Universidade do Algarve

Faculdade de Ciências do Mar e do Ambiente

The Cape Verde Coastal Ecosystem

A Study of Community Structure, Trophic Interactions and Exploitation Pattern

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Doctor of Philosophy Dissertation in Fisheries Science and Technology

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The Cape Verde Coastal Ecosystem

A Study of Community Structure, Trophic Interactions and Exploitation Pattern

O Ecossistema Costeiro de Cabo Verde

Estudo sobre a Estrutura de Comunidades, Interacções Tróficas e Padrão de Exploração

by / por

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Abstract

A mass-balance trophic model was created to describe the coastal ecosystem of the Cape Verde Archipelago for the time period from 1981 to 1985, using available estimates on biomass and catches. This time period was characterised by predominantly artisanal fisheries and a low level of motorisation; the initial phase of a more intensive fishery development.

Subsequently, a dynamic simulation model, using Ecopath with Ecosim, was used to simulate from 1986 to 2000, incorporating time series information on biomass, catches and catch per unit of effort (CPUE). Using this approach, a number of problems were detected in model specification such as incompatible biomass estimates for small pelagics and various demersal fish groups. Measures of fishing effort in fishing days or trips were found to be unreliable, in particular for the artisanal fishery. Available biomass estimates for small pelagic resources from acoustic surveys were questionable. Conventional biomass dynamic modelling was therefore applied to estimate biomass of small pelagics and provide indications for management purposes, including parameter estimation and risk assessment using the "frequentist" and Bayesian approaches. After a number of adjustments applied to the initial model, the approach used in simulation was to fit the model to observed catch estimates by adjusting effort, placing less emphasis on fitting to CPUE and biomass estimates. A reasonable overall fit to time series of catches was obtained for 18 fish groups, using only two overall trends for artisanal and industrial effort and three specific trends for small pelagics, yellowfin, and skipjack, which are the main targets of the indutrial fisheries.

The observed decrease in abundance of important predators such as yellowfin and skipjack tuna resulted in decreased predation on neritic pelagic species and some demersal fish groups, but this was compensated by higher fishing mortality over the study period. Consequently, the model estimated an almost constant biomass of neritic fish species from 1986 to 2000. Overall fish biomass decreased by 10 percent, including pelagic migratory species. Relative fishing effort was assumed to have almost tripled over the time period from 1986 to 2000 (effort directed towards small pelagics was assumed to have increased by a factor 5), but this resulted in only a 19 per cent increase in catches. Thus, previous assessments of potential fish harvest, ranging from 25 000 to 58 000 tonnes, appear to have been overly optimistic.

Alternative methodologies were applied to assess the fishery resources in Cape Verde in order to gain further understanding on the dynamics of the system and the effects of fishing over time as well as to validate the results of ecosystem modelling. This included non-parametric multidimensional scaling (MDS), generalized linear models (GLM), size spectra analysis, and a type of time series analysis (MAFA). The changes observed over time in the size structure and species composition/abundance of demersal fish communities were slight or even negligible. There was a tendency for lower abundance of demersal fish in recent years, but a decreasing trend was not apparent. It is however important to note that these results were based on trawl survey data, which concern a limited area of trawlable grounds in Cape Verde. In contrast, time series analysis of catch data indicates that a shift has occurred with decreasing catches of important pelagic species such as yellowfin and skipjack tuna and increasing catches of small pelagics and neritic tuna as well as some demersal species. Simulation with Ecosim gave similar results as they were based on the same catch data, but this is nevertheless reassuring as alternative methods gave consistent results.

We believe that this study has been successful in taking the first steps towards an ecosystem approach to assess the effects of fishing in Cape Verde, but further research is necessary to resolve some crucial issues such as the conflicting results on demersal fish biomass and the possible over-exploitation of demersal predators and moray eels. Priority should also be given to the definition of fleet components, effort and CPUE estimation, and the study of increasing efficiency, using the available catch and effort data.

Resumo (Português)

Foi criado um modelo trófico para descrever o ecossistema costeiro do arquipélago de Cabo Verde para o período de 1981 a 1985, utilizando estimativas disponíveis de biomassa e capturas. Este período pode ser considerado a fase inicial de desenvolvimento das pesquerias, sendo de carácter artesanal predominante e com baixo uso de motores na pesca. Verifica-se que as capturas têm variado ao longo do tempo do estudo, entre 1986 e 2000, mas existe uma evolução de aumento gradual de 7 000 a 10 000 toneladas composto maioritariamente por pequenos (cavala preta) e grandes (atuns) pelágicos.

Subsequentemente, um modelo dinâmico de simulação (Ecopath com Ecosim) foi utilizado para simular as séries temporais de biomassa, capturas e captura por unidade de esforço (CPUE) no período de 1986 a 2000. Como resultado desta abordagem foram detectados vários problemas na especificação dos parâmetros do modelo tal como estimativas incompatíveis de biomassa para os pequenos pelágicos e vários grupos demersais de peixes. Determinou-se que as medidas do esforço da pesca em dias ou viagens de pesca não são fiáveis, particularmente no caso da pesca artesanal. As estimativas disponíveis de biomassa para os pequenos pelágicos, baseados em campanhas acústicas, parecem ser pouco fiáveis. Portanto, um modelo dinâmico de produção geral foi aplicado para obter estimativas de biomassa para os pequenos pelágicos e mais ainda, indicadores e recommendações para a gestão deste recurso, utilizando métodos "bootstrap" e Bayesianos. Depois de aplicar um número de ajustes ao modelo ecossistémico inicial, o método utilizado na simulação foi a de ajustar às capturas como objectivo principal, dando menos ênfase às observações de CPUE e biomassa. Conseguiu-se um ajuste razoável às 18 séries temporais de captura de peixes, utilizando dois padrões de evolução do esforço para a frota artesanal e industrial e três padrões específicos para os pequenos pelágicos, rabil, e gaiado, sendo estes os principais alvos da pesca industrial.

A diminuição observada na abundância de predadores importantes tais como a albacora e o gaiado resultou numa diminuição da predação em peixes costeiros, tanto pelágicos como demersais. No entanto, esta diminuição de mortalidade foi compensada por um aumento da mortalidade por pesca durante o período em estudo. Consequentemente, o modelo estimou uma biomassa quase constante de espécies costeiros entre 1986 e 2000. A biomassa total dos peixes diminuiu em 10 por cento, incluindo os grandes migradores pelágicos. É suposto que o esforço relativo de pesca quase triplicou durante o período de 1986 a 2000 (em relação aos pequenos pelágicos o esforço é suposto ter aumentado com um factor 5), mas este aumento significativo de esforço resultou apenas num aumento de 19 por cento nas capturas.

Várias metodologias alternativas foram aplicadas para avaliar o estado dos recursos marinhos em Cabo Verde a fim de melhorar o conhecimento sobre a dinâmica do sistema e determinar os efeitos da pesca como também proceder à validação dos resultados do modelo ecossistémico. Estes métodos incluiram análise multivariada (MDS), modelos lineares generalizados (GLM), análise de estrutura de tamanhos (size spectra), e análise de séries temporais (MAFA). Não foram observadas alterações significativas em relação à estrutura de tamanhos e na composição/abundância de comunidades demersais de peixes durante o período em estudo. Verificou-se uma tendência para a diminuição na abundância de peixes demersais em anos recentes, mas o padrão não era significativo. É de salientar que estes resultados foram baseados em dados de campanhas de arrasto, o que abrange uma área limitada de fondos adequados em Cabo Verde. Por outro lado, a análise de séries temporais de capturas indicam uma alteração importante com a diminuição dos grandes migradores pelágicos, nomeadamente a albacora e gaidao, e um aumento nas capturas de pequenos pelágicos, atuns costeiros e algumas espécies demersais. A simulação com Ecosim deu resultados similares, com base nos mesmos dados de capturas, o que demonstra resultados consistentes apesar de utilizar métodos alternativos.

