sharks (Chapter 2). In the worst cases, all elasmobranchs are reported together as a single item: "various elasmobranchs". Without statistics by species or species groups it is very difficult to implement most fishery models or to get any insight into the dynamics of the stocks. This sets obvious constraints on our ability to do assessment and management. The lack of statistics by species or species groups has a large economic component: it is not cost effective to sort catches by species when they all attain the same price. Nonetheless, it has been shown that whenever a specific market is developed for an elasmobranch species, catch information becomes readily available.

Lack of information on some relevant areas of elasmobranch population dynamics is also a constraining factor for their assessment and management. First, stock-recruitment relationships have never been documented with hard data for any elasmobranch, although an almost proportional relationship is suspected due to the reproductive strategies of the group (Holden 1973, Hoff 1990). Secondly, there is a general shortage of hard evidence about density-dependent mechanisms regulating elasmobranch population size. Thirdly, the spatial structure and dynamics of most stocks are almost totally unknown. This often overlooked issue is of particular importance to fisheries management both at the local and international level. Inadequate knowledge of migration routes, stock delimitation and movement rates amongst these stocks/substocks, can seriously undermine otherwise "solid" assessment and management regimes. Finally, the difficulties in finding adequate models for elasmobranchs are exacerbated by considerable gaps in our understanding of their biology. The life cycles of most species, even in terms of the basic parameters of age, growth and reproduction, have just started to be unveiled during the last fifteen years or so, and only for a handful of elasmobranch stocks, mainly those of commercial importance in developed countries. This situation has hindered the use of the more "biologically correct" age-structured models.

### 1.3.4 Economics.

Sharks and rays are not a highly priced fishery product (e.g. in the Taiwanese gillnet fisheries of the Central Western Pacific, prices for shark in trunk attain only 20% and 60% of the price of whole tunas and mackerels respectively (Millington 1981)). Some exceptions are sport fisheries, which can be of considerable economic value, certain species for whom a trendy gastronomic demand has recently developed in some parts of the world (i.e. mako and thresher sharks in USA), or those species which unfortunately are highly-sought only for their teeth and jaws, like the great white shark. Shark's fins for oriental soup stand as the only highly-priced elasmobranch product; a kilo of top-quality dry fins can fetch more than $100 US.

A number of the problems associated with elasmobranch exploitation can be traced back to economic factors. In particular, two economic constraints surrounding elasmobranch fisheries cause what I call the "tragedy of sharks". First, research and management for sharks and rays are hampered by the low economic value of this group: in a time when budgets allocated for scientific research and for resource management continue to decrease, priorities are given to resources economically more important than elasmobranchs, i.e. salmons, prawns, tunas, etc. The second, is the high price attained by shark fins in the international market. This high price stimulates increasing exploitation and is also the force behind "finning" practices in many fisheries. Within this context, "finning" (consisting in cutting-off the fins from the shark and dumping the carcass to the sea) is an excessively wasteful habit which is unfortunately very common among fishermen throughout the world: when sharks are caught as a bycatch, the extra high-profits obtained from cutting the fins off the shark are difficult to forgone in the name of conservation. Apart from the ethical issues of this practice (many times the shark is dumped still alive), finning is responsible for high death rates of sharks at sea. Hence, the dynamics of the general low price of elasmobranchs and the high price of shark fins keep sharks caught hopelessly in-between. At present, this dilemma seems to have no viable solution that is consistent with both economic and conservation interests. In addition, the economic incentive behind finning practices is indirectly responsible for the fact that a substantial part of the shark catch never makes it to the official statistics; as no shark carcasses are brought to port, in most cases shark fin landings are not converted to live weight and accounted for. This is probably one of the major reasons why official statistics do not truly reflect the recent increase in elasmobranch exploitation worldwide.

## 1.4 Thesis outline.

Considering the range of problems associated with elasmobranch fisheries, it is not surprising that there is a history of failed sustainability in their exploitation (Anderson 1990, and Holden 1977) provide a list of failed elasmobranch fisheries). In recent years however, there has been growing international concern over the conservation of some elasmobranch stocks and it seems that now, more than ever, there is a need for a more detailed approach to the problem of elasmobranch assessment and management.

This thesis is conceived within the context of elasmobranchs as a special case of fisheries management requiring specific attention. Although each of these chapters constitutes a separate and independent study, they are all liked by the common purpose of contributing towards the establishment of general rules for the rational exploitation of elasmobranchs. In this sense, the concept of tackling fishery resources in a taxonomic/specific basis is not a novelty. As the widespread application of traditional fishery models to all types of resources has frequently proven to be an unsuccessful strategy, more and more fishery scientists are turning towards developing methods tailored to the specific needs of the resource of their concern (see IWC 1987, Punt 1988).

The first study, Chapter 2, serves two purposes, it is a background and framework for the thesis, and constitutes a major reference source for elasmobranch fisheries worldwide. This overview integrates in a single volume the most important information available about fisheries for elasmobranchs around the world, and provides a preliminary analysis of the global situation. It contains detailed analyses of elasmobranch fisheries in countries that have significant shark and ray catches, and gives the first ever estimate of global bycatches in high-seas fisheries of the world. Chapter 2 was published during 1994 by the Food and Agriculture Organization of the United Nations, as FAO Fisheries Technical Paper 341, and has been received with great interest by the scientific community.

In Chapter 3, I contribute towards the understanding of the population dynamics of sharks, and its relation with exploitation. In this study, a simple deterministic simulation model with explicit age structure is used to analyse the effects of density-dependent fecundity upon the ability of a shark population to sustain fishing mortality. This is done by simulating the natural growth of a population and its long term trajectory under different scenarios of exploitation and fecundity increases. In addition, the use of the model as an aid in management decision making is exemplified with a case from a tropical shark fishery. Ultimately, this chapter illustrates the biological limitations that constrain most elasmobranch fisheries.

The search for adequate fishery models for the assessment of elasmobranch populations is treated in Chapter 4. This is a large Monte Carlo analysis of the performance of three fisheries models for the estimation of assessment and management parameters of elasmobranch fisheries. This modelling exercise aims at determining if simple fishery models can be used for shark assessment and management parameter estimation, which model is best, and how robust such models are. A full age-structured stochastic simulation model of a shark population is used to generate catch and cpue data via a stochastic harvesting submodel. A total of 10 different scenarios including variations in stock recruitment relationships, fishable stock size, spatial behaviour of the sharks, and data quality, are analysed to test the robustness of the fishery models. These are the surplus production models of Schaefer (1957) and Fox (1971), and the delay-difference model of Deriso (1980) and Schnute (1985). The Monte Carlo analysis includes several alternatives for implementing the estimations; performance is examined by comparing the estimates of stock assessment and management parameters from each fishery model against the known values of the simulated populations.

Finally, Chapter 5 is an example of the practical difficulties faced during real elasmobranch fisheries assessment work. In this chapter I analyse the multispecies shark fishery of Yucatan through the application of one of the fishery models studied in chapter 4, and suggest some possible solutions to overcome the shortcomings of the available data.

# CHAPTER 2

# TRENDS AND PATTERNS IN ELASMOBRANCH EXPLOITATION: AN OVERVIEW OF WORLD FISHERIES FOR SHARKS, RAYS AND RELATIVES.

## 2.1 Introduction.

The apparent fragility of elasmobranchs and the past history of collapses in their fisheries (see Anderson (1990) for a review) are causes for concern. This is specially true now that the continuing increase in their catches (see 2.2) and the ever demanding market for shark fins may be endangering the sustainability of these fisheries. In recent years, there has been growing international concern over the state of elasmobranch stocks, and some conservationist movements are starting to ring the alarm over the plight of sharks and rays around the world. Unfortunately, whilst the eventual adoption of any harsh unilateral conservation measure (i.e. embargoes like those of the tuna-dolphin controversy) could have negative effects in the fisheries of many countries for which elasmobranchs are of considerable importance, the reality is that the present impacts of fisheries on shark and ray stocks on a global scale are difficult to assess. This happens because there is not only a lack of information on the size of most elasmobranch stocks, but because there is no readily available basic information about their fisheries worldwide. Much of the existing information about shark and ray fisheries is not only disperse, but is usually unpublished and kept by those concerned with their study or management in many laboratories around the world. Even the real magnitude of the total world catches of sharks and rays is uncertain, mainly because of poor knowledge of the total levels of bycatches and discards.

This chapter is oriented towards alleviating the lack of baseline information about shark and ray fisheries worldwide. The present overview of elasmobranch fisheries puts together for the first time most of the existing information about the characteristics and diversity of their fisheries, the species under exploitation, the extent of the catches, the level of bycatches and discards in the high seas, and about management measures currently in use for elasmobranch fisheries.

### 2.1.1 Organisation of this work.

Section 2.2.1 partially describes the scale of global elasmobranch fishing by examining the official statistics worldwide. This section consists of an overview of the catch statistics by FAO major fishing areas including short-term projected catches, and an overview of the trends in the most important fisheries for elasmobranchs in the world on a country basis. For this review, countries with official elasmobranch catches of 10,000 t/yr or more are called "major" elasmobranch-fishing countries.

Sections 2.2.2 and 2.2.3 deal with the major fisheries for elasmobranchs, and the bycatches and discards at sea, respectively. Although it is difficult to distinguish between directed and incidental fisheries, especially when dealing with fishes that are seldom targeted and/or caught alone as is the case of sharks and rays, I will use the following two main divisions for the treatment of elasmobranch commercial fisheries: I will treat under the name "direct", all fisheries that target elasmobranchs, together with all coastal fisheries and small scale multispecies fisheries which catch elasmobranchs incidentally. Typically, the catches from these two sources are mixed together in the official statistics of most countries and it becomes necessary to treat them together. On the other hand, there is a group of large-scale long-range fisheries that mainly target high value species on the high seas. These fisheries very frequently catch elasmobranchs incidentally but usually discard these bycatches for various reasons. They comprise essentially a different category of fisheries in which the elasmobranchs are not only being wasted, but the actual numbers of elasmobranchs caught are also poorly known and usually do not make it to the catch statistics. Most cases in this category are high-seas large scale fisheries with driftnets and longlines carried out by a few countries and targeting high profile resources such as tunids, billfishes, salmonids and squid. These fisheries are suspected of causing substantial kills of elasmobranchs, mainly sharks. This has raised concern over the conservation of these fishes, although on a very different scale than the concern over marine mammals, which are also frequently taken as bycatches in these fisheries. Depending on the amount of information available, the species, catches, gears, fishing units, localities, levels of exploitation and existing management or conservation measures, are summarised for each case.

## 2.2 Characterisation of elasmobranch fisheries.

### 2.2.1 The Official Statistics.

The data used in this analysis is taken from official fishery statistics of each country. The first source is the compilation of Compagno (1990) who analysed FAO data for the period 1947-1985. FAO figures since 1970 have been updated using Fisheries Yearbooks for 1988-1991 (FAO 1990-1993) and data provided directly from the FAO statistical database (David Die, FAO, pers. comm. August 2, 1993). Additional sources are: Fishery Statistical Bulletins for the South China Sea Area years 1976-1990 (SEAFDEC 1977-1993), the Fisheries Yearbook of Taiwan Area for 1970 and 1988-1990 and the Mexican Fishery Statistical Yearbooks 1976-1990 (Secretaria de Pesca 1979-1992). After the thorough review of FAO data done by Compagno (1990), the information is here updated and expanded, including explicitly the catches of Taiwan and estimates of the catches of the People's Republic of China.

#### 2.2.1.1 Trends and outlooks by FAO Major Fishing Areas.

Total world elasmobranch catches reported for the period 1947-1991 (fig. 2.1) amounted to a record of 704,000 t in 1991. Roughly, four periods with different trends can be identified. Poor growth in catches between 1947 and 1954, a sustained increase of production during 1955-1973 followed by a period of sluggish production for most of the 70's and finally renewed growth in catches during the last years 1984-1991.

Catches by FAO Major Fishing Areas from 1967 to 1991 are summarised in table 2.1. An attempt is made to rank these Areas according to their elasmobranch catches. Because the sizes, coastline lengths and human populations of each Area vary notably, a rough index of relative production was devised for comparison purposes. This index is defined as the average total elasmobranch catch of each Area divided by the square root of the surface of that Area in $km^2$. A better index might have been to use the extension of continental shelf for each Area, but it was not possible to obtain these data. Arbitrarily, values of the index below 5 were considered indicative of low relative production, those between 5 and 10 intermediate and those of more than 10 as high. Additionally, the trend in catches during the

Table 2.1 Elasmobranch catches by FAO Statistical Area 1967-1991. Mean catch, variation and Index of Relative Production (IRP) are given for the last 25 yr, and catch trends for the last 10 yr. (All weights in tonnes, live weight.)

| | F.A.O. Major Fishing Areas | Area Million Km2 | Mean Catch '000 t | Coefficient of Variation | I.R.P. Avg Catch/SqrtAre | Trend 82-91 '000 t/y |
|---|---|---|---|---|---|---|
| 27 | NE Atlantic Ocean | 16.9 | 94.8 | 12% | 23.07 | 0.26 |
| 61 | NW Pacific Ocean | 20.5 | 102.3 | 10% | 22.60 | -0.29 |
| 51 | W Indian Ocean | 30.2 | 97.6 | 19% | 17.75 | 1.16 |
| 21 | NW Atlantic Ocean | 5.2 | 26.5 | 57% | 11.61 | 5.48 |
| 37 | Mediterranean & Black Seas | 3.0 | 18.2 | 29% | 10.50 | -0.76 |
| 71 | W Central Pacific Ocean | 33.2 | 59.1 | 38% | 10.26 | 5.00 |
| 41 | SW Atlantic Ocean | 17.6 | 34.2 | 30% | 8.15 | 0.60 |
| 57 | E Indian Ocean | 29.8 | 42.9 | 32% | 7.87 | 1.34 |
| 34 | E Central Atlantic Ocean | 14.0 | 28.6 | 29% | 7.63 | -0.65 |
| 87 | SE Pacific Ocean | 16.6 | 21.4 | 32% | 5.24 | -0.39 |
| 31 | W Central Atlantic Ocean | 14.7 | 17.4 | 47% | 4.54 | 0.77 |
| 77 | E Central Pacific Ocean | 57.5 | 21.1 | 34% | 2.79 | 0.08 |
| 81 | SW Pacific Ocean | 33.2 | 10.4 | 47% | 1.81 | 0.55 |
| 67 | NE Pacific Ocean | 7.5 | 4.8 | 60% | 1.74 | 0.20 |
| 47 | SE Atlantic Ocean | 18.6 | 6.6 | 42% | 1.53 | 0.07 |

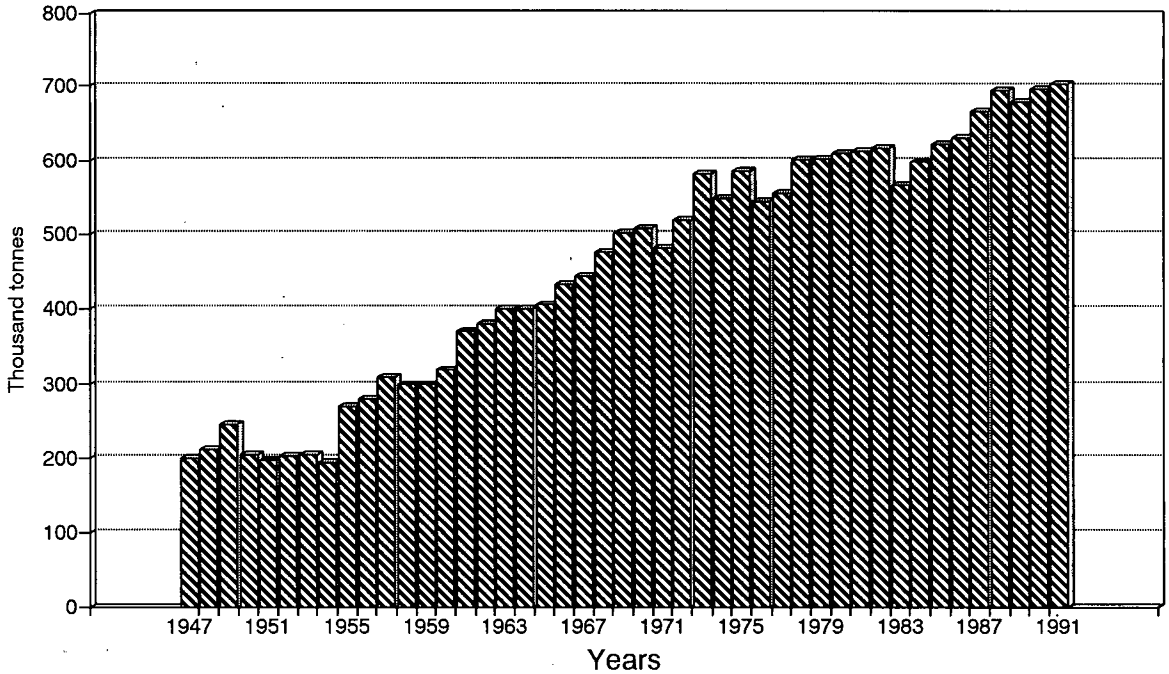



Figure 2.1 World reported catch of elasmobranch fishes 1947-1991. (Data from FAO, SEAFDEC, Fishery Yearbooks for Taiwan Area, and Secretaría de Pesca).

lasts 10 years recorded for each Area, is expressed as the slope of a linear regression fitted to the data by least squares.

In the Western Atlantic Ocean, all the Areas have fairly high increasing trends, especially Area 21 (North West Atlantic) which has the highest increasing trend overall. These three Areas show strong variations in their catches. Area 21 had the highest variability, with recent years apparently recovering production from a dramatic drop suffered in the late 70's following high yields in the early 70's. Area 21 had a marginally high index of relative production (IRP), but considering that a good part of this area includes arctic waters practically void for fishing, we should not expect a much higher future IRP from this Area. In the Western Central Atlantic (Area 31), there was a trend of moderate increase in catches, while the IRP indicated a low elasmobranch yield. This agrees with Stevenson (1982) who suggests that elasmobranch resources in this Area could have been under-utilised. Perhaps there is still a potential for expansion of catches in this Area, mainly for countries of the Caribbean region. For Area 41 (South Western Atlantic), elasmobranch catches also show a moderate increasing trend after variable catches in the 60's. Average catch of elasmobranchs in Area 41 is the highest in the Western Atlantic but this is also the largest area. Hence, it has only an intermediate IRP. Small increases in catches might still be possible here in the future. In comparison, catches in Area 31 have been the lowest in the Western Atlantic, while in the first half of the period and during the last two years, Area 21 had the highest yields.

For the Eastern Atlantic Ocean, Area 27 ( North Eastern Atlantic) had by far the largest catches in the Atlantic as well as the third largest and the second least variable catches in the world. According to the IRP, this Area has the highest production of elasmobranchs worldwide but further expansions in the catches should probably not be expected. In fact, the catch trend hardly increased as production has fallen since 1988, perhaps showing that the high levels of exploitation in this Area are not sustainable. The Central Eastern Atlantic (Area 34) shows a medium level of variation in elasmobranch production. Catches in this Area increased during the early 70's but the recent trend shows a slow decline. This is an Area with an intermediate IRP, thus a good recovery in catches could be possible. For the Mediterranean Sea (Area 37), production was relatively variable during the period examined, but its recent trend of declining catches is the steepest. Because of the small size and the high density of human settlements of this Area, fishing is intense and the IRP for elasmobranchs is the third highest in the Atlantic Ocean. Very likely, shark and ray stocks here are close to full exploitation. In Area 47 (South Eastern Atlantic) catches have been fairly variable. It has the second smallest mean catch of elasmobranchs and the lowest IRP in the world, showing the most possibilities for increased exploitation of elasmobranchs in the future. From the four Areas of the Eastern Atlantic, Area 27 dominated the catches with an elasmobranch production superior to those of the other three Areas together.

There are only two FAO Areas in the Indian Ocean. The Western Indian Ocean (Area 51) has the second highest average yield in the world. This Area has shown reasonably low variability in catches. Catches increased steadily up to the early 70's but fell dramatically during 1983. Judging from the recent increasing trend in production, the situation seems to be recovering but catches have not yet reached previous levels. The IRP of Area 51 is the third highest in the world. Most of the catches in this Area are taken in the northern region by Pakistan, India and Sri-Lanka. Stocks in the northern region might be close to over-exploitation, but given the large extension of this Area and the low catches from its southern portion, it might present some possibilities for increasing elasmobranch exploitation especially from oceanic species. Area 57 (Eastern Indian Ocean) shows more variable catches with a growing trend. It has an intermediate IRP and higher yields are expected here. In the Indian Ocean, Area 51 produces on average more than double the catches of Area 57.

In the Western Pacific Ocean, catches in Area 61 (North Eastern Pacific) had a decreasing trend and the lowest variability of elasmobranch catches in the world. This Area had the highest average yields in the world and the IRP was accordingly very high, marginally second to that of the North Eastern Atlantic. Therefore, catches in this area might not increase substantially in the future and stocks may even be presently overexploited. Area 71 (Central Western Pacific) showed the second highest increasing trend in catches, reaching in the last few years catches five times those of the mid-60's. The IRP in this area is relatively high and might indicate that yields could probably not be expanded much more. In the South Eastern Pacific (Area 81), catches have varied substantially, with a low positive trend in recent catches. Average catches and the IRP are very low. One of the possible reasons for this is the relatively small extension of coastline inside this Area, together with correspondingly few human settlements. The potential of this area to significantly increase catches will depend mainly on the capabilities of the stocks of oceanic and deep water elasmobranch species to sustain fisheries. Of the three Areas of the Western Pacific, Area 61 is the most important in elasmobranch exploitation having produced on average almost twice the catches of Area 71 and about ten times those of Area 81.

Finally, for the three areas of the Eastern Pacific, Area 67 (North Easter Pacific) has the smallest average catches and the highest variation in the world. The IRP is the second smallest of all Areas and the trend of recent catches is moderately positive. Larger catches could be obtained here in the future. Area 77 (Central Eastern Pacific) has somewhat variable catches with a very low increasing trend and a very low IRP. Area 77 is the largest in the world but the associated low human population density might account for the low IRP. The potential for increasing catches here is probably good especially in Central American countries and in the vast oceanic waters. The South Eastern Pacific (Area 87) is the only Area of the East Pacific with a negative trend in catches and has an intermediate IRP. Further increases in the catches should be possible here. Of the whole Eastern Pacific, Areas 77 and 87 have almost the same average catches during this period, amounting to about four times those of Area 67.

Assuming that recent catch trends will remain without major changes in each of FAO's Major Fishing Areas, reported catches of elasmobranchs in the world can be expected to reach between 755,000 t and 827,000 t by the year 2000. These forecasts are based on 5 year step "jackknife" linear regression analyses of elasmobranch catches since 1967 in each FAO Major Fishing Area.

2.2.1.2 Catches by countries.

Data for the period 1947-1991 indicate that 26 countries presently harvest or have harvested recently more than 10,000 t/yr of elasmobranch fishes, i.e. there are 26 "major elasmobranch-fishing countries". Although there are no official statistics for the elasmobranch catches of the People's Republic of China, they also surpass 10,000 t/yr (see section 2.2.2.4.), and therefore qualify the PRC as one of the major elasmobranch-fishing nations.

Historical catch statistics for the 25 major elasmobranch-fishing countries for which data are available, are shown in table 2.2. Japan has traditionally been the overall major fisher of elasmobranchs in the world with average catches of 65,000 t/yr. Indonesia, India, Taiwan and Pakistan follow with catches between 33,000 t/yr and 43,000 t/yr. France, the UK, the former USSR and Norway, report harvests of between 21,000 t/yr and 27,000 t/yr. Mexico, Brazil, South Korea, Nigeria, Philippines, Sri-Lanka and Peru caught between 11,000 t/yr and 18,000 t/y. A large group of countries formed by Spain, USA, Malaysia, Argentina, Thailand, Australia, Italy, New Zealand and Ireland followed with average catches between 4,000 and 10,000 t/y.

Even though there is great variability in the development of individual elasmobranch fisheries, some patterns can be identified from these data. About one third of the major elasmobranchs-fishing countries show recent levelling-off trends in their catches, probably signalling full exploitation of shark and ray resources. Seven countries show falling trends while nine others have a definite rise in catches (fig. 2.2).

Elasmobranch production is specially high in Indonesia, where catches have rocketed since the early 70's with no sign for a slow-down at all. Taiwan, the USA, Spain and India are other examples of countries with increasing fisheries for sharks and rays. Japan, historically the leader in elasmobranch production, has a clear trend of decreasing catches. Norwegian catches show a clear trend of increase until the early 60's, but this has since switched to a sharp decrease. The same is true for the former USSR catches, which had a growing period from the early 60's to the mid-70's but have substantially decreased since, without recovering to former levels. Catches in the UK have a very slight almost imperceptible decreasing trend. Pakistan had a powerful trend of increasing catches until the late 70's, but dramatically dropped in the early 80's to make a slow but sustained comeback. The range of causes for the decreasing trends is not easy to find in all cases, but possible explanations for some cases are given below in the individual country accounts (2.2.2).

The reported statistics indicate that during the last 15 years sharks have been slightly more important in the catches than other elasmobranchs. The average reported catch of sharks and batoids is 285,433 t/yr and 180,196 t/yr respectively, with 190,159 t/yr reported as "various elasmobranchs". After reallocating the reported catches in this last category to

Table 2.2 Reported world catches in commercial elasmobranch fisheries (thousand tonnes, live weight). (Data from Compagno, 1990 and FAO, unless otherwise indicated). (T.W.F. = total world fisheries, T.W.CUPL = total world cupleoid fisheries, T.ELAS = total world elasmobranch fisheries. EL/FISH = T.ELAS as % of T.W.F., CUPL/FISH = T.W.CUPL as % of T.W.F.).

| YEAR | T.W.F. | T.W.CUP | T.ELAS. | EL/FISH % | CUPL/FISH % | USA | MEX (p) | BRA | PERU | ARG | USSR | UK | EIRE | NORW | SPAIN |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1947 | 20000 | 3481 | 201 | 1.0 | 17.4 | 13.1 | | | 1 | 6.9 | | 27.1 | | 10.8 | 10.4 |
| 1948 | 19600 | 3486 | 211 | 1.1 | 17.8 | 12.8 | | | 1.4 | 5.1 | | 29.8 | | 10.7 | 10.4 |
| 1949 | 20100 | 3724 | 245 | 1.2 | 18.5 | 11.2 | | | 1.2 | 2.4 | | 30.7 | | 10 | 10.6 |
| 1950 | 21100 | 4081 | 204 | 1.0 | 19.3 | 6.1 | | | 1.3 | 1 | | 29.2 | | 12 | 10.8 |
| 1951 | 23600 | 4392 | 197 | 0.8 | 18.6 | 12.8 | | | 1.1 | 1.2 | | 32.6 | | 14 | 11.6 |
| 1952 | 25200 | 5440 | 203 | 0.8 | 21.6 | 3.1 | | | 2.5 | 1.7 | | 30.8 | | 15.3 | 10.1 |
| 1953 | 25900 | 5500 | 204 | 0.8 | 21.2 | 2 | | | 3 | 2.9 | | 28.8 | | 15.5 | 10.8 |
| 1954 | 27600 | 5760 | 194 | 0.7 | 20.9 | 2.8 | | | 4.5 | 2.4 | | 27.8 | | 18.8 | 10.9 |
| 1955 | 28900 | 6410 | 270 | 0.9 | 22.2 | 2.8 | | | | 2.2 | | 28.6 | | 19.1 | 10.8 |
| 1956 | 30500 | 7020 | 280 | 0.9 | 23.0 | 3.3 | 4.1 | | 3.3 | 3.8 | | 27.1 | | 22.8 | 11.7 |
| 1957 | 31500 | 7230 | 310 | 1.0 | 23.0 | 14.3 | 4.5 | | 3.5 | 4.1 | | 29.1 | | 20.9 | 14.1 |
| 1958 | 32800 | 7450 | 300 | 0.9 | 22.7 | 16.6 | 5.6 | | 3.4 | 4.6 | | 29.2 | | 24.4 | 14.2 |
| 1959 | 36400 | 9060 | 300 | 0.8 | 24.9 | 16.6 | 4.6 | 4.6 | 4.2 | 4 | | 27.2 | | 22 | 15.4 |
| 1960 | 39500 | 10290 | 320 | 0.8 | 26.1 | 16.6 | | 5 | 7.2 | 2.4 | | 25.7 | | 29 | 14 |
| 1961 | 43000 | 12620 | 370 | 0.9 | 29.3 | 5.7 | 3.6 | 5.9 | 3.8 | 2.9 | | 27.8 | | 45.6 | 14.3 |
| 1962 | 46400 | 14730 | 380 | 0.8 | 31.7 | 9 | 3.4 | | 5.4 | 3.9 | | 23.6 | | 38.7 | 10.6 |
| 1963 | 47600 | 14930 | 400 | 0.8 | 31.4 | 9 | 3.5 | 7.6 | 5.1 | 6.2 | | 23.5 | | 51.6 | 11.4 |
| 1964 | 52000 | 18730 | 400 | 0.8 | 36.0 | 8.6 | 4.4 | 8.9 | 6.1 | 6.9 | 0.1 | 35.7 | | 45.7 | 13.8 |
| 1965 | 52400 | 17442 | 405 | 0.8 | 33.3 | 8.6 | 5.1 | | 7.6 | 7.2 | 3.7 | 24.7 | | 32.2 | 11.4 |
| 1966 | 57300 | 19426 | 433 | 0.8 | 33.9 | 6.3 | 5.3 | 10.6 | 9.9 | 7.7 | 20.8 | 24.5 | | 27.6 | 11.5 |
| 1967 | 60400 | 20308 | 444 | 0.7 | 33.6 | 7.3 | 6.5 | 13 | 19.6 | 10.1 | 20.1 | 25.6 | | 27.7 | 10.8 |
| 1968 | 63900 | 21117 | 476 | 0.7 | 33.0 | 7.3 | 6.3 | 12.5 | 24.7 | 13.7 | 31.9 | 25.9 | | 25.3 | 11.1 |
| 1969 | 62700 | 18786 | 502 | 0.8 | 30.0 | 7.3 | 8.9 | | 14.7 | 10.8 | 40.1 | 23.8 | | 21.5 | 9.9 |
| 1970 | 70388 | 22209 | 508 | 0.7 | 31.6 | 1.7 | 9.1 | 12.6 | 19 | 8.7 | 26.3 | 22.3 | 1.7 | 44.1 | 9.9 |
| 1971 | 70747 | 20241 | 482 | 0.7 | 28.6 | 1.5 | 9 | 12.6 | 11.3 | 10 | 48.3 | 26.3 | 1.7 | 29.8 | 0 |
| 1972 | 66121 | 14288 | 519 | 0.8 | 21.6 | 1 | 8.4 | 3.2 | 10.5 | 9.6 | 55.3 | 26.6 | 1.5 | 31.1 | 11.4 |
| 1973 | 62824 | 12073 | 583 | 0.9 | 19.2 | 1.8 | 14.1 | 15.6 | 21.5 | 13.4 | 47.1 | 26 | 1.5 | 30.5 | 0 |
| 1974 | 66597 | 14631 | 549 | 0.8 | 22.0 | 2.2 | 16.6 | 9.5 | 16.8 | 14.3 | 55.3 | 24.1 | 1.7 | 30.6 | 0.6 |
| 1975 | 66487 | 14373 | 586 | 0.9 | 21.6 | 1.7 | 14.3 | 9.9 | 14.6 | 13.8 | 58.5 | 26.5 | 1.8 | 35.9 | 1 |
| 1976 | 69930 | 15371 | 544 | 0.8 | 22.0 | 4.1 | 16.1 | 6.1 | 10.5 | 10.6 | 29.4 | 26.6 | 1.9 | 24.8 | 0.7 |
| 1977 | 69226 | 13043 | 556 | 0.8 | 18.8 | 4.7 | 15.6 | 7.3 | 13.8 | 9.6 | 13.7 | 28.1 | 1.8 | 21.9 | 0.4 |
| 1978 | 70596 | 14493 | 600 | 0.9 | 20.5 | 5.9 | 21.5 | 9.3 | 15.6 | 12.5 | 25.7 | 27.2 | 1.5 | 21.5 | 3.7 |
| 1979 | 71331 | 15790 | 603 | 0.8 | 22.1 | 11.1 | 24.6 | 21.9 | 13.8 | 10.0 | 16.2 | 24.2 | 1.7 | 20.0 | 0.9 |
| 1980 | 72141 | 16070 | 609 | 0.8 | 22.3 | 11.2 | 26.6 | 23.3 | 13.3 | 11.3 | 12.6 | 21.6 | 1.8 | 15.6 | 2.1 |
| 1981 | 74884 | 16920 | 612 | 0.8 | 22.6 | 11.0 | 35.7 | 25.8 | 19.1 | 8.3 | 12.5 | 20.3 | 2.5 | 8.9 | 2.4 |
| 1982 | 76810 | 17867 | 617 | 0.8 | 23.3 | 11.7 | 34.6 | 31.3 | 18.8 | 12.8 | 9.2 | 18.9 | 3.2 | 9.6 | 6.3 |
| 1983 | 77591 | 17455 | 568 | 0.7 | 22.5 | 12.4 | 31.4 | 29.1 | 14.9 | 9.5 | 11.2 | 18.8 | 6.8 | 9.8 | 6.1 |
| 1984 | 83989 | 19607 | 598 | 0.7 | 23.3 | 9.3 | 34.1 | 25.2 | 34.4 | 10.2 | 9.5 | 21.2 | 9.4 | 10.1 | 5.7 |
| 1985 | 86454 | 21101 | 623 | 0.7 | 24.4 | 11.9 | 33.3 | 29.6 | 16.8 | 15.3 | 10.2 | 23.0 | 11.8 | 7.8 | 13.7 |
| 1986 | 92822 | 23955 | 630 | 0.7 | 25.8 | 12.1 | 29.4 | 25.7 | 23.3 | 16.1 | 17.5 | 21.5 | 7.3 | 6.5 | 15.8 |
| 1987 | 94379 | 22375 | 666 | 0.7 | 23.7 | 15.2 | 27.9 | 27.8 | 23.1 | 15.3 | 18.1 | 25.9 | 11.4 | 5.1 | 22.0 |
| 1988 | 99016 | 24388 | 694 | 0.7 | 24.6 | 17.2 | 34.6 | 24.3 | 26.6 | 21.1 | 20.9 | 24.6 | 8.9 | 5.2 | 16.7 |
| 1989 | 100208 | 24800 | 679 | 0.7 | 24.7 | 20.4 | 33.1 | 24.9 | 25.0 | 16.5 | 12.0 | 21.2 | 6.2 | 8.0 | 21.7 |
| 1990 | 97434 | 22183 | 695 | 0.7 | 22.8 | 34.6 | 38.1 | 24.7 | 12.6 | 16.7 | 6.0 | 21.7 | 5.0 | 11.1 | 14.7 |
| 1991 | 96926 | 21407 | 704 | 0.7 | 22.1 | 35.5 | 34.0 | 25.2 | 5.7 | 17.6 | 3.1 | 20.4 | 4.0 | 12.3 | 15.9 |
| MEAN | 57896 | 14357 | 455 | 0.8 | 24.4 | 9.8 | 17.4 | 16.4 | 11.7 | 8.8 | 22.7 | 25.7 | 4.3 | 21.4 | 9.8 |
| %variation | 43 | 46 | 37 | 14 | 20 | 76 | 72 | 55 | 72 | 59 | 75 | 14 | 80 | 56 | 57 |
| % of worldwide elasmobranch catch, 1987-1991 | | | | | | 3.57 | 4.88 | 3.69 | 2.71 | 2.54 | 1.75 | 3.31 | 1.03 | 1.21 | 2.65 |
| % importance of elasmobranchs in country, 1987-1991 | | | | | | 0.42 | 2.36 | 3.00 | 0.29 | 3.19 | 0.11 | 2.63 | 3.03 | 0.44 | 1.22 |

(p) data from Secretaria de Pesca

(s) data from SEAFDEC

(s/f) data from SEAFDEC and FAO

(t/f) data from Fishery Yearbooks for Taiwan Area and FAO

Table 2.2 Continued

| YEAR | ITALY | FRA | NIG | PKST | INDIA | SRILK | THAI (s) | MALAY (s) | INDONE (s/f) | S KOR | JAPAN | PHILIPP (s) | TAIWA (t/f) | AUST | N ZEL |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1947 | | 20.5 | | | | | 1 | | | | 73.2 | | | | |
| 1948 | | 16 | | 1.5 | | | 2 | | | 14.6 | 86.1 | | | | |
| 1949 | | 16.7 | | 9.1 | | | 3 | | | | 118.5 | | | | |
| 1950 | | 13.7 | | | | | 2 | | | | 100.7 | | | | |
| 1951 | | 13.5 | | | | | 2 | | | | 85.7 | | | | |
| 1952 | | 13.1 | | 9.8 | | 0.6 | 2 | | | | 89.1 | | | | |
| 1953 | | 14.4 | | 10.8 | 15.9 | 0.7 | 2.2 | | | 10.5 | 97.4 | | 10.7 | | |
| 1954 | | 13.7 | | 9.8 | 16 | 3.1 | 2.3 | | | 9.2 | 102.9 | | | | |
| 1955 | | 14.9 | | 11.7 | 20.4 | 2.5 | 1.6 | | | 10.8 | 97.2 | | | | |
| 1956 | | 15.2 | | 9.7 | 21.9 | 3 | 1.6 | | | 14.8 | 92.6 | | | | |
| 1957 | | 15.2 | | 17.6 | 23.1 | 3.9 | 3.1 | | | 12.2 | 93.8 | | | | |
| 1958 | | 15.2 | | 9.5 | 24.3 | 4.3 | 2.7 | | | 10.2 | 82.9 | | | | |
| 1959 | | 15.1 | | 9.8 | 23.5 | 4.3 | 2.8 | | | 7.6 | 86 | | 16.5 | | |
| 1960 | | 16.7 | | 11.3 | 35.6 | 7.1 | 4.3 | | | 10.9 | 83.9 | | 17.1 | | |
| 1961 | | 34.3 | | 9.4 | 33.6 | 8.5 | 4 | 3.2 | | 8.7 | 78.3 | | 18.9 | | |
| 1962 | | 33.1 | | 22 | 40.8 | 10.3 | 4.5 | 3.2 | | 9.9 | 81.5 | | 19.7 | | |
| 1963 | | 35.5 | 0.3 | 25.2 | 43 | 12.1 | 5.1 | 4.4 | | 9.4 | 77.4 | | 17.1 | | |
| 1964 | | 37.4 | 0.3 | 26.2 | 34.9 | 11.2 | 5.8 | 4.7 | | 12.6 | 69 | | 18.8 | | |
| 1965 | | 29.5 | | 28.2 | 31.4 | 11.8 | 12.4 | 4.6 | | | 66.9 | | 20.2 | | |
| 1966 | | 36.3 | | 37.2 | 37.4 | 11.6 | 12.8 | 6.4 | | 6.3 | 71.1 | | 22.9 | | |
| 1967 | | 33.1 | | 38.4 | 29.6 | 16.3 | 8 | 7 | | 5.6 | 67.5 | | 26.0 | | |
| 1968 | | 27.4 | | 40.3 | 31.2 | 14.7 | 12.3 | 6.5 | | 18 | 56 | | 33.1 | | |
| 1969 | | 39 | | 42.5 | 8.75 | | 18.8 | | | | 59.3 | | 32.8 | | |
| 1970 | 4.8 | 28.2 | 30.4 | 39.8 | 44.1 | 12.5 | 22.4 | 3.6 | | 14.2 | 61.8 | 6.9 | 36.3 | 7.8 | 2.6 |
| 1971 | 5.0 | 25.2 | 9.4 | 41.8 | 41.3 | 9.8 | 12.5 | 6.4 | 10.3 | 12.3 | 50.2 | 7.3 | 39.7 | 7.4 | 3.1 |
| 1972 | 5.4 | 25.7 | 10.2 | 62.9 | 45.2 | 11.5 | 14.4 | 6.7 | 9.2 | 7.2 | 52.2 | 8.2 | 41.4 | 7.4 | 2.4 |
| 1973 | 4.6 | 27.3 | 10.4 | 74 | 60 | 17.9 | 13.6 | 7.7 | 16.3 | 19.3 | 49.4 | 9.0 | 38.1 | 3.0 | 2.6 |
| 1974 | 5.1 | 25.6 | 11.2 | 34.8 | 60.1 | 15.7 | 13.7 | 8.2 | 18.5 | 18.9 | 45.7 | 9.4 | 45.8 | 4.3 | 3.5 |
| 1975 | 4.8 | 23.9 | 12.5 | 36.6 | 61 | 13.1 | 12.1 | 8.5 | 27 | 22.5 | 46.2 | 10.4 | 62.4 | 2.9 | 3.0 |
| 1976 | 5.6 | 26.8 | 19.4 | 40.3 | 49.1 | 15.6 | 11.4 | 12.2 | 28.7 | 18.7 | 52.9 | 9.1 | 59.9 | 4.5 | 4.4 |
| 1977 | 5.6 | 23.2 | 19.9 | 64.1 | 45.6 | 11.3 | 12.2 | 12.2 | 29.5 | 17.4 | 59.7 | 8.9 | 56.4 | 6.9 | 5.3 |
| 1978 | 4.8 | 27.8 | 20.3 | 71.9 | 49.9 | 12.6 | 9.8 | 13.7 | 30.3 | 18.2 | 51.2 | 21.2 | 48.1 | 8.0 | 4.2 |
| 1979 | 4.5 | 31.9 | 20.9 | 74.7 | 40.9 | 12.8 | 9.3 | 11.9 | 33.3 | 19.0 | 53.0 | 9 | 43.7 | 7.5 | 4.4 |
| 1980 | 5.1 | 35.0 | 21.5 | 65.0 | 49.7 | 14.2 | 9.5 | 10.9 | 42.9 | 18.0 | 54.3 | 9.7 | 52.3 | 9.4 | 6.6 |
| 1981 | 3.9 | 42.0 | 11.9 | 62.9 | 50.0 | 21.3 | 10.2 | 11.5 | 43.2 | 21.5 | 49.0 | 12.6 | 43.7 | 9.5 | 7.3 |
| 1982 | 4.8 | 32.8 | 14.0 | 68.8 | 47.8 | 20.1 | 9.6 | 9.9 | 45 | 20.5 | 47.6 | 11.4 | 47.2 | 9.6 | 8.0 |
| 1983 | 6.5 | 39.2 | 12.0 | 18.2 | 51.4 | 19.2 | 8.5 | 10.3 | 49.9 | 22.3 | 43.7 | 8.2 | 43.5 | 9.4 | 9.9 |
| 1984 | 12.2 | 34.1 | 13.0 | 20.9 | 54.0 | 14.7 | 8.1 | 10 | 52.8 | 20.5 | 45.7 | 11.3 | 48.5 | 7.1 | 11.5 |
| 1985 | 14.3 | 33.1 | 14.2 | 29.5 | 50.5 | 15.1 | 9.2 | 10.3 | 54.3 | 22.9 | 39.4 | 10.9 | 55.8 | 7.5 | 11.1 |
| 1986 | 13.4 | 36.4 | 9.3 | 27.4 | 49.1 | 15.5 | 13.5 | 11.2 | 55.1 | 21.0 | 44.4 | 18.1 | 46 | 10.6 | 8.3 |
| 1987 | 9.8 | 36.6 | 9.5 | 28.6 | 57.9 | 16.1 | 14.4 | 11.7 | 58.2 | 16.2 | 42.9 | 16.2 | 50.1 | 13.5 | 9.5 |
| 1988 | 10.4 | 34.4 | 9.5 | 30.3 | 73.5 | 16.7 | 11.4 | 16.8 | 63.9 | 21.7 | 28.6 | 17.9 | 43.9 | 14.2 | 13.0 |
| 1989 | 8.4 | 34.0 | 6.9 | 27.6 | 66.3 | 17.0 | 11.2 | 13.4 | 74.9 | 20.8 | 33.9 | 19.0 | 54.8 | 8.3 | 10.8 |
| 1990 | 9.6 | 34.0 | 8.4 | 40.0 | 51.2 | 15.3 | 11.0 | 16.8 | 73.3 | 15.7 | 32.1 | 18.4 | 75.7 | 6.7 | 12.3 |
| 1991 | 13.7 | 25.7 | 7.2 | 45.1 | 52.9 | 18.4 | 11.8 | 16.9 | 79.8 | 17.3 | 33.8 | 19.0 | 68.6 | 7.6 | 13.7 |
| | 7.4 | 26.7 | 12.6 | 33.0 | 41.6 | 11.9 | 8.4 | 9.4 | 42.7 | 15.2 | 65.2 | 12.4 | 39.9 | 7.9 | 7.2 |
| | 47 | 33 | 54 | 63 | 36 | 47 | 62 | 43 | 49 | 34 | 34 | 37 | 42 | 36 | 53 |
| | 1.51 | 4.79 | 1.21 | 4.99 | 8.78 | 2.42 | 1.74 | 2.20 | 10.18 | 2.67 | 4.98 | 2.63 | 8.52 | 1.46 | 1.73 |
| | 1.89 | 3.78 | 2.92 | 7.42 | 1.72 | 8.76 | 0.43 | 2.46 | 2.41 | 0.66 | 0.31 | 0.85 | 3.50 | 4.80 | 2.19 |

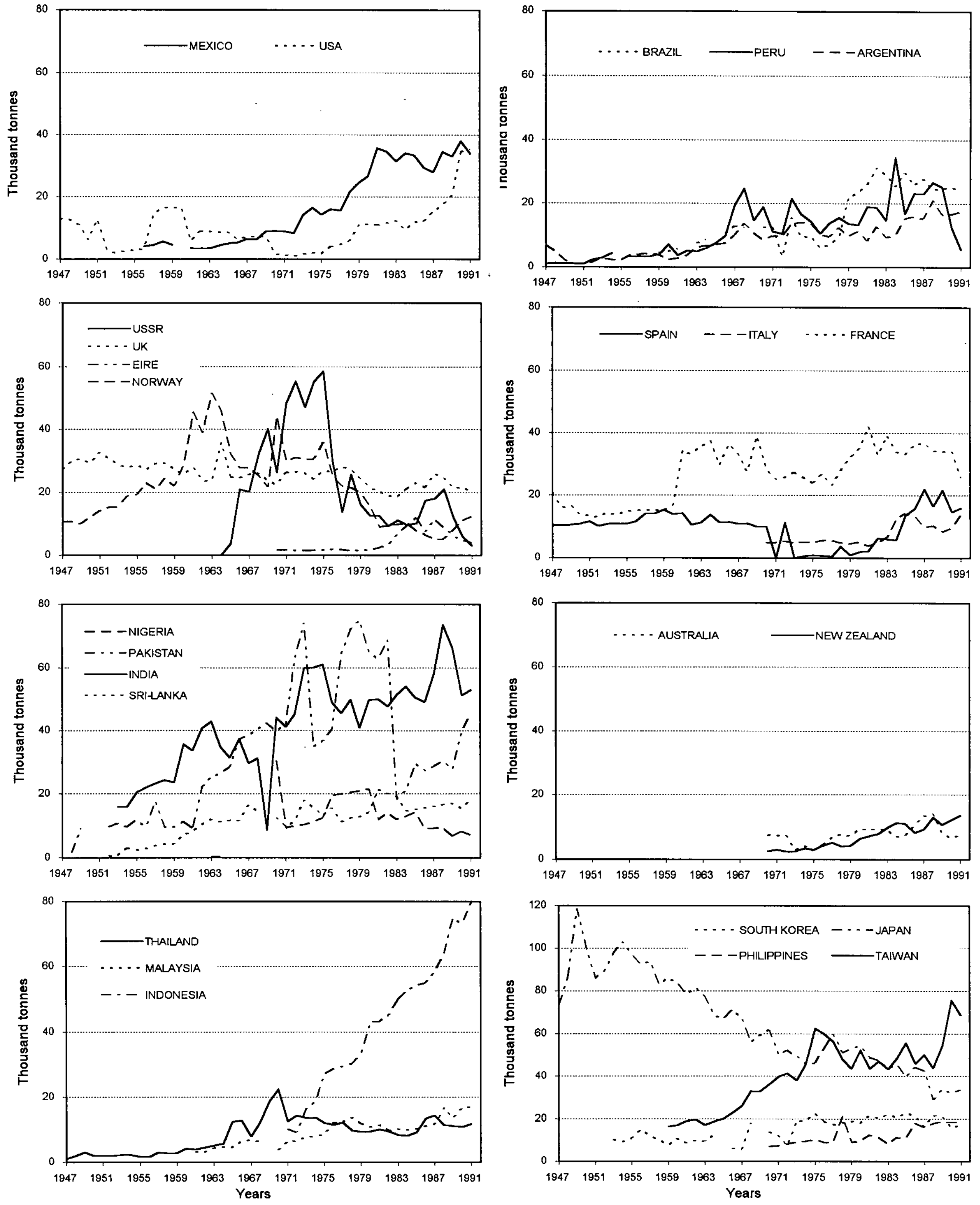



Figure 2.2 Historical catches of elasmobranchs for the 25 major elasmobranch-fishing countries, arranged by geographical area.

either sharks or rays with the aid of ancillary information from section 2.2.2., and then splitting the remaining 94,139 t/yr of "various elasmobranchs" in equal parts, a total of 393,741 t/yr (about 59.5 % of total elasmobranchs) can be attributed to sharks and 262,046 t/yr to skates and rays (about 39.5 %) whereas only less than 1% are chimaeras and elephant fishes.

### 2.2.2 Major Fisheries for Elasmobranchs.

Two main sources were used for the information in this section. First, literature on the subject was consulted for each case as extensively as possible. Much information probably remains in the form of unpublished reports from different governmental offices around the world. Secondly, in an attempt to fill in some of the many gaps of information, a questionnaire was designed and sent to officers or scientists in all the major elasmobranch-fishing countries. However the success of this approach was poor. The extent of published work on elasmobranchs in each country and the level of response to the questionnaire is reflected in the quantity of information that is presented under each country's account.

#### 2.2.2.1 America.

USA.

General overview.

The USA is one of the few countries with reasonably detailed information on elasmobranch fisheries. Nevertheless, no comprehensive account of these fisheries on a national basis seems to exist. Main fisheries for elasmobranchs of the USA have traditionally been centred on sharks, although batoids have also been fished through history. Rays and skates were recorded in U.S. commercial catches as early as 1916 (Martin and Zorzi, 1993), mainly as a bycatch of more important fisheries. However, the first directed fisheries for elasmobranchs in this country seem to have been those for the soupfin shark *Galeorhinus galeus* (then *zyopterus*) in California, and the fishery for large sharks off Salerno in Florida. Both flourished as a consequence of the high demand for shark liver oil in the 40's-50's and died mainly because of the laboratory synthesis of vitamin A in 1950.

Until recently and according to FAO statistics, the commercial catches of elasmobranchs in the USA were together with those of Argentina, the least important among major elasmobranch-fishing countries in America. However, this situation has changed since the early 90's. Elasmobranch production has varied considerably for the last 40 years oscillating around 10,000 t/yr until the late 80's. Two periods of very low catches were 1952-1956 and 1970-1977, while 1958-1960 saw some of the highest yields. The post-war peak of 17,000 t has been broken since 1988 (fig. 2.2). Catches rapidly increased during the mid 70's and soared in the mid-80's. Still, elasmobranchs are only a minor fishery in the USA as catches during 1987-1991 averaged only to 0.42 % of the total fisheries production while they contributed 3.57% of the total reported elasmobranch catch in the world (table 2.2).

According to Compagno (1990), the recent rise in catches might reflect a change in consumer preferences that has made shark meat fashionable and acceptable to the public as a direct result of the infamous "Jaws" films. This would have prompted a whole new group of fisheries directed to sharks in the USA. According to Cook (1990), recent changes in international shark-fin markets have further increased the demand for sharks in the USA. Amongst these new fisheries, those for the thresher shark, *Alopias vulpinus,* the Pacific angel shark *Squatina californica* and the shortfin mako *Isurus oxyrinchus,* are the most important in the west coast. For the Gulf of Mexico and east coast of USA, most of the recently rising shark fisheries have a diverse catch of coastal sharks, reported as unclassified sharks. This difference in detail of the reported catches on both sides of the USA probably follows because of the existence of well defined markets and prices for many species of elasmobranchs on the west coast, which are lacking in the east coast. According to the National Oceanic and Atmospheric Administration (NOAA 1991), in the east coast fisheries only mako sharks have a separate price from the remaining "unclassified sharks".

Data from FAO shows that up until 1980 elasmobranch catches in the USA were about evenly distributed in both sides of the country. Since 1981 however, the east coast has contributed the bulk of the catches thanks to a large expansion of fisheries for sharks and rays (fig. 2.3). This new growth led to the recent implementation of management strategies for large shark fisheries in the east coast.

Overall, the two most important elasmobranch groups in the fisheries of the USA are the

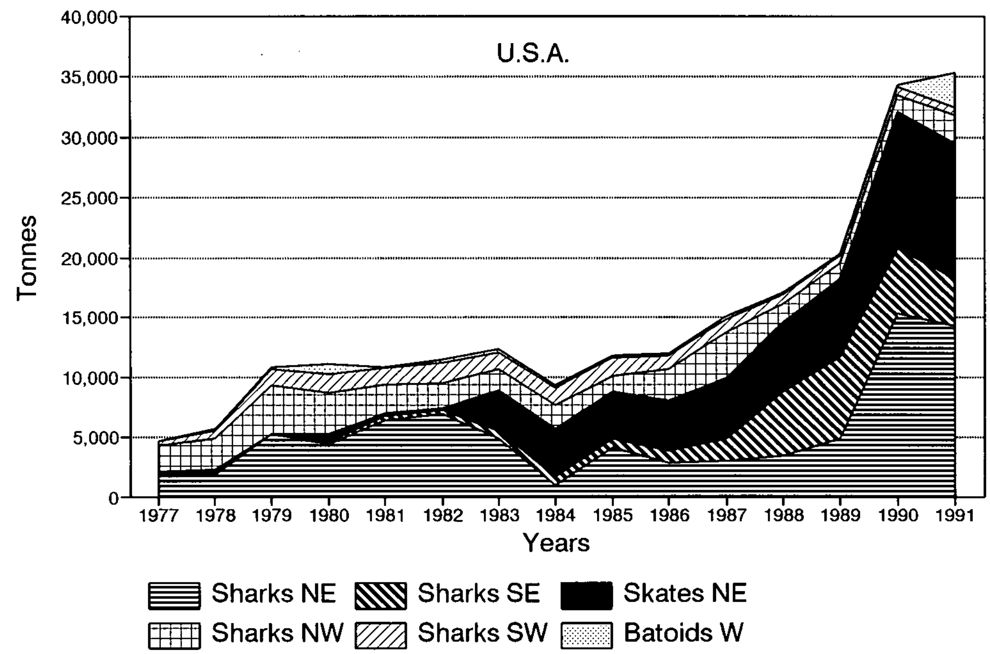



Figure 2.3 Elasmobranch catches of the USA by major groups and regions as reported by FAO, during 1977-1991.

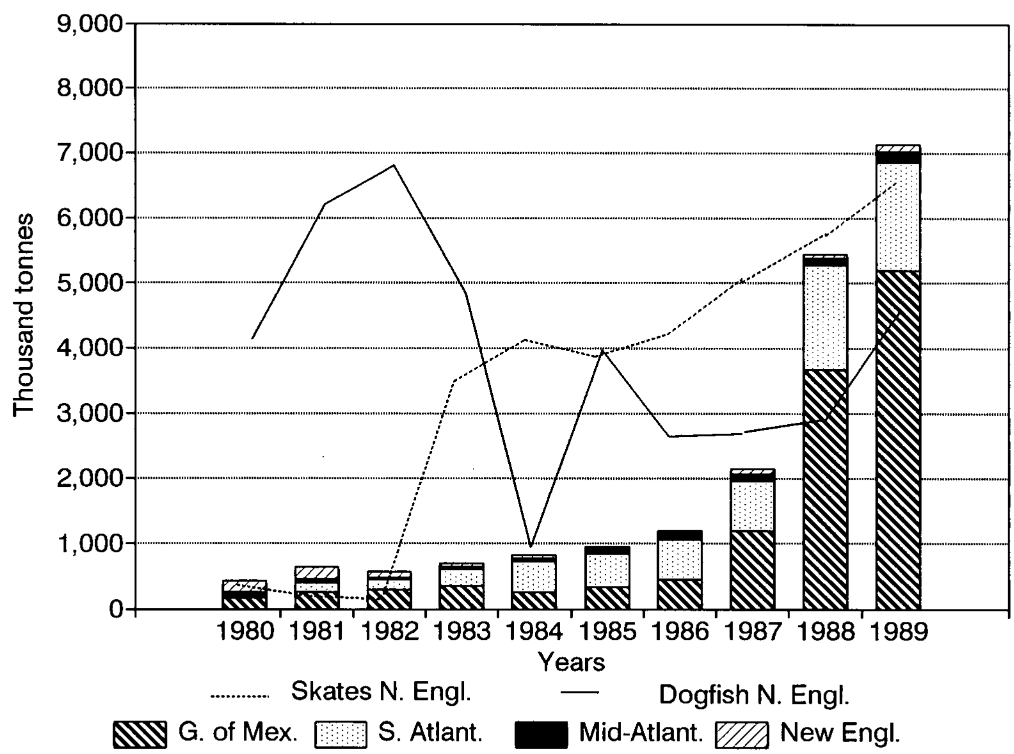



Figure 2.4 Elasmobranch catches from the east coast of the USA during 1980-1989. Bars represent shark fisheries. (Data from FAO and Hoff 1990).

dogfishes (mainly *Squalus acanthias*) and the skates. Dogfish and skate catches from the waters within FAO Area 21 (roughly corresponding to the New England and Mid-Atlantic regions of the National Marine Fisheries Service [N.M.F.S.] of the USA) and dogfish catches in FAO Area 67 (roughly corresponding to the coasts of Washington and Oregon) have dominated the elasmobranch production of the country until very recently. Dogfish catches off the Northeast USA (Area 21) were the major part of total elasmobranch catches during 1979-1983, fell down in 1984 and slowly recovered since 1985. Skate catches in this region have increased tremendously since 1983, and were the second most important group in 1989 with almost one third of the total elasmobranch catches of the country (figs. 2.3 and 2.4). Dogfish catches off the northwest USA (taken mainly in Washington) had fairly variable yields, contracting during the mid 80's, partially recovering in 1986-1987 only to subsequently fall. Most of the dogfish on the east coast and skates on both sides of the country are taken by trawlers, while dogfish in the northwest coast are apparently harvested with gillnets and trawl nets. Although both rays and dogfish are low priced resources when compared with some other elasmobranchs (i.e. mako or thresher sharks) they are available in such large quantities that they become profitable for fishing enterprises.

The dogfish and ray resources of the USA appear to be without specific management regimes. In the best cases, some stocks are included in general management schemes for ground fish resources. Grulich and DuPaul (1987) estimate that the spiny dogfish stocks of the USA east coast could support a harvest of about 24,000 t/yr in the mid-80's. However, recent studies suggest that the biomass of the *Squalus acanthias* stock sustaining most of this fishery, although increasing recently, is highly variable from year to year (Silva 1993). This could mean that high levels of exploitation would not be sustainable, preventing steady supply to a large market.

The East Coast.

Throughout this century, the single most important fishery for sharks in the East Coast of USA was that for large sharks of Salerno Florida during the period 1935-1950 (major accounts in Springer [1951, 1960]). The fishery was centred on production of vitamin A from shark liver oil and was closed as a consequence of the industrial synthesis of this vitamin. Fins and hides were also utilised. The fleet was centred at Salerno but it usually extended operations west to the Mississippi river during summer, and after 1945 it expanded to include boats in the Carolinas, the Florida keys and the Gulf coast of Florida. The Caribbean and West Indies also provided catches to this fishery. In the later years, approximately half of the catches came from the Gulf of Mexico. This fishery had up to 16 boats of 12-15.5 m operating concurrently, each fishing with two bottom longlines of at least 200 hooks, in depths up to 91 m. Floating longlines and bottom gillnets were also used occasionally. Sandbar sharks, *Carcharhinus plumbeus,* composed most of the catches, which peaked at 10,514 sharks in 1947.

In recent times, the second most important elasmobranch fisheries in the USA after dogfish and rays, are the growing fisheries for large sharks in the Gulf of Mexico and South Atlantic. While catches of large sharks have remained practically unchanged in the Mid-Atlantic and New England regions, shark catches in the Gulf of Mexico and South Atlantic regions underwent radical changes, with an eightfold increase in yield from 1984 to 1989 (fig. 2.4). This trend, caused mainly by the development of a stable market, began in 1985 when fishermen started to target sharks with gillnets and longlines. The landing of previously discarded shark bycatches from other fisheries also became attractive.

According to NOAA (1991), directed fisheries for sharks in the east coast include: a monofilament 18-64 cm mesh driftnet fishery apparently directed to schooling blacktip sharks (*Carcharhinus limbatus*) in Florida; a May-November gillnet fishery in the east coast of Florida catching mostly gray sharks *Carcharhinus* spp.; a driftnet fishery for tunas, billfishes and sharks in the Atlantic, Gulf of Mexico and Caribbean; pelagic longlines for tunas, billfishes and sharks in the Atlantic, Caribbean and Gulf of Mexico (this fishery deploys gear in a mechanised operation involving large vessels and thousands of hooks); a recent fishery for sharks with bottom longlines sets manually up to 100 hooks from each small boat; and a pelagic hook and line fishery for tunas, billfishes and sharks in the Gulf of Maine, South New England and the Mid Atlantic.

Lawlor and Cook (1987) report that the seasonal East Florida longline fishery for sharks is carried out from boats 11-15.5 m long with 2-4 fishermen using bottom and/or surface longlines in 1-2 days operations. The longline's mainline varies from 1.6 to 10 km in length and is made of 4.8-6.4 hard-lay tarred nylon, from which 300-500 gangions of 3.6 m long multistrand steel cable fall, armed with a 3/0 or 3.5/0 shark hook each. Buoys are attached to the mainline on 28-30 m leaders for bottom longlines and for pelagic longlines with 10-30 m leaders. Bluefish, bonito, mackerel, mullet and squid are the most common bait. Apparently, about 110 boats work full-time and year-round in this fishery following migrating sharks along the coast. Additional information (NOAA 1991) indicates that 124 vessels target sharks in the USA east coast, with longliner catches during 1989 adding up to 6,140 t while gillnetters caught 621 t.

Some sharks in the east coast of USA are also landed as bycatch in the following fisheries: the Gulf of Mexico tuna fisheries; the Gulf of Mexico and south Atlantic coast snapper-grouper bottom longline fishery; swordfish gillnet fishery of Massachusetts and Rhode Island (up to 15 vessels) and the gillnet fisheries of Maine, Virginia, New York and New Jersey. According to the reports of Hoff (1990) and NOAA (1991), the main species caught in the South Atlantic and Gulf of Mexico with gillnets are *Carcharhinus plumbeus, C. limbatus, C. leucas, C. altimus, C. brevipinna, Galeocerdo cuvieri, Carcharias taurus, Negaprion brevirostris, Sphyrna lewini* and *S. mokarran*. Those captured with longlines are mainly *C. plumbeus, C. limbatus, C. isodon, C. acronotus, C. leucas, C. brevipinna, C. obscurus, Rhizoprionodon terraenovae, Carcharias taurus* and *Sphyrna lewini* (a glossary of english and latin species names is given in appendix 1). Russell (1993) reports *C. limbatus, Mustelus canis* and *Rhizoprionodon terraenovae* as the most common species in shark longliners in the northern Gulf of Mexico. Data from NOAA (1991) shows that ex-vessel prices for sharks in the Gulf of Mexico and southeast USA almost doubled from an average price in constant US dollars of \$0.57/kg in 1979 to \$1.12/kg in 1986, the average since 1983 being approximately \$ 1.00/kg. Meanwhile, the prices for fins have risen nearly an order of magnitude since 1985. In general, higher prices are paid for dressed carcasses and also for sharks fished in waters more than 3 miles from the coast as opposed to those caught inside the 3-mile State waters limit. Only the mako shark attains a higher price than the rest of the species which are treated as "unclassified shark".

Hoff (1990) stresses that important bycatches of several species of sharks are taken regularly by the shrimp trawling fisheries of the north Gulf of Mexico. Unfortunately, most of the catches are discarded because of a lack of market for them (GMFMC 1980). The only published estimates (NOAA 1991) indicate that the incidental catch of sharks in the Gulf of

Mexico shrimp fishery is of about 2,800 t/yr. Most individuals are juveniles caught in nursery areas and this kill might represent an important threat for recruitment to the breeding stocks in future years. Escapement of larger specimens will probably increase if the regulations for the mandatory use of turtle excluder devices (TED) are approved. Overall, total yearly discards of sharks in all fisheries of the east coast of USA averaged 16,000 t (NOAA 1991).

The great increase in shark exploitation both by commercial and recreational fishermen in the east coast of the USA led to the establishment of a management regime based on catch quotas and bag limits since april 1993. This management regime took an outstanding 10+ years for implementation due to, among other things, lack of appropriate data on abundance, biology, distribution, life history and catches of sharks, needed for stock assessment. Given concerns about possible overexploitation of shark stocks during the late 80's, an assessment was performed with the very limited available information. The shark Fisheries Management Plan divides shark species in three management units according to their habitat, as shown in table 2.3. The estimated levels of MSY, which appear rather small when compared with catches in neighbouring Mexico, are about 3,400 t for large coastal sharks and about 3,600 t for small coastal sharks (Parrack 1990, NOAA 1991). A number of management measures in effect since April 1993 include: 1993 commercial quotas (in dressed weights) of 2,436 t for large coastal sharks and 580 t for pelagic species; recreational bag limits of four sharks/vessel/trip for large coastal and pelagic sharks combined, and five sharks/person/day for small coastal species; commercial fishing only by permit; fins landed in proportion to carcasses; release of shark bycatches ensuring maximum probability of survival; compulsory submission of sales receipts and logbooks from selected commercial and recreational operators; accommodation of observers in selected commercial boats; and banning of shark catches for foreign vessels in USA waters (NMFS 1993).

The West Coast.

Holts (1988) and Cailliet et al. (1993) review the shark fisheries of the west coast of the USA. Aside from the spiny dogfish fisheries which dominate the catches of the west coast, an important group of directed fisheries for sharks suddenly rose in California at the end of the 70's. However, some of these fisheries declined within the following ten years. These

Table 2.3 Sharks species considered in each of the USA east coast management unit (from NOAA 1991).

| | FAO Common Name | Scientific Name |
|---|---|---|
| Large Coastal Sharks | Sandbar | *Carcharhinus plumbeus* |
| | Blacktip | *Carcharhinus limbatus* |
| | Dusky | *Carcharhinus obscurus* |
| | Spinner | *Carcharhinus brevipinna* |
| | Silky | *Carcharhinus falciformis* |
| | Bull | *Carcharhinus leucas* |
| | Bignose | *Carcharhinus altimus* |
| | Copper | *Carcharhinus brachyurus* |
| | Galapagos | *Carcharhinus galapagensis* |
| | Night | *Carcharhinus signatus* |
| | Caribbean reef | *Carcharhinus perezi* |
| | Tiger | *Galeocerdo cuvier* |
| | Lemon | *Negaprion brevirostris* |
| | Sandtiger | *Carcharias taurus* |
| | Bigeye sand tiger | *Odontaspis noronhai* |
| | Nurse | *Ginglymostoma cirratum* |
| | Scalloped hammerhead | *Sphyrna lewini* |
| | Great hammerhead | *Sphyrna mokarran* |
| | Smooth hammerhead | *Sphyrna zygaena* |
| | Whale | *Rhincodon typus* |
| | Basking | *Cetorhinus maximus* |
| | Great White | *Carcharodon carcharias* |
| Small Coastal Sharks | Atlantic sharpnose | *Rhizoprionodon terraenovae* |
| | Caribbean sharpnose | *Rhizoprionodon porosus* |
| | Finetooth | *Carcharhinus isodon* |
| | Blacknose | *Carcharhinus acronotus* |
| | Smalltail | *Carcharhinus porosus* |
| | Bonnethead | *Sphyrna tiburo* |
| | Sand devil | *Squatina dumeril* |
| Pelagic Sharks | Shortfin mako | *Isurus oxyrinchus* |
| | Longfin mako | *Isurus paucus* |
| | Porbeagle | *Lamna nasus* |
| | Thresher | *Alopias vulpinus* |
| | Bigeye thresher | *Alopias superciliosus* |
| | Blue | *Prionace glauca* |
| | Oceanic whitetip | *Carcharhinus longimanus* |
| | Sharpnose sevengill | *Heptranchias perlo* |
| | Bluntnose sixgill | *Hexanchus griseus* |
| | Bigeye sixgill | *Hexanchus vitulus* |

fisheries originated mainly as a response to changes of trends in consumer preference which increased demand and sparked a soar in prices for some species. Total catches (excluding dogfish) increased through the late 70's to a peak of about 1,800 t in 1982 but have since varied with a decreasing trend (table 2.4). Cailliet et al. (1993) consider market fluctuations and susceptibility to overexploitation of some stocks as the main reasons for diminishing catches.

The first fishery to rise was that for the common thresher *(Alopias vulpinus).* This was centred between San Diego and Cape Mendocino, and initiated operations with 15 large-mesh driftnet vessels in 1977. Ex-vessel prices for this species increased from US$0.64/kg in 1977 to US$3.52/kg in 1986. Soon, the thresher shark fishery was displaced by the more valuable swordfish fishery and the thresher shark became a secondary target. This lead to political upheaval and spurious management of the fishery and resulted in the loss of the thresher populations (see Bedford 1987 for a detailed account). Catches peaked in 1982 at 1,083 t when more than 200 vessels were operating, but slowly declined thereafter. During 1986, a limited area and season legislation was passed, causing a further decline in catch until the directed fishery for this species was outlawed in October 1990. At present, only incidental catches in the swordfish fishery are permitted, and they account for almost 300 t/yr (D. Holts, NMFS Southwest Fisheries Centre, pers. comm., July 22, 1991). Throughout most of the fishery catches were composed mainly of young sharks 1-2 years old, but also included a few *A. superciliosus* and *A. pelagicus.* Bedford (1987) reports that market sampling data showed decreasing modal sizes through time along with dropping CPUE indices since the mid-80's. Unpublished data (Holts, pers. comm. op. cit.) shows the mean length of catches clearly declined during the same period.

Another important recent development in the west coast was the fishery for Pacific angel shark *(Squatina californica).* This began as a very localised operation in Santa Barbara in 1977 (166 kg landed), underwent great expansion in 1981 (158 t landed), reached a peak in 1986 (563 t landed) and suffered a steady fall in the following three years (121 t in 1989) (table 2.4; Cailliet et al. 1993). Ex-vessel prices climbed from US$0.33/kg in 1978 to US$0.99/kg in 1984 (Holts 1988). Pacific angel sharks were taken initially as bycatch of the Pacific halibut fishery with bottom set trammel nets. As markets and demand expanded, they began to be targeted with single-walled gillnets made of nylon twine (No. 24 to No.30)

Table 2.4 Shark landings, in dressed weight (kg), west coast USA (adapted from Cailliet et al. 1993)

| NAME | 1976 | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 | 1989 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Spiny dogfish * | 2,647,724 | 2,639,478 | 2,940,474 | 4,341,382 | 3,242,281 | 2,193,992 | 2,085,296 | 2,450,906 | 3,474,589 | 2,512,781 | 2,339,761 | 3,695,570 | 3,404,150 | 2,952,019 |
| Common thresher | 16 | 58,803 | 137,141 | 333,963 | 819,925 | 879,679 | 1,083,510 | 797,838 | 754,814 | 694,060 | 551,685 | 349,622 | 290,457 | 297,405 |
| Pacific angel | 313 | 166 | 37,402 | 56,138 | 49,950 | 118,054 | 144,351 | 159,510 | 287,496 | 561,966 | 563,473 | 426,845 | 223,072 | 121,786 |
| Shortfin mako | 9 | 9,040 | 12,456 | 16,042 | 70,523 | 125,227 | 239,565 | 146,621 | 110,786 | 97,667 | 207,053 | 277,857 | 222,105 | 176,298 |
| Soupfin | 82,805 | 73,623 | 79,936 | 100,715 | 87,222 | 116,836 | 113,078 | 79,974 | 253,459 | 110,622 | 89,512 | 103,371 | 66,554 | 77,559 |
| Blue | 1,041 | 44,658 | 16,300 | 38,121 | 87,227 | 92,116 | 26,258 | 6,348 | 1,789 | 1,070 | 1,294 | 1,774 | 3,301 | 6,184 |
| Leopard | 0 | 10,109 | 15,870 | 12,243 | 18,199 | 22,419 | 32,082 | 45,994 | 31,411 | 34,366 | 29,885 | 25,138 | 18,949 | 22,913 |
| Bigeye thresher | 4,507 | 0 | 0 | 0 | 4,922 | 4,786 | 16,466 | 48,354 | 33,945 | 54,313 | 20,968 | 11,488 | 5,463 | 10,030 |
| Brown smooth-hound | 21,287 | 120 | 3,344 | 1,108 | 2,625 | 10,733 | 2,389 | 6,402 | 3,673 | 15,124 | 6,132 | 5,864 | 7,047 | 4,979 |
| Smooth hammerhead | 0 | 844 | 465 | 138 | 0 | 1,026 | 847 | 20,194 | 3,101 | 1,780 | 1,647 | 820 | 244 | 72 |
| Grey smooth-hound | 27 | 0 | 15,320 | 5,469 | 345 | 0 | 1,144 | 479 | 3,108 | 851 | 230 | 0 | 9 | 187 |
| Horn | 6,624 | 525 | 124 | 9,559 | 3,843 | 1,038 | 3,424 | 220 | 278 | 165 | 89 | 24 | 62 | 15 |
| Great white | 0 | 0 | 0 | 1,030 | 754 | 19 | 3,656 | 288 | 2,770 | 1,299 | 419 | 610 | 997 | 596 |
| Pelagic thresher | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4,959 | 0 | 291 | 108 | 1,041 | 350 | 113 |
| Sevengill | 0 | 0 | 38 | 0 | 247 | 1,550 | 927 | 788 | 128 | 405 | 25 | 77 | 10 | 6 |
| Cow | 23 | 0 | 113 | 132 | 199 | 350 | 603 | 571 | 605 | 194 | 199 | 163 | 71 | 59 |
| Salmon | 0 | 35 | 0 | 0 | 0 | 0 | 452 | 104 | 0 | 915 | 1,022 | 116 | 122 | 159 |
| Swell | 0 | 0 | 1,269 | 0 | 74 | 0 | 0 | 0 | 101 | 9 | 0 | 0 | 1 | 2 |
| Sixgill | 0 | 0 | 0 | 9 | 5 | 144 | 0 | 58 | 44 | 2 | 0 | 25 | 9 | 61 |
| Dusky | 0 | 92 | 47 | 0 | 0 | 89 | 0 | 54 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unspecified sharks | 264,432 | 255,775 | 272,615 | 381,794 | 525,831 | 263,743 | 124,269 | 82,343 | 82,343 | 87,766 | 61,356 | 80,512 | 21,188 | 13,631 |
| Total | 3,028,809 | 3,093,269 | 3,532,914 | 5,297,842 | 4,914,174 | 3,831,799 | 3,878,318 | 3,852,008 | 5,044,440 | 4,175,643 | 3,874,858 | 4,980,917 | 4,264,160 | 3,684,074 |
| Total excluding dogfish | 381,084 | 453,790 | 592,440 | 956,460 | 1,671,892 | 1,637,808 | 1,793,021 | 1,401,102 | 1,569,852 | 1,662,863 | 1,535,097 | 1,285,347 | 860,009 | 732,055 |

* includes catches from Canadian waters (approx. 50% during 1983-89)

366-549 m long and 13 meshes deep (mesh sizes between 30.5 and 40.6 cm) (Richards 1987). Vessels were usually those of the halibut fishery, which use hydraulic gear retrievers. Operations were centred in the Santa Barbara-Ventura region and the Channel Islands in waters less than 20 m deep, no more than 1.6 km from shore. In the opinion of Cailliet et al. (1993), the drop in catches since 1986 is due to a combination of declining availability of the species and changes in the market as cheaper imports of shark meat became available. The only regulations applied to this fishery are still those which pertain to the set-net fishery for halibut in California, neglecting the need for separate management of this elasmobranch resource.

A shortfin mako (*Isurus oxyrinchus*) fishery also started in California as a valuable bycatch of the driftnet fishery for swordfish and common thresher shark of the late 70's. Catches increased steadily from 1977 through 1982 when they reached 239 t, then underwent a period of lower catches possibly attributed to changes in fishing strategy or environmental conditions (Holts 1988), but peaked again in 1987 at 277 t. Since then, catches have declined once more (table 2.4). The bycatch of makos in the driftnet fishery is low and since 1988 a closely controlled experimental fishery was started with longlines targeting this species. Under this regime, six vessels using 4.8-8.2 km stainless steel cable longlines near the surface, are allowed to fish in time/area closures away from sport fishing grounds. Additionally, an 80 t TAC has been established and a market for the substantial blue shark bycatch must be developed to utilise this resource. Bycatches of shortfin mako in the driftnet fishery are also allowed. Although the shortfin mako fishery is mainly sustained by very young sharks averaging 9-14 kg dressed weight, there is no apparent decline in the mean size of the catches, populations look healthy and even might be relatively unexploited (Holts 1988, Cailliet et al. 1993).

In addition to the three fisheries mentioned above which constitute the main "new" shark fisheries in the last 15 years on the west coast, many other elasmobranchs are also taken commercially, mainly as a bycatch of other fisheries. Martin and Zorzi (1993) review the skate fisheries of California. Skates (mainly *Raja binoculata, R. inornata* and *R. rhina)* have been fished in California since at least 1916, averaging 96 t and 11.8% of the total commercial elasmobranch catches in California per annum. San Francisco and Monterey are the main landing ports making over 70% of the total. Technical constraints in the processing limit marketable skates to sizes of up to 1 kg, therefore most of the landings of *R. binoculata* and *R. rhina* are composed of immature individuals. Roedel and Ripley (1950) suggested that the skate resource might be underutilised, but it also seems to be presently missutilised. A market for larger skates should be developed in order to optimise use and management of these resources.

Another species of interest is the blue shark (*Prionace glauca).* Holts (1988) and Cailliet et al. (1993) summarise the available information. The blue shark is a major incidental catch of the driftnet fishery of California and a minor bycatch of the set-net fisheries for halibut and angel sharks. Mortality estimates for the driftnet fishery were of 15,000-20,000 (300 t) sharks annually in the early period, although changes in gear design have accounted for reductions in this mortality. The experimental longline fishery for mako sharks also takes incidental catches of blue sharks at a rate of four blue sharks for each mako. Nevertheless, the enforcement of rapid release of live sharks is expected to decrease mortality. A small experimental longline fishery with one vessel took place during 1980-1982 and catches of blue sharks peaked around 90 t in 1980 and 1981 (table 2.4). The main constraint for the development of a large scale fishery for blue sharks is the lack of a market. Blue shark meat is reportedly less palatable than that of other elasmobranchs. Attempts to initiate a fishery for salmon sharks *Lamna ditropis* in Alaskan waters was reported by Paust (1987) but no further records of this were found.

The single most important fishery for elasmobranchs in the history of the west coast was the California fishery for soupfin shark *Galeorhinus galeus* during the 1930's-1940's. Ripley (1946) gives a detailed description of this fishery. Stimulated by the discovery in 1937 that the soupfin sharks of that area were the richest source of high potency vitamin A in the world, the subsequent four years marked a tremendous increase in catches which reached over eight times those of pre-boom levels and averaged approx. 3,400 t/yr. Vessels from the northern halibut fishery switched to shark fishing and in a very short period all sorts of vessels modified their operations and joined the fishery totalling about 600 boats by 1939. Swift changes in gears from drift and set gillnets to machine-handled halibut longlines and back to "diver" gillnets and the posterior mechanisation of their operation occurred in a period of less than 3 years (detailed description of gear in Roedel and Ripley 1950). Northern California was the main fishing area with more than 70% of the catches, although fishing occurred throughout the entire coast mostly within 7.8 km from shore in waters up to 144 m deep. Since 1941, catches plummeted and never recovered their peak levels. The discovery of synthetic vitamin A prevented efforts to revive this fishery to its former glory, although a small fishery has remained up to present times. Catches since 1976 fluctuated between 66 and 253 t/yr (table 2.4). Activities are now centred around San Diego and Orange counties (Holts 1988) where catches are apparently an incidental product of net fisheries for halibut and angel shark. Only the general regulations for the latter fisheries "protect" soupfin shark populations. Holden (1977) estimated the north Pacific unexploited stock at 29,400 t, but it appears that stocks have not yet recovered to the former levels (Holts 1988). However, no recent assessments have been done for this species. Finally, a short lived small-scale harpoon fishery for basking sharks (*Cetorhinus maximus)* took place during the late 40's in Pismo Beach (Roedel and Ripley 1950) but perished also as a consequence of the fall of the liver oil industry.

Mexico.

Since the mid-70's, Mexican elasmobranch fisheries have been the largest in America (fig. 2.2). According to FAO statistics, there has been a general trend of increased catches of elasmobranchs in Mexico, from the typical 5,000 t/yr of the 50's to the recent yields varying around 30,000 t/yr since the early 80's. Judging from the trend of the last ten years, Mexican fisheries for sharks and rays have attained relative stability.

Elasmobranchs are a relatively important resource in Mexico, making 2.36 % of the total national catches during 1987-1991. This is comparable to other major elasmobranch-fishing countries, but is substantially higher than the 0.8 % contribution of elasmobranchs to world fisheries in the last 10 years.

Elasmobranch exploitation in Mexico can be traced back to at least the 1930's, but detailed statistics are difficult to find before the mid-70's. Walford (1935) reports "several tons" of shark fins from the west coast of Mexico being imported to California each year and Ripley (1946) refers to Mexican fisheries supplying shark liver oil to the USA industry. Mazatlan and Guaymas were the main landing points in the west coast. Catches peaked at 9,000 t in 1944 but declined to 480 t in 1953, after the fall of the shark liver oil industry (Castillo

1990). In the east coast during the 40's, a fleet targeting sharks based at Progreso, Yucatan had characteristics similar to the fleet of Salerno, Florida, and caught up to 3,200 t/yr since 1950 (GMFMC 1980).

Mexican fisheries for elasmobranchs are centred on sharks. Batoids are seldom exploited but considerable (and unknown) amounts are discarded in the extensive trawling operations for shrimp fisheries. According to data taken from the Mexican Ministry of Fisheries yearbooks for the period 1977-1991, sharks account for 94.8 % (29,036 t/yr) of elasmobranch catches while batoids only represent 4.2 % (1,272 t/yr).

Because of its larger shoreline, the Pacific coast contributes 60 % of total shark catches while the remaining 40 % comes from the Gulf of Mexico and Caribbean. No data on catches by species are available. Only small sharks (those measuring less than 1.5 m TL when caught and know locally as cazón) and large sharks (those larger than 1.5 m TL) are recorded in the statistics. Large sharks are 60 % of total shark catches and 2/3 of these are caught in the Pacific while only 1/3 are caught in the Gulf of Mexico and Caribbean. The remaining 40 % of the total shark catches are small sharks, 64 % come from the Pacific and 36 % from the east coast. There is some variability in the catches of large and small sharks from each coast, but overall, Mexican fisheries seem to have reached an equilibrium during the last 10 years (fig. 2.5). Meanwhile, batoid catches are slowly and steadily expanding.

Mexican shark fisheries are largely artisanal, multispecies, multigear fisheries. Bonfil et al. (1990), Castillo (1990) and Bonfil (*in press*) summarise most of the available information on elasmobranch fisheries in Mexico. It is estimated that approximately 2/3 of the shark catch is taken by small-scale fisheries. Vessels are generally fibreglass boats 7-9 m long with outboard motors using either gillnets or longlines depending on the customs of each region. Some vessels of 14-20 m are also used whereas only a few vessels in excess of 20 m take part in the fishery. Significant quantities of sharks and rays are also taken as incidental catches of large-scale trawl fisheries for shrimp or demersal fishes in both sides of the country. Large scale fisheries for tunas and billfishes in both coasts also contribute to the total catches. Sharks and rays are traditionally used for food in Mexico, either fresh, frozen or more commonly, salt-dried. Shark fins are an important export, hides are also intensively utilised and most offal is burned down to fish meal.

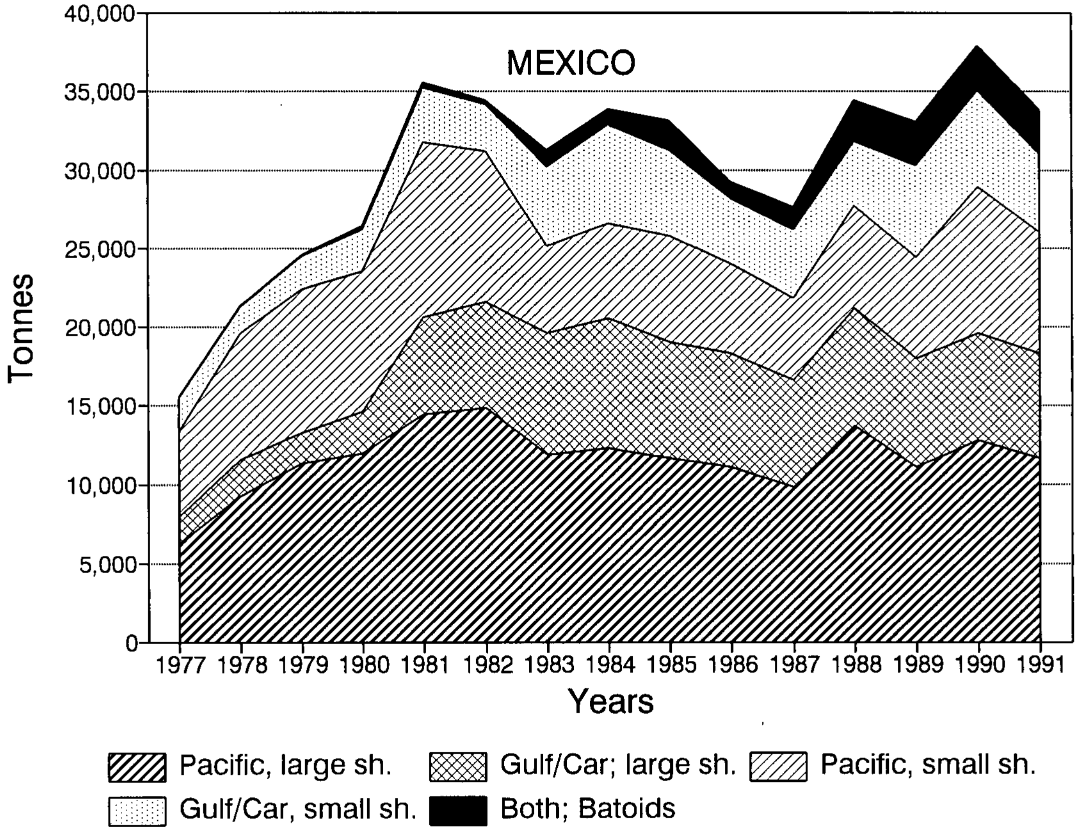



Figure 2.5 Elasmobranch catches in the Pacific and Gulf of Mexico/Caribbean coasts of Mexico during 1977-1991 (sh = sharks). (Data from Secretaría de Pesca, México).

The main fishing grounds in the Pacific are centred in the Gulf of California in the north and the Gulf of Tehuantepec in the south. However, most of the available information about these fisheries comes from the northern coast. Little is know from the shark fisheries in the Gulf of Tehuantepec apart from the total catches. In the northern region, sharks are mainly caught with monofilament longlines of 1-2 km and approximately 350 hooks, although small catches are taken with various gillnets of up to 2 km in length. There are reports that some 17 vessels, 44 m long and using longlines of up to 2,000 hooks targeted sharks and billfishes in the Pacific coast during 1987. It is unknown if these vessels are still in operation. Holts (1988) states that a similar number of Japanese-Mexican joint venture longliners caught 234 t/yr of sharks in Baja California during 1981-1983.

On the east, fishing grounds span the entire coastline. During 1976-1988, Veracruz and Campeche shared 58 % of the total shark catch while Tamaulipas and Yucatán made another 30 %. Longlines are preferred in Veracruz and Tamaulipas. Gillnets of 11-40 cm mesh size are the main fishing gear in the Bank of Campeche. Additionally, there is a substantial bycatch of mainly juvenile sharks in the semi-industrialised longline fisheries for red grouper and red snapper of the Campeche Bank but no estimates of the bycatch are available. The species caught in the different regions of the Mexican coast and the structure of such catches are only partially known. Most research has been done in the mouth of the Gulf of California in the west coast and in the southern States of Campeche, Yucatan and Quintana Roo in the east coast. Important landings in other areas of both coasts have been very poorly studied.

At least 44 species of sharks are reported in the commercial catches of Mexico. Available information indicates 12 as the most important for their contribution to the catches in the area of La Paz, Baja California Sur and Sinaloa, whereas there are 15 main species in the Gulf of Mexico and Caribbean (table 2.5). Most of the catches of large sharks consist of *Carcharhinus* spp, *Sphyrna* spp and other carcharhinids, while the small shark catches are a mixture of *Mustelus* spp. and *Rhizoprionodon* spp., with juveniles of the large sharks sometimes contributing an important part of the total. For the Sinaloa coast in the central Pacific *Rhizoprionodon longurio, Sphyrna lewini, Nasolamia velox, Carcharhinus limbatus, C. falciformis, C. leucas* and *Galeocerdo cuvieri,* are the most important species. Galván-Magaña et al. (1989) report *Mustelus lunulatus*, *Heterodontus mexicanus* and *Sphyrna lewini*

Table 2.5 Shark species found in the commercial fisheries of Mexico.

| FAMILY | | SPECIES | PACIFIC | GULF OF MEXICO /CARIBBEAN |
|---|---|---|---|---|
| Hexanchidae | 1 | *Heptranchias perlo* | | X |
| | 2 | *Hexanchus griseus* | | X |
| | 3 | *Hexanchus vitulus* | | X |
| Echinorhinidae | 4 | *Echinorhinus cookei* | X | |
| Squalidae | 5 | *Centrophorus granulosus* | | X |
| | 6 | *Centrophorus uyato* | | X |
| | 7 | *Squalus cubensis* | | X |
| | 8 | *Squalus mitsukurii* | | X |
| Squatinidae | 9 | *Squatina californica* | X* | |
| Heterodontidae | 10 | *Heterodontus mexicanus* | X* | |
| Ginglymostomatidae | 11 | *Ginglymostoma cirratum* | X | X* |
| Rhiniodontidae | 12 | *Rhiniodon typus* | X | X |
| Alopiidae | 13 | *Alopias vulpinus* | X* | |
| | 14 | *Alopias superciliosus* | X | X |
| Lamnidae | 15 | *Isurus oxyrinchus* | X | X |
| Triakidae | 16 | *Mustelus californicus* | X | |
| | 17 | *Mustelus canis* | | X* |
| | 18 | *Mustelus lunulatus* | X* | |
| | 19 | *Mustelus sp. ?* | | X |
| | 20 | *Triakis semifasciata* | X | |
| Carcharhinidae | 21 | *Carcharhinus acronotus* | | X* |
| | 22 | *Carcharhinus altimus* | X | X |
| | 23 | *Carcharhinus brevipinna* | | X* |
| | 24 | *Carcharhinus falciformis* | X* | X* |
| | 25 | *Carcharhinus leucas* | X* | X* |
| | 26 | *Carcharhinus limbatus* | X* | X* |
| | 27 | *Carcharhinus longimanus* | | X |
| | 28 | *Carcharhinus obscurus* | X | X* |
| | 29 | *Carcharhinus perezi* | | X |
| | 30 | *Carcharhinus plumbeus* | | X* |
| | 31 | *Carcharhinus porosus* | X | X |
| | 32 | *Carcharhinus signatus* | | X |
| | 33 | *Galeocerdo cuvieri* | X* | X* |
| | 34 | *Nasolamia velox* | X* | |
| | 35 | *Negaprion acutidens* | X | |
| | 36 | *Negaprion brevirostris* | | X* |
| | 37 | *Prionace glauca* | X* | |
| | 38 | *Rhizoprionodon longurio* | X* | |
| | 39 | *Rhizoprionodon terraenovae* | | X* |
| Sphyrnidae | 40 | *Sphyrna lewini* | X* | X* |
| | 41 | *Sphyrna media* | X | |
| | 42 | *Sphyrna mokarran* | X | X* |
| | 43 | *Sphyrna tiburo* | X | X* |
| | 44 | *Sphyrna zygaena* | X | |

* Main species in the commercial catches.

as the most important sharks in the area of La Paz, B.C. Experimental catches of longliners in the Pacific caught mainly pre-adult and adult *Alopias vulpinus* and *Carcharhinus limbatus* (Velez et al. 1989). For the east coast, the most important species are *Carcharhinus falciformis, C. leucas, C. obscurus, C. plumbeus, C. limbatus, Rhizoprionodon terraenovae, Sphyrna tiburo, Mustelus canis*, *C. brevipinna, Negaprion brevirostris, Sphyrna mokarran, Sphyrna lewini, Galeocerdo cuvieri* and *Ginglymostoma cirratum.* With the exceptions of *C. obscurus*, *C. plumbeus* and *Ginglymostoma cirratum*, all the important species of the east coast are known to be heavily exploited as juveniles and sometimes even as newborns at least in some part of their range.

There are only a few preliminary assessments of the status of some shark stocks for the east coast. Alvarez (1988) reports that according to surplus production models, the stocks of *Sphyrna tiburo* and *Rhizoprionodon terraenovae* in Yucatan are close to optimal exploitation levels; results of the yield-per-recruit model suggest *Sphyrna tiburo* is at the optimum exploitation level whereas *Rhizoprionodon terraenovae* seems to be already overexploited. For the production models, catch and effort were estimated in a very rough way and for the dynamic model, growth and mortality were estimated via length frequency analysis. Bonfil (1990), estimated growth via vertebrae readings and using the yield-per-recruit model diagnosed growth overfishing for the *Carcharhinus falciformis* stock of the Campeche Bank. This results mainly from the very high bycatches of newborns and juveniles of this species in the local red grouper fishery.

There have been a number of permanent research programmes for shark fisheries in Mexico since the early 80's. Despite this, to date there is no specific management for elasmobranch fisheries in Mexico. A number of concerns have been expressed about some undesirable practices in the fisheries. At least, *Carcharhinus falciformis, C. acronotus, Rhizoprionodon terraenovae* and *Sphyrna tiburo* are being heavily exploited as juveniles in Campeche and Yucatan, hence opening the possibility of a future collapse of the stocks. Additionally, there are suggestions that strong decreases in the abundance of juveniles of *C. leucas, C. limbatus, C. acronotus, C. perezi* and *Negaprion brevirostris* have occurred in some coastal lagoons of the Yucatan Peninsula due to heavy fishing with set nets (Bonfil *in press*). It is very likely that this situation is commonplace in most coastal lagoons along the coast of Mexico. Additionally, the killing of large quantities of pregnant females of

*Rhizoprionodon longurio* in Sinaloa, on the west coast, is another cause for concern. Although information is limited, it is likely that many stocks in the Gulfs of California and Tehuantepec are close to the optimum exploitation level or even overfished. However, no assessments are known to date in those areas. Limited or non-existent information about the size of the stocks and about the actual levels of mortality makes the adequate appraisal of the status of Mexican fisheries difficult.

As in other countries, socio-economic and health problems related to the fisheries further complicate the management of elasmobranchs in Mexico. The chances of curtailing the fishing of juvenile sharks in Mexico will be constrained by the problems related to the artisanal nature of many of the fishing fleets (loss of income for large numbers of fishermen) and the high esteem that small sharks have on the Mexican table. The higher concentration of heavy metals in older sharks also makes the harvesting of juveniles preferable.

Peru.

From the mid-sixties and until very recently, the elasmobranch catches of Peru were the third largest in America and contributed 2.71 % of the world elasmobranch catch. Nevertheless, elasmobranchs are of minor importance in Peruvian fisheries and represent only 0.29 % of the total fishery production (table 2.2). The elasmobranch fisheries of Peru had a fairly steady trend of slow development in the 50's and early 60's. Since the mid-60's catches have oscillated around 18,000 t, peaking at more than 30,000 t in 1984 and unexpectedly crashing in 1990-1991 (fig. 2.2). There may be a link between recently declining elasmobranch catches and the eruption of cholera in Peru during 1990.

According to FAO statistics, elasmobranch yield in Peru is strongly dominated by smooth-hounds. During the period 1977-1991, smooth-hounds of the genus *Mustelus* were the most important species in the elasmobranch catches making 56 % (10,219 t/yr) of the total and accounted for 25,000 t in 1984 when record elasmobranch catches of Peru reached 34,400 t (fig. 2.6). Several unspecified rays make 25 % (4,640 t/yr) of the total catches. Their landings have increased significantly since 1984, making them the second most important elasmobranch group in Peru. *Rhinobatos planiceps* and angel sharks *Squatina* spp. are also important species in the catches with averages of 10 % (1,908 t/yr) and 3 % (560 t/yr) respectively. The yields of these two groups showed variable trends in this period. An assorted group of elasmobranchs made the remaining 6 % (1,133 t/yr). Apart from FAO statistics, nothing else is known about the elasmobranch fisheries of Peru.

Brazil.

Brazilian elasmobranch catches are the third highest in America, after Mexico and the USA. It appears that Brazilian elasmobranch fisheries have attained a relative stability. After an slow but steady start through the sixties and a brief fall in the 70's, the catches of sharks and rays from Brazil show a major leap in the early 80's. Yields have varied since then, up to a 30,000 t maximum (fig. 2.2). Sharks and rays contributed 3 % to the total fisheries of Brazil during 1987-1991 while making 3.69 % of the world catches of elasmobranchs (table 2.2).

Brazilian fisheries statistics do not differentiate elasmobranchs by species. At least 30 elasmobranchs are common in the commercial catches in the southeast, but most of the landings come dressed without guts, head or fins, making it difficult to sort by species (Tomas 1987). Some of the species reported for the commercial catches are: *Mustelus schmitti, Galeorhinus galeus, Prionace glauca, Isurus oxyrinchus, Squatina guggenheim, Squatina* sp, *Pristis* spp., *Rhinobatos percellens, R. horkelii, Dasyatis* spp, *Gymnura* spp and *Myliobatis* spp.

According to FAO data, during the period 1977-1991, Brazilian landings were dominated by an assorted group of species corresponding to 72 % (17,919 t/yr) of the elasmobranch catch. Yields for this group of elasmobranchs grew rapidly from less than 1,000 t in 1978 to more than 23,000 t in 1982 and have remained close to 20,000 t/yr since then (fig. 2.7). All the sharks known to occur in Brazilian catches are included in this group. According to Batista (1988), landings of *Galeorhinus galeus* have increased since 1970 due to growing activities of trawlers in south east Brazil. The second most important group during this period were the skates and rays contributing 17 % (4,254 t/yr) of the catches. Landings of this group expanded slowly, as well as those of guitarfishes *Rhinobatos* spp. which averaged 7 % (1,683 t/yr) of the total elasmobranch catch. Small catches of sawfishes *Pristis* sp. have been steadily landed averaging 4 % (1,014 t/yr) of the catches.

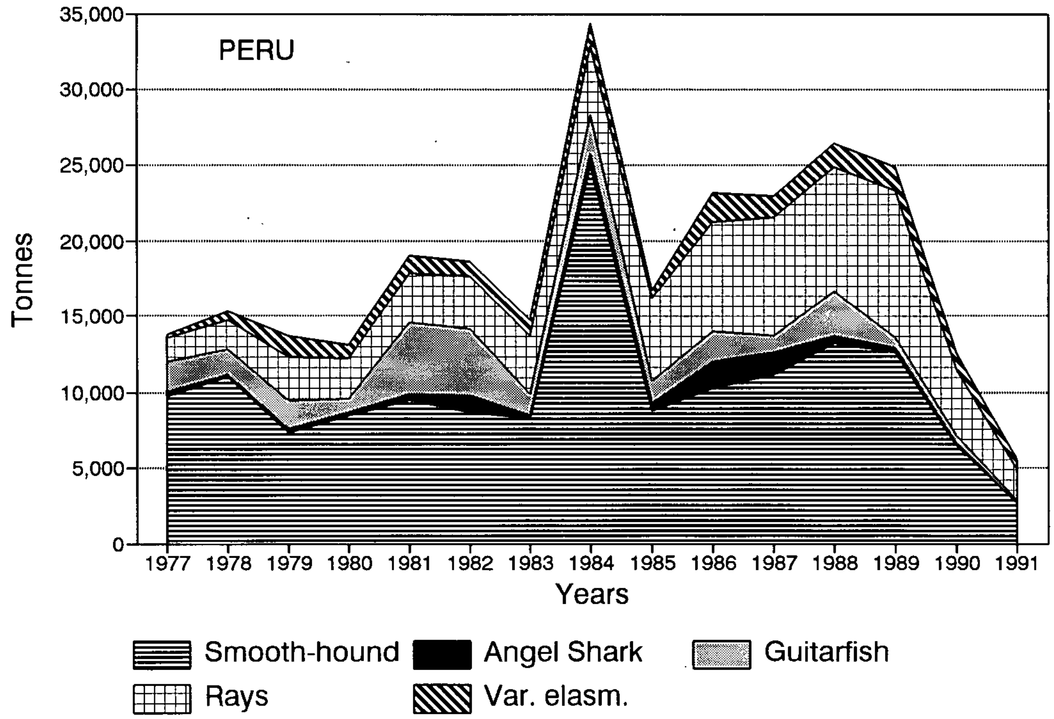



Figure 2.6 Elasmobranch catches of Peru, by species groups, during 1977-1991 (Data from FAO).

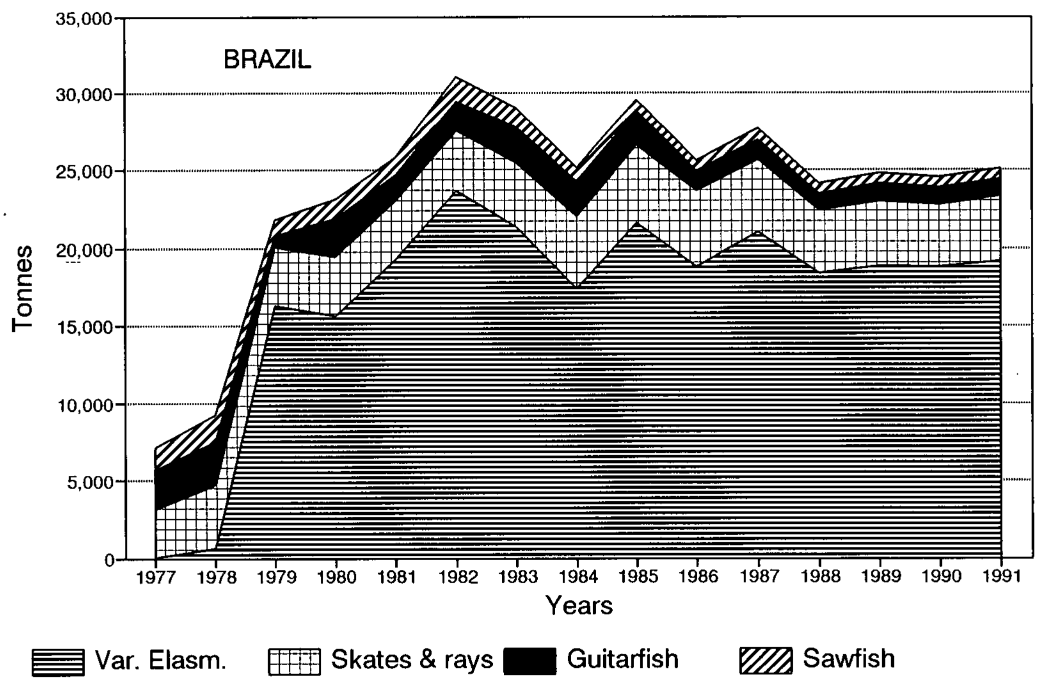



Figure 2.7 Elasmobranch catches of Brazil, by species groups, during 1977-1991. (Data from FAO).

Vooren and Betito (1987) report on at least 25 species of small sharks and 24 of batoids for waters less than 100 m deep in the southeastern continental shelf. Swept area biomass estimates indicate 20,000 t available in winter and 13,000 t in summer of which 90% are composed of 16 small sharks and 8 batoid species of commercial value. Apparently, the only traditional use for elasmobranchs in Brazil has been for food, but Göcks (1987) and Jacinto (1987) not some efforts for the utilisation of hides and other parts.

In the north of Brazil, at least two kinds of fisheries land elasmobranchs (R. Lessa, pers. comm. February 1992). An industrial longline fishery for tunas with up to 50% bycatches of sharks, captures mainly *Prionace glauca, Carcharhinus longimanus, Carcharhinus* spp., *Sphyrna* spp., *Isurus* spp., *Alopias* spp., *Pseudocarcharias kamoharai* and *Galeocerdo cuvieri.* This fishery landed an average of 144 t/yr of sharks between 1985-1990. About 60 % of these were sharks less than 1.5 m TL. Artisanal fisheries for *Cynoscium acoula* and *Scomberomorus* spp. catch *Carcharhinus porosus, Rhizoprionodon* spp., *Sphyrna* spp., *Isogomphodon oxyrhynchus* and *Pristis peroretti.* There is a high incidence of juveniles in this fishery which is carried out using small driftnets of about 1 km long and 6 m deep. On the north shore, between the Amazon river and Recife, elasmobranch catches may be as high as 60% of the total in this artisanal fishery. Incidental catches of small sharks and rays in the *Brachyplatystoma,* shrimp and snapper fisheries of the north of Brazil are reported by Evangelista (1987). Apparently most of the bycatches were formerly discarded but are now beginning to be utilised.

Vooren et al. (1990) summarise information on demersal fisheries for elasmobranchs during 1973-1986 on the continental shelf off the southern port of Rio Grande. Elasmobranchs account for 7.3 % of the total catches, 13.1 % of the otter trawlers catches, 7.1 % of the paired trawlers catches and 5.4 % of small-scale fisheries catches. Otter trawling operations are carried out with 440-480 HP boats in 11-13 day trips at depths between 40-100 m, while paired trawling is done by 340-370 HP boats in 9-11 day trips at depths less than 40 m. Small-scale fisheries include beach seining and trammel nets in waters less than 10 m deep and gillnetting by 11-16 m boats with 100-130 HP motors in waters 8-40 m deep. Small sharks average 46.3 % of elasmobranch catches, while angel sharks, guitar fishes and rays account for 24.85 %, 24.5 % and 5 % respectively. *Mustelus schmitti* and *Galeorhinus galeus* make most of the catches of "caçoes" or small sharks, which show increased landings from 1,414 t in 1973 to 3,217 in 1986, but, according to SUDEPE (1990) landings dropped to 2,023 t in 1989. The proportion of small sharks in the catches of the small-scale and pair trawler fishery increased during this period but decreased in the otter trawl fishery. This resulted in an almost equal proportion of landings by each type of fishery in 1983-1986. The CPUE of small sharks in t/trip of both types of trawlers tended to increase throughout the period of the study. Angel shark (*Squatina guggenheim* and *Squatina* sp.) landings increased from 822 t in 1973 to 1,777 t in 1986. As with the small sharks, the proportion of catches contributed by small-scale fisheries and pair trawlers increased while that of otter trawlers decreased. Still, about 50 % of the total landing of angel sharks comes from the latter. While CPUE of angel sharks in otter trawlers showed an overall increase, paired trawlers' CPUE increased until 1983 and decreased afterwards. Landings of guitar fish *Rhinobatos horkelii,* varied between 600 t and 1,925 t. Most of these came from the small scale fisheries (50 %) and paired trawlers (32 %), while otter trawlers contributed very small catches (13 %). Data of CPUE showed a slight decrease up to 1982 for both types of trawlers, increasing to 1984 and falling again afterwards. Landings of rays, mainly genus *Dasyatis* and *Gymnura* and to a lesser extent *Myliobatis*, grew from 36 t in 1973 to 484 t in 1986. Small-scale fisheries averaged 18 % of these catches, paired trawling 53 % and otter trawling 34 %. CPUE for rays in trawl fisheries were variable with an increasing trend.

The apparent decline of some of these populations in the last period of the above study seems to be confirmed by a switch from trawling to bottom longlines and gillnets (the latter specifically aimed at *Squatina* and *Galeorhinus*) which started in 1986 due to decreasing CPUE's. This switch was also coupled with additional fishing for angel sharks by shrimp trawlers from other areas during the shrimp off season (C.M. Vooren, Universidad de Rio Grande, pers. comm. Dec. 23 1991).

Amorim and Arfelli (1987) and Arfelli et al. (1987), report some bycatches of large sharks taken in south and southeast waters by tuna longliners. *Prionace glauca* (accounting for 33 % of total catches of this fleet in 1985) and *Isurus oxyrinchus* (accounting for 3.2 % of total catches of this fishery 1971-1985 ) are caught throughout the year but mainly during April-July and May-November respectively. Landings of blue sharks are composed mainly of 20-40 kg sharks dressed weight (no head, fins or guts) and accounted for 553 t and 462 t in 1984 and 1985 respectively. Blue shark CPUE has varied from 0.4 kg/100 hooks in 1971 (when their capture was avoided by skippers) to 27.6 kg/100 hooks in 1985. Shortfin mako catches varied between 21 t (1971) and 73 t (1981), their mean weight in the catch varying between 42 kg and 60 kg throughout 1985. Their meat is the most valued among elasmobranchs in Brazil, and is used locally as well as exported to USA.

A good deal of research on elasmobranchs is done by several Brazilian Universities, governmental and non-governmental organisations. However, according to Lessa (pers. comm., op.cit.), at present there are no management measures for elasmobranchs in Brazil, although some local groups intend to raise governmental attention into the status of these fisheries. Some plans to implement reporting by species are also underway. Lessa points out that elasmobranch stocks exploited in the north coast artisanal fishery are thought to be underexploited, those utilised by the tuna longline fishery are sustainably exploited, whereas the south Brazil demersal stocks are almost surely overexploited.

Argentina.

Argentina has some of the few expanding fisheries among major elasmobranch-fishing countries in America. After a temporary drop in the late 40's, attributed to the general collapse of shark liver oil fisheries around the world, shark and ray yield had a slow but steady growth from the early 50's to the mid 60's (fig. 2.2). Since 1967, yields fluctuated closely to 10,000 t but have increased since 1981. Despite the relatively low catches, which account only for 2.54 % of the world elasmobranch catch during 1987-1991, elasmobranchs are reasonably important for Argentinean fisheries contributing 3.19 % of the total yields during this period. This is the highest relative importance for elasmobranchs in American major elasmobranch-fishing countries.

During the period 1977-1991 the most important species in the elasmobranch catches were: *Mustelus schmitti* which averaged 49 % (6,790 t/yr) of the total elasmobranch catch; several rays at 20 % (2,722 t/yr), unclassified elasmobranchs with 23 % (3,160 t/yr) and elephant fishes (*Callorhinchus* sp.) with 8 % (1,048 t/yr). Of these groups, the smooth-hounds and "various elasmobranchs" had a growing trend in catches while the elephant fishes and rays had a tendency to decrease. Argentina is one of the few countries with important catches of chimaeriformes in the world (fig. 2.8).

Crespo and Corcuera (1990) give a very detailed description of the fisheries for sharks of Claromeco and Necochea, Buenos Aires Province. In this northern Argentina fishery, *Galeorhinus galeus, Mustelus schmitti, Carcharias taurus* and *Squatina argentina* are caught with gillnets. About 23 wooden and iron vessels from 8-44.9 m in length take part in the fishery. They use nylon monofilament gillnets (2-3 mm twine) with mesh sizes 19-21 cm and dimensions 55-71 m long 3.8 m deep per panel (8-25 panels). These gillnets are set on the bottom between 0.5 and 25 nm from the coast in depths varying from 2-70 m. Typical about 6-15 *Squatina argentina* and 1-20 of the other sharks are caught per panel. Ex-vessel prices are US$3-4/kg for undamaged *Galeorhinus* destined for export (mainly to Italy) and US$1-2.5/kg for damaged ones that are consumed salt-dried in the local market. In an unusual note, these authors report extensive damage to shark catches by marine mammals! Sea lions bite off the belly of entangled sharks eating only the liver.

Menni et al. (1986) indicate the presence of more sharks species in northern Argentina. In addition to the species mentioned above, they report *Mustelus canis, M. fasciatus, Squalus blainvillei, S. cubensis* and *Notorhynchus cepedianus* in the commercial catches of Buenos Aires province. *Mustelus schmitti* accounts for 92 % of their shark samples, at commercial landing sites. The remaining species are less than 1 % of the shark catches except *S. cubensis* which made up 2 %. Government statistics of shark landings at Mar del Plata port averaged 5,890 t during 1971-1980, this is about 1/2 of the average total elasmobranch catch of Argentina during that period. About 93 % of this catch is made of 'gatuzos' (predominantly *Mustelus schmitti,* with some quantities of *M. canis* and small numbers of *M. fasciatus).* 'Cazones' (mainly *Galeorhinus galeus* but including some large *M. canis)* contribute the remaining 7 %. Apparently, the remaining species are not recorded in the statistics.

2.2.2.2 Europe.

Norway.

Norway has had some of the most important shark fisheries in the North Atlantic. Norwegian commercial fisheries for elasmobranchs have been quite variable since the end of World War II, with an increasing trend up to 1963, followed by a general decrease to levels around

7,500 t/yr since 1981 (fig. 2.2). Catches rose again in the last three years on record. Judging from recent trends, elasmobranchs are not an important fishery resource for Norway. Sharks and rays represent only 0.44 % of the total fisheries production of this country. Moreover, Norwegian elasmobranch fisheries only contribute 1.21 % to the world elasmobranch yield 1987-1991 (table 2.2).

The largest part of the elasmobranch catches has typically been spiny dogfish *Squalus acanthias.* Nevertheless, there were important fisheries for porbeagles in the 60's and for basking sharks until the mid 80's. While marketing and economical constraints have traditionally inhibited basking shark fisheries (Maxwell 1952; O'connor 1953; Kunslik 1988), apparently the porbeagle, *Lamna nasus,* fishery declined at least in part as a result of over-exploitation (Gauld 1989, Myklevoll 1989a, Anderson 1990).

Norwegian elasmobranch fisheries seem to be recovering after a prolonged decline. For the first time in almost 20 years, catch trends are on the rise. FAO data for the period 1978-1991 (fig. 2.9) show catches of spiny dogfish declining from more than 12,000 t in 1978 to 2,986 t in 1986 then rise up to 9,627 t in 1991 and averaging 5,715 t/yr (53 % of elasmobranch catches for this period). Catches of basking sharks, *Cetorhinus maximus,* show a pattern similar to that of spiny dogfish catches, although their recovery is more modest. Basking shark catches fell from 11,335 t in 1979 to only 352 t in 1987, but were of 1,932 t in 1990 and averaged 3,929 t/yr (36 %) during this period. Rays catches are fairly stable at around 1,115 t/yr (10 %). Small quantities of porbeagles are still caught, averaging 67 t/yr (less than 1 %).

Although there are some incongruencies among different sources of data on the Norwegian directed fishery for porbeagles of the 60's (Gauld 1989; Anderson 1990), it is clear that this fishery caught large amounts of sharks. The summary of this fishery is based on Aasen (1963) and Myklevoll (1989a). Operations started as a coastal activity but, since 1930, the fishing grounds steadily expanded from Norwegian waters northwest to the Orkney-Shetland area and the Faroe Islands, then southerly into Irish waters and finally stretched to the Atlantic coasts of Canada and northeastern USA. Distant water operations were carried out by specialised freezer vessels 43-50 m long, deploying longlines with up to 5,000 hooks in waters 10-30 m deep. Sharks less than 10 kg in weight were discarded because of a lack

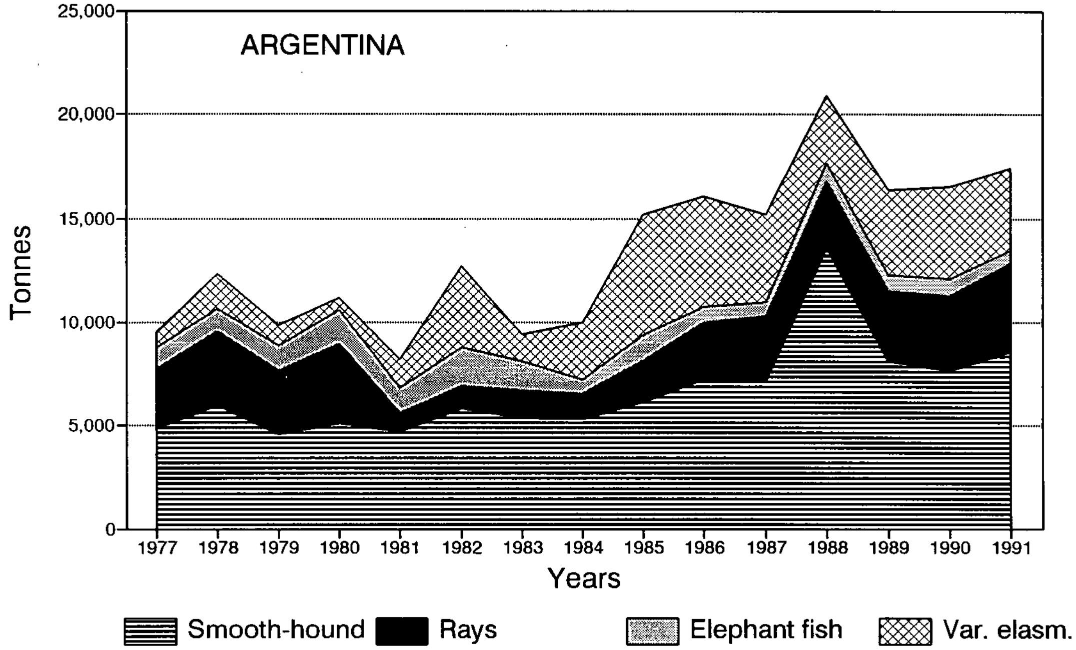



Figure 2.8 Elasmobranch catches of Argentina, by species groups, during 1977-1991. School shark catches, too small to show (Data from FAO).

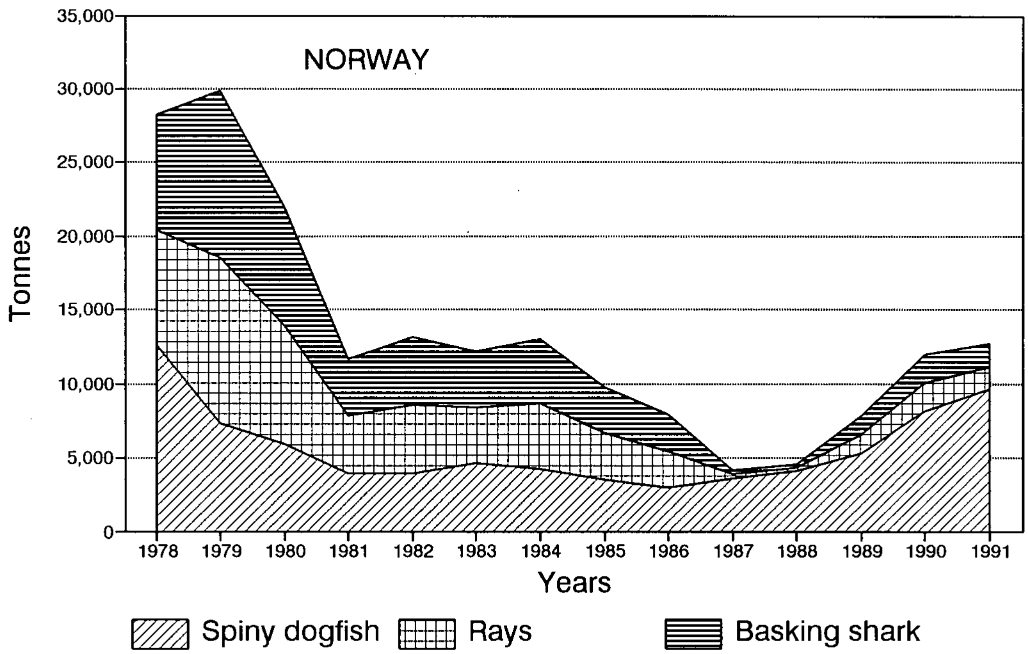



Figure 2.9 Elasmobranch catches of Norway, by species groups, during 1978-1991. Porbeagle catches, too small to show (Data from FAO).

of market. The coastal fleet was composed of wooden boats 23-30 m long which kept the catch on ice. Dressed carcasses of porbeagles were exported frozen to Italy, while fins were marketed to the Far East. Once the NW Atlantic porbeagle stocks reached unprofitable size in 1965, the fleet moved out to catch mako sharks in North West Africa. Presently, only bycatches of porbeagles from purse-seining, trawling and gillnet fisheries are landed. Norwegians do not even take their 200 t porbeagle TAC in EC waters.

The basking shark fishery (documented by Kunzlik 1988 and Myklevoll 1989b) has roots in the 16th century when the dried strips of meat were used as food. This has traditionally been an important directed fishery in Norway. Inspired by a great demand for liver oil, the big expansion of the fishery started in 1960. Small wooden vessels 15-25 m long and armed with harpoons, operated mainly during April-August. Experiments to use the flesh of basking sharks into fishmeal and to give various uses to the hides failed. Consequently, in an attitude comparable to that of the "finning" fishermen of other areas, Norwegian fishermen took just the liver for oil extraction and discarded the carcasses at sea. Lately they also take the fins and export them to the Orient. During the period 1959-1980, catches ranged between 1,266 and 4,266 sharks per year, but have declined since. EEC agreements with Norway limited their catches to a TAC of 400 t/yr of livers since 1978. This corresponds approximately to 2,400 t/yr whole weight considering livers to be 1/6 of whole weight. Socio-economic constraints including limited markets and an ageing fleet, and the erratic distribution of the sharks are identified as reasons for the decline of the fishery. The Norwegian fishery for basking sharks does not even take the allowed catch in EEC waters. The oil from the livers is sold for extraction of squalene, a hydrocarbon used in cosmetic and aviation industries, but since richer sources have been found in deep-sea sharks of the genus *Centrophorus*, the market for basking sharks is shrinking. In general, the dynamics of Norwegian elasmobranch fisheries seem to be strongly influenced by economic and social factors (Myklevoll 1989a, 1989b, 1989c). Many of these fisheries in Norway have declined in part or totally because of reasons external to the dynamics of the resource (e.g. market forces).

Holden (1977) and Myklevoll (1989d) summarise most of the Norwegian fishery for spiny dogfish *Squalus acanthias* in the northeast Atlantic, which dates back to 1931. Expansion of the markets led to catches of 8,767 t by 1937, and a peak of almost 34,000 t in 1963.

Since then, catches have slowly fallen to a level of less than 6,000 t in the 80's. During the period 1950-1970, Norwegian longliners fished mainly in their coastal waters during winter and in Scottish waters during summer-autumn. The fishery aimed at exporting most of the catches to fish and chips shops in England. Until the early 70's, this fishery constrained the expansion of the British fishery, due to more competitive products with larger size, better appearance and lower price. In recent years, the migration of large numbers of spiny dogfish into unusually northern parts of Norway has produced an incentive for the fishery. This might account for the increase in catches observed during 1989-1991.

During the first half of this century, Norway had a fishery for greenland sharks, *Somniosus microcephalus*, both as a specialised activity and in combination with sealing. Judging from the data reported by Myklevoll (1989c), this fishery peaked in 1917, when 17,049 hectolitres of livers were landed. Probably because of falling market prices for the product, the fishery ceased in 1960. Fisheries for skates and rays have never been developed as a targeted activity in Norway, all catches are incidental to spiny dogfish, ling, halibut and trawl fisheries (Myklevoll 1989e). Skates and rays of no commercial value and small specimens are commonly discarded

Despite having developed several specific fisheries for sharks, Norwegian research in elasmobranchs has been relatively poor. Out of the three most important shark fisheries of Norway (spiny dogfish, porbeagle and basking sharks), only the spiny dogfish was studied in any dept in a research programme which lasted from 1958 to 1980. This effort produced the first known assessment of an elasmobranch fishery (Aasen 1964). Aasen estimated a maximum equilibrium yield of 50,000 t/yr for what he considered a single stock of spiny dogfish for Northern and Western Europe. By 1961, this yield level was already surpassed. Porbeagles were briefly studied while the fishery was in expansion and this produced one of the first attempts to estimate growth in sharks from vertebral rings (Aasen 1963). Only very limited research was ever done on basking sharks.

There is evidence that Norwegian vessels take part in the orange roughy fisheries of New Zealand. However no details about these activities could be found. The final use given to the probably large bycatches of deep sea sharks in this fishery is unknown (see section 2.2.3.4.).

Former USSR.

Although the USSR does not exist any more as such, the elasmobranch fisheries of what used to be the Soviet Union are treated here because of the importance of its catches. For the sake of simplification, the name 'former USSR' will be used.

Former USSR fisheries for elasmobranchs were not recorded separately from the rest of the fisheries catches of this country before 1964 in FAO yearbooks. Since the beginning of records, the catches soared reaching a total of 59,000 t in 1975, only to fall down as sharply as they soared to a level of about 20,000 t in 1977. Since then, catches have been quite unstable, varying roughly between 10,000-20,000 t/yr (fig. 2.2). Due to the disappearance of the Soviet Union, catches plummeted in 1990-1991. Because of the huge fisheries production of the former USSR, elasmobranchs contributed only 0.11 % of the total catches for 1987-1991, which is the lowest among major elasmobranch- fishing countries. The former USSR contribution to world elasmobranch fisheries was of only 1.75 % in the same period.

The elasmobranch catches of the former USSR were made up from contributions of its enormous fishing fleet, which worked around the world. A great variety of species are masked under the two main headings that were reported: rays and various elasmobranchs. The ever changing characteristics of former USSR fisheries which depended largely on agreements with various nations, makes their analysis difficult.

Data from FAO (fig. 2.10) show that from 1978 to 1991, rays accounted for 66 % (8,761 t/yr) of the total former USSR elasmobranch catches, while various elasmobranchs represented 31 % (4,109 t/yr). Small catches of *Squalus acanthias* accounted for the remaining 3 % of the total (327 t/yr). Most of the elasmobranch catches of the former USSR were probably taken in large trawling operations as suggested by their large catches of batoids. Rays were taken mainly in FAO areas 21 (37 %), 47 (26 %), 27 (15 %) and 37 (10 %), with the remaining (12 %) taken in areas 34, 41, 51 and 71. Catches of various elasmobranchs came chiefly from areas 37 (37 %), 47 (31 %) and 34 (25 %), with the rest of catches (7 %) contributed by areas 27, 51, 71 and 81. Catches of these two groups in Area 37 correspond to thornback ray *Raja clavata* and spiny dogfish *Squalus acanthias* fisheries in the Black

Sea.

Ivanov and Beverton (1985) indicate that Crimean and Caucasian fishermen have specialised fisheries for thornback ray and spiny dogfish in the Black Sea. Thornback rays are fished with baited longline and incidentally caught in bottom gillnets set for spiny dogfish. The latter is also taken by trawl in the northwestern coast of the Black Sea and by bottom longlines and fixed nets along the coasts of Crimea and Caucasia. After the continuous shrinkage of elasmobranch catches in former USSR fisheries until 1982, catches (mainly of batoid fishes) were slowly increasing again, when political events practically shut down all fisheries.

United Kingdom.

The United Kingdom has one of the most stable elasmobranch fisheries in the world. The records show an almost steady trend of catches slowly decreasing from 30,000 t/yr in the early post-war years, to the current level of about 22,000 t/yr (fig. 2.2). During 1978-1991 total elasmobranch catches varied between 20,000 t and 25,000 t mainly because of changes in the catches of spiny dogfish *Squalus acanthias* which averaged 63 % (13,820 t/yr) of total elasmobranch catches (fig. 2.11). Almost 47 % percent of spiny dogfish catches during this period were caught in England and Wales, with an equal amount caught in Scottish waters, while the remaining 6 % came from Northern Ireland. Catches of rays averaged 36 % (7,877 t/yr) of all elasmobranchs and have remained fairly constant with a slight tendency to increase. Approximately 49 % of the ray catches are taken in Scotland and the same amount in England-Wales, while Northern Ireland contributes only about 2 %. Less than 1 % of the total elasmobranch catch of the UK is made up of Scyliorhinids, Squaloids and unspecified elasmobranchs. As a group, chondrichthyans are relatively important to UK fisheries making up 2.63 % of the total catches during 1987-1991.

Holden (1977) summarises information for the spiny dogfish (*Squalus acanthias*) fishery. This species has been fished in England since the beginning of the century but catches did not exceed 2,850 t until 1931; Scottish catches appear on record in 1954. Total spiny dogfish catches in the UK remained between 6,000-10,000 t/yr during the 60's and peaked at 19,400 t in 1978. During 1950-1970 most of the spiny dogfish were caught in amounts

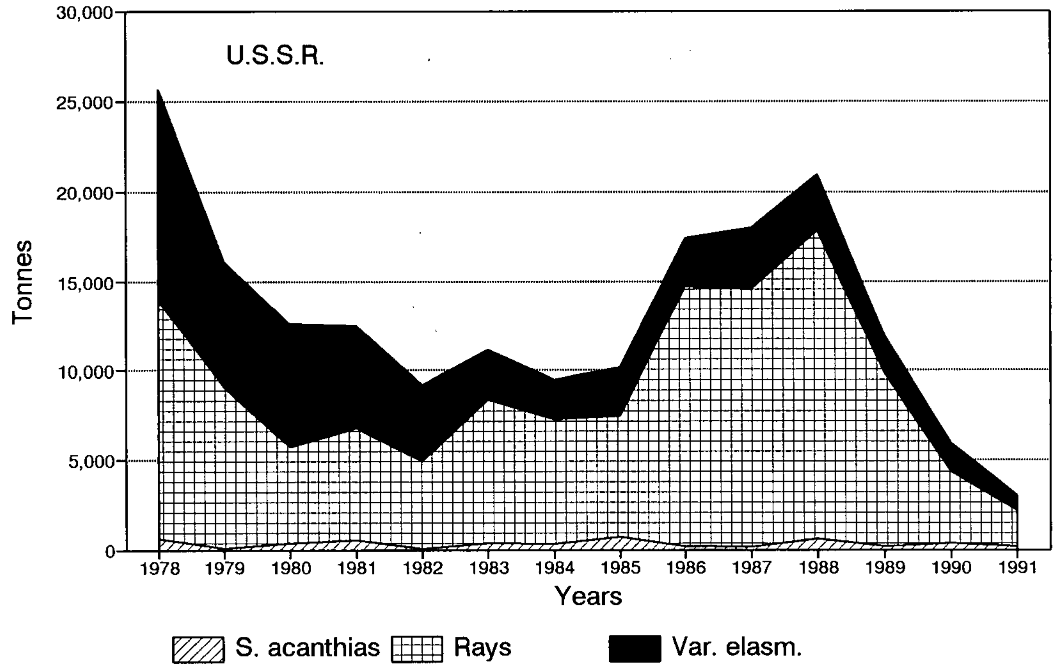



Figure 2.10 Elasmobranch catches of former USSR, by species groups, during 1978-1991. (Data from FAO).

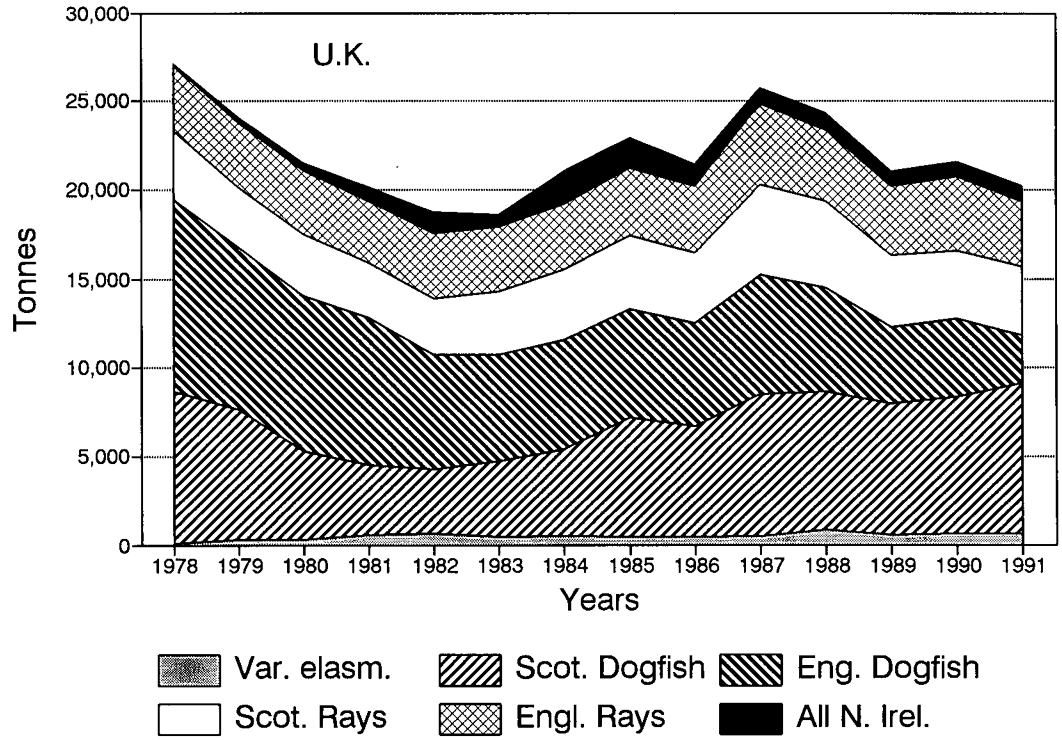



Figure 2.11 Elasmobranch catches of U.K., by country and species groups, during 1978-1991. (Data from FAO).

dictated by local market demand, or either incidentally by trawlers aiming for species such as cod, haddock and hake.

According to Kunzlik (1988), there were fisheries for basking sharks *Cetorhinus maximus* in the UK during the 40's, mainly on the west coast of Scotland. Most of these fisheries were short lived because of marketing difficulties and economic failures (Maxwell 1952). Basking sharks were hunted mostly during the summer with hand or whaling harpoons from vessels adapted from other fisheries, but catches never surpassed 300 sharks per year (approx. 600 t/yr). As in the case of Norwegian and other basking shark fisheries in the world, they were mainly aimed at the livers. Present catches are minimal, since 1983, only a single boat fishes opportunistically for basking sharks in Scotland.

Porbeagle sharks have been sporadically landed in small quantities (no more than 30 t/yr) in the UK, mainly on an incidental basis. The only exception is a single opportunistic event during 1987-1988 when porbeagles were unusually abundant for a couple of months in the Shetlands and 35-45 t were taken in four months (Gauld 1989).

Although UK catches of skates and rays are larger in the North Sea, most of the available information is from the Irish Sea. British fisheries for skates and rays in the Irish Sea are sustained mainly by four species, in order or importance: *Raja montagui, R. clavata, R. brachyura* and *R. naevus* (Holden 1977). Fishing pressure has apparently caused a decline in some of the local stocks. Brander (1977, 1981) considers that skates and rays of the Irish sea are in need of immediate management measures in order to allow stocks to recover and attributes the disappearance of *Raja batis* from the Irish sea to excessive commercial fishing. According to data summarised in Ryland and Ajayi (1984), stocks of rays in the Bristol Channel which used to provide 27% of the UK ray catch, were halved during 1964-1974. For the North Sea, Vinther and Sparholt (1988) roughly estimate the biomass for *R. radiata* and for all rays during the mid 80's as 160,000-252,000 t and 294,000-464,000 t respectively. Data presented by these authors suggest declines in the abundance of *R. batis, R. clavata, R. naevus* and increases in abundance of *R. radiata.* Later estimates of the biomass of *R. radiata* are of 100,000 t (Sparholt and Vinther 1991).

Research on elasmobranchs is comparatively abundant in Britain: however, management seems to be a long neglected area. During a period of years, a good amount of research was devoted to the stocks of spiny dogfish in the UK (Holden 1968, Holden and Meadows 1962, 1964). However, despite the general guidelines proposed by Holden based on his assessment of the fishery, it seems that regulation was never implemented for this stock.Despite the availability of a reasonable number of basic studies on rays in the UK, there seems to be no management specifically directed to these fishes. This might be due, at least partially, to the complications of setting management regulations for multispecific fisheries (especially bottom-trawl fisheries) due to technical interactions.

Ireland.

Elasmobranch fisheries of the Irish Republic have been of minor importance worldwide until 1985, when catches attained more than 10,000 t/yr (fig. 2.2). In the period 1987-1991 they only contributed 1.03 % to the world catch of elasmobranchs. Despite this low profile, elasmobranch fisheries have been relatively important for Ireland in recent times, representing 3.03 % of the total fisheries production of this country. This figure ranks relatively high compared to other major elasmobranch-fishing countries (table 2.2).

Rays have been exploited for a long time in Ireland in small quantities. Spiny dogfish is the other main elasmobranch and has gained much attention since the beginning of the 80's. Since 1983, spiny dogfish catches have made the major proportion of the total elasmobranch catch (fig. 2.12). During the period 1978-1991 rays and dogfish were equally represented in the elasmobranch catches of Ireland with 3,048 t/yr and 3,067 t/yr respectively. While the catches of rays have remained practically unchanged since 1978, dogfish catches increased tremendously in less than five years, had a small fall in 1986 and recovered and fell again after three years. Statistics for 1989-1991 suggest a relative stability has been achieved in this fishery. Fahy (1989a,b, 1991) and Fahy and Gleeson (1990) cover most of what is known from the recent elasmobranch fisheries of Ireland and most of the following information is taken from these sources.

Recording of ray landings goes back to 1903. No more than 600 t/yr were recorded before 1940, when catches began to rise slowly (partially due to increased consumption in Ireland), up to the late 70's when they sharply increased reaching 3,000 t in 1985. Ray catches have traditionally been taken in greatest quantities (around 50 % of the total) in the east coast of Ireland, since 1975 about 25 % has been taken from the north coast, and the rest came from the south and west coasts. Most of the landings are not sorted by species but by a casual process defined by similarities in size and appearance. At least 18 trawling vessels catch rays from eastern Irish ports. Thirteen otter trawlers and four beam trawlers operate from the southeast, but more vessels are suspected to take part in the fishery. Although most of these vessels catch rays incidentally to prawns and bottom teleosts, a small ray fishery appears to be run on a seasonal basis by some of the southeast vessels. At least nine species of rays are found in the commercial catches. *Raja brachyura, R. clavata, R. naevus* and *R. montagui,* are the most common rays, roughly in order of importance, and *R. microocellata, R. batis, R. fullonica, R. undulata* and *R. alba,* are caught only sporadically. Landings are mostly composed of small (less than 60 cm TL) and medium sized rays (60-70 cm TL) accounting for 60-80 % of the weight. Most species are totally recruited to the fishery after age 2, but *R. naevus* recruits at age 3. In the east coast, at least 50 % of the catches of *R. clavata* and *R. brachyura* are made of 0-2 year old. Total mortality estimates for the most important species mentioned above range from 0.54-0.74. Although the populations are heavily exploited specially in the southeast fishery, they continue to produce good yields.

There are dogfish fisheries all around the country but they have concentrated on the west coast. Catches were high in the north (Co. Donegal) during 1982-1985 but contributions in the south (Co. Kerry) increased during 1986-1987 as a result of effort being shifted to the south due to decreasing catches in the north. Dogfishes in the past were considered a nuisance, but now the fishery is specifically directed at them. In the west coast fisheries, otter trawlers fish mainly male dogfish in waters sometimes exceeding 100 m deep, while monofilament gillnets of 6.4 cm mesh size are used in shallow waters where they catch great proportions of pregnant females. Spiny dogfish in west Ireland fully recruit to the fishery at around 17 yr of age and total mortality coefficients have been estimated at 0.24 and 0.30 for females and males respectively. Fahy and Gleeson (1990) report that monthly CPUE of gillnetters in Carrigaholt has plummeted by 80-90 % in a two-year period. Available information is insufficient to make definite conclusions about depletion of the stocks, but it seems that these are close to being overfished. Total female spawning biomass for Carrigaholt was estimated at 5,700 t by Fahy and Gleeson. Most of the catches are destined for export but no information on the genesis of this fishery could be located.

A fishery for basking sharks began in 1947 at Keem Bay on the west coast of Ireland (Kunzlik 1988). Initially harpoons and nets were used, but by 1951 only nets were used. These were either encircling nets or entangling nets built of sisal with mesh sizes of 33 cm and set perpendicular to the shore. Initially, the liver was the only target of the fishery but in later years fins and meat were also used for human consumption. In 1973 harpoons were reintroduced to this fishery and another harpoon fishery started in the south east coast of Ireland. The west coast fishery reached its highest yields (around 1,500 sharks annually) during the early 50's and declined after 1955, probably as a response to the shrinking market for livers which brought down fisheries for sharks all over the world. Catches remained below 100 sharks/year during most of the period 1963-1973 and increased to almost 400 sharks in 1975 when the last records were taken. Some trials to develop a commercial blue shark fishery with longlines off the south coast of Ireland were under way during 1990 (Crummey et al. 1991).

France.

French elasmobranch catches suggest another relatively stable fishery. Two periods of relatively sustained catches are identifiable. From 1948 to 1960 catches oscillated near 15,000 t/yr, then shifted in 1961 to higher, more variable catches around 35,000 t/yr (fig. 2.2). Elasmobranchs represent 3.78 % of the total fishery production of France, the highest among European countries and rather high in global terms. French catches make 4.79 % of the world elasmobranch yield.

Between 1978 and 1991, French catches of skates and "various dogfishes" were stable. Spiny dogfish, "various elasmobranchs" and porbeagle catches showed a slight declining trend (fig. 2.13). During this period, skates averaged 42 % (14,499 t/yr) of the total elasmobranch catches while spiny dogfish, various dogfishes, various elasmobranchs and porbeagles averaged 32 % (10,806 t/yr), 18 % (6,139 t/yr), 6% (2,103 t/yr) and 2 % (531 t/yr) respectively. Spiny dogfish and skates are caught by French vessels mainly in the Northeast Atlantic but small catches of skates are also taken in the northwest Atlantic and the Mediterranean Sea. According to Gauld (1989), a small flotilla of French vessels based

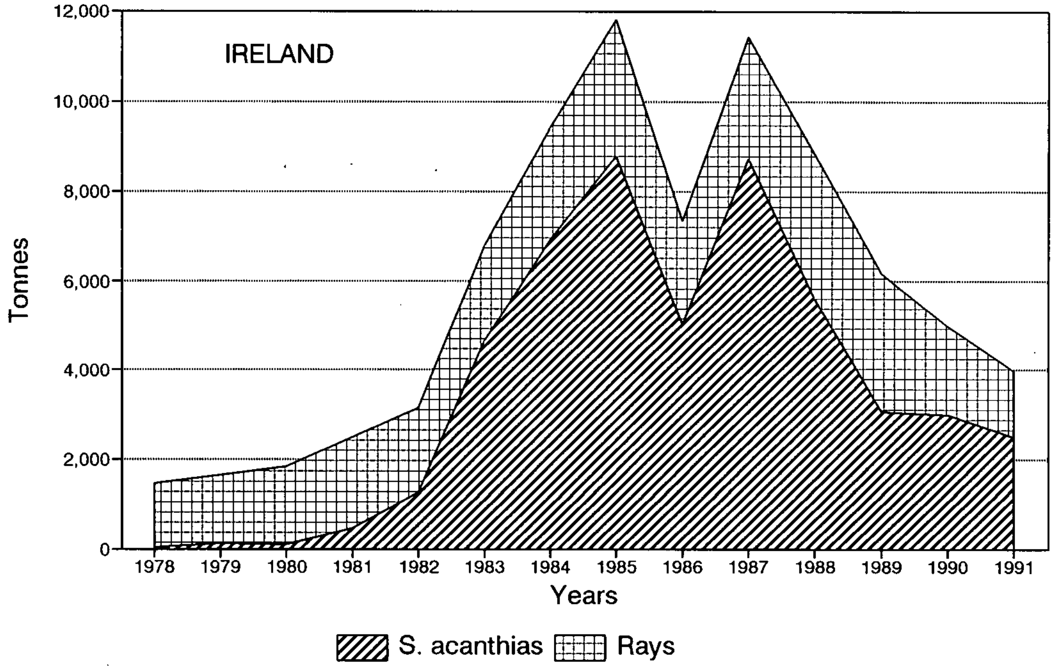



Figure 2.12 Elasmobranch catches of Ireland, by species groups, during 1978-1991. (Data from FAO).

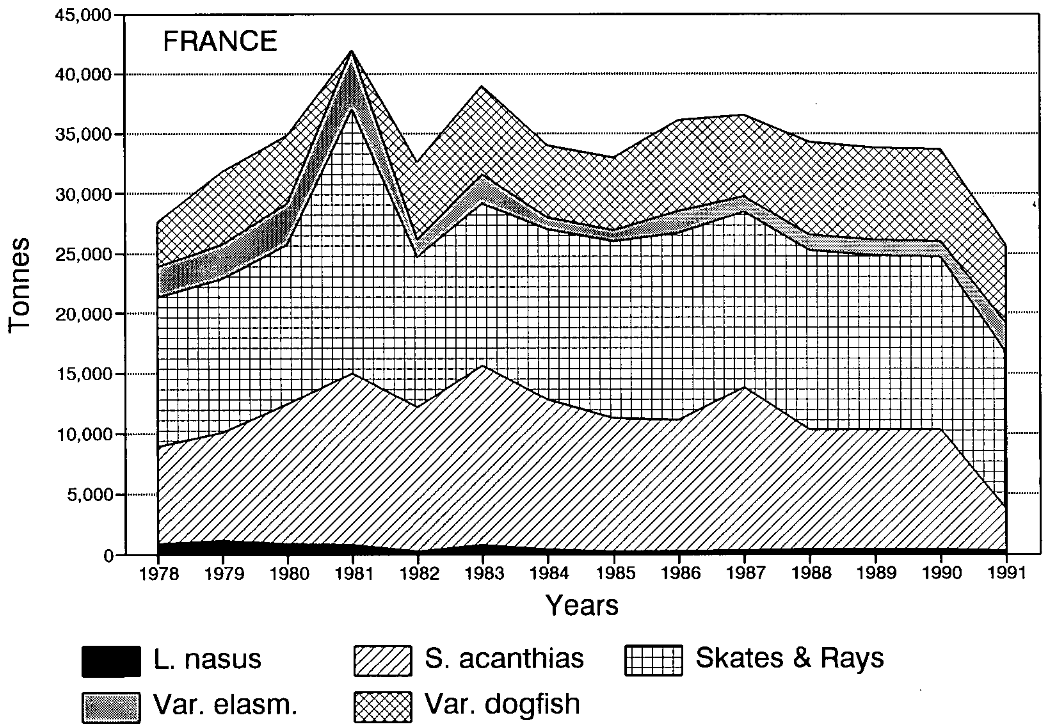



Figure 2.13 Elasmobranch catches of France, by species groups, during 1978-1991. (Data from FAO).

in Britannia specifically target porbeagles with longlines in the Bay of Viscay and Irish waters, making about 75 % of the total porbeagle catches of France. The remainder is landed as a bycatch of trawl and seine fisheries.

Tetard (1989a, 1989b) summarises information about shark and batoid fisheries for France. His information allows further separation of catch statistics into species or species groups. The following is from his account. The catch of batoids of France includes at least eight species of skates and rays. The separation of ray species is possible because each species attains a different price. *Raja naevus* and *R. clavata* are the most important species accounting respectively for about 25 % and 17 % of the batoid landings during 1978-1987. *Raja montagui* and a group formed by *R. batis* and *R. oxyrinchus* make 4 % and 3 % respectively. *Dasyatis pastinaca, Myliobatis aquila* and *Raja fullonica* are species of minor importance contributing only 1 % of the catches. Finally a group of unidentified rays makes the remaining 50 %. Most of the French catches of rays are taken around the Celtic Sea and the English Channel and to some extent in the Irish sea and the North of the Bay of Biscay. Rays are mostly caught by bottom trawling operations. *Raja clavata* is actively sought for its highly desired meat. Tetard highlights the almost complete disappearance of *R. alba* from the catches and the apparently declining catches of *R. clavata.* Meanwhile, yields of *R. naevus* seem to be increasing. He also notes that an uncited study indicates that the yield per recruit of *R. naevus* is at an optimal value. Judging from Tetards' account, it appears that no management regulations are in place for any of these species in French waters.

French shark landings are chiefly composed of spiny dogfish and catsharks. The latter are mainly *Scyliorhinus canicula* with a minor component of *S. stelaris.* Catshark catches are all incidental to trawler and longline fisheries and make about 32% of the shark catch. The spiny dogfish fishery is one of the few directed fisheries for sharks in France, accounting for almost 57 % of all shark landings. During 1987, approximately 27 longliner boats 8-25 m long (three of them with automatic longliners), were potentially targeting spiny dogfish. However, about 80 % of the landings came from bottom trawlers. The main fishing grounds of the French fishery for spiny dogfish are the Celtic Sea, and formerly, Northern Irish waters and the North Sea. Tope, *Galeorhinus galeus,* ranks third in importance among shark catches with about 6 % of the total, but catches seem to be in clear decline. The French fishery for porbeagles is also a directed shark fishery which represents about 3 % of the shark catch. Some shortfin mako sharks are caught incidentally in the longlines of this fishery. About 75 % of the landings are from longliners and the rest from trawlers. Main fishing grounds for the French porbeagle fishery are offshore waters from Spain to Ireland in winter, closer to shore and around the Channel Islands in spring. Smooth-hounds, *Mustelus mustelus* and *M. asterias* make about 1 % of the total French shark catch. Some minor quantities of blue shark and angel shark, *Squatina squatina,* are landed incidentally to longline and trawl fisheries respectively.

France is both the major producer and importer of shark in Europe. Because of its high exports of mainly porbeagle and tope to Italy, France has a deficit of shark meat, thus imports have increased since 1982 (9,000 t in 1986). However, some problems related to mercury content of shark meat seem to limit French exports to Italy constraining the fishery for porbeagle sharks. The home market is also increasing. The meat of *Lamna nasus*, *Squalus acanthias* and *Galeorhinus galeus* has good internal demand for human consumption as "salmonette" (saumonette) in school refectories and restaurants. The internal demand for *Squalus acanthias* is not met by French landings and considerable quantities have been imported from the UK.

Spain.

Spanish elasmobranch catches were steady during 1947-1971 when yields varied within a narrow band of 10,000-15,000 t/yr. This was followed by a sudden collapse in the early 70's and a later slow recovery in the 80's up to the recent level oscillating wildly at 15,000-20,000 t/yr (fig. 2.2). Elasmobranchs only make 1.3 % of the total fishery production of Spain and contribute only 1.2 % of the world elasmobranch catch (table 2.2).

Disaggregated data for the years 1978-1991 indicates that the major reason for the recent increase in catches are the skate fisheries which have grown consistently in the period 1980-1987 (fig. 2.14). The bulk of skates comes from operations in the Northwest Atlantic (average of 80 % of skate catches) and the rest from the northeastern Atlantic. No information on the relative importance of the species in the catches is available. Shark catches taken mainly in the Northeast Atlantic have also increased in a similar way. These include shortfin makos *Isurus oxyrinchus,* porbeagles *Lamna nasus,* small-spotted catshark *Scyliorhinus canicula* and some squaloids. Various species of rays are fished in small quantities mainly in the Mediterranean Sea along with unspecified elasmobranchs, which are also caught in the central eastern Atlantic (FAO Area 37). In this area, skates make up 63 % (7,125 t/yr) and unspecified sharks 21 % (2,259 t/yr) of elasmobranch catches, the contribution of "various elasmobranchs" was only 11 % (1,168 t/yr).

All elasmobranch fisheries in Spain are incidental catches of either trawl or longline fisheries (R. Muñoz-Chápuli, pers. comm. Jan. 2 1992). Muñoz-Chápuli (1985a) reports on the landings of Spanish commercial bottom trawlers operating in depths up to 500 m. *Scyliorhinus canicula* dominates the landings from the mouth of the Mediterranean, the southern coast of Spain and coasts of northwest Africa. *Centrophorus granulosus* and *Squalus blainvillei* are also landed from these areas. In the entrance of the Mediterranean, *Galeus melastromus* is also important in the catches while another 11 species are caught in smaller amounts in both regions (table 2.6). Muñoz-Chápuli (1985b) reports that landings from longline vessels fishing in the eastern Atlantic from the Azores to the Cape Verde Islands, are dominated by *Prionace glauca, Isurus oxyrinchus* and *Sphyrna zygaena,* while 13 other species are of minor importance (table 2.6). Very likely, both reports reflect not only the abundance of the species in such areas but also the selection of species on board. Spanish swordfish longliners caught 304 t of shortfin makos and 20 t of porbeagles from the north and central east Atlantic during 1984 (Mejuto 1985). Makos were more abundant during Sept-Dec and catches were mainly composed of sharks 100-240 cm FL with males more than doubling the numbers of females. Porbeagle catches were more abundant in March, September and October and individuals were mostly 150-225 cm FL, with males doubling the numbers of females.

Italy.

Judging from the historical imports of sharks from Norway (porbeagles), France (porbeagles and tope) and Argentina (smooth-hounds), elasmobranch meat seems to be well appreciated in Italy. Nonetheless, sharks and rays have been for long of minor importance in Italian fisheries. Catches did not exceed 6,000 t/yr until the mid 80's when more than 10,000 t/yr were taken (fig. 2.2). Currently, elasmobranchs represent only 1.89 % of the total

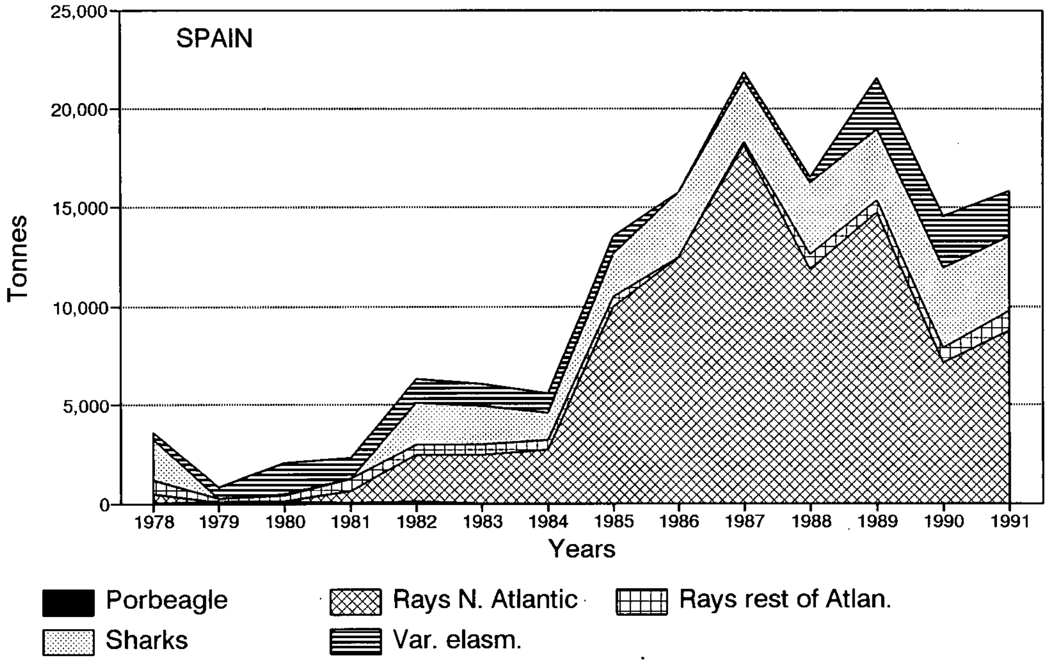



Figure 2.14 Elasmobranch catches of Spain, by species groups, during 1978-1991. (Data from FAO).

Table 2.6 Shark species reported in Spanish commercial fisheries (adapted from Muñoz-Chapuli 1985 a,b).

| Demersal | Pelagic |
|---|---|
| *Hexanchus griseus* | *Lamna nasus* |
| *Heptranchias perlo* | *Isurus oxyrinchus* |
| *Squalus acanthias* | *I.paucus* |
| *S. blainvillei* | *Alopias vulpinus* |
| *Centrophorus granulosus* | *A. superciliosus* |
| *C. lusitanicus* | *Carcharhinus brevipinna* |
| *Deania calcea* | *C. falciformis* |
| *Dalatias licha* | *C. longimanus* |
| *Squatina squatina* | *C. obscurus* |
| *S. aculeata* | *C. plumbeus* |
| *Galeus melastomus* | *C. signatus* |
| *Mustelus mustelus* | *Prionace glauca* |
| *M. asterias* | *Galeorhinus galeus* |
| | *Sphyrna zygaena* |
| | *S. lewini* |

fishery catches of Italy. Furthermore, the Italian catch of sharks and rays makes only 1.51 % of the world elasmobranch catch (table 2.2).

During the period 1978-1991, smooth-hounds *Mustelus* spp. averaged 52 % (4,463 t/yr) of the elasmobranch catches, rays contributed 38 % (3,340 t/yr) and various elasmobranchs 10 % (860 t/yr). Catches of all elasmobranch groups grew similarly during the expansion of the fishery which peaked in 1985 and then declined to bounce back in 1990-1991 (fig. 2.15). Smooth-hounds were all fished in Mediterranean waters, along with 91 % of the ray catch. The rest of rays were caught in FAO Areas 34, 47, 48, 51 and 21. Catches of various elasmobranchs were taken in FAO Area 34 (70 %) and Areas 47 (7 %), 51 (16 %) and 41 (7 %). Small catches of blue sharks *Prionace glauca* are landed as a bycatch of the drift longline swordfish and albacore fisheries of the Gulf of Taranto, where averages of 14.5 t/yr and 4 t/yr were landed respectively in each fishery during 1978-1981 (De Metrio et al. 1984). During this period, an average of 12 boats fished for swordfish from April to August using between 700 and 1000 hooks (Mustad no. 1) per boat. Additionally, an average of 44 boats fished for albacore during August to December with 2,000 hooks (3 cm long) per boat. Due to the different hook size and probably also to seasonal cycles of the species, the swordfish boats caught blue sharks of 25 kg average weight whereas blue sharks from the albacore boats averaged 3 kg. De Metrio et al. (1984) report that the meat of *Prionace glauca* is fraudulently sold in Italy as *Mustelus*. It is therefore very likely that the blue shark catch is probably reported under *Mustelus* spp. in the official statistics.

2.2.2.3 Africa and Indian subcontinent.

Information about elasmobranch fisheries in this region is particularly scarce. Statistics from most of these major elasmobranch-fishing countries give little detail of the catch composition and literature sources are not only limited but also very difficult to obtain. This is probably directly related to the economic development of these countries which are located in one of the most economically depressed areas of the world.

Nigeria.

Nigeria is the only African nation with major elasmobranch fisheries. FAO statistics for

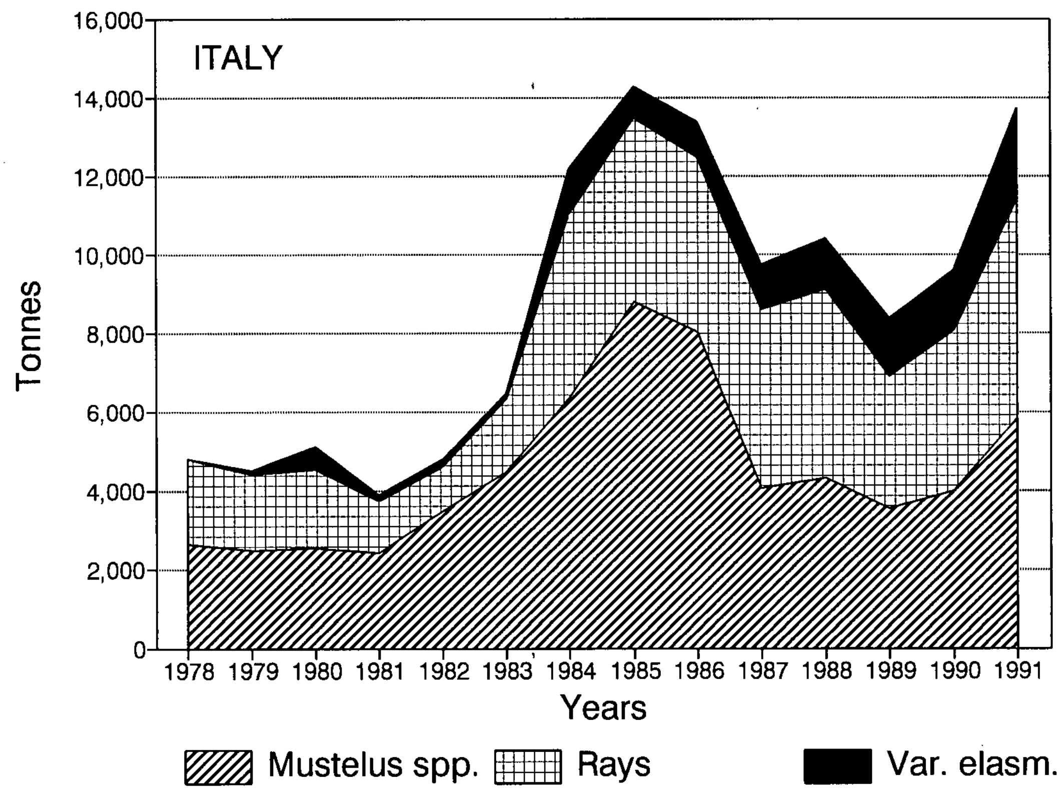



Figure 2.15 Elasmobranch catches of Italy, by species groups, during 1978-1991 (Data from FAO).

Nigeria are poor and have only appeared regularly since 1970. Nigeria has an unstable fishery with an overall trend of decreasing catches falling from the more than 30,000 t/yr caught in the early 70's to less than 10,000 t since 1986 (fig. 2.2). However, without any background information it is difficult to venture into an interpretation of these falling catches. Despite the fall in yields, elasmobranchs continue to be a relatively important resource for Nigeria, recently (1987-1991) contributing 2.92 % of the total fishery production of this country. The catch of sharks and rays of Nigeria contributes only 1.91 % to the world total.

FAO data from 1977-1991 show that most of the catches are not recorded by species. A group of "various elasmobranchs" accounts for 89 % (15,827 t/yr) of the catches, while Squalidae and a group of skates and rays accounts for less than 1 % (7.6 t/yr) and about 10 % (1,703 t/yr) respectively (fig. 2.16). No further information about the characteristics of the fishery, species, research or management of elasmobranchs in Nigeria could be found.

Pakistan.

Elasmobranch fisheries of Pakistan have been fairly unstable. They were of prime importance on a global scale until recently, when yield plummeted. Elasmobranch catches of Pakistan grew almost exponentially from the late 40's to a first peak of about 75,000 t in 1973, dropped about 50% the following three years and then recovered to peak levels for another 6 years. Yield collapsed in 1983 but has recovered over the last 10 years to the present levels of about 45,000 t (fig. 2.2). Given the scarcity of direct information on Pakistani fisheries it is very difficult to assess the reasons for these dramatic changes in elasmobranch catch. The relative importance of elasmobranchs in Pakistan is among the highest in the world, representing 7.42 % of the total national catches during 1987-1991. This level must have been at least double during the bonanza of the late 70's. Pakistan contributes 4.99 % of the world elasmobranch production (table 2.2).

Grey sharks (Carcharhinidae) and batoids constitute most of the catches, averaging 45% (20,200 t/yr) and 54% (24,380 t/yr) of the elasmobranch yield respectively during 1977-1991. Since 1987, catches of sawfishes (Pristidae) and guitarfishes (Rhinobatidae) have been reported separately but they only account for <1% and 1% of the elasmobranch catches respectively (fig. 2.17). While grey sharks catches declined steadily during the late 70's and

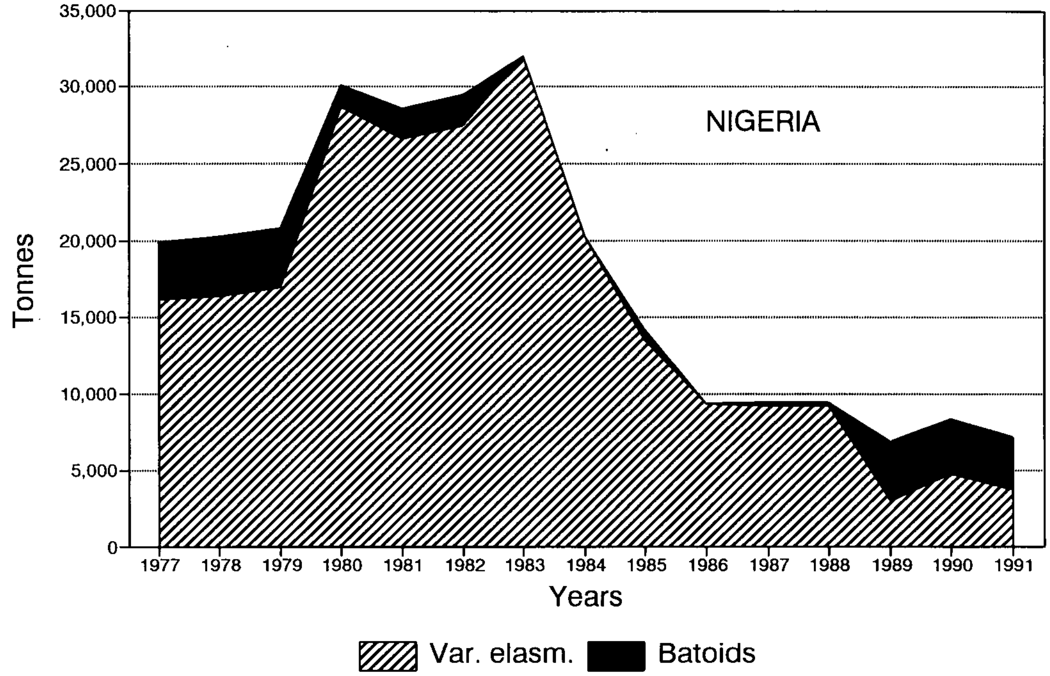



Figure 2.16 Elasmobranch catches of Nigeria, by species groups, during 1977-1991. (Data from FAO).

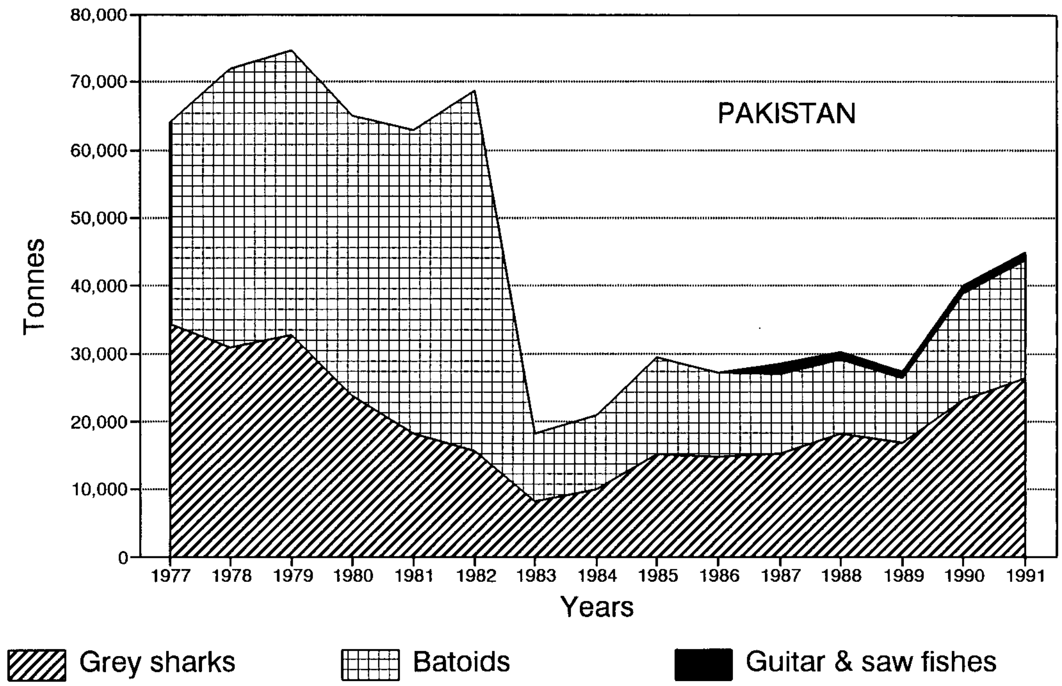



Figure 2.17 Elasmobranch catches of Pakistan, by species groups, during 1977-1991. (Data from FAO).

early 80's, batoid catches dropped abruptly by 43,000 t in a single year (1983) thus causing the overall collapse. Grey sharks have since been the major group in the elasmobranch catches.

Detailed information about Pakistani elasmobranch fisheries is very poor. A report from the Indo Pacific Tuna Development and Management Programme (IPTP 1991) is about the only source of information. According to this document, the port of Karachi is the only landing site for the mechanised gillnet fleet in the entire coast of the Sind province. Sharks are caught mainly by pelagic gillnet boats fishing as far as Somalia and the waters off Yemen and Oman, although small catches are also landed by bottom gillnetters working in coastal areas of Pakistan. The estimated number of mechanised gillnetters in Pakistan in 1989 was 394 vessels, 185 in Sind province and 209 in Baluchistan province. Vessels in Karachi range in sizes from 20 to 25 m in length and 5 to 7 m in breath and use diesel engines of 88-135 HP. These fisheries are very important socio-economically, employing considerable numbers of fishermen. Small boats carry 15-17 crew and make trips of about 10 days, whereas larger boats carry up to 25 fishermen and stay at sea for 20-30 days and occasionally 60 days. Catches are usually salt dried in the larger vessels and kept on ice in the smaller ones. Gillnets are hand-woven out of multifilament polyamide twine and are 80 meshes deep and 2.5-9 km long (average of about 5.2). Mesh sizes are 10-16 cm and mainlines of 14-16 mm diameter. Shark catches are sorted into eight different size categories, but the species are not separated. Effort in this fishery increased from 23,000 fishing days in 1988 to 28,000 in 1989 then fell to 26,000 in 1990. Estimates indicate that about 93 % of the shark catch comes from pelagic driftnet vessels as opposed to demersal or mixed fishing vessels. The yield of sharks for this driftnetting fleet is estimated at about 3,860 t/yr during 1988-1990. Shark yield during this period was correlated with distance to fishing grounds. The largest catches coming from Somalian waters, the most distant fishery. Shark yields decreased by about 44 % from 1989 to 1990, although this decline was not exclusive to shark catches, teleost (tunas etc.) yields fell 32 % during the same period.

Some efforts to introduce longline fishing for sharks, rays and other species in Pakistan are summarised by Prado and Drew (1991). Apparently gillnets are much more favoured in Pakistan because of their higher catch rates of valuable species.

India.

By tradition, India has had important fisheries for elasmobranchs. The fisheries had a relatively steady growth up to the mid 70's, followed by a period of stability in catches during most of the 80's, then a tremendous increase in catches in 1987, resulting in India becoming one of the top three elasmobranch producers in the world during the last ten years (fig. 2.2). The importance of the Indian yield of sharks and rays is highlighted when we consider it represents 8.78 % of the world elasmobranch catch. Still, due mainly to large inland fisheries, elasmobranchs do not rank very high in the domestic fisheries, making only 1.72 % of total fish catches of India in 1987-1991. Catches are not sorted into species or taxonomic groups in the statistics but are only divided into FAO areas. Approximately equal amounts of sharks and rays (about 26,000 t/yr) were obtained in each Area for the period 1977-1991, with catches from the west coast only slightly larger than those of the east coast during 1977-1991 (fig. 2.18).

There is a relatively high number of published articles on elasmobranch exploitation and utilisation in India, especially for the 80's. Appukuttan and Nair (1988) indicate that during 1983-1985 sharks comprised 55 % of the elasmobranch catch of the country. The main fishing areas in order of importance were Gujarat, Maharashtra, Kerala Andhra Pradesh, Karnataka and Tamil Nadu. Important fishing grounds for sharks are reported for Ashikode, Kerala Province (Anon. 1983).

Sharks catches are all incidental to other fisheries in India (Appukuttan and Nair 1988). Sharks are mainly taken with longlines which vary regionally in design, but they are also taken as bycatch of trawlers using disco nets off Ratnagiri (Maharashtra), with bottom set gillnets in Porto Novo (Tamil Nadu) and the shrimp trawlers of Kerala (Devaraj & Smita 1988; Shantha et al. 1988; Rama Rao et al. 1989; Kulkorni & Sharangdher 1990).

Rays are caught with bottom set gillnets in Gujarat, northwest India and Cudalore and are abundant in the outer shelf and slope off Kerala and Karnatakta (Devadoss 1978; Kunjipalu & Kuttappan 1978; Sudarsan et al. 1988). Devadoss (1984) indicates that batoids make up to 10% of bycatches in Calicut; 90% of the bycatch comes from trawlers, 8% from gillnets and 2% from hook and lines. Both sharks and rays are abundant in Lakshakweep and form important bycatches in trawling fisheries in Krishnapatnam (Swaminath et al. 1985; James 1988).