Considera-se que este estudo foi bem sucedido como primeira tentativa de fazer um abordagem ecossistémica para avaliar os efeitos da pesca em Cabo Verde, mas é necessário investigar e ressolver alguns assuntos cruciais tais como os resultados contraditórios referente à biomassa dos peixes demersais e a possível sobre-exploração de predadores demersais e moreias. Considera-se de prioridade a definição de componentes da frota, melhorar as estimativas de esforço e CPUE, e o estudo da eficiência crescente, utilizando os dados disponíveis de captura e esforço.

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When undertaking a study that has to do with the ecosystem approach to fisheries, it is only natural that such a study builds upon the work undertaken by and collaboration of many persons and institutions.

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I dedicate this to

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Chapter 1. General Introduction

Our perception of the bountiful seas has changed much over time. In 1883, one of the best-known scientists, Thomas Huxley (Huxley, 1884 in Haddon, 2001), stated that:

> "I believe then that the cod fishery, the herring fishery, pilchard fishery, the mackerel fishery, and probably all the great sea-fisheries are inexhaustible; that is to say nothing we can do seriously affects the number of fish. And any attempt to regulate these fisheries seems consequently, from the nature of the case to be useless."

This view by Huxley was countered by Ray Lankester, who voiced his concerns for the removal of spawning stock and called for consideration of non-target species, since all species had their place in the complex interactions of living beings within their area (Botsford *et al.*, 1997). Despite appeals for ecosystem management at the end of the 20th Century and the development of multispecies models, few fisheries are actually managed on this basis (Sainsbury, 1998; Christensen, 2004).

Overall abundance was considered to be essentially unlimited and renewal from year to year constant. The explanation for the large fluctuations in catches from the traditional fishing grounds in northern European waters was the "migration theory" (Ulltang, 1998). According to the theory, change in adult migration behaviour in relation to physical oceanographic variability generated fluctuations in landings. Johan Hjort made a fundamental contribution to fishery science by explaining the fluctuations in landings by "year class" variations in stocks with a limited geographical distribution (local stocks), which was in contrast to the migration theory (Hjort, 1914 in Ulltang, 1998). The stage was set for a long debate on the relative importance of environmental effects and climate for the fluctuations of fish stocks (Botsford *et al.*, 1997; Jackson *et al.*, 2001; Jacobson *et al.*, 2001; Klyashtorin, 2001; Pauly *et al.*, 2002).

Regarding the effects of fishing on the ecosystem, the first known incident is in 1376 when the British Parliament received protests from fishermen concerned about the damage being done to their fisheries by trawling (Groot, 1984 in Humborstad, 2004). Centuries would pass until the first scientific survey in 1938 concluded that there was no evidence of long-term effects of trawling (Graham, 1955 in Humborstad, 2004). This view prevailed until recently, at least until the early 1990s, when trawling was considered to have little detrimental impact, or even a beneficial impact, analogous to the ploughing of fields. Recent results suggest that a more appropriate analogy would be the clear cutting of forests in the course of hunting for deer (Pauly *et al.*, 2002).

Present day reality is quite different in that many important fisheries resources world-wide can be characterised as seriously depleted or in danger of depletion due to poor management practices and excessive fishing pressure (Pauly *et al.*, 1995, 1998, 2002; Grainger and Garcia, 1996; Botsford *et al.*, 1997; Jackson *et al.*, 2001; FAO, 2002-a; Baum *et al.*, 2003; Christensen *et al.*, 2003; Myers and Worm, 2003). A recent assessment by the FAO found that around a quarter of the world's fisheries resources are considered "overexploited" and half are "fully exploited" (Garcia and Moreno, 2001; FAO, 2002-b). Unsustainable fishing practices coupled with an excessive level of investment in fishing capacity have resulted in serious degradation and low yields in the stocks of developed countries, creating new pressures on the resources of developing countries (Pauly *et al.*, 2002). These pressures are largely trans-national, highlighting the importance of regional and international aspects for resource management (Bonfil *et al.*, 1998; FAO, 2002-b; Pauly and Palomares, 2002).

The concept of sustainability has become an integral part of modern day fisheries management, but this concept has been criticised on numerous occasions. The reasoning behind these criticisms has been that it is difficult, if not impossible, to define sustainable harvest levels in highly complex and dynamic ecosystems (*e.g.* Sainsbury, 1998). Criticisms appear to be associated with the common practice, at present, of managing fisheries at the limit of collapse, maximizing resource utilisation and maintaining effort relatively stable. Another criticism stresses that the concept is fundamentally flawed, because there is little point in sustaining stocks whose biomass is but a small fraction of its value at the onset of industrial fishing (Pauly *et al.*, 2002). Rebuilding of marine ecosystems to previous historical levels is considered more important by these authors (Pitcher, 1996; Pauly *et al.*, 2002), but defining the previous level (*e.g.* the shifting baseline syndrome) and whether it is possible (*e.g.* habitat changes, regime shifts, catastrophic shifts, alternate states) is under debate. There is however no doubt that a drastic reduction of fishing capacity is necessary, globally.

So what is the reason for the numerous failures in spite of considerable efforts and investments in fisheries research and management? Two classical examples of collapses are the Peruvian anchovy fishery and the northwestern Atlantic cod stocks. But there is still much debate about the driving force that led to these and other collapses, although there is consensus that incorrect assessment advice and poor management procedures played a role (Walters and Maguire, 1996; Botsford *et al.*, 1997; Gascuel *et al.*, 1998; Longhurst, 1999; Sainsbury, 1998; Ulltang, 1998; Pauly *et al.*, 2002). A fundamental flaw in the management process has been termed the ratchet effect (Botsford *et al.*, 1997), where managers under political pressure will allow harvests to increase for their short-term benefits to society (jobs and profits) when fishery scientists cannot specify with certainty that the next increase will lead to over-fishing and collapse.

Stock assessment methods range from simple to highly complex approaches, depending on the type of data available or the school of thought (Sparre and Venema, 1992; Hilborn and Walters, 1992; Funk *et al.*, 1998; Quinn II and Deriso, 1999; see also[1]). Recent developments in

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[1] "Assessment and Management of New and Developed Fisheries in Data-Limited Situations", 21st Lowell Wakefield Fisheries Symposium, 22-25 October 2003, Anchorage, Alaska; http://www.uaf.edu/seagrant/Conferences/symposia.html#dls

stock assessment methods have placed much emphasis on uncertainty and risk analysis as this has been identified as a weak point (Punt and Hilborn, 1997; Hilborn and Liermann, 1998; Sainsbury, 1998). Another interesting aspect has been the development of meta-analytic, comparative and demographic approaches used in a Bayesian (or not) context in order to reduce uncertainty in stock assessment (Cortés, 1998; Hilborn and Liermann, 1998; McAllister *et al.*, 2001; Myers, 2001; MacKenzie *et al.*, 2003). Concurrently, procedures for the evaluation of a management process through simulating and testing have been developed (Punt and Hilborn, 1997; Sainsbury, 1998; Butterworth and Punt, 1999; McAllister *et al.* 1999; Sainsbury *et al.*, 1999). This includes a range of key aspects of the process such as the monitoring programme, data collection, stock assessment model and alternative hypotheses regarding the population dynamics of the stock in question. Other aspects considered are the specification of alternative management strategies, selection and evaluation of performance measures, feed-back system and decision rules and the evaluation of all these through simulation in order to create a robust operational management strategy.