Recent reports (Dahlgren 1992) indicate that directed fisheries for sharks are starting to develop on a seasonal basis on the east coast of India. About 500 vessels, both sail-powered and motorised fish for sharks with bottom or drift longlines in the coasts of Orissa Andhra Pradesh and Tamil Nadu. Bottom longlines are usually set in waters 80-150 m deep and occasionally as deep as 500 m. Bull sharks and tiger sharks are commonly caught in bottom longlines. The longlines have up to 400 hooks and the meat is usually salted on board. In Orissa alone, about 200 boats are engaged in drift longlining on a seasonal basis (December-March). The most common species in drift longlines are silky sharks and scalloped hammerhead sharks.

Catch composition data are not readily available, but the multispecies nature of these fisheries is evident from the literature. Appukuttan & Nair (1988) report that more than 20 species of sharks (mainly carcharhinids and sphyrnids) are known to be common in the catches. From their data for Pamban and Kilakkarai, *Rhizoprionodon acutus, R. oligolinx, Carcharhinus limbatus, C. sorrah, C. hemiodon, Sphyrna lewini* and *Eusphyra blochii* seem to be the most important species in the catches. Other sharks found in Indian catches are *C. melanopterus* and *Scoliodon laticaudus* (Devadoss 1988). Some important batoids are: *Dicerobatis eregoodoo, Rhynchobatus djiddensis, Rhinobatus granulatus, Himantura uarnak, H. bleekeri, Dasyatis sephen, D. jenkinsii, Aetobatus narinari, A. flagellum, Aetomylus nichofii* and *Mobula diabolus* (Devadoss 1978, 1983; Kunjipalu & Kuttappan 1978).

Although localised assessments of the state of the fisheries for elasmobranchs exist (Santhanakrishnan 1983, Krishnamooorthi et al. 1986, Devadoss et al. 1988, Sudarsan et al. 1988), there are no overall studies of the elasmobranch fisheries of India (Appukuttan & Nair 1988). Devadoss (1983) reports ray resources off Calicut apparently overfished by 1980, while according to Reuben et al. (1988) shark and ray resources of Northeast India were still underexploited in 1985. Devadoss et al. (1988) performed local assessments using Schaefer's model and provide suggestions of effort increase/decrease for the different areas studied.

The present situation in Indian elasmobranch fisheries needs careful monitoring. The high catches of elasmobranchs in India, which peaked at 73,500 t in 1988, suggests a very high level of exploitation. It is unlikely that such large yields will be sustainable over a long period of time, specially under the light of the 1983 collapse of neighbouring Pakistani elasmobranch fisheries.

Sri Lanka.

The elasmobranch fisheries of Sri Lanka appear on record since the early 50's. Their development has been slow, growing from less than one tonne in 1952 to the current level of about 15,000 t/yr (fig. 2.2). Sri Lankan elasmobranch fisheries are the smallest among major elasmobranch-fishing countries in the Indian Ocean. However, sharks and rays are quite important at a national level, contributing 8.76 % of the total catches during 1987-1991. Remarkably, this is the highest relative importance of any elasmobranch fishery in the world. The catch of sharks and rays of Sri Lanka represents 2.42 % of the world elasmobranch catch for the period 1987-1991 (table 2.2).

Information on catch composition is very poor for Sri Lankan elasmobranch fisheries. FAO data indicates that catches were commonly grouped in a single "various elasmobranchs" category until 1987. Since then, an entry reported as *Carcharhinus falciformis* constitutes the major part of the catches. On the other hand, information from the National Aquatic Resources Agency (NARA) of Sri-Lanka (P. Dayaratne, NARA, Colombo, Sri Lanka, pers. comm. February 1992) indicates that *C. falciformis* comprises only 75% of the shark catches, with *C. longimanus, C. sorrah, Sphyrna lewini, Alopias pelagicus* and *Isurus oxyrinchus,* ranking high among the remaining 25%; hence catches reported as *C. falciformis* by FAO here loosely labeled "Carcharhinid sharks" on figure 2.19.

There are few directed fisheries for elasmobranchs in Sri Lanka. Some estimates (P. Dayaratne, pers. comm. op. cit.) indicate that approximately 85% of the elasmobranch yields are bycatches from other fisheries, which use mainly bottom and drift gillnets. Both the directed and incidental catches of elasmobranchs come from small-scale fisheries. Drifting shark longlines are used in offshore ( >40 km from shore) EEZ waters in the directed fishery. Bottom set gillnets operate in coastal areas up to 25 km from shore (P. Dayaratne

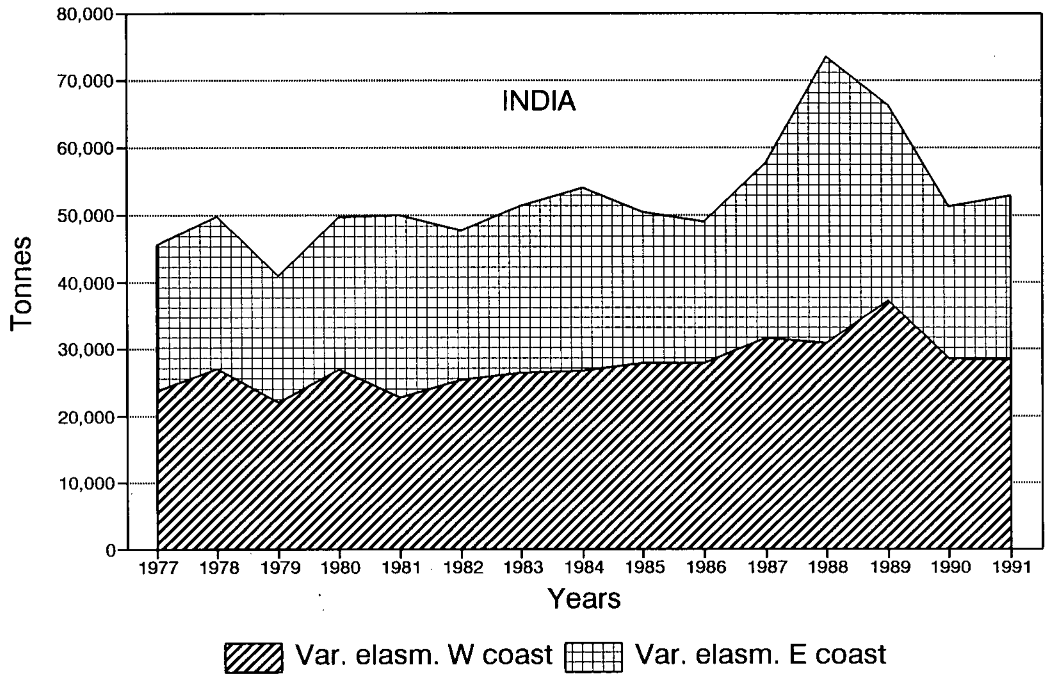



Figure 2.18 Elasmobranch catches of India, by region, during 1977-1991. (Data from FAO).

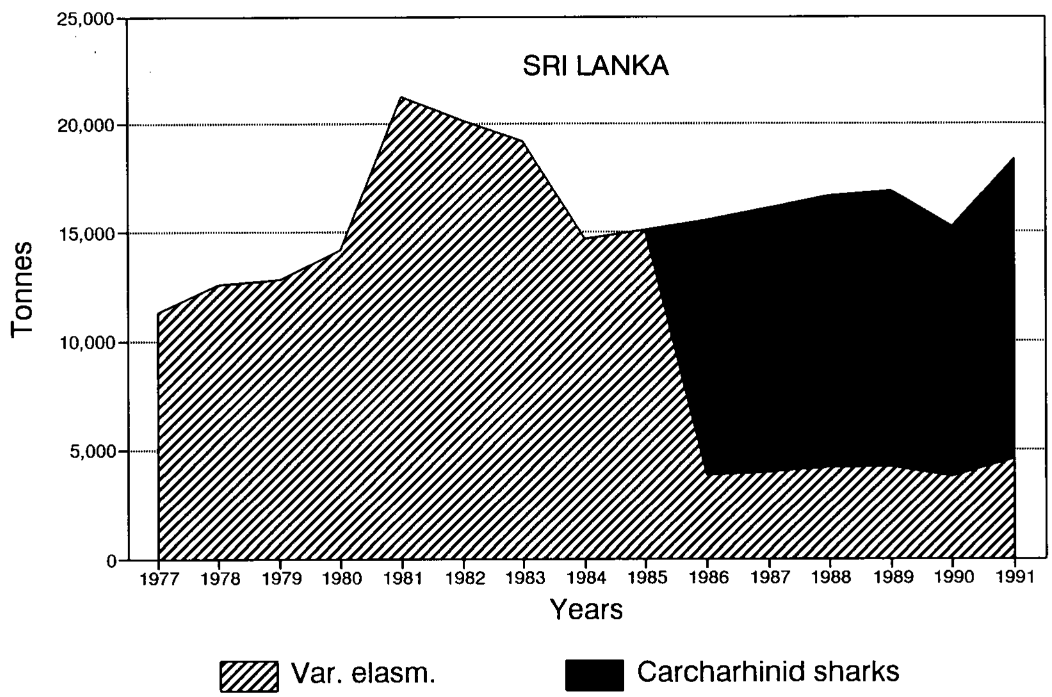



Figure 2.19 Elasmobranch catches of Sri Lanka, by species groups, during 1977-1991 (Data from FAO).

pers. comm. op. cit.). Pajot (1980) reports that elasmobranchs comprise 26.62 % of the total catch in weight for the large-mesh small-scale driftnet fisheries of Sri Lanka.

There is some detailed information about the pelagic tuna fisheries of Sri Lanka in which substantial amounts of sharks are caught incidentally. Most of these information has been produced by the IPTP/NARA tuna sampling programme (IPTP 1989, Dayaratne and Maldeniya 1988, Dayaratne and de Silva 1990, Dayaratne 1993a,b). The sampling programme started in Kandakuliya in the northwest, Negombo in the west and Beruwala in the southwest coast of Sri Lanka during 1986. Two additional locations were added in the south coast (Matara and Hambantota) in 1987. Roughly, three types of vessels operate in the pelagic tuna fisheries: small outboard motored boats roughly 5 m in length, stationary diesel motored vessels of about 9 m length and 3.5 t of displacement, and the larger 11 m long 11 t net tonnage vessels with stationary diesel motors. By far, the most numerous are the 3.5 t vessels with about 2,000 units. They usually carry a crew of four and about 40 panels of net. Over 1,000 of these boats spend more than one day offshore per trip. In contrast, there are only 70 of the 11 t vessels, but these usually carry 50-60 panels of net and are capable of making offshore trips 6-8 days long. Gillnets are the most popular gear and they have been used for many decades by Sri Lankan fishermen. Each piece of net measures 500x100 meshes of size 90-180 mm (140-152 the most common), making a total of 3-4.5 km of net per vessel. In general, the yield and catch rate of sharks in this fishery are quite variable, but both have a clear increasing trend. Total shark catch grew from 1,569 t in 1986-1987 to 2,155 in 1987-1988 in the northwest, west and southwest coasts. For the west and south coasts, total shark catches increased from 3,159 t to 4,374 t, to 8,676 t during 1989-1991. Overall shark catch rates increased from about 10 kg/day/boat in 1986 to about 35-40 kg/boat/day in 1988. These increases in shark yields and CPUE reflect expansion of the fishing grounds to offshore areas, increase in time spent at sea for each trip and changes in the fishing gear which involve fewer vessels fishing only with gillnets and more vessels switching to multiple-gear fishing. The percentage importance of sharks in the catch of each gear type is about 15 % for driftnets, about 28 % for vessels using driftnets/longlines/handlines, about 40 % for driftnets/longlines/troll lines, and about 45 % in driftnet/longline vessels. Elasmobranch catches for each gear type in 1991 were: driftnet 313 t; driftnet/longline 3,569 t; driftnet/longline/handline 513 t and driftnet/longline/troll line 1,110 t. The species composition of sharks in the pelagic tuna fishery is dominated by grey sharks (Carcharhinidae) which constitute about 85% of the shark catch, followed by hammerheads (3.5 %), thresher sharks (1 %), mackerel sharks (0.7 %) and other sharks and rays comprising the remaining 10.3 %. The catches of sharks in this fishery are estimated visually by weight. However, there are currently plans to include three species of sharks (*Carcharhinus falciformis, C. longimanus* and *Prionace glauca)* in the sampling campaigns in the near future (J. Morón, IPTP, pers. comm. December 1993).

According to Dayaratne (pers. comm. op. cit.) elasmobranchs are at present sustainably exploited in Sri Lanka. There are no management measures for these fisheries, nor are there any presently under consideration. So far, there is no evidence of any conservation problem or endangered species. Nonetheless, figures show that sharks and rays represent an important fishery for Sri-Lanka and they should be managed carefully. This summary indicates that at least the pelagic fishery is presently at a developing stage. It seems this is the ideal time to start research efforts aimed towards the management of the resource.

2.2.2.4 Asia.

Japan.

According to recorded statistics, Japan has taken the world's largest catches of elasmobranchs, although these have followed a clear decreasing trend after the initial explosive growth of the late 40's when a record 118,900 t were caught (fig. 2.2). Despite this contraction in catches, Japan's elasmobranch yield still ranks among the top seven in the world with 37,300 t in 1991 and contributed 4.98 % of the total world elasmobranch catch in the period 1987-1991. This is quite high when compared with most other countries. Taniuchi (1990) reports that the relative importance of sharks (which traditionally make the majority of elasmobranch catches) dropped from 4.3% of the total fish catches in 1949 to 0.3% in 1985. Taniuchi remarks that both a decrease in the relative cash value of elasmobranchs and a reduction of the Japanese elasmobranch stocks seem responsible for the general decline in these fisheries. At present, elasmobranchs constitute 0.31 % of the total Japanese catches, one of the lowest among major elasmobranch-fishing countries (data from FAO for 1987-1991). Taniuchi also reports a sharp reduction of Japanese catches of spiny dogfish *Squalus acanthias* from more than 50,000 t in 1952 to less than

10,000 t in 1965 and points out that this might represents a reduction of spiny dogfish stocks; catches of other sharks has not followed the same trend. Aside from possible stock reduction, it should be considered that as Japan's economy has expanded tremendously since the post-war period, changes in purchase power might have modified consumer preferences thus explaining the decreased demand for elasmobranchs. This hypothesis seems to be confirmed by the large amounts of sharks that are discarded at sea in various Japanese fisheries (see below and next section).

Japanese elasmobranch yield is chiefly a bycatch of other fisheries. Some exceptions are a trawl fishery for skates and rays in the East China Sea, a salmon shark fishery off northeast Japan in the Oyashio Front (Paust, 1987) and a winter fishery in Hokkaido for *Raja pulchra* (Ishihara 1990). Additionally, small scale coastal gillnet fisheries take up to 3,817 t of sharks, which accounts for less than 0.01% of the total coastal gillnet catch in Japan (Anonymous 1986). Several trends can be identified in the data presented by Taniuchi (1990) and Ishihara (1990) for the period 1976-1985 (fig. 2.20). Sharks accounted for 83% of the elasmobranch catches of Japan and batoids for 17%; at least 63% of the shark catches were taken as bycatch of tuna longline operations around the world, while the remaining 37% came from various unspecified sources. Of the average 25,000 t/yr of sharks landed by the tuna longline fleet, 58% came from offshore areas, 33% from the high seas and only 9% from coastal waters presumably the Japanese E.E.Z. Additionally, shark catches equivalent to approximately 2.8 times the landed shark bycatch of the longline tuna fishery is discarded at sea. Of the approximately 9,000 t/yr catch of batoids, 50% was fished in the East China Sea, 35% in Hokkaido and 8% in the Sea of Japan. Japan holds some of the largest high-seas fisheries for tunas and billfishes in the world. These produce substantial incidental catches of sharks, some of which are utilised. An account of these fisheries is given under section 2.2.3.

Data from FAO for the period 1977-1991 indicate that sharks are taken mainly in the northwest Pacific (Area 61) where Japanese catches are rapidly declining (fig. 2.21). Approximately 8,000 t/yr are taken in the rest of the Pacific with fairly constant trend and very small catches are also caught in the Indian and Atlantic Oceans. All batoid catches come from the northwest Pacific.

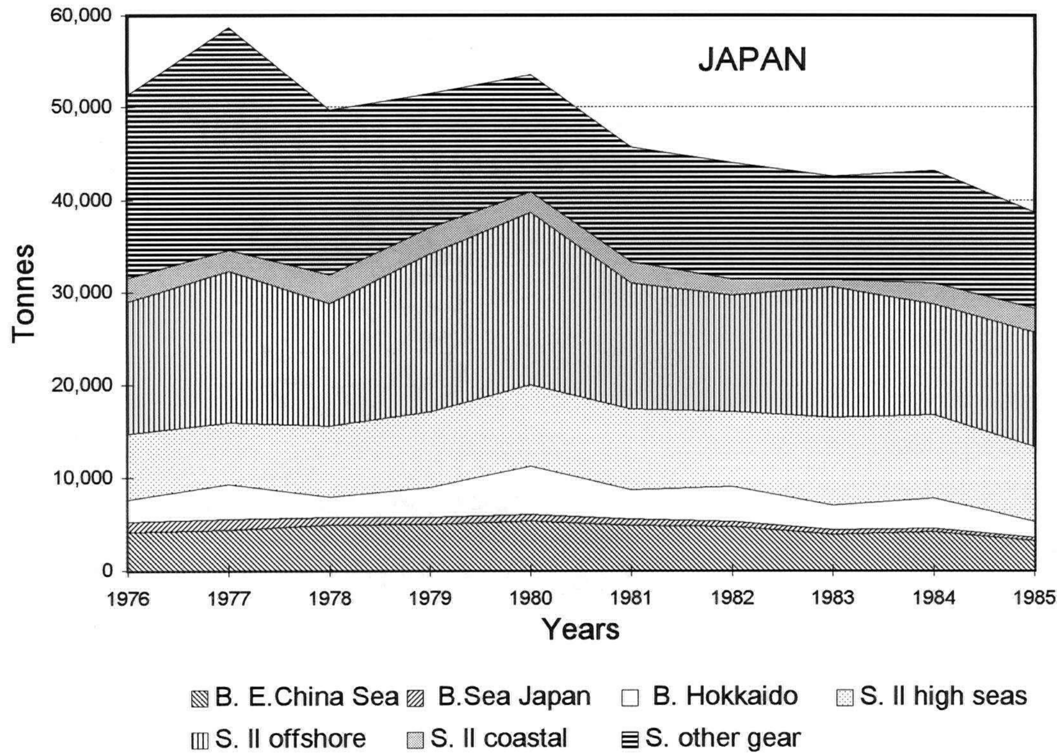



Figure 2.20 Elasmobranch catches in different fisheries of Japan during 1976-1984 (S=sharks, B=batoids, ll=longline). (Data from Taniuchi (1990) and Ishihara (1990)).

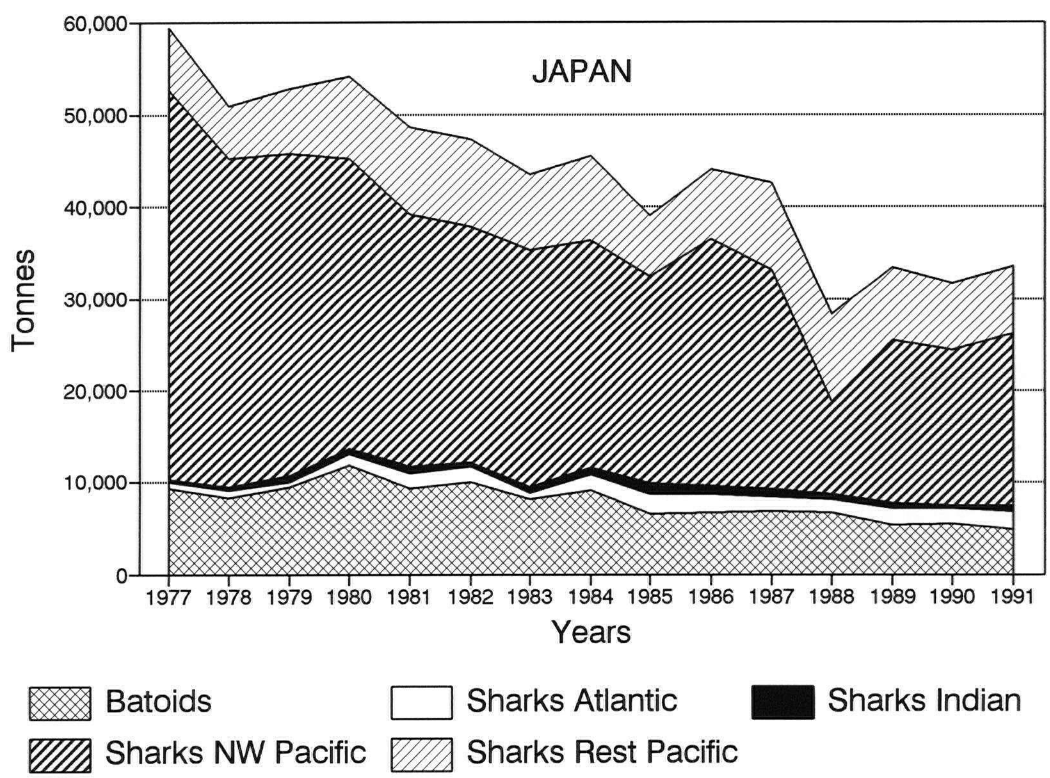



Figure 2.21 Elasmobranch catches of Japan, by species groups and region, during 1977-1991. (Data from FAO).

Detailed information on the species composition of the catches is not available for Japanese statistics after 1968. However, Taniuchi (1990) presents data for the period 1951-1967 and reports spiny dogfish *Squalus acanthias* as the main species in the catch up to 1958, followed in importance by blue shark *Prionace glauca* and salmon shark *Lamna ditropis*. The same author lists 25 shark species captured by tuna longline vessels. Considering the large contribution of the shark bycatches of longline fisheries to the total shark catch, and the research cruises data reported by Taniuchi (1990), the most important species in the overall shark catches in Japan should be, by importance, the blue shark *Prionace glauca,* the silky shark *Carcharhinus falciformis,* the oceanic whitetip shark *C. longimanus* and the shortfin mako *Isurus oxyrinchus.* However, this estimate might be affect by the selection and discard at sea and by the species composition of that part of the total shark catch that does not come from the tuna longliners. In the East China Sea, *Raja boesemani, R. kwangtungensis* and *R. acutispina* are respectively the most important species in the batoid catch (Yamada 1986).

The uses given to elasmobranchs in Japan vary from human consumption of meat and even cartilage in various traditional dishes, to industrial and medicinal uses of liver oil compounds and leather from hides. However, Japanese fishermen consider sharks a nuisance as they damage fishing gear and hooked tunas and billfishes, and are even considered competitors for some valuable fish stocks (Taniuchi 1990). No management measures are known to exist for elasmobranch fisheries in Japan.

South Korea.

The records of South Korean elasmobranch fisheries are intermittent and limited to FAO statistics. However, this country has taken more than 10,000 t/yr of elasmobranchs since at least 1948 and catches show an increasing trend oscillating around 20,000 t/yr since the mid-80's(fig. 2.2). The recent catch of sharks and rays of South Korea contributes 2.67 % of the total world elasmobranch catch (table 2.2). Given the large fisheries production of South Korea, elasmobranchs are of minor importance representing only 0.66 % of the total catches (1987-1991).

The elasmobranch fisheries of South Korea are very poorly documented. There are no reports on catch composition by species. From FAO data (1977-1991), two major categories are identified as batoids and "various elasmobranchs", the latter probably referring to sharks (fig. 2.22). During this period, batoids constituted 73 % of the elasmobranch catch and were taken chiefly in the Pacific Ocean (94 %), with small catches in the Atlantic (4 %) and the Indian Oceans (<1 %). Various elasmobranchs were also taken mainly from the Pacific Ocean (88%) and in small quantities from the Atlantic (9%) and Indian Oceans (3%). Although batoids comprise the majority of the elasmobranch catch according to FAO statistics, this only represents the actual landings of elasmobranchs and does not include discards. South Korean markets may, to some extent, influence the discard procedures of elasmobranchs at sea. The Korean longlining tuna fleet is known to catch and probably discard large numbers of sharks in the high seas of the world (see section 2.2.3).

People's Republic of China.

There is no information on the elasmobranch fisheries of the People's Republic of China in FAO statistics. Attempts to obtain information directly from the fisheries agency of China received the answer that no information on elasmobranch fisheries exists there. However, it is known that China has been exporting increasing quantities of shark fins to Hong Kong during the past few years, so that a harvest of sharks must exist there even as an incidental catch. A rough estimate based on data from the Southeast Asian Fisheries Development Center (SEAFDEC) on shark fins exports into Southeast Asian countries (P. Wongsawang, SEAFDEC, Samutprakan, Thailand, pers. comm. April 1992) indicates that China's shark catch apparently grew from less than 100 t in 1981 to somewhere between 17,000 t and 28,000 t in 1991, depending on which conversion factor is used for the estimation (fig. 2.23). These figures are minimum estimates of the real catches of sharks in China as an unknown part of the production might be consumed domestically. Thus, the actual catches are expected to be much higher. According to Cook (1991), due to recent relaxation in import and consumer restrictions in China, demand for the traditional shark fin soup has soared, creating extra demand for the product. In addition to the expansion of imports of this commodity mentioned by Cook, this demand for shark fins must be causing increased exploitation of elasmobranchs.

Zhow and Wang (1990) provide some information that confirms the existence of catches of

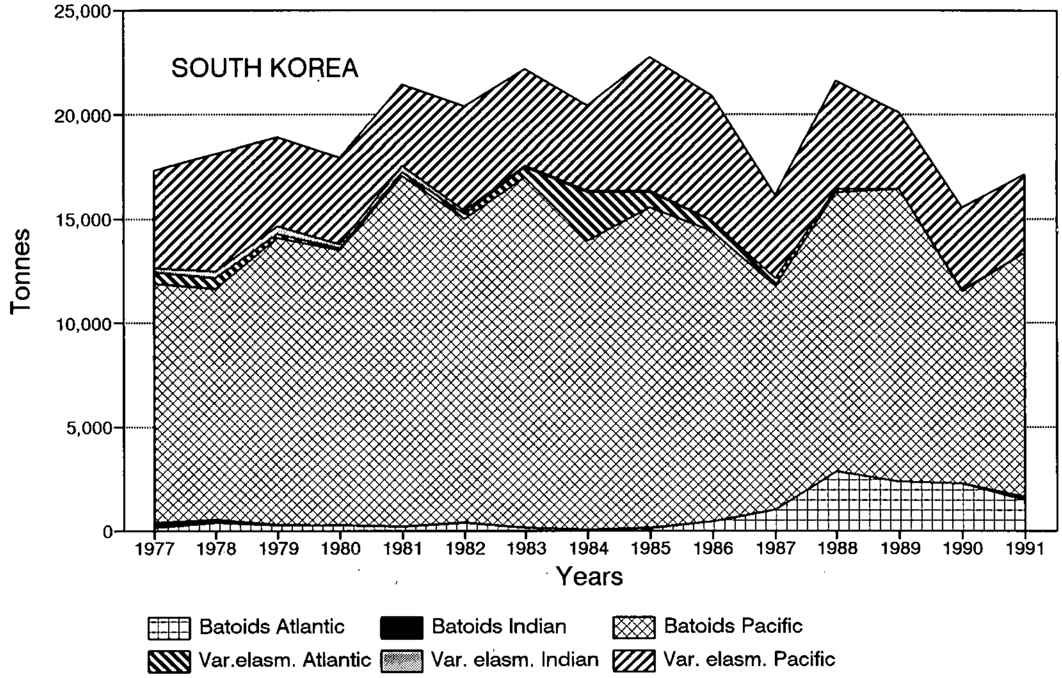



Figure 2.22 Elasmobranch catches of South Korea, by species groups and region, during 1977-1991. (Data from FAO).

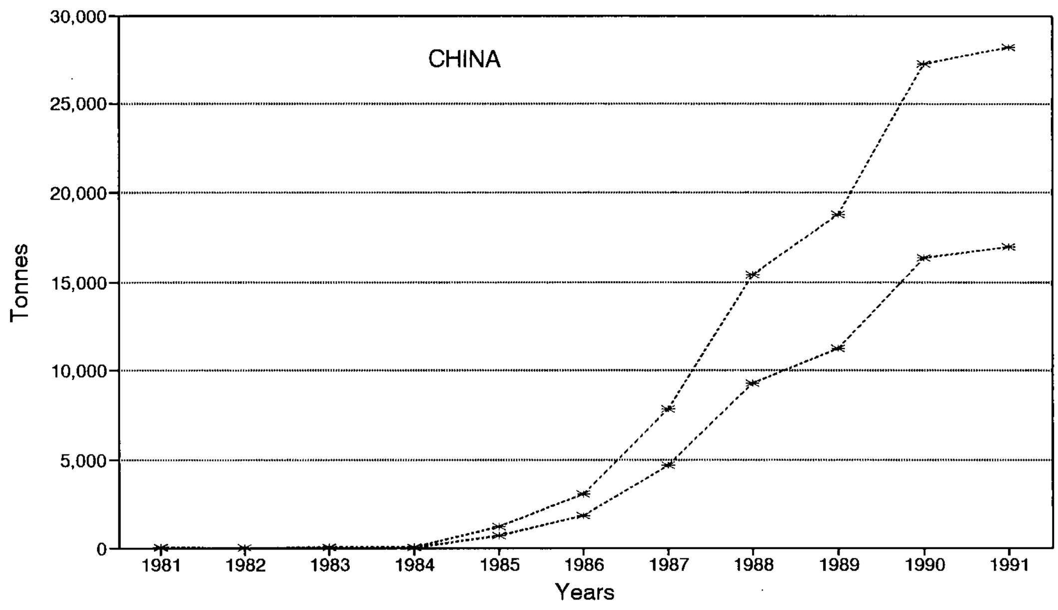



Figure 2.23 Estimated shark catches for the People's Republic of China from fin exports, using 3% and 5% conversion factor. (Fin export data from P. Wongsawang, pers. comm.)

sharks and rays in the People's Republic of China. Sharks and rays are targeted in some instances using driftnets, set gillnets and longlines. There are more than 3.5 million gillnets fishing in Chinese waters. Driftnets range from 30 mm to 360 mm mesh size, but probably those targeting elasmobranchs belong to the upper part of this range. Driftnets target sharks in Xiapu and Jinjiang, Fujian Province. Set gillnets occur in mesh sizes 30-320 mm and are used in shallow waters to target among many other species, *Triakis scyllium* and *Squalus fernandinus* in Haiyang, Shandong Province. Set longlines of different types are used to catch various elasmobranchs. They all vary between 388 and 500 m long. *Prionace glauca* and *Carcharhinus* spp. are targeted with longlines in Hui'an, Fujian Province, "various sharks" are caught in Yangjiang, Guangdong Province and "various rays" in Changdao, Shandong Province. A variation of longlines called rolling lines are used to catch rays in Haixin, Hebei province, Minhou, Fujian Province, and Rudong, Jiangsu Province. Rolling lines consist of non-baited sharp hooks narrowly spaced on the main line.

Taiwan.

This country has some of the most important elasmobranch catches in the world, comprised mainly of sharks. There is no comprehensive information on elasmobranch catches before the 70's for Taiwan, but data from the Fisheries Yearbooks of Taiwan Area indicate that large quantities of elasmobranchs have been harvested since the 50's (fig. 2.2). Total elasmobranch catches fluctuated around 45,000 t/yr during the period 1979-1988. This was followed by a substantial increase of catches in 1989 and especially 1990 when yield soared to more than 70,000 t as a result of increased catches of large sharks (fig. 2.24). These variations in catch probably represent changes in the discard rates of the distant fleet. Elasmobranchs comprised about 3.5 % of the total fisheries catches of Taiwan from 1987-1991. The majority of these catches were large sharks, i.e. approximately 81% of the total elasmobranch catch during 1978-1990. Small sharks account for approximately 14 %, while rays are of very little importance contributing only about 5 %. The main species in the elasmobranch catch are hammerhead sharks (*Sphyrna lewini*, *S. zygaena)*, grey sharks (*Carcharhinus plumbeus*, *C. falciformis)*, mako sharks (*Isurus oxyrinchus)*, blue sharks (*Prionace glauca)* and thresher sharks (*Alopias superciliosus*, *A. pelagicus)* (C.T. Chen, National Taiwan Ocean University, pers. comm. January 1992).

Most of the shark catches of Taiwan are obtained outside Taiwanese waters by the various far-seas tuna fleets. During 1988-1990, approximately 85% of the large shark and 70% of the small shark catches came from these operations. In contrast, most of the ray catches (53%) in the same period were taken in Taiwanese waters by local fisheries. The Taiwanese far-seas fleet is difficult to monitor as it operates in all the oceans of the world and is composed of multiple sizes and types of vessels, such as longliners, driftnetters, and purse seiners (Ho, 1988). It is known that important shark catches are taken by large-scale driftnetters targeting sharks specifically in Indonesian waters in the region of the Arafura, Banda and Timor Seas.

Taiwan operated an important fishery for sharks in Northern and North Western Australia waters from 1972 to 1986. This was mainly composed of driftnetters setting multifilament nylon nets of between 3 and 16 km long, 17-30 m deep and with mesh sizes of 140-170 mm. Vessels ranged in tonnage between 160 and 380 tonnes (Okera et al. 1981, Stevens 1990). In addition, Taiwanese pair trawlers fishing for demersal fish obtained shark bycatches in approximately the same grounds of the driftnetters. The catches of driftnetters were 80 % sharks; of these, *Carcharhinus tilstoni* and *C. sorrah* were the main component (55% of total catches), the remaining part were tuna and mackerel (Stevens 1990). According to Okera et al. (1981), between 3,500 and 14,800 t/yr of sharks were taken by these driftnetters during the period 1975-1980, however, Stevens and Davenport (1991) report catches equivalent to between 7,200 and 11,200 t/yr live weight for the same period. Meanwhile, catches from pair trawlers averaged $\approx$ 2,300 t/yr of sharks, with up to 7,300 t taken in 1974 (Okera et al. 1981). Limits on number of vessels and fishing areas as well as a catch quota of 7,000 t processed weight were imposed to this fishery in 1979 by the Australian government. The Taiwanese shark driftnet fleet pulled out of the fishery in 1986 following the imposition of a maximum gillnet length of 2.5 km by Australian authorities, which led to unprofitable business (Stevens 1990); Taiwan has since continued the fishery in Indonesian waters. At least 7,000 t/yr of sharks were taken by the Taiwanese fleet in Australian EEZ before 1987, but it is unknown how much they take presently in Indonesia. If the SEAFDEC figures reported for Taiwanese large-scale gillnet shark catches correspond solely to the fishery in Indonesian waters, then 19,636 t were taken there in 1987. Also, bycatches of sharks in other important large-scale Taiwanese fisheries, for example the tuna longline fishery, the Indian Ocean driftnet fishery and North Pacific squid driftnet fishery, might account for part of the shark catches of this country but these remain unknown. These high-seas fisheries are treated in detail in section 2.2.3.

According to data from the Fisheries Yearbooks of Taiwan Area, during 1988-1990 the main fishing localities for large sharks were Ilan Hsien and Pingtung Hsien accounting for 32 % (2,109 t/yr) and 49% (3,246 t/yr) of the large sharks caught in Taiwanese waters. Keelung Hsien was the main site for catches of small sharks and rays with 37% (991 t/yr) and 73% (875 t/yr) of the local catches of each group respectively.

Most of the Taiwanese shark catches are taken by large-scale fisheries, particularly with longlines. According to SEAFDEC data, about 90% of the 9,529 t domestic elasmobranch catches (those taken in the South China Sea Area) in 1988 came from large-scale fisheries. For sharks, large-scale longlines and hook and lines accounted for 62% of the catches while gillnets and otter trawls accounted for less than 20% each (table 2.7). Only 5% of the shark catch came from small-scale gillnet fisheries and less than 1% from traps and longlines. For rays, otter trawl was the most important large-scale gear with 23% of the catch, but gear classified as large-scale "others" provided 58%. Gillnets contributed to 7% of the small-scale catch. The remaining 11% of ray catches was taken using small-scale gillnets and traps.

It is unknown if any stock assessment has been done for the Taiwanese fisheries. Nevertheless, elasmobranch stocks in Taiwan are believed to be overexploited and tiger sharks (*Galeocerdo cuvieri*) are considered endangered species (C.T. Chen, pers. comm. op. cit.). Despite this, no management measures exist at present or are being considered in the near future for the elasmobranch fisheries of Taiwan.

Malaysia.

Malaysian elasmobranch fisheries are one of the smallest among Asian major elasmobranch-fishing countries together with those of the Philippines and Thailand. Malay catches of sharks and rays make only 2.46 % of the world elasmobranch catch. The development of the fishery in Malaysia shows a steady trend of slow growth from 1961 until the current level of about 15,000 t/yr (fig 2.2). Elasmobranchs represent 2.2 % of the total fishery production of Malaysia. Rays dominate the catches. SEAFDEC data indicate that from 1976-1991 rays represented an average of 60 % of the elasmobranch yield and sharks the remaining 40 %. Catches of sharks showed overall a very slightly declining trend, while ray catches expanded mainly from 1986-1991 (figure 2.25). Main species in the ray catches are *Rhyncobatus djiddensis* (which is processed as "shark fin" together with other ray species), *Gymnura* spp. and *Dasyatis* spp. *Scoliodon sorrakowa*, *Chiloscyllium indicum* and *Sphyrna* spp. are the most common species in the shark catches (C. Phaik, pers. comm. February 1992).

Elasmobranch catches in Malaysia are predominantly bycatches of trawl fisheries (95% of the catch) with only a small amount taken in small scale directed fisheries (5%). In both coasts of Peninsular Malaysia and the Sabah coast, between 60% and 70 % of the local shark catches were taken with trawls, while those of rays were in the order of 72-93%. Purse seines caught less than 1% of sharks in Peninsular Malaysia. In the waters of Sarawak, large scale otter trawls take 70% of the local catches of rays and 30% of those of sharks. In this area, other various large-scale gears accounted for less than 1% of catches of both sharks and rays.

Malaysian small-scale fisheries for elasmobranchs, although not as important as large-scale fisheries for their contribution to total elasmobranch catches, are very diverse. During 1988, they comprised 70% of the Sarawak shark catches using mainly gill nets (54%) and longlines and hook and line (15%), with traps making a very small contribution (table 2.7). Ray were taken in small-scale fisheries using hook & line and longlines (17%) and gillnets (11%); small catches were also taken with traps. For both coasts of Peninsular Malaysia and Sabah, small scale fisheries with gill nets took between 15% and 28% of the shark catches, while hook & line and longlines accounted for about 9% of the catch in Peninsular Malaysia and 25% in Sabah. Catches of rays from small-scale fisheries in Sabah and the west coast of Peninsular Malaysia were taken mainly by hook & line and longlines and to a lesser extent by gillnets traps and other gear. The opposite was found in the east coast of Peninsular Malaysia, where most of the small contribution (5%) of small scale fisheries to the total rays catch came from gillnets.

Because of the incidental nature of elasmobranch catches, the most important fishing

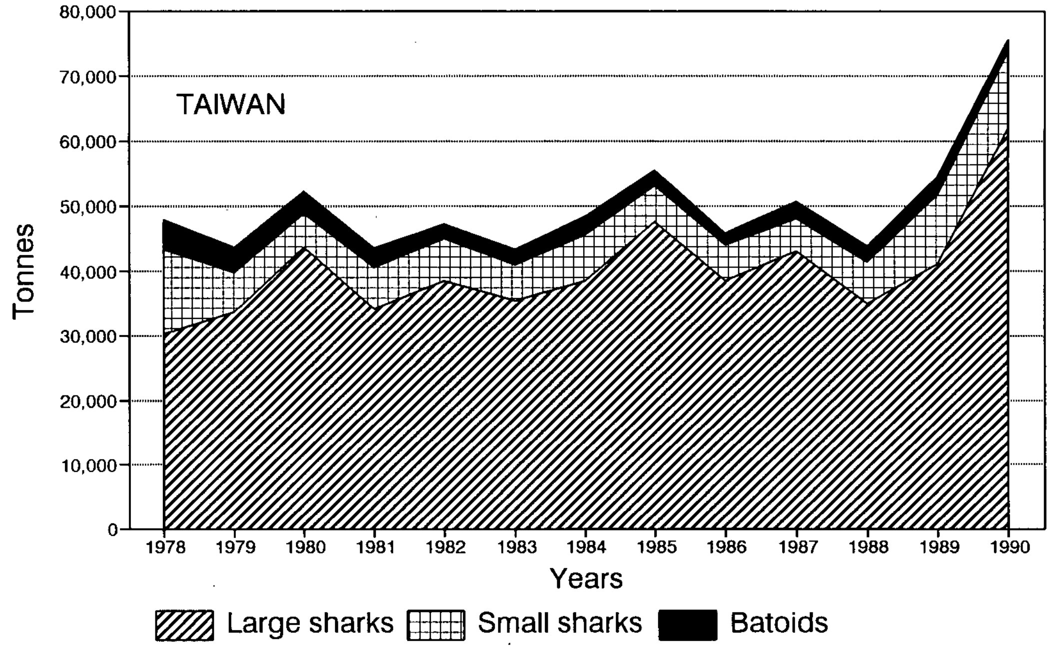



Figure 2.24 Elasmobranch catches of Taiwan, by species groups, during 1978-1990. (Data from FAO).

Table 2.7 Percentage catches of sharks and rays according to fishing gear and zones in Taiwan (Prov. of China) and Malaysia (data from SEAFDEC 1988).

| | TAIWAN | | PENINSULAR MALAYSIA | | | | INSULAR MALAYSIA | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| TYPE OF FISHERY | | | WEST COAST | | EAST COAST | | SABAH | | SARAWAK | |
| AND GEAR | SHARKS | RAYS | SHARKS | RAYS | SHARKS | RAYS | SHARKS | RAYS | SHARKS | RAYS |
| LARGE SCALE | | | | | | | | | | |
| Purse seine | - | - | 0 | - | 0 | - | - | - | - | - |
| Trawl | - | - | 63 | 80 | 70 | 93 | 60 | 72 | - | - |
| Otter trawl | 11 | 23 | - | - | - | - | - | - | 30 | 70 |
| Gill net | 17 | 7 | - | - | - | - | - | - | - | - |
| Hook & line | 62 | 0 | - | - | - | - | - | - | - | - |
| Others | 4 | 58 | - | - | - | - | - | - | 0 | 0 |
| SMALL SCALE | | | | | | | | | | |
| Gill/drift net | 5 | 4 | 28 | 4 | 20 | 5 | 15 | - | 54 | 11 |
| Hook/long line | 0 | - | 8 | 16 | 9 | 0 | 25 | 26 | 15 | 17 |
| Trap | 0 | 7 | - | 0 | 0 | 0 | 0 | 0 | 1 | 2 |
| TOTAL CATCH (mt) | 8588 | 941 | 1359 | 6125 | 1111 | 2303 | 910 | 596 | 1872 | 2546 |

grounds are those of the trawling fishery, mainly peninsular Malaysia and Sarawak. Average catches for 1976-1989 indicate that sharks are taken mainly in Sarawak (1,869 t/yr or 15% of total elasmobranch catch), the west (1,363 t/yr or 11%) and east (1,169 t/yr or 9%) coasts of Peninsular Malaysia and in lower quantities in Sabah (778 t/yr, 6%). Catch trends for sharks in these areas show a decrease in west Peninsular Malaysia, relatively sustained yields in Sarawak and Sabah and variability in east Peninsular Malaysia (fig. 2.25). The west coast of Peninsular Malaysia is the most important fishing area for rays (3,457 t/yr, 28% of total elasmobranch catches) followed by Sarawak (2,004 t/yr, 16%) and the east coast of Peninsular Malaysia (1324 t/yr, 11%), with Sabah contributing only 573 t/yr (5%). The trend of the catches indicates growth in ray yields in both coasts of Peninsular Malaysia, relative stability in Sabah and strong variability in Sarawak. At present, there are no management measures for elasmobranchs, and only ray catches are indirectly controlled via the licence restrictions for trawl fisheries.

Philippines.

The elasmobranch catches of the Philippines were of minor importance before the late 70's and although variable, show a growing trend and recent stability around 17,000 t/yr since 1986 (fig 2.2). Sharks and rays comprised only 0.85 % of the total national catches. According to SEAFDEC data, rays are slightly more important than sharks in the catches with an average of 53% of the elasmobranch yields in the period 1977-1991; the catches of both groups had a growing trend during this period. Philippine catches account for 2.63 % of the world elasmobranch yield.

Judging from the catches of 1988 (17,879 t), small scale fisheries provide the large majority of elasmobranch catches in Philippines (table 2.8). In Luzon, large scale trawlers accounted for 30% of the local shark catches but only 6% of rays, with purse seiners contributing around 3% of both groups' catches. In Visayas, trawls were the main gear in large scale fisheries for rays (23%) but accounted only for 1% of shark catches. Large scale purse seining took 11% and 8% of the shark and ray catches respectively in Visayas. Catches from small-scale fisheries for both sharks and rays in Luzon and for sharks in Visayas were mainly taken by hook & line and longlines (38%-76%) but also by gillnets (8%-30%). The opposite happens in Visayas where gillnet catches of rays were greater than those from hook & line and longline (42% vs. 22%). In Visayas and Luzon, "other gear" made small contributions (< 13%) to the catches of both sharks and rays, and traps were used to obtain minor catches of rays (< 8%). Elasmobranch catches in Mindanao were all from small scale fisheries: gillnets were the main gear for rays (81%) and hook and line for sharks (57%). Small scale gear classified as "other" were the second most important for catching both groups in Mindanao (28% of sharks, 10% of rays). Gill nets took 15% of the small-scale shark catches and traps less than 1%. For rays, hook & line and longlines were the third most important gear in this area with 7% of the catches and traps and otter trawls contributed minimum catches.

The composition of batoid and shark catches by area in the Philippines is shown in figure 2.26 based on SEAFDEC data. Most of the catches of both sharks and rays are taken in Mindanao, averaging 3,185 t/yr (24% of total elasmobranch catches) and 2,724 t/yr (21%) respectively. The yield of sharks and rays in Mindanao has grown since the late 70's. Luzon is the second area in importance with 1,993 t/yr of sharks (15%) and 2,312 t/yr of batoids (18%). Shark catches in Luzon have decreased from the levels of the late 70's while batoid yields have grown recently after a decrease in the early 80's. Yield of sharks and rays in Visayas is the lowest in the Philippines with averages of 1,108 t/yr (8%) and 1,856 t/yr (14%) respectively; yields of both groups decreased shortly during the early 80's.

Little is known about the species composition of elasmobranch catches in the Philippines. Warfel and Clague (1950) report tiger sharks as the prime catch in shark longlines around the Philippines during exploratory fishing. Other sharks found in the survey include at least six species corresponding to the genus *Carcharhinus*, plus *Sphyrna zygaena, Scyliorhinus torazame, Hexanchus griseus* and an unidentified nurse shark. The species taken by gillnets were *Pristis cuspidatus* and *Rhynchobatus djiddensis.* Additionally, Encina (1977) reports on an budding dogfish fishery catching *Squalus acanthias* and *Centrophorus* spp. all around the Philippines, primarily directed towards squalene oil extraction.

Thailand.

Now one of the more modest major elasmobranch-fishing countries in Southeast Asia, Thailand has an elasmobranch fishery that grew considerably in the 60's but declined since

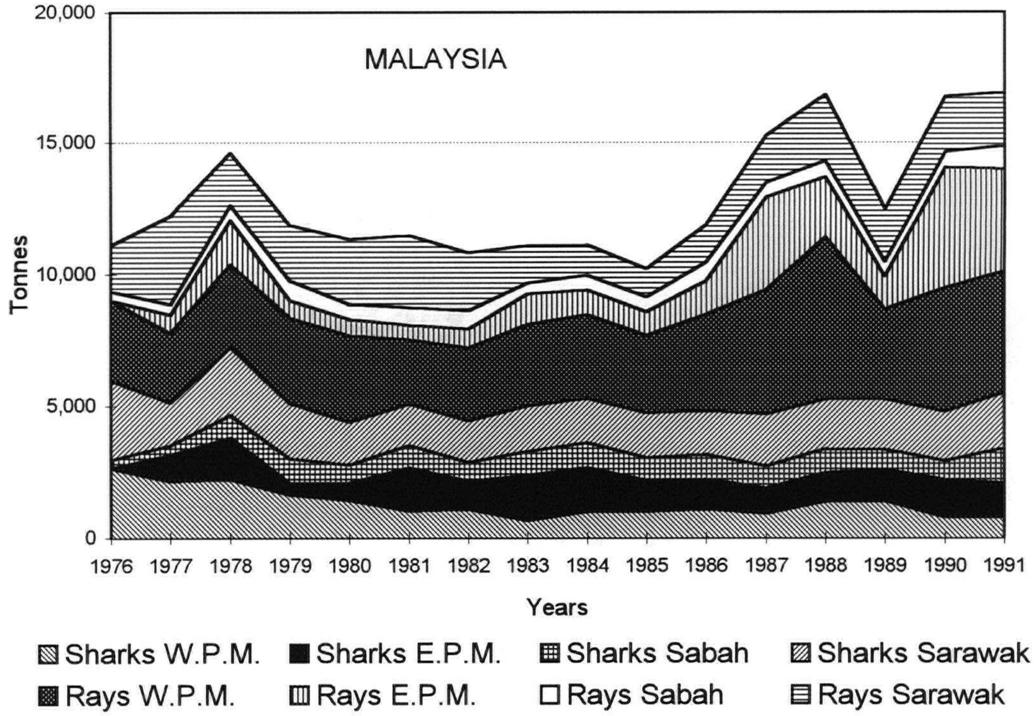



Figure 2.25 Elasmobranch catches of Malaysia, by species groups and region, during 1976-1990 (E.P.M.=eastern penninsular Malaysia, W.P.M.=western penninsular Malaysia). (Data from SEAFDEC).

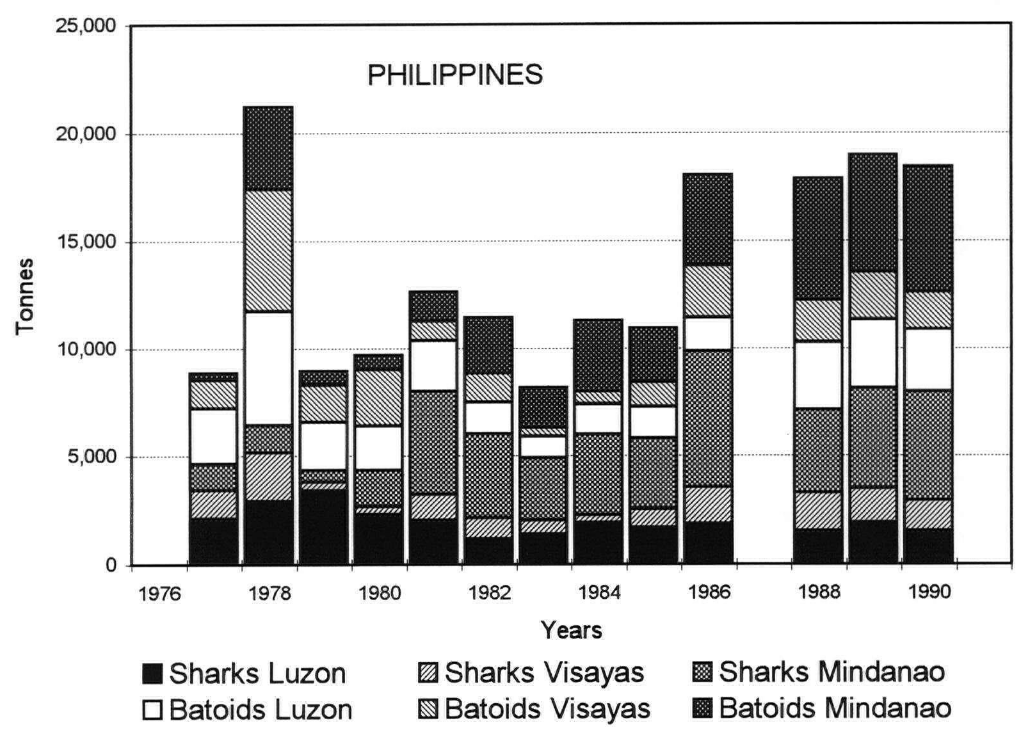



Figure 2.26 Elasmobranch catches of Philippines, by species groups and region, during 1976-1990. (Data from SEAFDEC).

the early 70's (fig. 2.2) mainly as a consequence of over-exploitation by trawlers in the Gulf of Thailand (Menasveta et al. 1973, Pope 1979). More recently, there were signs of an apparent recovery but catches fell again since 1988 and the present state of the stocks is uncertain. Sharks and batoids represent a minor fishery in Thailand contributing only 0.43 % of the total national fishery production during 1987-1991, and only 1.74 % of the world elasmobranch catch (table 2.2).

Rays dominate the catches, and are a bycatch of the predominately trawling Thai fishery. SEAFDEC data show that average catches of rays for the period 1976-1991 accounted for 64 % of the elasmobranch yield, while the rest were sharks. Estimates of the Thai Department of Fisheries show that approximately 95 % of the shark catch is made up of sharks smaller than 1.5 m TL, mainly *Carcharhinus* spp., while the main batoid species in the catch are *Dasyatis* spp. and various eagle rays. (P. Saikliang, D.O.F. pers. comm. December 1991).

The main fishing grounds for sharks and rays are in the Gulf of Thailand. During 1976-1989 catches from the Gulf averaged 2,955 t/yr of sharks (28% of all elasmobranchs caught) and 4,885 t/yr of rays (46%), while the Andaman Sea only produced 1,042 t/yr of sharks (10%) and 1,709 t/yr of rays (16%). There was no trend in shark catches during this period in the Gulf of Thailand but there was a decreasing trend in the Andaman Sea. Rays catches grew considerably in the Gulf of Thailand but showed no trend in the Andaman Sea (fig 2.27).

Thai elasmobranch fisheries are chiefly a large-scale activity. From a total of 11,438 t of elasmobranchs taken in 1988 by Thailand, most of the catches on both coasts of the country came from large-scale trawlers. Otter trawls provided 63% and 82% respectively, of the shark and ray catches of the Gulf of Thailand and 92% and 64% of those in the Andaman Sea coast. Additionally, pair trawls in the Gulf of Thailand took around 10% of both sharks and rays (table 2.8). In the Gulf of Thailand, large-scale gillnets accounted for 22% of shark catches but only for 1% of those of rays. Further, purse seiners contributed with very small catches of both groups. In the Andaman Sea, small shark catches were taken by large-scale gill nets. Small-scale elasmobranch fisheries in Thai waters are relatively important for their catches of rays with gill nets in the Andaman Sea, where they contribute almost 30% of the local ray catches. Small catches (less than 1 % to 7% of local

Table 2.8 Percentage catches of sharks and rays according to fishing gear and zones in Philippines and Thailand (data from SEAFDEC 1988).

| TYPE OF FISHERY AND GEAR | PHILIPPINES LUZON SHARKS | PHILIPPINES LUZON RAYS | PHILIPPINES VISAYAS SHARKS | PHILIPPINES VISAYAS RAYS | PHILIPPINES MINDANAO SHARKS | PHILIPPINES MINDANAO RAYS | THAILAND GULF SHARKS | THAILAND GULF RAYS | THAILAND INDIAN OCEAN SHARKS | THAILAND INDIAN OCEAN RAYS |
|---|---|---|---|---|---|---|---|---|---|---|
| LARGE SCALE | | | | | | | | | | |
| Purse seine | 3 | 2 | 11 | 8 | - | - | 1 | 0 | - | - |
| Trawl | 30 | 6 | 1 | 23 | - | - | 12 | 10 | - | 0 |
| Otter trawl | - | - | - | - | - | - | 63 | 82 | 92 | 64 |
| Gill net | - | - | - | - | - | - | 22 | 1 | 4 | - |
| Hook & line | 2 | - | - | - | - | - | - | - | - | - |
| Others | - | 0 | 0 | - | - | - | - | - | - | - |
| SMALL SCALE | | | | | | | | | | |
| Otter trawl | - | - | - | 1 | - | 0 | - | - | - | - |
| Gill/drift net | 21 | 30 | 8 | 42 | 15 | 81 | 1 | 3 | - | 29 |
| Hook/long line | 38 | 42 | 76 | 22 | 57 | 7 | 0 | 4 | 4 | 7 |
| Trap | - | 7 | - | 3 | 0 | 1 | - | - | - | - |
| Others | 6 | 12 | 3 | 4 | 28 | 10 | - | - | - | - |
| TOTAL CATCH (mt) | 1513 | 3132 | 1742 | 1924 | 3879 | 5689 | 3436 | 5963 | 408 | 1631 |

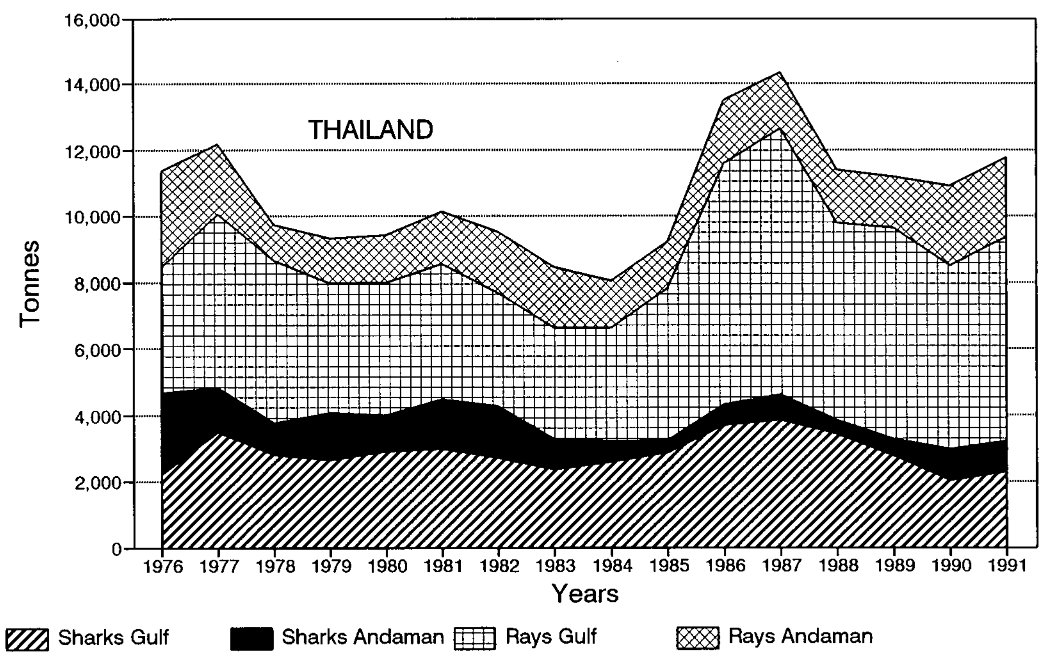



Figure 2.27 Elasmobranch catches of Thailand, by species groups and region, during 1976-1991. (Data from SEAFDEC).

catches) of both groups are taken also in small-scale hook & line and longline fisheries in both coasts. In the Gulf of Thailand, small-scale gillnets take only small catches of sharks and rays.

There seem to be no recent stock assessments for the area. Studies based on 1963 and 1966-1972 research cruises' swept area estimates (Menasveta et al. 1973), indicated total standing stocks of 2,880 t for sharks, 4,404 t for rays and 1,988 t for rhinobatids in the whole Gulf of Thailand, as well as an estimated 5,000 t potential yield for all elasmobranchs. The study highlighted severe reductions in standing stocks of rays over that period and classified elasmobranch stocks as "heavily exploited, if not too heavily already". However, these estimates might have been too conservative as total Gulf catches of elasmobranchs from Thailand and Malaysia were 10,439 t in 1977, 10,959 t in 1978 and 7,621 in 1979, maintaining a level of about 8,000 t/yr for another 6 years, and rising above 10,000 t/yr in the late 80's. Nevertheless, the reductions in catch rates shown by Pope (1979) are evidence that the stocks of both sharks and rays have indeed declined dramatically in the area.

Indonesia.

There is no information on the elasmobranch fisheries of Indonesia before 1971, but records show they have expanded tremendously since then. Indonesian elasmobranch fisheries have the highest sustained growth rate of any elasmobranch fishing country and they are currently the largest in the world. Indonesian catches amounted to almost 80,000 t in 1991 and there are no signs yet of any levelling off (fig. 2.2). Indonesian fisheries for sharks and rays represent 10.18 % of the world's elasmobranch commercial catch. Despite this, elasmobranchs are of only moderate importance in Indonesia, contributing 2.41 % to the total fisheries of this country in the period 1987-1991. Contrary to most major elasmobranch fishing countries in the region, which harvest larger quantities of rays than sharks or similar quantities of both, elasmobranch catches in Indonesia are dominated by sharks, which accounted for 66 % of the average elasmobranch catches during 1976-1991.

According to SEAFDEC data (1976-1989) the most important areas for shark fishing in Indonesia are in the western part of the country, namely Java (9,727 t/yr on average and

21% of total elasmobranch yields), Sumatra (7,837 t/yr, 17%) and Kalimantan (5,870 t/yr 12%), with the eastern provinces of Bali-Nusa Tengara, Sulawesi and Molluca-Irian Jaya, accounting for 1,796 t/yr (3.8%), 3,157 t/yr (7%) and 1,983 t/yr (4.2%) respectively. This pattern is similar for batoid catches, but in this case Sumatra is at the top with 6,404 t/yr (13% of total elasmobranch catches), followed by Java with 4,670 t/yr (11%) and Kalimantan with 2,987 t/yr (6%). In the east, Sulawesi ranks first with 1,329 t/yr (3%), Bali-Nusa Tengara second with 957 t/yr (2%) and Molluca- Irian Jaya third with 518 t/yr (1%). The catches of sharks and rays show increasing trends over the period in all provinces (fig. 2.28). Eastern provinces could be the most suitable for future increases in the fishery. However, in addition to the Indonesian catches, large quantities of sharks are also harvested by Taiwanese driftnet vessels in eastern Indonesian waters since these fleet abandoned the Australian EEZ in 1987. The Taiwanese vessels were capable of taking at least 7,000 t/yr of sharks, and catches in the area between north Australia and Indonesia were in the region of 25,000 t/yr before 1979 (Stevens, 1990).

In the light of the overall catches of elasmobranchs taken in Indonesian waters, it is surprising that yields from Indonesia keep growing year after year. There are apparently no research or management programmes for elasmobranchs in Indonesia and the question of the sustainability of shark fisheries in the area becomes more intriguing and relevant as catches keep growing. Much attention should be paid to this fishery if Indonesia has any interest in continuing it into the next century.

### 2.2.2.5 Australian subcontinent.

#### Australia.

Elasmobranch fisheries in Australia are small and barely classifiable as "major fisheries", having only temporarily produced more than 10,000 t/yr during the late 80's (fig. 2.2). They only contribute 1.46 % to the world elasmobranch catch (1987-1991). Nevertheless, Australian shark fisheries are among the most documented and one of the few managed elasmobranch fisheries in the world. This is probably directly related to the importance of elasmobranchs in the catches of Australian fisheries. FAO data for 1987-1991 indicate that elasmobranchs contribute 4.8 % of the total fisheries of Australia, the third highest percent

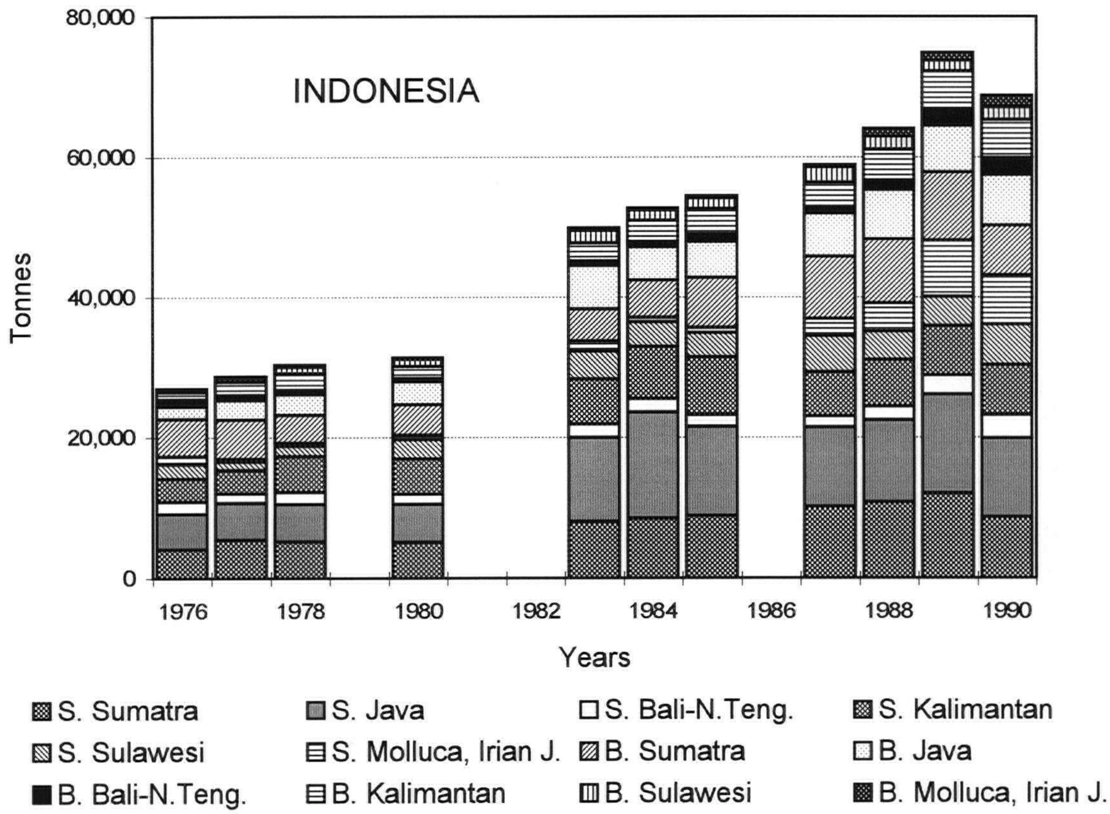



Figure 2.28 Elasmobranch catches of Indonesia, by species groups and region, during 1976-1990 (B=batoids, S=sharks). (Data from SEAFDEC).

importance in the world. Additionally, these are very old fisheries that form part of the fishing tradition of the country. Stevens (1990) reviews Australian shark fisheries and reports that their history dates back to the end of the 19th century, when fisheries for school shark liver oil and fins already existed in southeastern Australia.

FAO data are not reported by species or species groups and it is only possible to get the geographical composition of the catches from this information. The large majority of the catches come from Area 57 probably reflecting mainly the southern shark fishery for *Mustelus antarcticus* and *Galeorhinus galeus*. Small catches of elasmobranchs come from Area 81 while catches in Area 71 are negligible (fig. 2.29).

Historically, the most important elasmobranch fishery in Australia has been the southern shark fishery which provides the major part of the total elasmobranch catches of the country. Information for this particular fishery is summarised by Walker (1988), Anonymous (1989) and Stevens (1990). School sharks *Galeorhinus galeus* were the original target species, at least since 1927, when records began to be taken regularly. However, other species taken in the fishery are the gummy shark *Mustelus antarcticus,* the sawsharks *Pristiophorus cirratus* and *P. nudipinnis* and the elephant fish *Callorhynchus millii.* Management of the fishery began as early as 1949 when a minimum size of 91 cm TL was introduced for school sharks in Victoria. Protection of nursery areas in coastal lagoons followed later. The fishery expanded from coastal to offshore operations in the mid-40's and catches grew gradually until 1969. Yield was temporarily reduced following a combined effect of the introduction of monofilament gillnets and a ban by the government of Victoria of school sharks longer than 104 cm TL due to impermissibly high concentrations of mercury in their meat. The introduction of gillnets was intended to boost the decreasing catches of school sharks, but this also brought about large bycatches of gummy sharks, which were previously regarded as an undesirable species. Due to the size restrictions on school sharks and the availability of gummy sharks, the latter were displaced as the main species in the fishery. Soon after, revised size limits allowed school sharks between 71-112 cm TL to be taken again in the Victorian fishery and total catches rose once more attaining a peak of 3,754 t (dressed weight) in 1986, with both species contributing approximately equal parts to the catch. Since then, catches have slowly fallen as a result of management of the fishery.

Most of the catch in the southern shark fishery is taken with monofilament gillnets and longlines, but small catches are also taken by trawlers. Gillnets vary geographically in mesh size but they are all between 15 cm (legal minimum) and 20.23 cm, with 17.78 cm as the most common mesh size. Gillnets are typically 1.7 m in height and geared with a hanging coefficient of 0.6 (Kirkwood & Walker 1986). Gillnets account for about 90% of the gummy shark and approximately 75% of the school shark catches. Longlines are typically 10 km long and rigged with several hundreds of hooks. Although less important for their contribution to the total catches, their usage has grown lately especially in Tasmania. The most important fishing grounds for gummy shark are primarily in Bass Strait and secondarily in South Australia. The opposite was true for school shark until recently, when Tasmanian catches almost equalled those of each of the other areas. During 1987, total shark catches were distributed by gear and area as follows: in Bass Strait, gillnets 47.3%, longlines 7.4%; in South Australia, gillnets 27.3%, longlines 1.3%; in Tasmania, gillnets 10.9%, longlines 10.4% (Anonymous 1989).

The southern shark fishery is a model of honest concern over elasmobranch resources. Fishing effort has expanded in all areas while gillnet CPUE (kg/km/hr) has dropped for both species. This has recently led fisheries scientists to suspect that both stocks are in decline. As a result, a monitoring program and a special research group have been set up for the study of the fishery and several projects are being funded by the fishing industry and government agencies. The approach is comprehensive, with research spanning from biological studies (Moulton et al. 1992) and the construction of databases to building specific simulation models for the management of the fishery (Walker 1992, Sluckzanowski et al. 1993) and economic analyses (Campbell et al. 1991). The biology of the species is well documented and suggests single breeding populations for each species in the whole area. However, some concerns have been raised recently about the spatial structure and dynamics of the stocks. Current investigations concentrate on the spatial dynamics of the stocks and the vulnerability of juvenile school sharks to commercial and sport fisheries in nursery areas of Tasmania. The recent concerns about possible overexploitation of the stocks led to management measures aimed at reducing effort by about a 50% through an elaborate licensing procedure. Unfortunately, longline effort was not considered in the scheme and grew rapidly as a result of the restrictions imposed to gillnetters, causing that overall effort reductions fell short of expected levels. The southern shark fishery continues to be intensively studied and monitored.

There is also a smaller shark fishery in the south western and southern coast of Western Australia. Catches are dominated by *Furgaleus macki* and *Mustelus antarcticus* but substantial catches of *Carcharhinus obscurus* are also obtained (Lenanton et al 1990). Catches are about 1,600 t/yr and reportedly 10% of the Australian catch of gummy shark comes from this fishery. Management measures include license limitations, gear restriction and a recent ban which prohibits shark fishing in waters from Shark Bay northward to North West Cape (Anonymous 1992).

The northern Australia shark fishery was initiated in 1974 by Taiwanese gillnetters exploiting sharks, tuna and mackerel in offshore areas of the Arafura sea. Taiwanese pair-trawlers fishing in the same areas also caught sharks as bycatch (see account of Taiwanese fisheries above). Sharks made up approximately 80% of the catch with 55% composed by *Carcharhinus tilstoni* and *C. sorrah.* At the beginning of the 80's Australian fishermen became interested in these resources and small fisheries spread in inshore waters from the Northern Territory to the north of Western Australia and Queensland. Catch composition is similar to that of the offshore Taiwanese fishery and landings have fluctuated between 50 and 400 t/yr (Stevens 1990). Although stocks declined due to overexploitation by the Taiwanese fleet, the latter moved out to Indonesia in 1987 and the stocks are believed to be recovering. No management measures for the small domestic fishery are thought necessary at the moment. This fishery has also been closely monitored and several research projects have been conducted by the Northern Territory Department of Primary Industry and Fisheries and the CSIRO.

The future development of an Australian shark fishery in the north of Australia is constrained by high concentrations of mercury and selenium in most species of carcharhinids and sphyrnids. Lyle (1984) estimated that only 49% of the catch in weight could be retained if the maximum level of mercury is set to 0.5 mg/kg. In addition, market restrictions have precluded the catches from entering the main market for shark meat in Melbourne (Rohan 1981). Some recent arrangements have been made in the northern shark fishery to prevent overexploitation. Several endorsements have been allocated in different areas under Commonwealth jurisdiction since January 1992.

New Zealand.

Elasmobranch fisheries in New Zealand were under 10,000 t/yr until recently. Although current catches are not much larger, there is an overall increasing trend in yield since the late 70's (fig. 2.2). Elasmobranch fisheries are moderately important for New Zealand with catches making 2.19 % of the total national fishery production. New Zealand fisheries for sharks are another good example of continuous research and management. On a global scale, these fisheries are very small, contributing only 1.73 % of the world elasmobranch yield (table 2.2).

According to FAO data for 1977-1989, the catches of the different elasmobranch groups in New Zealand are quite variable. Dogfish catches (mostly *Squalus acanthias)* show a tremendous increase while catches of smooth-hounds show a clear decline. Batoid and elephant fish catches grew moderately and the catch of grey sharks (mostly *Galeorhinus galeus*) grew considerably then contracted during this period (fig. 2.30).

Recent information from the N.Z. Ministry of Agriculture and Fisheries indicates that during 1989-1992, approximately 15 % of the catch was composed of elephant fishes (*Callorhinchus milli*) and chimaeras (*Hidrolagus* spp.), 18 % was tope (*Galeorhinus galeus)*, 12.5 % was rig (*Mustelus lenticulatus),* 33 % was spiny dogfish (*Squalus acanthias)*, 17.5 % was the skates *Raja nasuta* and *R. innominata* and the remaining 4 % was comprised by 13 species of large or deepwater sharks and at least three species of batoids. About 40% of the total elasmobranch yield is a bycatch of trawl fisheries, while the remaining 60% is mainly taken directly with longlines and setnets. Elephant fishes are caught mainly in the coast of Canterbury and tope and rigs are caught all around New Zealand.

Francis and Smith (1988) analyse the catches of rig around New Zealand and summarise some information about this fishery. The rig fishery is strongly seasonal and concentrated during the austral spring and summer months. Catches are mostly exported to Australia. Almost 90 % of the catches were a bycatch of trawling fisheries during the mid 60's, but the increase in demand and introduction of monofilament gillnets changed the pattern of exploitation and presently setnets account for 80% of the landings on this species. Francis and Smith report that CPUE declined in three of the five zones analysed during 1974-1985

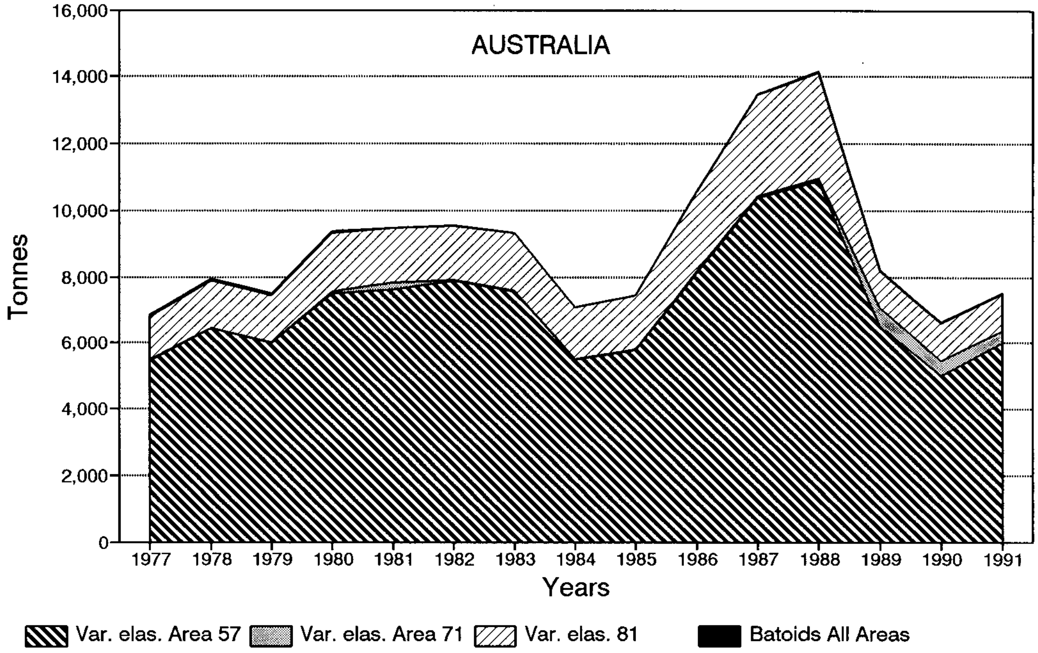



Figure 2.29 Elasmobranch catches of Australia, by FAO statistical areas, during 1977-1991. (Data from FAO).

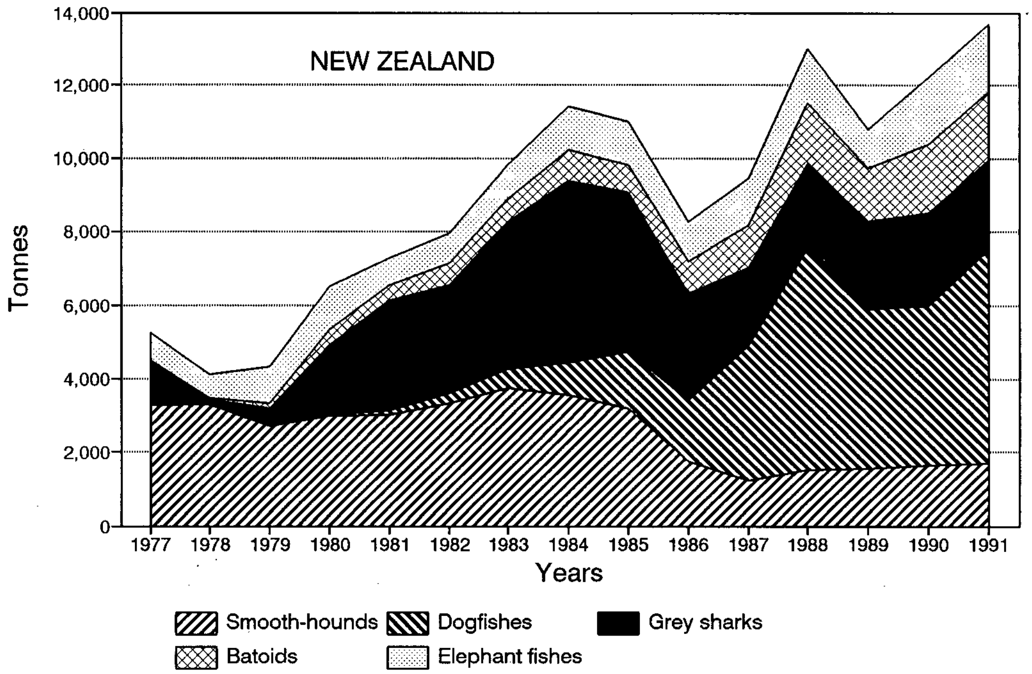



Figure 2.30 Elasmobranch catches of New Zealand, by species groups, during 1977-1991. (Data from FAO).

and that in several areas stock sizes appear to be down to one third of their original sizes. Presumably, this is partly the reason for the imposition of management regulations in this fishery.

Management measures for the main elasmobranch species in New Zealand include revisable TAC's, a percentage of which go to ITQ holders. For the year 1992, TAC's were 636 t for elephant fishes, 2,070 t for rig and 3,087 t for tope (Annala 1993). In addition, basking sharks can only be taken as a bycatch and there are current proposals to include more elasmobranch species under the quota management system. Research in New Zealand has concentrated in rig and spiny dogfish (Francis and Mace 1980, Hanchet 1988, Francis 1989, Massey and Francis 1989, Hanchet 1991, Francis and Francis 1993).

Some small quantities of livers from deep water squaloid sharks are currently utilised from the bycatches of the orange roughy (*Hoplostethus atlanticus)* deep trawl fisheries of New Zealand (King & Clark 1987), although large quantities of the sharks are also discarded at sea (see next section on bycatches of large-scale fisheries). Results from research cruises indicate that the stock of these deep sea sharks could sustain yields of no more than 2,250 t/yr.

2.2.3 Bycatches and Discards of Elasmobranchs at Sea.

Several large-scale fisheries operating in the high-seas around the world are known to capture elasmobranchs, particularly sharks, as a substantial bycatch. Although sharks are retained and utilised in some of these fisheries, most frequently they are simply thrown overboard, sometimes after their valuable fins have been chopped off.

The survival chances of these bycatches vary depending on the type of gear used. Trawl and gill nets, and perhaps purse seines too, almost certainly cause 100% deaths among sharks caught. While survival is higher in longlines because they permit sharks some limited movement and thus some respiration, survival rates depend highly on the metabolism and endurance of individual species. Overall, it is believed that most of the bycatches of sharks in large-scale fisheries face a very high mortality. This might not be true in the case of batoids which generally have very different mobility requirements in order to respire.

However, their catches are normally very small in large-scale high-seas fisheries due to their more demersal habits.

The amount of elasmobranchs killed in large-scale high seas fisheries and the rate of discards are poorly understood and have never been systematically assessed. Reports on the sharks taken by the countries involved in these fisheries do not reflect the real levels of incidental catches but most frequently only represent the amounts retained. The major purpose of this section to present the available information on the most important large-scale fisheries of the world and evaluate as far as possible the extent of their elasmobranch bycatches, the amounts taken and the total discards.

Until very recently, there were two main large-scale fisheries catching and discarding significant numbers of elasmobranchs in their operations, namely driftnet and longline fisheries. Due to international pressures and following UN resolution 44/225, all large-scale driftnet fisheries were phased out of international waters at the end of 1992. They are still discussed here however, due to the importance of their bycatches. In addition to longline and driftnet fisheries, other large-scale fisheries with minor elasmobranch bycatches (tuna purse seine and pole and line fisheries) are briefly discussed. The deep trawl fisheries for orange roughy are treated briefly because of their potential high impact on deep water shark populations. The following accounts focus on assessing the species of elasmobranchs caught and their catch rates in each of these fisheries. Incidental catches are estimated where no estimates already exist and these are then compared with reported landings for each fishery or country in order to assess the quantities of elasmobranchs wasted each year and not included in the official statistics of world fisheries.

2.2.3.1 Drift gillnet fisheries.

For the last few decades, several countries, chiefly Japan, Korea and Taiwan, developed large-scale fisheries using drift gillnets in the high seas of many oceans. Typically, vessels deployed several kilometres of gillnet which trapped very efficiently the relatively dispersed resources they were aiming for. Unfortunately, they also captured many other non-target species which were commonly discarded, sometimes in very large quantities. The concern of environmentalists over the impact of drift gillnets on oceanic fauna has been focused mainly in the more appealing marine mammals. However, it is now known that sharks were among the most frequently caught non-target organisms in some of these fisheries. Despite this, little attention has been paid to the effect of drift gillnets on shark populations. Although all large-scale driftnet fisheries have stopped on the high seas of the world since December 1992, an attempt is made here to assess the magnitude of their kills of sharks and rays. Though most of this kind of mortality has ceased, its effects may still be felt over subsequent generations of elasmobranchs. The details given below should both provide important reference information and stand as testimony to a recent environmental issue.

In this section, I analise the most important large-scale driftnet fisheries. The description of these fisheries is based primarily on the recent review by Northridge (1991) and the bulletins of the International North Pacific Fisheries Commission (INPFC) ( Myers et al. 1993, Ito et al. 1993). Readers are referred to those reports for more detailed information on these fisheries.

North Pacific Ocean.

Until recently, there were three main large-scale driftnet fisheries in the North Pacific, namely the salmon fishery, the flying squid fishery and the large-mesh fishery for tunas and billfishes. Together, these three fisheries made the North Pacific the most heavily exploited area of the world with driftnets. This probably reflected the geographic location of the three main countries involved in driftnetting on a large-scale.

a) Salmon fishery.

The Japanese fleet was the largest in this fishery. Canadian and USA fishermen still hold considerable numbers of boats, but are restricted to small driftnets (< 500 m per vessel) and fishing exclusively in the coastal waters of their EEZ

There were two Japanese fisheries for salmon. The "mothership" fishery that operated in the international waters of the North Pacific, south of the Aleutians and on the Bering Sea, and the land based fishery that occurred in the high-seas east of Japan (Fig. 2.31). In general, during the past two decades the Japanese salmon fishery showed a consistent

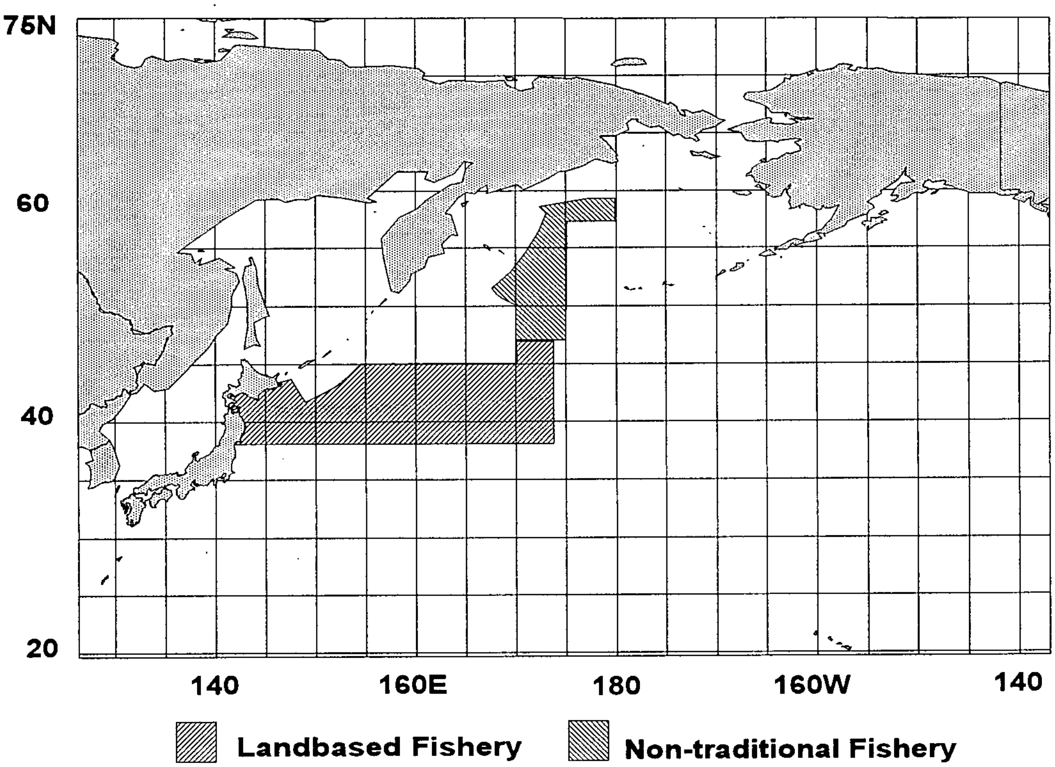



Figure 2.31 Generalized area of operation of the Japanese landbased and non-traditional (ex-mothership) fisheries in 1990. (Based on INPFC 1993).

decline in effort that involved contractions in number of vessels, fishing area and fishing season.