1.1 Ecosystem Approach to Fisheries

The concept of fish stock became the core concept of theoretical developments and empirical studies during much of the 20th Century, referring to a basic population of fish that can be represented by life history parameters relating to the population rather than its individuals. Johan Hjort´s theory of "year classes", whose success dictated the observed fluctuations in landings, implied a basic population that led to the concept of stock (Degnbol, 2001). Research was generally structured according to species at the national and international level. The researchers involved became advocates of methods and measures that seemed best adapted for particular species that they studied, and the overfishing problem became thus divided (Graham, 1948 in Degnbol, 2001). Fisheries science focussed on maximising the yield of single species or stocks and Maximum Sustained Yield (MSY) became the central goal of management, ignoring the wider impacts of fishing such as accidental bycatch, habitat degradation and ecosystem attributes such as functionality and productivity (Botsford *et al.*, 1997; Jennings and Kaiser, 1998; Degnbol, 2001; Pauly *et al.*, 2002).

However, the scope of international fisheries management changed in the early 1990s when two new considerations entered the scene: the precautionary approach and the need to include considerations on the ecosystem effects of fishing into management (Degnbol, 2001). The need to reduce the alarming trend of depletion and degradation was recognized in many international fora, most recently at the 2002 World Summit for Sustainable Development (WSSD) in Johannesburg, which pledged to maintain or restore stocks to levels that can produce the maximum sustainable yield with the aim of achieving these goals for depleted stocks on an urgent basis and where possible not later than 2015 (FAO, 2003; Garcia *et al.*, 2003). This choice of the MSY as a goal is unfortunate, because it is in conflict with another resolution by the WSSD: to encourage the application by 2010 of the ecosystem approach to fisheries, noting the Reykjavik Declaration on Responsible Fisheries in the Marine

Environment (Article 30d). The problem is that MSY as a goal has been associated with many failures in management (Botsford *et al.*, 1997; Mace, 2001; Pauly *et al.*, 2002) and it appears not to be a good management objective in an ecosystem context, because it is achieved by eliminating the large predators and simplifying the system (Mendelssohn, 1980; Gislason, 1999; Hollowed *et al.*, 2000).

The term ecosystem approach to fisheries (EAF) implies the goal of conserving the structure, diversity and functioning of ecosystems through management actions that focus on the biophysical components of ecosystems (including protected areas) as well as fisheries management, which aims to meet the goals of satisfying societal and human needs for food and economic benefits through management actions that focus on the fishing activity and the target resource (not only maximising yield!) (FAO, 2003). Unfortunately, objectives such as maintaining high employment, high profits, and high stock sizes are often in conflict with one another in an ecosystem context (Gislason, 1999; Mace, 2001), thus the importance and difficulty of defining operational objectives and reference points.

1.2 Science and Tools for the Ecosystem Approach

According to Degnbol (2001), fisheries science is now facing two fundamental problems in order to adapt to current management requirements in the context of EAF.

- "Fisheries research is approaching the limits of cost efficiency relative to the value of fisheries and can still not deliver the goods in terms of numerical predictions."
- "The models and concepts of fishery biologists are becoming increasingly alien to stakeholders. This gap is not just a question of lack of understanding or education on the side of fishers but is rather associated with the basic scales at which the resource basis for fisheries is observed and understood."

Concerning the latter point, Degnbol (2001) refers to the global or large-scale approach by fisheries science in contrast to the detailed spatial and temporal scale of fishing activities. Degnbol (2001) puts forward two alternatives:

- "To internalise the issue in the same way as was done before when species interactions and uncertainty was internalised, that is by developing models with new layers of complexity which include all relevant processes and effects and thus enables ecosystem effects to be predicted within stochastic predictability."
- "To develop a fundamentally new approach which does not pretend to understand or measure causal relationships and all relevant processes in detail but identifies specific features which can be measured and indicate the pressures on the system."

It is quite obvious from the line of reasoning above that Degnbol considers the second alternative (*e.g.* ecosystem indicators) the most viable. However, the middle path with a blending of the two schools of thought, reductionist and holistic represented by these two alternatives, respectively, is another way forward and there are some examples of this.

The developments of methods and tools for the implementation of EAF has been an intensive field of research in recent years (Gislason *et al.*, 2000; Cury and Christensen, 2004). Multispecies and system models of varying complexity have been developed (Hollowed *et al.*, 2000). Many studies have dealt with the ecosystem effects of fishing in order to gain a better understanding of ecosystem dynamics and develop new complementary management tools, quantitative ecosystem indicators (Jennings and Kaiser, 1998). Current efforts are underway to develop the theory behind these indicators, considering changes in marine ecosystems from environmental, ecological and fisheries perspectives, including the robustness and usefulness of these proposed indicators (Jennings *et al.*, 2002; Rochet and Trenkel, 2003; Trenkel and Rochet, 2003). The goal is to define innovative or conjugate reference points that can serve as effective EAF management objectives and thus improve on the current state of fisheries resources.

Many question however whether it is possible to incorporate ecosystem objectives in management, as this would increase the uncertainty involved and predictions would be extremely difficult. Instead, it has been suggested that modelling efforts should include only crucial trophic interactions, indicators, or climate proxies and only if this decreases uncertainty (Livingston and Methot, 1998; Sainsbury, 1998; Beamish and McFarlane, 1999; Collie and DeLong, 1999; Hollowed *et al.*, 2000). A convincing argument by Christensen and Pauly (2004) is that ecosystem modelling is not intended to replace stock assessment, but rather complement it. In fact, ecosystem modelling often feeds on conventional single-species stock assessment results. The complementary use of these two types of tools, stock assessment and ecosystem modelling, represent tactical (setting of TACs) and strategic (policy exploration) thinking, respectively (Christensen and Pauly, 2004). It appears that this is also a question of changing paradigms amongst the stakeholders, including the people in industry, management and research.

1.3 Tropical Fisheries in (Mostly) Developing Countries

Fisheries science includes few contributions to the specific problems of tropical fisheries and the contribution of scientists from western industrialised countries can be characterised as the "tropicalization" of Beverton and Holt (Pauly, 1998), which is applying the same concepts and methods from temperate areas. However, as ageing of fish is difficult and prohibitively costly in most tropical fisheries, age-based models were converted to length-based models (Sparre and Venema, 1992). The most commonly used model (Jones Length Cohort Analysis) is nevertheless considered a poor alternative to age-based methods (Hilborn and Walters, 1992; Smith and Addison, 2003).

The relatively large number of species present in tropical fisheries and the importance of artisanal fishers in developing countries using a variety of gears and targeting numerous species is another problem, posing serious difficulties in terms of methodology. One approach used has been multispecies ecosystem modelling with Ecopath (Christensen and Pauly, 1993), which considers ecological groups rather than age structured population dynamics. This has resulted in numerous models that have been excellent as structuring tools, for hypothesis testing, identifying major gaps and research priorities as well as policy exploration, but their use in management is limited if not non-existent at present (Christensen, 2004).

Considering these limitations and difficulties in tropical fisheries, it is urgent to start on the application of management that is strongly precautionary and that can succeed in data-free or data-poor situations (Caddy and Mahon, 1995; Die and Caddy, 1997; Sainsbury, 1998; Caddy, 2002). It is particularly true that monitoring and research should be cost effective in developing countries. Putting this in simplified terms, a robust holistic approach would be more suitable rather than a complex reductionist approach. The developments in the field of quantitative ecosystem indicators for the formulation of new management rationality appear to be promising. Research in developing countries can make important contributions in this pioneering development, including experiences in the implementation of EAF management strategies.

1.4 The Cape Verde Archipelago

Ten major islands and various small islets, situated about 650 km west off the coast of Senegal, comprise the Cape Verde Archipelago. The total land area is 4033 km^2, consisting of a northern (Barlavento: windward) group of islands and a southern group (Sotavento: leeward) (Figure 1.1). The islands are of volcanic origin, rising from a depth of at least 3 000 metres and the continental shelves, generally narrow and irregular, are limited to a total area of 5 394 km^2 (Bravo de Laguna, 1985). The eastern islands Sal, Boavista, and Maio, form one system with a more extensive continental shelf compared to the other islands. On the other hand, the EEZ of Cape Verde covers an extensive area of 789 400 km^2 (Froese and Pauly, 2000), much of which is exploited by foreign fishing fleets only. The climate of Cape Verde is characterised by warm, relatively even temperatures (22^o - 27^{o}C) and very low and variable values of precipitation (about 400 mm per year).

Cape Verde became independent from Portugal in 1975 and has achieved a strong development performance record in areas such as health, education, economic growth and the establishment of a well-functioning democracy. Per capita GDP was estimated at US$1 260 in 2002 (population 440 000), one of the highest in Africa (www.worldbank.org). In 2000, agriculture and fisheries accounted for 10.9 percent of GDP, but these activities involved 53 percent of the active population (FAO/WFP, 2002). Fisheries accounted for 7 percent of export earnings and 1 percent of GDP. Agriculture is hampered by arid climatic conditions and erratic rainfall with production rarely covering more than 15 percent of the country's needs.

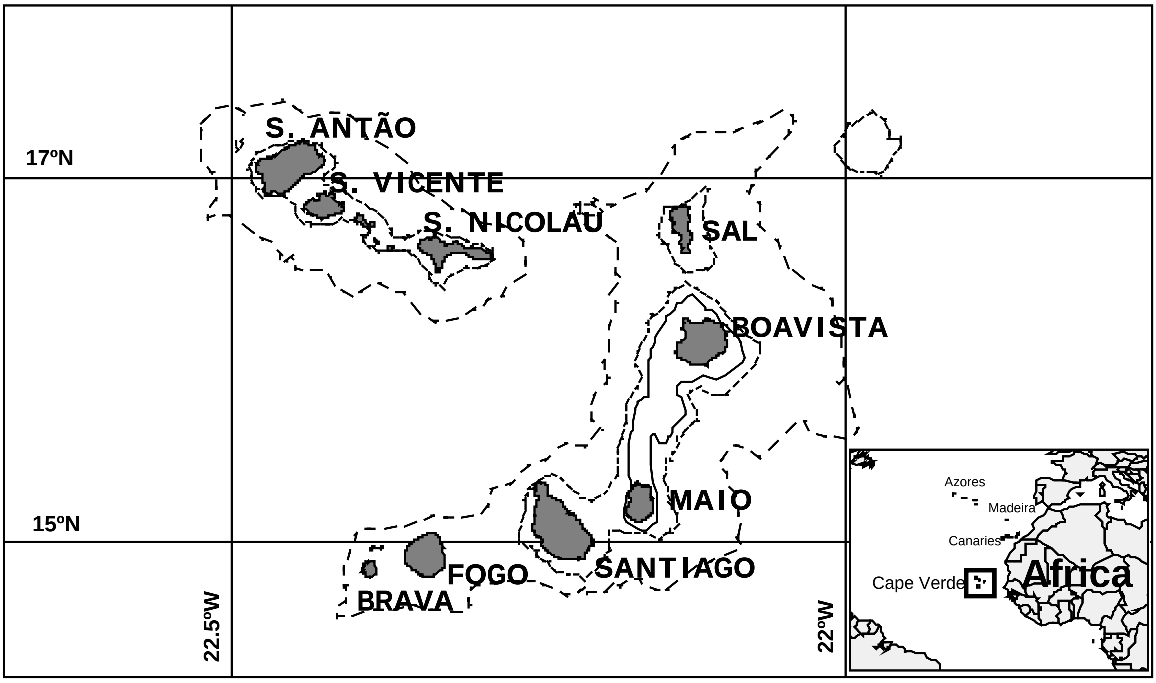

Figure 1.1 Map of the Cape Verde Archipelago including bathymetric lines corresponding to depths of 200, 1000, and 3000 metres.

Biogeography

The review by van der Land (1993) gives a general overview of the marine biota of the Cape Verde Islands, which provided most of the characteristics given in the following. The Archipelago is situated in the tropics and is rather isolated, far from the African mainland and separated by great depths. Moreover, marine habitats around the Cape Verde Islands are generally different from those along the West African coast, which are highly productive upwelling areas. These factors hamper the immigration of many organisms. On the other hand the Canary Current and the equatorial currents (North Equatorial Counter Current – NECC / North Equatorial Current – NEC) are expected to play an important role in immigration and colonisation. However, these distinct water masses result in occasionally adverse conditions (for either tropical or subtropical/temperate species), thus hampering the development of rich tropical biota.

There is some disagreement on the origins and age of Macaronesian[2] islands, but the shallow water fauna of the Cape Verde Islands is considered to be of relatively young age (less than 1 million years). Every island is considered to have risen independently from the ocean floor. In the case of Cape Verde, the eastern islands Sal and Boavista are older and much of the

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[2] Macaronesia: biogeographical province or region commonly accepted in ecology comprising Azores, Cape Verde, Canaries, and Madeira Islands.

volcanic topography has been eroded away by wind and time, resulting in more sandy sediments. Depending on what organisms are being considered, Cape Verde may be considered a biodiversity "hotspot" with a high level of endemism (*e.g.* Merino *et al.*, 2002; Roberts *et al.*, 2002).

Shelf conditions are characterised by predominantly sandy bottoms with rocky outcrops of volcanic rock, but there are also regions with carbonate rocky bottom sometimes shaped like reefs. In some places the bottom is more or less densely covered with calcareous nodules. The extensive shelf regions around Boavista and between Boavista and Maio would present ideal trawling grounds were it not for these outcrops of sharp rocks and nodules that fill up the trawls in minutes.