The mothership fishery consisted of processing ships that supported some 40 smaller "catcher" vessels. The fishing grounds were divided in subareas with different opening and closing seasons, although the total time span of the fishery only ran from May 31 to July 31. The fishery contracted its operations basically due to pressures from the USA, Canada and the former U.S.S.R. During the 1990 and 1991 seasons, the operations were converted to a landbased fishery ("non-traditional" landbased fishery) by eliminating the mothership boats. Catches peaked in 1956, when approximately 9.3 million tans were set, while only 238,700 tans were set in 1991, the last year of the fishery (F.A.J. 1991). Tans are independent net panels which constitute the working unit of driftnets and are typically 45-50 m long (already rigged) in the salmon fishery. The driftnet was 8-10 m deep and was constructed of nylon monofilament with mesh sizes in the range of 121-130 mm, each vessel deploying a maximum of 15 km of net in a dusk-to-dawn operation.

In the land based fishery, two types of vessels were known: coastal boats of <30 GT and medium size vessels of 30-127 GT. Effort in this fishery also declined significantly in the later days. The numbers of vessels in the fishery declined and the fishing area was reduced. Coastal vessels peaked in the mid-70's at 1,400 units but during 1978-1988 there were only 678 (Northridge 1991). Vessels over 30 GT were at their highest numbers in 1972-1974 with 374 boats, but were reduced to only 83 in 1991 (Myers et al. 1993). The total number of sets per season peaked at approximately 19,700 in 1966 but declined to about 4,100 (781,176 tans) in 1989 (F.A.J. 1990), with only 374,990 tans set during 1991 (F.A.J. 1991). The fishing season spanned from late May to the end of June during the last years of the fishery. Gillnets of the landbased fishery were similar to those of the mothership fishery but with smaller mesh sizes of 110-117 mm. Coastal vessels of <10 GT set less than 10 km of net per night while offshore vessels deployed up to 15 km of net.

Detailed reports on the bycatches of non-target species in these fisheries (see Northridge 1991, for a summary) are strongly biased towards studies dealing with marine mammals and birds: sharks are mentioned only as a side issue. However, the Fisheries Agency of Japan (F.A.J. 1987, 1988, 1989) reports bycatches of several non-target species in their driftnet research cruises for salmon. Their results for the years 1986-1988 are presented in table 2.9, together with the estimated total bycatch of sharks taken in the 1989 season when a total of 1'097,630 tans were set. Blue sharks are the most frequently reported shark species. The total bycatch in the fishery for 1989 is here estimated as 11,492 sharks of eight species or approximately 108 t.

These results should be taken with caution. First, the areas surveyed in the research cruises were apparently different from those of the commercial fishery, and there were some very small mesh sizes among some of the research driftnets. This probably has an effect on the catch rates of most species, both through changes in catchability of the gear and availability of each species (e.g. blue sharks are not expected to be caught in the Bering Sea in high numbers due to their more temperate distribution). Direct extrapolations of the research data to the total fishery might thus not be representative of the real situation. Secondly, most of the catch rates of sharks reported in table 2.9 seem too low compared with other studies.

Although there are no other direct reports for the salmon fishery, results from Canadian research cruises (LeBrasseur et al. 1987) can be used to derive alternative catch rates for sharks. These research cruises were designed to assess the salmon bycatches of the squid fishery but employed nets virtually identical to those of the commercial salmon fishery. Accordingly, their results could better reflect the catch rates of sharks in the commercial salmon fishery. The estimates obtained for blue and salmon sharks are one order of magnitude higher than those calculated from F.A.J. data, with values of 5,275 and 194 sharks/1000 km of net respectively (Table 2.10).

In general terms, the total catch of sharks in the Japanese salmon fisheries is believed to have been relatively small when compared with other driftnet fisheries in the north Pacific (see below). Even considering the alternative catch rates of 5,502 sharks per 1000 km of driftnet based on Canadian research data, some 300,000 individuals or approximately 1,237 t of sharks are estimated to have been caught during the 1989 season in this fishery. This relatively small catch is mainly a function of the size of the fishery, which as previously mentioned was contracting year by year. As a reference point, according to Shimada & Nakano (1992), some 34,000 large and adult salmon sharks were landed from the salmon driftnet fishery in Japan in 1960. Furthermore, reports for the early 80's (Paust 1987)

Table 2.9 Estimation of shark bycatches in the Japanese salmon fisheries, based on information from research cruises.

| Species | 1986 (24,549 tans) | 1987 (17,056 tans) | 1988 (17,805 tans) | Catch Rate a) (sharks/1000km) | Estimated Numbers in Catch 1989 b) Landbased | Mothership | Total | Likely weight (kg) c) per shark | in the catch |
|---|---|---|---|---|---|---|---|---|---|
| Unid. Lamnidae | 0 | 1 | 2 | 1.01 | 39 | 16 | 55 | 50 | 2,771 |
| *Lamna ditropis* | 25 | 26 | 23 | 24.91 | 973 | 394 | 1,367 | 50 | 68,359 |
| *Isurus oxyrinchus* | 13 | 1 | 2 | 5.39 | 210 | 85 | 296 | 50 | 14,780 |
| *Prionace glauca* | 142 | 188 | 79 | 137.69 | 5,378 | 2,179 | 7,556 | 2.42 d) | 18,287 |
| *Squalus acanthias* | 73 | 33 | 8 | 38.38 | 1,499 | 607 | 2,106 | 2 | 4,212 |
| *Isistius brasiliensis* | 1 | 1 | 0 | 0.67 | 26 | 11 | 37 | 0.75 | 28 |
| *Mustelus manazo* | 1 | 0 | 2 | 1.01 | 39 | 16 | 55 | 2 | 111 |
| *Triakis scyllium* | 0 | 0 | 1 | 0.34 | 13 | 5 | 18 | 2 | 37 |
| Totals | 255 | 250 | 117 | 209.39 | 8,179 | 3,313 | 11,492 | 159.17 | 108,586 |

a) assuming 50m tans in research cruises
b) based in effort reported by FAJ (1990)
c) Considering sizes expected for 110-130 mm mesh
d) Calculated from LeBrasseur et al. (1987) length frequency data, Pratt (1979) TL-FL relationship, and Strasburg (1958) L-W relationship.

Table 2.10 Alternative estimates of shark bycatches in Japanese salmon fisheries, based on Canadian research cruise (LeBrasseur et al. 1987).

| Species | Sharks caught (618 tans) | Catch rate per/1000km of net | Estimated numbers in 1989 catch | Likely weight (kg) per shark | in 1989 fishery |
|---|---|---|---|---|---|
| *Prionace glauca* | 163 | 5,275 | 289,504 | 2.42 a) | 700,601 |
| *Lamna ditropis* | 6 | 194 | 10,657 | 50 | 532,830 |
| *Squalus acanthias* | 1 | 32 | 1,776 | 2 | 3,552 |
| Total | 170 | 5,502 | 301,937 | 54.42 | 1,236,983 |

a) Calculated from LeBrasseur et al. (1987) length frequency data, Pratt (1979) TL-FL relationship, and Strasburg (1958) L-W relationship.

indicate 25,000 salmon sharks (*Lamna ditropis)* were taken each year by the Japanese salmon fishermen in the central Aleutian region. Considering the available effort statistics and the catch rates obtained from LeBrasseur et al. (1987), a total of less than 1,600 salmon sharks should have been taken in the area south of the Aleutians in 1989. This suggests that a reduction of about 95% in salmon shark fishing mortality accompanied the decline of the fishery.

Although there is not enough information to assess the level of catches and discards of sharks that took place in this fishery, it is possible that some of the salmon sharks would have been kept and utilised. This is suggested by reports of specific fisheries for this species taking place in NE Japanese waters off the Oyashio front (Paust 1987, Anon. 1988), which indicate that the salmon shark is appreciated by Japanese fishermen. On the other hand, the motivation to keep salmon sharks probably had to be weighted against the availability of space and the danger of spoilage of the valuable catches of salmon in the vessel's storage area.

In July 1991, all Japanese salmon driftnet fisheries in the high-seas ceased activities. Most of the fleet was disbanded although a minor part was reallocated to the Russian EEZ waters via a joint venture between Japan and Russia. There is still no official information available about this new salmon driftnet fishery but judging from the calculations made above the bycatches of elasmobranchs should be of relatively low importance.

b) Flying squid fishery.

Since the late 70's, a major driftnet fishery for flying squid (*Ommastrephes bartrami)*, was started consecutively by Japan, Korea and Taiwan (in order of importance) in the Central North Pacific. In 1990 almost 740 vessels from the three nations were operating.

Yatsu et al. (1993) summarise most of the information available for Japan which was the first country to begin fishing for flying squid in the central North Pacific in 1978. Japan limited the number of vessels and the area open to this fishery (fig. 2.32), with a north boundary which moved through the year to avoid catches of salmons which were prohibited to the entire flying squid fishery. There were two categories of Japanese vessels: 60-100

GT and 100-500 GT. The fishing season ran from June 1st to December 31st, although two types of licenses for 7 and 4 months were issued within the season. The driftnets were constructed of nylon monofilament (yarn 0.5 mm) and mesh sizes in the range 100-135 mm, with 115-120 mm the most commonly employed. Rigged tans were 9-10 m deep and 33-42 m in length. Each vessel set between 15 and 50 km of net, although some reports indicate that most common sets were close to 50 km.

Following Japan's initiative, Korea joined the fishery in 1979 (see Gong et al. 1993 for a full account). Korean squid driftnet vessels were mostly of c. 350 GT, but some boats exceeded 400 GT. The Korean fleet fished from April to early August in an area partially overlapping with the Japanese fishing grounds, and from early August to mid-December for smaller squid to the east of Japan (fig. 2.32). Korean driftnets had 50 m tans with mesh sizes of 76-155 mm. In the main fishing area they were commonly 105-115 mm, while those utilised in the grounds east of Japan were 86-96 mm. According to Gong et al. (1993), Korean vessels deployed about 28 km of driftnet in the early 1980's but increased to a high of 45 km in 1990.

Information on the Taiwanese squid fishery is scarce and most of this account is based on the brief communication of Yeh and Tung (1993). Taiwan joined the fishery in 1980. Vessels' size ranged from 100-700 GRT but most were 200-300 GRT. Driftnetters larger than 400 GRT were introduced mainly in 1984 while those larger than 600 GRT entered during the 1986-1987 season. Taiwanese driftnets for squid were apparently constructed of monofilament nylon. Their mesh sizes ranged from 76-120 mm, with each tan measuring between 15 and 40 m in length. Typical total lengths of driftnet deployed per boat were 31-41 km (Fitzgerald et al. 1993). Taiwanese vessels were allowed to fish year round (Pella et al. 1993) but the fishing season was apparently realised only from June to November (Yeh and Tung 1993) in an area very similar to the Korean grounds but extending westward to the Japanese EEZ (fig 2.32).

Effort statistics for these fisheries have been made available only very recently. According to data provided by Yatsu et al. (1993), Gong et al. (1993) and Yeh and Tung (1993), the total number of vessels from the three countries in the squid driftnet fishery for the period 1988-1990 were 792, 784 and 737 respectively. Data on the total number of tans deployed

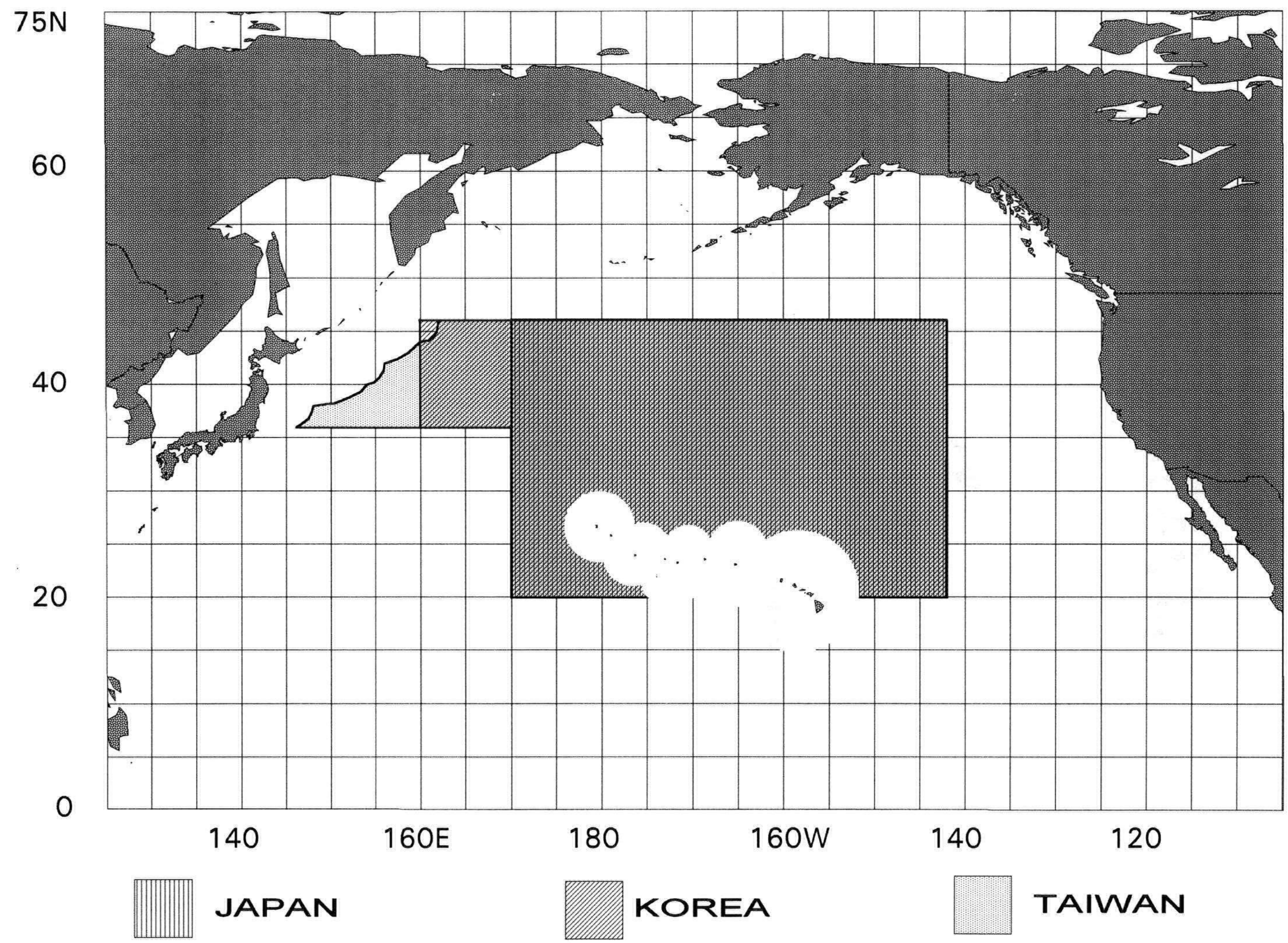



Figure 2.32 Legal boundaries of the Japanese, Korean and Taiwanese flying squid driftnet fisheries. (Redrawn from Pella et al. 1993).

by Japan and Korea are also available. Unfortunately, Taiwanese statistics do not separate effort between the squid fishery and the large-mesh driftnet fishery (examined in next section) as their boats carried both types of gear and deployed either one depending on the expected catch. Furthermore, Taiwanese effort statistics are given only in total vessel/days fished (table 2.11).

The total number of standardised tans set by the Taiwanese fleet in the squid fishery can be estimated with the aid of comparative data on typical total length of sets for vessels from each county. Fitzgerald et al. (1993) provide estimates of a total of 51-61 km of driftnet per Japanese vessel and a total of 31-41 km per Taiwanese vessel. Data from Yatsu et al. (1993) indicates that Japanese vessels deployed an average of 997.43 tans (50 m each) per fishing day during 1989 and 1990. The effort of Taiwanese vessels is here assumed to be allocated equally to the flying neon squid and the large-mesh fisheries. Assuming that the number of tans per vessel is equal in the Japanese and Taiwanese fleets, total efforts of 4'471,678, 5'616,888 and 3'595,855 standardised (50 m) tans of net are estimated here for the Taiwanese fleet in the squid fishery for the years 1988-1990 respectively. Accordingly, total effort for the three countries in this fishery can be approximated at 64'782,236 tans (3'239,112 km) for 1989 and 50'922,388 tans (2'546,119 km) for 1990.

There are several sources of information on catches of non-target species in these fisheries, chiefly from research cruises and more recently from observer programmes. Results from some research surveys enable an assessment of catch rates in numbers of sharks for blue, salmon and four other species of sharks, size structure and catch rate in kg/m for blue sharks, percentage distribution by mesh size for blue and unspecified shark species, and differences in blue shark catches between surface and subsurface squid driftnets (FAJ 1983, Murata and Shingu 1985, Murata 1986, 1987, Rowlett 1988, Murata et al. 1989, Yatsu 1989, Ito et al. 1990). However, results from these surveys suffer the same problems of the salmon fishery research surveys. Japanese and Korean research cruises utilise a variety of mesh sizes which extended above and below the size range of those utilised in the commercial fishery. The application of their results is therefore very limited for the purpose of assessing total catches of non-target species.

Far more useful information comes from the observer programmes on board commercial vessels. Data from Japanese observers for 1988 (FAJ 1989) indicate catch rates of 536 blue sharks per 1000 km of net. However, collective data from Japanese, Canadian and U.S. observers for 1989 (Gjernes et al. 1990) report 814 blue sharks per 1000 km of net.

Data for the 1990 observer programme (INPFC 1991) are more detailed and indicate that 12 elasmobranch species were taken as bycatch in the fishery. The catch rates for blue sharks was 718/1000 km of driftnet, followed by salmon sharks (55/1000 km of driftnet). Other large sharks species captured, perhaps by entangling, were common thresher (*Alopias vulpinus),* shortfin mako (*Isurus oxyrinchus),* white (*Carcharodon carcharias)* and basking shark (*Cetorhinus maximus)* (Table 2.12). Observer data from the Korean fleet for 1990 estimated a catch rate of 32.08 sharks and rays per 1000 poks (Korean tans) which is equivalent to 641.6/1000 km of net. This figure is slightly low, compared to the 785/1000 km of net estimated for the Japanese fishery.

Data on fishes in most of the observer programmes for the North Pacific driftnet fisheries are likely to be slightly underestimated. Only decked animals are taken into account, thus unknown numbers of 'dropoff' fishes were not included in the records. Despite this, observer programmes provide the best available information.

There are some independent estimates for elasmobranch bycatches in the squid driftnets. Yatsu et al. (1993) estimate a total incidental catch of 723,933 blue sharks, 56,029 salmon sharks and 11,322 various sharks and rays for the Japanese fleet during 1990, amounting to 7,415 t. Yatsu et al.'s estimation method takes into consideration sources of variability for cruises and sets sampled. However, their estimates of blue shark bycatch for 1989 are almost double those for 1990 highlighting the variability of estimations and the changes in fishing effort during the period. Wetherall and Seki (1992) using a stratified estimation arrive at a total bycatch of 1.2-1.4 million blue sharks for the Japanese fishery during 1989, while Northridge (1991) estimated the total catch of blue sharks for the entire flying squid fishery during 1989 at 2.44 million individuals (considering the same effort level of 1988 and catch rates derived from Gjernes et al. [1990]).

Using the latest effort statistics available for 1990 and the results from the Japan-USA observer programme (INPFC 1991), numbers of elasmobranchs by species and the likely

Table 2.11 Effort statistics for the flying squid driftnet fishery in the North Pacific for the period 1988-1990 (from Yatsu et al. 1993, Gong et al. 1993 and Yeh & Tung 1993).

| Year | | Japan | Korea | Taiwan | Total |
|---|---|---|---|---|---|
| | # boats | 463 | 150 | 179 | 792 |
| 1988 | days fished | - | - | 14,010 | |
| | total tans | 36,055,567 | 24,594,370 | - | |
| | # boats | 460 | 157 | 167 | 784 |
| 1989 | days fished | 33,646 | - | 17,598 | |
| | total tans | 34,385,032 | 24,780,316 | - | |
| | # boats | 457 | 142 | 138 | 737 |
| 1990 | days fished | 23,656 | - | 11,266 | |
| | total tans | 22,769,857 | 24,556,676 | - | |

Table 2.12 Estimation of bycatches of elasmobranchs in 1990 Squid driftnet fishery based on reports of observer programme on board commercial vessel (INPFC 1991).

| Species | Numbers observe (2,281,896 Tans) | Catch rate per/1000km of net | Numbers in Total catch | Likely mean Weight (kg)* | Weight in Total catch (kg) |
|---|---|---|---|---|---|
| Unidentified shark | 1,191 | 10 | 26,578 | 15? | 398,672 |
| *Prionace glauca* | 81,956 | 718 | 1,828,915 | 7 (1) | 12,802,407 |
| *Lamna ditropis* | 6,263 | 55 | 139,764 | 38.7 (1) | 5,408,866 |
| *Isurus oxyrinchus* | 71 | 0.622 | 1,584 | 40 | 63,377 |
| *Alopias vulpinus* | 48 | 0.421 | 1,071 | 40 | 42,846 |
| *Squalus acanthias* | 8 | 0.070 | 179 | 2 | 357 |
| *Carcharodon carcharias* | 7 | 0.061 | 156 | 50 | 7,811 |
| *Isistius brasiliensis* | 5 | 0.044 | 112 | 0.75 | 84 |
| *Euprotomicrus bispinatus* | 1 | 0.009 | 22 | 0.20 | 4 |
| *Cetorhinus maximus* | 1 | 0.009 | 22 | 500 | 11,158 |
| *Dasyatis violacea* | 8 | 0.070 | 179 | 10? | 1,785 |
| *Dasyatis brevis* | 1 | 0.009 | 22 | 10? | 223 |
| Unidentified ray | 8 | 0.070 | 179 | 10? | 1,785 |
| Totals | 89,568 | 785 | 1,998,783 | - | 18,739,376 |

* considering sizes expected for 100-135 mm. mesh
(1) from Yatsu et al. 1993

weight of their catches in the 1990 season are estimated here and summarised in table 2.12. These estimates indicate that a total of about 2 million sharks equivalent to 18,739 t were taken in the whole fishery. About 12,802 t of these or 1.8 million individuals were mostly very young blue sharks, which according to Nakano and Watanabe (1992) correspond mostly to sharks 1-2 years old. Unless otherwise stated, the estimates of the individual average weight for each species are based on approximations I made considering the relatively small mesh size of the nets. In any case these weights might be negatively biased. Accordingly, the present results can be taken as a minimum estimate of the real catch of elasmobranchs in the fishery. Of the approximately 18,700 t of sharks caught, some 8,400 t would have been taken by Japan, while Korea and Taiwan would have caught 9,000 t and 1,300 t respectively.

A great proportion of the elasmobranch bycatches were apparently dumped to the sea. Assessment of shark catches for the Japanese fleet in 1989 using the same procedure as above produced estimates of 1.8 million sharks with a total weight of almost 12,654 t. The reported total take of sharks by the squid fleet of Japan during 1989 is 237,734 individuals only (FAJ 1990). Assuming this figure is equal to the landed catch of sharks, about 1.56 million sharks weighing some 10,900 t were wasted in the operation. Some of the almost 95,000 salmon sharks estimated to be caught in the fishery, were probably utilised as this species is more appreciated in Japan. An appraisal of the amount of elasmobranchs actually discarded by the fleets of Taiwan and Korea is not possible due to the lack of information on their shark landings from the squid fishery. The total estimated catch of elasmobranch for the Korean and Taiwanese fleets in 1989, worked out in the same manner, amounts to 9,120 t and 2,067 t respectively.

The present estimates of elasmobranch bycatches are certainly rough due to the limitations of the available information. They do however highlight the problems found when trying to assess the magnitude of the elasmobranch bycatch and the proportions dumped to the sea. Estimates of total weight of the bycatch are very sensitive to the average weights for each species used in the calculations. This is particularly true in the case of blue shark which accounts for most of the bycatches in numbers. Ideally, the average weight of sharks used for a particular area should reflect the size composition for that area because of disctinct size segregation for many species, however, this detail in data is not available at present. Yatsu et al. (1993) use an average weight of 7 kg for blue sharks and this was followed here for the calculations of table 2.12, however, alternative calculations based on the length frequency reports for blue sharks of LeBrasseur et al. (1987) and morphometric equations for the species provided by Strasburg (1958) and Pratt (1979), produce an average weight of 2.4 kg/shark. The estimate of 2.4 kg/shark is consistent with the findings of Bernard (1986), Mckinnell et al. (1989) and Murata et al. (1989), for nets with the same characteristics as those from the commercial squid fishery.

The figures derived here appear slightly overestimated when compared with alternative figures. However, considering that observer programmes do not take into account any sort of dropouts from the nets, the present estimates could be closer to the real mortality inflicted by the driftnets and serve as an indication of the order of magnitude of the problem. If this is true, previous appraisals of blue shark catches in the whole fishery (Anon. 1988) seem to be highly overestimated.

Efforts to minimise the take of non-target species in the squid driftnet fishery were met with unsuccessful results. Data summarised by Gong et al. (1993) for Korean research experiments indicate that shark bycatches can drop by as much as 41% when subsurface driftnets are utilised instead of normal (surface) driftnets. Unfortunately, catch of the target species (neon flying squid) dropped by 73%, probably making operations with the subsurface driftnets unprofitable. As a result of international agreements, the squid driftnet fishery of the North Pacific ceased to exist at the end of 1992.

c) Large-mesh driftnet fishery.

A large-mesh driftnet fishery for skipjack, marlin, albacore and other tunas was initiated in the high seas of the North Pacific in the early 70's by Japan. However, this fishery came to an end on December 31 1992 together with all other high-seas driftnet fisheries in the area.

This fishery had its origins in the coastal Japanese bluefin tuna fishery of the 1840's. By the late 1980's it covered an area extending from 140°E to 145°W (fig. 2.33). The fishing grounds were divided into two subareas. A southern subarea open to fishing year round and a northern subarea with portions closed to fishing during specific months in order to avoid catches of salmonids. Recent reports indicate this fishery operated with vessels in the 100-

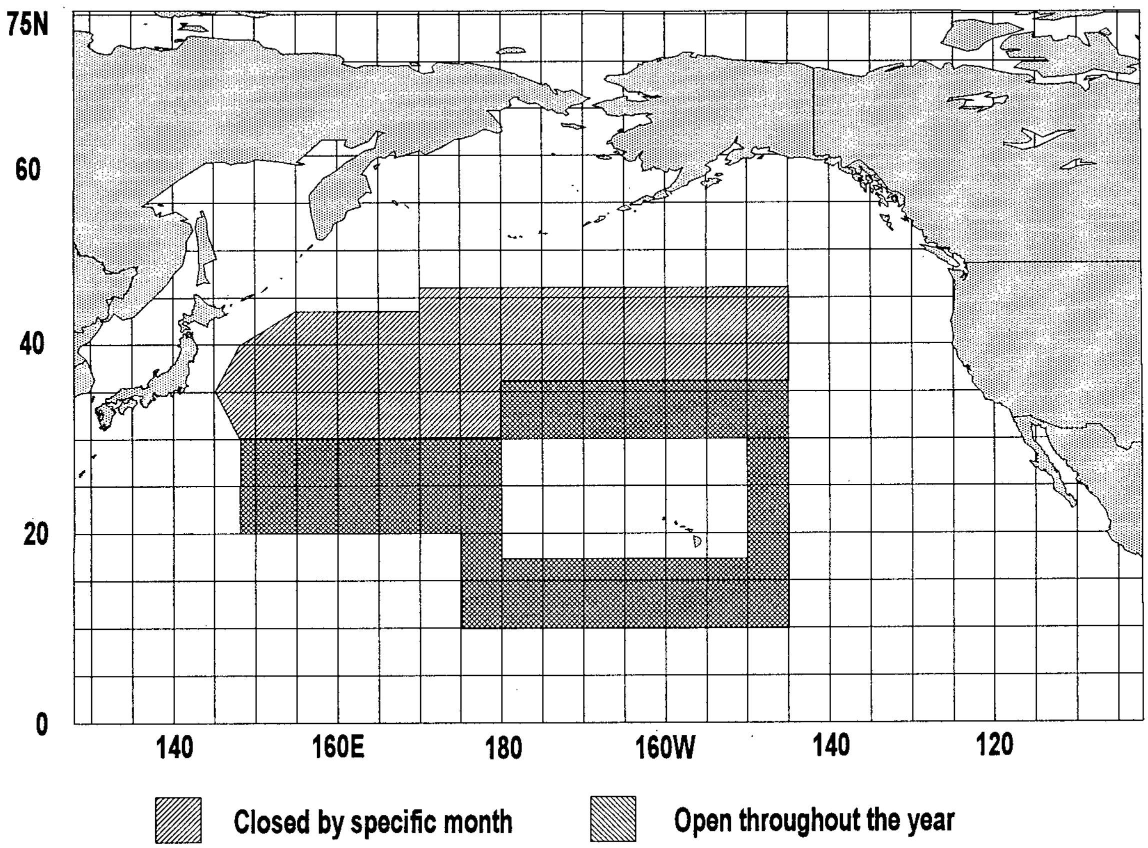



Figure 2.33 Area of operation of Japanese large-mesh drift net fishery. (Redrawn from Nakano et al. 1993).

500 GT range. Nets with small meshes were of nylon monofilament twines of 1.2 mm diameter while larger meshed nets used multifilament and multistrand twines. Although mesh size was restricted to be >150 mm, meshes as small as 113 mm were recorded; most driftnets had meshes of 180 mm (INPFC 1992). Tans were commonly 33-36 m in length. Japanese boats were restricted to deploy a maximum of 12 km of net at a time. According to recent figures, 459 vessels from Japan took part in the large-mesh driftnet fishery in 1988 with a total catch of approximately 40,000 t.

Taiwanese vessels also participated in this fishery, but information is scarce. Apparently, up to 123 vessels from Taiwan took part in this fishery during 1989. The Taiwanese fishing season spanned chiefly from June to December.

According to last available figures (Fitzgerald et al. 1993), Japanese vessels deployed a total of 4,682,630 standard (50 m) tans in this fishery during 1990. Taiwanese effort is here assumed to be the same of that estimated for the squid driftnet fishery (see above) due to the combined nature of these fisheries (Yeh and Tung 1993). The combined effort of both nations during 1990 was probably equivalent to a total of 413,924 km of large mesh driftnet.

Information on the kinds and numbers of elasmobranchs caught in this fishery has recently become available through the reports of the international observers programme (INPFC 1992). Catch rates and estimates of the total catches of sharks and batoids based on effort levels reported for 1990 indicate that about 150,000 sharks or 1,722 t, were taken as bycatch (table 2.13). The average weights of some species are taken from research cruises that utilised driftnets with mesh sizes 150-180 mm (FAJ 1983), while others are my best possible 'guesstimates' for the corresponding mesh sizes.

The estimated elasmobranch bycatch rate of 366 fish/1000 km for the large-mesh driftnets is about half that of the squid fishery. This difference is related to the different selectivity patterns of the nets involved, with larger meshes allowing for greater escapement of small non-target species. Blue shark catch rates are less than half of those observed in squid driftnets, while catch rate for salmon sharks are even lower. On the other hand, the average size of each specimen is expected to be larger in the large-mesh fishery. From the overall estimated catch of elasmobranchs in this fishery in 1990, approximately 974 t would have

Table 2.13 Estimated bycatches of elasmobranchs in the 1990 North Pacific large-mesh driftnet based on reports of the observer programme 1990 (INPFC 1992).

| Species | Numbers observed (513,367 Tans) | Catch rate per 1000km of net | Numbers in Total catch | Likely mean Weight (kg) | Weight in Total catch (kg) |
|---|---|---|---|---|---|
| Unidentified shark | 57 | 12.00 | 4,967 | 25 ? | 124,177 |
| *Prionace glauca* | 7,692 | 300 | 124,040 | 9.2 (1) | 1,141,168 |
| *Lamna ditropis* | 136 | 5.30 | 2,193 | 32.5 (1) | 71,276 |
| *Isurus oxyrinchus* | 592 | 23 | 9,547 | 30 ? | 286,395 |
| *Alopias vulpinus* | 6 | 0.23 | 97 | 167 (1) | 16,158 |
| *Squalus acanthias* | 1 | 0.04 | 16 | 2.5 | 40 |
| *Carcharodon carcharias* | 35 | 1.36 | 564 | 47.7 (1) | 26,922 |
| *Isistius brasiliensis* | 305 | 12 | 4,918 | 0.85 | 4,181 |
| *Euprotomicrus bispinatus* | 156 | 6.08 | 2,516 | 0.25 | 629 |
| *Cetorhinus maximus* | 2 | 0.08 | 32 | 550 | 17,738 |
| *Triakidae* | 3 | 0.12 | 48 | 3 | 145 |
| *Sphyrnidae* | 2 | 0.08 | 32 | 127 (1) | 4,096 |
| *Dasyatis violacea* | 73 | 2.84 | 1,177 | 12 ? | 14,126 |
| *Dasyatis brevis* | 8 | 0.31 | 129 | 12 ? | 1,548 |
| Unidentified ray | 69 | 2.69 | 1,113 | 12 ? | 13,352 |
| Totals | 9,137 | 366 | 151,390 | - | 1,721,953 |

(1) Derived from F.A.J. (1983).

been taken by Japan and 748 t by Taiwan.

There are no estimates in the literature to compare with the present results. Furthermore, there are no statistics for the amounts of elasmobranchs landed from the large-mesh fishery in Japan or Taiwan that allow an estimate of discards. Judging from the trends in other high-seas fisheries, it is very likely however, that most bycatches of sharks were not utilised but instead discarded at sea.

South Pacific Ocean.

Large-scale driftnet fishing stopped since 1991 in the South Pacific. Formerly, Japan and Taiwan fished chiefly for albacore with large-mesh driftnets (see Northridge (1991) for details). Due to pressure from coastal states in the area, an agreement was made to terminate these fisheries in the high-seas of the South Pacific by 1991. It is not clear if the agreement pertains only to the waters of the South Pacific Commission (SPC) (fig. 2.34) or if it also includes the Eastern South Pacific. Japan stopped all large-scale driftnet fishing in the area in 1990 (Nagao et al. 1993), but information on Taiwanese vessels is not at hand. However, available information suggests that elasmobranch bycatch in large-scale driftnets in the South Pacific should at present be nil or negligible even if vessels from Taiwan continue to fish there. A brief account of the few reports of elasmobranch bycatches in South Pacific driftnet fisheries is presented below.

Some reports of elasmobranch catch rates in the South Pacific are given in table 2.14 based on data from Sharples et al. (1990) and Watanabe (1990). Their sources of information are two research cruises conducted in the Tasman Sea and the Sub-Tropical Convergence Zone (STCZ) to the east of New Zealand between 30° and 45°S. Catch rates estimated from these data are 181 and 158 sharks/1000 km of net for the STCZ and the Tasman Sea respectively, or 5,035 kg/1000 km of net for the Tasman Sea. While total elasmobranch catch rates could seem relatively similar among both areas, strong differences in catch rates for individual species are evident when looking at detailed information (e.g. blue sharks are more frequently caught in the STCZ than in the Tasman Sea while the opposite is true for mako sharks). Additionally, the catch rate for the Tasman Sea is high compared to data from Coffey and Grace (1990). These differences illustrate the restrictions faced for

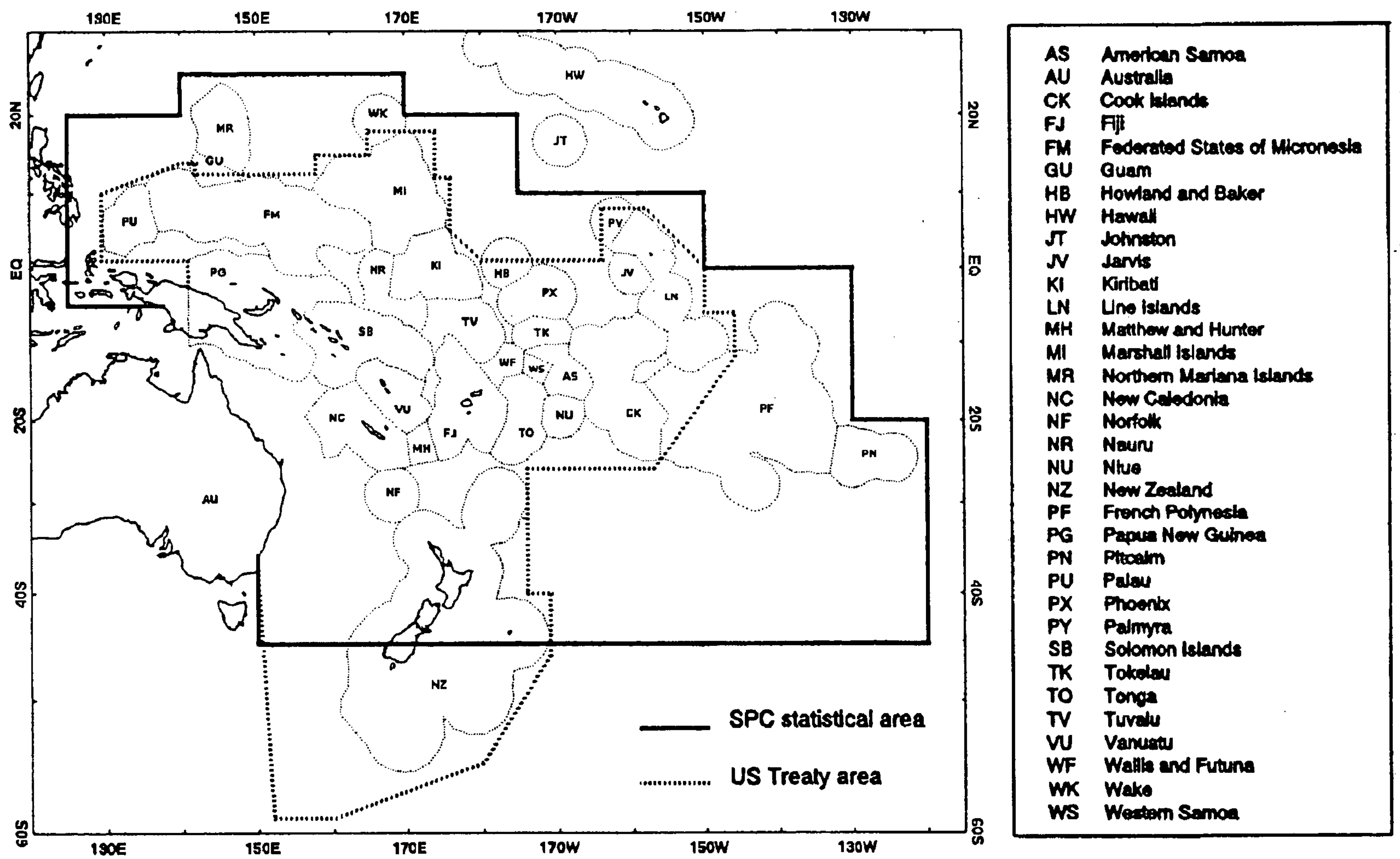



Figure 2.34 South Pacific Commission statistical area. (Taken from Lawson 1991).

Table 2.14 Reported bycatches of elasmobranchs in South Pacific driftnet fisheries.

| | STCZ (464 km of net)* | | TASMAN SEA (766 km of net)** | | | |
|---|---|---|---|---|---|---|
| Species | Numbers Caught | Catch rate (#/1000 km) | Numbers Caught | Catch rate (#/1000 km) | Mean Weight | Catch rate (kg/1000 km) |
| *Cetorhinus maximus* | - | - | 1 | 1.31 | - | - |
| *Prionace glauca* | 70 | 150.86 | 22 | 28.72 | 70 | 2,001 |
| *Lamna nasus* | - | - | 3 | 3.92 | - | - |
| *Isurus oxyrinchus* | 10 | 21.55 | 66 | 86.16 | 31 | 2,663 |
| *Isistius brasiliensis* | - | - | 10 | 13.05 | - | - |
| *Sphyrna zygaena* | - | - | 3 | 3.92 | 95 | 371 |
| *Dasyatis violacea* | 4 | 8.62 | 16 | 20.89 | - | - |
| Total | 84 | 181.03 | 121 | 157.96 | 195 | 5,035 |

* Data from Watanabe (1990)

** Data from Sharples et al. (1990)

extrapolations from catch rates to total bycatches when the catch rates are based on information limited to a particular area/fishery/season.

Coffey and Grace (1990) observing commercial vessels estimated catch rates of 48 sharks/1000 km of net and a total bycatch of 3,500 sharks in the Tasman Sea area for the 1990 season. Murray (1990) compiles data from several sources and provides information on percentage by weight of sharks in total catches of Japanese research campaigns using three types of driftnets along with total effort for each type of net. With this information, shark catch rates are here calculated assuming 50 m tans: for albacore nets, 16,362 kg/1000 km; for slender tuna nets, 14,618 kg/1000 km; and for pomfret nets, 21,781 kg/1000 km. Given the lack of estimates of the total amount of nets deployed in these fisheries shark bycatch is estimated using the percentages of sharks to the total albacore catch for the albacore nets mention above, and the reported albacore catches for driftnet fleets in the South Pacific for 1989 provided by Lawson (1991). The gross estimates of total shark bycatches are: Japan, 3,462 t, Korea, 48 t and Taiwan, 2,871 t. These figures add up to 6,381 t and correspond to the reported peak in albacore driftnet catches. Hence, total bycatch levels should have been smaller in the earlier and later years of the fishery. These estimated catches pertain only to the waters of the South Pacific Commission (fig 2.34) and are crude estimates limited by the available information. Furthermore, it is unknown if the data cited by Murray (1990) used for estimating bycatch percentages contain information from the whole South Pacific region or only from part of it. Geographical variations in abundance are likely to affect the bycatch levels considerably. Without any information about driftnetting activities in the rest of the South Pacific Ocean, I can only speculate that given the proportion of the South Pacific covered by the SPC area (about 2/3), the bycatch of elasmobranchs in the whole Southern Pacific could have been 50% more than that calculated here for the SPC zone, or a total of 9,572 t. Although uncertain, this elasmobranch bycatch level is about half of that for driftnets in the North Pacific Ocean.

Indian Ocean.

Several countries have extensive driftnet fisheries in the Indian Ocean. However, most of the coastal states in the area e.g. India, Pakistan, Sri Lanka, only fish within inshore waters with small and medium-scale fisheries already treated in section 2.2. The elasmobranch catches of these coastal states are assumed to be landed and therefore already reported in FAO statistics.

Taiwan is the only country known to have large-scale driftnet vessels fishing in the international waters of the Indian Ocean, but there is very limited information available. This tuna fishery started with one boat in 1983 and grew to a total of 139 vessels in 1988. Fishing apparently takes place from November to March with driftnets of 200-220 mm mesh size, 20-24 m depth with 20-25 or 37-47 km of net deployed per vessel. Fishing is mainly carried out in waters of the North West and South Central Indian Ocean. Hsu and Liu (1991) report sharks to be 23.76 and 29.57% of the total catches in numbers and weight respectively for the 1986-1987 fishing season, while during 1987-1988 this decreased to 0.52 and 2.07%. As no significant changes in fishing area were observed between both fishing seasons, this reduction in shark bycatches most likely reflects changes in discard rates. Multiplying the percentage composition of sharks to the reported total landings of 18,281 t in the 1986-1987 season (IPTP 1990), some 5,405 t of sharks are estimated to have been caught in the fishery. A total shark catch of 6,108 t is here estimated for the 1988-1989 season, assuming that the number of vessels increased by 13% from the 1986-1987 level.

Atlantic Ocean.

Until recently, the only known large-scale driftnet fisheries in the Atlantic were a French albacore fishery and an Italian swordfish fishery. However, Taiwanese driftnet vessels were thought to operate also in the Atlantic Ocean during the early 1990's. Many other fisheries with gillnets exist across the Atlantic and Mediterranean and in many cases they amount to large quantities of nets deployed per night. However, most of these fisheries are limited to coastal waters and fall out of the scope of this section. A summary of these smaller fisheries is given by Northridge (1991).

The French albacore fishery began in the Bay of Biscay in 1986 and 37 vessels were operating by 1989. These boats trolled during the day and used gillnets during the night. Fishing took place during June to September and extended from the Azores north and eastward following the movements of albacore. Nets were 20-36 m deep and 80-120 mm mesh size; the most successful were 90 mm in mesh. While French reports indicate driftnets' lengths of 2-6 km per vessel, Greenpeace claims that they are up to 20 km long. For shark bycatches, the only available information indicates they were in the order of 6-10%. Woodley and Earle (1991) observing several French boats report sharks (mostly *Prionace glauca)* as the most common bycatch, amounting to 6.2% of the albacore catch. Sharks caught were estimated to range between 40-250 cm but more commonly between ca. 125-200 cm. Woodley and Earle additionally estimate catch rates of 1,750 to 3,520 sharks/1000 km of net (including dropouts) together with a total catch of 22,015 to 44,282 sharks during the 1991 French albacore fishery. This is equivalent to some ca. 430-865 t of sharks assuming a mean total length of 175 cm for blue sharks. They report a discard of two sharks at sea but no further information is available on the disposition of the shark bycatches in this fishery. However, these shark catches could be already included in the reported "various elasmobranchs" of France which amount to almost 10,000 t/yr.

The use of driftnets in Italian fisheries for tuna and swordfish has a long history, but it was until the 1980's that the fishery expanded considerably as a consequence of governmental support. According to Northridge (1991), this was one of the largest driftnet fisheries in the world before it was banned. By 1989, 700 vessels were participating, 90% of them using nets of 12-13 km in total length with depths of 28-32 m and mesh sizes in the range 180-400 mm. A few vessels used less than 6 km of net, while a few others more than 20 km. The fishery pursued albacore and swordfish from Sicily and Calabria to the Ligurian Sea. While there is no information on catch rates of non-target species, several elasmobranchs have been reported to occur in this fishery. Species commonly caught include common thresher, blue and porbeagle sharks, as well as manta and common eagle rays. Another three sharks are reported as infrequent species and 10 more as occasional species (table 2.15). It is unknown whether most of the catches were kept or discarded, and it is impossible to estimate the amount of the total catch from available information. However, a large increase in landings of smooth-hounds took place concurrently with the expansion of the driftnet fishery and it is known that other sharks are commonly smuggled locally as smooth-hounds (De Metrio et al. 1984). Therefore, it is possible that a considerable part of the shark bycatch from this fishery was landed. Recent reports suggest that there are still some driftnetters in the Ligurian Sea using gear lengths above the permitted 2.5 km per vessel in the Area (ICCAT 1993a).

Table 2.15 Elasmobranchs caught in Mediterranean driftnets (adapted from Northridge 1991).

| | Common Name | Scientific Name |
|---|---|---|
| Species commonly caught | | |
| | Thresher shark | *Alopias vulpinus* |
| | Blue shark | *Prionace glauca* |
| | Porbeagle | *Lamna nasus* |
| | Manta ray | *Mobula mobular* |
| | Common eagle ray | *Myliobatis aquila* |
| Infrequent species | | |
| | Basking shark | *Cetorhinus maximus* |
| | Shortfin mako | *Isurus oxyrinchus* |
| | Smooth hammerhead | *Sphyrna zygaena* |
| Occasional species | | |
| | Bigeye thresher | *Alopias superciliosus* |
| | Spinner shark | *Carcharhinus brevipinna* |
| | Blacktip shark | *C. limbatus* |
| | Dusky shark | *C. obscurus* |
| | Sandbar shark | *C. plumbeus* |
| | Great white shark | *Carcharodon carcharias* |
| | Sharpnose sevengill shark | *Heptranchias perlo* |
| | Sand tiger shark | *Carcharias taurus* |
| | Smalltooth sand tiger | *Odontaspis ferox* |
| | Hammerhead shark | *Sphyrna spp.* |
| | Tope | *Galeorhinus galeus* |
| | Bull ray | *Pteromylaeus bovinus* |

Northridge (1991) reviews several reports of Taiwanese vessels fishing with large driftnets in different areas of the Atlantic Ocean. However, no further information on the issue is available.

Overview of driftnet fisheries.

High-seas driftnet fisheries were an important source of elasmobranch bycatches. The estimates presented above suggest that the total elasmobranch bycatch could have been between 3.28 and 4.31 million sharks and rays per year during 1989-1991, or in the order of 20,000-38,000 t/yr. Total discards of elasmobranchs at sea from driftnet fisheries could have been as high as 30,500 t/yr, but assuming all Taiwanese and French catches were kept, discards could have been lower at 20,803 t/yr. The estimates presented here are derived by adding estimates for each of the fisheries previously described and carries along accumulated uncertainty. In this sense this overall estimates should be treated with discretion and used only as a first approximation of the level of elasmobranchs removed by driftnets worldwide.

The North Pacific fisheries were without any doubt the most intensive and therefore the most important driftnet fisheries for their catches and waste of sharks and rays (table 2.16). In particular, the flying squid fishery with its high catch rates and massive effort killed more elasmobranchs than any other high-seas driftnet fishery ever known. Fortunately, for the purpose of appraising the world bycatches of elasmobranchs by high-seas driftnet fisheries, the North Pacific fisheries accounted for the largest proportion of the total and were also the best studied driftnet fisheries. This somehow reduces the level of uncertainty in the estimates of world catches of elasmobranchs with driftnets.

Blue sharks were undoubtedly the most common animal caught in driftnet fisheries because of their high abundance in pelagic habitats. Total numbers taken in 1989 are estimated to amount to 2.2-2.5 million sharks. They are caught worldwide very frequently in larger numbers than any other elasmobranch. Blue sharks may well be the most threatened elasmobranch by these fisheries, but much more information is needed to ascertain the magnitude of the impact of their removal.

Table 2.16 Summary of estimated bycatch of elasmobranchs in high seas driftnet fisheries.

| Fishery | Total catch in tonnes | | | Total catch in | Catch rates |
|---|---|---|---|---|---|
| | Lower level | | Upper level | number of individuals | (sharks/1000 km nets) |
| North Pacific Ocean | | | | | |
| salmon(89) | 108 | - | 1,237 | 11,492 - 300,000 | 210 - 5,502 |
| squid(90) | 7,415 | - | 18,739 | 2.0 - 2.44 Million | 536 - 814 |
| large mesh(90) | - | 1,722 | - | 151,390 | 366 |
| South Pacific Ocean(89) | 6,381 | - | 9,572 | 56,000 - 841,500* | 48 - 181 |
| Indian Ocean(89) | - | 6,108 | - | 537,000* | - |
| Atlantic Ocean(91) | 430 | - | 865 | 22,000 - 44,000 | 1,750 - 3,520 |
| Total | 22,164 | | 38,243 | 3,282,882 - 4,313,890 | |

* from extrapolation of average weight of large mesh fishery

The different levels of uncertainty surrounding the estimated catch rates for each fishery highlight the importance of cooperative observer programmes in high-seas fisheries worldwide: only those fisheries that had observer programmes, had enough information to allow derivation of reasonably good estimates of elasmobranch bycatch. Also, only in these cases was possible to have direct information of the species caught. Considering this, the best estimates are those for the North Pacific squid and large-mesh fisheries which were the only fisheries with observers on board. In contrast, larger uncertainties surround catch rate and total kill estimates from the rest of the fisheries analysed.

A consequence of the recent closure of all large-scale driftnet fisheries in the high-seas is that overnight, the mortality inflicted by these fisheries has ceased. This provides some relief to many populations of birds, mammals and other marine fauna. Unfortunately it only provides a small breathing space for elasmobranchs and particular sharks, which continue to be caught incidentally in very large numbers in other high-seas fisheries (see next section).

### 2.2.3.2 Longline fisheries.

The most important large-scale longline fisheries in the world are those for tunas and billfishes. These fisheries include the fleets of several countries and take place in all oceans of the world. Technological innovations such as the usage of deep longlines and the introduction of blast freezing capabilities on board, allow them to be among some of the most technically sophisticated and economically important fisheries in the world, especially those supplying the exclusive sashimi market.

Longlines are a relatively unselective gear and in many cases sharks account for a large part of the bycatches. Regularly, sharks are discarded as freezer space is limited and reserved for the valuable target species. The extent of the elasmobranch bycatch in large-scale longline operations is unknown and very difficult to assess because most of the international bodies engaged in the study and regulation of these fisheries (i.e. ICCAT, IPTP, SPC, IATTC) do not explicitly include sharks as an item in their statistics or research. This is further complicated by the absence of a comprehensive information of these fisheries on a global scale. In this section I attempt to summarise the most important characteristics of the major large-scale longline fisheries and to provide estimates of their bycatches and discards of elasmobranchs.

Atlantic Ocean.

Japan, Taiwan, Korea and Spain have the most important large-scale longline fisheries in the Atlantic Ocean. Several countries, like Canada, Cuba, USA, Italy, Morocco, Brazil and others, have longline fisheries in their own waters but their effort is very small and in some cases the elasmobranch bycatch is utilised and already included in official statistics.

Most of the information available about Atlantic high-seas fisheries comes from the International Commission for the Conservation of Atlantic Tunas (ICCAT). The information available is however of variable quality and this should be considered when interpreting the following summaries.

a) Japan.

Japanese longliners have fished for albacore (*Thunnus alalunga)* and yellowfin tuna (*Thunnus albacares)* in the Atlantic Ocean since the mid 1950's and for bigeye tuna (*Thunnus obesus)* since at least 1961. According to Susuki (1988), the fleet gradually expanded its geographical range from the Equatorial Western Atlantic grounds in 1956 to virtually all of the Atlantic by 1970 (fig. 2.35). Most recently, bigeye tuna comprises more than half of the total reported catches and is specifically targeted with deep longlines year-round in a vast area between 45°N and 45°S. These deep longlines were introduced by the Japanese fishery in 1977 and they also take welcomed bycatches of yellowfin tuna and swordfish (*Xiphias gladius).* Additionally, some effort is directed towards bluefin tuna (*Thunnus thynnus)* in the Mediterranean Sea (ICCAT 1991a).

The total number of Japanese longliners in the Atlantic during 1988 and 1989 was reportedly 183 and 239 vessels (NRIFSF 1992) with a total effort of 68'444,716 and 91'395,915 hooks respectively (ICCAT 1992). Recently published data indicate that nominal effort of the Japanese fleet continues to grow in the Atlantic Ocean, with 96,651,000 hooks set during 1990 (Uozumi 1993). Japanese reports of "other species" for 1988 and 1989 are

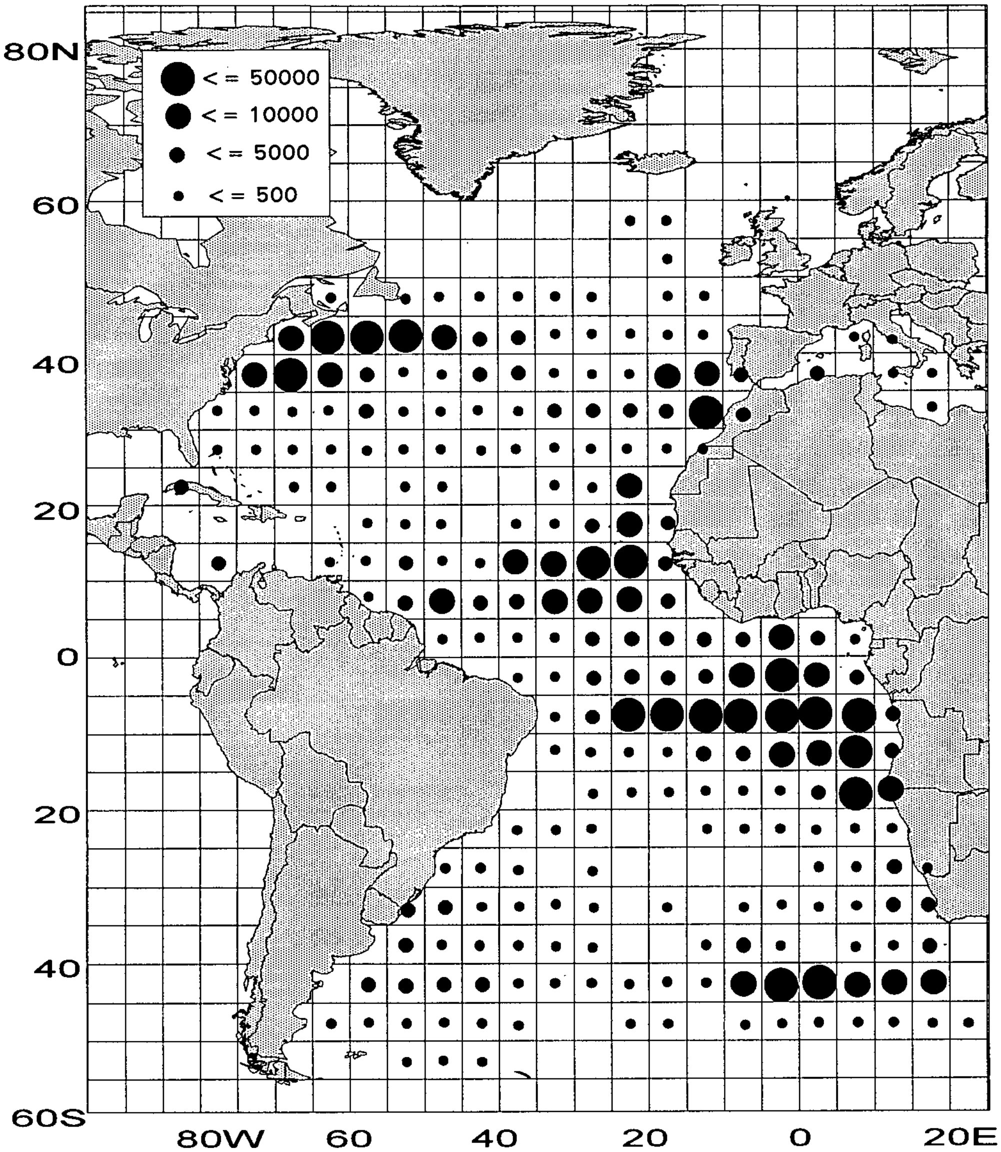



Figure 2.35 Effort distribution of Japanese longline fishery in the Atlantic Ocean in the 1980's. Keys indicate accumulated nominal hook numbers in thousands. (Redrawn from Nakano 1993).

366 and 500 t but there is no indication if these figures include any sharks or other elasmobranchs.

Hooking rates of sharks for different areas of the Atlantic Ocean where the commercial Japanese longliners operate are inadequately documented. With one exception, most of the available information pertains only to Japanese longlining activities in the North West Atlantic. Witzell (1985) estimates hooking rates of sharks in Japanese longliners at 1.31 sharks/1000 hooks (107 kg/1000 hooks) for the Gulf of Mexico and at 5.98 sharks/1000 hooks (378 kg/1000 hooks) for the USA Atlantic Coast. These are minimum estimates as they are based on Japanese logbook information and under-reporting is known to occur (Nakano 1993). In fact, reports from observers in Japanese longliners fishing in the Gulf of Mexico indicate higher hook rates, of 1.74 sharks/1000 hooks (Lopez et al. 1979). Au (1985) documents catch rates of between 1 and 5 sharks/1000 hooks as the most common for Japanese longliners in USA waters based on data from observers. Au also reports about 20 shark species in the bycatches.

Hoff and Musick (1990) provide monthly numbers of fish caught for 10 shark groups and numbers of sets made by Japanese longliners in the US EEZ in 1987. They report 8,330 sharks from more than eight species taken as bycatches in this fishery. Blue sharks comprise about 85% of the total in numbers, followed by porbeagle and shortfin mako sharks but give no indication of sharks' sizes or weights. Assuming an average of 2,206 hooks per set (derived from data of Lopez et al. 1979) the total hook rate can be accordingly estimated as 7.04 sharks/1000 hooks.

Hooking rates reported by Nakano (1993) for sharks in Japanese Atlantic operations range from 1 to 4.5 sharks /1000 hooks, with a rough average of about 2.1 sharks/1000 hooks. Nakano lists 11 elasmobranchs (10 sharks and 1 ray) identified during a research cruise in the Atlantic during the 1960's but does not give hooking rates by species. Although Nakano derives separate estimates for the North and South Atlantic, these hooking rates are underestimated because of the common underreporting of sharks in the logbooks. Most skippers do not report sharks at all whereas others record only sharks of economic value (Nakano 1993).

Information on shark bycatches of other longline operations seem to confirm the order of magnitude of the various hook rates estimated above for the Japanese fishery. Research cruises of the USA in the North Atlantic are documented by Sivasubramaniam (1963) and Brazilian tuna longliners in the Equatorial West Atlantic by Hazin et al. (1990). From the first report, hook rates of 10.35 sharks/1000 hooks can be derived for an area inside 0-80°W and 30-40°N. A smaller area inside this had catch rates for blue sharks and oceanic whitetip sharks (*Carcharhinus longimanus*) of 3.32 and 2.3 sharks/1000 hooks respectively. For the Brazilian longliners, averages can be calculated from the hook rates for six shark groups provided by Hazin et al. in 1° squares off Rio Grande do Norte. The results indicate an overall hook rate of 8.66 sharks/1000 hooks and can be further split into 3.94 for blue sharks, 4.17 for grey sharks (genus *Carcharhinus*), 0.27 for mako sharks, 0.08 for thresher sharks, 0.14 for crocodile sharks (*Pseudocarcharias kamoharai)* and 0.06 for oceanic whitetip sharks. Much higher hooking rates of up to 41.6 sharks/1000 hooks can be found in more coastal areas (Berkeley and Campos 1988).

Extrapolating from these hooking rates for specific areas to the total Atlantic is dangerous as the distribution of sharks is not homogeneous in space and time. Additionally, two different kinds of gear (regular and deep longline) are used in commercial longlining and they have different effect in the catches (Gong et al. 1987, 1989). On the other hand, this range of hooking rates can be used to place boundaries around the estimates. From the various reports listed above, there seems to be a general agreement that total hooking rate values for the Atlantic Ocean range roughly between 1 and 10 sharks/1000 hooks.

Given the scarce information available, hooking rates derived from Hoff and Musick (1990) are used here to estimate total catches of Japanese longliners in the Atlantic Ocean. They are the most updated based on data from Japanese longliners and fit well the overall range of hook rates available. Because the species composition of sharks changes according to the fishing grounds, and Japanese effort figures cannot be disaggregated, I make not attempt to estimate catch rates of individual species for the whole Japanese Atlantic longline fishery. The approximate weight of the catch is estimated with the figures of Hazin et al. (1990) of 40.91 kg per shark.

The total catch of sharks by Japanese longliners during 1989 in the Atlantic Ocean roughly estimated as outlined above is of 643,427 sharks or 26,322 t. The estimates for 1990 are

680,423 sharks or 27,835 t. However, these assessments are uncertain. The estimates for 1989 could be substantially smaller (14,619 t) if calculated using the 30% ratio of sharks to total tuna catches suggested by Taniuchi (1990), or even larger (40,149 t) if we assume the average weights reported by Witzell (1985) for the South East Atlantic USA waters. On the other hand, the average weight of 40.91 kg/shark used here seems to be reasonable, and is supported by the reports of Rodriguez et al. (1988) of a steady average weight of 48.9 kg/shark for the bycatches of the Cuban longline fleet operating in the tropical Atlantic during 1973-1985.

Witzell (1985) reports that the percentage of sharks killed in the Japanese longline fishery is only of 7.2 % in the Atlantic U.S. coast, thanks to the mandatory release of all bycatches and probably because most of the catches are blue sharks. This species, as well as other carcharhinid sharks, reportedly survives better in the longlines than lamnoid sharks (Sivasubramaniam 1963, Hoff and Musick 1990, Hazin et al. 1990). If this kill rate is common for the whole Japanese Atlantic fishery, only between 1,052 and 2,890 t of sharks died during 1989 operations. However, other reports indicate that the U.S.-enforced release of all shark bycatches in this fishery is not followed throughout the entire Atlantic (Nakano 1993). Moreover, the species composition of the bycatches is known to change latitudinally and this could alter the overall survival rates. Additional variations in the estimated bycatch of elasmobranchs are expected to be found if we consider the multiple areas and types of gears used by the Japanese longliners across the Atlantic Ocean. However, as long as more detailed information on areal, seasonal and gear-wise hooking rates is not available to assess these changes, it will be difficult to obtain better estimates.

The reported catch of elasmobranch by Japan in the Atlantic Ocean in 1989 is 1,540 t (see section 2.2). This figure falls close to the lower limit of the very ample range of elasmobranch catches estimated here. However, if we take an average of the different estimates provided above, at least 15,466 t of sharks would have been discarded. Most of them would have been finned prior to release, as acknowledged by Nakano (1993).

b) Korea.

The Korean longlining fleet had 29 vessels operating in the Atlantic in 1988 and 33 during

1989 (NFRDA 1992). This fleet uses deep longlines directed mainly at bigeye tuna since 1980. Both the number of vessels and the catches of Korea in the Atlantic have decreased since 1977. These vessels reported an effort of 21,968,198 hooks and a total "others" catch of 944 t for 1989 (ICCAT 1992). No information on the species included under "others" is available and no reports of elasmobranch bycatches for this particular fishery are known to exist.

According to the reported Atlantic fishing grounds of the Korean fleet during 1983-1985 (NFRDA 1988) most of the effort is localised between 20°N-20°S (fig. 2.36). Thus, it seems more appropriate to use the hook rates derived above from Hazin et al. (1990) for the equatorial Atlantic. This rough estimates indicate that 190,245 sharks (86,554 blue sharks, 91,607 grey sharks, 5,932 mako sharks, 1,758 thresher sharks, 3,076 crocodile sharks and 1,318 oceanic whitetip sharks) or some 7,783 t were caught during 1989 by Korean longliners in the Atlantic Ocean. This compares very high to the reported 143 t of various elasmobranchs taken in that year by South Korea in the Atlantic Ocean (FAO 1993). Presumably an elasmobranch discard of at least 97% is occurring in this fishery. The proportion of sharks released alive and the extent of finning practices in the Korean fishery are unknown.

c) Taiwan.

Longliners from Taiwan have fished for albacore in the South Atlantic since at least 1967 and in the North Atlantic at least since 1972. Typically, more than 80% of their catches are albacore, followed by bigeye tuna. During 1989, Taiwan deployed 3.6 million hooks in the North Atlantic and 68.7 million in the South Atlantic (ICCAT 1991b). According to Hsu and Liu (1992), in 1990 this increased to a total of 99.8 million hooks, 17.4 and 82.4 million in the North and South Atlantic respectively. Of these, 17.5 million hooks were from deep longlines directed towards bigeye and yellowfin tunas, while the remaining 82.2 million hooks were regular longlines fishing for albacore principally in the South Atlantic (fig. 2.37). Hsu and Liu also report a Taiwanese catch of 736 t of sharks and other fishes for 1990. During 1991, the number of vessels operating in the Atlantic fell about 10%. However, reported shark bycatches increased to 1,486 t (Hsu and Liu 1993). The reports of Hsu and Liu (1993) indicate that the variations in the reported bycatches of sharks from this fishery

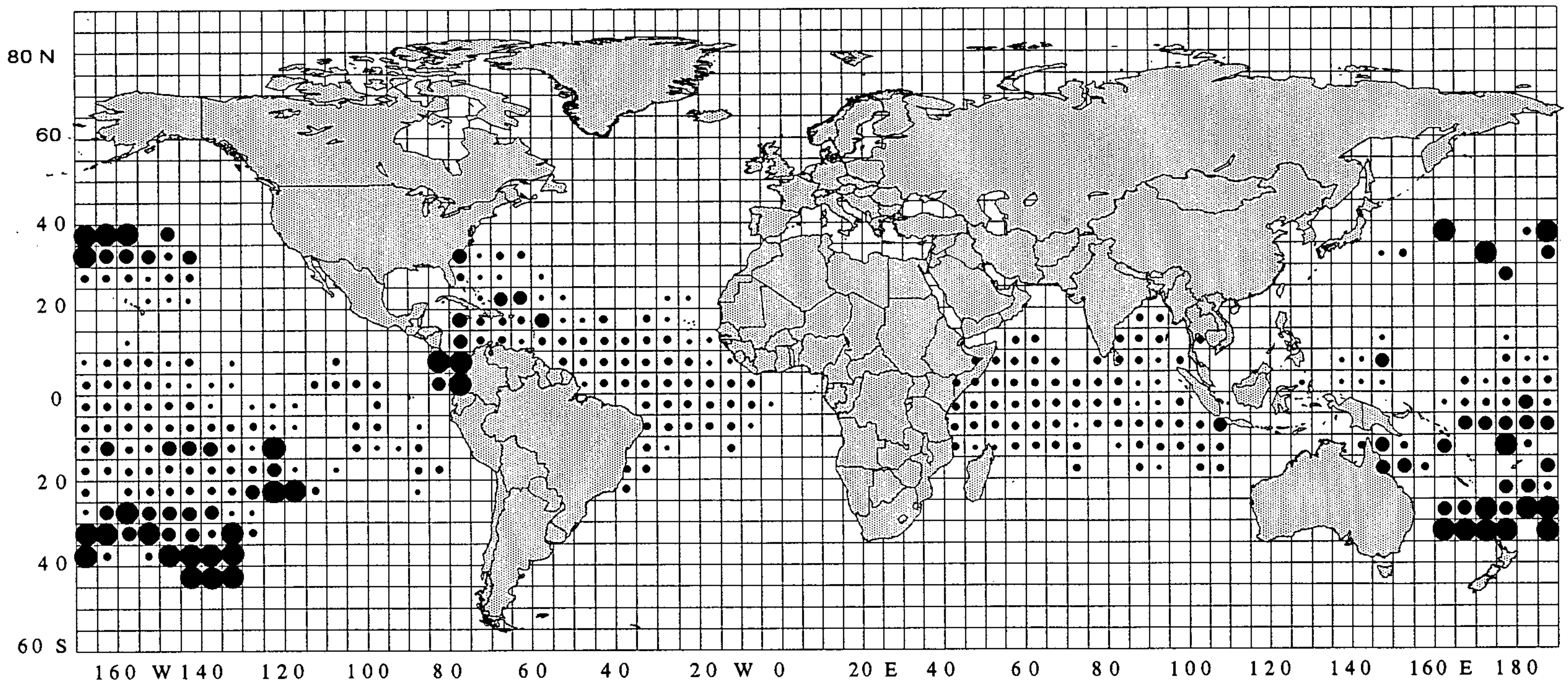



Figure 2.36 Distribution of Korean long-line catches, no units given. (Redrawn from NFRDA 1988).

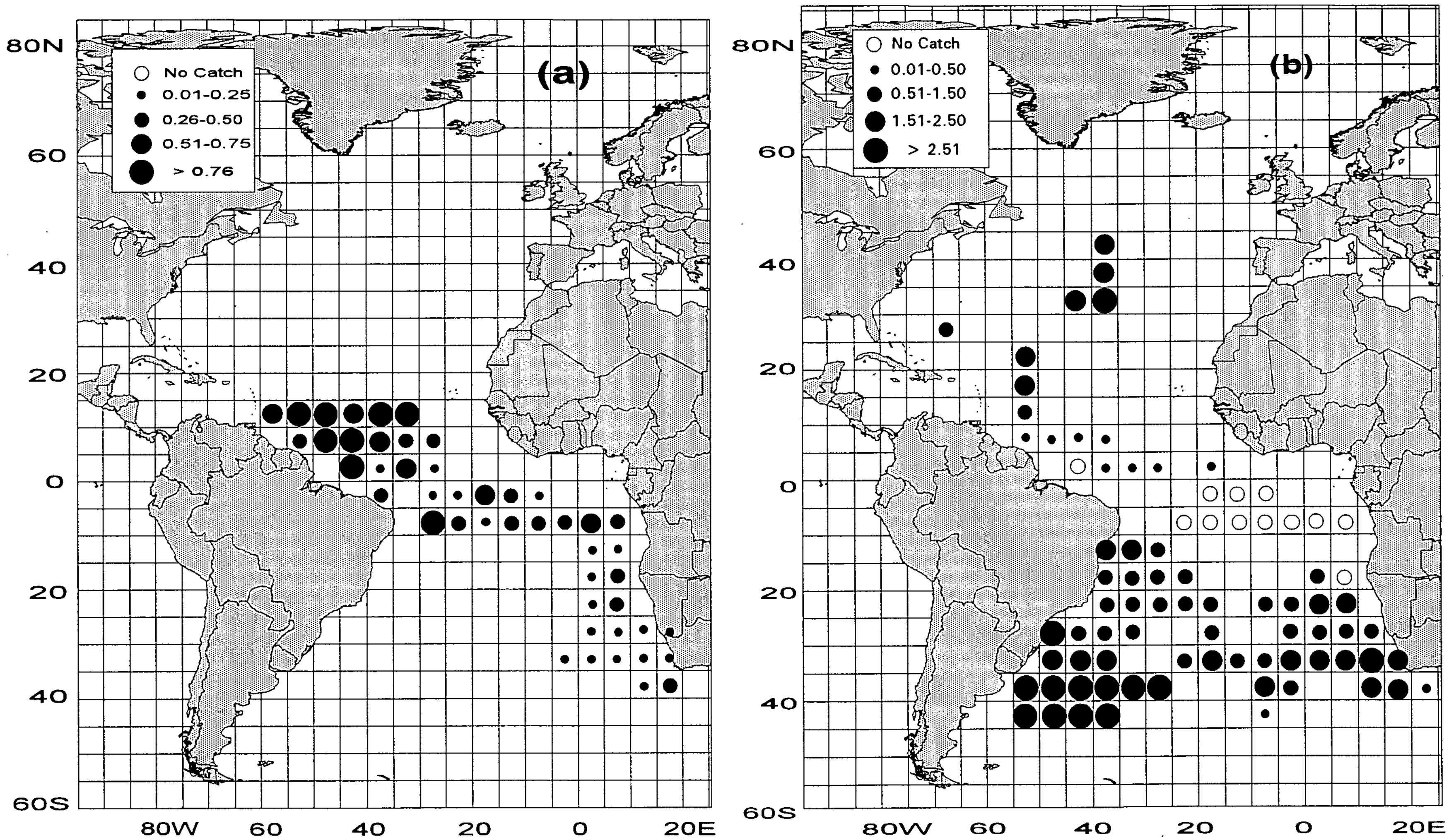



Figure 2.37 Distribution of nominal CPUE of bigeye tuna (a) and albacore (b) in the deep and regular longline fisheries of Taiwan in the Atlantic Ocean, 1990. (Redrawn from Hsu and Liu 1992).

are determined by the success in the catch of the target species. In years when tuna catches are relatively low, vessels tend to keep a larger proportion of the shark bycatch.

The reported catch of sharks in this fishery seems very small for the number of hooks deployed by the Taiwanese longlining fleet. The Taiwanese fleet fishes predominantly in the South Atlantic and for this reason the hooking rates derived from Hazin et al. (1990) seem more appropriate for the purpose of estimation. Nevertheless, as a large part of the effort takes place in temperate waters I do not attempt to brake down the bycatches into species. This way, it is roughly estimated that 864,268 sharks were caught in 1990 (probably equivalent to 35,357 t) by the Taiwanese longliners.

The real quantity of elasmobranchs taken by Taiwan from the Atlantic Ocean is unknown. The present analysis is certainly rough due to limited information. However, it suggests that a massive discard of around 34,000 t of sharks could be taking place in the fishery. As in the case of the other fisheries documented above, estimating the actual number of sharks released alive and discarded dead is very difficult with the available information.

d) Spain.

The Spanish longline fishery for swordfish in the Atlantic can be traced back to at least 1973 (Garces and Rey 1984). Fishing grounds for 1988-1991 were centred in the Eastern Atlantic between 55°N and 15°S (fig. 2.38), although some fishing has also been reported for the Mediterranean. Surface longlines are used in waters of the North Atlantic but deep longlines have been introduced in the Southeast Atlantic since the recent expansion of the fishery there. The deep longlines are consist of baskets of about 1,200 m of line between floats and have some 33 branch lines 15 m in length with the deepest hooks reaching down to between 360 and 470 m (Rey and Muñoz-Chapuli 1991). The Spanish fleet set 35,850,078 hooks in the Atlantic Ocean and 7,683,580 hooks in the Mediterranean Sea during 1989, with increases of 6.75 and 7.3% during 1990 respectively (ICCAT 1991a, 1992).

De Metrio et al. (1984) report hooking rates of blue sharks in swordfish longlines in the Mediterranean of 0.014 sharks/1000 hooks. However, their reports do not consider other shark species or the discards done at sea and are thus biased towards small hook rates.

Rey and Alot (1984) provide data from the Spanish swordfish fleet in the western Mediterranean. They indicate hooking rates of 6.34 blue sharks, 0.32 shortfin mako sharks, 0.21 smooth hammerhead sharks (*Sphyrna zygaena)* and 0.005 pelagic rays, per 1000 hooks.

Mejuto (1985) reports CPUE values of 138.8, 17.5 and 1.1 kg/1000 hooks for blue, shortfin mako and porbeagle sharks respectively in the north and north western grounds of the Spanish Atlantic swordfish fleet, based on a sample of 200 trips during 1984. This is equivalent to hook rates of 13.7, 0.259 and 0.016 sharks/1000 hooks respectively for each species. These catch rates take into consideration the discards at sea of blue sharks, which Mejuto estimates at 68.4% in weight. Mejuto also finds a linear relationship between catches of swordfish and discards of blue sharks, which is driven by the limitations in storage capacity and low value of blue sharks. He points out that in many cases fins are removed before discarding the sharks. More recently, Mejuto and Iglesias (1988) provide information from exploratory swordfish longlining carried out during 1986 in the Western North Atlantic. From their data, catch rates of 13.5 and 2.05 sharks/1000 hooks or 168 and 61.7 kg/1000 hooks can be calculated for blue and shortfin mako sharks respectively.

The elasmobranch bycatch of Spanish longliners includes more than the three species mentioned above. Muños-Chapuli (1985b) reports 16 species of sharks in the landings of vessels fishing between Cape Verde Island and the Azores. The blue shark, the shortfin mako and the smooth hammerhead shark *Sphyrna zygaena,* were, in order, the most abundant species in the catches (table 2.6 section 2.2.2).

Limited information from the southern Atlantic fishing grounds of the Spanish swordfish fishery where deep longlines are used, indicates important changes in the species composition. Rey and Muñoz-Chapuli (1991) report 14 elasmobranchs in this area from the catches of 16 nights of fishing of a single commercial longliner. From their data, average hook rates in sharks/1000 hooks are estimated as 20.6 for night sharks *Carcharhinus signatus,* 6.3 for silky sharks, 3.4 for bigeye thresher sharks, 2.9 for blue sharks, 2 for devil rays *Mobula sp.,* 1.8 for shortfin mako sharks, 0.3 for common hammerhead sharks and less than 0.3 each for *Sphyrna couardi, S. mokarran, S. zygaena, Centrophorus granulosus, Galeocerdo cuvieri, Isurus paucus* and *Carcharhinus plumbeus.* The overall hook rate of elasmobranchs is estimated at 38.8 fish/1000 hooks, which is fairly high compared to that for Spanish swordfish longliners in the North Atlantic. The different areas fished and gears used could explain these discrepancies, but the limited time window and number of operations observed by Rey and Muñoz-Chapuli could also be a significant source of bias.

The total catch of sharks in the Spanish fishery for 1989 can be estimated using the results of Mejuto (1985). His report does not only take into account the discards of blue sharks and provide catch rates in weight, but it covers a larger time frame and geographic area than other reports. Hence, for the effort levels of 1989, a total of more than 608,000 sharks weighing some 6,856 t would have been caught in this fishery (5,646 t in the Atlantic and 1,210 t in the Mediterranean; table 2.17). Given the discard rate of 68.3% for blue sharks in the spanish swordfish fleet reported by Mejuto, the total discard of blue sharks from the spanish fishery during 1989 could be as high as 4,134 t.

The results presented above should be used with caution as they are based on estimated data coming from only part of the geographical area fished by the Spanish fleet. They are useful only to get an idea of the extent of the elasmobranch bycatches and discards. Better estimates that take into account other species present in the catches and the geographical/seasonal variations of catch rates and species composition will have to await improved information from this or similar fisheries.

Indian Ocean.

A number of countries fish for tunids with longlines in the waters of the Indian Ocean. The three principal longline fleets are from Japan, Korea and Taiwan, which entered the fishery in 1952, 1963 and 1966 respectively. Indian longliners started fishing for tunas in 1986 but their catches, as well as those from other few countries, are very small in comparison (IPTP 1990). Most of the information about longline fisheries in the Indian Ocean is available through the reports of the Indo-Pacific Tuna Development and Management Programme (IPTP).

The Japanese fleet fished tropical areas for yellowfin, albacore and bigeye tunas at the beginning of the fishery but shifted to higher latitudes for southern bluefin and bigeye tuna

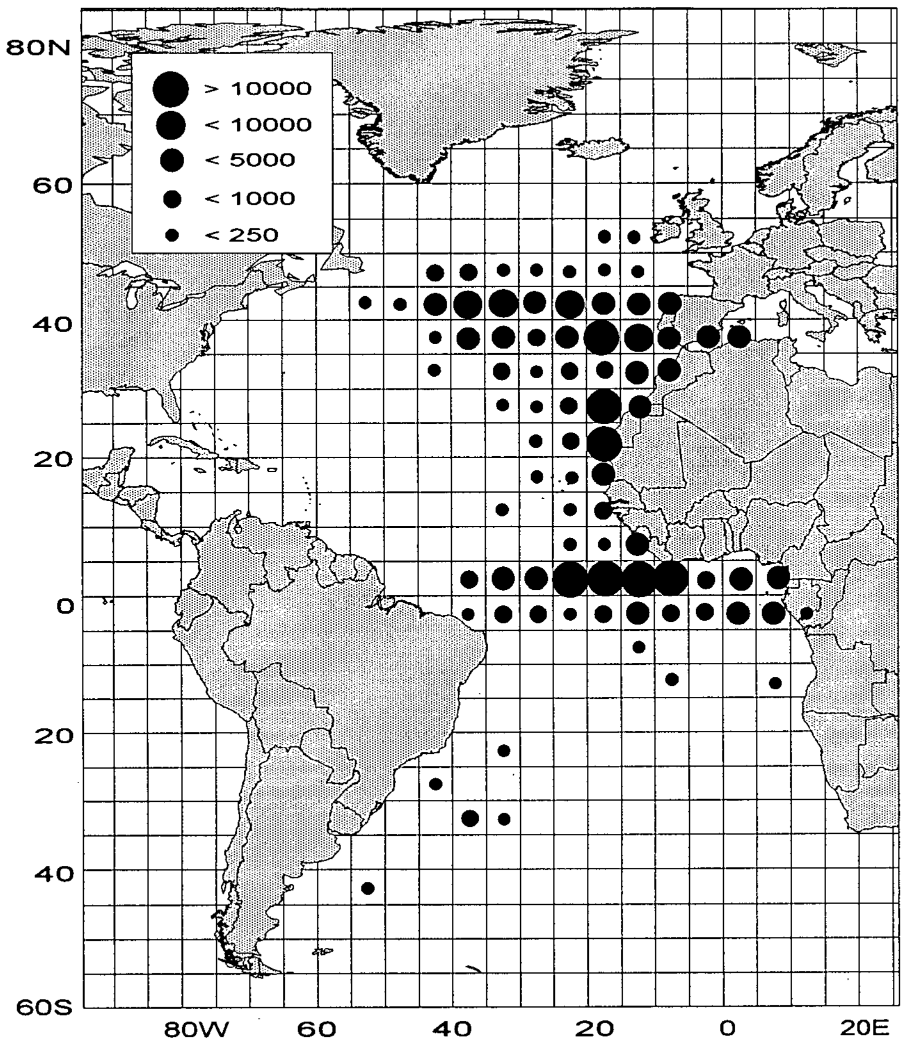



Figure 2.38 Distribution of effort (in thousands of hooks) by the Spanish swordfish longline fishery in the Atlantic Ocean during 1988-1991. (Redrawn from Mejuto et al. 1993).

Table 2.17 Catch rates and estimated total catch of sharks in the Spanish swordfish fishery.

| | Information from Mejuto (1985) | | | | Estimated total catch 1989 | | | |
|---|---|---|---|---|---|---|---|---|
| | Numbers | Weight (t) | Hook rate | CPUE | Mediterranean (7.68 M hooks) | | Atlantic (35.8 M hooks) | |
| Species | (17.344 M hooks) | | (sh/1000 h) | (kg/1000 h | Numbers | Weight(t) | Numbers | Weight(t) |
| *Prionace glauca* * | 237,660 | 2,408 | 13.703 | 138.8 | 105,286 | 1,067 | 491,244 | 4,977 |
| *Isurus oxyrinchus* | 4,488 | 304 | 0.259 | 17.5 | 1,988 | 135 | 9,277 | 628 |
| *Lamna nasus* | 272 | 20 | 0.016 | 1.1 | 120 | 9 | 562 | 41 |
| Totals | 242,420 | 2,732 | 14 | 158 | 107,395 | 1,210 | 501,083 | 5,646 |

* includes estimated discards (68.4%)

during the 1970's, while introducing deep longlining in tropical waters at the same time. Judging from data reported to IPTP, Japanese longliners have decreased their effort from 106.64 million hooks in 1986 to 74.861 million hooks in 1989. The data records of Japanese longliners in the Indian Ocean do not report any shark bycatches in the fishery. However, in FAO yearbooks Japan reports 675 t of "various elasmobranchs" caught in the Indian Ocean during 1989. Given that the only Japanese fishery taking place in those waters is the tuna longline fishery (except for three newly introduced purse seiners), the elasmobranch catches reported to FAO, although small, can be attributed mainly to shark bycatches of the longliners.

Taiwanese vessels take the largest catches of albacore but also fish for yellowfin and bigeye tunas primarily using deep longlines in tropical waters (fig 2.39). There were 199 vessels in the fishery in 1983, 127 in 1985, and 187 in 1988. The total effort in nominal hooks during 1988 was estimated at 107 million by Taiwanese researchers (IPTP 1990). On the other hand, unpublished data from IPTP indicate that Taiwan caught 33,052 sharks with a total weight of 1,216 t in this period, with a total effort of 130,235,742 hooks. For 1989 these values were 188,615 sharks or 7,474 t with an effort of 136,418,296 hooks.

Korean longliners operate primarily in the tropical Indian Ocean targeting bigeye and yellowfin tunas with deep longlines (fig. 2.36). The number of vessels has varied considerably, peaking in 1975 at 185, decreasing to 62 in 1985 and reaching 112 in 1988 (IPTP 1990). According to most recent available data for IPTP, Korean vessels caught 10,851 sharks in 1987 and the effort was 35,748,292 hooks.