No true coral reefs exist along the West African coast or in the archipelagos of the Gulf of Guinea and Cape Verde, but there are a number of sites with rich coral communities (Longhurst and Pauly, 1986). In Cape Verde, "Bahia das Gatas" in the S. Vicente Island is one such example of a shallow protected lagoon with an abundant coral growth, but this is actually one of few exceptions to the rule. The exposure to strong northeasterly winds and associated turbulence as well as oceanic swells appear to restrict the growth of coral as well as seagrasses. Instead, calcareous algae cover large areas from the littoral zone down to great depth. Demersal fish fauna is remarkably similar to other tropical islands with coral reefs (van der Land, 1993).

The biogeographical unit Macaronesia is widely accepted in ecology, characterised by its relic laurisilva vegetation (Lloris *et al.*, 1991). There is some disagreement on the status of region or province whether it concerns botany or zoology, respectively. However, from the ichthyological point of view the term Macaronesia does not constitute a biogeographical unit, but rather a segment or faunistic transition zone where organisms of diverse source meet (amphiatlantic, cosmopolitan, circumglobal), having a clear component of durable species of northern origin (Atlanto-Mediterranean) and others of meridional origin (Equato-Guinean) (Lloris *et al.*, 1991). In relation to Cape Verde, this area is considered a "branch" of the Equato-Guinean province with some connection to the Madeira District (the Canaries and Madeira Islands primarily) (Lloris *et al.*, 1991). As referred above, this is in complete agreement with the expected influence of the Canary Current and equatorial currents (both NEC and NECC) in immigration and colonisation.

The North Atlantic Tropical Gyral Province

Longhurst (1998) pointed out the inadequacies of classical biogeography of the marine environment, because of the inherent difficulties and conflicting viewpoints depending on the approach used. The preceding section is an example of the difficulty in reaching consensus and achieving operational concepts in classical biogeography. The availability of remote sensing data was a major factor in reactivating the search for a satisfactory way of defining ecological biogeochemical provinces (BGCPs), which is the approach used by Longhurst (1998).

Using this approach, the Cape Verde Islands are classified as part of the North Atlantic Tropical Gyral Province (NATR), which is a region with a consistently low and uniform surface chlorophyll field with a seasonal cycle of small magnitude. This region includes the continuation of the offshore Canary Current south of the Canaries and the flow into the North Equatorial Current (NEC). Only in the Canary Basin do we find patches of enhanced chlorophyll values, caused by the island effect downstream of the individual Canary Islands and in a few detached eddies and jets of upwelled water from the inshore Canary Current. There are indications that these island effects exist also in Cape Verde, but further studies are necessary (Stroemme *et al.*, 1982; Almada, 1993, 1994).

Where the Canary Current detaches from the coast, the convergence zone between it and poleward flow from the south is characterised by the persistent giant filament often observed seawards from Cape Blanc at about 21° (Mauritania). This giant filament off Mauritania, reaching as much as 450 km offshore, may appear as a persistent chlorophyll feature or it may be seen principally in the temperature field. The influence of this giant filament is seen to reach the Cape Verde Islands, possibly leading to an import of nutrients and chlorophyll production. Nevertheless, the primary production is relatively low with a mean daily estimate of 675 mgC m^{-2} day^{-1}, based on remote sensing data (www.seaaroundus.org).

Almada (1994) gives a good characterisation of oceanographic conditions in Cape Verde, which vary seasonally as a result of the displacement of the Azores and Santa Helena anticyclones. From December to June, the Azores anticyclone diminishes in strength and shifts to a more southerly position and the St. Helena anticyclone increases in strength. Under these conditions, the northerly winds intensify and the whole Cape Verde Archipelago is under stronger influence from the Canary Current. These waters are generally colder (≤ 23° C), transporting nutrients from the upwelling areas in northern Africa and creating more favourable conditions for primary production around the islands most affected by this north-eastern current (Barlavento). From July to November, the St. Helena anticyclone diminishes in strength and the Azores anticyclone intensifies and shifts position to a more northerly position (about 5º). As a result of these seasonal changes only the northern islands remain under the influence of the Canary Current (but somewhat weaker). The southern islands (Sotavento) come under the influence of the North Equatorial Current (NEC) and North Equatorial Counter Current (NECC), causing warmer waters to reach these southern islands (24° - 27°C). These warmer waters create ideal conditions for pelagic fish such as tuna, skipjack tuna in particular, coinciding with the season for the baitboat fishery. Thus, during the second half of the year the Archipelago may be divided in a northern and southern area of distinct oceanographic characteristics.

1.5 Objectives and Relevance

Marine life has often been studied in Cape Verde since the visit by Darwin onboard the vessel H.M.S. Beagle. Most of the studies have been undertaken from shore or on board vessels during their typically short visits. The position of the islands on the transatlantic route, as the ships took advantage of the trade winds, benefited Cape Verde on many occasions. In recent decades, this changed somewhat as several extensive surveys have been undertaken specifically to explore fisheries resources. An introduction will be given on the existing fisheries information in the following chapter, but this is to indicate that there are numerous studies and characterisations. Thus, groups of organisms such as corals, molluscs, algae, and fish are reasonably well known (Franca and Vasconcelos, 1962; van der Land, 1993; Reiner, 1996; Froese and Pauly, 2000). Phytoplankton and zooplankton communities have also been studied on several occasions (Paiva, 1963, 1971; Ribeiro, 1964; Marques, 1974; Neto and Paiva, 1981; Marques *et al.*, 1997). However, it has generally been difficult to identify and locate many of these studies as they have often been published as "grey" literature and are dispersed worldwide, depending on the institution that carried out the study. Emphasis was placed on fisheries related sources and references, which were more easily accessible. In this respect, the compilation of studies by Carvalho *et al.* (1999) is an excellent introduction to the current state of knowledge in Cape Verde.

A considerable part of this study consisted in compiling existing information on the coastal ecosystem in Cape Verde in order to apply an ecosystem approach to fisheries assessment. The purpose was to take advantage of existing data and carry out analysis using alternative tools and methods. More importantly, the present study includes the first attempts to undertake ecosystem modelling and the application of various ecosystem indicators (*e.g.* size spectra, biomass, etc.) to determine their usefulness for management in tropical areas such as Cape Verde. Assessment of small pelagic (bulk biomass) was also undertaken with conventional production modelling in order to complement the results of ecosystem modelling.

Cape Verde is a tropical scenario with the usual problems related to limited data and inconsistent, irregular sampling procedures and intensity. Hence, a pragmatic approach was adopted of using various methods comparatively, contributing to existing knowledge in Cape Verde and tropical areas in general. Also important was the identification of major gaps in our knowledge and priorities for future fisheries research. Emphasis was placed on studying ecosystem dynamics, including possible changes that have occurred since 1964, the year of the first quantitative trawl survey, and the possible effects of fishing. Thus, the null working hypothesis is that there has been no change over recent decades. If there has been change, then we attempt to answer the following questions:

a) Is the change observed due to fishing or environmental effects or possibly a combination of both?

b) Has there been a change (decrease or increase) in biomass for specific species or groups?
c) What have been the consequences of these changes in terms of trophic interactions?
d) What is the current level of fishing pressure on the ecosystem and the different fishery resources and what are the implications for potential harvest?

These are essential questions in order to improve on the present understanding of the Cape Verde coastal ecosystem, its form and functioning, with the prime objective of contributing to the sustainable management of its resources, bearing in mind the crucial importance that fisheries has in an island country such as Cape Verde.