The absence of Japanese reports of sharks taken in the fishery make it necessary to estimate their bycatches. Additionally, the analysis of apparent hooking rates derived from the reports of the Korean and Taiwanese fisheries were too low compared with alternative data from the Indian Ocean (see below) and similar fisheries in other oceans (i.e. Atlantic Ocean). The hooking rates calculated from the data reported above are 1.38 sharks/1000 hooks for Taiwan in 1989 and 0.3 sharks/1000 hooks for Korea in 1987. I considered that these catch rates reflect considerable under-reporting and are therefore of little use. The selection of catches and discard of sharks in high-seas tuna fisheries is a very common practice. In the following paragraphs, the available information on shark catch rates in the

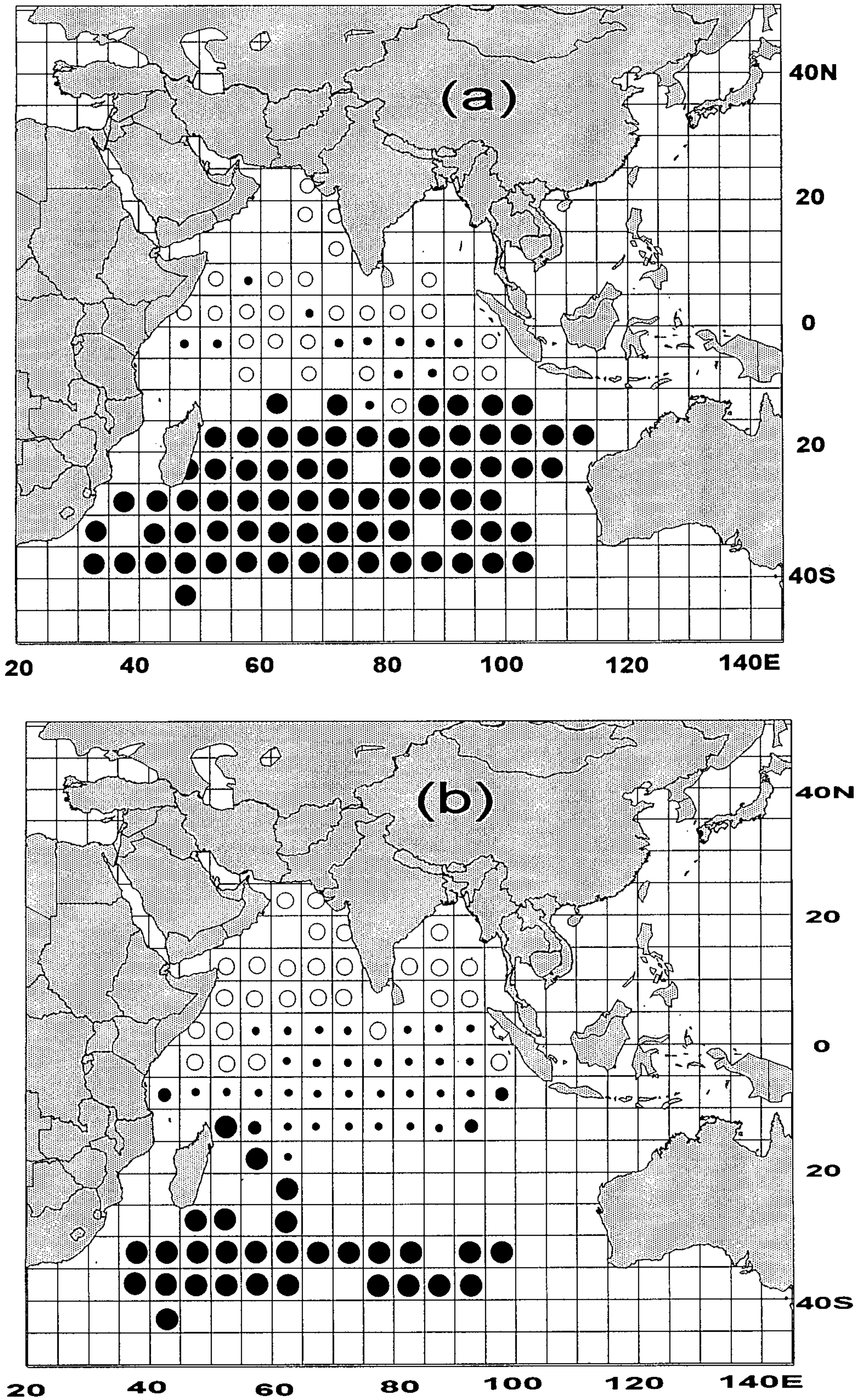



Figure 2.39 Distribution of Taiwanese catch per unit effort of albacore by (a) regular and (b) deep longline fisheries during 1988 in the Indian Ocean. (Redrawn from Hsu and Liu 1990).

Indian Ocean is analysed to provide alternative estimates of the bycatches of sharks in these three tuna fisheries.

Information on shark bycatches in the Indian Ocean longline fisheries is relatively abundant and allows for geographical partitioning in some cases. However, virtually no reports include data on hooking rates by species. The only indication of species composition comes from Taniuchi (1990) who reports the percentage of each species in the shark bycatches of research tuna longliners from Japan. His results indicate that 76.6% are blue sharks, 6.6% silky sharks, 6.5% shortfin mako sharks, 3.4% oceanic whitetip sharks and 6.8% unidentified sharks. Sivasubramaniam (1963) provides data on early research operations by Japanese and Taiwanese vessels that indicate bycatches of 10.83 sharks/1000 hooks for the eastern Indian Ocean (E of 60°E). Sivasubramaniam (1964), reports on commercial and research operations for six areas of the Indian Ocean and indicates that about 20 species of sharks occur in the bycatches, 11 of these sharks (mainly carcharhinids) are very common (table 2.18). The results of Sivasubramaniam indicate latitudinal changes in species composition of sharks and higher hooking rates for sharks north of the equator. He reports that the frequency distributions of hooking rates for sharks for six areas of the Indian Ocean show a range of 0-4 to 44.1-49 sharks/1000 hooks, with a modal class of 4.1-8 sharks/1000 hooks. Mimura et al. (1963) report hooking rates by area and season that average 5.1 sharks/1000 hooks (range 2.6-7.3).

On more recent reports, Pillai and Honma (1978) provide monthly catch rates of pelagic sharks in 10°x20° squares for the Japanese fleet in the Indian Ocean that range between 0.1 and 50 sharks/1000 hooks. Varghese (1974; cited by Pillai and Honma (1978)) reports hooking rates as high as 84 sharks/1000 hooks and an average weight of 57 kg/shark in the Lakshadweep Sea. According to Silas and Pillai (1982), hooking rates of sharks in the Indian Ocean vary from year to year and between areas, the highest being between 0.6 and 10 sharks/1000 hooks. They also report that in the area of the Southeast Arabian Sea, sharks were 63.8 and 57.8% of the total catch in number and weight respectively and had an average weight of 30 kg. Sivasubramaniam (1987) summarises data from Fisheries Survey of India tuna research cruises in the south west coast of India during 1983-1986. These results indicate hooking rates of 17.6 sharks/1000 hooks. James and Pillai (1987) review additional research cruises in areas of the Southeast Arabian Sea, Andaman Sea,

Table 2.18 Shark species commonly caught by tuna longlining in the Indian Ocean (adapted from Sivasubramaniam, 1964).

| Scientific name | Caught by longline (approx. mean wt.) |
|---|---|
| *Carcharhinus longimanus* | 30 kg |
| *C. falciformis* | 60 kg |
| *C. albimarginatus* | 40 kg |
| *C. melanopterus* | 35 kg |
| *Prionace glauca* | 50 kg |
| *Isurus oxyrinchus* | 75 kg |
| *Lamna ditropis* | 75 kg |
| *Galeocerdo cuvieri* | ? |
| *Sphyrna spp.* | 75 kg |
| *Alopias pelagicus* | 50 kg |
| *A. superciliosus* | 100 kg |

Western Bay of Bengal, and the Equatorial Region south of India, providing figures for the percentage contribution of sharks to the total catch averaging 39.8% (range 30.9-43.7%). They also report hooking rates that average 16.4 sharks/1000 hooks (range 7.4-29.7) in the Southeast Arabian Sea. James and Jayaprakash (1988) report two different studies comprising several areas around India that indicate hooking rates of 8.43 sharks/1000 hooks (range 3.3-14) and contribution of sharks to the catches of 32.1% (range 19.6-44.8) in one case and hooking rates of 7.6 sharks/1000 hooks (range about 1.5-9.5) and contributions of sharks to the catch of 17.4% in the other. Stevens (1992) reports hooking rates of 8.3 blue sharks and 3.5 mako sharks per 1000 hooks for a Taiwanese research longliner in south Western Australia.

Strong variations are evident in hooking rates across the Indian Ocean depending on location and season. Ideally, an overall estimate of elasmobranch bycatches should be built at least considering areal differences. Unfortunately, this is not possible given the aggregated nature of effort statistics for the fleets of Japan, Korea and Taiwan. Nevertheless, there seems to be a consensus around 1-10 sharks/1000 hooks as the most common hooking rate. Total catches of sharks in numbers for the whole Indian Ocean are thus roughly estimated using a hooking rate of 7.96 sharks/1000 hooks obtained by averaging the values derived from Sivasubramaniam (1963) and Mimura et al. (1963). These values do not only come from data pertaining to most of the Indian Ocean but also match with the most common hooking rates reported by different sources. The average weight of sharks taken in the fishery is estimated at 38.21 kg derived from the weight and numbers of sharks reported for Taiwanese longliners during 1988 and 1989. The estimated shark bycatches for the last available effort levels are: 596,267 sharks or 22,783 t for Japan during 1989, 248,735 sharks or 10,879 t for Korea during 1987 and 1,086,572 sharks or 41,518 t for Taiwan in 1989. The total catch of sharks in the Indian Ocean tuna longline high-seas fishery can be estimated at approximately 1,931,574 sharks or 75,180 t.

Based on the reported catches of elasmobranchs from each county, the corresponding discards of sharks are estimated at about 22,108 t by Japan, 9,089 t by Korea and 34,044 t by Taiwan. The percentages of these discards that survive are not known, but judging from the reports of Sivasubramaniam (1963; 1964) about 70-80% of the discards of carcharhinid sharks are expected to survive if released alive whereas hammerheads and mako sharks are dead on the line most of the time. However, the rate of finning although unknown, is expected to be high.

The various estimations provided here are limited by the variability of hooking rates reported for the Indian Ocean and the roughness of the effort statistics. They should be taken with caution and used as a first approximation to the level of elasmobranch bycatches and discards in these fisheries.

Tropical and South Pacific.

There are several fleets fishing for tuna in this area, home to many small insular countries. However, most of the longline operations are carried out, in order of importance, by Japanese, South Korean, Taiwanese and Australian vessels. In general, these fisheries are very poorly documented, making it very difficult to ascertain elasmobranch bycatches. Most of the available information for the central Pacific area is that submitted to the South Pacific Commission (SPC) by the fishing nations and made available through the Forum Fisheries Agency (FFA) (P. Tauriki, FFA, P.O Box 629, Honiara, Solomon Islands, pers. comm. June 1992). Additionally, Australian and New Zealand sources provide some information pertaining their EEZ's.

There seems to be a gap in information for those areas of the eastern Pacific where neither Australia nor New Zealand has jurisdiction. Furthermore, the coverage of the fishing fleets by the FFA data is partial (Lawson 1991). Hence, the effort levels for the entire central and south Pacific area are unknown and suspected to be larger than those available through the sources used here. The area considered in this section as "Tropical and South Pacific", is that comprised by waters south of 20°N.

Japanese fishermen started experimenting with longlines in the western central Pacific as early as the 1920's and had 72 vessels active by 1939. However the peak in the expansion of this fishery occurred during the late 1960's, covering most of the central and south Pacific (Suzuki 1988, Lawson 1991). At present, at least 406 vessels are suspected to operate in the region. The FFA database indicates that Japan deploys more than 70% of the total effort in the area, with 31,143 fishing days in 1989.

The South Korean longline fleet appeared during 1958 and is reported to have 124 vessels at present in the area. According to South Korean records (NFRDA 1988), longliners from this country fish largely for tunids in the South Pacific (fig. 2.36). The South Korean effort in the FFA zone is reported as 6,312 fishing days for 1989.

The Taiwanese fleet is poorly documented and there is not even an estimate of the number of vessels in the region. A partial coverage of the Taiwanese effort indicates their presence in the waters north of Papua-New Guinea, but they are also known to fish around Fiji and American Samoa (Lawson 1991). According to FFA data, the Taiwanese fleet accumulated an effort of 4,163 fishing days in 1989.

The Australian longline fisheries for tuna expanded in the 1980's but date back to the 1960's. During 1989, Australian long-liners put a total effort of 2,244 fishing days (P. Tauriki, FFA, pers. comm). In addition to the above mentioned fleets, a few vessels from China, Fiji and Tonga participate in the fishery. However, in 1989 their effort only accounted for a total of 558 fishing days. The geographical distribution of total longline effort during 1990 available to the SPC is shown in figure 2.40. Most of the fishing effort takes place between 15°N and 15°S.

The reported catch of sharks for 1989 in this area was 426 t. Approximately 375 t corresponds to Taiwanese, 35 t to South Korean, and 12 t to Japanese vessels. Although the number of hooks deployed by each country were not available, the total for all longliners amounts to 98,832,500 hooks during 1989. The number of hooks per country can be estimated using the reported fishing days of each fleet. The corresponding estimated catch rates in kg/1000 hooks are 0.167 for the Japanese, 2.5 for South Korean, 40.5 for the Taiwanese and 0 for the Australian fleet. This is equal to an overall catch rate of 4.31 kg of shark per 1000 hooks. Such minuscule catch rates, grossly equivalent to less than 0.5 sharks/1000 hooks when compared to other reports, are almost certainly a result of severe under-reporting, presumably due to discard processes. This is evident also in the cross-comparison of the estimated catch rates for each of the mentioned countries.

Saika and Yoshimura (1985) plot hooking rates for the most common shark species caught by Japanese research longliners in the western equatorial Pacific. These are approximately

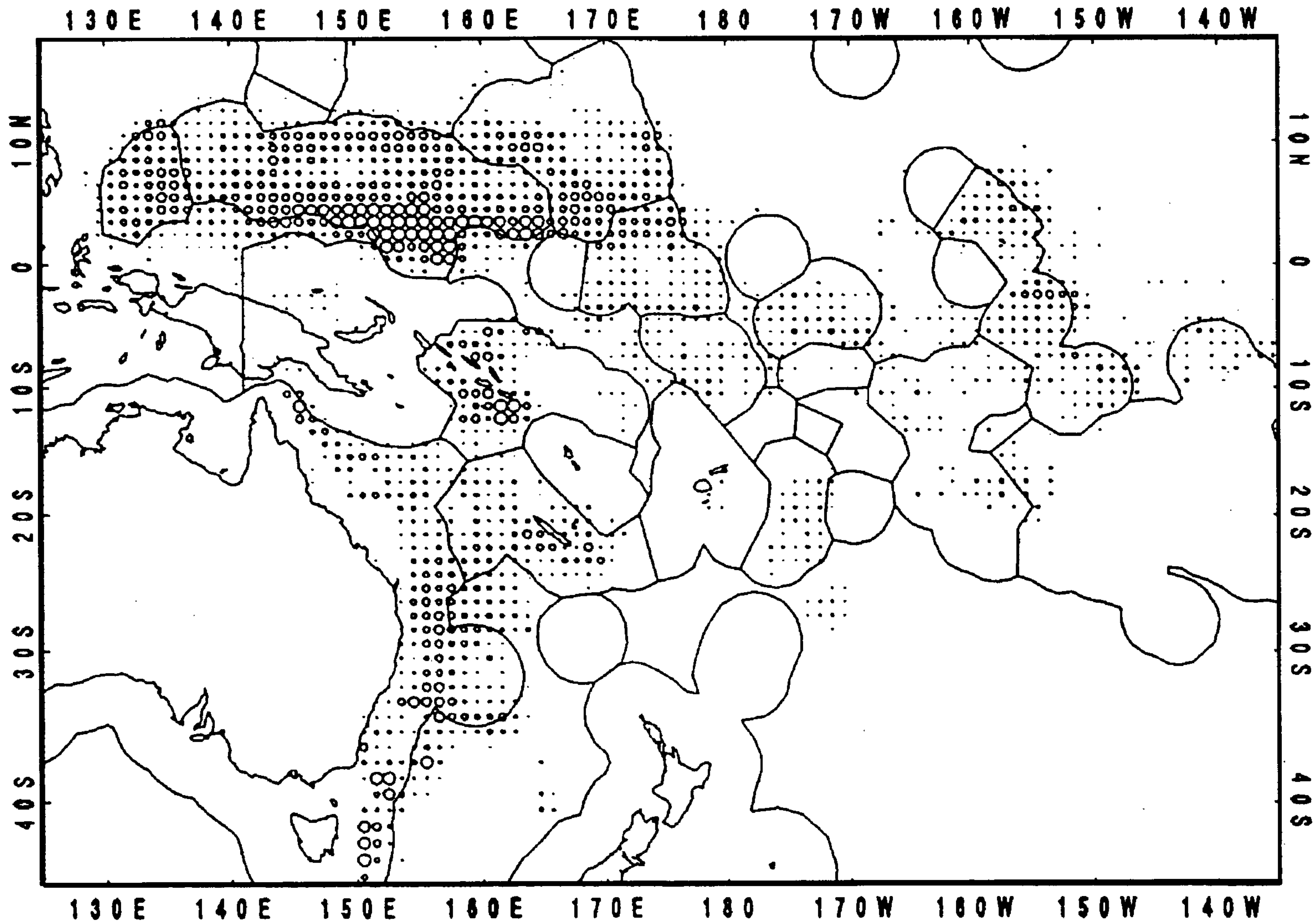



Figure 2.40 Distribution of longline effort in the SPC area during 1990, units not given. (Taken from Lawson 1991).

0-14/1000 hooks for oceanic whitetip and for silky sharks, 0-16/1000 hooks for blue sharks and 0-2/1000 hooks for shortfin mako sharks.

An overall hooking rate of 20.45 sharks/1000 hooks can be estimated for waters below 22°N from the report of Strasburg (1958) on research and commercial cruises in the eastern equatorial Pacific. This can be further split into 4.14 blue, 5.46 oceanic whitetip, 10.07 silky and 0.78 unidentified sharks, per 1000 hooks.

Stevens (1992) reports on the bycatches of blue and mako sharks of longliners off Tasmanian waters based on information collected by observers on board Japanese vessels fishing for southern bluefin tuna (*Thunnus maccoyii*). He provides hooking rates of 10.4 blue and 0.5 mako sharks per 1000 hooks and estimates that 1,594 mako and 34,000 blue sharks weighing 24 and 275 t respectively, are caught each fishing season in this fishery. Hooking rates for other species are not available but Stevens mentions that thresher, porbeagle, school (*Galeorhinus galeus*), black (*Dalatias licha*), crocodile (*Pseudocarcharias kamoharai*), hammerhead, velvet dogfish (*Zameus squamulosus*) and grey (*Carcharhinus*) sharks are also present in the bycatches of Japanese longliners in the Australian Fishery Zone. Stevens also provides data for the bycatches of blue sharks in New Zealand waters: Japanese and Korean fisheries have hooking rates of 4.8 and 1.3 blue sharks/1000 hooks respectively in Northern New Zealand, whereas the Japanese catch rate in southern New Zealand is 4 blue sharks/1000 hooks. Stevens draws attention to the strong under-reporting of shark bycatches in Japanese logbooks and reports that fins are cut off from the sharks before being discarded. This suggests that mortality in this fishery might be equal to the total bycatch.

Ross and Bailey (1986) report hooking rates for mako sharks in the Korean and Japanese fisheries for albacore of northern New Zealand and for the southern New Zealand Japanese fishery for southern bluefin tuna. Averages are 0.43 and 0.34 sharks/1000 hooks for the northern and southern fisheries respectively. The total catch of mako sharks can be estimated at 334 t processed weight based on their data, however, because about 50% of the shark's weight is lost during processing, the live weight of the mako shark bycatch should be approximately 668 t. Although Ross and Bailey do not provide further information, it is suspected that this figure only represents the reported catch and does not consider discards.

The total bycatch of sharks in the SPC zone can be roughly estimated using the figures from Strasburg (1958) and a conservative guess of 20 kg/shark to calculate the total weight of the catch. Even though this catch rate might appear to be comparatively too high, the distribution of effort in these fisheries (see figure 2.40) justifies the usage of hooking rates from the Equatorial Pacific. The results, presented in table 2.19, indicate that approximately 2,021,711 sharks or 40,434 t were caught in 1989 of which probably almost 50% were silky sharks. Japan contributes the majority of the elasmobranch catches and also has the highest discard rate. The total discard is estimated at 40,000 t.

The shark bycatches in the whole Tropical and South Pacific might still be higher. Judging from size of the statistical area covered by the SPC (fig. 2.34) and the maps of CPUE of the South Korean longline fleet for the years 1983-1985 (see figure 2.36) and considering the partial coverage of the SPC area by FFA statistics (SPC 1991), it is estimated that the South Korean fleet deployed twice as many hooks in the whole central and south Pacific as those reported by the FFA. This applies also to the Japanese and Taiwanese fleets. Under this assumption, the total catch of sharks in the central and south Pacific outside the SPC zone could be around 1,097,288 sharks or 21,946 t; 16,422 t by Japan, 3,328 t by South Korea and 2,196 t by Taiwan. These figures assume an extra effort of 92,598,173 hooks (1989) and a total hooking rate of 11.85 sharks/1000 hooks. This hooking rate tries to take into consideration the possible occurrence of effort outside equatorial areas and was calculated by averaging the total hooking rates obtained from Strasburg (1958) for the equatorial zone, from Stevens (1992) for Tasmanian waters, and from Ross and Bailey (1986) and Stevens (1992) for New Zealand waters. The same estimated weight of 20 kg/shark as above was used. Accordingly, it is estimated that some 62,380 t of sharks were caught as bycatch of longline fisheries in the whole central and south Pacific in 1989. According to FAO statistics, the joint reported catch of elasmobranchs from the West Central, South Western and South Eastern Pacific of Japan, Taiwan and Korea is only 4,409 t for 1989. These figures suggest that some 58,000 t of sharks may be discarded in these fisheries.

The estimates obtained above are relatively more uncertain than those calculated in

Table 2.19 Estimated bycatch of sharks in tuna longline fisheries of the Central and South Pacific (SPC zone), based on the results of Strasburg (1958).

| | Strasburg's data | | Estimated Catch in 1989 | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Numbers caught | Hook rate | Total | | Japan | S. Korea | Taiwan | Australia |
| Species | (216,172 hooks) | (#/1000 hooks) | numbers | weight (t) | weight (t) | weight (t) | weight (t) | weight (t) |
| *Carcharhinus falciformis* | 2,176 | 10.07 | 994,854 | 19,897 | 13,950 | 2,827 | 1,865 | 1,005 |
| *Carcharhinus longimanus* | 1,181 | 5.46 | 539,946 | 10,799 | 7,571 | 1,535 | 1,012 | 546 |
| *Prionace glauca* | 896 | 4.14 | 409,646 | 8,193 | 5,744 | 1,164 | 768 | 414 |
| Various sharks | 169 | 0.78 | 77,266 | 1,545 | 1,083 | 220 | 145 | 78 |
| Totals | 4,422 | 20.46 | 2,021,711 | 40,434 | 28,349 | 5,746 | 3,789 | 2,043 |

previous sections for other high-seas longline fisheries. This is an unfortunate consequence of the limited information available, both about the real effort levels of each fleet and about hooking rates in the South Pacific. Again, these estimates can be used for comparative purposes and as preliminary information to be revised whenever appropriate baseline data become available.

North Pacific.

This is another region where longline fisheries are very poorly documented.The reports of NFRDA confirm that Korean longliners fished in the central north Pacific during 1983-1985 (fig. 2.36). Figures from Suzuki (1988) also indicate that the Japanese longline fleet covers good part of the North Pacific. However, Taiwanese vessels do not have a high-seas longline fishery in this area (Nakano and Watanabe 1992). There are no statistics available, at least in English, of the amount of effort deployed by longliners in the North Pacific.

Nakano and Watanabe (1992) estimate longline effort of the Korean fleet at 14-19 million hooks per/yr for the period 1982-1988. Using this estimate and statistics from the Fishery Agency of Japan they arrive at a total effort of 258'422,780 hooks deployed during 1988 in the entire North Pacific by Japan and Korea. Their estimate of about 3,274,609 blue sharks caught by longline fisheries in the North Pacific during 1988 is based on latitudinal stratification of effort and hooking rates. Because of the geographical coverage considered in the previous section for the Tropical and South Pacific, only waters north of 20°N are included here as "North Pacific". From the data of Nakano and Watanabe, it is possible to estimate a total effort of 105'885,418 hooks and a bycatch of 2'964,500 blue sharks for this portion of the North Pacific during 1988.

Data derived from the reports of Strasburg (1958) produce overall hooking rates of 18.45 blue, 0.07 oceanic whitetip and 0.84 unidentified sharks (total of 19.36 sharks/1000 hooks) for the eastern north Pacific north of 22°N. Tabulated data from Sivasubramaniam (1963) allows the calculation of hooking rates of 6.79 blue and 0.35 oceanic whitetip sharks/1000 hooks for two combined areas of the North Pacific above 20°N. Saika and Yoshimura (1985) present data on shark bycatches of Japanese research cruises from 1949-1979 in the Western Pacific. Their maps of hooking rates indicate values of approximately 0-3 oceanic whitetip, 0-0.5 silky, 0-2 shortfin mako and 0-30 blue sharks per 1000 hooks for the region north of 20°N. The most common hooking rate values plotted for blue sharks appear to be around 10 sharks/1000 hooks, whereas those for the other three species probably are less than 1 shark/1000 hooks. On the other hand, Nakano et al. (1985) report numbers of blue sharks caught and number of stations sampled for longline cruises during 1978-1982 in the western north Pacific. These longlines had between 1500-1800 hooks. Assuming a mean of 1,650 hooks per station, hooking rates averaged to 17.62 blue sharks/1000 hooks, quite similar to the figure calculated from Strasburg's data.

The estimated total bycatch of sharks by tuna longlines in the North Pacific is comparatively very high. Based on the hooking rates derived above from Strasburg (1958) and the effort estimated by Nakano and Watanabe (1992), a total of 2,050,136 sharks would have been caught during 1988 in the North Pacific. Roughly, 1'950,000 of these would be blue sharks, 7,250 oceanic whitetip sharks and about 90,000 unidentified sharks (table 2.20). These estimates are conservative when compared to the estimates of Nakano and Watanabe for blue sharks taken in the same area. Assuming an average weight of 20 kg/shark regardless of species, the estimated total bycatch would be of 41,000 t. Catches by country are difficult to estimate since it is impossible to split by country the effort estimates of Nakano and Watanabe. A very crude estimate based on the proportions of effort indicates that about 7.35% of the catches could pertain to South Korea and the rest to Japan.

There is no information on the discards of sharks from these fisheries, or the percentage of sharks released alive. Given the partitioning of FAO statistical areas in the Pacific Ocean, it is almost impossible to assign catches of elasmobranchs reported by Japan and Korea, to that part of the North Pacific above 20°N. However, even the total reported "various elasmobranchs" catches for FAO areas 61, 67 and 77 by Japan and Korea (5,537 t and 2,927 t respectively), the estimated discard would be of about 22,000 t.

Overview of longline fisheries.

High-seas longline fisheries for tunas and billfishes are a very large source of bycatches and discards of elasmobranchs in the world. Despite the uncertainty surrounding the different estimations, it is evident that the amount of effort exerted by longlining fleets (worldwide total of about 750 million hooks/yr) is the main reason for the high bycatch estimates. The best estimates allowed by the quality of baseline information, are presented in table 2.21. The grand total of elasmobranchs caught incidentally by longlining fleets in all the high-seas of the world is estimated at almost 8.3 million fishes or an astonishing 232,425 t. This represents almost a third of the total world catch of elasmobranchs reported in commercial fisheries by FAO in 1991.

The relative importance of shark bycatches in number of fishes is almost equally distributed in the longline fisheries of the world. The fisheries of the Atlantic, Indian, Tropical and South Pacific and North Pacific Oceans each account for about 2 million elasmobranchs. However, the total weight of bycatches in the Atlantic and Indian Oceans is estimated to be almost double that for the whole Pacific Ocean (table 2.21). This is due to the different mean weights used in the calculations and does not necessarily represent a real difference in weight of the catches. Specifically, the mean weight of 20 kg/shark used for the Pacific Ocean fisheries is very conservative.

The levels of discard and survival of released sharks are also uncertain. The accumulated estimates of discards from the longline fisheries treated above amount to a total of 204,347 t. It is unknown what proportion of these discards survives the gear but some reports indicate it could be as high as 66 % (Berkeley and Campos 1988). Nevertheless, there are numerous accounts of finning in the literature (Mejuto 1985, Stevens 1992, Nakano 1993) and given the increase in shark fin prices in the late 1980's it would be naive to think that a great part of the sharks released will actually survive. Further research is needed to clarify the extent of the real kills of sharks in longline fisheries.

The bycatches of blue sharks in longline fisheries is very large. Although species breakdown was not always possible for reasons explained above, an approximation can be done for those areas where only total shark bycatch was estimated if we conservatively consider 40% of the total bycatch to be blue sharks. Adding this figure to the numbers already estimated of blue sharks caught in those fisheries where species breakdown was possible, we find a total estimate of 4'075,162 blue sharks caught incidentally in the high-seas longline fisheries of the world.

Table 2.20 Estimated bycatch of sharks in the North Pacific by the longline fleets of Japan and Korea, based on the results of Strasburg (1958).

| Species | Strasburg's data* Numbers caught (87,595 hooks) | Hook rate (sharks/1000 hooks) | Estimated Catch in 1988 Total numbers | weight (t) ** |
|---|---|---|---|---|
| *Prionace glauca* | 1,616 | 18.45 | 1,953,432 | 39,069 |
| *Carcharhinus longimanus* | 6 | 0.07 | 7,253 | 145 |
| Various sharks | 74 | 0.84 | 89,452 | 1,789 |
| Totals | 1,696 | 19.36 | 2,050,136 | 41,003 |

* for cruises north of 21 N
** assuming 20 kg/shark

Table 2.21 Selected estimates of shark bycatches in high seas longline fisheries.

| Area | Number of individuals | Total catch in tonnes |
|---|---|---|
| Atlantic Ocean | 2,305,940 | 76,318 |
| Indian Ocean | 1,931,574 | 75,180 |
| South/Central Pacific Ocean | 1,996,350 | 39,927 |
| North Pacific (above 20N) | 2,050,135 | 41,000 |
| Total | 8,283,999 | 232,425 |

The order of magnitude of present estimates seems to be in general agreement with previous assessments. As a reference point, Taniuchi (1990) estimates a total shark catch from Japanese longliners of 90,000 t using a ratio of shark-catch/target-species catch for the tuna and billfish longline fishery. The worldwide elasmobranch bycatch estimated here for Japanese longliners is of 115,441 t. There is however a good degree of uncertainty introduced by the quality of the baseline information available for the present estimations. For example, the hooking rates used here range between 7.04-20.45 sharks/1000 hooks, whereas Taniuchi (1990) plots hooking rates for Japanese research longliners ranging between 2.7 and 8 sharks/1000 hooks. Only reliable effort figures and updated hooking rates by region will allow to make reasonably better estimates of the bycatches.

In contrast with driftnet fisheries, there is no observer programme for any of the high-seas longline fisheries in the world. This accounts for much of the uncertainty surrounding the estimates of non-target species caught in longline fisheries. It is worth noting that most of the international tuna organizations and the governments of longline fishing nations mandating logbook reports from longliner fleets, still do not require or enforce the reporting of bycatches of sharks or other elasmobranchs. Some of these organizations are taking steps to change this situation (ICCAT 1993b, Nakano 1993). This should help reduce the uncertainty about the real levels of bycatches and discards in the near future. Considering the common underreporting of elasmobranchs in longliner logbooks (Stevens 1992, Nakano 1993), observer programmes are undoubtedly the best way to tackle this crucial information problem.

#### 2.2.3.3 Purse Seine Fisheries.

Most of the large-scale purse seine fisheries for tuna occur in tropical waters where the relatively shallow schooling behaviour of some tuna species makes them easy to catch. The main species targeted by purse seine are the yellowfin (*Thunnus albacares*) and skipjack (*Katsuwonus pelamis*) tunas, although some other species are also captured in smaller quantities. Purse seines are very unselective gears, so that other fish and non-fish species (e.g. marine mammals) commonly associated with the tuna schools are frequently caught in the fishing operation.

Major tuna purse seine fisheries are fairly localised. They are centred in four main areas (fig. 2.41): the Eastern Tropical Pacific (ETP) off Mexico down to the north of South America; the Western Central Pacific (WCP) from the Philippines and Papua-New Guinea to Polynesia; the western Indian Ocean (WIO) around the Seychelles and the Eastern Tropical Atlantic (ETA) around the Gulf of Guinea. Additionally, there is some tuna purse seining off Venezuela in the Western Atlantic Ocean.

The ETP purse seining fishery began during the 1950's. It expanded largely in the 1960's and 1970's but declined temporarily in the early 1980's. Presently, about 280,000 t of yellowfin tuna are caught by purse seiners in this region (Sakagawa and Kleiber 1992). The fleet used to be dominated by USA vessels but since the early 1980's many of these were reallocated to the WCP fishery and now Mexican vessels are dominant.

Tuna purse seining was developed in the WCP by Japanese and USA vessels in the 1970's. In contrast to the ETP, effort here is largely directed towards skipjack tuna although yellowfin tuna is also caught in large amounts. The Japanese fleet fishes mainly log-associated schools whereas USA boats concentrate on free-swimming schools (Sakagawa and Kleiber 1992). Korean and Taiwanese purse seiners have joined the fishery since the late 1970's (Suzuki 1988). These four countries comprise the major part of the fleet, with smaller numbers of vessels operating under the flags of Australia, Indonesia, Philippines, Marshall Islands, New Zealand, Solomon Islands and the former U.S.S.R. The total catch of tunas by purse seiners in the WCP during 1989 was 576,204 t, at least 73% was skipjack tuna (Lawson 1991).

In the WIO, the fishery was started by a Mauritius-Japan purse seiner in 1979, followed by French vessels in 1980. By 1984 all the French fleet moved from the Atlantic to the WIO, together with part of the Spanish fleet. During 1989, France, Spain, Panama, Japan, Mauritius, U.S.S.R. and Cayman Island had 49 purse seiners in this fishery, with the first two countries dominating the fleet. The total catches in the WIO were some 220,000 t of tunas (yellowfin and skipjack mainly, but also some bigeye) in 1989 (IPTP 1990).

Purse seine fishing for tunas in the tropical Atlantic was initiated by French fishermen in the early 1960's in the coastal waters of the Gulf of Guinea. African coastal states, Spanish, and

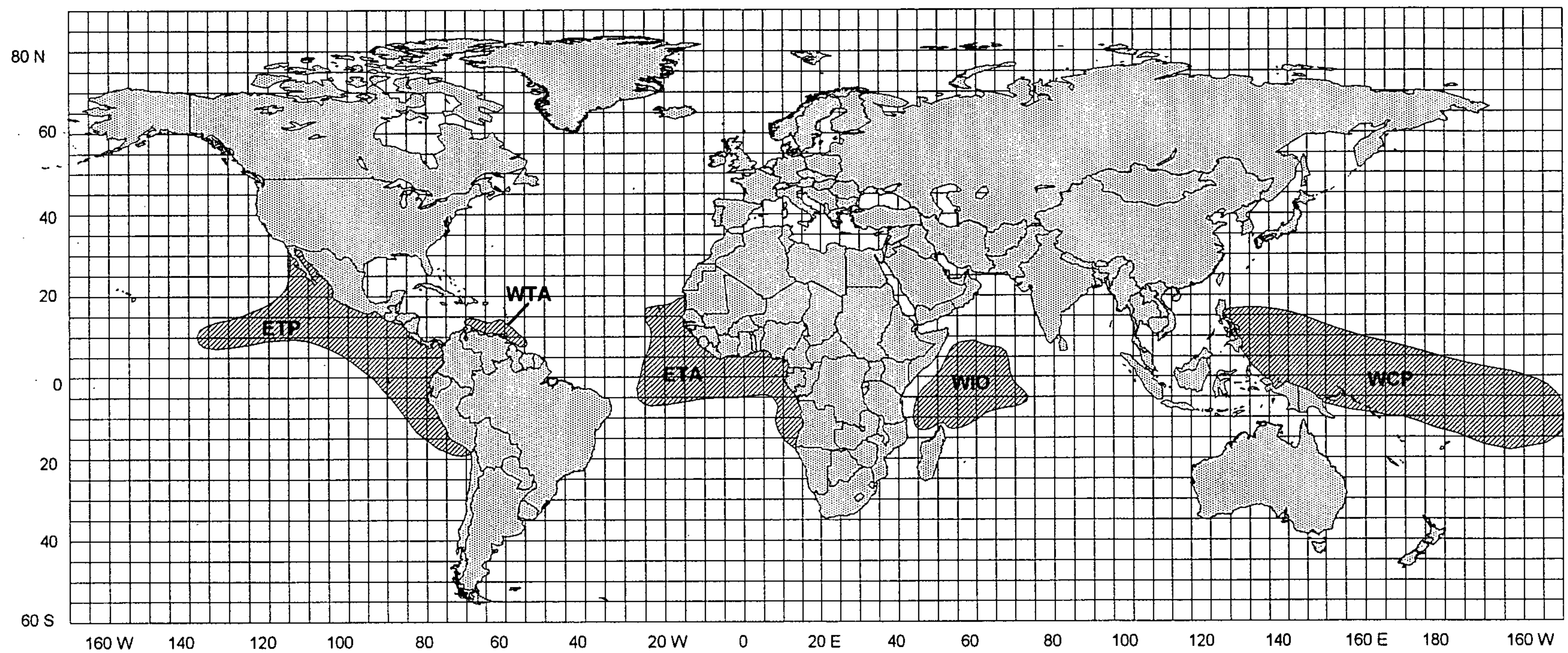



Figure 2.41 Major areas of Tuna purse seine fisheries in the world.

USA fleets joined later. The fishery expanded to offshore areas at the end of the 1970's and it currently accounts for more than 80% of the yellowfin tuna catches of the Atlantic (Suzuki 1988). At present the majority of the catches are taken by the Spanish and French-Ivorian-Senegalese-Moroccan (FISM) fleets, with small amounts contributed by Venezuelan, U.S.S.R. and Japanese boats. Yellowfin and skipjack are the main targeted species, with minor bycatches of bigeye tuna. A total of 167,800 t of tunas were caught by purse seiners in the tropical Atlantic during 1989; at least 90% of this came from the eastern Atlantic (ICCAT 1991a, 1991b, 1992).

Information on the elasmobranch bycatches in purse seine tuna fisheries is appallingly scarce. Even though the presence of sharks in the purse seine catches is documented at least since the mid-1960's, it has received very poor attention in the literature. Bane (1966) reports several large silky and other sharks, and manta rays taken in a purse seine set off Gabon in 1961. Bane also mentions *C. limbatus, C. plumbeus* and *Rhizoprionodon acutus* are associated with tuna schools in the area. Yoshimura and Kawasaki (1985) report 183 silky sharks caught by purse seine fishing in the WCP and length frequency histograms indicate that most silky sharks were between 60 and 170 cm TL with a mode of 110-130 cm TL. For the Indian Ocean, LaBlache and Karpinski (1988) based on observer programme data, report bycatch rates of 6% of the total catch for purse seiners that had bycatches. They consider various teleosts, including undersized and damaged tuna, as part of the bycatch. Oceanic whitetip sharks were the second major part (12%) of the bycatch.

The most detailed account of sharks associated with tuna schools is provided by Au (1991) for the ETP. According to Au, sharks form associations with yellowfin tunas that could be of an opportunistic predator-scavenger nature. The association of sharks with yellowfin tunas, measured as percentage of sets having sharks, is 40% for log-associated tuna schools, 6-21% for free schools and to 13% for dolphin-associated schools. Apparently, these associations are limited by the swimming speed of sharks. The silky shark was the most common elasmobranch in the bycatches with up to 500 individuals caught per set. Various other carcharhinids, oceanic whitetip, sphyrnid, alopid, lamnid, blue and whale sharks were also caught together with several batoids and mobulids. Unfortunately, Au's report fails to provide any useful measure of the numbers of sharks caught by purse seine fisheries (i.e. catch of sharks per unit of effort, or proportion of elasmobranch catch to tuna catch). Although he lists average numbers of each shark species per set, those values represent only purse seine sets that caught that particular species. Without any reference in his paper to the total numbers or weights of sharks in the full sample, his results are of very limited use for the purpose of estimating shark bycatch rates.

The total bycatches of elasmobranchs in purse seine fisheries can be estimated in a very rough way using the information on shark and tuna catch provided by Lablache and Karpinski (1988). From their data, shark catch is calculated to be 0.51 % of the total tuna catch kept by the purse seiners. Using this proportion and the reported tuna catches listed in each fishery, the estimated total catch of sharks in purse seine fisheries during 1989 is of 6,345 t: 856 t in the tropical Atlantic, 1,122 t in the Western Indian Ocean, 2,939 t in the Western Central Pacific and 1,428 t in the Eastern Tropical Pacific.

The above estimates assume that the amount of sharks caught is directly proportional to tuna catches. Although this is a very wild assumption, it is worthwhile giving it a thought. Purse seining is essentially an active fishing mode that takes advantage of the schooling behaviour of fish. Sharks are known to gather around tuna schools, especially log-associated schools (Au 1991) so that shark catch will always depend on tuna catch. Contrary to the case of passive gears (longline, driftnet), shark catches in purse seine do not occur without tuna catches (fishermen never set the gear if there are no tuna schools). Accordingly, it seems more appropriate to relate the shark catch to the tuna catch rather than to an effort variable (usually days at sea for purse seiners) as in the case of passive gears where there is competition for a hook or space in the gill net. The main weakness of the present estimations, is to base the calculations in a single (and poorly representative) account of the proportion of shark catch to tuna catch in purse seine operations, and the extrapolation of Western Indian Ocean data to other geographical areas. These assessments will improve only when more information on catch rates becomes available and as our understanding of the seasonal and spatial changes in the shark-tuna associations increases.

There are no records of the condition of the elasmobranchs caught in tuna purse seine operations, but it is very likely that all of them die either by suffocation or crushing, when they do not manage to bite their way out of the nets. Although Bane (1966) reports that the shark catches were sold at shore in the Gulf of Guinea, this seems to be an exception under an experimental fishing campaign. Most of the shark catches in commercial tuna purse seine fisheries are probably discarded. However, this cannot be confirmed from the available information.

2.2.3.4 Other miscellaneous fisheries.

The preceding sections treated those fisheries responsible for the largest bycatches and discards of elasmobranchs (mainly sharks) on a global scale. However, there are other fisheries which incidentally take elasmobranchs and are worth mentioning.

Pole and line fisheries for tunas take some shark bycatches while fishing tuna schools (Anderson and Teshima 1990). Unfortunately, almost nothing is known about the catch rates. Bane (1966) mentions sharks taken by "tuna clippers...at the surface on live bait", which suggests pole and line fishing: 131 sharks were taken at six stations by this method. It is possible, due to the global scale of pole and line fisheries for tunas that their bycatch of sharks could be significant, perhaps in the order of magnitude of the bycatch from purse seiners. On the other hand, pole and line gear may avoid the capture of sharks and survival of discards could be high. Both factors would minimise the impact of the pole and line fishery on sharks.

The orange roughy (*Hoplostethus atlanticus)* fishery of New Zealand takes deep water squaloid sharks and other elasmobranchs in their bottom trawl nets. Although there are no estimates of catch rates or proportion of the catches of these sharks for the commercial trawlers, there is some information from research vessels. At least 21 elasmobranchs (11 selachians, 4 batoids and 6 holochephalans) have been identified in deep water trawl surveys around New Zealand (Robertson et al. 1984). There are eight squaloid sharks that would have commercial importance, of which *Deania calcea* is the most abundant in the North Island, *Etmopterus baxteri* in the South Island and *Centroscymnus spp.* in the central areas. Surveys in the North Island indicate that *Deania calcea* constitutes a larger part of the total catches than either of the most important commercial species, the orange roughy and the hoki (*Macruronus novaezelandiae)*, (Clark and King 1989). Although catch rates in commercial trawling operations should be smaller than in research cruises due to more targeted fishing, it is possible that the bycatches of elasmobranchs constitute between 10 and 50% of the orange roughy catches. According to recent FAO statistics, the orange roughy catches in New Zealand waters were of around 44,000 t/yr during 1984-1989. The total bycatch of squaloid sharks could therefore be between 4,400 and 22,000 t/yr in this fishery. King and Clark (1987) estimate the MSY for these shark stocks as 2,250 t/yr. Evidently, the current catches exceed by far the estimated MSY. Most of the catches are thrown overboard as there is no market for them, although small quantities are used for fishmeal and liver oil extraction. Given the depth from which these sharks are brought up (600-1,200 m) and the type of gear employed, all catches are probably dead when returned to the sea.

The impact of this level of bycatch on the local stocks of deep-sea sharks is poorly understood. One can postulate that it is likely to be highly damaging and unlikely to lead to sustainable exploitation. However, this is difficult to verify when there is virtually no information about the abundance, biology and population dynamics of these deepwater species. More research is needed on the actual levels of bycatch, survival of discards and about the population dynamics of these deep shark populations.

2.2.3.5 Overview.

Estimating the total bycatch and discard of elasmobranchs in high-seas fisheries worldwide is difficult because neither of these processes are adequately documented. The rates of discard, finning and survival are virtually unknown. There are large uncertainties about the catch rates and effort levels by region. We should expect qualitative and quantitative variations in the elasmobranch bycatches within each ocean due to areal and seasonal changes in availability of the different species. Unfortunately, these sources of variability could not be taken into account in the present work with the available information. For these reasons, the results obtained here should be used with caution as they are only indicative of the order of magnitude of the bycatches.

Present results indicate that a very large amount of elasmobranchs are caught incidentally in the high-seas fisheries of the world. The estimated grand total of elasmobranch bycatch at the end of the 1980's, is around 260,000 and 300,000 t or 11.6-12.7 million fish per year. Most of these catches are sharks, predominantly blue sharks.

Longline fisheries are the most important source of shark kills in the high-seas, mainly because of the magnitude of their effort. They contribute about 80% of the estimated total elasmobranch bycatch in weight and about 70% in numbers of fish. There is large uncertainty around the bycatch estimates for this type of fisheries. However, the figures are based on the best available information and they seem to compare well with the few reference points at hand.

The former high-seas driftnet fisheries ranked second for their contribution to the total elasmobranch bycatches. Since these fisheries were terminated worldwide at the end of 1992, they are now one less problem to worry about in terms of sea-life conservation. It would be interesting to know the fate of the vessels formerly engaged in driftnet fisheries since 1992: it is possible that this effort has been redirected to fisheries which might still impact elasmobranchs and the other species previously affected by gillnetting activities.

Discards from high-seas fisheries also appear to be very high. The figures suggest that up to 230,000-240,000 t of elasmobranchs are discarded every year in the various high-seas fisheries. Most of the discards are probably dead, almost certainly those caught in the driftnet, purse seine and orange roughy fisheries. For longline fisheries, survival depends on whether fishermen release sharks readily and unharmed. Nevertheless, common finning practices make dubious that survival is high in longline operations.

Available information on purse seine and pole-and-line tuna fisheries and the deep trawl fisheries for orange roughy make it very difficult to assess the importance of their bycatches of sharks and rays. Presently, they seem to share a minor part of the total bycatch of elasmobranchs but there is a big gap in direct information on this subject. More, basic research, is needed in this field.

There is another substantial source of bycatch and waste of sharks and rays around the world. This is the incidental catch of bottom trawling vessels fishing for shrimps and fishes in continental shelves around the world. The assessment of the elasmobranch bycatches in these fisheries is out the scope of this work primarily because of the extreme difficulty in gathering information about them, and the magnitude of the work involved. These fisheries are known to be of high impact to local populations, specially in the case of rays (see accounts of British and Thai fisheries in sections 2.2.2.2 and 2.2.2.4). Some of these catches of elasmobranchs are landed and reported under official statistics of the fishing country. However, a large proportion is just dumped at sea, and is never accounted for.

Species of elasmobranchs under pressure from high-seas fisheries.

Blue sharks are the most common elasmobranch caught incidentally in high-seas fisheries. Present estimates indicate that 6.2-6.5 million blue sharks are taken annually worldwide in these fisheries. Although this is apparently the first estimate of total catches for blue sharks in all high-seas fisheries of the world, there are a couple of partial estimates which can be used for comparison. Stevens (1992) estimates that the Japanese longline fisheries annually take a total of 433,447 blue sharks. This figure appears small compared with that estimated here. However, he considers a conservative hooking rate of only 1 shark/1000 hooks. Nakano and Watanabe (1992) estimate that all the high-seas fisheries of the North Pacific Ocean caught 5 million blue sharks during 1988. In this case, their estimate appears high against the present results. Hence, the assessment of blue shark bycatch performed here seems to be within reasonable values.

Our current knowledge prevents an assessment of the impact that the removal of 6 million blue sharks annually has on high-seas ecosystems or on the blue shark populations. There is virtually nothing known about the size of the stocks of blue sharks anywhere in the world, and the biology of most stocks is poorly understood. Nakano and Watanabe (1992) performed the only assessment known to date of the impact of high-seas fisheries in blue shark stocks. After estimating bycatches and using cohort analysis, they consider that the catch levels during the late 1980's did not have a significant impact on the populations of the North Pacific. However, Wetherall and Seki (1992) and Anonymous (1992) consider that appropriate information is lacking for an assessment of this kind. More research is badly needed both to assess the real bycatch levels in each fishery and their impacts on the different populations.

Silky sharks are probably the second most common shark bycatch, specially in longline and purse seine fisheries. As in the case of blue sharks, appropriate information is lacking to assess the impacts of the removal levels. In any case, their growth and reproduction compare poorly to those of blue sharks, i.e. silky sharks have slower growth, a later sexual maturation and are much less fecund (see Pratt and Casey [1990] for a compilation of life history parameters of sharks). Hence, they are expected to be less resilient to exploitation than blue sharks. Again, more research is needed before any conclusions can be drawn about the effects of these fisheries on silky shark populations. Local stocks of *Deania calcea, Etmopterus baxteri* and *Centroscymnus spp.* in New Zealand could be added to the list of elasmobranchs under possible threat by large-scale fisheries.

## 2.3 Discussion.

### 2.3.1 Current Situation of Elasmobranch Fisheries.

Several features were identified throughout this review. Fisheries for sharks and rays are very common throughout the world and very diverse in regard to the species taken and to the types of fishing gears and vessels used. Unfortunately, this diversity contributes to the difficulty for keeping the appropriate statistics of yield and abundance essential for the study of these fisheries. This is particularly evident in the scarcity of information available for most of the cases reviewed here. Very few countries have sufficient information about their shark and ray fisheries for assessment purposes, and in most cases the information is still dispersed. Statistics for elasmobranchs around the world need to be improved: catch should be reported by major species and species groups; the elasmobranch bycatch from high seas large-scale fisheries should also be compulsory reported. The latter could be achieved by the establishment of observer programmes for most high-seas fisheries. Additionally, more compilation and review work needs to be done on a country and regional basis to set the ground for a better appraisal of exploitation levels and in order to make an overall assessment of the status of elasmobranch stocks around the world.

Another important characteristic brought out by the review is the predominantly incidental nature of the catches of elasmobranchs. The number of fisheries which specifically and primarily target sharks or rays around the world can probably be counted with fingers. Most of the cases examined in the preceding sections indicate that the vast majority of fisheries for sharks and rays, and surely most of the world catches, are in fact the product of fisheries for other species. This makes their assessment but especially their management very difficult. Few managers will constrain economically or socially important fisheries in order to manage elasmobranchs sustainably .

The results from the analyses of yield trends in each FAO Major Fishing Area of section 2.2.2.1 suggest that an expansion of the catches could be achieved in some Areas and to a lesser extent on a global scale. Nevertheless, local stocks in several parts of the world (North Indian Ocean, North Sea, North East Atlantic) are probably overexploited and catches there are expected to decrease. However, these analyses are very rough and must be used with caution. In this context, a better index of relative production (IRP) could be developed in order to make a better "quick and dirty" assessment of the possibilities for elasmobranch exploitation in the world. A simple improvement would be to incorporate in the IRP the area of continental shelf of each Major Fishing Area in order to weight the harvest of sharks and rays taken, in a similar way in which the total surface of sea of each Area was used here.

The increasing global trend in reported shark and ray catches suggests that overall yields could be expected to continue rising as there is no sign of decline in yield. This would be misleading if interpreted uncritically. A closer look at the elasmobranchs fisheries in various countries reveals changes in the types of fisheries and species exploited. While some fisheries for elasmobranchs collapse, others are developed elsewhere. This indicates that exploitation levels are not being sustained in all cases. Almost 30 % of the major fishing countries analysed in section 2.2.1.2 show a falling trend on catches. It is very likely that the increase in world catches might have components other than an absolute increase in catch. Possible reasons for an apparent increase could be improvements in the reporting of catches and increased landings of bycatches in other fisheries.

The likelihood that elasmobranchs will be sustainably exploited in the near future is not very promising. There is in general a lack of management and research directed towards these fragile resources. This raises serious doubts about the future of shark and ray fisheries. Only 3 out of 26 major elasmobranch-fishing countries (Australia, USA and New Zealand) are known to have management and research programmes for their elasmobranch fisheries.

Surprisingly, not one of these three countries plays a leading role in worldwide elasmobranch yield. Moreover, for the few countries that do have some fisheries information this indicates apparent problems of over-exploitation for some elasmobranch stocks (e.g. shark fisheries in souther Brazil, in both coasts of the USA and in southern Australia). Unfortunately, many of the countries playing the major roles in elasmobranch fisheries worldwide have very limited or non-existent research programmes and probably no management for these resources. If this situation continues unattended, stocks will eventually be driven to such low population levels that fishing will probably cease for a very long time. A particular case that needs close monitoring is the fishery in Indonesia, which has grown incredibly quickly in the last 20 years and will probably collapse dramatically in the absence of management.

World catches of elasmobranchs are substantially higher than reflected by the different kinds of official statistics. Statistics reported to FAO amount to just below 700,000 t for 1991. The results presented here suggest that the total catch (as opposed to landings) could be closer to 1 million t, if we include the estimated catch of the People's Republic of China and the bycatch from large-scale high-seas fisheries. However this figure <u>does not take into account</u> discards from innumerable bottom trawl fisheries around the world. Recreational fisheries are also not included since there is little information available. However, there are very important recreational fisheries for elasmobranchs in specific parts of the world (e.g. USA, South Africa, Australia). Hoff and Musick (1990) estimate that the mortality of sharks in recreational fisheries of the eastern USA alone, can be more than 10,000 t/yr. The real total level of sharks, rays and chimaeras caught around the world is probably closer to 1.35 million t or more per year, twice the official statistics.

### 2.3.2 Conservation of elasmobranchs.

The bycatch of elasmobranchs in high-seas fisheries around the world seems to be a major source of concern for conservation due to the very high numbers of sharks killed. Blue sharks in particular might be facing extreme pressure in many parts of the globe because of these fisheries, but more specific studies are needed in order to address the real situation.

The possible threat of elasmobranch overexploitation from high-seas fisheries is actually only part of a complex technical interaction. There is substantial gear and catch damage caused by sharks in most of these fisheries (Taniuchi 1990, Sivasubramaniam 1963,1964, Pillai and Honma 1978, Berkeley and Campos 1988) which translates directly into economic loss for the fishing industries.

A possible way to solve this dual problem could be to install shark deterrent devices in passive fishing gears (these account for most of the elasmobranch kill). The Natal Shark Board in South Africa is currently testing a promising non-lethal electroacoustics device to protect bathers from shark attacks. Another possibility would be to design new selective fishing gear that could substantially reduce shark hooking rates. However, for the time being the only viable alternative is the implementation of suitable bycatch quotas for elasmobranchs in the high-seas fisheries of the world through international agreement, and their enforcement via observer programmes.

The concern over elasmobranch exploitation arises not only from theoretical considerations about their biological and ecological traits, but also for historical reasons. The record of fisheries for sharks and rays includes several cases of collapse and rapidly falling catch rates, reminding us of the fragility of these resources (Holden 1977). Documented accounts include the California fishery for soupfin sharks and the spiny dogfish fishery of British Columbia in the 40's, the school shark fishery of Southern Australia in the 50's, the porbeagle shark fishery in the Northwest Atlantic and the spiny dogfish fishery in the North Sea during the 60's (Anderson 1990). Although the underlying reasons for some of these collapses are partly understood (and are sometimes independent of high levels of exploitation), and despite the fact that decreasing CPUE's are a natural characteristic of fisheries development, these failures constitute a warning against careless exploitation in view of the special biological attributes of sharks and rays discussed above.

Effective protection of sharks and rays from the potential impacts of large-scale fisheries is not an impossible task. The efforts of international collaboration that regulated the catches of salmonids, marine birds and marine mammals in the North Pacific Ocean and the recent banning of all driftnet fisheries in the high-seas of the world are testimony to the reality of effective protection for marine fauna. The strong pressure that some countries are imposing on fleets that continue to take some dolphins in purse seine tuna operations are another clear example that, when the will is there, effective protection can be achieved.

The road to effective management and protection of elasmobranchs depends largely on education and awareness. This is the only way in which it will be possible to stimulate in fishermen, scientists, the public and governments the will needed to achieve real protection and management of sharks and rays. Efforts of this type have already met with some success. The South African Government has recently protected the white shark; the government of Australia forbids the killing of grey-nurse sharks and is considering protection of white sharks; California just passed legislation banning the catch of white sharks in their waters. Various recent scientific meetings have focused on the issue of elasmobranch conservation. During 1991, the international meeting "Sharks Down Under" was held in Sidney, Australia, focusing attention on the need for the conservation of elasmobranchs by hosting a Conservation Workshop as the opening event. The American Elasmobranch Society held a Symposium on Conservation of Elasmobranchs during its 1991 meeting and is presently setting up a Conservation Committee at the international level. The Species Survival Commission of the IUCN has recently formed a Shark Specialist Group. Evidently, international concern about the future of elasmobranchs and the extent of their exploitation is starting to pick up. This events suggest that it might be possible to achieve proper fisheries management for sharks and rays in the near future. However, without significant increases in funding to enable more effective research on elasmobranch, the goal of sustainable exploitation might never be reached. In addition, the conflicting demands of conservation and the socio-economic concerns of fishermen also require directed research efforts.

## 2.4 Summary and conclusions.

Elasmobranch fisheries are a traditional and common activity of little importance globally but providing important sources of hard currency, protein and employment to many local communities around the world. These fisheries are particularly important in places such as Sri Lanka, Pakistan and Australia. The type of exploitation of elasmobranch ranges from subsistence fisheries with artisanal gears and vessels, as is the case of some sail-powered boats in India, to the highly industrialised fisheries with longlines, gillnets or trawls of long-range fishing nations like Japan, Taiwan, Spain and the former Soviet Union.

There are 26 countries that can be considered major elasmobranch-fishers, that is they harvest or have recently harvested more than 10,000 t/yr of elasmobranchs. Among these, Japan, Indonesia, India, Taiwan and Pakistan have the highest average elasmobranch yields. About 30% of these 26 countries show recent falling trends in yield. The analysis of IRP's (Index of Relative Production) by FAO Major Fishing Areas suggests that further increases in exploitation of sharks and rays could be possible in the South East Pacific (Area 87), North East Pacific (Area 67) and the South East Atlantic (Area 47). However this analysis is very simplistic and needs to be taken with caution.

Although there are some cases of specific fisheries for elasmobranchs (south Australian shark fishery, fisheries for sharks in Argentina and Mexico, basking shark fisheries of Norway, etc.), the larger part of the yields of sharks and rays in the world are produced as incidental catches in other fisheries. This poses particular problems for assessment and management for several technical and economic reasons.

Official fisheries statistics do not properly reflect the amounts of sharks and rays actually harvested every year in the world's oceans. Although official figures report about 700,000 t/y of elasmobranchs caught at the end of the 80's, the actual level is at least of 1'000,000 or possibly 1'350,000 t/yr.

The bycatches of sharks in large-scale high-seas fisheries around the world is very large, amounting possibly to almost 50 % of the reported catches from commercial fisheries. The numbers of sharks caught annually in these fisheries during 1989-1991 are roughly estimated here at about 11.6-12.7 million. The longline fisheries for tunas of Japan, Korea and Taiwan account for most of these bycatches. More detailed information is needed to properly address the magnitude of this problem and its effects upon shark populations. Observer programmes need to be implemented soon for these fisheries in order to obtain reliable information about yields, discards, and the extent of finning practices.

There are serious deficiencies not only in the reporting rates but also in the handling of the reported statistics. The statistics discriminate very poorly the types of elasmobranchs caught in each fishery, and this is of particular concern because it makes more difficult appraisals of any kind. Fisheries statistics need to be improved both in coverage of the fisheries and the dissaggregation of species. This probably implies improvement of quality control in the catches to allow for development of new markets.

There is a generalised lack of research and management for shark and ray fisheries even in major elasmobranch-fishing countries. Very few fisheries are under specific management and this is a reason for further concern over their sustainability. Management of elasmobranch fisheries should ideally start very early in the development of the fisheries given their extreme fragility and the difficulties in reducing fishing effort once it grows beyond optimal levels.

# CHAPTER 3

# DENSITY-DEPENDENT FECUNDITY IN ELASMOBRANCHS AND ITS IMPLICATIONS IN FISHERIES MANAGEMENT: A DETERMINISTIC AGE-STRUCTURED SIMULATION MODEL.

*Representation is a compromise with chaos.*
*Bernard Berenson*

## 3.1 Introduction.

### 3.1.1 Biological characteristics of the group in relation to exploitation.

The evolution of elasmobranch reproductive systems has resulted in strategies that are in great contrast with those of most bony fishes. Teleosts are extremely fecund, and they generally produce several million eggs which are released to the environment with the hope that an uncertain number will eventually develop into recruits. In contrast, elasmobranchs have very distinct reproductive strategies comparable in many ways to mammalian reproduction: they concentrate large efforts in producing a limited quantity of offspring which are recruited to the population as soon as they are born. In comparison with teleosts, uncertainty in recruitment should be greatly reduced in elasmobranchs (Holden 1973). The resulting (inferred) close relationship between parent stock and recruitment in elasmobranchs has been considered of foremost importance for their assessment and management by those working with fisheries for this group (Holden 1968, 1974). Even though elasmobranch reproduction has often been considered a limiting factor for their sustained exploitation (Holden 1973, 1977, Hoenig and Gruber 1990, Pratt and Casey 1990), few studies have actually explored in detail the extent of such constraints.

The study of the importance of elasmobranch fecundity has also relevance in the understanding of elasmobranch population dynamics particularly in relation to density-dependent mechanisms. The analyses of different stocks of spiny dogfish *Squalus acanthias* (to date the best studied elasmobranch) have resulted in alternative hypotheses each supporting either changes in fecundity, growth, immigration or juvenile mortality, as the underlying compensatory mechanism responsible for the relative resilience of this species to exploitation.

On the one hand, Holden (1973) proposes density-dependent changes in reproduction as the compensatory mechanism for the Scottish-Norwegian stock of spiny dogfish. This is reinforced by the results of Gauld (1979) who presents some evidence of changes in fecundity for the same stock. Notably, Gauld's data only demonstrate changes in the number of ova produced per female, but fail to provide evidence for increased number of <u>embryos</u> per female. In contrast to Holden's view, Woods et al. (1979) suggest that density-dependent fecundity is not enough to provide effective compensation in the relatively resilient British Columbian spiny dogfish stock, and they favour a reduction in natural mortality as the compensatory mechanism. Fahy (1989) analyses and excludes both hypotheses in addition to compensatory growth, and proposes that rapid immigration/re-colonization from un-depleted nearby stocks might be the answer to the recovery of some stocks of spiny dogfish.

There is additional data from other elasmobranch groups raising doubts over the value of fecundity increases. Using an analytical model, Brander (1981) demonstrates that fecundity changes have minimal or no effect in the capability of adult stocks of Irish Sea skates to support heavy fishing, and that juvenile survival is the key factor. Walker (1992), on building a simulation model for the gummy shark (*Mustelus antarcticus*) fishery of southern Australia reports not finding any evidence of changes in fecundity as a result of exploitation.

3.1.2 Definition of the problem.

A simulation approach is a viable alternative to analyse the dynamics of a population in a fast and cheap way. It offers the advantage of being easily adapted to fit different scenarios and could therefore be used to test various hypotheses. In this sense, it could also provide important information for future research-planning and management.

During the present study I built an age-structured model of an elasmobranch population with the following aims a) to evaluate the potential effectiveness of changes in elasmobranch fecundity as a compensatory response to increased fishing mortality, and b) to assess the implications of the results, and the potential use of the model, for fisheries management.

## 3.2 Construction of the model.

### 3.2.1 Model-building considerations.

A model is any representation or abstraction of a given system or process. The type and complexity of models depends on the field of research and the particular problem to be analysed. In terms of Holling's (1978) classification, problems in population modelling generally lay in the area of low quality/quantity of relevant data. For the particular case of elasmobranch populations, we must consider in addition to this, the poor understanding of the biological and fisheries processes involved (e.g. the explicit form of the stock-recruitment relation, the underlying density-dependence mechanism, or the actual levels of exploitation of a particular stock).

The complexity of a model (understood as the number of variables included) is not always directly related to its performance and usefulness. Ludwig and Walters (1985, 1989) give examples of simple models outperforming more complex ones, whilst Hilborn and Walters (1992) support the existence of an "optimal" model size, which will be specific for each case study. It seems pointless to construct a large and extremely detailed model that includes every single variable that is likely to affect the system. Very frequently, the uncertainty surrounding the estimation of some of these variables, only reduces the ability of the model to produce useful information. The secret in successful model building resides in locating the most important variables and ignoring the ones that do not add significantly to the performance of the system. On the other hand, we must also consider the level of resolution needed from the model in relation to management or informational needs. Starfield and Bleloch (1986) recommend a compromise of simplification for model building, and in their own words: "choosing the appropriate level (of resolution) is thus a pragmatic compromise between the complexity ... and the need to solve a problem with limited data and in a reasonable amount of time...". This is precisely the philosophy adopted throughout this chapter.

### 3.2.2 Biological considerations.

#### 3.2.2.1 Early life natural mortality.

One of the most common assumptions in fisheries science is that the natural mortality of a stock remains constant through time and across age classes (Caddy 1991). However, not surprisingly, recent research indicates that natural mortality is not constant across ages in wild shark populations (Manire & Gruber 1991). Sharks are a very healthy group of organism and to date there are no accounts of mortality due to disease or parasitism in wild shark populations. Although most sharks species have very few predators (mainly other sharks) and their natural mortality levels are thought to be very low, shark pups are exposed to relatively high natural mortalities mainly due to predation from larger shark species or to cannibalism by the adult part of the population. As the young sharks approach a certain size threshold (approx. 1 m TL is suggested by Branstetter (1990)), their natural mortality decreases because they are able to deter predators or escape them thanks to increased swimming speed.

Although it would seem more adequate to use a size-specific natural mortality during this study, an age-specific mortality schedule has been chosen in order to simplify the model. Effectively, this implies assuming that there are no changes in individual growth rate, except when this is explicitly modelled.

Natural mortality for sharks has been estimated to range between 0.048 and 0.2 (Holden 1968, Pauly 1978, Grant et al. 1979, Bonfil 1990). In the present model, the natural mortality coefficient for early juveniles is set to a value 2-3 times larger than that chosen for adults. Mortality is reduced with increasing age, then becomes nearly constant for most preadult and adult age classes.

#### 3.2.2.2 Life history functional relationships.

The great majority of the commercially important shark species are viviparous. In the batoids, only the skates are oviparous species of significance to fisheries. Therefore, I will centre my analysis on viviparous species.

There are several reports of positive correlation between mother size and litter size (the number of embryos in a litter) for viviparous sharks. This relationship between fecundity and mother size may either assume a linear (Olsen 1954, Parsons 1983, Simpfendorfer 1992) or exponential (Walker 1984, Lenanton et al. 1990) form. In the present model, I assume a linear relationship because this appears to be the most common case in the sharks so far studied.

Size at birth is also positively correlated to mother size in some sharks (Olsen 1984, Hanchet 1988, Peres & Vooren 1991). However, producing larger newborn sharks might mean smaller litter sizes due to space limitations in the female body cavity, as reported for *Rhizoprionodon terraenovae* (Parsons 1983). A positive correlation between mother size and the size of the newborns is advantageous because it can mean decreased mortality by predation for larger pups. Smaller newborns will not only have higher natural mortality than larger newborns but assuming equal growth rates, they will remain for longer under the size threshold that allows them to escape or deter predators. Once this is considered, the most important consequence of a correlation between mother size and the size of pups at birth is that the average natural mortality of the first age class in a given year will be dependent on the size structure of the female parent stock. However, the incorporation of this relationship between mother size and early natural mortality of the pups is not attempted in the present model.

### 3.2.3 General characteristics of the model.

As a first simplification, the model considers only the female part of the population. This is a common practice in demographic models (Krebs 1978) and is based on the assumption that male availability is never limiting for reproduction. For the model, fertilization takes place at the start of the year and gestation time is exactly one year. Hence, newborns are recruited to the population at the beginning of every year. A gestation time of one year is the most common among sharks (see table 2 of Pratt & Casey 1990), specially among carcharhinids which are usually the most important commercial species in the tropics.

A deterministic approach is used in this study in order to keep the model tractable, and because of computational restrictions (the whole model is implemented in Quattro Pro 3.0 and run on a 386/25 PC computer). In addition, stochastic processes in fish populations are usually associated with recruitment, and viviparous sharks are likely to show small recruitment variation as compared to bony fishes, so that for the present purpose these can be ignored. In Chapter 4, a shark population model incorporating stochastic variability in recruitment is implemented on another platform (BASIC).

The simulations are broken into three phases. In the first, the different parameters of the model are entered (natural mortality and fecundity arrays, number of age classes, etc.) and a new population is allowed to grow to its equilibrium size. During the second phase, fishing is introduced via an array of age-specific fishing mortalities. After a 100 year period of constant fishing, the size of the remaining stock is compared to the virgin population. In the third phase, fishing mortality, fecundity, or any other initial conditions are changed to simulate different type of stock or different exploitation and density-dependent response scenarios. The model is run again under these new conditions and the results of each run are stored for later analysis. For simplicity, the change in fecundity of the exploited population is simulated as an immediate response process by which fecundity is increased as soon as exploitation begins.

3.2.4 Formulation.

The simulated population is a closed system with no emigration/immigration in which the number of fish $N$ in a cohort of age $a+1$ during the year $t+1$ are calculated as:

$$N_{t+1,a+1} = N_{t,a} e^{-M_a} \qquad 3.1$$

where $N_{t,a}$ is the number of sharks of age $a$ at the beginning of year $t$, and $M_a$ is the natural mortality of fish aged $a$ until the beginning of age $a+1$. The total number of fish in the population in year $t+1$ is given by:

$$N_{t+1} = [\sum_{a=1}^{A_{\max}} N_{t,a} e^{-M_a}] + R_{t+1} \quad 3.2$$

where $A_{max}$ denotes the maximum age after which all sharks die and $R$ the number of recruits born into the population at the beginning of the year (i.e. number of sharks aged 0, assuming single pulse birth).

The number of recruits in a year $t+1$ is determined by the number of individuals in each mature age class during the previous year ($t$) and surviving to the beginning of $t+1$, times the age specific fecundity:

$$R_{t+1} = \sum_{a=A_{mat}}^{A_{\max}} N_{t,a} e^{-M_a} \varphi_a \quad 3.3$$

where $\varphi_a$ is average fecundity in number of female newborns per mother per year, and $A_{mat}$ is the age of first maturity of the population.

Total biomass is calculated using the equation:

$$B_t = \sum_{a=0}^{A_{\max}} N_{t,a} \bar{W}_a \quad 3.4$$

where $B$ is total biomass, and $W_a$ is average weight at age.

The population growth according to this model has an undesirable feature, it follows an exponential behaviour which is not realistic. There are two easy ways of including density dependence in order to simulate a more 'realistic' logistic growth. One is via a stock-recruitment relationship, and another is to incorporate density-dependent natural mortality; the second approach is chosen here. The usage of a stock-recruitment relationship was discarded because it obscures the effects of one of the variables we want to change in the model, i.e. density-dependent fecundity. Additionally, stock-recruitment relationships have not been documented yet for wild elasmobranchs. On the other hand, the inclusion of density-dependent natural mortality is a plausible alternative, often recommended for theoretical models of fish populations (Ricker 1940; Kesteven 1947).

The following equation is used to adjust the initial values of age specific natural mortality (from here onwards called the baseline natural mortality):

$$M_{t,a}=M_a(\alpha+pN_{t-1}) \qquad 3.5$$

where $M_{t,a}$ denotes the natural mortality for sharks of age $a$ in year $t$, and $\alpha$, $p$ are constants. This makes the change in $M$ a linear function of population numbers which is the simplest form of expressing density dependence (Beverton and Holt 1957).

For the second phase of the model, fishing mortality is introduced into the equation for numbers-at-age:

$$N_{t+1,a}=N_{t,a}e^{[M_a(\alpha+pN_{t-1})+F_aF\%]} \qquad 3.6$$

here $F_a$ is the age specific fishing mortality coefficient, and $F\%$ a factor allowing the baseline fishing mortality value to be changed interactively for each run of the model in order to simulate different scenarios of fishing intensity. In analogy with fishing mortality changes, fecundity is increased by multiplying the baseline fecundity values by a factor depending on the desired percentage increase.

3.2.5 Initial parameters.

The baseline input values for the model during the baseline run, are listed below. These parameter values were chosen to approximate the known features of sharks' biology, and simulating a fishery that would harvest all ages but with some level of selectivity in which very young (small) or very old (large) sharks are less susceptible of being captured.

$A_{max}$ = 25 years

$A_{mat}$ = 10 years

Fecundity array = [0,0,0,0,0,0,0,0,0,0,0.75,1,1.5,2,2.5,3,3.5,4,4.5,5,5,5,5,6,6,6]

M (baseline) = 0.4 for age 0

M (baseline) = 0.3 for age 1

M (baseline) = 0.2 for age 2

M (baseline) = 0.15 for ages 3-4

M (baseline) = 0.12 for ages 5-6

M (baseline) = 0.09 for ages 7-25

F = 0.08 for ages 0-6

F = 0.12 for ages 7-20

F = 0.04 for ages 21-25

The values for $p$ and $\alpha$ for the density-dependent natural mortality function were "hand-tuned" to allow a population growth that would resemble a typical logistic curve while keeping the upper and lower limits of $M_a$ within reasonable bounds. The parameter values were

$$p=0.6, \text{ and } \alpha=1 \times 10^{-6}$$

The actual $M$ values used by the calculations in the model at each population size are illustrated in figure 3.1.

All simulations were initialised by 'stocking' 10,000 new recruits (age 0) per year, until the year when the first cohort reached maturity. From then onwards, the population was left to grow alone.

3.2.6 Sensitivity analysis.

A sensitivity analysis is the easiest way to investigate the behaviour of the model under different scenarios, and to understand which are the most important variables affecting the output. For this purpose, both small and large changes in the input variables are analysed when possible.

The choice of a dimensionless variable as the output is frequently recommended in order to simplify the sensitivity analysis in nonlinear models (Starfield & Bleloch 1986). For the present study, such a strategy is also very useful for comparing the results from different runs. Two new variables are introduced: *N%* is the ratio between total population numbers after 100 years of fishing and the virgin population numbers; similarly, *B%* expresses the same concept in terms of total biomass. These two variables provide an easy way to compare the effect of changes in fecundity on the long term sustainability of a fishery under different scenarios of fishing intensity. Furthermore, comparing *B%* to *N%* also gives some insight about possible changes in population size structure as a result of exploitation.

The sensitivity analysis is structured in the following way: runs 1-7 test the separate effects of different changes in fishing intensity and increases in fecundity, as compared to baseline results; runs 8-9 explore the effects of reductions in the age of first sexual maturation, whereas runs 10-12 look at selective fishing patterns; run 13 considers the effect of having a flat age distribution at the virgin population size; runs 14-15 include changes in the natural mortality assumptions of the model; and finally runs 16-18 are designed to observe the effect of the number of age classes of the population (i.e. longevity).

## 3.3 Results.

### 3.3.1 Baseline run.

The population resulting from the baseline run of the model attains an equilibrium size of about 1.05 million sharks in 79 years (fig. 3.2). This population displays the stable (stationary) age structure (fig. 3.3) expected for a virgin population free of disturbances. Fishing begins at this point under a constant regime using the baseline values of *F*. The following 100 years when the fishery take place, show a slightly oscillating decay of the stock (fig. 3.4).

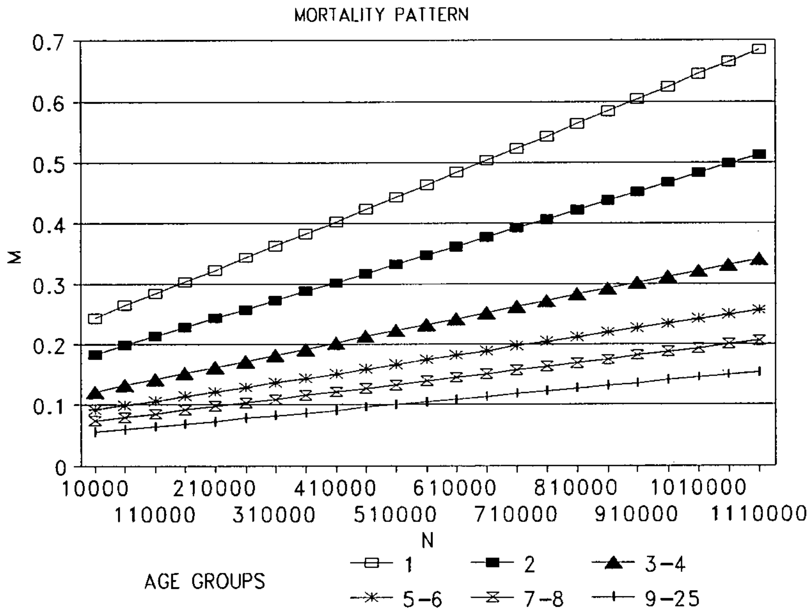



Figure 3.1 Values of the natural mortality coefficient used by the density dependent function of the model.

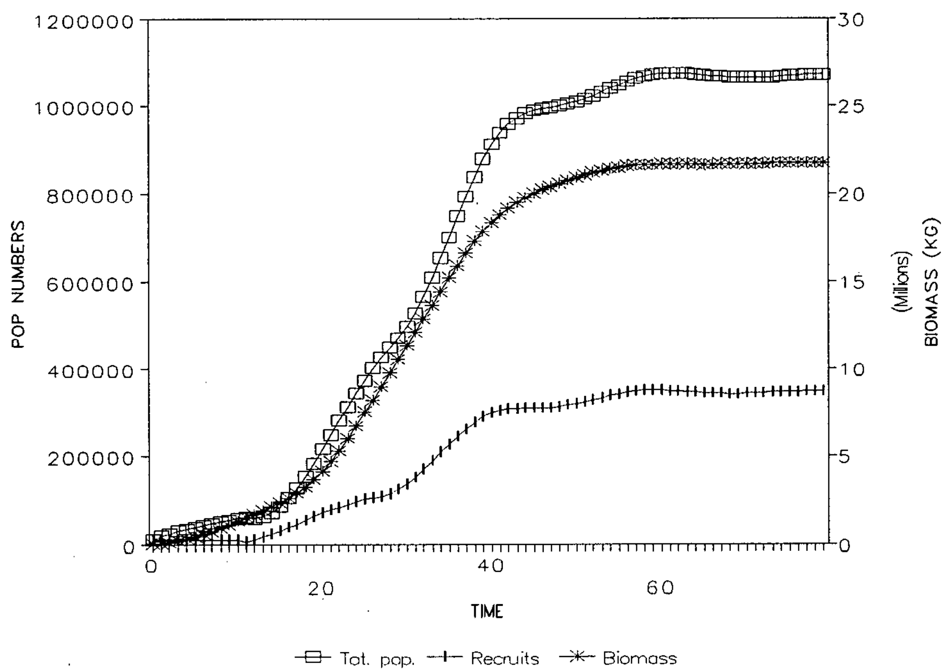



Figure 3.2 Growth of the simulated elasmobranch population according to parameters defined in the text.

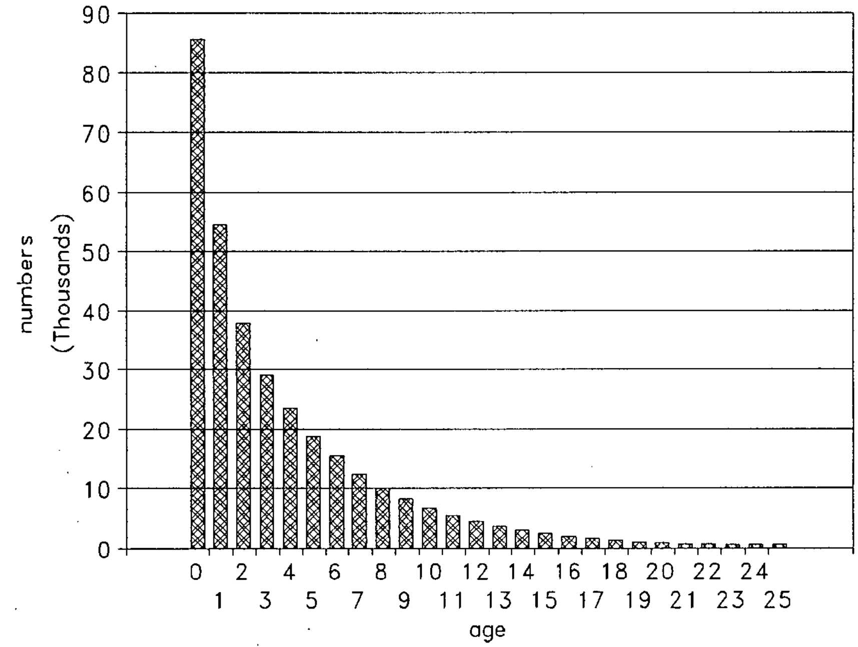



Figure 3.3 Stationary (stable) structure of the simulated population at the asymptotic size (virgin population), defined by the mortality and natality schedules.

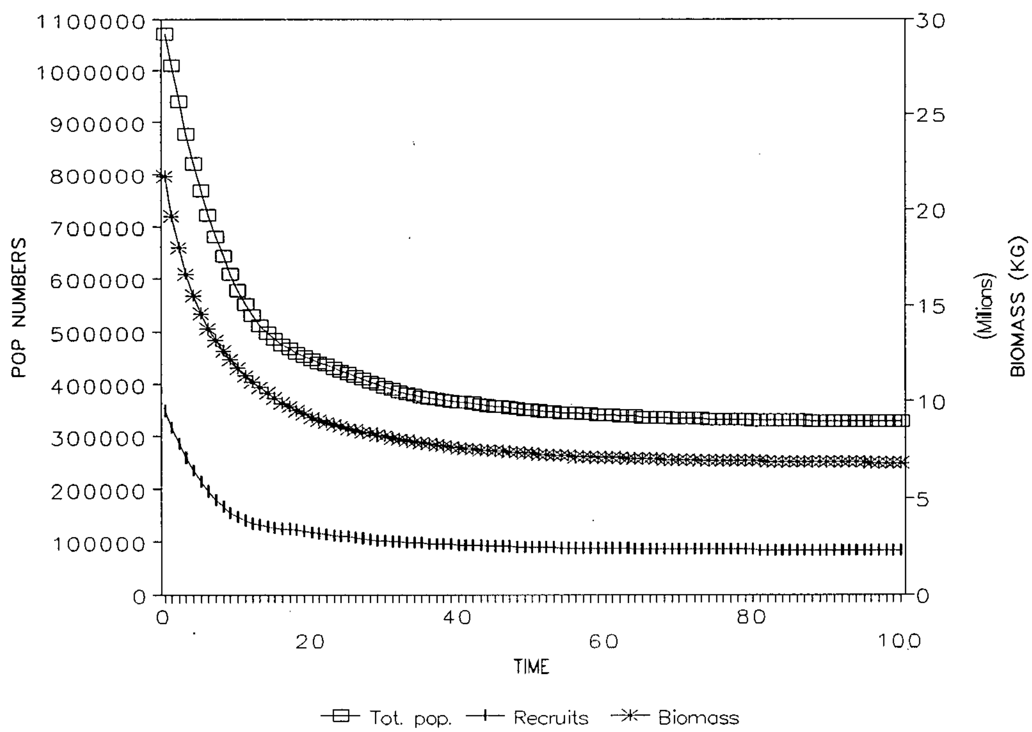



Figure 3.4 Decay of the simulated population under the baseline fishing mortality pattern.

3.3.2 Sensitivity analysis.

The results of the sensitivity analysis are shown in table 3.1. The first important outcome from runs 1-7, is that the simulated population is much more sensitive to changes in fishing mortality than to increases in fecundity. A fecundity increase of 400% has less impact on *B%* than a 40% decrease in fishing mortality. Note that the effect of fecundity increases is felt much more in the population numbers than in terms of biomass.This is not surprising considering that the fecundity increase is translated directly as a gain in the number of newborn sharks, which proportionally contribute less to the bulk of the biomass. Because fishing mortality is assumed constant throughout the 100 yr period considered here, the greater part of the newborns never grows old enough to contribute substantially to the total population biomass.

The model is also more sensitive to reductions in the age of first maturity than to increases in fecundity (runs 8 and 9). A 10% decrease in $A_{mat}$ has a stronger effect than a 20% increase in fecundity. We can expect reduced age of maturity through density-dependent changes in growth. In contrast to the increases in fecundity, an increase in growth has a positive impact in the total biomass.

The effects of selective fishing indicate that protecting a few juvenile age classes by excluding them from the fishery (run 10) is more beneficial in terms of biomass and almost as beneficial in numbers, as protecting all of the adult stock (run 12). In comparison, fishing only in the middle portion of the age distribution has the less positive effect for the conservation of the stock. This probably happens because a major part of the reproductive potential of the population is eliminated by fishing the less fecund but more abundant age classes.

Run 13 shows that the long-term size of the population is insensitive to its virgin size structure. This is largely an effect of the time span considered in the present simulations. According to demographic theory, any population will eventually attain a stable age distribution, regardless of its initial age structure, if it is subject for long enough to a constant system of mortality and natality schedules (Krebs 1978). The sensitivity of the model to different assumptions about the natural mortality (runs 14 and 15) is in agreement with the

Table 3.1 Results of sensitivity analysis of the model. *B%*= proportion of virgin biomass remaining after 100yr of fishing; *N%*= proportion of virgin population remaining after 100yr of fishing. Numbers in parentheses are percentage change from values of the baseline run.

| DEPARTURES FROM BASELINE RUN | | | *B%* | *N%* |
|---|---|---|---|---|
| run # | Changes in F | Increase in fecundity | | |
| Baseline | None | None | 0.314 | 0.306 |
| 1 | -95% | 0% | 0.966 (+208%) | 0.963 (+215%) |
| 2 | -40% | 0% | 0.589 (+88%) | 0.580 (+90%) |
| 3 | +40% | 0% | 0.082 (-74%) | 0.079 (-74%) |
| 4 | +100% | 0% | 0.00045 (-99.9%) | 0.0005 (-99.8%) |
| 5 | 0% | 20% | 0.365 (+16%) | 0.379 (+24%) |
| 6 | 0% | 60% | 0.433 (+38%) | 0.497 (+62%) |
| 7 | 0% | 400% | 0.568 (+81%) | 0.974 (+218%) |
| 8 | Age of maturity reduced | 10% ($A_{mat}$= 9) | 0.376 (+20%) | 0.400 (+31%) |
| 9 | | 50% ($A_{mat}$= 5) | 0.549 (+75%) | 0.843 (+175%) |
| 10 | Selective fishing | Only ages 10-25 | 0.703 (+124%) | 0.650 (+112%) |
| 11 | | Only ages 7-16 | 0.601 (+91%) | 0.582 (+90%) |
| 12 | | Only ages 0-9 | 0.635 (+102%) | 0.667 (+118%) |
| 13 | Flat age distribution (same numbers in each age class) at equilibrium | | 0.314 (0%) | 0.306 (0%) |
| 14 | Baseline M = 0.12 for all ages | | 0.395 (+26%) | 0.393 (+28%) |
| 15 | M not density-dependent | | 0.380 (+21%) | 0.389 (+27%) |
| 16 | F = M = 0.12 for all ages; fecundity = 5 for all mature ages ( > 9yr) | | 0.465 | 0.464 |
| 17 | F, M and fecundity as above. Only 21 age classes, $A_{mat}$= 8yr | | 0.571 (+23%) ** | 0.571 (+23%) ** |
| 18 | As above, but only 16 age classes, $A_{mat}$= 6yr | | 0.672 (+45%) ** | 0.672 (+45%) ** |

** Percentage change from run number 16.

results of the changes in fishing mortality from the initial runs (1-7).

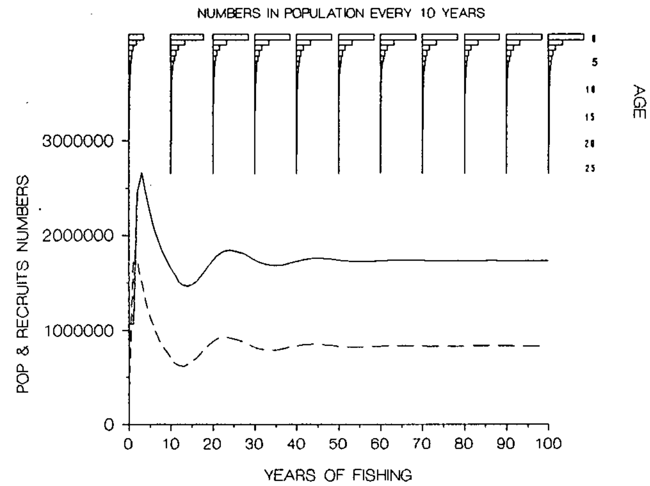
Finally, the results of the relative importance of longevity in the model indicate that shorter lived species, which have earlier ages at first maturity than longer lived ones, have a higher net reproductive rate, and are therefore better suited to sustain exploitation (runs 16-18).

Summarising the previous results, over the range of values analysed the model is more sensitive to variations in mortality and age of first maturity than to increases in fecundity. If all the assumptions of the model hold, this suggests that for shark fisheries, research efforts should concentrate on getting good estimates and controls of mortality, rather than on detailed investigations of the fecundity of the stocks.

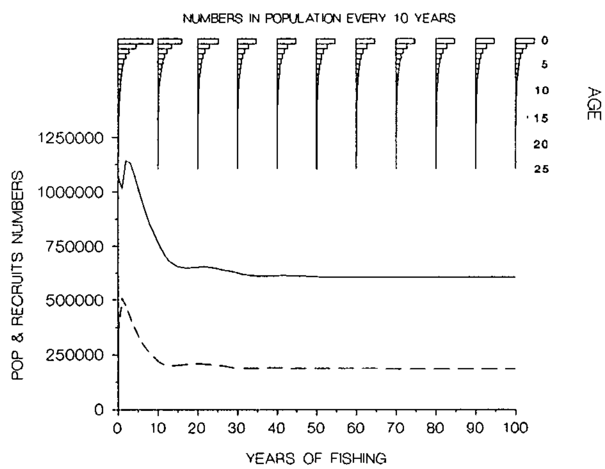
3.3.3 The value of fecundity increases.

The behaviour of the model was further investigated in detail with an extensive analysis which included a total of 160 runs under a wide range of combinations between fishing mortality and theoretical increases in fecundity. Figure 3.5 shows three examples of the dynamics of the model under some of these scenarios. The age structure of the population is sampled every 10 years and is shown at the top of each graph. An increase of 400% the baseline fecundity combined with very light fishing (0.05 times the baseline level) causes the population to increase and attain a new equilibrium (fig. 3.5a). However, changes of 1.6 times the baseline fecundity produce decreased equilibrium population sizes when fishing mortality increases to 0.9 times the baseline value (fig. 3.5b). After 70 years of heavy exploitation (F-tripled), the older age classes have disappeared from the population (fig. 3.5c). It is worth noting that in the two last cases above, the increase in fecundity has an impact on recruitment and on the size of the population during the first few years, but as time goes by this effect is more or less diluted as a consequence of heavy fishing.

The interaction between fishing mortality and fecundity is better illustrated through diagrams such as figure 3.6. This shows a three-dimensional surface plot of population trajectories for 100yr, over a wide range of fishing mortalities when fecundity has increased 100%. The x-y plane shows the corresponding contour diagrams with isolines of population size. The 3D surface of population numbers is steeper than that of biomass and has a rising peak at




a)

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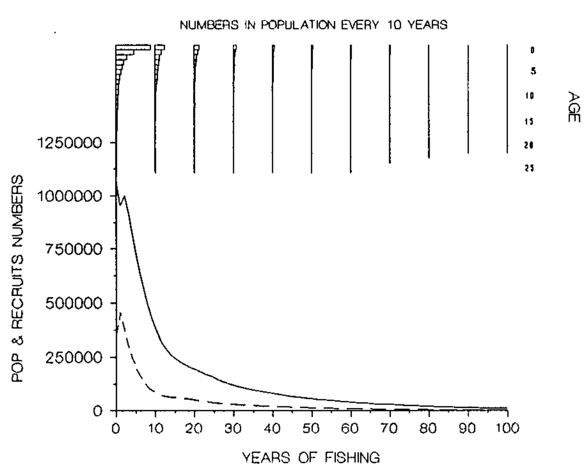



c)

Figure 3.5 Total numbers (___) and recruitment (---) trends of the simulated shark population. The top histograms show the structure of the population every 10 years. a) F=0.05 times baseline, fecundity=5 times baseline; b) F=0.9 times baseline, fecundity=1.6 times baseline; c) F=3 times baseline, fecundity=1.6 times baseline.

low fishing intensities never observed in the biomass surface. This happens because when fecundity increases, the young age classes account proportionately much more in terms of numbers than in biomass. The contour diagrams shown here could have other applications. Based on them, it is possible to find easily the combination of fishing mortality that would allow to sustain a population at a desired level in the long term, if we know the increase in fecundity we can expect as a result of exploitation. This could be a rough guide used either for purposes of conservation and management of a resource, or for the culling of shark populations.

In a similar way, we can look at the effect of a whole range of fecundity increases for a fixed value of fishing mortality using the same type of diagrams as above. In figure 3.7 population abundance is plotted against different increases in fecundity when fishing mortality is 1.8 times the baseline value. This figure illustrates very well the limited long-term value of changes in fecundity when fishing mortality is high. Although there is a considerable benefit in terms of population numbers during the first 10-15 years (purely contributed by very young age classes), fishing mortality rapidly smooths down the differences, and after 100 years of fishing there is relatively little difference between the population sizes from the different fecundity increases. The greater sensitivity of the model to fishing mortality than to fecundity increases is also illustrated when comparing the flatness of figure 3.7 against the steepness of figure 3.6a.

The terminal long-term effects (after 100 yr of fishing) of the different scenarios considered so far, are summarised in figure 3.8. At the baseline level of fishing mortality, only increases in fecundity of 4 to 5 times would allow the population to remain close to its virgin size. At this same level of fishing and in the absence of any increase in fecundity, the population would be reduced to about 30% the virgin size. If we consider 100% as the likely biological limit of an increase in fecundity in a real shark population (fecundity factor of 2 in the figure), we can see that the population would be almost wiped out just by an increase in fishing mortality of the same magnitude (F factor of 2).

Finally, figure 3.9 illustrates how changes in fecundity and fishing mortality affect *N%* much more than they affect *B%*. In this case the axes of the diagram are switched to observe more clearly the different effects of increased fecundity in population numbers and biomass.

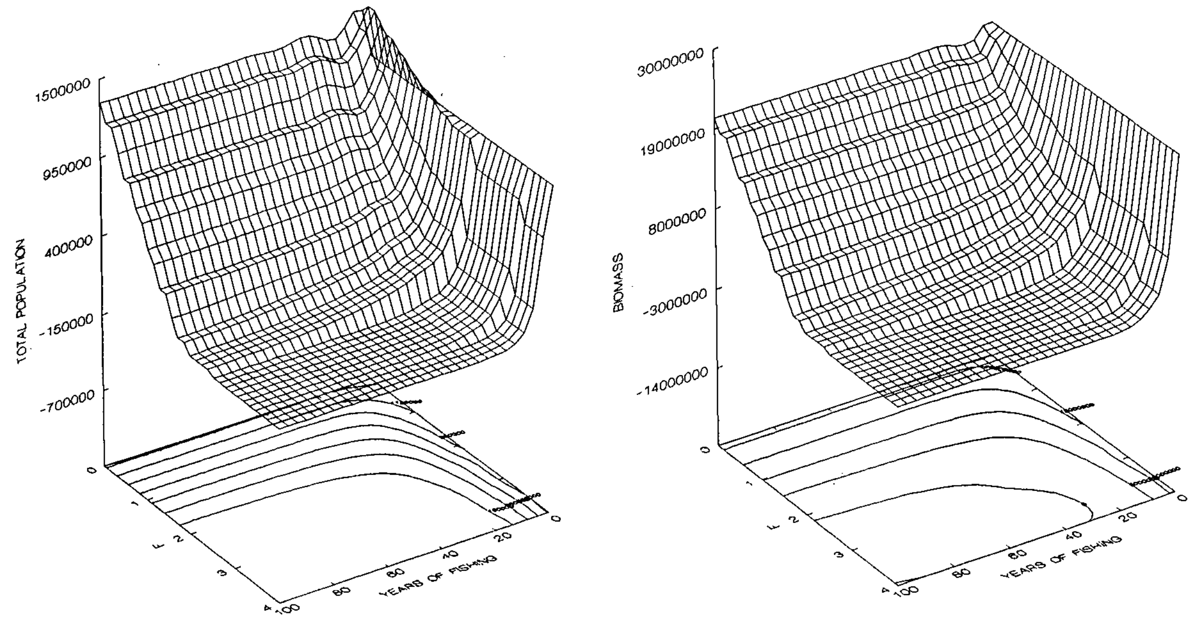



Figure 3.6 Response surfaces of population numbers (left) and biomass (right) to different fishing regimes during 100 years, when fecundity is increased 100% from baseline values. Values of F are multipliers of the baseline fishing mortality. Note the steepness of both surfaces as F decreases, indicating the sensitivity of the model to changes in F. The axes in the figures have been switched to offer the best view of the surfaces.

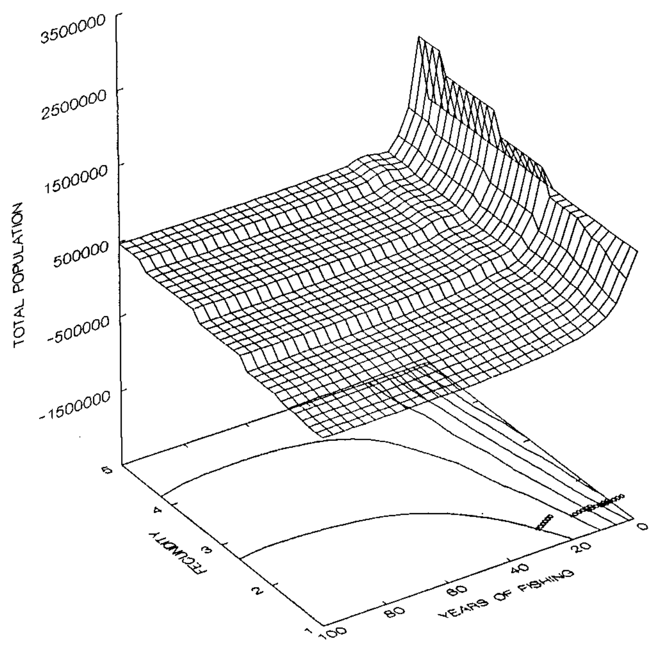



Figure 3.7 Response surface of population numbers to changes in fecundity during a 100 years' period when F is 1.8 times the baseline value. Compare the flatness of the surface with that of figure 6. The axes in the figure have been switched to offer the best view of the surface.

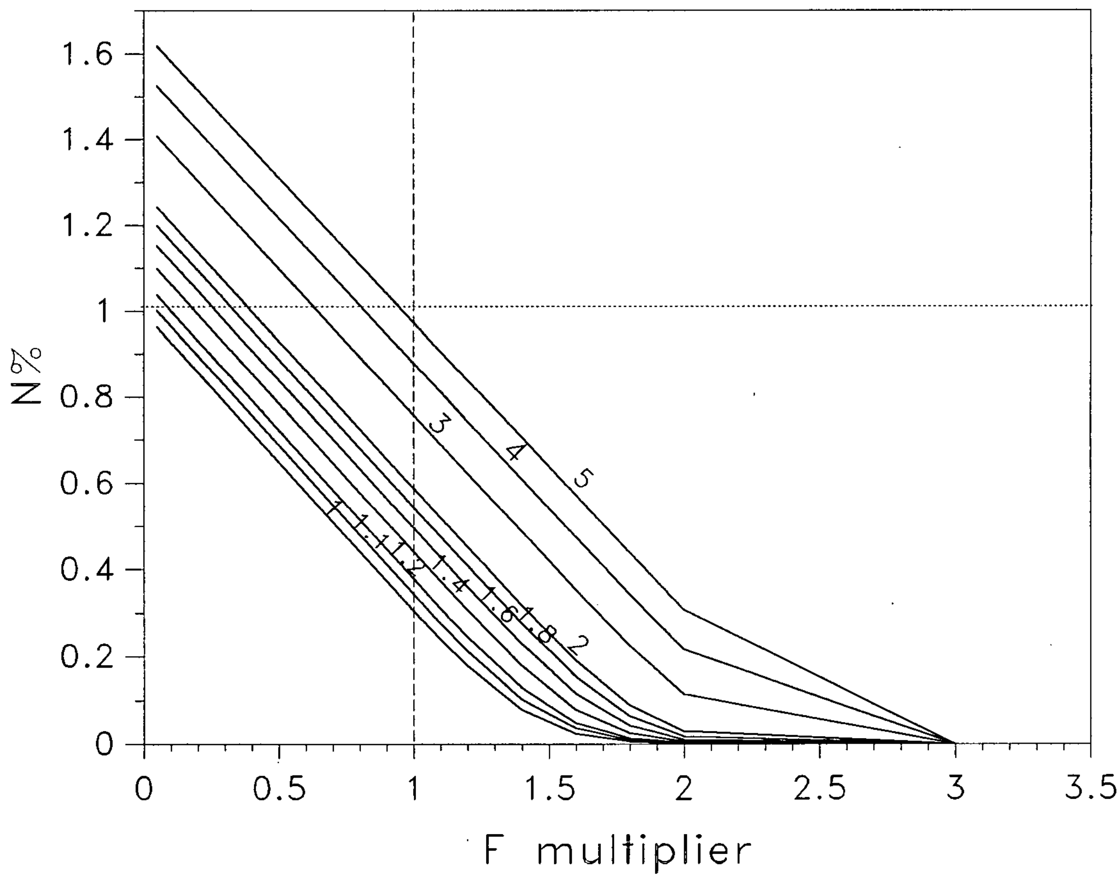



Figure 3.8 Proportion of the virgin population after 100 yr of fishing (*N%*) against fishing intensity (expressed as mulipliers), for different fecundity increase multipliers (numerals on each line). ( ---- baseline fishing mortality; .... virgin population size in numbers).

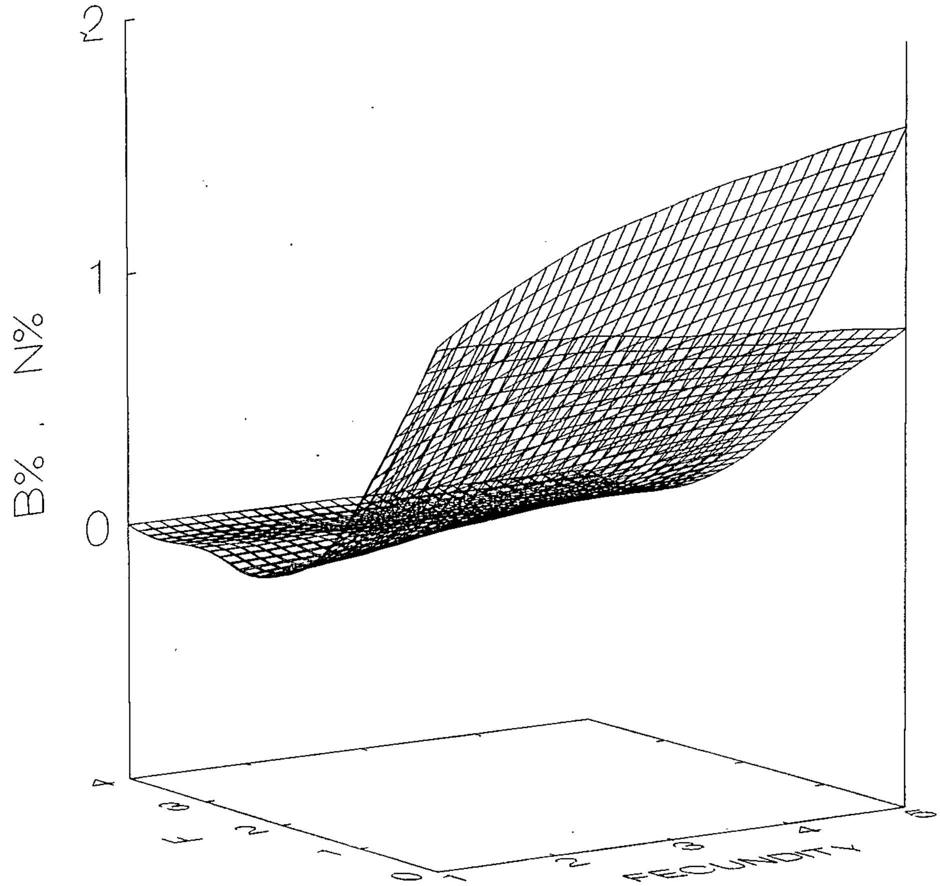



Figure 3.9 Response surfaces of *N%* (upper) and *B%* (lower) as a function of fecundity increases and **F** values (both plotted as multipliers). Y axis reversed in order to facilitate view.

### 3.3.3 A tropical paradigm with management applications.

In this section the model is used to analyse a real tropical shark fishery and to explore the possible management applications of the model.

The data comes from the silky shark (*Carcharhinus falciformis*) fishery of Yucatan. This species grows at least to 23 years of age and females attain maturity at around 12 years. The sex ratio at birth is very close to 1:1, birth takes place during a well defined 2-3 month period, gestation time is approximately 12 months, and maximum fecundity is around 14 embryos per female (Bonfil 1990). There is no length-fecundity relationship described for this species. Therefore, values were simulated by allocating a fecundity of 2 for the first time mothers (age 12) and increasing this by 2 with each year of age until a fecundity of 12 was reached. Fecundity is maintained in this level for most females with the exception of the last 2 age groups which have a fecundity of 14. By halving the above mentioned figures, only female embryos are considered in the actual modelling. The entire reproductive cycle is here assumed to take 1 year, although some studies suggest that females might have a protracted cycle with a year of rest between pregnancies (Branstetter 1987). The growth in weight shown in figure 3.10 is used for the calculation of biomass.

The fishery of Yucatan is composed of two fleets. A small scale directed fishery with gillnets capturing large adult sharks, and a large scale red grouper fishery using hook and line that catches large quantities of young silky sharks incidentally (Bonfil et al. 1990). There are some estimates of natural and total mortality for this stock (Bonfil 1990) based on Pauly's formula (Pauly 1980), and catch curves constructed through the ELEFAN programme, respectively. From these, age specific values of both parameters were 'guesstimated' for the different portions of the population. The natural mortality value of 0.2 derived by Bonfil (1990) was thought to be representative of most part of the population. However, considering the high natural mortality of young large tropical sharks (Manire & Gruber 1991), baseline natural mortality values chosen for age classes 0-2 were of 0.5, 0.4, and 0.3 respectively. For ages 3-6, a value of 0.25 was adopted.

The $\alpha$ and $p$ parameters of the density-dependent mortality relationship were hand-tuned as described in the previous section. The values used for the silky shark were of $2.5 \times 10^{-7}$

and 0.55 respectively. The actual natural mortality values used by the model are shown in figure 3.11.

Total mortality values estimated from the catch curves were used to subtract the baseline natural mortality estimates and obtain the corresponding estimates of fishing mortality. The latter were included in the model as baseline fishing mortality values and are thought to reflect well the relative magnitude of fishing mortality for the different parts of the population in this fishery. These values are 0.05 for age 0, 0.08 for age 1, 0.15 for ages 2-8, 0.05 for ages 9-16, and 0.23 for the remaining age groups.

The model is used to explore four different scenarios with management alternatives in mind. The first choice (scenario 1) maintains the ***status quo***. Scenario 2 considers protecting the juvenile portion of the stock by totally banning the bycatch of sharks in the hook and line fishery, thus effectively reducing the fishing mortality of ages 0-9 to zero. Scenario 3 reduces by half the fishing mortality of the preadult and adult sharks caught by the directed gillnet fishery, perhaps by imposing a TAC. Hence, mortality for ages 18-23 is reduced to 0.11. Finally, scenario 4 is a total ban of the directed fishery with gillnets which reduces to zero the fishing mortality of the age classes considered in scenario 3.

Figure 3.12 shows the population trajectory for the next 100 years under the four scenarios. If fishing is kept at the present level (scenario 1), the population will decline by 50% in 10 years time and will almost disappear in 100 years (fig. 3.12a). If the stock is to be conserved, protecting the juvenile portion of the population (scenario 2) is by far the best option for management. This measure allows the gillnet fishery to be maintained while keeping the stock very close to the virgin population level (fig. 3.12b). This result is not surprising. Bonfil (1990) reports a situation of growth overfishing in the silky shark fishery of Yucatan. The management alternative of scenario 2 is effectively equivalent to the well known effect of increasing the age of recruitment in a fishery suffering growth overfishing (in the context of the Beverton & Holt type of models), which is exemplified graphically in figures 1-5 of Beddington & Cook (1983). In contrast, a reduction of 50% or a total banning of the gillnet fishery will not produce significant benefits towards the long term conservation of the stock. Actually, for the long-term size of the population, these measures are almost as bad as maintaining the ***status quo***.

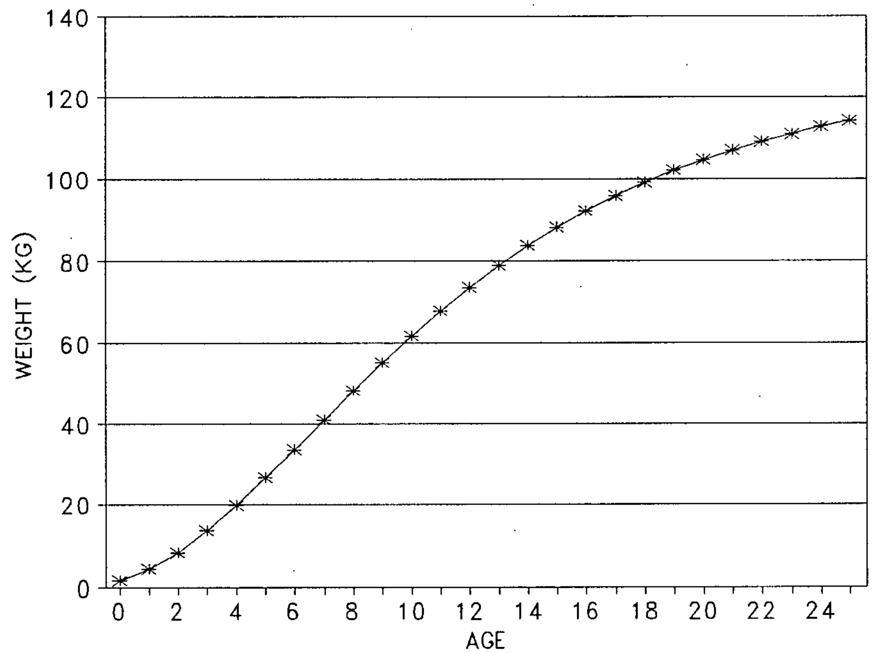



Figure 3.10 Growth in weight of *Carcharhinus falciformis* from Yucatan, used for the calculation of population biomass.

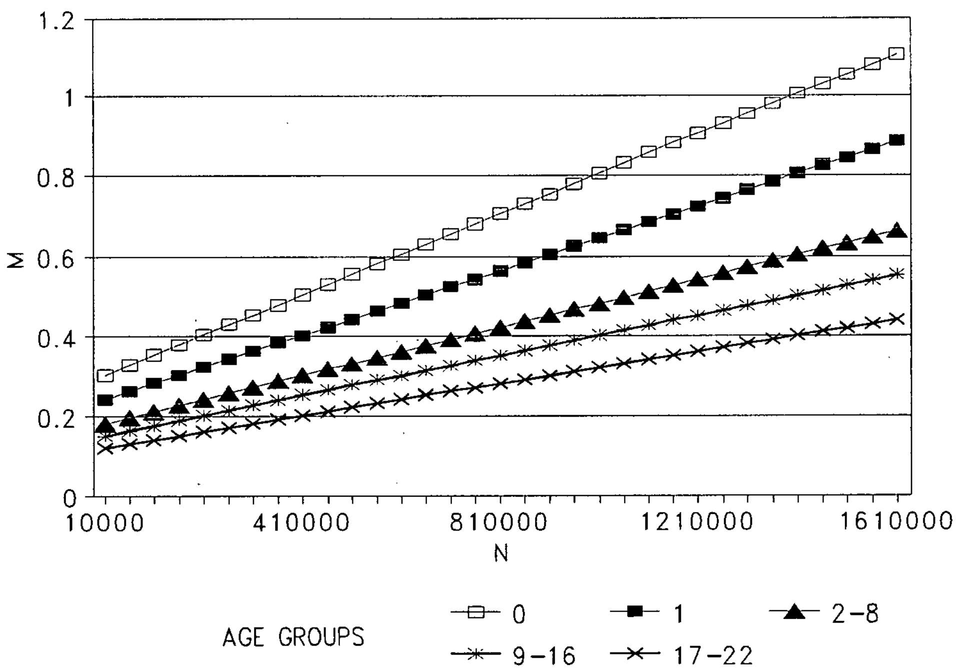
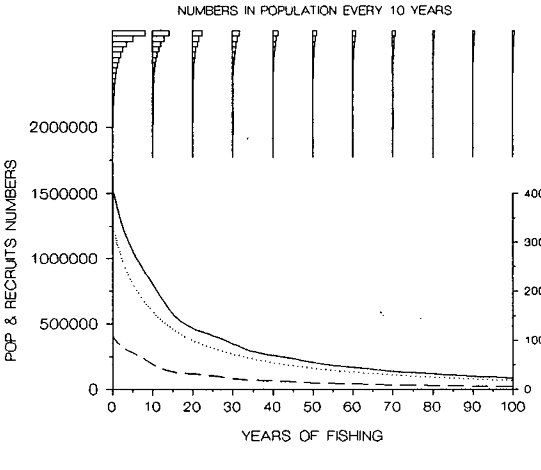



Figure 3.11 Age-specific density-dependent mortality coefficients used for the simulation of the silky shark (*Carcharhinus falciformis*).

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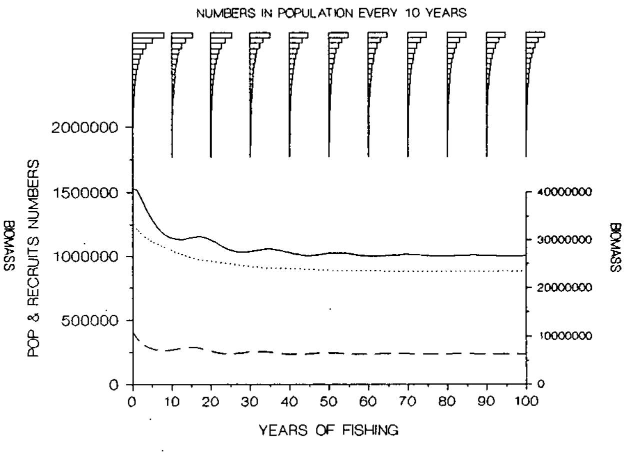


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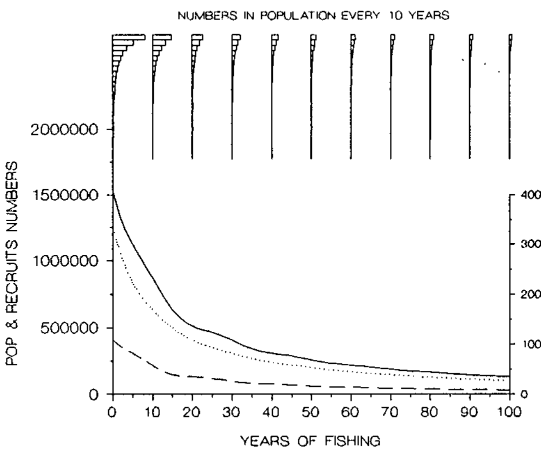


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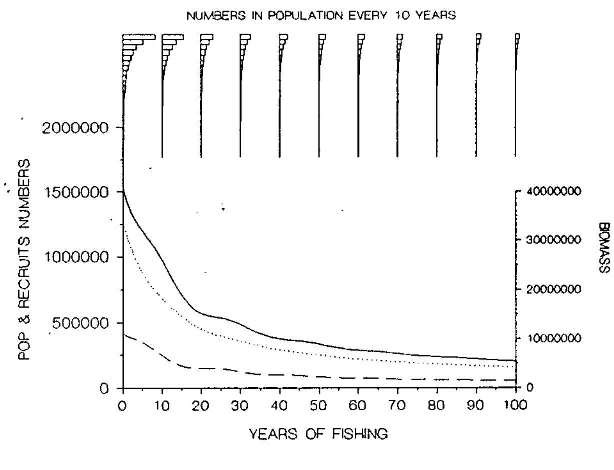



d)

Figure 3.12. Forecasted evolution of the silky shark fishery of Yucatan under 4 different management scenarios: a) no change in estimated fishing mortality; b) a total ban of the bycatch of juveniles in the red grouper hook & line fishery; c) a reduction of 50% in the fishing mortality from the gillnet fishery for adults; d) a total ban of the gillnet fishery for adults. (___ tot. numbers; --- recruits; .... tot. biomass).

Although the present results are subject to uncertainty in the various estimates of mortality, the conclusion derived from the model about the general behaviour of this fishery should be relatively robust in the sense that even if the individual age specific values of *M* might not be accurate, the general pattern of fishing mortality for the different segments of the population is probably good enough to warrant consistency of the present results.

The four management scenarios are also analysed under different increases in fishing mortality and fecundity. Three possible population responses are considered in terms of fecundity increase: no change, an increase of 40%, and an increase of 100%. For each population response, fishing mortality is either left as in the estimated ***status quo*** or it is doubled. The four management scenarios are then considered for each of these six combinations of *F* and fecundity. The results of this analysis indicate that if the population could actually respond with a fecundity increase of 100% under the ***status quo*** fishing mortality, management scenario 4 would maintain about a 50% of the virgin stock numbers and about 30% of its biomass (figure 3.13). In all cases, management scenario 2 is still the best option for the conservation of the stock. Furthermore, this is the only management scenario allowing for fishing mortality to be doubled without the stock being virtually driven to extinction, independently of the level of fecundity of the population.

## 3.4 Discussion.

### 3.4.1 Simulation results and documented changes in fecundity.

While the present study considers fecundity increases of up to 400% as a possible compensatory mechanism, the only documented case of apparent changes in fecundity of an elasmobranch population (Gauld 1979) suggests fecundity increases of only between 22% and 74% for different age groups for the Scottish-Norwegian stock of spiny dogfish (*Squalus acanthias)*. Independently of the significance of Gauld's findings which do not seem to suggest increases in the actual fertility of the population, they are far from the large fecundity increases found here to be of any value for compensation against light fishing (400% at baseline *F*; table 3.1). Gauld also studied the space limitations of thespiny dogfish body cavity (without allowing any space for food) and found that up to a 130% increase in fecundity could be afforded in terms of volume by this species. However, he considers that

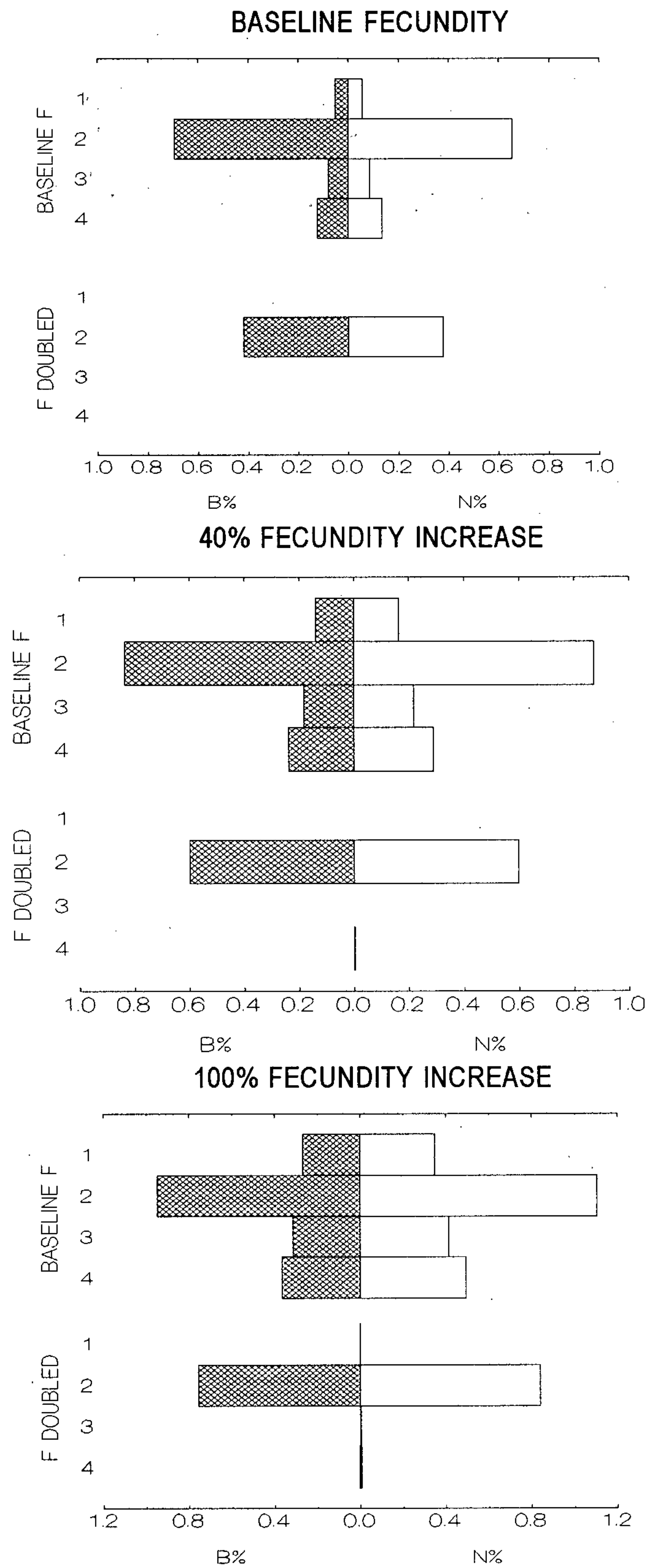



Figure 3.13 Proportion of the biomass (**B%**) and numbers (**N%**) from the virgin stock left after 100y of fishing under 4 management alternatives (numbered 1-4 in the Y axis), under initial and doubled fishing mortalities, for 3 different fecundities: Top- baseline fecundity; Centre- increase of 40% in fecundity; Bottom- increase of 100% in fecundity. (For explanation of the management alternatives see text).

physiological factors such as gas exchange between embryos and uterus will probably prevent such a high increase in fecundity.

In placental species such as the majority of the carcharhinid sharks, the amount of reserves required by the mother to nurture two times the normal quantity of embryos might pose a limit on fecundity increases. The elimination of toxins would represent an additional problem. Oviparous species such as most skates and some orectolobiformid sharks do not have space limitations for eggs production, however, fecundity increases in these species would be probably limited by the speed at which eggs can be laid and by the energy required to substantially increase production of the high quality egg yolk that feeds the embryos throughout their development.

The present analysis demonstrates that independently of the potential capability of elasmobranchs to substantially increase their fecundity, the value of such a strategy is seldom sufficient to compensate for increases in fishing mortality. Adequate fisheries management must take into consideration this biological constraint of elasmobranchs.

### 3.4.2 Trade-off between fecundity and early life natural mortality.

Given the importance of mortality during the first years of life highlighted during the present results, it is interesting to speculate on the possible secondary effects of fecundity increases. As outlined during the initial model building considerations, the time span and the magnitude of the vulnerability to predators is a function of the size of newborn sharks. If increases in fecundity encounter physiological or spatial limitations, this can be partially solved by reducing the average size of the litter at birth. However, this would reduce the success of the density-dependent fecundity strategy of the species. The net result would be that the average natural mortality of the litter, which as suggested by the present results is more important than fecundity, would be probably increased by the reduction in size at birth.

Future versions of the model, hopefully implemented on a more efficient platform, could incorporate the relationship between natural mortality of newborns and their size at birth. This could be achieved by calculating a weighted average coefficient of natural mortality for pups during a few of their early years of life, based on the size frequency distribution in the litters. The frequency distribution of sizes at birth would in turn be a function of the density of the litter and the structure of the parent stock.

### 3.4.3 Compensatory mechanisms in elasmobranch populations.

The present study has provided evidence that increased fecundity alone is very unlikely to be the factor accounting for the relative resilience of some elasmobranch stocks. Unreasonably large increases in fecundity are effective to counteract for the losses, only under very light fishing mortalities. The Scottish-Norwegian stock of spiny dogfish, which has been proposed as a case of increased fecundity due to exploitation is known to be under heavy fishing since a long time ago (Aasen 1963, Holden 1968), and it is doubtful that changes in fecundity as those reported by Gauld (1979) could explain its apparent resilience.

The findings presented here provide further support to the conclusions of Brander (1981): mortality of the young age classes is more important for the capability of a stock to withstand fishing, than the fecundity of the species. My study further supports the results of Woods et al. (1979), regarding the need of unreasonably high increases in fecundity in order to compensate for fishing in the British Columbian spiny dogfish stock. It is more probable that the underlying mechanism involved in the eventual success of an elasmobranch population to withstand a fishery, if any, is a combination of responses in which not only fecundity increases up to a reasonable limit, but also growth speeds up and reduces the age at first maturity. This would probably be complemented with density-dependent natural mortality of the stock, specially for the early ages. In highly mobile species, the immigration of individuals from nearby stocks could also contribute to compensate for fishing mortality.

### 3.4.4 Considerations for fisheries management and research.

The present model highlights the need to obtain more accurate estimates of both natural and fishing mortality in order to be able to make better simulations of an elasmobranch stock. The model also suggests that budgeting great amounts of effort and money to very detailed studies of fecundity and the reproductive cycle of the species would be a wrong allocation of research resources.

Although the present model is a useful tool for a rough evaluation of the outcome of very general management alternatives, the adequate management of an elasmobranch population through the utilization of a model like this will require major improvements. First of all, the natural uncertainty of the population dynamics and the fishing process should be incorporated by doing a fully stochastic model. A module linking effort with fishing mortality would be the best way to integrate in the model variables susceptible of direct measurement in a real fishery system. Another important addition for management considerations would be the inclusion of explicit estimates of the catch, to give an idea of the socio-economic impact of the management alternatives in the short and the long term. Unfortunately, including all these improvements would require the estimation of additional parameters and it would build on the complexity of the model, and this is beyond the scope of the present study.

If the above suggestions are eventually incorporated, the model could then be used to better explore the outcome of different regulation measures depending on the priorities of the managers. For example, in the case of the silky sharks of Yucatan, the aim could be to maintain the population at not less than 50% of the virgin stock. By banning the bycatch of juveniles from the hook and line fishery and without expecting any changes in fecundity, the management objective could be achieved even by increasing the catches of the directed gillnet fishery for large sharks.

The usage of this type of model for short term forecasts is not recommended. Model approximations in the short term are subject to oscillating behaviour, as shown during the first 20 to 30 years of fishing in some runs of the model. As the time span is extended, these oscillations become less important and virtually disappear, without serious effects on the long-term output considered here (*N%, B%*). Furthermore, the forecasts of this type of models for the first few decades should be very sensitive to input parameters.

3.4.5 From model to reality.

There are several assumptions that still need to be validated or modified for the model to be more realistic (although here we must remember that realistic models are not always better models). First, the existence of a density-dependent natural mortality relationship has not been conclusively demonstrated yet in any elasmobranch and will probably remain without substantial proof for many years. In addition, the shape of this density-dependent mechanism could take other forms rather than the linear function used here (Walker 1992). Very likely, an increase in the fecundity of a viviparous species would have to be regulated through a true density-dependent mechanism and would involve a time lag which could be considerable for species with late first sexual maturation, and not as in the present model in which fecundity increase is an automatic switch that turns on as soon as the fishing boats appear on scene.

The explicit representation of a stock-recruitment relationship could add a new dimension to the model. This would be an alternative for building a "realistic" population that grows in a logistic fashion, without using a density-dependent natural mortality function as it was done here. It must be noted, however, that a stock-recruitment relationship is in fact a very specific case of density-dependent natural mortality in which only the mortality of the pre-recruits is considered.

# CHAPTER 4

# A MONTE CARLO ANALYSIS OF FISHERY MODELS FOR SHARKS.

## 4.1 Introduction.

### 4.1.1 Problems for shark fisheries assessment and management.

Elasmobranchs, particularly sharks, are believed to be incapable of sustaining intensive fisheries (see Chapters 1, 3). Under this circumstance it would be very useful to find methods to balance between the exploitation of sharks and rays and their conservation.

Unfortunately, the assessment and management of elasmobranch fisheries has proved extremely difficult. Perceived problems of lack of adequate fishery models have precluded the utilization of simple surplus production models needing only catch and CPUE data. On the other hand, the shortage of appropriate detailed biological/ecological data makes difficult the application of assessment procedures based on age-structured data.

The perception that surplus-production models are not appropriate for sharks and rays stems mainly from the early work on elasmobranch fisheries. While studying fisheries for the North Sea spiny dogfish *Squalus acanthias*, Holden (1977) stated that the assumptions of surplus production models do not hold regarding immediate response in the rate of population growth to changes in population abundance, and independence of the rate of natural increase from the age composition of the stock. Holden bases these conclusions mainly on the time delays caused by the typically long reproductive cycles of this species and the suspected very direct relationship between stock and recruits. Because such life history characteristics are common to most elasmobranchs, this has caused surplus-production models to be disregarded for assessing elasmobranch fisheries. In other words, Holden's thoughts have been echoed in the works of other scientists who opted for the more detailed approach offered by age-structured models (e.g. Wood et al. 1979, Walker 1992).

What is often overlooked is that there are also problems in applying these realistic age-structured models. Age-structured data is much more difficult and expensive to obtain. Furthermore, the life cycles of most species, even in terms of the basic parameters of age, growth and reproduction, have just started to be unveiled during the last 15 years, and this only in the case of a handful of stocks (see Pratt and Casey (1990) for a review). In addition, there are some relevant areas of elasmobranch population dynamics which are still largely a mystery. For example: directly derived stock-recruitment relationships have never been documented for any elasmobranch, although a very strong relationship is suspected due to the reproductive strategies of the group (Holdend 1973, Hoff 1990); the size, structure and spatial dynamics of most stocks/substocks of elasmobranchs are almost totally unknown. Inadequate knowledge of migration routes, stock delimitation and movement rates amongst them, can seriously undermine otherwise "solid" assessments and management regimes.

4.1.2 Holden's view and modern methods in fisheries science.

Considering the problems outlined above, there seem to be two main ways to approach the problems of elasmobranch fisheries assessment and management. The first is to invest millions of dollars and wait several years in order to collect sufficient age-structured data for assessment, and to gain a better understanding of elasmobranch biology that warrants confident usage of age-structured models. The second option is to find alternative simple models for elasmobranch study and management. In this sense, it is worthwhile reconsidering the views of Holden under the light of new advances in fisheries science.

A paramount obstacle for the use of **classic** surplus-production models in the 60's and part of the 70's was the equilibrium constraint. Back then, scientists were forced to assume that populations were in equilibrium at all exploitation levels (i.e. that every catch observed was sustainable) in order to simplify the process of fitting surplus-production models to data. The dangerous consequences of this assumption are well known and explicitly warned against in fishery text books (Pitcher and Hart 1982, Hilborn and Walters 1992). However, the computer revolution has helped to overcome the equilibrium constraint through non-linear optimisation routines which are available to virtually any fishery scientist in the world. The diversity of approaches this offers for fitting surplus production models has translated into a new era of popularity for the utilization of what are presently known as **dynamic** surplus production models (Punt 1988, Prager 1994, Polachek et al. 1993). Perhaps the most interesting outcome of all this is the view that most of the problems associated with successfully applying surplus production models are due to the quality of the fisheries data (Hilborn 1979), and the finding that simple surplus production fishery models can sometimes perform better than the more elaborate and biologically detailed age-structured approaches (Ludwig and Walters 1985, 1989, Punt 1991).

These challenging and relatively new points of view open the question of whether dynamic surplus production models can be successfully applied to elasmobranch fisheries. In other words, we need to find out whether good quality data and the emancipation from the equilibrium assumption warrants the use of surplus production models for elasmobranch fisheries assessment and management.

The present study aims towards advancing our understanding of elasmobranch fisheries assessment and management methods by answering three questions: can simple models that use catch and effort data be used for shark populations?, if yes, which is the best from two surplus production and one partially age-structured models?, and finally, are such models applicable under different scenarios of stock/recruitment relationship, spatial behaviour, stock productivity, and data quality? (i.e. how robust are they?).

## 4.2 Methods.

### 4.2.1 General approach.

The present study relies on the "operating model" approach of Linhardt and Zucchini (1986). This approach involves constructing a detailed, credible model to represent how the system behaves in reality (the **operating model**), including all relevant aspects of biological and non-biological dynamics (i.e. age-structure, fishing systems, data collection, etc.). The second step is to build **estimation procedures** for the system, based usually on simpler models that are feasible for practical situations. Finally, the operating model is used to generate large data sets in a stochastic manner, and these data are used by the estimation procedures to find parameter estimates for each data set. These estimates are then compared to the 'real' values from the operating model, providing a direct means of evaluating the performance (bias, uncertainty, parameter confounding) of each estimation procedure.

The method used to generate the large number of data sets from the operating models is known as a Monte Carlo process. This consists of creating data sets that differ from each other by the effect of one or more types of random errors, generally assumed to be either normally or lognormally distributed. Monte Carlo analysis represents the state of the art in fisheries and has been extensively used for testing various types of fisheries models (Hilborn 1979, Ludwig and Walters 1985, Punt 1988, Fournier and Warburton 1989, Polachek et al. 1993).

The procedure for the study is summarised in figure 4.1 and consists on first constructing a series of simulation models of shark populations involving stochastic effects that reflect 'natural' variations of biological processes. An integrated stochastic fishery subsystem (including 'natural' variations in catchability) is then used to harvest the population and generate time series of catch and CPUE data from each simulation. A total of one hundred Monte Carlo simulations (trials) are performed with each operating model and the data are fed to each of the three fishery estimation models under scrutiny. Finally, the assessment and management parameters obtained from each estimation model are compared against the real values known from the operating model. This comparisson allows to calculate performance statistics for the 100 trials. These statistics indicate the precision and bias in parameter estimates for each of the different estimation procedures.

4.2.2 A set of simulation models of a shark population under exploitation.

The different shark population models considered in this study arise from assuming different types of stock recruitment relationship and spatial behaviour (as represented by the relationship between catchability and stock biomass). Because these two traits stand as some of the less known characteristics in the case of elasmobranchs and are also of great relevance for the determination of key fisheries parameters, it is important to explore the effect of uncertainty in these traits on the estimation of assessment and management parameters for elasmobranchs. The two variations of the stock recruitment relationship are the familiar models of Beverton and Holt (1957) and Ricker (1954). The different spatial

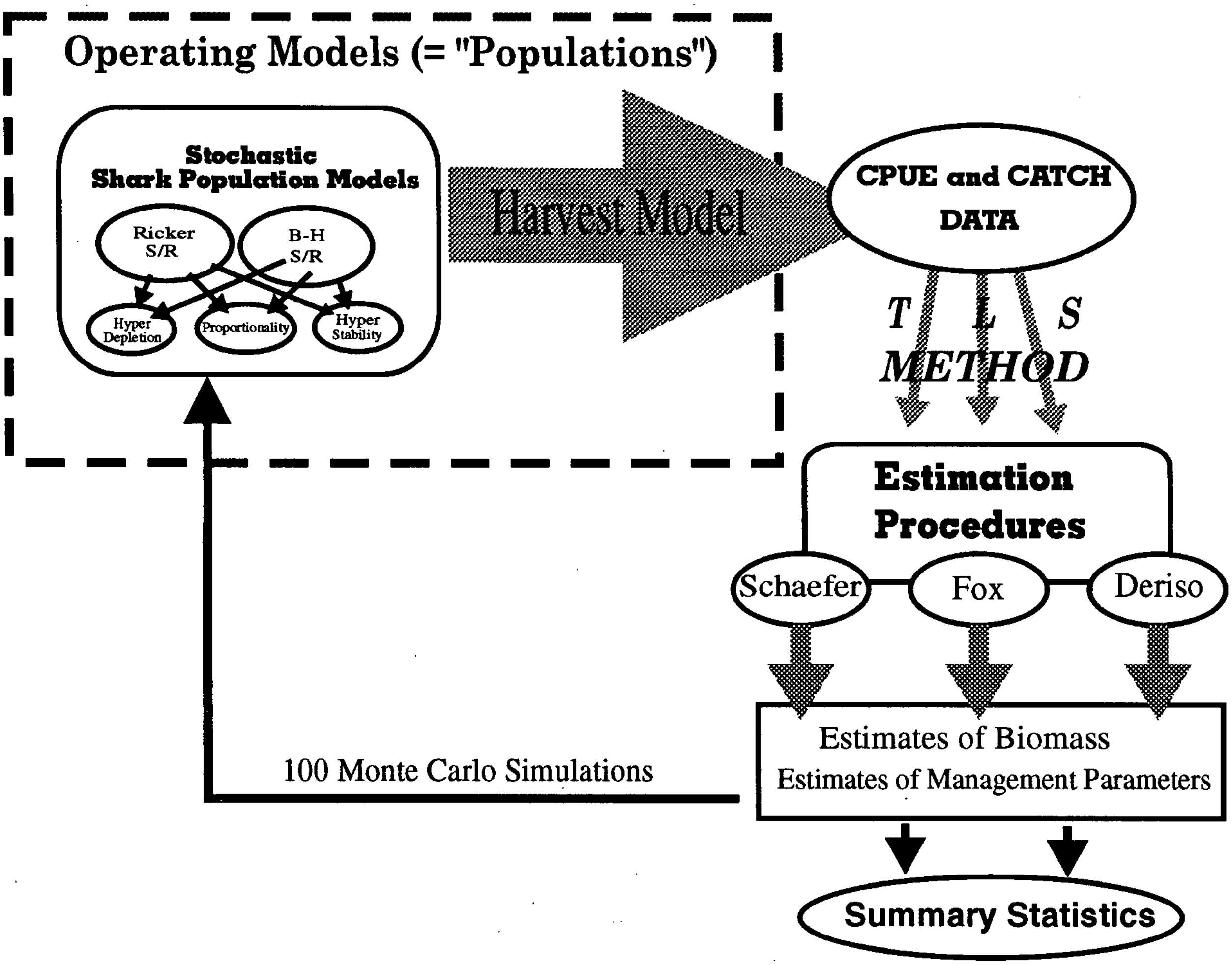



Figure 4.1 Schematic representation of the approach taken in this study for testing the performance of three fisheries models for the estimation of assessment and management parameters for shark fisheries. The process is repeated 100 times (Monte Carlo simulations).

behaviour models are detailed in the following section. Additionally, two sizes of the fishable stock are considered for each shark population by selecting one of two possible ages at first capture. This means that for a given shark population, each 'fishable stock' has a different rate of increase and MSY level. Each operating model (OM) is composed of a simulated population and a particular fishing sub-model.

In agreement with the model developed in chapter three, only the female part of the population is considered (males are assumed not to be limiting for recruitment). The population is simulated by a set of difference equations. The shark population model is fully age-structured (26 age classes) with age-specific fecundity and density dependent natural mortality. Vulnerability to the fishing gear is also age-specific and the two alternative ages of first capture are 4yr and 7yr (fig. 4.2). Growth is slow and sexual maturity is attained relatively late (age 10) in an attempt to mimic the known biology of the carcharhinid sharks, which are the most commonly exploited around the world.

The population is modelled in the following manner:

In the absence of fishing, total number of sharks in the entire population next year $N_{t+1}$ are calculated according to the set of equations:

$$N_{t+1} = \sum_{a=1}^{A_{\max}} [N_{t,a} e^{-M_a}] + R_{t+1} \qquad (4.1),$$

$$R_{t+1} = \frac{dS_{t+1}}{b+S_{t+1}} . e^{-\omega_t} \qquad \text{Beverton-Holt} \quad (4.2a)$$

or

$$R_{t+1} = a' S_{t+1} e^{-b' S_{t+1}} . e^{-\omega_t} \qquad \text{Ricker} \quad (4.2b),$$

| Age | fec. | M | vul(1) | vul(2) | Weight |
|---|---|---|---|---|---|
| 0 | 0 | 0.4 | 0 | 0 | 0.005 |
| 1 | 0 | 0.3 | 0 | 0 | 0.03 |
| 2 | 0 | 0.2 | 0 | 0 | 0.07 |
| 3 | 0 | 0.15 | 0 | 0 | 0.14 |
| 4 | 0 | 0.12 | 0 | 1 | 0.21 |
| 5 | 0 | 0.1 | 0 | 1 | 0.29 |
| 6 | 0 | 0.09 | 0 | 1 | 0.37 |
| 7 | 0 | 0.09 | 1 | 1 | 0.44 |
| 8 | 0 | 0.09 | 1 | 1 | 0.52 |
| 9 | 0 | 0.09 | 1 | 1 | 0.58 |
| 10 | 0.8 | 0.09 | 1 | 1 | 0.64 |
| 11 | 1 | 0.09 | 1 | 1 | 0.69 |
| 12 | 1.5 | 0.09 | 1 | 1 | 0.74 |
| 13 | 2 | 0.09 | 1 | 1 | 0.78 |
| 14 | 2.5 | 0.09 | 1 | 1 | 0.81 |
| 15 | 3 | 0.09 | 1 | 1 | 0.84 |
| 16 | 3.5 | 0.09 | 1 | 1 | 0.87 |
| 17 | 4 | 0.09 | 1 | 1 | 0.89 |
| 18 | 4.5 | 0.09 | 1 | 1 | 0.91 |
| 19 | 5 | 0.09 | 1 | 1 | 0.92 |
| 20 | 5 | 0.09 | 1 | 1 | 0.93 |
| 21 | 5 | 0.09 | 1 | 1 | 0.94 |
| 22 | 5 | 0.09 | 1 | 1 | 0.95 |
| 23 | 6 | 0.09 | 0.97 | 0.97 | 0.96 |
| 24 | 6 | 0.09 | 0.93 | 0.93 | 0.97 |
| 25 | 6 | 0.09 | 0.93 | 0.93 | 0.97 |

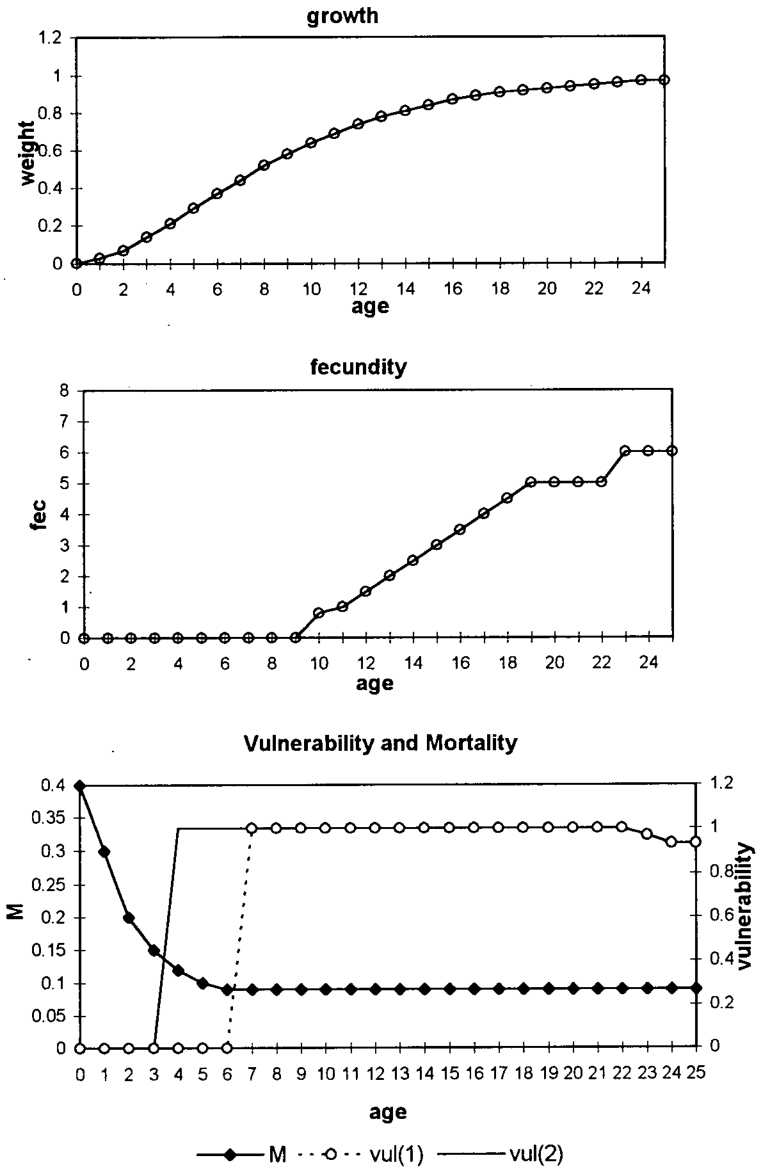



Figure 4.2 Characteristics of the simulated age structured shark population. The two vulnerability schedules differ only in the age of entry to the fishery.

$$S_{t+1} = \sum_{a=A_{mat}}^{A_{\max}} N_{t,a}\, e^{-M_a}\, \varphi_a \tag{4.3}$$

Here, $N_{t,a}$ is the number of organisms of age $a$ at the beginning of year $t$, $M_a$ is the natural mortality coefficient of fish aged $a$, $A_{max}$ denotes the maximum age after which all organisms die, $A_{mat}$ is the age of first sexual maturity, $S_{t+1}$ is the parental stock at the end of the year measured as total number of fertilised eggs or embryos produced, $R_{t+1}$ is the number of sharks actually recruited to the population at the beginning of the year (assuming single pulse birth), and $\varphi_a$ is the product of the age specific fecundity times the number of broods per year. Recruitment is described using either the Beverton and Holt (4.2a) or the Ricker recruitment model (4.2b), where $d$, $b$, $a'$, $b'$ are constants. Natural variation in the recruitment process is represented by the term $\omega$, an independent normally distributed random variable with mean 0 and variance $\sigma^2$.

As the model only tracks the female part of the population, age specific fecundities are set to about half the values commonly found in carcharhinid sharks. The number of broods per year is here assumed to be one, although in reality this number ranges from less than 0.5 to 2 or more depending on the species.

For simplicity, the fishery submodel assumes a type I fishery (Ricker 1975) with a fishing season short enough to allow ignoring natural mortality during harvest. For each simulation, once the population stabilises at carrying capacity, the fishery subsystem produces 20 years of catch and CPUE data using the following equations:

$$C_t = \sum_{a=0}^{A_{\max}} [N_{t,a}(1 - e^{-(q_a f_t e^{-v_t})})\bar{W}_a\ ] \tag{4.4}$$

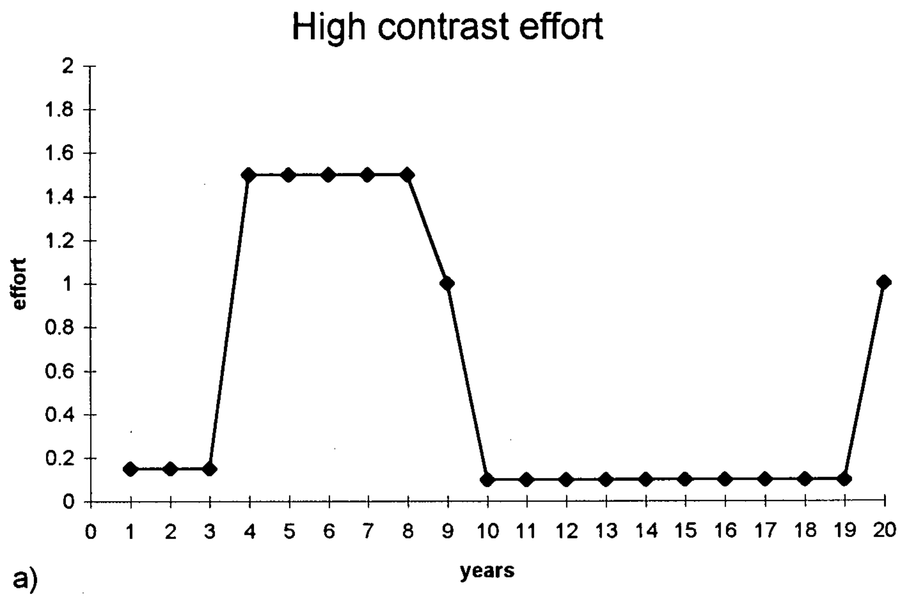
$$CPUE_t = \frac{C_t}{f_t} \tag{4.5}$$

where $C_t$ is total catch in weight during year *t*, $q_a$ is the age-specific catchability coefficient for age *a* (calculated by multiplying the overall baseline catchability coefficient $q$=0.3 by the vulnerability at age; see fig. 4.2), $f_t$ the total effort in year *t*, $W_a$ is the average weight at age *a*. Natural variation in the fishing system is represented by *v*, a normally distributed random variable with mean 0 and variance $\sigma^2$ .

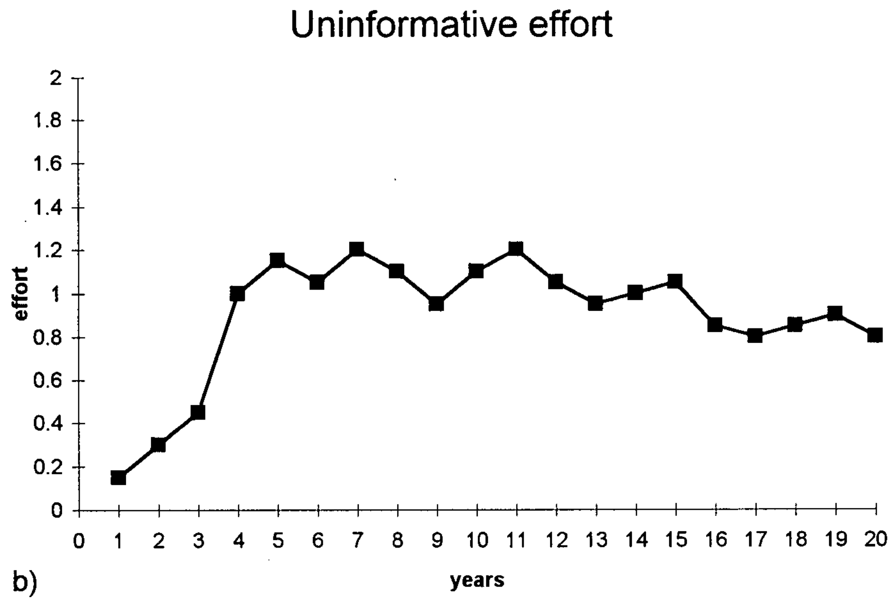
Hilborn (1979) has shown that data contrast (i.e. the occurrence of strongly dissimilar values in proximity) has a substantial effect on the ability of fishery models to overcome possible confounding effects and provide accurate estimates of the parameters. Here, two patterns of fishing effort, each providing a different amount of contrast, were used in the harvesting submodels. The first effort sequence included an initial brief period of light effort followed by a longer period of very high effort, and then a very long recovery period of low efforts (fig. 4.3a). Such an effort pattern is known to provide high contrast in both effort and CPUE (Ludwig and Walters 1985). The second, less informative effort pattern was used in a subset of simulations to test the robustness of the estimation models to data quality (fig. 4.3b). This latter effort sequence is somewhere between a 'one way trip' and 'up and down the isocline' of Hilborn and Walters (1992) and consists of a rapidly increasing effort at the beginning of the series, and slowly declining efforts as catches diminish.

During fishing, population numbers are simulated with variations of equations 4.1 and 4.3:

$$N_{t+1} = \sum_{a=0}^{A_{\max}} [N_{t,a} e^{-(M_a + q_a f_t e^{-v_t})}] + R_{t+1} \tag{4.6}$$




a)




b)

Figure 4.3 Effort patterns used to simulate the harvesting process in the operating models, a) High contrast effort, b) Uninformative effort.

$$S_{t+1} = \sum_{a=A_{mat}}^{A_{\max}} N_{t,a} e^{-(M_a + q_a f_t e^{-v})} f_a \tag{4.7}$$

Given that estimates from the fishery models pertain only to the exploitable part of the population, the population model keeps track only of biomass corresponding to the exploited part of the stock. The equation used for biomass calculation is:

$$B_t = \sum_{a=k}^{A_{\max}} N_{t,a} \bar{W}_a \tag{4.8}$$

where $B_t$ is total biomass in year $t$, and $k$ is the age of recruitment to the fishing gear.

A total of 100 different sets of values for the random variables $v$ and $\omega$ were used to produce the 100 Monte Carlo simulations for each operating model. In order for the model to simulate small enough variations in agreement with the life history characteristics of elasmobranchs, these random variables both had standard deviation 0.05 in all trials.

4.2.3 Spatial behaviour and its representation in the operating models.

Three types of spatial behaviour (characterised by the relationship between CPUE and abundance of fish) were simulated in the operating models. Traditionally, fisheries models assume that fishing effort is distributed at random in relation to the fish (Hilborn and Walters 1992) resulting in a CPUE that is proportional to abundance. This is known as the proportionality assumption and is the cornerstone for the traditional usage of commercial CPUE data as an index of fish abundance. Two other types of relationship between CPUE and abundance are shown in figure 4.4.

Hyperstability occurs when catchability changes inversely with biomass, causing CPUE to remain high while abundance drops. The overall result is an artificial appearance of stability in relative abundance. This phenomenon is probably the most common in real fisheries and

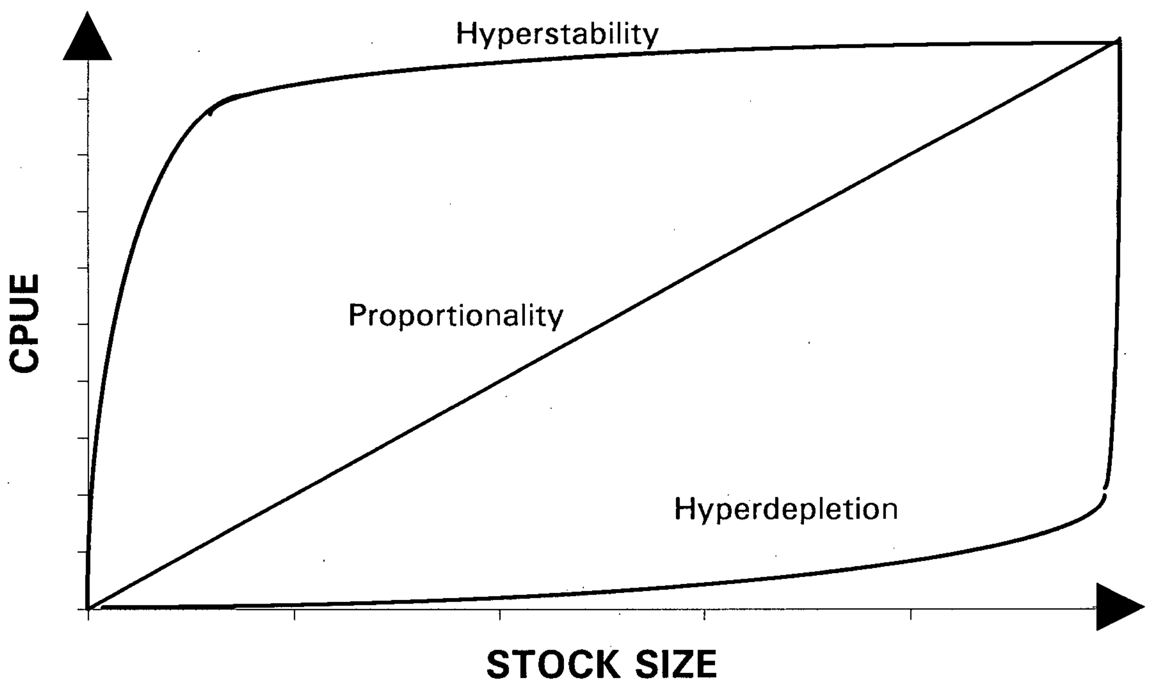



Figure 4.4 Theoretical relationships between cpue and abundance.

is know to be true for species with strong schooling behaviour such as clupeids and some tunids, or when fish congregate at known times and places for reproductive purposes (e.g. orange roughy in the Chatham Rise of N.Z.). Hyperstability is also common when fishermen have large handling times with respect to search times, and when they do not search fish randomly.

In contrast, a situation of hyperdepletion is characterised by a catchability coefficient that drops rapidly as abundance decreases. In this case, CPUE falls faster than the real abundance, creating an appearance of depleted stocks when in reality abundance is still high. Although hyperdepletion appears to be a less frequent situation in real fisheries (Hilborn and Walters 1992) it can be found when fish have virtual sanctuaries (spatially secluded areas unaccessible to exploitation) and fishing proceeds by depleting those areas where fish are vulnerable to the gear.

The three types of abundance-CPUE relationship outlined above can be modelled by density-dependent catchability functions. Several models of density dependent catchability were explored for the representation of hyperdepletion and hyperstability in the operating models, including those of Fournier and Doonan (1987) and Csirke (1989). However the first model did not reproduce hyperstability satisfactorily and the latter could not represent the hyperdepletion case. The function chosen for the present study (Walters pers. comm.) is:

$$q(B_t) = q_a\left(\frac{Q}{1 + (Q-1)\dfrac{B_t}{K}}\right) \tag{4.9}$$

where $q_a$ is the age-specific catchability, $B_t$ is biomass at time $t$, $K$ the carrying capacity or virgin biomass, and $Q$ a constant. As shown in figure 4.5, when $Q = 1$ catchability is constant and the relationship is the usual proportionality model between CPUE and abundance, if $Q > 1$ catchability increases as biomass decreases keeping CPUE high and we have hyperstability. Finally the hyperdepletion case occurs when $0 < Q < 1$ and catchability increases with biomass, hence CPUE drops rapidly with biomass.

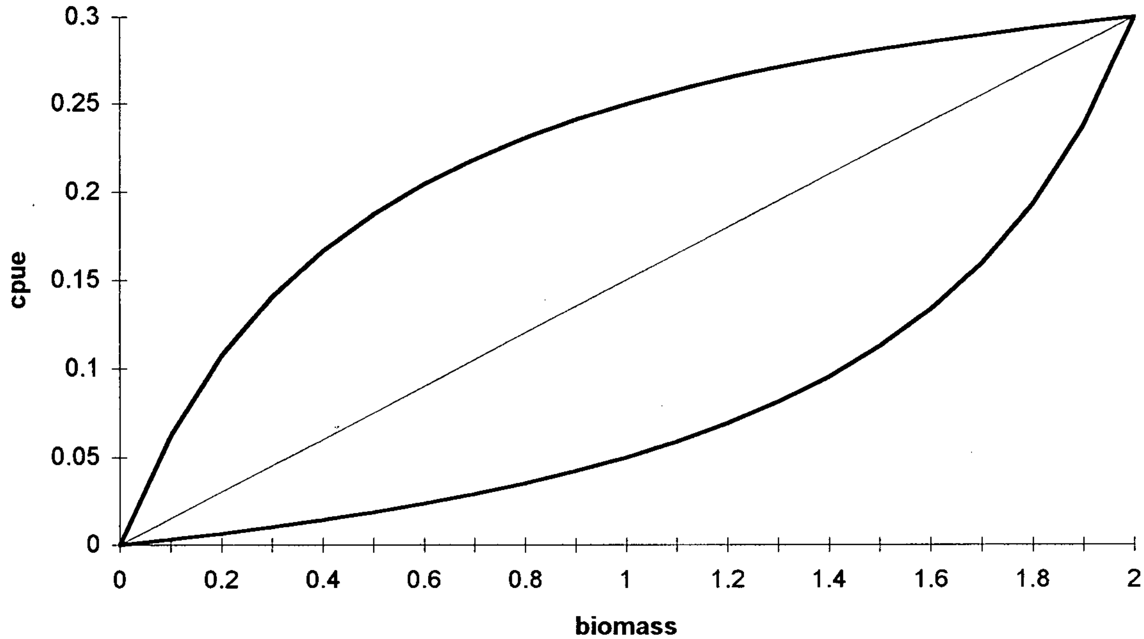



Figure 4.5 Hyperstability (top), proportionality (centre) and hyperdepletion (bottom) as simulated with equation 4.9. (top, $Q$=5; centre, $Q$=1; bottom, $Q$=0.5).

During the Monte Carlo trials, equation 4.9 is incorporated both in the simulations of the operating model while generating fishery data, and in the estimation procedure of each fishery model considered. The former is accomplished by substituting the value of $q_a$ in equation 4.4 by the value of $q(B_t)$. A similar procedure can be used when recovering parameters from the fishery models in the calculation of biomasses from observed catches using the catch equation. This allows to evaluate whether the fishery models can properly estimate the benchmark parameters for assessment and management while estimating the additional parameter Q which defines conditions of hyperdepletion or hyperstability.

4.2.4 Fishery models examined.

Given that an analysis of this kind is numerically very intensive, and considering the different scenarios to be explored during the study, it was necessary to keep everything within a manageable size, thus only three fishery models were included for the study. These are the dynamic versions of the surplus production models of Schaefer (1954) and Fox (1970), and the delay-difference model of Deriso (1980) and Schnute (1985). The Schaefer and Fox models are special cases of the generalised surplus production model of Pella and Tomlinson (1969), and this model was initially considered instead. However, exploratory trials indicated that its performance was inferior to either the Schaefer or the Fox models, possibly due to the extra parameter that has to be estimated in order to fit the Pella and Tomlinson model. Therefore, the latter model was abandoned. The delay difference model of Deriso-Schnute was chosen for its intuitive suitability to the biological characteristics of elasmobranchs, in particular because it takes account of reproductive delays in the population dynamics. Compared to surplus production models, the delay-difference model represents the next step in sophistication and biological realism, thus bridging a gap between surplus production and age-structured models. The delay-difference model is very attractive because of its relative simplicity. For all the above reasons, it seems worth testing the Deriso-Schnute approach alongside those of Schaefer and Fox in order to evaluate the performance of a model carrying some age-structure information against the 'simpler' and 'unrealistic' surplus production models.

4.2.4.1 Surplus production models.

All surplus production models are based on the simple assumption that biomass dynamics of single stocks can be represented by the following self explanatory equation:

*new biomass = old biomass + surplus production - catch* (4.10)

where

*surplus production = recruitment + growth - losses from natural mortality* (4.11)

These models simplify biological processes by including natural sources of increase (additions in the form of reproduction and individual body growth) and loss (natural mortality by predation, illness, competition, etc.) into the single surplus production term. The various surplus production models differ only in the particular form chosen to represent this process.

**Schaefer model.**

The surplus production model of Schaefer (1954) is based on the logistic model for population growth. The difference equation form of the model is:

$$B_{t+1} = B_t + rB(1 - \frac{B_t}{K}) - C_t \qquad (4.12)$$

where $B$ is biomass, $r$ is the intrinsic rate of population growth, $K$ is the carrying capacity and $C$ is catch.

Analytical derivations found elsewhere (e.g. Punt 1988), show that for the Schaefer model the management benchmark of optimal catch ($C_{opt}$) or maximum sustainable yield (MSY), and the stock size at MSY ($B_{MSY}$) are given by the following formulae:

$$C_{opt} = \frac{rK}{4} \tag{4.13}$$

$$B_{MSY} = \frac{K}{2} \tag{4.14}$$

Considering the form of the catch equation assumed in equation 4.4, we have:

$$C_{opt} = B_{MSY}(1 - e^{-(qf)}) \tag{4.15}$$

substituting equations 4.13 and 4.14 in equation 4.15 and solving for $f$ we obtain an explicit form of the optimal effort ($f_{opt}$):

$$f_{opt} = -\frac{\ln(1 - \frac{r}{2})}{q} \tag{4.16}$$

**Fox model.**

The model of Fox (1970) assumes a Gompertz form for the surplus production function. A convenient expression of this model in difference equation form is given by Punt (1988) as:

$$B_{t+1} = B_t + rB(1 - \frac{\ln B_t}{\ln K}) - C_t \tag{4.17}$$

The equations for optimal catch and biomass at MSY for this form of the Fox model are given by:

$$C_{opt} = \frac{rKe^{-1}}{\ln K} \qquad (4.18)$$

$$B_{MSY} = Ke^{-1} \qquad (4.19)$$

By a similar procedure as outlined above for the Schaefer model, it can be shown that the optimal effort in this particular form of the Fox model is equal to:

$$f_{opt} = -\frac{\ln(1 - \frac{r}{\ln K})}{q} \qquad (4.20)$$

4.2.4.2 Delay difference model.

The delay difference model developed by Deriso (1980) and generalised by Schnute (1985) has three key assumptions: 1) Growth in mean body weight at age, for all ages fully vulnerable to fishing gear, follows a linear relationship:

$$\bar{W}_a = \alpha + \rho \bar{W}_{a-1} \qquad (4.21)$$

where $W$ is average weight, the index $a$ is age and $\alpha$, $\rho$ are constants. 2) All fish aged $k$ and older are equally vulnerable to fishing (knife edge selection), and 3) All fish aged $k$ and older have the same natural mortality rate.

If we assume that the total survival rate can be decomposed into constant natural survival and variable harvest survival we have:

$$S_t = \psi\,(1-h_t) \tag{4.22}$$

where $S_t$ is the total survival rate, $h$ is harvest rate, and $\psi$ is the natural survival rate.

We can decompose the biomass equation 4.8 into the sum of biomasses of all ages $k+1$ and older plus the number of recruits $R_t$ to the fishery times their average weight (at age of recruitment) $w_k$:

$$B_t = \left[\sum_{a=k}^{A_{\max}} N_{t,a}\,\bar{W}_a\right] + w_k R_t \tag{4.23}$$

Given that each $N_{t,a}$ can be written as survivors from last year at age $a-1$, and each of the average weights at age can be written using equation 4.21, substituting both into equation 4.23 and rearranging gives:

$$B_t = S_{t-1}\left(\alpha \sum N_{a-1,t-1} + \rho \sum N_{a-1,t-1}\, w_{a-1}\right) + w_k R_t \tag{4.24}$$

where the sums are over ages $k+1$ to age maximum. Factoring out terms that do not depend on age, results in sums over ages $k$ and older for year $t-1$, thus:

$$B_t = S_{t-1}\alpha N_{t-1} + S_{t-1}\rho B_{t-1} + w_k R_t \tag{4.25}$$

Total numbers are:

$$N_t = S_{t-1} N_{t-1} + R_t \tag{4.26}$$

The last two equations combined together produce the delay-difference model with the advantage that population numbers can be completely ignored. This is accomplished by first writing the term $\alpha N_{t-1}$ above in terms of recruits and survivors from $N_{t-2}$:

$$\alpha N_{t-1} = \alpha S_{t-2} N_{t-2} + \alpha R_{t-1} \tag{4.27}$$

Then in a similar manner, the term $\alpha S_{t-2} N_{t-2}$ can be expressed in terms of a biomass equation for $B_{t-1}$ and $N_{t-2}$ as:

$$\alpha S_{t-2} N_{t-2} = B_{t-1} - \rho S_{t-2} B_{t-2} - w_k R_{t-1} \tag{4.28}$$

which if substituted into equation 4.27 and the result substituted into equation 4.25, finally gives us, after some algebra, the delay-difference equation:

$$B_t = S_{t-1} B_{t-1} + \rho S_{t-1} B_{t-1} - \rho S_{t-1} S_{t-2} B_{t-2} - (w_k - \alpha) S_{t-1} R_{t-1} + w_k R_t \tag{4.29}$$

A form of the delay-difference model recommended by Schnute (1985) avoids the explicit calculation of $\alpha$ by using a substitution based on equation 4.21:

$$B_t = S_{t-1} B_{t-1} + \rho S_{t-1} B_{t-1} - \rho S_{t-1} S_{t-2} B_{t-2} - \rho w_{k-1} S_{t-1} R_{t-1} + w_k R_t \tag{4.30}$$

which is the form employed in the present study.

Recruitment in the delay-difference model was predicted using the following equations:

$$R_{t+1} = \frac{dS_{t-k+1}}{b+S_{t-k+1}} \qquad \text{Beverton \& Holt} \qquad (4.31)$$

$$R_t = S_{t-k+1}\, e^{(a' - b' S_{t-k+1})} \qquad \text{Ricker} \qquad (4.32)$$

where *d* is the maximum recruitment rate when $S_{t-k+1}$ is large, *b* is the spawning stock size needed to produce a recruitment of half the maximum, *a'* is the maximum productivity per spawner when the spawning stock is low, and *b'* measures density-dependent reductions in productivity as stock size increases (Hilborn and Walters 1992).

In the case of the Deriso-Schnute model, management benchmarks can be found by iteration. This is accomplished by trying different exploitation rates to calculate $C_{opt}$ from one of the following formulae for equilibrium biomass $B_e$ (Hilborn and Walters 1992):

$$B_e = \frac{a}{G} - \frac{b}{(1-h)} \qquad \text{Beverton-Holt .} \qquad (4.33)$$

$$B_e = \frac{[a' + \ln(1-h) - \ln G]}{[b'(1-h)]} \qquad \text{Ricker} \qquad (4.34)$$

where

$$G = \frac{[1-(1-\rho)\psi(1-u)+\rho\psi^2(1-u)^2]}{w_k-\rho w_{k-1}\psi(1-u)} \tag{4.35}$$

Once the optimal catch and corresponding optimal exploitation rate $U_{opt}$ have been found, the optimal effort is calculated as:

$$f_{opt} = -\frac{1}{q}\ \ln(1-U_{opt}) \tag{4.36}$$

Fixing parameter values for the Deriso-Schnute model.

Fitting the delay-difference model as expressed in equation 4.30 to catch and CPUE data requires estimating a total of seven parameters: $\rho$, $w_k$, and $\psi$, are needed by the delay-difference equation; two additional parameters (*d, b* or *a', b'*) are needed for the stock recruitment model; finally the catchability $q$ and the initial stock size $B_0$ are required in order to relate CPUE to biomass. The procedure is simplified by fixing four of the parameters (*d* or *a'*, $\rho$, $w_k$ and $\psi$) prior to the Monte Carlo trials, thus avoiding the need to estimate them during model fitting. This leaves only three parameters to be estimated, $B_0$, $b$ (or $b'$) and $q$. Because fitting the surplus production models requires estimation of only three parameters, this strategy gives a handicap to the Deriso-Schnute model and allows a fair comparison between models.

All fixed parameters were derived from the data used to simulate the populations. The growth parameter $\rho$, was calculated from a Ford-Walford plot of the weights at age presented in figure 4.2, while $w_k$ was taken directly from the same data. The natural survival $\psi$ is constant after age 6 (fig. 4.2), and is calculated as $e^{-M}$.

The estimation of the parameters *d* and *a'* of the stock recruitment models can be avoided by assuming that the population was sustainable (i.e. at equilibrium) when exploitation began: at equilibrium, the losses due to natural mortality have to be equal to the gains in the form of new recruits. Such an assumption is true in our particular case, and is a reasonable guess for other cases if we know the stock has not been fished previous to data gathering. This allows to fix one point in the stock-recruitment function (the maximum level of recruitment) eliminating the need to estimate directly the parameters *d* or *a'*. Thus, recruitment at equilibrium $R_0$ must be equal to numbers dying ($N_o(1-\psi)$). The parameter $N_0$ is calculated from the model estimate of $B_0$ and the average equilibrium weight equation provided in page 340 of Hilborn and Walters (1992). This is used as the recruitment value in the first year of the calculations, because we are assuming these recruits were produced by the virgin spawning population. The parameter *d* or *a'* of the stock-recruitment relationship (maximum recruitment level) can then be expressed in terms of $R_0$, $B_0$ and the parameter *b* or *b'* of the stock-recruitment model:

$$d = R_0 \frac{(b + B_0)}{B_0} \qquad \text{Beverton-Holt} \qquad (4.37)$$

$$a' = \ln\left(\frac{R_0}{b'}\right) + B_0 b' \qquad \text{Ricker} \qquad (4.38)$$

avoiding in this way their independent estimation.