Chapter 2. Fisheries and Survey Data

2.1 Introduction

Fishing activities in Cape Verde probably started together with its discovery and settlement by African slaves in the 15th Century. Historical accounts of frequent attacks by English, French, and Dutch pirates were occasions for slaves to run away to the remote interior regions of Santiago Island where communities of free Africans were formed as early as the mid 16th Century (Almeida, 1997). These people became known as "badius" from the Portuguese word for vagabond or runaway. They survived on subsistence agriculture, which is possible on Santiago Island, probably supplemented with seasonal fishing activities.

On a larger scale, the American whaling industry began to recruit Cape Verdeans as early as the 18th Century, starting an all-male pattern of immigration to America. Before 1750, many crewmen aboard Nantucket whaling ships were Cape Verdeans and during the period from 1825 to 1875, an average of 100 American whaling ships called at Cape Verde each year (Almeida, 1997). Many of the best harpooners, steersmen, and all-round whalemen had for long been Portuguese-speaking Africans (Almeida, 2004).

At the local level, the first tuna fishing and canning company was established in 1927 and since then, several small companies were established in various islands, producing canned tuna, lobster and dried fish for domestic and export markets (Massuti, 1965-a). The method used for catching tuna was fishing with live bait using pole and line from relatively small and ill-equipped vessels. In 1963, Castro and Bastos (1967) undertook a mission to Cape Verde and mention that several French vessels exploited the abundant lobster resources using traps. They expressed concern about the fact that many of the French vessels were exploiting these resources illegally as only 3 vessels were authorised to do so. Vieira (1985) refers to the exploitation of coral over the period from 1850 to 1900, which was initiated by the Spanish and continued by the Italians. Apparently, this exploitation was intense and quickly became unprofitable due to the limited resources and unsustainable fishing methods (van der Land, 1993).

In relation to artisanal fisheries, an estimated 1 443 fishermen and 345 boats were active in the beginning of the 1960s (Massuti, 1965-a). The methods employed were primitive, consisting in handline fishing from small wooden boats powered by oars and sails, with few having motors. However, the interest in this sector of the fisheries and the livelihood of the people involved was limited. Instead, the primary goal was to develop industrial fisheries and overcome the existing limitations in relation to lobster and tuna fisheries, in particular (Frade, 1954; Postel, 1954; Massuti, 1965a; Castros and Bastos, 1967). The situation changed in the early 1980s in the context of a FAO/UNDP technical assistance project, which gave priority to artisanal fisheries (Hanek *et al.*, 1984).

Oceanographic studies concerning Cape Verde date back to historical times. For example, the Cape Verde Archipelago has been the target or on the route of numerous international surveys. These visits started with the famous ocean expeditions of the 19th Century, including the H.M.S.[3] Beagle with Charles Darwin as the expedition naturalist (1832), the United States Expedition (1838), the English vessel H.M.S. Herald (1852), the English vessel H.M.S. Challenger (1873), the French vessel Talisman (1883), Yacht Princesse Alice of Monaco (1901), and the French vessel President Théodore Tissier (1936) (Reiner, 1996). Generally, the objective was to gain a better understanding of the oceans and the living and non-living resources therein. More recently, more emphasis has been placed on the study of fishing possibilities in Cape Verde, specifically.

In the following, an introduction is given on the fishery resources in Cape Verdean waters, making use of the available information from surveys and fisheries statistics.

2.2 Materials and Methods

Fisheries data

Fisheries statistics are available starting in 1981 with the implementation of a national statistical collection system for fisheries (Shimura, 1980, 1984). The FAO played a key role in establishing this system as well as providing technical assistance. One of the first major tasks was a nation-wide census survey covering fishermen, boats, gears, catches, and effort as well as a variety of socio-economical factors (Hanek *et al.*, 1984).

The methodology used for subsequent estimation of catch and effort has been regular census surveys (not as complete as the first one) and random stratified sampling for artisanal catch and effort estimates, considering gear, landing site, island, and month. This is the classical approach of estimating mean catch by gear and extrapolating to total catch by landing sites using effort information (Stamatopoulos, 2002), adopted to the conditions in Cape Verde, which has been implemented by the FAO in numerous countries. Sampling coverage has improved progressively over time in an attempt to cover both the larger and smaller landing sites in each island (Hanek *et al.*, 1984; Tavares, 1989, 1994; Monteiro, 1999-a; Bellemans and Monteiro, 2000). It is important to point out that not all islands were sampled in the beginning. In the case of industrial or semi-industrial fisheries, the method of obtaining statistics has been by full enumeration, where information on catch and effort has been compiled for every fishing trip.

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[3] Her Majesty's Ship

The responsibility for the collection, processing, and dissemination of fisheries statistics was given to the "Instituto Nacional de Investigação das Pesca," created in 1986, which was substituted by the "Instituto Nacional de Desenvolvimento das Pescas (INDP)" in 1992. For previous statistics on fisheries catches, Watanabe (1981) made a compilation for the period from 1956 to 1979 based on various sources.

Table 2.1. List of surveys undertaken in Cape Verde waters, indicating the survey objective and gear used as well as the main reference.

Research Vessel	Year	Objective	Method	Reference
Gérard Trèca	1952	Exploratory survey: tuna	Trolling	Postel, 1954
Baldaque da Silva	1957 1958 1959	Exploratory survey: fishery resources	Pole and line, handlines, nets, lobster traps	Costa, 1962; Franca *et al.*, 1962; Soares and Vasconcelos, 1962
Walter Herwig	1964	Exploratory survey: fishery resources	Pelagic and bottom trawls	Massuti, 1965-b
Ernst Haeckel	1976	Exploratory survey: demersal fish	Bottom trawl	Danke and Koch, 1987
Dr. Fridtjof Nansen	1981	Exploratory survey: fishery resources	Acoustics, pelagic and bottom trawls	Stroemme *et al.*, 1982
Playa de Tamaris	1982	Exploratory survey: large pelagics	Longline	Torres, 1982
Fengur	1984 1985	Demersal fish survey	Bottom trawl and handline	Magnússon and Magnússon, 1985, 1987-a, 1987-b
Fengur	1988	Demersal fish survey	Bottom trawl	Palsson, 1989
Islândia	1994	Demersal fish survey	Bottom trawl	Thorsteinsson *et al.*, 1994
Islândia	1994 1995 1996 1997	Commercial fishing: demersal fish	Bottom trawl	INDP
Islândia	1995 1996 1997	Exploratory survey: small-scale resources	Handline	Oddson and Monteiro, 1995; 1996; 1997
Islândia	1997	Exploratory survey: tuna	Longline	Giani *et al.*, 1999
Capricórnio	1997	Acoustic survey: small pelagics	Acoustics	Marques *et al.*, 1997
Arquipélago	2000	Deep-water resources	Deep longline	Menezes *et al.*, 2001

Survey data

During the great oceanic expeditions of the 19th Century and early 20th Century, Cape Verde was visited on a number of occasions. As a result of these explorations, descriptions of the findings were made available, but the surveys that took place after 1950 started to explore fishing possibilities in the Cape Verde Archipelago as the main objective. A list of these surveys is given in Table 2.1, but this should not be regarded as complete as various international oceanographic surveys were not included such as the Capricorne and CANCAP surveys (van der Land, 1987; Almada, 1994). Instead, emphasis is placed on surveys that consider fishery resources.