**Non-delay model and misspecification of stock-recruitment relationship.**

The delay-difference model of Deriso-Schnute includes a reproductive delay in the stock/recruitment relationship of *k-1* yr, that is, the age of recruitment minus one year. In order to see how important it is to consider this delay in the model, during the initial stages of this study I used a modified model that calculates recruitment as a function of last year's biomass. I refer to this model modification as the non-delay Deriso-Schnute model throughout this study. The original model considering the reproductive delay is referred to as the Deriso-Schnute model or the delay-difference model.

The choice of the explicit stock-recruitment function to be used in the Deriso-Schnute model is another point of uncertainty during model fitting. A cross examination of the effects of choosing the wrong stock-recruitment function for fitting the Deriso-Schnute model was performed during trials with simulated populations that had proportionality between CPUE and biomass. This analysis involved generating data with one of the two stock-recruitment functions and performing the estimations using the other (wrong) stock-recruitment function in the Deriso-Schnute model. This approach provides an opportunity to evaluate the effects of such a model misspecification in the performance of the delay-difference model.

4.2.5 Benchmarks for model performance.

Model performance for stock assessment in this study was measured by three benchmarks of biomass estimation: the size of the stock at the beginning of exploitation $B_0$, the terminal stock size after 20yr of fishing $B_{end}$, and the total estimated absoluted discrepancy $TD$, a measure of the biomass estimates for the entire 20yr time series. During each simulation trial, estimation bias was thus calculated for each benchmark as that proportion of its real value represented by its corresponding estimate. In the case of $B_0$ and $B_{end}$ this is only the estimate divided by the corresponding real value. The measure of performance for the entire 20yr time series $TD$, is calculated as:

$$TD = \sum_{t=1}^{20} \left| \frac{\hat{B}_t - B_t}{B_t} \right| \tag{4.39}$$

where $t=1$ corresponds to the first year of fishing, $B_t$ is estimated biomass and $B_t$ is the true value of the biomass.

During the present study, the performance criteria for management parameter estimation of the fishery models are the traditional benchmarks of optimal effort $f_{opt}$ and optimal catch $C_{opt}$, estimated for each Operating Model as in Appendix 4. The measure of estimation bias is the proportion of the true management benchmark represented by its corresponding estimate, just as explained above for the biomass parameters.

4.2.6 Fitting fishery models to data.

**Error types and the TLS fitting procedure.**

The method used to fit the fishery models to each time series of simulated fishery data was the Total Least Squares method (TLS) of Ludwig et al. (1988). This procedure attempts to combine two different approaches for dealing with errors during fit of models to catch and effort data.

Most fitting procedures have traditionally assumed that departures from model predictions (the differences between the data values and the values predicted by the models) are attributable to natural variability in the catch-effort "observation" system. Such errors can arise due to randomness in the catchability or in the effective effort applied. This is known as the observation error assumption and has been used by Pella and Tomlinson (1969), Butterworth and Andrew (1984), Polachek et al. (1993) and Prager (1994), among others. An alternative approach is to assume the errors to be all due to biological processes such as recruitment, growth, or survival. This less common practice is known as the process error assumption and is found in the works of Schnute (1977) and Ludwig and Walters (1985). The method of TLS takes account of both types of errors by minimizing a weighted sum of squares that involves both observation errors and process errors:

$$TLS = \frac{1}{1-\lambda}\sum_t \omega^2 + \frac{1}{\lambda}\sum_t v^2 \tag{4.40}$$

where $v_t$ and $\omega_t$ are the observation and process errors respectively, and $\lambda$ is the ratio of the variance of the $v_t$ to the total variance $v_t + \omega_t$. Following the results of Ludwig et al. (1988) and Punt (1988), the parameter $\lambda$ is fixed at 0.5 because it cannot be satisfactorily estimated from the data.

There are at least two different assumptions about the form of the observation error $v_t$. Ludwig et al. (1988) recommend assuming multiplicative errors associated to the effective "observed" effort:

$$f_{obs} = f_{true} e^{v_t} \tag{4.41}$$

where $f_{obs}$ is the observed effort, $f_{true}$ is the true effort and $v$ (the observation error) is a normally distributed random variable with mean 0 and variance $\sigma_v^{\ 2}$. Another possibility is to assume additive errors perhaps associated with variations in catchability, thus:

$$q_{obs} = q_{true} + v_t \tag{4.42}$$

where $q_{obs}$ is the observed catchability, $q_{true}$ the nominal catchability and $v_t$ is as above.

Although data are simulated by the operating populations using only equation 4.37, both assumptions were implemented in the estimation procedures during early trials in order to evaluate the effects of each assumption in performance of the TLS method.

The process errors appear as part of the biomass dynamics:

$$B_{t+1} = P(B_t) e^{\omega_t} - C_t \tag{4.43}$$

where production $P$, is a function of biomass $B$, $C$ is the catch, and $\omega_t$ is a normally distributed random variable with mean 0 and variance $\sigma_w^2$.

During the actual fitting process, the $v_t$'s are treated as additional (nuisance) parameters to be estimated, that is, the estimation procedure searches for the best values of $v_t$. Once the observed effort or catchability is corrected with these estimates according to 4.41 or 4.42, the biomass can be estimated from the catch equation using the observed catch and estimates of $q$. Finally, the process error $\omega_t$ can be estimated from 4.43. Note that only $n$-$1$ process errors can be estimated from each time series of $n$ years of data because both $B_{t+1}$

and $B_t$ appear in 4.43.

Initial biomass assumption.

Normally, the size of the stock at the beginning of harvest ($B_0$) is estimated by the fitting procedure together with the entire biomass series and the rest of the model parameters. However, it is possible to omit this additional parameter when there is information about the history of the fishery suggesting that the stock was at its virgin size when exploitation began. In the case of surplus production models, this implies assuming that the first biomass in each time series is equal to the carrying capacity of the system ($B_0$=$K$) and incorporating this explicitly in the fitting algorithm. According to Punt (1991) the $B_0$=$K$ assumption can help improve results under some situations. The benefits of this simplification in the fitting process are investigated throughout most of the simulations.

**Non-linear search algorithms.**

Fitting the fishery models under test to the simulated data involves minimising equation 4.40 via some non-linear optimisation algorithm that searches for the best values of $B_0$, $b$ or $b'$, $q$ and all $v_t$'s. Such an algorithm must be able to perform numerical calculation of derivatives due to the complicated functions obtained as a solution to equation 4.43 especially in the case of the delay-difference model. The first optimisation choice for the present study was the Fletcher-Powell algorithm because of its speed and availability through free distribution programmes on disk . However, after a few trials it was evident that this algorithm could not converge to a solution for too many cases. The downhill simplex algorithm of Nelder and Mead (Press et al. 1990) was then implemented because of its renowned robustness. This algorithm improved the rate of success in finding solutions for difficult cases but at a great cost in time due to the slowness of the method. However, still the rate of success (convergence to a reasonable solution) was not satisfactory. The Marquardt algorithm was finally implemented and used with better success for the totality of the study. Still, for some types of trials success rates as low as 50% were observed.

**Estimation bias.**

In model fitting, it is customary to assess the bias of a particular estimator. Better parameter estimates can be obtained from point estimates after correction for bias. One of the most popular methods of assessing and correcting estimation bias in recent times is the bootstrap (Efron 1982). However, this technique involves a very intensive process (250-1000 simulations per trial) which is extremely expensive in computing time. The nature of the present study, involving 100 trials per operating model, prevented using bias-corrected estimates of the management and assessment benchmarks. Nevertheless, other studies suggest that estimation bias is negligible at least for the Schaefer surplus production model (Prager et al. 1994).

### 4.2.7 The types of trials performed.

#### 4.2.7.1 Nomenclature.

Throughout this work, operating models will be called 'operating populations' or OP's to avoid confusion between so many types of models: fishery, estimating, and operating models. Operating populations designated only by the name of the type of stock-recruitment function used (Ricker and Beverton-Holt ) refer to operating populations considering CPUE-biomass proportionality and using the informative effort pattern to generate the fishery data. Operating populations using the less productive fishable stock are here designated as 'unproductive', whilst those using the non-informative effort sequence are known as 'uninformative'. Finally, OP's designating the hyperdepletion and hyperstability cases are named using precisely these two terms.

#### 4.2.7.2 Sequence of trials.

The different operating populations considered, along with the number of estimation models and the different variations for the observation error estimation tested in this study, made the number of possible combinations too large. Computational time constraints and the need to keep manageable the amount of results to be analysed, dictated a hierarchical approach for carrying out the trials. This strategy allowed to include all OP's and examine all possible variations of the estimation procedure while discarding along the way procedures that proved to be unsatisfactory and did not warrant further analysis.

First, the two alternative assumptions for the observation error (multiplicative in effort or additive in catchability, eqs. 4.41 and 4.42) were investigated with the Ricker and Beverton-Holt OP's. These trials involved estimating parameters with the Schaefer and Fox surplus production models and the delay and non-delay versions of the Deriso-Schnute model. After comparing the performance of both observation error assumptions, only the optimal of these assumptions (i.e. multiplicative) was kept for the remaining part of the study. Further trials with the Ricker and Beverton-Holt OP's included the delay-difference model with misspecified stock-recruitment function and the $B_0=K$ trials. From this point onwards, only the Schaefer, Fox and Deriso models were evaluated. The tests with the hyperstability and hyperdepletion OP's included only the multiplicative observation error assumption. Another stage of trials designed to further test the robustness of the Schaefer, Fox and delay-difference models involved simulations with the unproductive OP's, both with and without the $B_0=K$ assumption. Finally, these three models were tested with unproductive and uninformative versions of the Ricker and Beverton-Holt OP's. A summary of the different types of trials according to the sequential order in which they were performed is provided in tables 4.1 and 4.2.

## 4.3 Results

Preliminary trials with the Deriso-Schnute model highlighted the importance of getting reasonably accurate estimates of the fixed parameters that are not estimated during the non-linear fit of the model to data. Errors in the values of the age of recruitment to the fishery (wrong value by 1 or 2 ages), the natural survival, or the different weights required during estimation ($w_k$, $w_{k-1}$ and average weight) all resulted in biassed estimates of management parameters and very frequently in failure of the estimation procedure to converge. In fact, a great amount of time had to be spent finding the right values of the fixed parameters and checking the implementation of the model procedure in order to get reasonable results with this model, due to its relative complexity.

### 4.3.1 Simulated populations and fishery data time series.

Figure 4.6 shows the typical trajectories of population growth for each of the two operating populations (Ricker and Beverton-Holt). This general behaviour was also observed for the

Table 4.1 Types of trials performed with informative effort pattern and productive stock (age or recruitment = 7yr). A= additive observation error; M= multiplicative observation error; S=Schaefer; F=Fox; D=Deriso-Schnute; Dn=non-delay Deriso-Schnute; D**= misspecified Deriso-Schnute.

<table>
<tr><th colspan="2">Operating Population</th><th colspan="2">Estimation Procedure</th></tr>
<tr><th>Spatial behaviour</th><th>Stock-recruitment</th><th>Assumptions</th><th>Fishery models</th></tr>
<tr><td rowspan="6">Propor-tionality</td><td rowspan="2">Beverton & Holt</td><td>A</td><td>S, F, D, Dn</td></tr>
<tr><td>M</td><td>S, F, D, Dn, D**</td></tr>
<tr><td rowspan="2">Ricker</td><td>A</td><td>S, F, D, Dn</td></tr>
<tr><td>M</td><td>S, F, D, Dn, D**</td></tr>
<tr><td>Beverton & Holt</td><td>M ($B_0$=K)</td><td>S, F, D, D**</td></tr>
<tr><td>Ricker</td><td>M ($B_0$=K)</td><td>S, F, D, D**</td></tr>
<tr><td rowspan="2">Hyper Stability</td><td>Beverton & Holt</td><td>M</td><td>S, F, D</td></tr>
<tr><td>Ricker</td><td>M</td><td>S, F, D</td></tr>
<tr><td rowspan="2">Hyper Depletion</td><td>Beverton & Holt</td><td>M</td><td>S, F, D</td></tr>
<tr><td>Ricker</td><td>M</td><td>S, F, D</td></tr>
</table>

Table 4.2 Types of trials performed with clue proportional to biomass and an unproductive stock (age of recruitment = 4yr). Key as in table 4.1.

<table>
<tr><th colspan="2">Operating Population</th><th colspan="2">Estimation Procedure</th></tr>
<tr><th>Effort pattern</th><th>Stock-recruitment</th><th>Assumptions</th><th>Fishery models</th></tr>
<tr><td rowspan="4">Informative Effort</td><td rowspan="2">Beverton & Holt</td><td>M</td><td>S, F, D</td></tr>
<tr><td>M ($B_0$=K)</td><td>S, F, D</td></tr>
<tr><td rowspan="2">Ricker</td><td>M</td><td>S, F, D</td></tr>
<tr><td>M ($B_0$=K)</td><td>S, F, D</td></tr>
<tr><td rowspan="4">Uninformative Effort</td><td rowspan="2">Beverton & Holt</td><td>M</td><td>S, F, D</td></tr>
<tr><td>M ($B_0$=K)</td><td>S, F, D</td></tr>
<tr><td rowspan="2">Ricker</td><td>M</td><td>S, F, D</td></tr>
<tr><td>M ($B_0$=K)</td><td>S, F, D</td></tr>
</table>

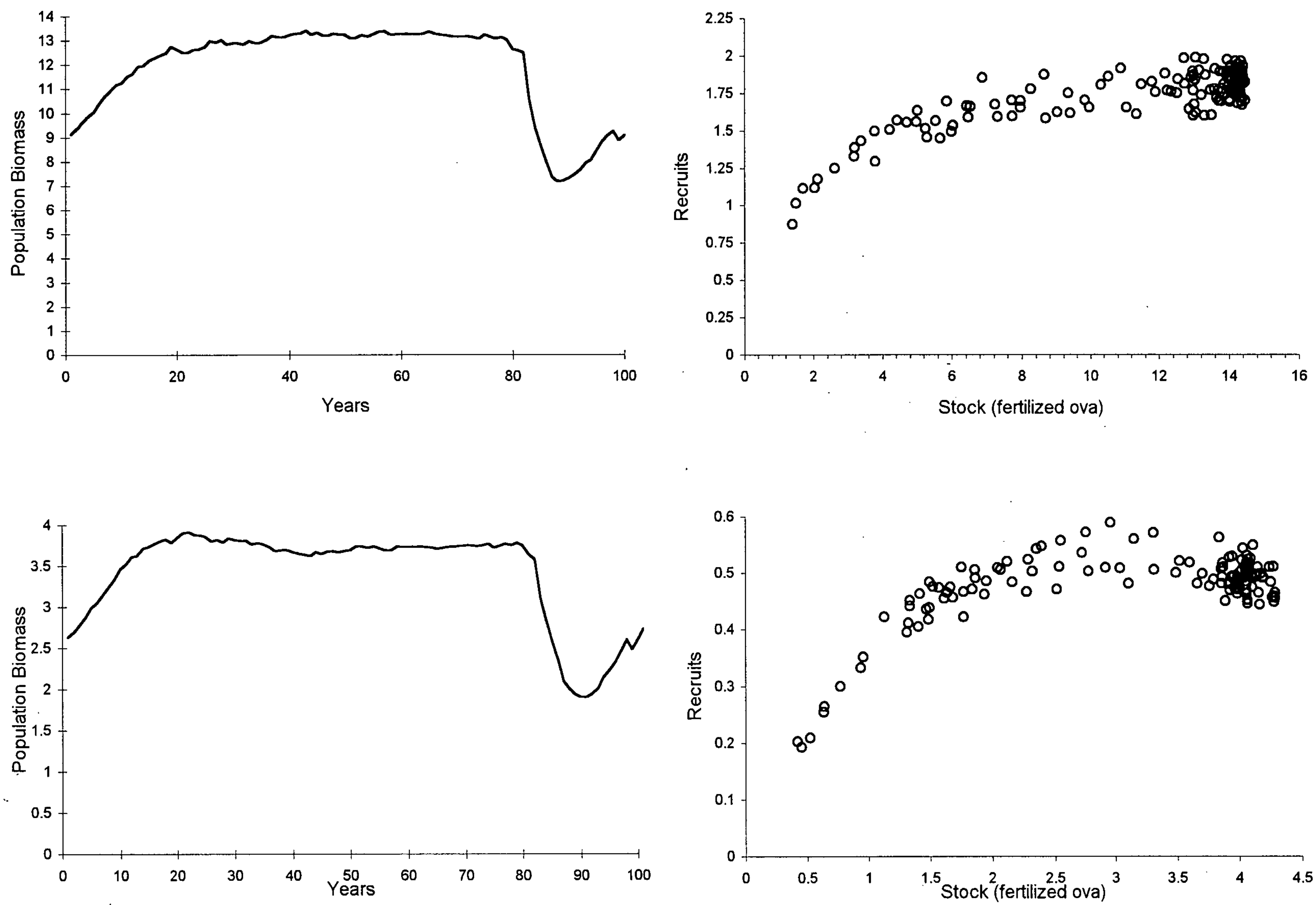



Figure 4.6 Total biomass trajectory of the simulated shark populations simulated stock-recrutiment data.
Top figures, Beverton-Holt OP; Bottom figures, Ricker OP. Fishing starts at year 80.

hyperstability and hyperdepletion cases, because the differences in spatial behaviour between operating populations becomes evident only during the exploitation phase.

Examples of the 20yr time series of biomass and CPUE for four variations of the Beverton and Holt OP are shown in figure 4.7. The Ricker OP populations behaved in a similar fashion differing only in absolute values. An informative pattern (Ludwig and Walters 1985) of rapid stock decline followed by a recovery when fishing subsides was satisfactorily reproduced under the proportionality and hyperdepletion scenarios. However, this was not the case for the hyperstability case or when data were generated using the uninformative effort series. The increase in catchability associated with lower biomasses in the hyperstability case produced a larger fishing mortality despite the terminal mild effort used, thus preventing the recovery of the population. This figure also illustrates very clearly the differences in the CPUE-biomass relationship for each of the three spatial assumptions in the OP. For the two 'proportionality' cases, where $q$ remains constant over all stock levels, the expected close correspondence between CPUE and biomass is evident. In contrast, for the hyperstability and hyperdepletion cases the biomass trajectory is poorly indexed by CPUE.

4.3.2 Tests with the 'proportionality' operating populations.

All estimation results throughout this study are presented as modified box plots showing the distribution of the benchmark parameters as measured by the median and quartiles. In each of the $f_{opt}$, $C_{opt,}$ $B_{0,}$ and $B_{end}$ box plots, the horizontal line at unity represents the absence of bias, i.e. a perfect answer. For the total expected discrepancy TD, the target value is zero (a perfect biomass fit).

4.3.2.1 Tests of observation error assumptions and different versions of the Deriso-Schnute model.

The performance of estimation procedures using the assumption of multiplicative observation errors in effort is better in all cases with the exception of some of the benchmarks for the Deriso-Schnute model (figs. A.1 and A.2, Appendix 1). Not only the bias is greater in the large majority of the estimates from procedures using the additive

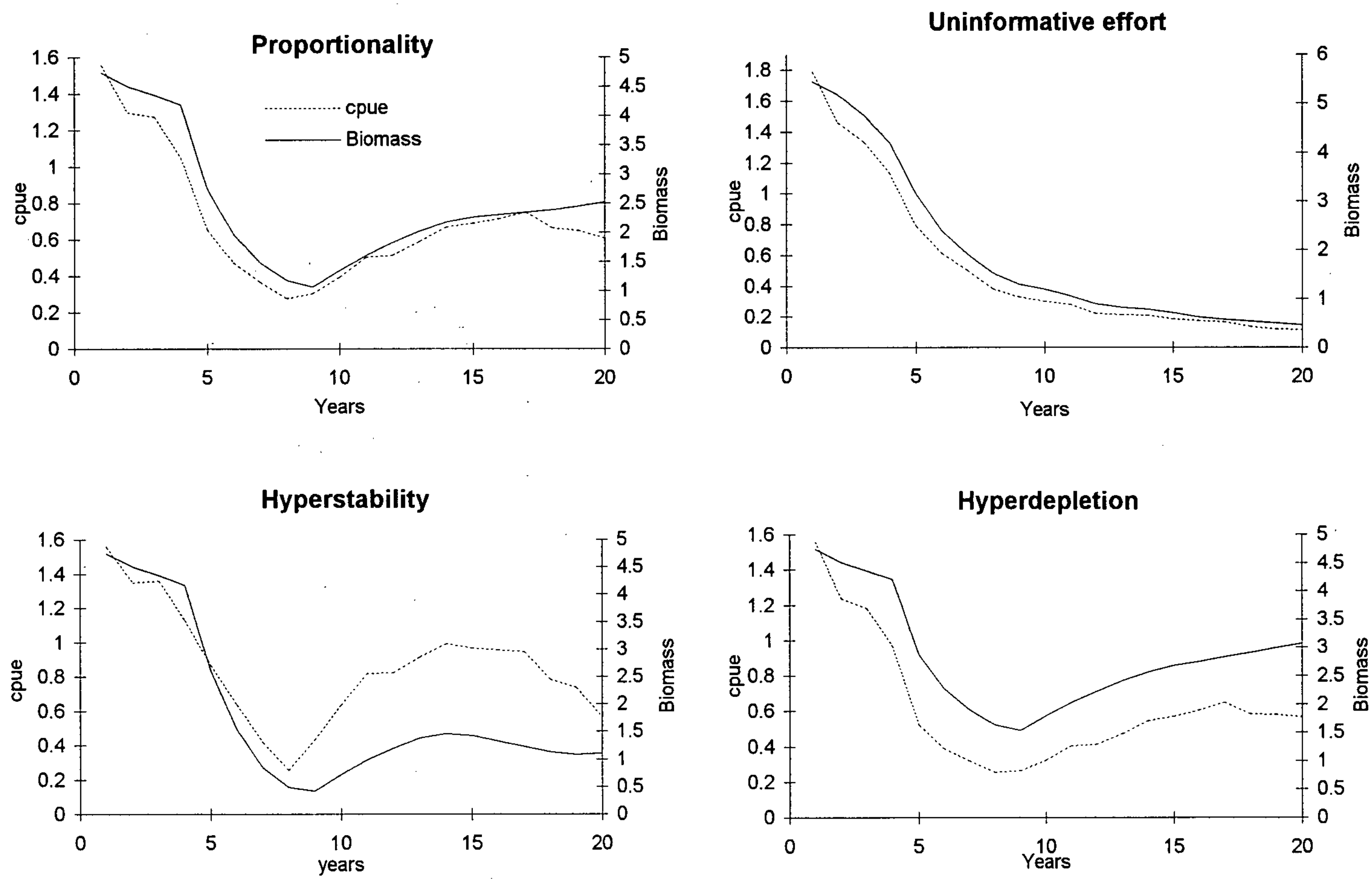



Figure 4.7 Examples of one realization of biomass and cpue trends for different types of operating populations.

observation error assumption, but the uncertainty in the estimates and the number of unsuccessful fits are also usually larger than for estimates obtained by using the multiplicative observation error assumption. This is particularly noticeable in the results for the Ricker OP. The additive error in catchability assumption was therefore abandoned after these trials.

The non-delay Deriso-Schnute model is generally inferior to the delay-difference Deriso-Schnute model (which accounts for the reproductive delay). In particular, the former model could not be used to obtain estimates of the management benchmarks for the Beverton-Holt OP, and estimates for the Ricker OP are inferior to those from the delay-difference model (figs. A.1 and A.2, Appendix 1). The differences in biomass assessment between the two models are minor. The non-delay Deriso-Schnute model was discarded at this stage.

4.3.2.2 Tests between the Schaefer, Fox, and Deriso-Schnute model.

The Deriso-Schnute model performs far better than any of the surplus production models for the estimation of management parameters, particularly for the estimation of optimal effort (fig. 4.8). While the Schaefer and Fox models are reasonably good for estimating $C_{opt}$, their performance for $f_{opt}$ estimation is dismal especially in the Fox model and with the exception of the Scheafer estimate for the Beverton-Holt OP.

The three models led to satisfactory estimation of stock assessment benchmarks, with all estimates of $B_0$ and $B_{end}$ having relatively small biasses. There are small negative biasses in the Deriso-Schnute model estimates of $B_0$ and $B_{end}$. In contrast, estimates from both surplus production models have small positive biasses. Among the surplus production models, the Fox model performs better than the Schaefer model for biomass assessment. According to the total discrepancy, all models manage to obtain relatively good fits to the overall biomass time series (a *TD* of 2 indicates a 10% average absolute error) with the Schaefer model having the largest discrepancies. The Deriso-Schnute model appears to be more robust: its performance is more consistent across operating populations than in the Schaefer and Fox models. Notably, the biomass time series of the Ricker OP seemed to be more difficult to assess for the surplus production models, than those of the Beverton-Holt OP (figs. 4.9a and b).

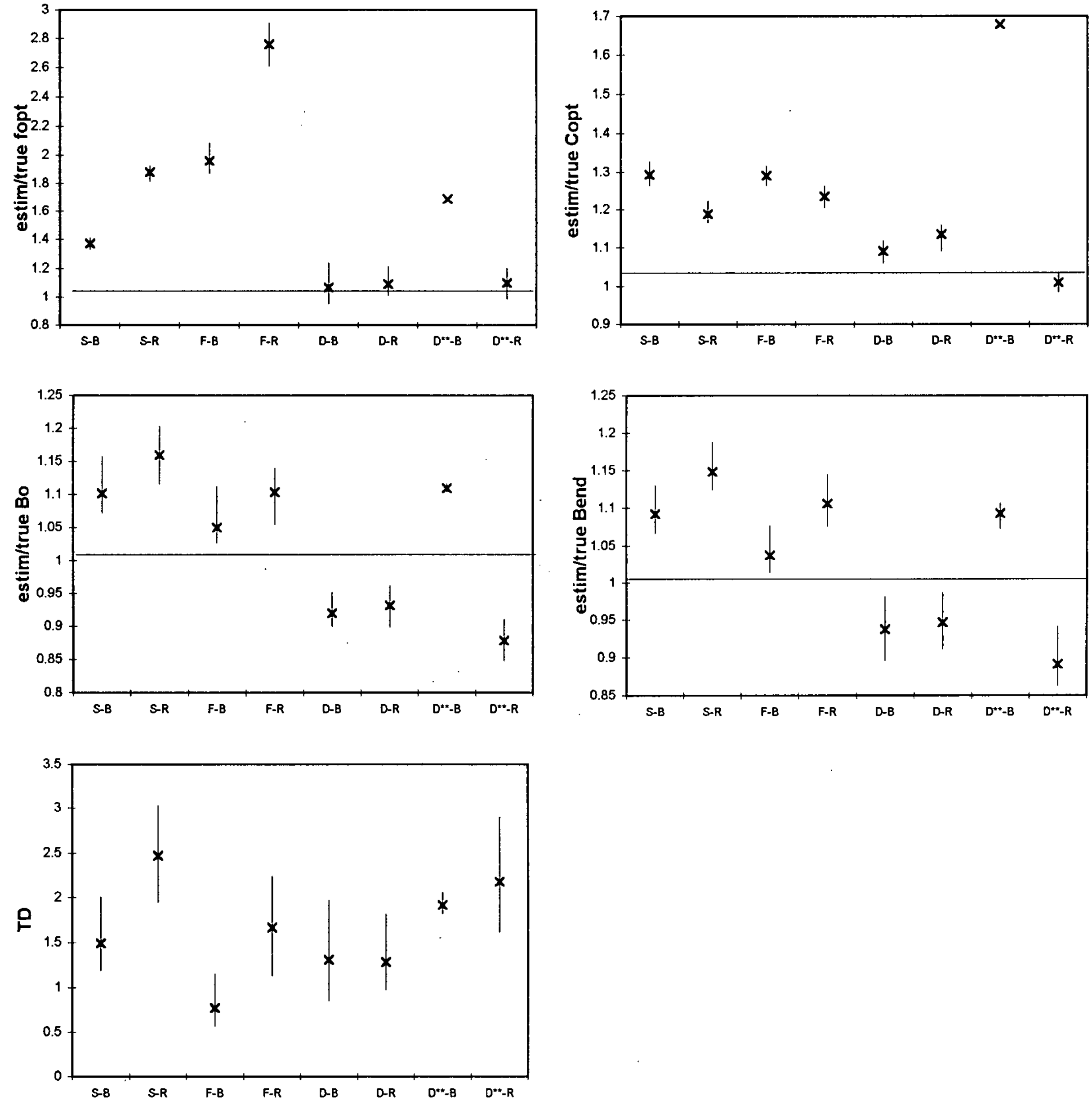
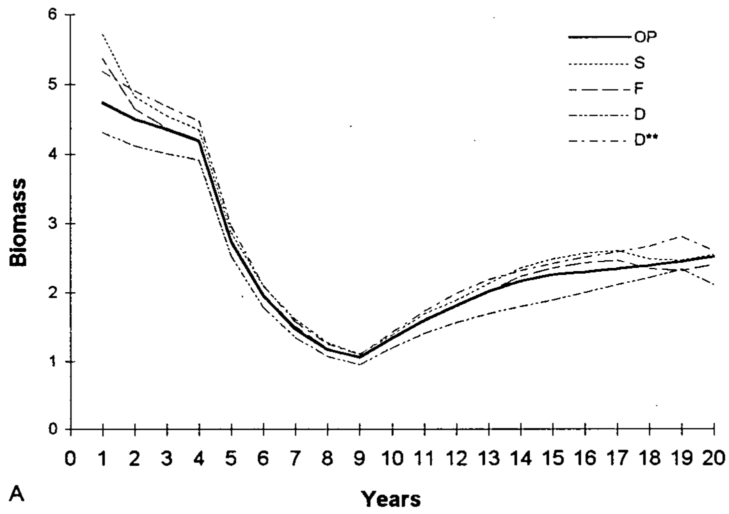
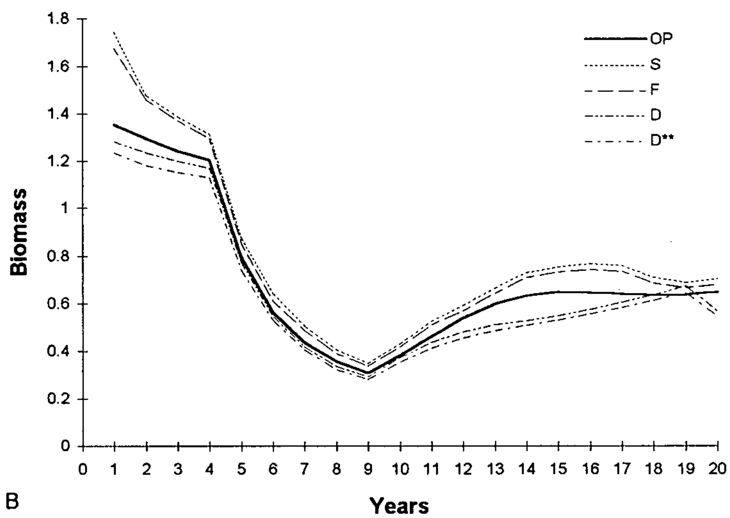



Figure 4.8 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the multiplicative observation error assumption. Monte Carlo simulations performed with the proportionality OP, high contrast effort, and productive stock. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.




A

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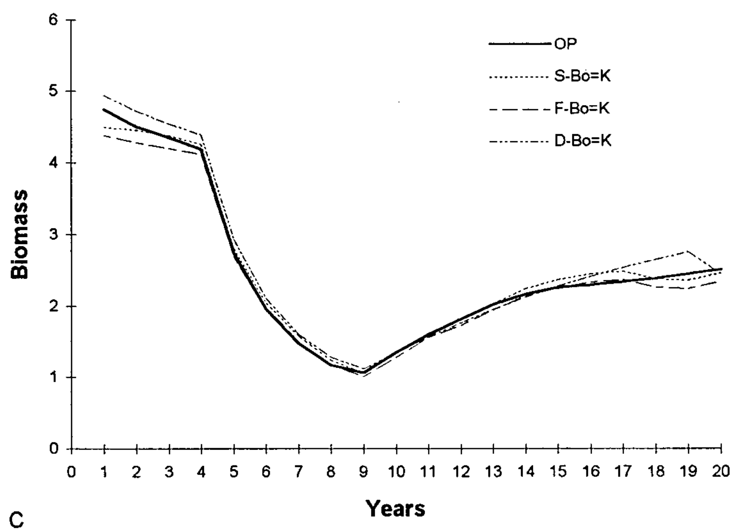



C

Figure 4.9 Examples of model fits to biomass time series. A) Beverton-Holt OP; B) Ricker OP; C) Beverton-Holt OP with the *Bo=K* assumption. Legend key: OP is true biomass, S is Schaefer model, F is Fox model,D is Deriso-Schnute model; and D** is Deriso-Schnute model with mis-specified stock-recruitment function.

Figure 4.8 also illustrates the consequences of misspecifying the stock-recruitment function in the Deriso-Schnute model. While biomass estimates are only slightly degraded by this misspecification, the estimates of management parameters for the Beverton-Holt OP are noticeably degraded (over +60%) when the wrong stock-recruitment function is assumed.

4.3.2.3 The $B_0$=$K$ assumption.

In general terms, using the $B_0$=$K$ strategy during estimation produces mixed results (fig. 4.10). This assumption improves all estimates of $f_{opt}$ for the Schaefer and Fox models, slightly degrades $C_{opt}$ estimates of the Schaefer model but marginally improves those of the Fox model. The $B_0$=$K$ assumption either degrades or does not improve results for the Deriso-Schnute model: it does not lead to better estimates of management parameters for the Beverton-Holt OP, and slightly degrades $f_{opt}$ estimates for the Ricker OP. On the other hand, the $B_0$=$K$ assumption leads to improved performance in biomass estimation for all models (fig. 4.9c). The bias in $B_0$ and $B_{end}$, and the total discrepancy are significantly reduced for the three estimation procedures. The $B_0$=$K$ assumption is analysed during further tests of the proportionality OP's because of its potential to improve biomass assessment estimates.

4.3.3 Changes in the spatial behaviour of the stock: hyperstability and hyperdepletion.

The hyperstability and hyperdepletion cases present a formidable problem to all fishery models. In all hyperstability cases, the three models perform dismally for the estimation of management parameters, with the exception of the Fox model estimations of $C_{opt}$ (fig. 4.11). An effort to improve management parameter estimation consisted of calculating the optimal exploitation rate (which is a more stable parameter than $f_{opt}$ because of its independence from $q$ estimates). However, this only helps to improve the performance of the Fox model with the Beverton-Holt OP. Biomass assessment is also very poor in all estimation procedures. Whilst most estimates of $B_0$ are satisfactory, this is due to the usage of the $B_0$=$K$ assumption in all hyperstability and hyperdepletion trials (section 4.2.3.2). Note that the estimates from the Deriso-Schnute model have larger uncertainties than the other models in all cases of hyperstability, suggesting a greater difficulty of the delay-difference model to cope with the extra load of estimating the additional parameter $Q$. With the

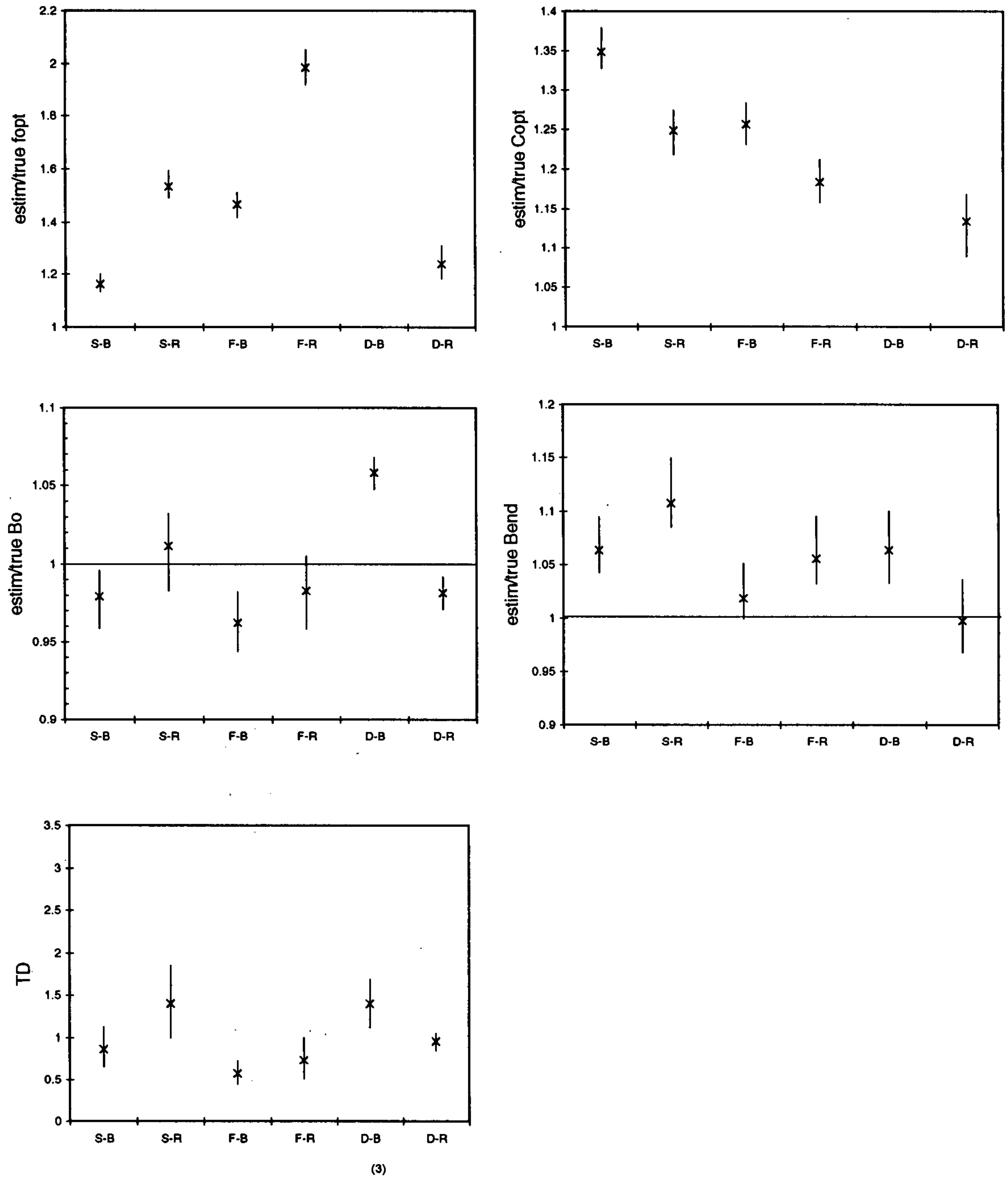



Figure 4.10 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using multiplicative observation error and the $B_0=K$ assumptions. Monte Carlo simulations performed with the proportionality OP and the high contrast effort pattern. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.

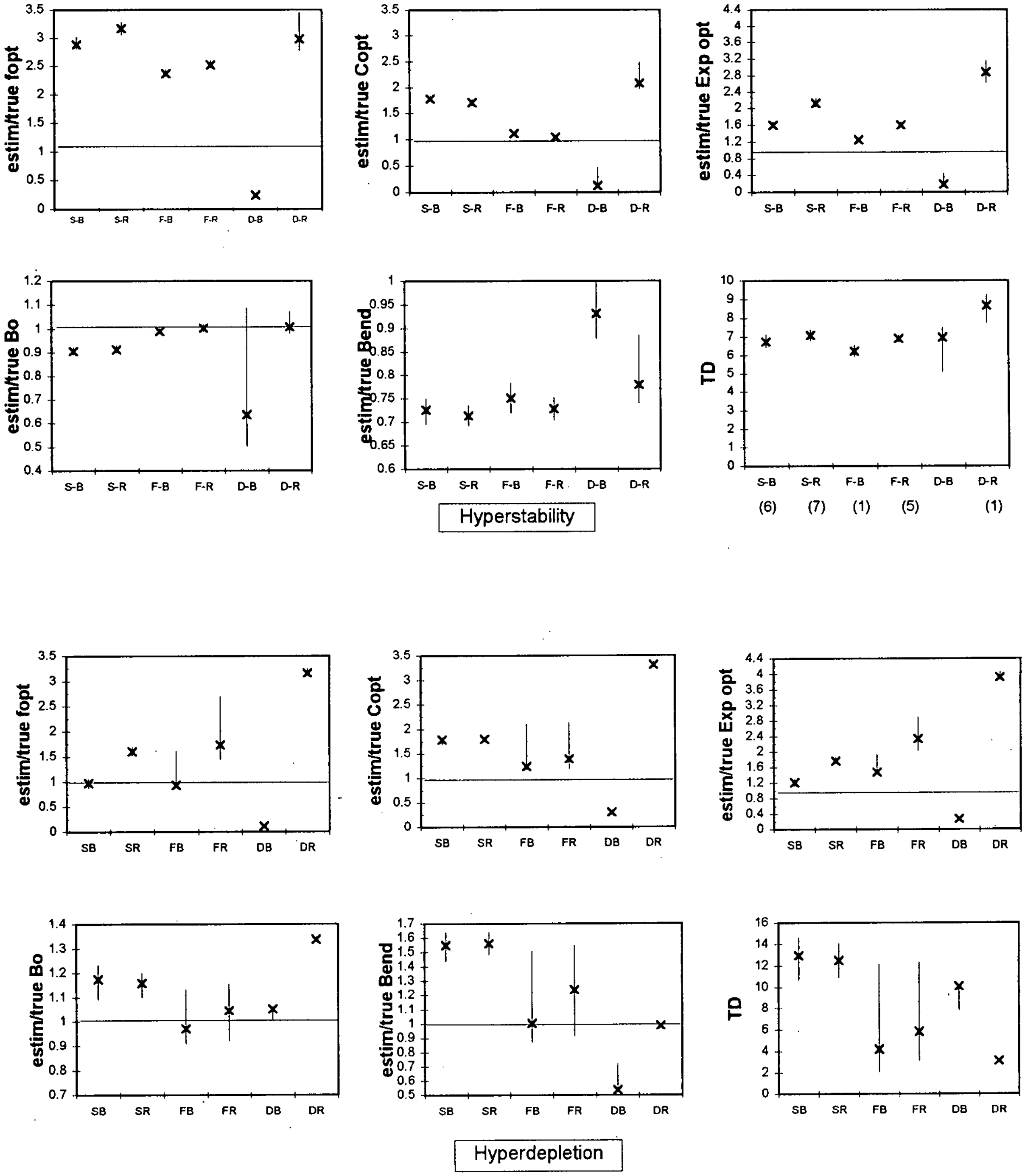



Figure 4.11 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the multiplicative observation error assumption. Monte Carlo simulations performed with the Hyperstability and Hyperdepletion OPs and high contrast effort pattern. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.. ()= number of failed trials.

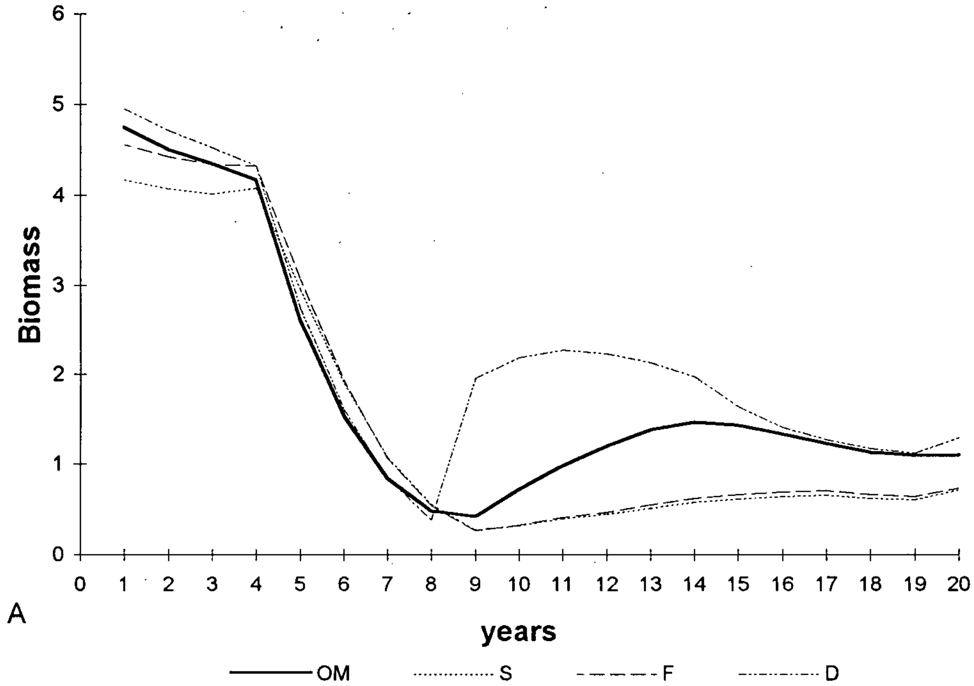
exception of the Deriso-Schnute estimates for the Beverton-Holt OP, all estimates of $B_{end}$ have relatively large negative biases. The overall poor performance of the three fishery models for biomass assessment under hyperstability conditions (fig. 4.12a) translates into very high total discrepancy values.

Management parameter estimates for the hyperdepletion OP are also very poor for all estimation models. Although the Schaefer model appears to provide good estimates of $f_{opt}$ for the Beverton-Holt OP, the corresponding $C_{opt}$ estimates are highly positively biassed. The Fox model suffers from highly imprecise estimates and the Deriso-Schnute model has the worst performance of the three models. Biomass estimation was also dismal for all models (fig. 4.12b). The Deriso-Schnute model is able to produce a good biomass fit (low *TD* score) for the Ricker OP but its performance collapses for the Beverton-Holt OP.

4.3.4 Unproductive stocks.

When the original operating populations are simulated shifting the age of entry to the fishery towards lower values (age 4 as opposed to age 7) the productivity of the fishable stock decreases. This can be seen by analysing the values of the corresponding true optimal catch and the optimal effort (table A.1). The estimation of management parameters for these unproductive OP's is best in the Deriso-Schnute and then in the Schaefer model (fig. 4.13). The Fox model performs dismally in the estimation of $f_{opt}$. The reduction in stock productivity seems to be relatively beneficial for the performance of surplus production models: in comparison with the results for the productive OP, estimates of management parameters generally improved for the surplus production models but slightly deteriorated for the delay-difference model.

Biomass assessment for the unproductive stock scenario is relatively good in all models. Overall, the Deriso-Schnute model outperforms the surplus production models, having remarkably good biomass fits (fig. 4.14a). The surplus production models had comparatively larger variability than the Deriso-Schnute model in their biomass estimates. Notably, the Ricker OP seemed to present more challenges to the surplus production models than the Beverton-Holt OP.



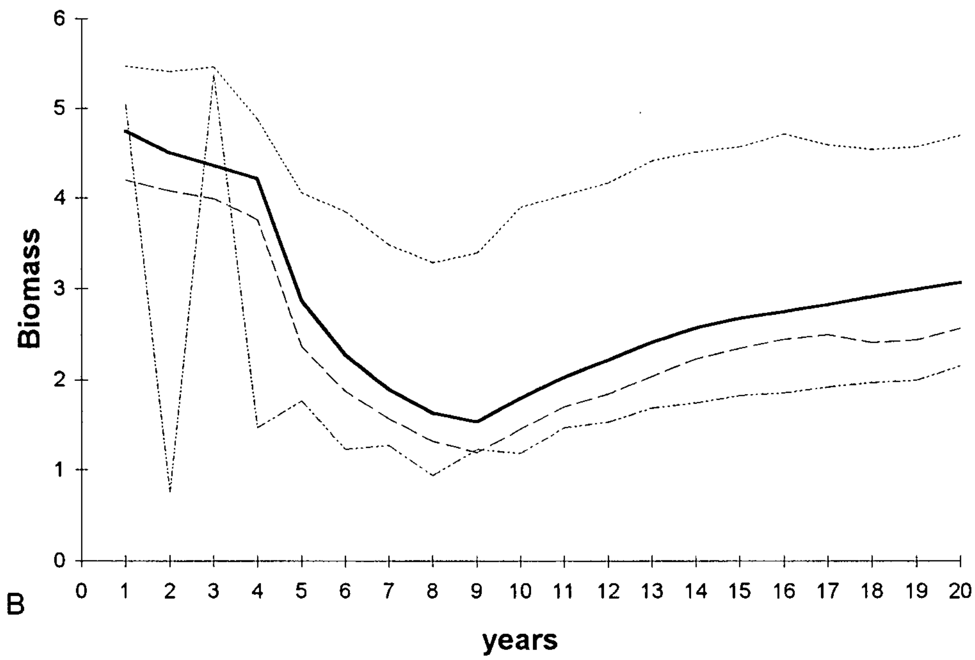




Figure 4.12 Examples of model fits to biomass time series for the Beverton-Holt OP. A) Hyperstability Case; B) Hyperdepletion Case. Legend key as in fig. 4.9. By chance, fig B shows a relatively good fit of the Fox model, however, note large uncertainty of TD values for this model in fig. 4.11.

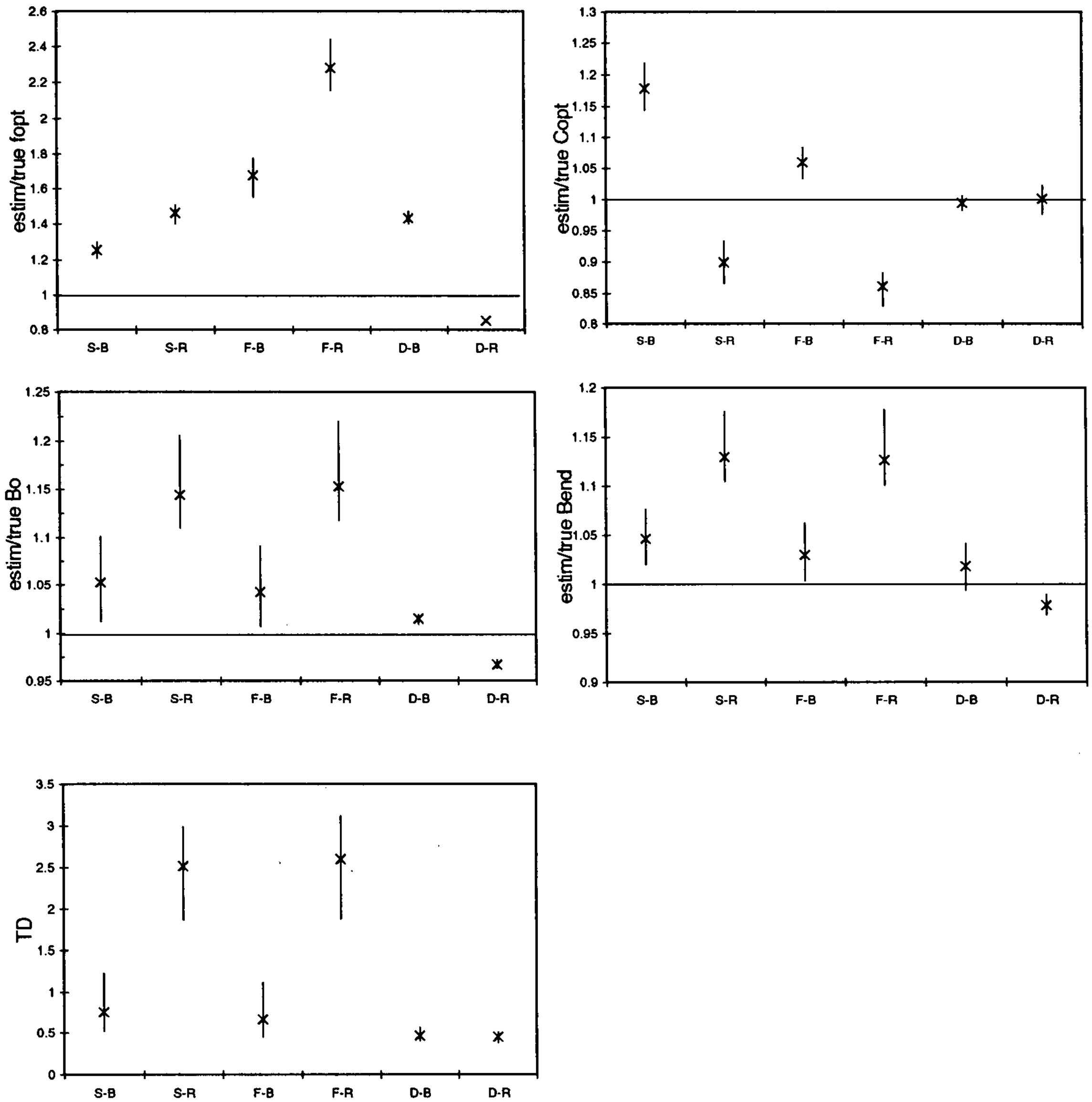



Figure 4.13 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the multiplicative observation error assumption. Monte Carlo simulations performed with the unproductive stock (age of recruitment =4yr) and proportionality OP. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.

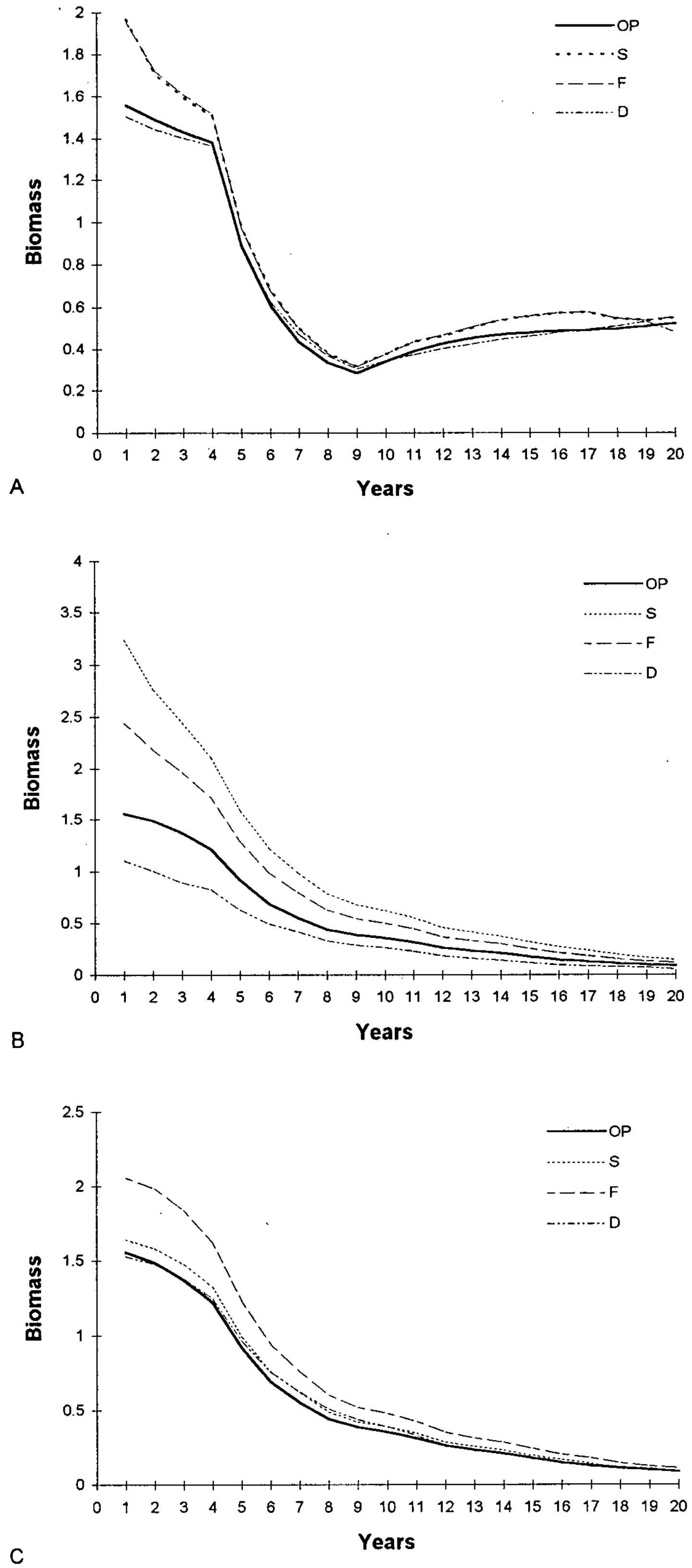



Figure 4.14 Examples of model fits to biomass time series. A) Ricker OP with low productivity; B) Same as above, with uninformative effort pattern; C) As above, with the Bo=K assumption. Legends as in fig. 4.9

Using the $B_0=K$ assumption in the unproductive stock scenario improves considerably the results of the Schaefer and Fox models but has negative effects in the performance of the Deriso-Schnute model (fig. 4.15). Management parameter estimates for both OP were acceptable for the two surplus production models but not for the Deriso-Schnute model (specially $f_{opt}$ for the Beverton Holt OP). Biomass assessment are also improved for the Schaefer and Fox models but degrade slightly for the Deriso-Schnute model (see *TD* scores).

4.3.5 Trials with uninformative data.

Figure 4.16 shows the negative effect of having data that do not provide sufficient contrast for parameter estimation. Although these data pertain to the relatively easier to fit unproductive stock OP, the performance of all models deteriorates significantly in comparison with results shown in figure 4.13. None of the three models is able to provide unbiased estimates of both management benchmarks: good estimates of $f_{opt}$ were not matched with good estimates of $C_{opt}$. The estimation of biomass suffered similar loss in performance for all models (fig. 4.14b), with some *TD* values reaching levels as high as those found for the hyperstability and hyperdepletion cases. Under this scenario, only the Deriso-Schnute model yields more or less consistent biomass fits for both operating populations.

The $B_0=K$ assumption helps to improve results sufficiently only for the Deriso-Schnute model (fig. 4.17). Estimates of management benchmarks are only within acceptable values in the Deriso-Schnute model, and in the Fox model Beverton-Holt OP. Under the difficult situations of low contrast data, this strategy helps improve most biomass estimates including those of the Deriso model (fig. 4.14c), which were previously deteriorated by this strategy in the unproductive stock with high contrast effort scenario (figs. 4.13 and 4.15). However, the improvement is not sufficient for the biomass estimates of the Fox model for the Ricker OP. When the $B_0=K$ assumption is used, once again the Deriso-Schnute model is the best estimation procedure for both operating populations performing satisfactorily for management and assessment parameter estimation.

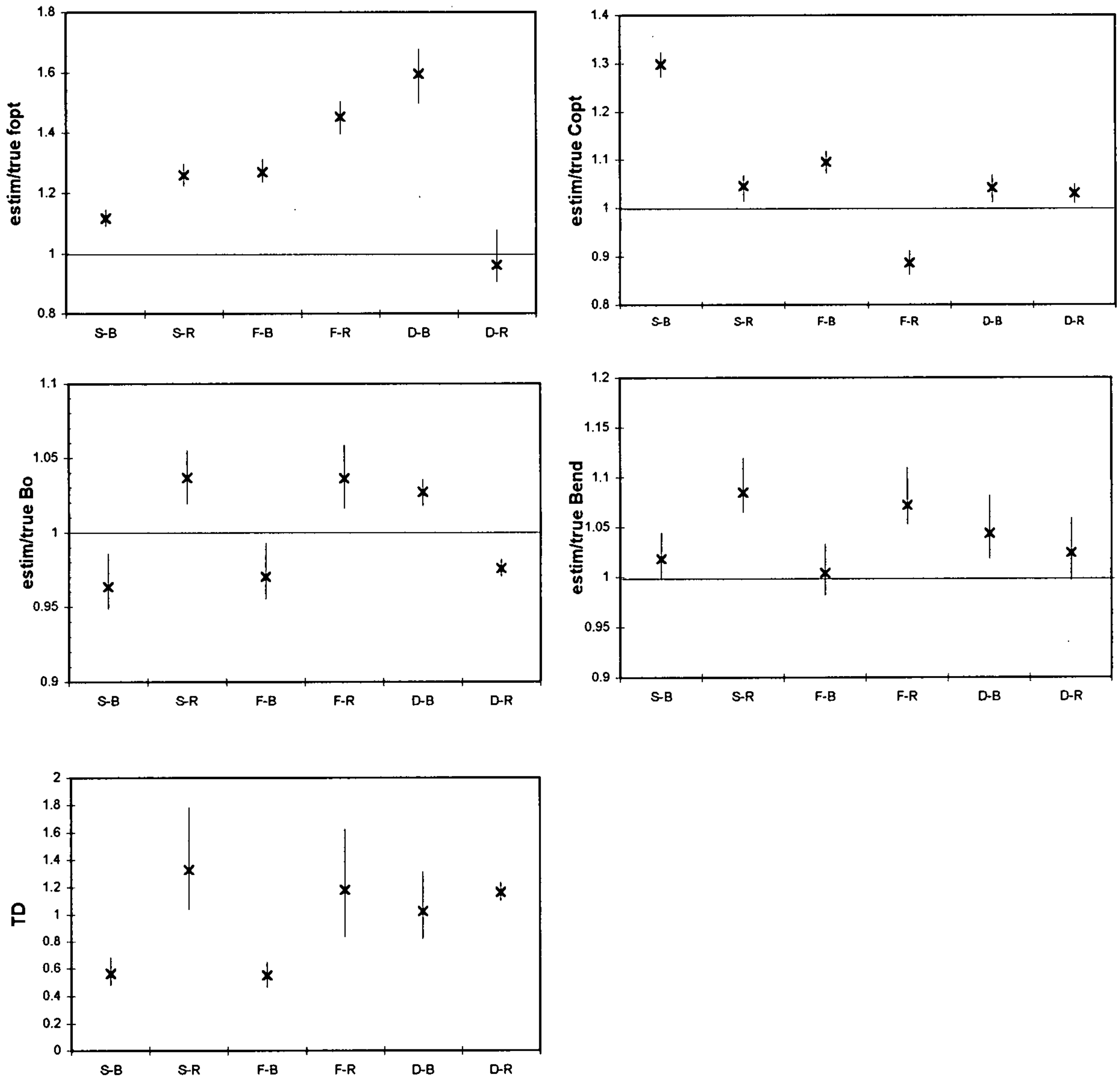



Figure 4.15 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the $B_0=K$ and the multiplicative observation error assumptions. Monte Carlo simulations performed with the high contrast effort pattern, unproductive stock (age of recruitment =4yr) and the proportionality OP. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.

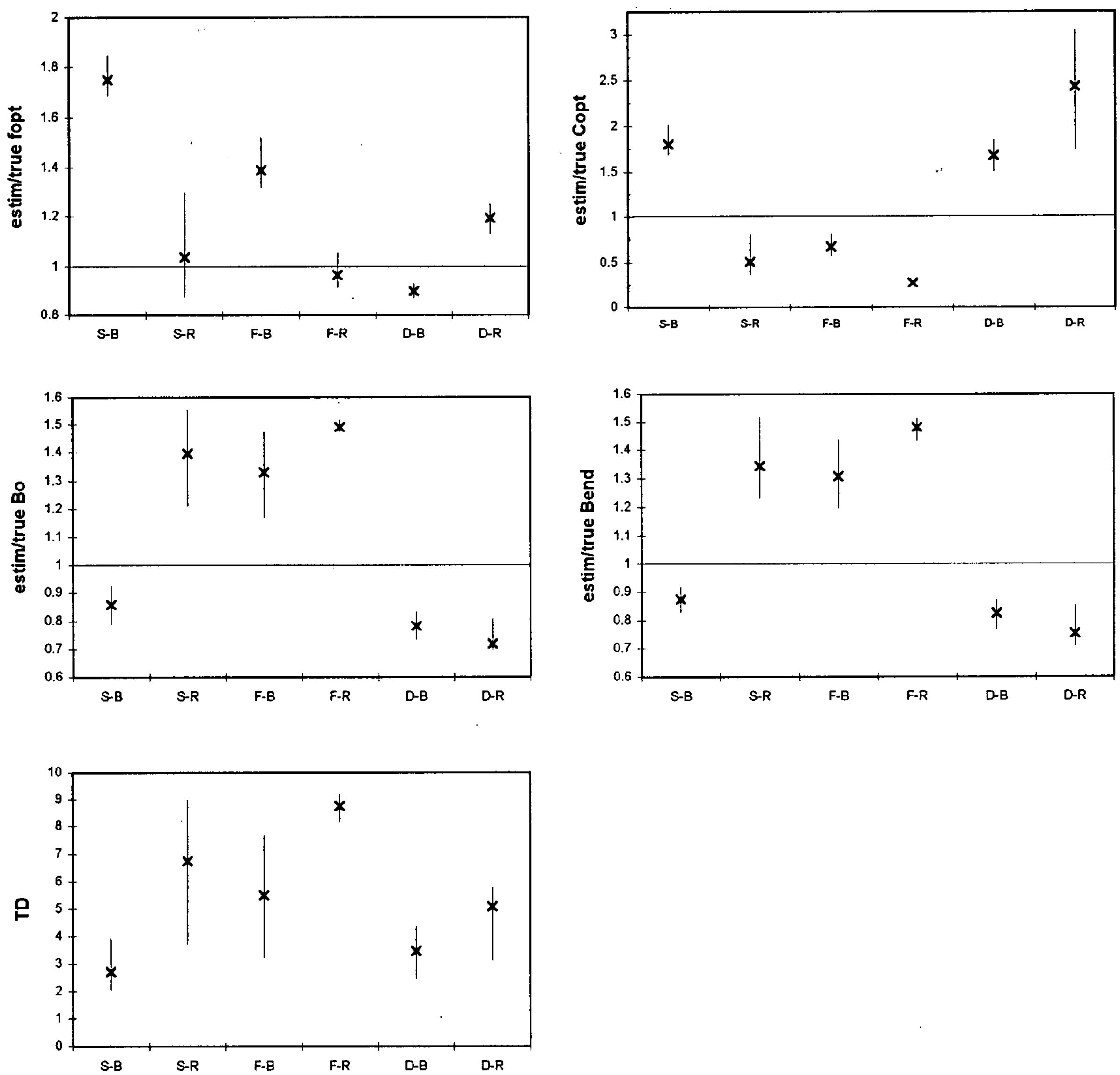



Figure 4.16 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the multiplicative observation error assumption. Monte Carlo simulations performed with the uninformative effort pattern, the unproductive stock (age of recruitment =4yr) and the proportionality OP. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.

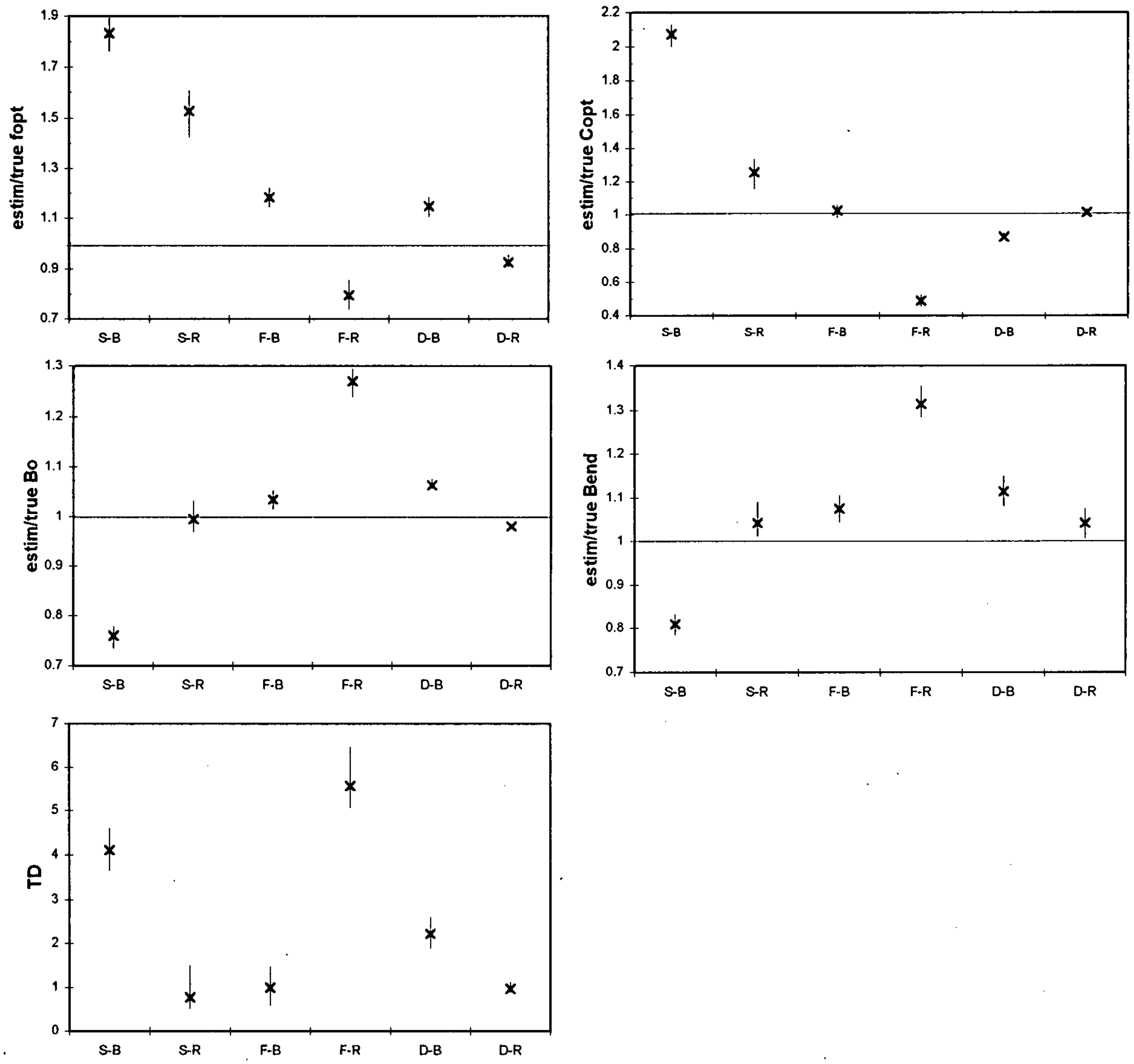



Figure 4.17 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the multiplicative observation error and the $B_0=K$ assumptions. Monte Carlo simulations performed with the uninformative effort pattern, the unproductive stock (age of recruitment =4yr) and the proportionality OP. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.

## 4.4 Discussion.

### 4.4.1 Unsuccessful estimation assumptions and model misspecifications.

The reason why the usage of additive observation errors in the estimation procedure is inferior to the multiplicative error assumption is that the additive error assumption is a misspecification that cannot properly mimic the simulated error values (see fig. 4.18). For a given value of $v_t$, equation 4.40 determines a distribution of $q$ estimates that does not match the normal curve simulated in the data.

The failure of the non-delay version of the Deriso-Schnute model highlights the importance of a conscious and careful implementation of the delay-difference model. The poor performance of the model with this misspecification, adds to the difficulties encountered during the independent estimation of the fixed parameters and other model implementation details described above (section 4.2.4.2) and demonstrates that the Deriso-Schnute model as implemented here with several fixed parameters, is prone to fail when there are model departures. The experience gained during this study allows to provide a few lines of advice to new users of this model: whenever the results from the Deriso-Schnute model are very sensitive to starting values, or when the model converges to absurd parameter estimates or cannot converge at all, there is likely to be a wrong fixed parameter value or some misspecification in the model. Unfortunately, as it has been shown here, there are also many instances in which wrong specifications of the model will provide some kind of answers, usually grossly biassed, without any way to diagnose this situation. Notably, misspecification of stock-recruitment relationship is likely to happen in many instances while working with real fish stocks, with potentially negative consequences on the estimation of management parameters. As there is no way to detect these cases from a simple analysis of the results, it is recommended to do a careful determination of the parameters to be fixed, before implementation of the model .

### 4.4.2 Changes in the CPUE-biomass relationship.

None of the three estimation procedures examined here is able to cope with the complexities of the hyperdepletion and hyperstability situations. This is in agreement with

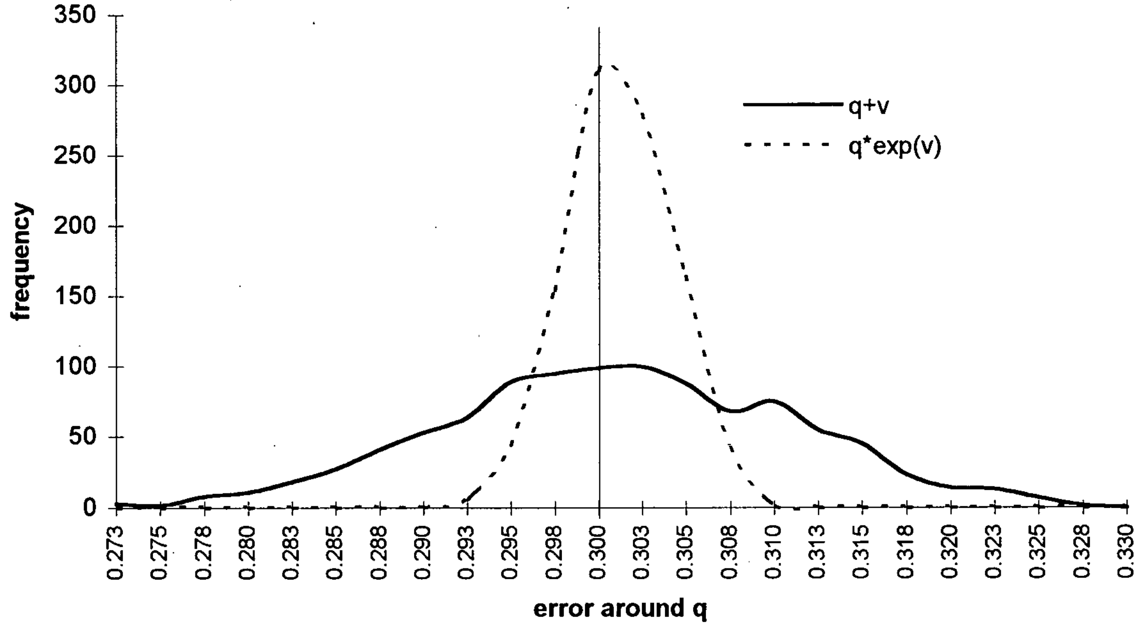



Figure 4.18 Distribution of errors around a *q* value of 0.3, according to the two observation error assumptions considered in the study: additive and multiplicative. All data are simulated with the multiplicative model.

similar studies that have shown the virtual impossibility of finding values of the extra parameter Q which determines the degree of hyperdepletion or hyperstability (Reed and Simmons 1994). The inability of models to correctly estimate this value is due to parameter confounding. In the present context, this means that there is simply not enough information in the data for the model to separate the effects of Q from the effects of parameters that determine the shape of the production function and the total stock size. The only solution to this problem is to provide the models with additional information. Fournier and Doonan (1987) successfully fitted a generalised Deriso-Schnute model to hyperdepletion and hyperstability data but they used a simpler parameterisation, additional information on average weight and length, and a Bayesian approach for their estimations. The key to their success however, is the additional information embedded in the average weight and length data. In the light of the work of Fournier and Doonan, delay-difference models seem to be a promising approach for situations of biomass-dependent catchability when more information than just total catch and an abundance index are available for the assessment.

4.4.3 Performance of the estimation procedures, and choice of the best model.

The performance of the surplus production models during the present study indicates their inability to produce unbiased estimates of management parameters for a simulated age-structured shark population. Perhaps more importantly, surplus production models show a persistent (and frequently large) positive bias in their estimates of management parameters over a wide range of situations. There was only one instance where both models performed reasonably well for management parameter estimation, the unproductive OP with high contrast data and the $B_0=K$ assumption.

There is also a striking inconsistency in the behaviour of surplus production models when different stock-recruitment functions are used. Changes from a Beverton-Holt to a Ricker OP caused a shift on the relative accuracy for biomass estimation from surplus production models. This shift is characterised by a tendency towards larger estimates, frequently leading to greater positive biasses and consequently higher *TD*'s. This was generally accompanied by a worsening of the estimates of management parameters, especially $f_{opt}$, except for the uninformative data scenario. It is unclear if the different scale of each operating population was responsible for this behaviour. The Schaefer model was able to correctly estimate management parameters for most of the Beverton-Holt OP's (except when data contrast was poor) but failed very frequently with the Ricker OP's. This suggests the Schaefer model could be used for rough estimates of management parameters for elasmobranch populations in situations of high contrast in the fishery data, provided there is a fairly good indication that the stock-recruitment relationship indeed follows a Beverton-Holt curve.

The present study seems to be the first of its kind specifically oriented towards shark populations, however it can be contrasted against other investigations comparing the applicability of surplus production models to age-structured populations. My results are in conflict with findings from Ludwig and Walters (1985) who report that a generalised Ricker surplus production model performs remarkably well for $f_{opt}$ estimation using data generated by the Deriso-Schnute model. Punt (1988, 1991) also reports surplus production models as the most reliable for management of Cape hake resources simulated with a fully age-structured operating population. Similar positive results are reported with a Schaefer estimator for a swordfish age-structured simulation model (Prager et al. 1994).

The divergence of the above studies with my results reopens an interesting question: are the views of Holden about model inadequacy responsible for the failure of surplus production models observed here? Unfortunately, this question cannot be solved with a simple comparison between the present study and results from the literature. The operating populations, fishery models, estimation procedures, minimization algorithms and the standards for accepting/rejecting models are different in each case, making difficult to understand the true reasons for their diverging results. Ludwig and Walters (1985) use only a process error estimator, later found to be grossly biassed (Ludwig et al. 1988, Polachek et al. 1993), whilst Prager et al. (1994) used a deterministic operating population and an observation error estimator. In order to investigate if surplus production models are inadequate for shark-like populations, a better approach would be to perform parallel simulations with similar age-structured operating models, one simulating a typical shark population (long lived, slow growing, low fecundity, etc.) the other simulating a typical bony fish population (fast growth, early maturity, high reproductive rate, etc.), and using exactly the same estimation procedures, contrast in data and techniques for model fitting.

There seems to be more hope for the usage of surplus production models for shark biomass assessment than for management parameter estimation. Although generally inferior to the Deriso-Schnute model, both surplus production models are capable of estimating biomass benchmarks and obtaining good biomass fits for most of the scenarios analysed (excluding hyperstability and hyperdepletion). Notably, the Fox model performs slightly better for this task than the Schaefer model. The $B_0$=$K$ assumption proved to be a good remedy for the performance of surplus production models under the difficult situations of uninformative data. In this context, it seems adequate to use either of the surplus production models, for the assessment of shark stocks when data for the Deriso-Schnute model is not available or the model is not applicable (i.e. multispecies fisheries).

Results presented here indicate that the Deriso-Schnute model is by far the best choice for biomass assessment as well as for the estimation of management parameters in shark-like populations. This model consistently outperformed the Schaefer and Fox models for both Ricker and Beverton-Holt OP's. Moreover, its robustness (under the ideal conditions of a simulation study) was evidenced by its superior performance under different scenarios of stock productivity and quality of fisheries data.

The clear superiority of the Deriso-Schnute model over surplus production models for the estimation of management benchmarks seems surprising under the light of findings from Ludwig and Walters (1985). According to their study, the Ricker surplus production model outperforms the Deriso-Schnute model for the estimation of optimal effort even when data are generated with the latter model. Punt (1988, 1991) also found that surplus production models perform better than the Deriso-Schnute model for fisheries management parameter estimation. Leaving aside the speculation that perhaps the Ricker surplus production model is also far superior to the models of Schaefer and Fox, this apparent contradiction highlights the limits of simulation results. Each study is valid under the particular conditions where it was carried out, and extrapolations must be done very cautiously. Undoubtedly, tackling this problem requires performing a study of very large proportions, in which all kinds of fishery models and estimation methods are tested at once under the same conditions. Ideally, such a study would have to be done every time a new stock or resource needs to be assessed and managed. For the particular case of elasmobranch fisheries, this is probably the subject of a whole dissertation analogous to the work of Punt (1988) for South African hake resources. Another solution, offering the advantage of training and communication among different parties, is to organise this work as an exercise for a whole group of fishery biologists and managers during a focused workshop. Ultimately, checking the of validity of the present results with more tests would emphasise the views of Hilborn and Walters (1992) that it should be customary to routinely perform Monte Carlo analyses of individual assessments, and to concentrate efforts not in finding a good fit to the data but in finding all possible alternative models (and parameters) that could explain such data equally well.

4.4.4 Significance for the assessment and management of real shark fisheries .

The results from the present study might mislead the uncritical reader. Even though the Deriso-Schnute model outperformed both surplus production models throughout most of the study, this does not guarantee that the Deriso-Schnute model will provide more accurate and reliable estimates of biomass and management benchmarks than surplus production models under a real situation. The apparent robustness and reliability of the model seems to be challenged by its sensitivity to wrong choice (estimation) of fixed parameters, particularly the age of entry to the fishery, highlighted during the early exploratory trials mentioned at the end of section 4.2.4.2. The results of the stock-recruitment function misspecification (section 4.4.1) also cast doubt on the applicability of this model in real situations.

The main problem is that the controlled conditions under which the Deriso-Schnute model has been tested here will hardly be met in a real fishery. The parameters known 'a priory' (actually estimated or 'stolen' from the simulated data) and fixed here for the delay-difference model are precisely some of the most uncertain in most real sharks stocks and many fish populations: the value of natural mortality, the type of stock recruitment function, and the exact age of entry to the fishery. Unfortunately, slight departures from the real values proved to be bad for estimation during the present tests. In all likelihood, real applications of the Deriso-Schnute model will be vulnerable to large bias in the estimation of management parameters because of uncertainties in the fixed parameter values. In coincidence with the present study, Fournier and Doonan (1987) find that errors in the estimation of the age of recruitment to the fishery can badly bias the results from delay-difference models, specially when this age is underestimated. Good determinations of the underlying type of stock-recruitment relationship are also needed before applying the Deriso model to real fisheries.

### 4.4.5 The $B_0=K$ strategy.

The effect of using the $B_0=K$ strategy during model fitting was not clear in the present study. The benefits of this assumption were not generalised for the three models, nor were they consistent across the different types of trials. Furthermore, this assumption produced negative effects in some cases.

These results are in agreement with previous findings. Punt (1988) reports a surplus production model with the $B_0=K$ strategy as the optimal model to estimate management parameters for a hake fishery. However, analysis of his tabulated results reveals that for a few cases the strategy was detrimental. Prager et al. (1994) found contradictory results and did not reach any conclusions from analyses of a similar constrain on $B_0$. The unpredictability of the effect of the $B_0=K$ strategy, suggests a cautious approach for its usage. A thorough examination of its effects for the particular system under study via a Monte Carlo analysis is imperative.

### 4.4.6 Summary and conclusions.

The present study provides insight into several areas of fisheries assessment and management: the applicability of surplus production models and delay-difference models to shark populations and their robustness under different scenarios; the applicability of surplus production models to age-structured populations; the effects of different assumptions about the form of the observation error on estimation; the sensitivity of the delay-difference model to different model misspecifications; and the ability of the three models to estimate parameters in the absence of proportionality between CPUE and biomass, and under situations of uninformative data.

In addition, these results are a starting point for further developments on the assessment and management of elasmobranch fisheries. Future work could build upon the findings presented here by pursuing more detailed/comprehensive studies in specific areas highlighted through this work and abandoning futile approaches that have been shown here to be unsuccessful.

None of the models analysed here should be used for estimation of stock size or management parameters under situations of suspected hyperdepletion or hyperstability when only catch and abundance data are available. The performance of the estimation procedures under this situations can lead to grossly biassed parameter estimates.

The usage of surplus production models, in particular the Fox model, for the estimation of management parameters in elasmobranch populations is not recommended. Prime reasons for this are large positive biasses in parameter estimation and lack of robustness.

Although the Schaefer and Fox surplus production models are not recommended for management parameter estimation, they can be used more confidently for the rough estimation of stock size in shark fisheries that meet the proportionality requirement. For situations of uninformative fishery data, the $B_0=K$ strategy could help provide better biomass estimates.

The Deriso-Schnute model is the most promising technique from those analysed here, for the estimation of assessment and management parameters in shark-like fisheries. This model provides the least biassed estimates of management and biomass benchmarks under most scenarios tested. This, however, is conditioned to stringent requirements: catchability should remain constant as abundance changes, and there must be a very good knowledge of life history parameters for the stock (i.e. parameters to be fixed in the model).

# CHAPTER 5

# ELASMOBRANCH STOCK ASSESSMENT AND MANAGEMENT IN THE REAL WORLD:
# THE MULTISPECIES SHARK FISHERY OF YUCATAN, MEXICO.

## 5.1 Introduction.

The practice of elasmobranch fisheries assessment frequently faces a large set of problems. First, there are many areas in which our knowledge of sharks is very limited, specially in relation to key population dynamic processes. Secondly, there is very limited practical experience in relation to proven methods for elasmobranch fisheries assessment. Additional problems seldom considered when discussing shark management are the definition of the size, distribution and dynamics of unit stocks, as well as the long-range seasonal migrations (specially in the case of large sharks) and other spatial dynamics. However, some of the problems related to population dynamics and model choice for elasmobranch fisheries assessment have already been discussed in previous chapters.

Here I focus on a discussion of some practical problems likely to be found when attempting to do assessments of real elasmobranch fisheries, especially in the developing world (which produces 2/3 of the total world elasmobranch catch), and I suggest some alternatives for overcoming these problems. The discussion is led by an analysis of data from a real case study, the multispecies shark fishery of Yucatán, México. A secondary objective is thus to do an assessment of this fishery, which is characterised by a typically poor database. As it happens, the only management recommendations that can be made at this stage are those aimed at improving our ability to do an assessment of this fishery. The measures suggested here can serve as general guidelines for solving similar problems likely to occur in many other tropical fisheries for elasmobranchs.

5.2 The shark fishery of Yucatan: a typical case study.

The overview presented in chapter 2 illustrates how common it is for most shark and ray fisheries throughout the world to have a very limited pool of information for assessment purposes. In general, the only statistics available are aggregated catches of shark or ray species (in the worse cases they are both lumped together) and these usually contain no further detail (i.e. spatial information). Catch per unit effort (CPUE) or fishery-independent abundance indices are very hard to find for elasmobranch fisheries, not only in the developing world but on a global basis (Bonfil 1994). The shark fishery of Yucatan is a pretty typical example of the above situation, and thus the shortcomings found for the assessment of this fishery probably mirror the problems to be encountered in many shark or ray fisheries in the world.

The shark fishery of Yucatan has been described by Bonfil et al. (1988, 1990). There are approximately 25 species in the catches but seven are the most important. Several fleets contribute to the total shark catch: a small-scale gillnet fishery spread along the coast makes up a large proportion of the landings; important catches are also taken incidentally in the large-scale hook and line finfish fisheries based at Progreso; finally a few vessels from the large-scale fishery contribute additional catches while exclusively fishing for sharks with gillnets.