Data standardisation

Trawl survey catches were standardised (kg/nm^2) using the swept-area method taking into consideration haul duration, average trawling speed, and a fixed value for the horizontal net opening (Table 2.2). This standardisation was not possible on a haul-by-haul basis, as data on speed was often lacking and the net opening was not measured.

Table 2.2. Gear characteristics used in the standardisation of survey catches.

	Year	Vessel	Net opening (m)	Velocity (knots)	Swept area per hour (nm^2)
Survey	1964	Walter Herwig	16	3.75	0.032
	1976	Ernst Haeckel	16.5	4.5	0.040
	1981	Fridtjof Nansen	18.5	3.75	0.037
	1985	Fengur	8.5	3	0.014
	1988	Fengur	16	3.2	0.028
	1994	Islândia	16	3.2	0.028
Fishing	1994 –1997	Islândia	16	3.2	0.028

2.3 Results and Discussion

Fisheries data

The evolution of total catches in Cape Verde show an increasing trend over the period from 1950 to the present, but it is difficult to assess how strong this increase has been (Figure 2.1). For the period before 1980, the estimates from Watanabe (1981) appear to be more reliable, compared to the FAO estimates, as these were based on a compilation from various sources. FAO total catch estimates for the period from 1981 to 1985 were relatively high and were based on the results of the project that was implementing a statistical collection system (Hanek *et al.*, 1984). These high catch estimates appear to have been the result of extrapolation errors for the catches of S. Antão and S. Vicente Islands between 1981 and 1983, in particular (Carlos Monteiro[4], *pers. comm.*). From 1986 onwards, the INDP estimates are considered reliable, providing the FAO with the same estimates.

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[4] Head of the Statistics Department, INDP

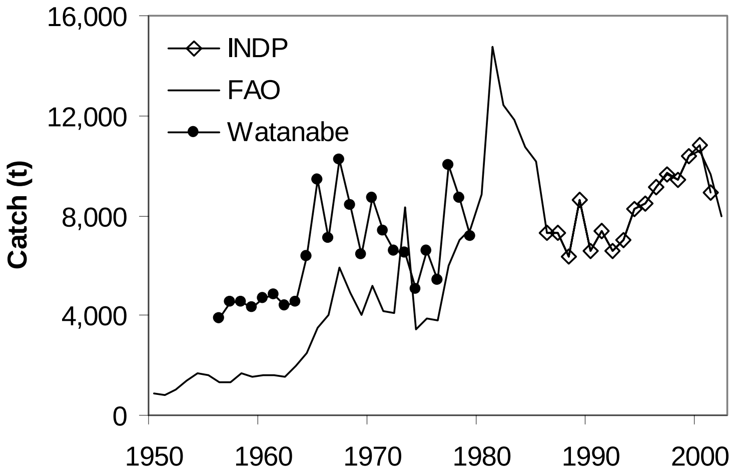

Figure 2.1. Total catch estimates in Cape Verde based on different sources (see text for explanation).

The total catch estimates for the period from 1981 to 1985 were revised in an attempt to correct them. This was important as the first extensive demersal surveys were undertaken in this period, making it essential to have a reliable source of fisheries data for the same period. Estimated catch rates in the artisanal fishery show that these were much higher in S. Antão by a factor of two to three in relation to the other islands (Hanek *et al.*, 1984) (Table 2.3). Tavares (1989) showed that total catch estimates were sensitive to extrapolation on the basis of boats, ports, or island, which shows that sampling coverage was not adequate for this time period. Various sources were consulted (Hanek *et al.*, 1984; INDP, 1986; Tavares, 1989; Medina and Tavares, 1992; Carvalho, 1994), but as the raw data were not available, a pragmatic and simple approach was used to correct the series. The overall average catch rate for the period 1981 to 1984 was 7.2 tonnes per boat per year and this was used to adjust the catches in S. Antão in 1981, 1982, and 1983 as well as the catches in S. Vicente in 1983, based on the number of boats fishing. Figure 2.2 shows that this approach had a strong impact on the total catch estimates between 1981 and 1983, but these are still relatively high considering that fisheries were under-developed at the time.

Based on the revised time series, catches have varied from about 7 000 to 10 000 tonnes per year over the last two decades. Artisanal catches account for a substantial proportion of the total catches (≈ 62%), but this has varied over time (Figure 2.3). The increasing trend for the period 1986 to 2000 was initially the result of increasing industrial catches, but the importance of artisanal catches has increased in recent years.

Table 2.3. Estimated catch rates in the artisanal fishery given for each island (tonnes per boat per year). Values given in parenthesis were considered erroneous and adjusted using an overall average of 7.2 tonnes per boat per year.

Island	1981	1982	1983	1984
S. Antão	(16.9)	(12.4)	(12.9)	7.6
S. Vicente	6.9	5.8	(14.4)	7.4
S. Nicolau	7.0	6.4	9.4	6.7
Sal	7.1	6.8	7.4	6.2
Boavista	6.4	3.1	3.1	6.9
Maio	7.1	9.5	16.4	6.7
Santiago	3.9	2.7	4.8	4.3
Fogo	1.4	2.5	7.2	5.8
Brava	7.1	6.7	7.6	5.9

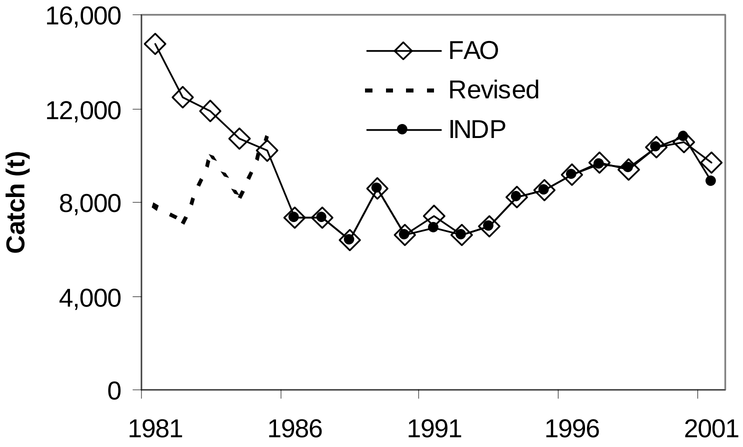

Figure 2.2. Revised total catches for the period 1981 to 2001.

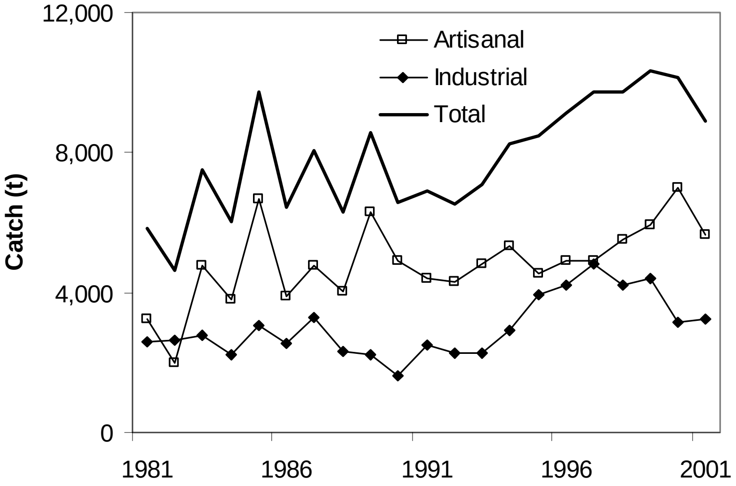

Figure 2.3. Artisanal and industrial fishery catches in Cape Verde.