The reliable record of shark catches in Yucatan goes back only to 1977, when the Ministry of Fisheries was established in Mexico. However, Bonfil et al. (1990) assembled historical records back to 1956 based on the application of conversion factors to different shark by-products from several sources of governmental statistics. This historical record indicates that shark catches were relatively small before 1982, seldom reaching more than 600 t/y. Catches grew very rapidly after 1980 and have been varying around some 2,100 t/y since 1982 (fig. 5.1).

The application of VPA and related methods for the analysis of this fishery is not feasible at the moment because only one shark species from Yucatan has been aged (but not validated) and there are no time series data on catch-at-age for any shark species. In fact, the single species approach might not be the appropriate procedure for this highly multispecific fishery in which setting management regimes for one species without affecting the whole of the fishery might be difficult to implement. In any case, the lack of catch and abundance information for individual species precludes the application of even the most simple single species models.

The above characteristics of lack of detailed information by species, high species diversity in the catches, and multiple types of gears and vessels involved in the fishing activity, are commonplace in most of the tropical elasmobranch fisheries of the world. There is however one type of information that is readily available in Yucatan, and probably elsewhere: aggregated total catch data for sharks by year. If some kind of abundance index could be found, then a rough assessment is possible by applying dynamic surplus production models to these aggregated-species data. Although aggregated-species surplus production models have several pitfalls (see Pauly 1979) they provide a useful first (and sometimes the only possible) approximation of the status of the stocks and have been used before, among others, by Brown et al. (1976) for fisheries of the Georges Bank, by Ralston and Polovina (1982) for handline fisheries of Hawaii, by ICSEAF for the management of Cape hake resources of southwestern Africa (Hilborn and Walters 1992), and Polacheck et al. (1993) for hakes off Namibia.

I obtained information on shark catches per fishing trip from two fishing companies that allowed access to their internal records and payment slips. Two time series of CPUE were thus built for two neighbouring locations of the eastern Yucatan coast. Data from El Cuyo covered the years 1983-1988, whereas data from Rio Lagartos were for the years 1988-1992 (fig. 5.2). The proximity of the two localities and the radius of action of these artisanal vessels indicates that both communities share a common fishing ground (fig. 5.3). Thus, it was in principle plausible to try to paste both series together to form a single time series of 10 yr. This new time series of CPUE could then be used as an index of abundance for the multispecies stock after making the usual leap of faith of assuming that there is proportionality between commercial CPUE and fish abundance.

## 5.3 Methods.

In order to join together the two time series of CPUE, the data first had to be standardised.

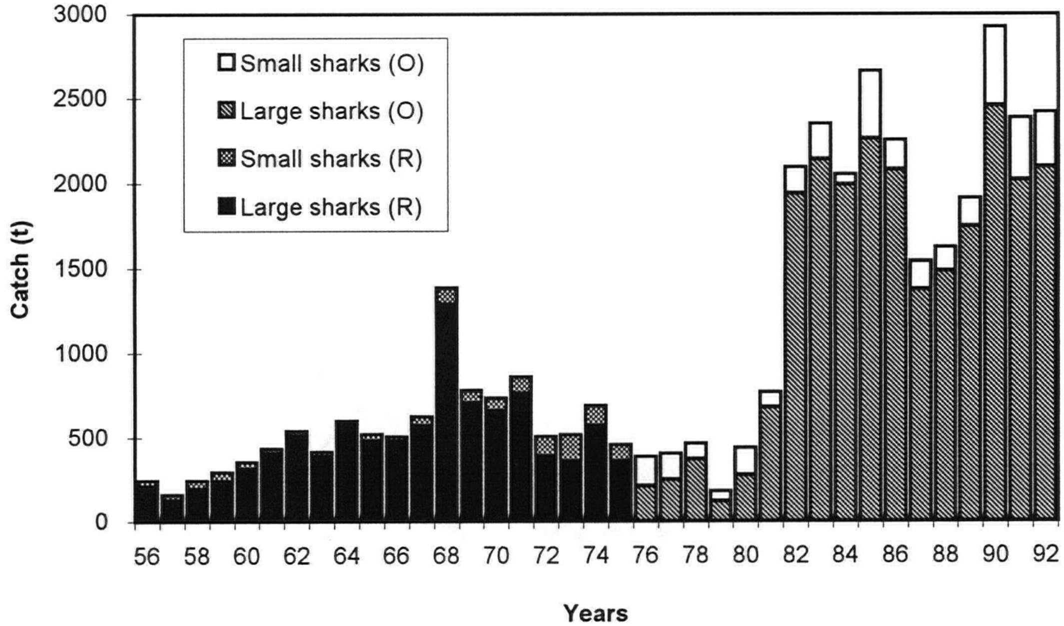



Figure 5.1. Historical catches of the Yucatan shark fishery. R = Reconstructed from shark by-products; O = Official data, Ministry of Fisheries (25 species of which 7 are common).

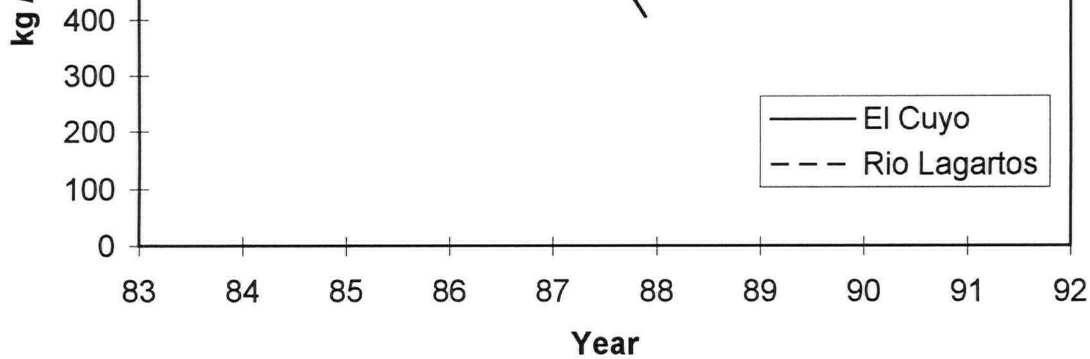



Figure 5.2. CPUE time series for shark fisheries in two neighbouring localities of Yucatan.

This was done by obtaining a better measure of fishing effort for both time series, and then proceeding to standardise the data. Interviews with fishermen in both localities revealed that while a typical vessel in Rio Lagartos used gillnets totalling 16 panels of net of approx. 40 fath. each, El Cuyo vessels used only eight of these panels per boat during the mid 1980's. By translating the CPUE for each locality from catch-per-trip to catch-per-net, and then standardizing the two time series, these matched perfectly (fig. 5.4). The resulting composite historical data set for the fishery is characterised by an increasing trend in CPUE during the first two years, followed by a decreasing CPUE. The terminal four years of data show no trend in CPUE.

A dynamic Schaefer surplus production model was fitted to the time series of total shark catches for Yucatan and these CPUE data, using the total least squares method (Ludwig et al. 1988). The technical details of this procedure are given in the previous chapter (sec. 4.2.6)

Two sets of trials were performed. In the first set, the Schaefer model was fitted to the entire time series. For the second set of trials, the first two years of CPUE data (increasing CPUE) were left out of the analysis. A growing CPUE early in the time series can be explained in the present context by several alternative hypotheses: i) a period of low effort that allows stock recovery after heavy exploitation; ii) an open system where there is net immigration; iii) absence of proportionality between CPUE and biomass; or iv) errors early in development of the catch reporting system. Available data suggest that the effort levels were not low in this fishery during the phase of growing CPUE and that the stock had not been fished hard in prior times, so the possibility of a recovering stock is excluded. Alternatives ii) and iii) violate the assumptions of surplus production models and could therefore not be evaluated in the analysis. On the other hand, the reporting system of the fishing companies which provided the data had been in place for many years without change, and as their records were kept for book-keeping purposes, they are thought to be reliable.

## 5.4 Results.

Table 5.1 shows the results of the first set of trials using the complete time series of CPUE and catch. Attempts to fit the Schaefer model to these data produced highly uncertain

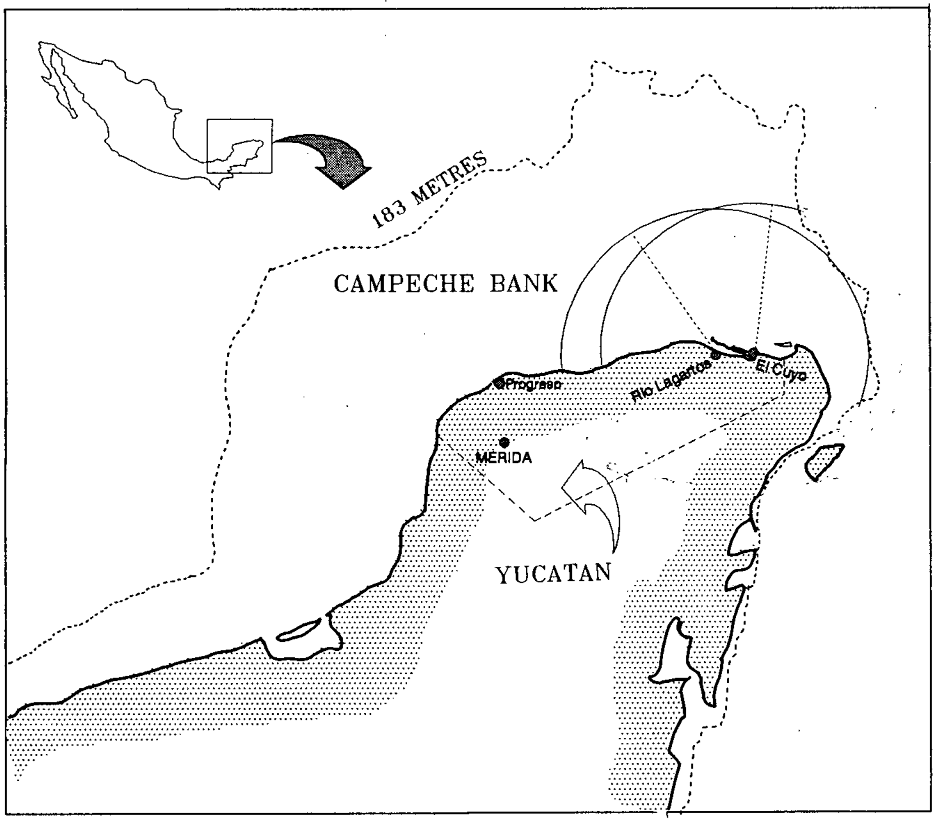



**Figure 5.3 Aproximate range of action of the artisanal shark fisheries of Rio Lagartos and El Cuyo. Additional constrains are imposed by depth. Deployment of nets is not possible beyond 75 m.**

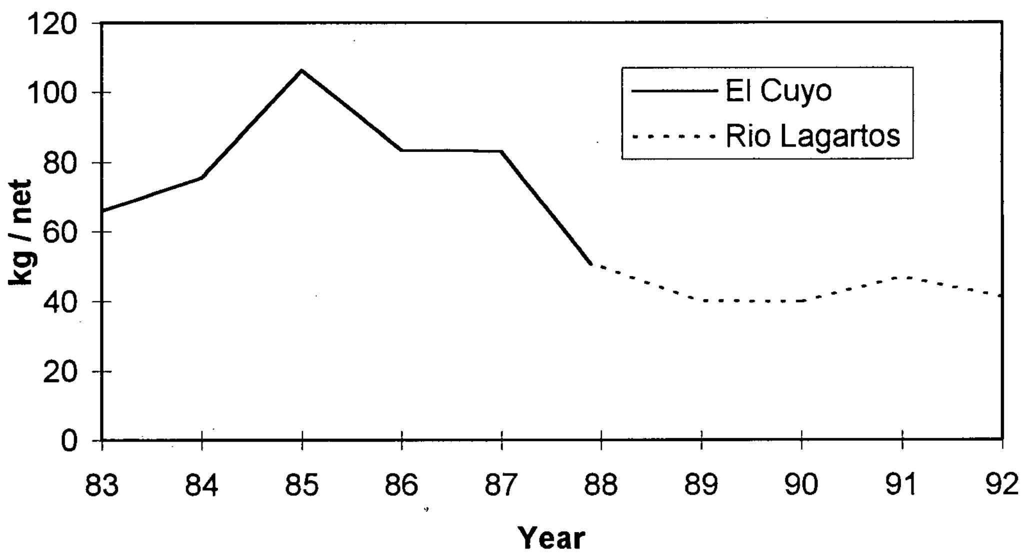



Figure 5.4. Standardized CPUE time series for shark fisheries in two neighbouring localities of Yucatan.

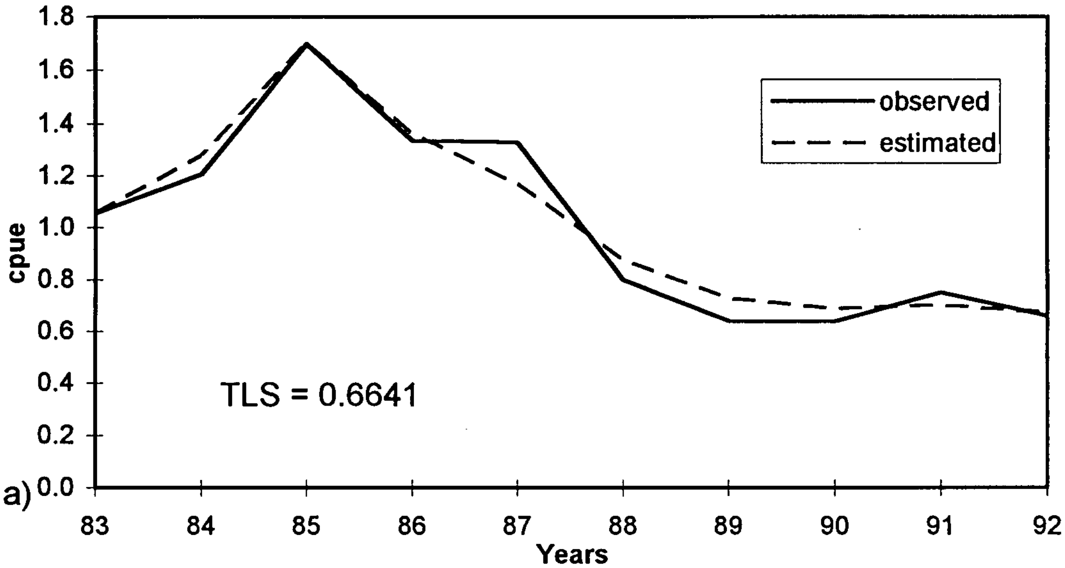
Table 5.1. Results of first set of trials: Total Least Squares (TLS) fits of the Schaefer model to CPUE and catch data for the 10 yr time-series from the Yucatan shark fishery.

| Trial | ---Starting values------ | | | -------Parameter estimates----------- | | | | TLS | |
|---|---|---|---|---|---|---|---|---|---|
| number | r | K | q | r | K | q | Bo | score | Constraints |
| 1.1 | 0.2 | 25 | 0.01 | 0.019561 | 35.739452 | 0.009810 | 122.580 | 0.452700 | none |
| 1.2 | 0.03 | 30 | 0.01 | 0.002883 | 20.234309 | 0.002786 | 424.480 | 0.449500 | none |
| 1.3 | 0.3 | 30 | 0.01 | 0.002883 | 20.234309 | 0.002786 | 424.480 | 0.449500 | none |
| 1.4 | 0.3 | 50 | 0.005 | 0.028215 | 60.901740 | 0.006933 | 171.000 | 0.453100 | none |
| 1.5 | 0.2 | 100 | 0.01 | 0.000210 | 1.454529 | 0.002764 | 430.980 | 0.448767 | none |
| 1.6 | 0.2 | 25 | 0.01 | 0.200027 | 41.026883 | 0.027499 | 39.576 | 0.664143 | none |
| 1.7 | 0.25 | 10 | 0.1 | 0.878897 | 10.000000 | 0.138291 | 9.350 | 0.546733 | K frozen |
| 1.8 | 0.25 | 15 | 0.1 | 0.523457 | 15.000000 | 0.092199 | 13.700 | 0.515239 | K frozen |
| 1.9 | 0.523 | 20 | 0.092 | 0.347191 | 20.000000 | 0.066545 | 18.730 | 0.496242 | K frozen |
| 1.10 | 0.347 | 25 | 0.066 | 0.233732 | 25.000000 | 0.049758 | 24.850 | 0.483547 | K frozen |
| 1.11 | 0.2 | 25 | 0.05 | 0.200000 | 32.593812 | 0.035117 | 34.580 | 0.475505 | r frozen |
| 1.12 | 0.15 | 25 | 0.1 | 0.150000 | 39.000853 | 0.026613 | 45.400 | 0.469012 | r frozen |
| 1.13 | 0.15 | 25 | 0.09 | 0.476546 | 16.026933 | 0.090000 | 14.050 | 0.512209 | q frozen |
| 1.14 | 0.1 | 15 | 0.12 | 0.693299 | 12.007724 | 0.120000 | 10.720 | 0.533486 | q frozen |
| 1.15 | 0.05 | 50 | 0.05 | 0.215046 | 26.611502 | 0.050000 | 24.700 | 0.482166 | q frozen |
| 1.16 | 0.3 | 10 | 0.1 | 0.000000 | 51.030000 | 0.028000 | ------ | 0.404230 | Bo=K |
| 1.17 | 0.3 | 25 | 0.03 | 0.000000 | 50.925988 | 0.028467 | ------ | 0.404234 | Bo=K |
| 1.18 | 0.05 | 15 | 0.15 | 0.000000 | 51.149185 | 0.028333 | ------ | 0.404236 | Bo=K |
| 1.19 | 0.5 | 20 | 0.05 | 0.000000 | 50.952946 | 0.028446 | ------ | 0.404236 | Bo=K |
| 1.20 | 0.5 | 40 | 0.1 | 0.000000 | 40.000000 | 0.036652 | ------ | 0.413799 | Bo=K, K frozen |
| 1.21 | 0.1 | 75 | 0.01 | 0.000000 | 75.000000 | 0.018953 | ------ | 0.412556 | Bo=K, K frozen |
| 1.22 | 0.5 | 75 | 0.01 | 0.000000 | 75.000000 | 0.018953 | ------ | 0.412556 | Bo=K, K frozen |
| 1.23 | 0.3 | 35 | 0.01 | 0.082005 | 35.000000 | 0.041765 | ------ | 0.408888 | Bo=K, K frozen |
| 1.24 | 0.3 | 40 | 0.01 | 0.000000 | 40.000000 | 0.036478 | ------ | 0.410514 | Bo=K, K frozen |
| 1.25 | 0.3 | 45 | 0.01 | 0.000000 | 45.000000 | 0.032373 | ------ | 0.405681 | Bo=K, K frozen |
| 1.26 | 0.3 | 50 | 0.01 | 0.000000 | 50.000000 | 0.029017 | ------ | 0.404261 | Bo=K, K frozen |
| 1.27 | 0.3 | 50 | 0.05 | 0.131621 | 29.760354 | 0.050000 | ------ | 0.413875 | Bo=K, q frozen |
| 1.28 | 0.1 | 15 | 0.05 | 0.131207 | 29.768486 | 0.050000 | ------ | 0.413875 | Bo=K, q frozen |
| 1.29 | 0.3 | 50 | 0.03 | 0.000000 | 48.570130 | 0.030000 | ------ | 0.404439 | Bo=K, q frozen |
| 1.30 | 0.3 | 50 | 0.045 | 0.099888 | 32.884498 | 0.045000 | ------ | 0.410625 | Bo=K, q frozen |
| 1.31 | 0.15 | 100 | 0.05 | 0.150000 | 30.859003 | 0.047192 | ------ | 0.413867 | Bo=K, r frozen |
| 1.32 | 0.05 | 100 | 0.05 | 0.050000 | 42.219089 | 0.034365 | ------ | 0.406138 | Bo=K, r frozen |
| 1.33 | 0.025 | 45 | 0.03 | 0.025000 | 46.055521 | 0.031496 | ------ | 0.404955 | Bo=K, r frozen |
| 1.34 | 0.025 | 47 | 0.03 | 0.000000 | 50.909983 | 0.028478 | | 0.404233 | Bo=K |
| 1.35 | 0.2 | 100 | 0.001 | 0.000000 | 361.720456 | 0.003638 | ------ | 0.467299 | Bo=K |
| 1.36 | 0.3 | 10 | 0.01 | 0.850560 | 10.000000 | 0.138877 | ------ | 0.523538 | Bo=K, K frozen |
| 1.37 | 0.3 | 15 | 0.01 | 0.472999 | 15.000000 | 0.097656 | ------ | 0.479044 | Bo=K, K frozen |
| 1.38 | 0.3 | 20 | 0.01 | 0.309643 | 20.000000 | 0.073396 | ------ | 0.437818 | Bo=K, K frozen |
| 1.39 | 0.3 | 25 | 0.01 | 0.198556 | 25.000000 | 0.059078 | ------ | 0.422255 | Bo=K, K frozen |
| 1.40 | 0.3 | 100 | 0.1 | 0.441921 | 15.514177 | 0.100000 | ------ | 0.461611 | Bo=K, q frozen |
| 1.41 | 0.3 | 10 | 0.1 | 0.000015 | 44.885062 | 0.000006 | 195534 | 1.197700 | none |
| 1.42 | 0.3 | 5 | 0.5 | | | failed trial | | | none |
| 1.43 | 0.1 | 7 | 0.1 | | | failed trial | | | none |
| 1.44 | 0.05 | 25 | 0.5 | | | failed trial | | | none |
| 1.45 | 0.3 | 10 | 0.05 | | | failed trial | | | none |
| 1.46 | 0.3 | 10 | 0.01 | | | failed trial | | | none |
| 1.47 | 0.15 | 100 | 0.3 | | | failed trial | | | Bo=K |
| 1.48 | 0.1 | 75 | 0.5 | | | failed trial | | | Bo=K, K frozen |

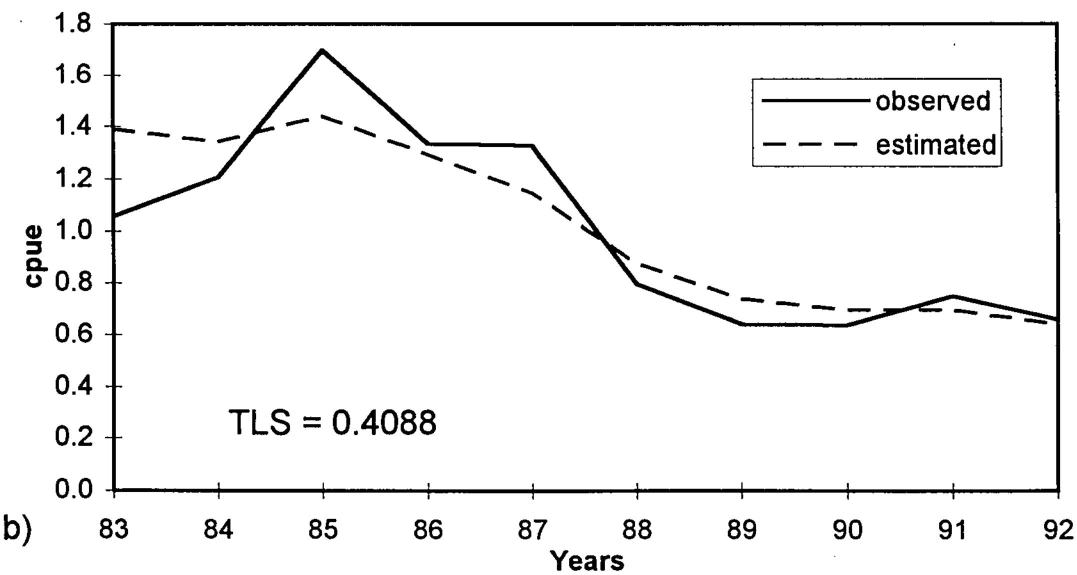
parameter estimates which were not unique (final estimates strongly dependent on the starting estimates in the non-linear search procedure). Noticeably, several combinations of *r, K* and *q* could explain equally well the data, i.e. the values of the minimization criterion are similar while parameter values are markedly different. This was specially evident during unconstrained trials, where estimates of *r* varied by up to two orders of magnitude with little change in the fitting criterion (trials 1.1-1.5). In addition, for some trials the non-linear minimization just failed completely (trials 1.42-1.48).

Strategies aimed at helping the estimation were unsuccessful. First, one parameter at a time was frozen during minimization, but the fit produced biomass estimates that were greater than *K* (trials 1.7-1.15). Using the $B_0=K$ strategy, alone or in combination with freezing one of the parameters, provided the best range of values for the minimization criterion (TLS between 0.40 and 0.42; trials 1.16-1.34). However, this did not remove the uncertainty in the results, and very frequently led to *r* estimates of zero. Additionally, the best values of the fitting criterion did not correspond with the best fits to the CPUE series (fig. 5.5), raising doubts about their reliability.

Results from the trials based only on data corresponding to the eight years of decreasing CPUE are shown in table 5.2. The effect of ignoring the increasing part of the CPUE series was an improvement in performance of the TLS function towards lower minima. However, this was largely and effect of fewer values being considered in the TLS score. In fact the strategy did not remove the problems encountered before. There were many instances in which the answers were nonsensical ($r = 0$; biomasses much larger than *K*). In fact, all the fits with the 'best' TLS scores (those between 0.16 and 0.18) resulted in values of $B_0$ greater than *K* (trials 2.3-2.7, 2.17-2.21, 2.24-2.25, 2.28-2.34). In many cases the non-linear optimisation failed (trials 2.46-2.74). Alternately freezing one of the parameters and letting the others free during estimation did not eliminate the above problems. Despite this, some relatively well behaved results were obtained with the $B_0=K$ strategy (trials 2.35-2.45). However, even this strategy resulted into uncertainty in the parameter estimates (specially for *r*), *r* values of zero, and some dependence of results upon starting values. In addition, the TLS scores from the $B_0=K$ fits were not the overall lowest and the estimated CPUE did not match the data sufficiently well (fig. 5.6).




a)




b)

Figure 5.5. Examples of fits to the entire CPUE series: a) 'good' visual fit, but high TLS score (trial 1.6). b) Low TLS score, but poor visual fit at the beginning of the time series (trial 1.23).

Table 5.2. Results of second set of trials: TLS fits of CPUE and catch data for the downward portion of the time series (years 85-92) from the shark fishery of Yucatan.

| Trial | ---Starting value | | | -------Parameter estimates----------- | | | | TLS | |
|---|---|---|---|---|---|---|---|---|---|
| number | r | K | q | r | K | q | Bo | score | Constraints |
| 2.1 | 0.2 | 100 | 0 | 0.000000 | 76.235340 | 0.035560 | 44.919 | 0.335130 | none |
| 2.2 | 0.2 | 100 | 0 | 0.000000 | 68.765576 | 0.035561 | 44.930 | 0.335130 | none |
| 2.3 | 0.1 | 50 | 0 | 0.056635 | 301.466232 | 0.001049 | 1652.334 | 0.169462 | none |
| 2.4 | 0.15 | 25 | 0 | 0.056417 | 104.694688 | 0.003060 | 566.742 | 0.171341 | none |
| 2.5 | 0.4 | 50 | 0 | 0.047788 | 136.044866 | 0.002087 | 815.991 | 0.171559 | none |
| 2.6 | 0.05 | 100 | 0 | 0.050290 | 295.266865 | 0.000992 | 1715.660 | 0.170342 | none |
| 2.7 | 0.05 | 100 | 0 | 0.055384 | 229.563378 | 0.001359 | 1276.732 | 0.169738 | none |
| 2.8 | 0.5 | 100 | 0 | 0.000000 | 87.221379 | 0.038150 | 44.964 | 0.351572 | none |
| 2.9 | 0.5 | 100 | 0 | 0.000000 | 73.026995 | 0.035561 | 44.940 | 0.335130 | none |
| 2.10 | 0.2 | 100 | 0 | 0.000000 | 72.890042 | 0.035566 | 44.934 | 0.335130 | none |
| 2.11 | 0.2 | 100 | 0 | 0.000000 | 66.021038 | 0.035565 | 44.952 | 0.335131 | none |
| 2.12 | 0.5 | 10 | 0 | 0.698674 | 10.000000 | 0.126087 | 14.845 | 0.308084 | K frozen |
| 2.13 | 0.5 | 40 | 0 | 0.000000 | 40.000000 | 0.037380 | 44.915 | 0.343349 | K frozen |
| 2.14 | 0.25 | 55 | 0 | 0.000000 | 55.000000 | 0.035561 | 44.933 | 0.335130 | K frozen |
| 2.15 | 0.2 | 55 | 0 | 0.000000 | 55.000000 | 0.038938 | 44.988 | 0.362495 | K frozen |
| 2.16 | 0.2 | 100 | 0 | 0.000000 | 100.000000 | 0.038926 | 45.001 | 0.362495 | K frozen |
| 2.17 | 0.05 | 100 | 0 | 0.050000 | 335.978637 | 0.000865 | 1966.673 | 0.170244 | r frozen |
| 2.18 | 0.01 | 45 | 0 | 0.010000 | 162.578179 | 0.000440 | 3859.212 | 0.173141 | r frozen |
| 2.19 | 0.005 | 45 | 0 | 0.005000 | 246.209719 | 0.000150 | 11363.246 | 0.173500 | r frozen |
| 2.20 | 0.01 | 100 | 0 | 0.010000 | 156.217911 | 0.000454 | 3783.909 | 0.173000 | r frozen |
| 2.21 | 0.1 | 100 | 0 | 0.100000 | 221.932509 | 0.002127 | 817.088 | 0.172706 | r frozen |
| 2.22 | 0.2 | 100 | 0 | 0.200000 | 36.071652 | 0.020644 | 85.801 | 0.191134 | r frozen |
| 2.23 | 0.3 | 100 | 0 | 0.300000 | 22.799622 | 0.040789 | 44.031 | 0.211097 | r frozen |
| 2.24 | 0.3 | 100 | 0 | 0.102745 | 50.372810 | 0.010000 | 174.736 | 0.176955 | q frozen |
| 2.25 | 0.3 | 100 | 0 | 0.080424 | 83.518130 | 0.005000 | 348.137 | 0.172791 | q frozen |
| 2.26 | 0.3 | 100 | 0 | 0.545006 | 12.222566 | 0.100000 | 18.584 | 0.269733 | q frozen |
| 2.27 | 0.3 | 100 | 0 | 2.553657 | 4.441209 | 0.300000 | 6.846 | 0.432039 | q frozen |
| 2.28 | 0.05 | 100 | 0 | 0.058387 | 165.024835 | 0.002000 | 851.029 | 0.171158 | q frozen |
| 2.29 | 0.05 | 40 | 0 | 0.022187 | 75.410945 | 0.002000 | 851.029 | 0.173978 | q frozen |
| 2.30 | 0.1 | 75 | 0 | 0.045419 | 136.430525 | 0.002000 | 851.029 | 0.171526 | q frozen |
| 2.31 | 0.05 | 75 | 0 | 0.067337 | 92.750194 | 0.004000 | 426.181 | 0.172800 | q frozen |
| 2.32 | 0.05 | 75 | 0 | 0.081621 | 70.560995 | 0.006000 | 290.278 | 0.173444 | q frozen |
| 2.33 | 0.05 | 75 | 0 | 0.090476 | 57.312158 | 0.008000 | 218.146 | 0.175086 | q frozen |
| 2.34 | 0.2 | 50 | 0 | 0.108509 | 43.766158 | 0.012000 | 145.907 | 0.178428 | q frozen |
| 2.35 | 0.05 | 100 | 0 | 0.043290 | 39.572226 | 0.034777 | ------ | 0.289850 | Bo=K |
| 2.36 | 0.3 | 100 | 0 | 0.041891 | 40.806946 | 0.033655 | ---------- | 0.290269 | Bo=K |
| 2.37 | 0.05 | 25 | 0 | 0.048187 | 38.899841 | 0.035379 | ---------- | 0.289851 | Bo=K |
| 2.38 | 0.5 | 50 | 0 | 0.044640 | 39.310043 | 0.034922 | ---------- | 0.289876 | Bo=K |
| 2.39 | 0.01 | 25 | 0 | 0.009746 | 44.859617 | 0.030582 | ---------- | 0.290192 | Bo=K |
| 2.40 | 0.01 | 50 | 0 | 0.010134 | 44.657819 | 0.030771 | ---------- | 0.290196 | Bo=K |
| 2.41 | 0.5 | 100 | 0 | 0.000000 | 46.569233 | 0.029440 | ---------- | 0.290406 | Bo=K |
| 2.42 | 0.5 | 25 | 0 | 0.000000 | 48.685076 | 0.027848 | ---------- | 0.290799 | Bo=K |
| 2.43 | 0.2 | 100 | 0 | 0.000000 | 46.984687 | 0.029162 | ---------- | 0.290416 | Bo=K |
| 2.44 | 0.2 | 45 | 0 | 0.000000 | 47.462990 | 0.028727 | ---------- | 0.290478 | Bo=K |
| 2.45 | 0.3 | 100 | 0 | 2.805020 | 4.257789 | 0.300000 | ---------- | 0.419515 | Bo=K, q frozen |
| 2.46 | 0.25 | 5 | 0 | | 38.3939485 | 0.038195 | | 0.30823266 | Bo=K |
| 2.47 | 0.1 | 8 | 0 | | 73.9878294 | 0.019037 | | 0.31627416 | Bo=K |
| 2.48 | 0.15 | 9 | 0 | | 41.4014195 | 0.035247 | | 0.30106583 | Bo=K |
| 2.49 | 0.15 | 10 | 0 | | 40.7081229 | 0.035852 | | 0.30103718 | Bo=K |
| 2.50 | 0.15 | 15 | 0 | | 41.1427534 | 0.035449 | | 0.30100048 | Bo=K |

Table 5.2. Continued......

| Trial number | ----Starting values-- r | K | q | -------Parameter estimates---------- r | K | q | Bo | TLS score | Constraints |
|---|---|---|---|---|---|---|---|---|---|
| 2.51 | 0.15 | 20 | 0.05 | 0.024643 | 41.6875158 | 0.03495 | ---------- | 0.300655551 | Bo=K |
| 2.52 | 0.15 | 30 | 0.05 | 0.04394 | 39.4014528 | 0.03514 | ---------- | 0.289970479 | Bo=K |
| 2.53 | 0.15 | 9 | 0.1 | 0.051294 | 38.3594583 | 0.035895 | ---------- | 0.28986273 | Bo=K, Vi=0.05 |
| 2.54 | 0.15 | 30 | 0.05 | 0.046412 | 39.1574264 | 0.035146 | ---------- | 0.289848786 | Bo=K, Vi=0.05 |
| 2.55 | 0.15 | 5 | 0.05 | 2.201323 | 5 | 0.234194 | ---------- | 0.389843718 | Bo=K, K frozen |
| 2.56 | 0.15 | 7.5 | 0.05 | 1.297722 | 7.5 | 0.157274 | ---------- | 0.386253703 | Bo=K, K frozen |
| 2.57 | 0.15 | 10 | 0.05 | 0.824437 | 10 | 0.130818 | ---------- | 0.362737404 | Bo=K, K frozen |
| 2.58 | 0.15 | 12.5 | 0.05 | 0.598343 | 12.5 | 0.108831 | ---------- | 0.33863909 | Bo=K, K frozen |
| 2.59 | 0.15 | 15 | 0.05 | 0.451837 | 15 | 0.092444 | ---------- | 0.321597552 | Bo=K, K frozen |
| 2.60 | 0.15 | 20 | 0.05 | 0.283087 | 20 | 0.069817 | ---------- | 0.303091551 | Bo=K, K frozen |
| 2.61 | 0.15 | 25 | 0.05 | 0.185024 | 25 | 0.055709 | ---------- | 0.294965128 | Bo=K, K frozen |
| 2.62 | 0.15 | 30 | 0.05 | 0.120665 | 30 | 0.046221 | ---------- | 0.2914473 | Bo=K, K frozen |
| 2.63 | 0.15 | 35 | 0.05 | 0.075169 | 35 | 0.039443 | ---------- | 0.290109528 | Bo=K, K frozen |
| 2.64 | 0.15 | 40 | 0.05 | 0.041771 | 40 | 0.034382 | ---------- | 0.289856726 | Bo=K, K frozen |
| 2.65 | 0.15 | 45 | 0.05 | 0.015441 | 45 | 0.030458 | ---------- | 0.290156523 | Bo=K, K frozen |
| 2.66 | 0.15 | 50 | 0.05 | 0 | 50 | 0.027304 | ---------- | 0.29079023 | Bo=K, K frozen |
| 2.67 | 0.15 | 60 | 0.05 | 0 | 60 | 0.02249 | ---------- | 0.29435403 | Bo=K, K frozen |
| 2.68 | 0.5 | 25 | 0.001 | 0.198587 | 25 | 0.052056 | ---------- | 0.30082778 | Bo=K, K frozen, Vi=0.05 |
| 2.69 | 0.15 | 25 | 0.01 | 0.056376 | 126.986224 | 0.002535 | ---------- | 0.171548756 | none |
| 2.70 | 0.2 | 10 | 0.3 | | | failed trial | | | none |
| 2.71 | 0.3 | 20 | 0.03 | | | failed trial | | | none |
| 2.72 | 0.15 | 10 | 0.01 | | | failed trial | | | K frozen |
| 2.73 | 0.5 | 20 | 0.3 | | | failed trial | | | K frozen |
| 2.74 | 0.3 | 20 | 0.1 | | | failed trial | | | K frozen |
| 2.75 | 0.5 | 20 | 0.03 | | | failed trial | | | K frozen |
| 2.76 | 0.1 | 20 | 0.03 | | | failed trial | | | K frozen |
| 2.77 | 0.5 | 25 | 0.3 | | | failed trial | | | K frozen |
| 2.78 | 0.05 | 25 | 0.03 | | | failed trial | | | K frozen |
| 2.79 | 0.5 | 30 | 0.1 | | | failed trial | | | K frozen |
| 2.80 | 0.25 | 35 | 0.03 | | | failed trial | | | K frozen |
| 2.81 | 0.01 | 100 | 0.1 | | | failed trial | | | r frozen |
| 2.82 | 0.5 | 50 | 0.3 | | | failed trial | | | Bo=K |
| 2.83 | 0.2 | 100 | 0.5 | | | failed trial | | | Bo=K |
| 2.84 | 0.01 | 100 | 0.3 | | | failed trial | | | Bo=K |
| 2.85 | 0.05 | 10 | 0.05 | | | failed trial | | | none |
| 2.86 | 0.05 | 10 | 0.2 | | | failed trial | | | none |
| 2.87 | 0.05 | 100 | 0.05 | | | failed trial | | | none |
| 2.88 | 0.05 | 50 | 0.05 | | | failed trial | | | none |
| 2.89 | 0.05 | 40 | 0.05 | | | failed trial | | | none |
| 2.90 | 0.5 | 10 | 0.2 | | | failed trial | | | none |
| 2.91 | 0.5 | 10 | 0.05 | | | failed trial | | | none |
| 2.92 | 0.2 | 45 | 0.2 | | | failed trial | | | none |
| 2.93 | 0.05 | 100 | 0.05 | | | failed trial | | | r frozen |
| 2.94 | 0.01 | 45 | 0.1 | | | failed trial | | | r frozen |
| 2.95 | 0.005 | 45 | 0.05 | | | failed trial | | | r frozen |
| 2.96 | 0.005 | 100 | 0.1 | | | failed trial | | | r frozen |
| 2.97 | 0.005 | 100 | 0.05 | | | failed trial | | | r frozen |
| 2.98 | 0.005 | 45 | 0.1 | | | failed trial | | | r frozen |

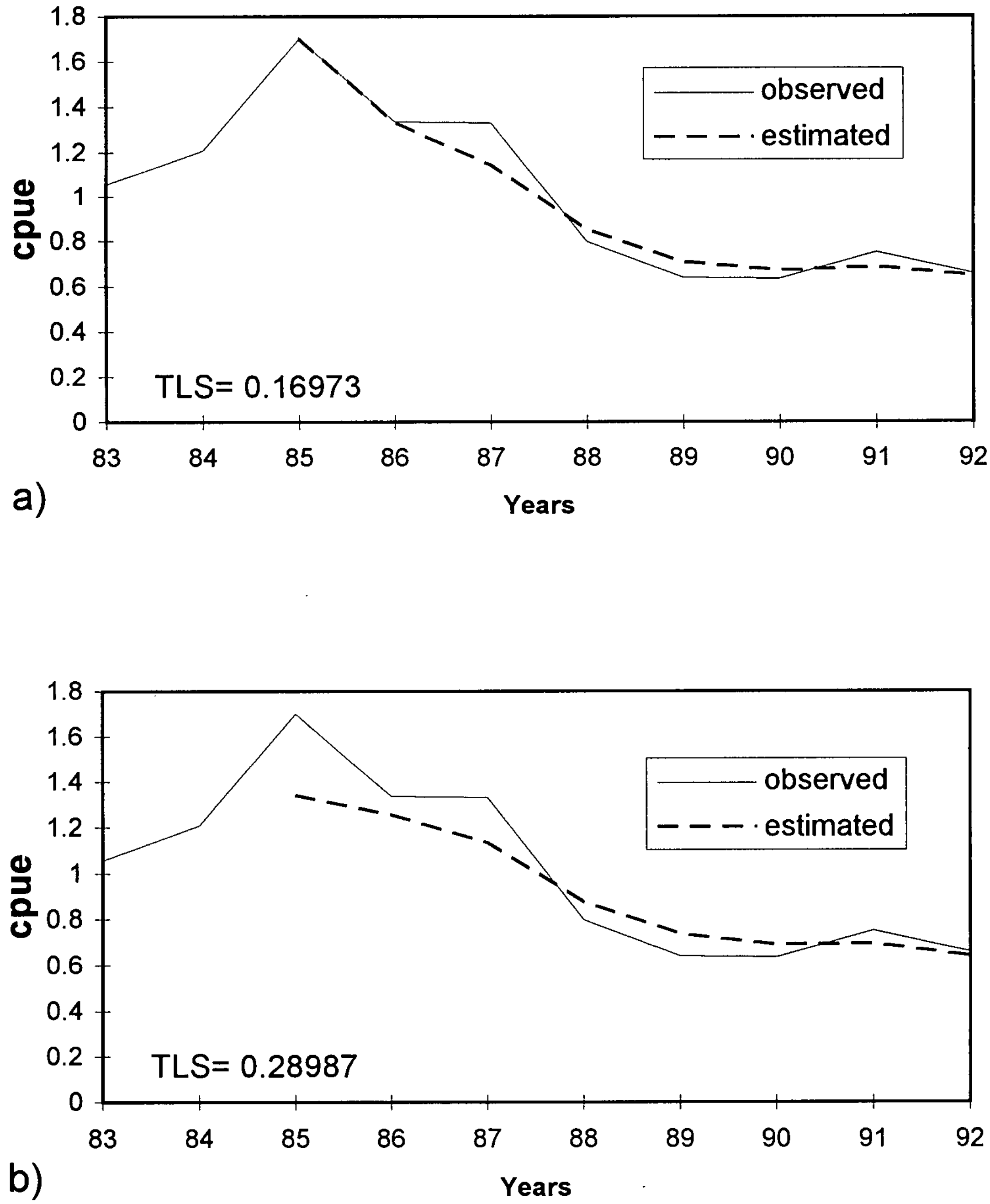



Figure 5.6. Examples of fits to the downward portion of the CPUE series (years 85-92) : a) unconstrained fit (trial 2.7); b) using the *Bo-K* constraint (trial 2.38).

## 5.5 Discussion.

### 5.5.1 Shortcomings of the data.

Failures during the first set of trials are not surprising. The first two data points in the time series define an upwards trend in CPUE (the model reads this as abundance) that cannot be explained properly in the light of the effort information embedded in the data. Only a recovering stock (i.e. under low effort for a sufficiently long recent period of time after heavy exploitation) can show an increase in abundance consistent with the surplus production model. Failing to detect this from the data, the model either predicts biomasses in excess of $K$ or tries to ignore the upwards trend in CPUE (fig. 5.5b).

In fact, the 'unexplainable' increasing trend in CPUE should be enough to cast serious doubts about the reliability of the entire data set. It is very possible that interactions between the spatial distribution of fishermen's effort and the distribution of sharks is resulting in CPUE not being proportional to fish abundance. Several scenarios where this can happen are listed by Hilborn and Walters (1992). These authors stress that commercial CPUE data are seldom an adequate index of abundance. For the sake of this discussion, I will resort to an old explanation for this increasing CPUE at the beginning of the time series which considers that this trend represent a 'learning period' in which fishermen were becoming more and more efficient; this learning period ends when CPUE starts to drop. We can then justify ignoring the first two data points, and work under the assumption that the downward portion of the CPUE series does in fact represent an index of abundance.

Many of the problems encountered during parameter estimation are expected given the characteristics of the data set. There are several reasons why the Yucatan data are not very useful for assessment. First, this is a short time series with only ten, or in the worst case eight data points. It would be indeed surprising to get reliable answers with such a short time series of data. Nevertheless, the most detrimental attribute of these data is not the small quantity, but the poor quality of the information they contain. There is not enough contrast in the Yucatan CPUE and catch data to let the model differentiate clearly what type of population it is dealing with: is this a large stock with very low productivity, a small and very productive stock, or something else? Ludwig and Walters (1985) suggest a rule of thumb for contrast in the data: ratios of at least 4 should be observable in the time series between largest and smallest stock biomass and effort, in order to have enough contrast for parameter estimation of surplus production models. This ratio is only 2.5 in the Yucatan CPUE data.

The lack of contrast in the Yucatan data is also evident in the tendency of the model to return a $r$ value of zero. The Schaefer model fails to estimate $r$ properly because the effects of $r$ are not evident in the early part of one-way trip data series. According to Hilborn (1979), in order to obtain information about $r$ it is necessary first to bring down the stock to low levels of abundance and then allow it to recover under fairly low fishing effort. In the Yucatan data, there is absolutely no recovery period for the stock. This data set is indeed a very good example of what is known as the one way trip: a typical pattern of continuously growing fishing effort accompanied by a decreasing CPUE (fig. 5.7). One way trips are characteristically difficult for estimation purposes because of their lack of information about key features of the stock.

Even though the results of using the $B_0=K$ strategy during the second set of trials offer the best TLS scores for a credible set of results, they should be regarded with suspicion. They do not only rely totally on the $B_0=K$ strategy in order to make sense, but when the fit is unconstrained, these results still depend on the starting values and have uncertainty in their parameter estimates (particularly for the parameter $r$ which varies between 0.0, 0.009, and 0.044).

It would be ill advised to set management measures about the adequate level of effort in this fishery expecting to obtain the predicted *MSY* based on estimates of the optimal effort derived from the present analyses. Most likely such recommendations would not produce the expected results in the short run, not only because of the shortcomings of the Schaefer model for management parameter estimation identified in the previous chapter, but because of the uncertainty surrounding the results of $f_{opt}$ and *MSY* from these trials (see below). Management parameters derived from surplus production models are specially sensitive to values of $r$, the intrinsic rate of increase (Hilborn and Walters 1992). Due to the uncertainty in $r$ identified here, *MSY* and $f_{opt}$ estimates derived from these results will be intrinsically unreliable.

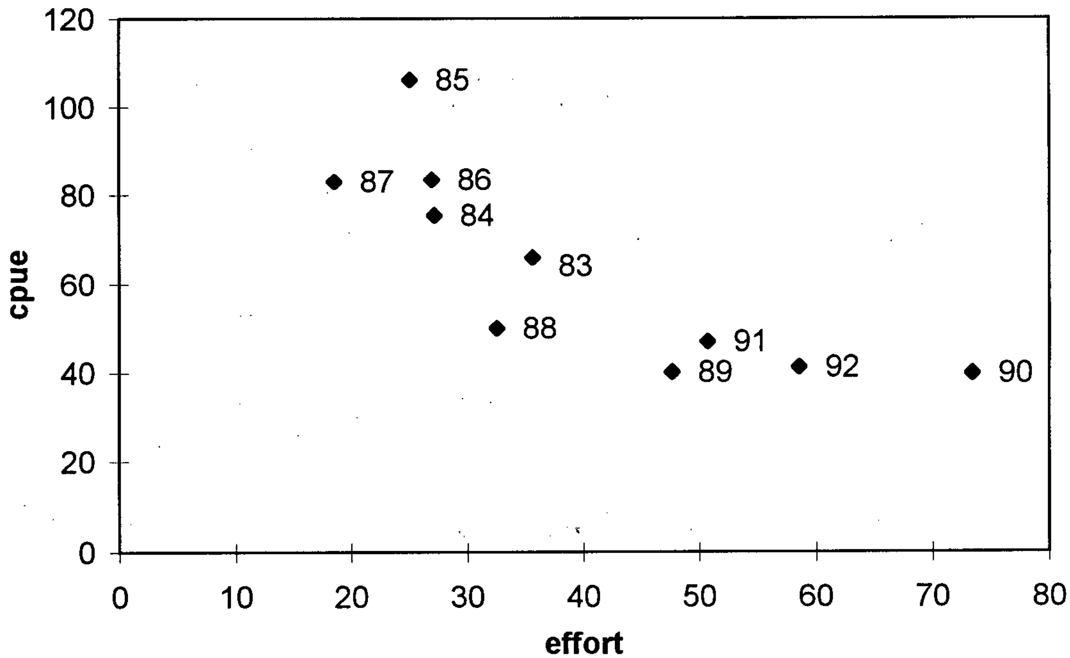



Figure 5.7. Plot of CPUE and effort for the shark fishery of Yucatan. (effort estimated from the total catch and the available CPUE series)

5.5.2 Harnessing uncertainty.

In general, it is possible to know or at least to set bounds, to *MSY* and $f_{opt}$ even though the Schaefer model parameters are not well known. A scatter plot of the range of *r* and *K* estimates obtained from different fits of the model will usually produce a well defined parameter space corresponding to a relatively narrow band of *MSY* values. Similarly, a scatter plot of *r* against *q* will help set bounds to the true value of $f_{opt}$. During exploratory trials, this method worked well when applied to fake data with good contrast generated from a Schaefer model and analysed with the same kind of procedure used for the Yucatan data.

Figure 5.8 is a plot of all *r* and *K* estimates from the second set of trials (8yr time series, table 5.2) excluding all fits producing *r* values of zero. This scatter plot defines a parameter space which suggests that for the Yucatan fishery, most *MSY* values are between 0.25 and 4 thousand tonnes, although the large majority of the results are under the 2 thousand tonnes *MSY* isoline. Unfortunately, the Yucatan data show a pathological situation that leaves still a lot of uncertainty around the *MSY* estimates. There are two problems with these data: first, the results lead to *MSY* values that can be as low as 0.2 or as large as 4 (thousand tonnes); and second, there are two very distinct *r-K* tradeoffs in the results. Circles indicate the results from unconstrained fits, whereas crosses indicate results from fits using the $B_0$=*K* constraint. The range of *MSY* values obtained with the *Bo=K* strategy span several orders of magnitude and tend to a *MSY* of zero as *K* increases towards values greater than 40. Basically, it is still impossible to say with certainty what the *MSY* level is for this fishery. The same pathological situation can be observed for the sets of values of *r* and *q*, which define two different tradeoffs depending on whether the $B_0$=*K* assumption was used or not (fig. 5.9)

A similar analysis illustrates very well the effects of another potential source of uncertainty in the Yucatan data. Figure 5.10 is a scatter plot of the same results of table 5.2, but this time the *MSY* isolines are calculated using three different relations between CPUE and biomass. These relations are defined by the following equation:

$$CPUE = qB^{\beta}$$

where $\beta$ is a constant defining proportionality ($\beta = 1$), hyperdepletion ($\beta > 1$) or hyperstability

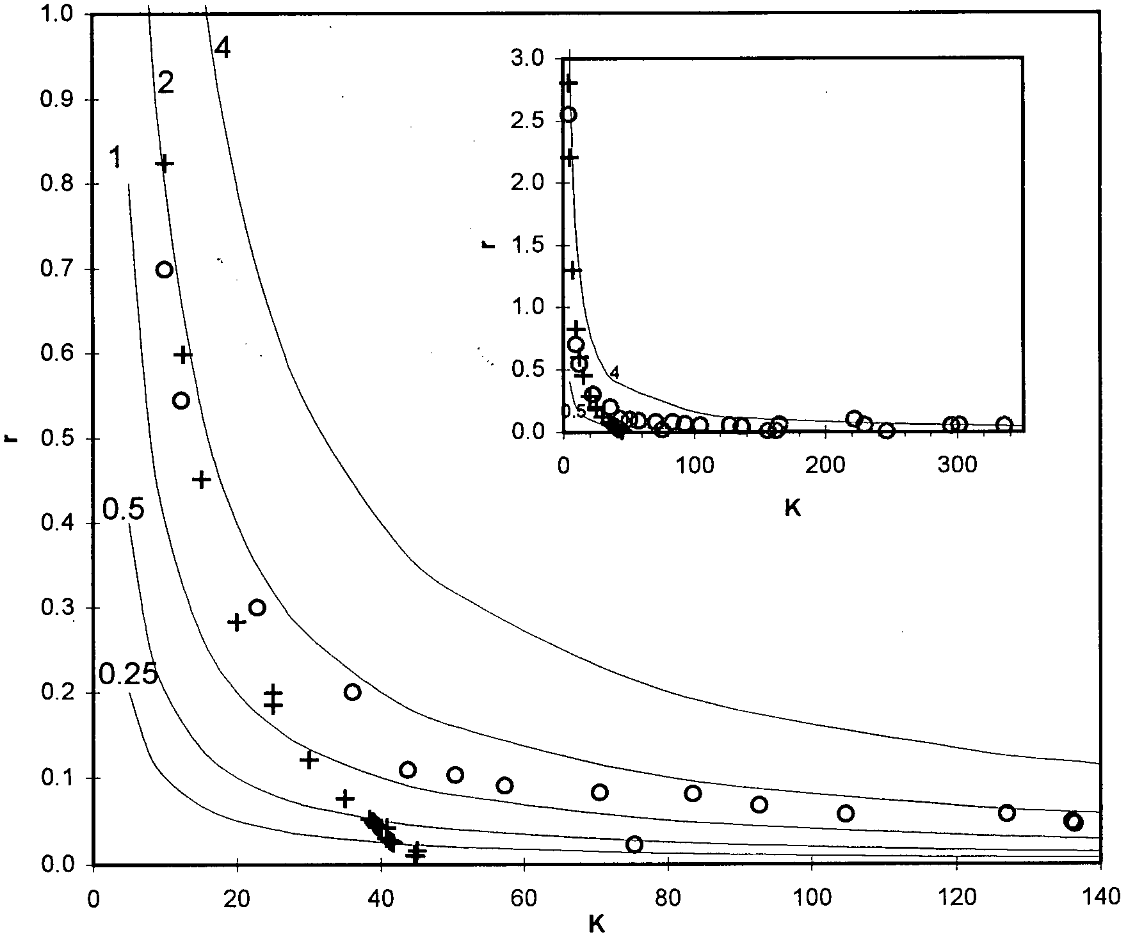



Figure 5.8. The uncertainty around the *MSY* value is roughly bounded by *MSY* isolines of less than 2 (thousand tonnes). Pairs of estimated *r* and *K* values for the Schaefer model are plotted for runs using the *Bo=K* assumption (crosses) and for runs not using this assumption (circles). Inset shows the complete range of estimated parameter values.

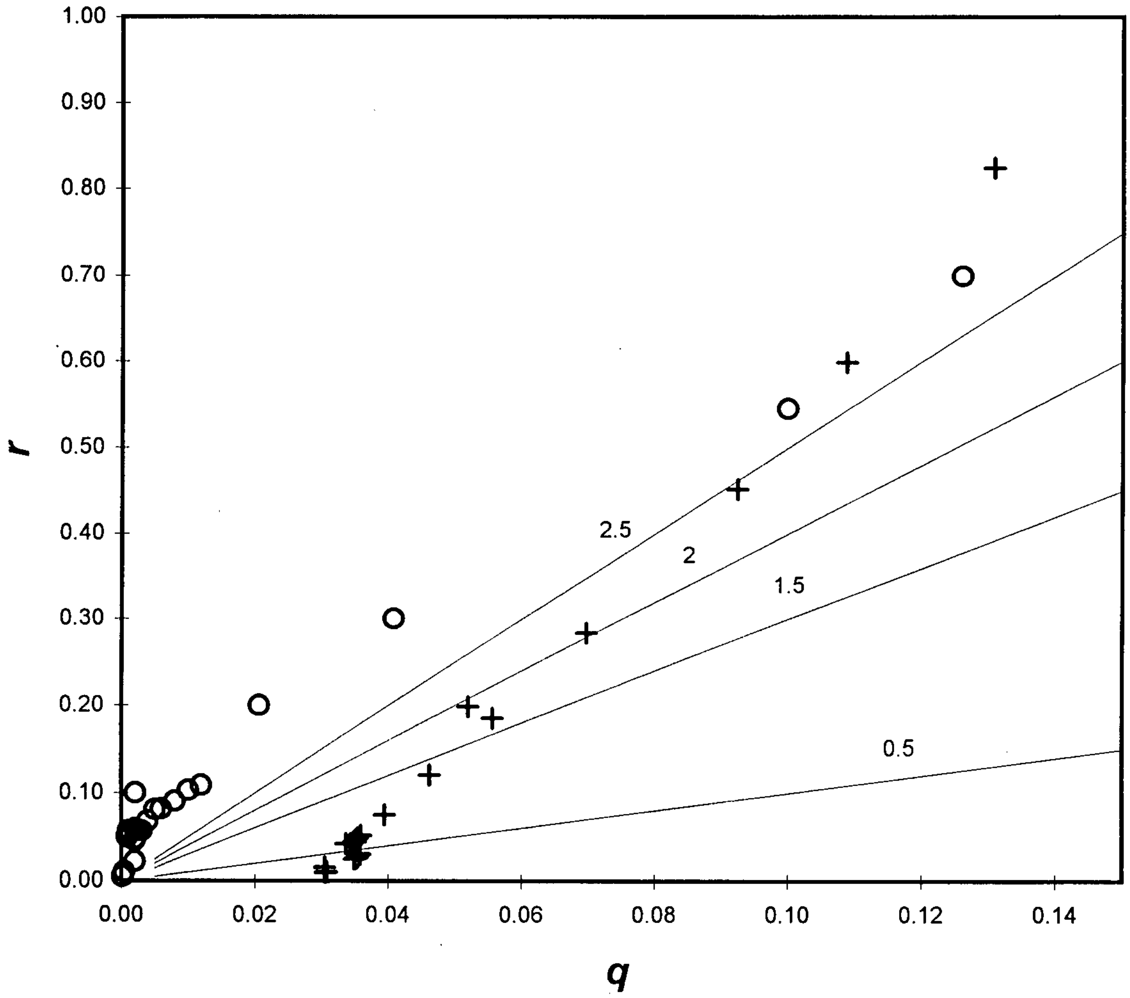



Figure 5.9 Scatter plot of $r$ and $q$ values obtained from fits with (crosses) and without (circles) the $Bo=K$ assumption. The straight lines are lines of equal *fopt* values.

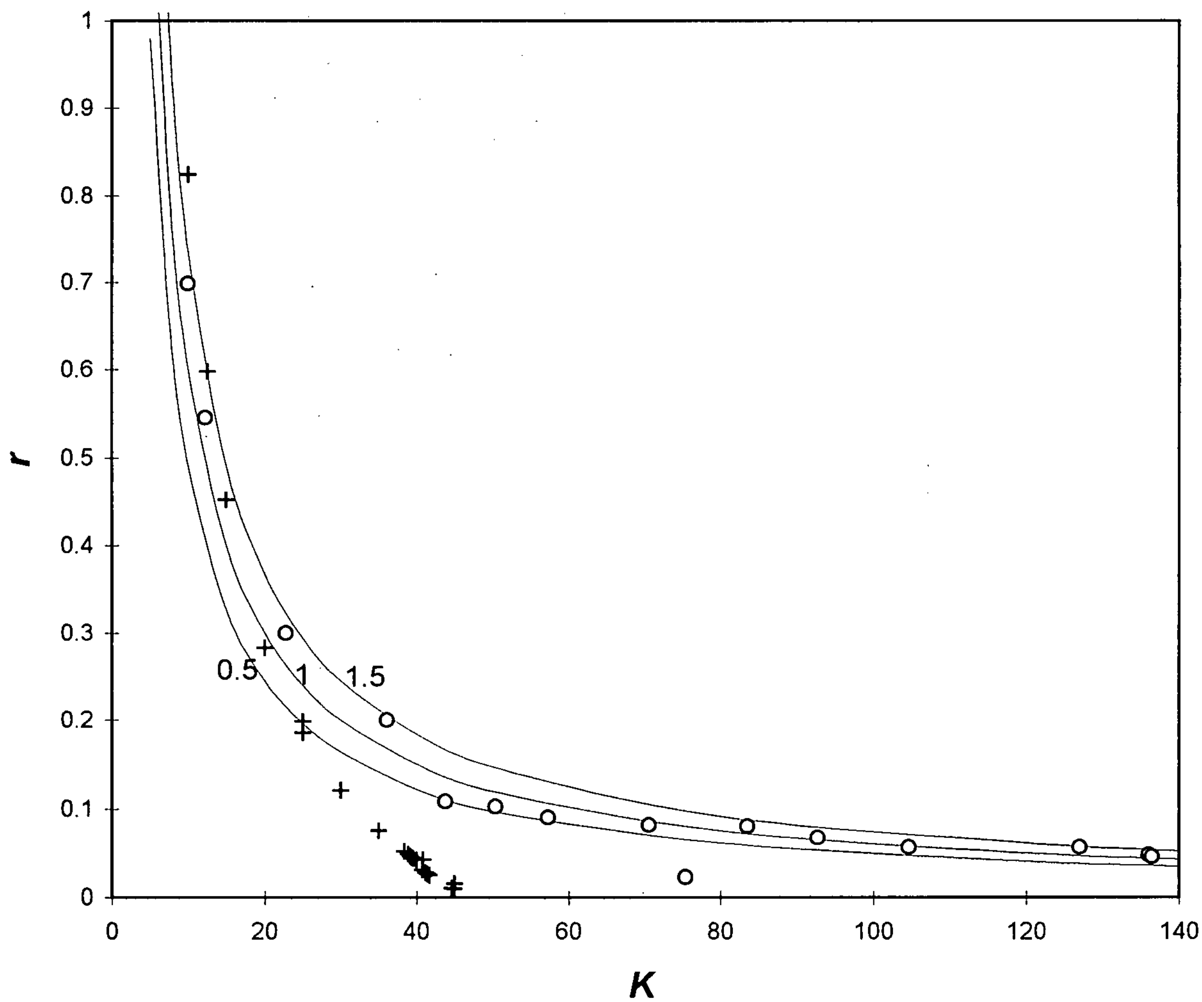



Figure 5.10. Uncertainty in the CPUE-Biomass relationship. Isolines of *MSY* = 1.5 thousand tonnes, for different values of the parameter beta representing proportionality (1), hyperdepletion (1.5) and hyperstability (0.5). Numbers in each isoline are $\beta$ values. Key to symbols as in figure 5.8.

($\beta$ <1). By calculating the $r$ and $K$ values that result in a $MSY$ isoline of value 1.5 thousand tonnes using values of $\beta$ of 1, 0.5, and 1.5, it is shown that the Yucatan data could fit any of these lines. From this, it is evident that the data could be bounded by an infinite number of $MSY$ isolines of various values, depending on what is the true value of $\beta$.

### 5.5.3 How can we improve future assessments of the Yucatan shark fishery?

Management advice for this system has to be carefully geared towards improving the assessment of the fishery, i.e. we need to ensure a better contrast in the data and we need to know if CPUE is proportional to abundance. In other words, it is necessary to explicitly design experimental management actions that would provide us with i) clearly contrasting effort levels, and ii) an abundance index that is truly proportional to stock size. Unfortunately, this implies taking very serious decisions about fishing patterns for the oncoming years. The formal and thorough way to proceed for making such decisions is through the application of adaptive policy analysis (Walters 1986). However, discussing adaptive management design is a complex topic outside the scope of the present study. It should suffice to say here that a proper statistical evaluation of all the likely hypotheses and all possible management decisions should be made before any decision is taken over whether or not to set experimental management actions, and before deciding which adaptive policies, if any, should be tried. My approach here is thus to simply outline in very broad terms which are the types of actions that would provide the Schaefer model with the information it needs for the proper assessment of the Yucatan fishery.

The simple collection of another ten years of CPUE and catch information without any planning on the quality of these data will only produce a 20 yr uninformative time series. What is needed are fishing patterns that will provide enough information about the size of the stock ($K$), the catchability of the fishing system ($q$), and the productivity of the stock ($r$). The trials presented above demonstrated particular problems to estimate the value of $r$. This fact, and the importance of $r$ for management parameter estimation ($MSY$ and $f_{opt}$) make it natural to think first about obtaining information on $r$. Nevertheless, to gain information about $r$ it would be desirable to first drive the stock down considerably and then let it recover. Apart from leading us to the conditions we need for obtaining information about $r$, this would provide us with valuable information about $K$-$q$ first. The recommended recipe would thus be, to fish harder for a few years in order to bring down the abundance in a relatively short time and thus provide the model with information about $K$ and $q$ (Hilborn, 1979). This should be followed by a significant reduction in effort for a number of years that allows the stock to rebuild. This phase of stock recovery would provide the model with very valuable information about the intrinsic rate of growth $r$. A test fishery would probably be the best way to monitor the recovery of the stock.

The final question that needs to be addressed is whether the commercial CPUE data from this fishery are a good index of stock abundance. I have shown in the previous chapter that even the best fitting procedures for surplus production models fail to provide unbiased estimates whenever hyperdepletion or hyperstability are present. Under the current circumstances in the Yucatan fishery, the first step should be to establish a system for the collection of spatial effort and CPUE information. This should be followed by the mapping of this information in order to identify trends and patterns in the allocation of effort and its relation to shark distribution. This can in turn provide some clues about the relationship between abundance and CPUE.

If the above analysis indicates a likelihood of hyperdepletion or hyperstability in this fishery, then it would be necessary to develop a careful and detailed fisheries data gathering system. One approach is to carry fishery-independent surveys of shark abundance; however the operational cost of this alternative would likely be prohibitive. Another, perhaps more feasible option, is to employ a good number of fishermen as sampling devices in an exercise on experimental fishing. Implementing this alternative will certainly require compensation to participating fishermen for their opportunity costs, but these would likely be much smaller than the costs of carrying fishery independent surveys and would offer an advantageous interaction between researchers, managers and fishermen which will prove to be essential for the success of future management actions and policy making. In either case, the spatial distribution of the fishing effort has to be such that fishing takes place evenly throughout the entire range of distribution of the sharks and not only following the concentrations of sharks, which is probably what is happening in the fishery now, as this is the most common behaviour among fishermen that know their business.

## 5.6 Summary and conclusions.

The limitations in the data encountered during the present study of the Yucatan shark fishery are likely to be found whenever stock assessment and management of elasmobranch fisheries is attempted in most tropical countries. In the great majority of the cases there will be no detailed fishery statistics by species and no biological information. The high cost of obtaining species-specific catch data and the complications of discriminating effort for a species in a fishery that is multispecific, are likely to constrain any attempts to perform the more sophisticated single-species analyses in tropical elasmobranch fisheries. I have shown that it is possible to obtain some CPUE information by direct search among the local fishing companies and use this for surplus production analyses that can provide a great deal of insight about the problems for assessing the fishery. This should also be a feasible way to get started in any other country with a significant elasmobranch fishery.

The usage of aggregated-species surplus production models illustrated here should only be a first and rough step towards the assessment of elasmobranch fisheries. Their major benefit is perhaps to emphasise the pitfalls of the available information and this might help to set management strategies that allow for better information. For some cases with better data conditions, where fishery independent abundance information or better contrasted CPUE data is available, the kind of analyses illustrated here might prove to yield satisfactory results. For difficult cases, the fitting strategies used here ($B_0=K$ constraint; freezing one parameter at a time) might help in the analysis of the data. If such investigations stress problems of poor data contrast, then a formal analysis of adaptive management strategies should be carried out. In such a case, it should be kept in mind that the level of participation of fishermen into any organised data gathering process is going to be constrained by many cultural, social and economic factors not considered here. Further research including a case study of an actual attempt to implement the kind of policies suggested here for improving the data for assessment, would be a very important step forward to evaluate the validity and feasibility of such policies.

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APPENDIX 1. List of common and latin names of the elasmobranchs mentioned in the text.

| Latin name | Common name | Latin name | Common name |
|---|---|---|---|
| *Aetobatus narinari* | Spotted eagle ray | *Mustelus henlei* | Brown smooth-hound |
| *Aetomylaeus nichofii* | Banded eagle ray | *Mustelus lenticulatus* | Rig |
| *Alopias pelagicus* | Pelagic thresher shark | *Mustelus lunulatus* | Sicklefin smooth-hound |
| *Alopias superciliosus* | Bigeye thresher shark | *Mustelus manazo* | Starspotted smooth-hound |
| *Alopias vulpinus* | Common thresher shark | *Mustelus mustelus* | Smooth-hound |
| *Callorhynchus millii* | Elephant fish | *Mustelus schmitti* | Narrownose smooth-hound |
| *Carcharhinus acronotus* | Blacknose shark | *Myliobatis aquila* | Common eagle ray |
| *Carcharhinus albimarginatus* | Silvertip shark | *Nasolamia velox* | Whitenose shark |
| *Carcharhinus altimus* | Bignose shark | *Negaprion acutidens* | Sicklefin lemon shark |
| *Carcharhinus brachyurus* | Copper shark | *Negaprion brevirostris* | Lemon shark |
| *Carcharhinus brevipinna* | Spinner shark | *Notorynchus cepedianus* | Bradnose sevengill shark |
| *Carcharhinus falciformis* | Silky shark | *Odontaspis ferox* | Ragged-thooth shark |
| *Carcharhinus galapagensis* | Galapagos shark | *Odontaspis noronhai* | Bigeye sand tiger shark |
| *Carcharhinus hemiodon* | Pondicherry shark | *Prionace glauca* | Blue shark |
| *Carcharhinus isodon* | Finetooth shark | *Pristiophorus cirratus* | Longnose sawshark |
| *Carcharhinus leucas* | Bull shark | *Pristiophorus nudipinnis* | Shortnose sawshark |
| *Carcharhinus limbatus* | Blacktip shark | *Pseudocarcharias kamoharai* | Crocodile shark |
| *Carcharhinus longimanus* | Oceanic whitetip shark | *Pteromylaeus bovinus* | Bull ray |
| *Carcharhinus melanopterus* | Blacktip reef shark | *Raja alba* | White skate |
| *Carcharhinus obscurus* | Dusky shark | *Raja batis* | Blue skate |
| *Carcharhinus perezi* | Caribbean reef shark | *Raja binoculata* | Big skate |
| *Carcharhinus plumbeus* | Sandbar shark | *Raja brachyura* | Blonde ray |
| *Carcharhinus porosus* | Smalltail shark | *Raja clavata* | Thornback ray |
| *Carcharhinus signatus* | Night shark | *Raja fullonica* | Shagreen ray |
| *Carcharhinus sorrah* | Spot-tail shark | *Raja inornata* | California ray |
| *Carcharhinus tilstoni* | Australian blacktip shark | *Raja microocellata* | Painted ray |
| *Carcharias taurus* | Sandtiger shark | *Raja montagui* | Spotted ray |
| *Carcharodon carcharias* | Great White shark | *Raja naevus* | Butterfly skate |
| *Centrophorus granulosus* | Gulper shark | *Raja oxyrinchus* | Longnosed skate |
| *Centrophorus lusitanicus* | Lowfin gulper shark | *Raja radiata* | Thorny skate |
| *Centrophorus uyato* | Little gulper shark | *Raja rhina* | Longnose skate |
| *Cephaloscyllium ventriosum* | Swell shark | *Raja undulata* | Undulate ray |
| *Cetorhinus maximus* | Basking shark | *Rhincodon typus* | Whale shark |
| *Chiloscyllium indicum* | Slender bambooshark | *Rhinobatos granulatus* | Granulated guitarfish |
| *Dalatias licha* | Kitefin shark | *Rhinobatos horkelii* | Brazilian guitarfish |
| *Dasyatis brevis* | Whiptail stingray | *Rhinobatos percellens* | Chola guitarfish |
| *Dasyatis jenkinsii* | Pointed-nose stingray | *Rhinobatos planiceps* | Pacific guitarfish |
| *Dasyatis pastinaca* | Common stingray | *Rhizoprionodon acutus* | Milk shark |

| Latin name | Common name |
|---|---|
| *Dasyatis sephen* | Cowtail ray |
| *Dasyatis violacea* | Pelagic stingray |
| *Deania calcea* | Birdbeak dogfish |
| *Echinorhinus cookei* | Brambell shark |
| *Euprotomicrus bispinatus* | Pigmy shark |
| *Eusphyra blochii* | Winghead shark |
| *Furgaleus macki* | Whiskery shark |
| *Galeocerdo cuvier* | Tiger shark |
| *Galeorhinus galeus* | Tope, school or soupfin shark |
| *Galeus melastomus* | Blackmouth catshark |
| *Ginglymostoma cirratum* | Nurse shark |
| *Heptranchias perlo* | Sharpnose sevengill shark |
| *Heterodontus franciscanus* | Hornshark |
| *Heterodontus mexicanus* | Mexican hornshark |
| *Hexanchus griseus* | Bluntnose sixgill shark |
| *Hexanchus vitulus* | Bigeye sixgill shark |
| *Himantura bleekeri* | Whiptail stingray |
| *Himantura uarnak* | Honeycomb stingray |
| *Isistius brasiliensis* | Cookiecutter shark |
| *Isogomphodon oxyrhynchus* | Daggernose shark |
| *Isurus oxyrinchus* | Shortfin mako shark |
| *Isurus paucus* | Longfin mako shark |
| *Lamna ditropis* | Salmon shark |
| *Lamna nasus* | Porbeagle shark |
| *Mobula diabolus* | Devil ray |
| *Mustelus antarcticus* | Gummy shark |
| *Mustelus asterias* | Starry smooth-hound |
| *Mustelus californicus* | Grey smooth-hound |
| *Mustelus canis* | Dusky smooth-hound |
| *Mustelus fasciatus* | Striped smooth-hound |
| *Rhizoprionodon longurio* | Pacific sharpnose shark |
| *Rhizoprionodon oligolinx* | Grey sharpnose shark |
| *Rhizoprionodon porosus* | Caribbean sharpnose shark |
| *Rhizoprionodon terraenovae* | Atlantic sharpnose shark |
| *Rhynchobatus djiddensis* | whitespotted shovelnose |
| *Scoliodon laticaudus* | Spadenose shark |
| *Scoliodon sorrakowa* | Milk shark |
| *Scyliorhinus canicula* | Lesser spotted dogfish |
| *Scyliorhinus stelaris* | Nursehound |
| *Scyliorhinus torazame* | Cloudy catshark |
| *Somniosus microcephalus* | Greenland shark |
| *Sphyrna lewini* | Scalloped hammerhead shark |
| *Sphyrna mokarran* | Great hammerhead shark |
| *Sphyrna tiburo* | Bonnethead shark |
| *Sphyrna zygaena* | Smooth hammerhead shark |
| *Squalus acanthias* | Spiny or picked dogfish, spurdog |
| *Squalus blainvillei* | Longnose spurdog |
| *Squalus cubensis* | Cuban dogfish |
| *Squalus fernandinus* | Japanese spurdog |
| *Squalus mitsukurii* | Shortspine spurdog |
| *Squatina aculeata* | Sawback angelshark |
| *Squatina argentina* | Argentine angelshark |
| *Squatina californica* | Pacific angel shark |
| *Squatina dumeril* | Sand devil shark |
| *Squatina squatina* | Angelshark |
| *Triakis scyllium* | Banded houndshark |
| *Triakis semifasciata* | Leopard shark |

APPENDIX 2. Calculation of true values of $f_{opt}$ and $C_{opt}$ for the operating populations.

A formal estimation of the true values of the management benchmarks of $f_{opt}$ and $C_{opt}$ implies numerical calculation of their values by iteration for each single simulation trial. This is necessary because the stochastic nature of each simulation trial produces variations in the particular level of effort needed to produce a long term maximum yield, rendering the levels of $f_{opt}$ and $C_{opt}$ to be different for each simulated population. The numerical search consists of choosing a constant level of effort and fishing the population for a long period of time (300 years in this study) then calculating the average catch after the new equilibrium (catches for the years 100-300 in this case). This process is repeated trying different levels of effort in discrete step sizes. The level of effort producing the highest average long term catch is the optimal effort and the corresponding catch is MSY.

Due to the enormous amount of computational time demanded for applying this approach to 100 Monte Carlo simulations for each operating population, a shortcut strategy was devised. A total of 11 different estimates of the management parameters were obtained for each operating population by using 11 different seeds to restart the pseudo-random number generator in the program. Inside each of the 11 trials, the same sequence of pseudo-random numbers was used for each of the effort levels by using the same seed. The median and quartiles of the 11 estimates of $f_{opt}$ and $C_{opt}$ were saved from each operating population and are shown in table A.1. Given the small variation in the estimates, the median values were used as the true values of $f_{opt}$ and $C_{opt}$ for each operating population.

Table A.1 Median and quartiles of true $f_{opt}$ and $C_{opt}$ values for the different operating populations as characterised by cpue-biomass relationship, stock-recruitment function and age of entry to the fishery ("productivity").

| **Operating population** | | | **Optimal effort** | | | **Optimal Catch** | | |
|---|---|---|---|---|---|---|---|---|
| **cpue/biomass** | **S-R** | **$A_k$** | **Median** | **Quartiles** | | **Median** | **Quartiles** | |
| Proportional | B&H | 7yr | .3720 | .3700 | .3720 | .1978 | .1973 | .1984 |
| " | Ricker | 7yr | .2640 | .2625 | .2645 | .0573 | .0572 | .0574 |
| " | B&H | 4yr | .2799 | .2789 | .2809 | .1827 | .1824 | .1833 |
| " | Ricker | 4yr | .2050 | .2040 | .2050 | .0535 | .0533 | .0536 |
| Hyper Stability | B&H | 7yr | .1960 | .1940 | .1965 | .1982 | .1977 | .1988 |
| " | Ricker | 7yr | .1750 | .1745 | .1760 | .0575 | .0574 | .0577 |
| Hyper Depletion | B&H | 7yr | .5890 | .5630 | .5920 | .1979 | .1976 | .1985 |
| " | Ricker | 7yr | .3750 | .3730 | .3780 | .0574 | .0572 | .0575 |

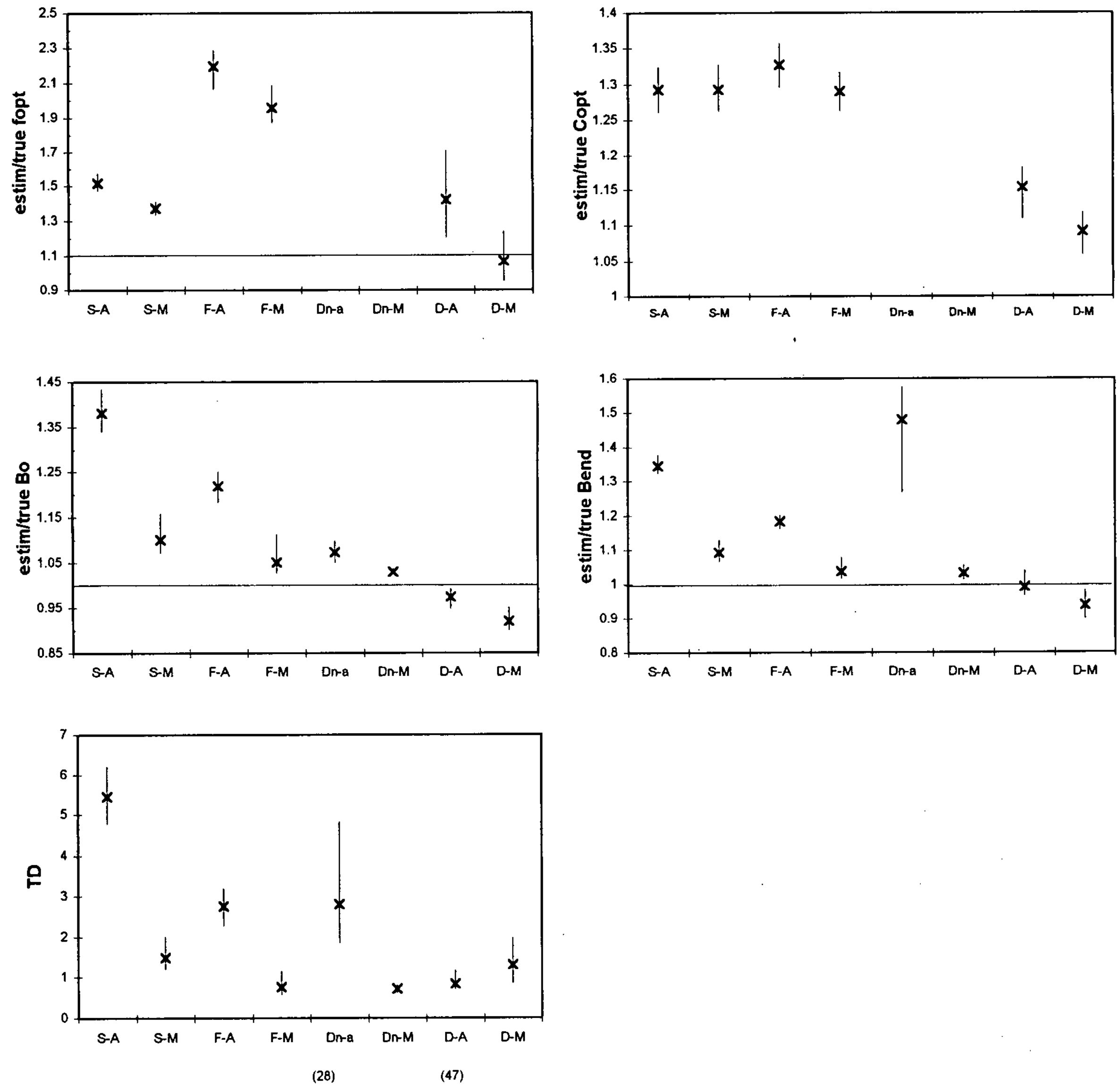


Figure A.1 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model using the two observation error assumptions. Monte Carlo simulations performed with the proportionality Beverton-Holt OP and high contrast effort. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1.. For each estimation model, A means additive, M means multiplicative observation error assumption. ()= number of failed trials.

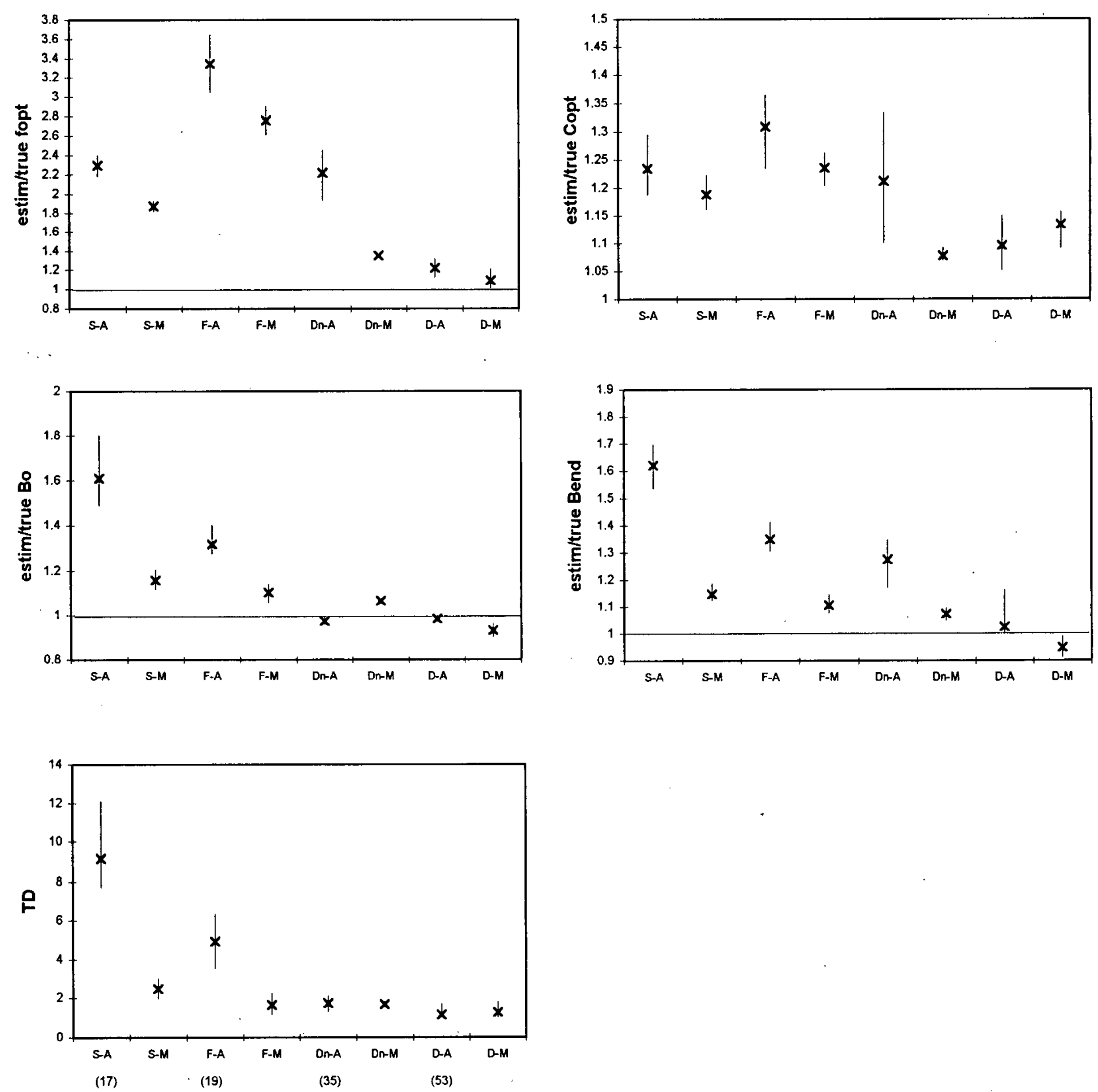


Figure A.2 Modified box plots showing distribution of estimates of management and biomass assessment benchmarks for each estimation model and the two observation error assumptions, using the $B_0$=$K$ assumption. Monte Carlo simulations performed with the proportionality Ricker OP and high contrast effort. Codes in the x axis correspond to letters used to name estimation procedures in table 4.1. For each estimation model, A means additive, M means multiplicative observation error assumption. ()= number of failed trials.