Catch per unit of effort (CPUE) has been relatively stable in the artisanal fisheries, but this was not the case in the industrial fisheries (Figure 2.4). It is important to note that effort is measured differently for these two categories. In the artisanal fisheries effort is measured as fishing trips that are usually, but not always, equivalent to fishing days. In the industrial fisheries effort is measured in fishing days as trips have a longer duration. There was a clear decrease in industrial CPUE at the same time as effort was increasing (Figure 2.5). Also, the decrease in catches in 2001 (Figure 2.2 & 2.3) was the result of less effort in both the artisanal and industrial fisheries (Figure 2.5).

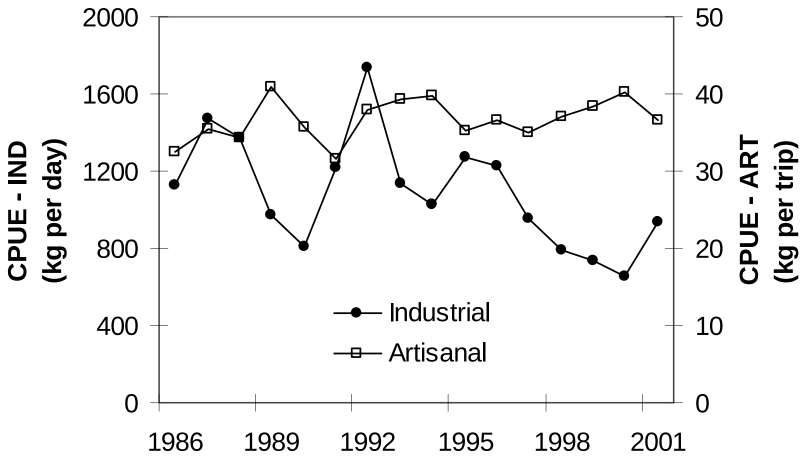

Figure 2.4. Artisanal (Art) and industrial (Ind) catch per unit of effort (CPUE).

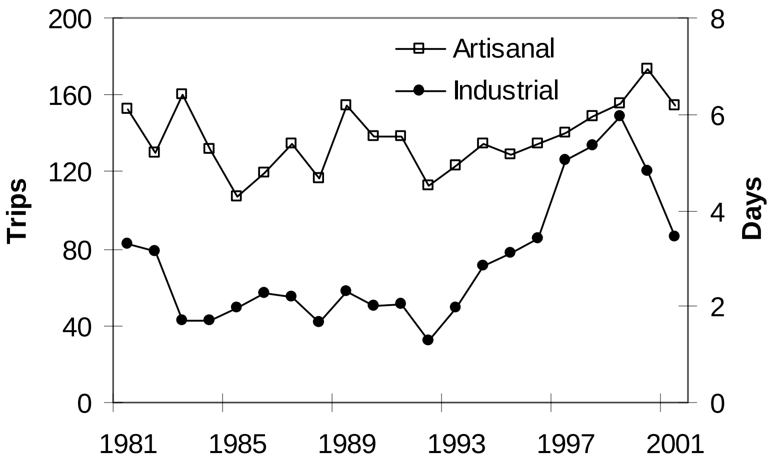

Figure 2.5. Artisanal and industrial effort (in thousands). In the artisanal fisheries effort is measured as fishing trips, while in the industrial fisheries it is measured in fishing days.

Catches are dominated by pelagic and/or migratory species such as tuna and various small pelagic species, accounting for around 80 percent. Important large pelagic species are yellowfin tuna, *Thunnus albacares,* and skipjack tuna, *Katsuwonus pelamis,* as well as wahoo, *Acanthocybium solandri*. Small pelagics are dominated by *Decapterus macarellus*, *Selar crumenophthalmus*, and *Spicara melanurus* in order of decreasing importance, although catches of the latter have increased in recent years (Appendix A). Relative species composition has been more or less stable in the artisanal fisheries (Figure 2.6). There is some doubt about the quality of the species composition data in the beginning of the period, but there appears to be a tendency for an increasing importance of demersal species and a decreasing importance of tuna.

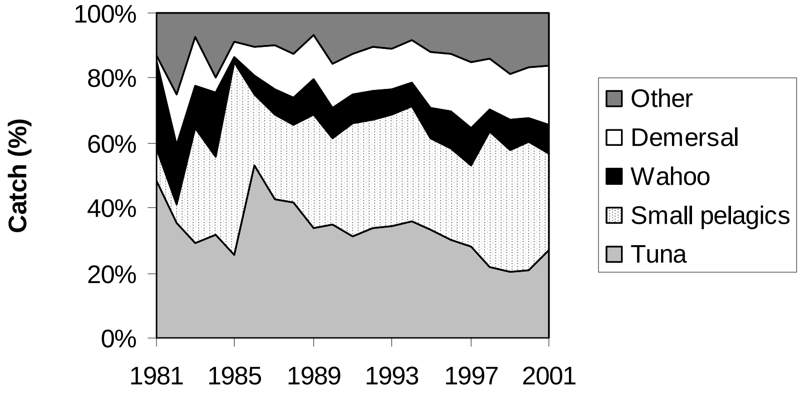

Figure 2.6. Composition of artisanal catches by species groups.

On the other hand, the species composition of industrial catches has changed drastically over the last two decades (Figure 2.7). Small pelagics, mostly *Decapterus macarellus* (or mackerel scad), have substituted tuna as the most important species group in the catches. This dramatic change was driven by a shift in target away from tuna in the industrial fishery. Two large Japanese freezer vessels started buying mackerel scad in the early 1990s to be used as bait elsewhere by the Japanese fleet (Almada, 1997). Mackerel scad has traditionally been fished for local consumption and is used as preferred bait in the tuna pole and line fishery. But the good prices offered by the Japanese and the introduction of small semi-industrial purse seiners (≈ 11m) in the same period (Bouwsma, 2003-b) made mackerel scad more attractive. This Japanese market disappeared however in the late 90s, which resulted in the increasing importance of tuna in recent years. However, the situation did not revert back to a reliance on tuna as market conditions continue to be difficult.

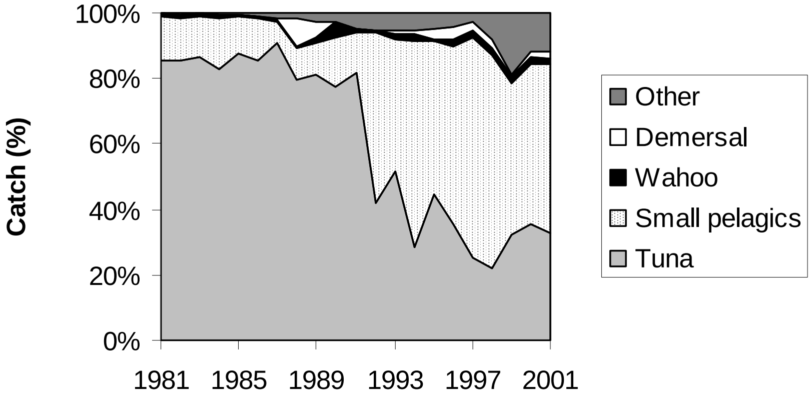

Figure 2.7. Industrial catches by species groups.

Another observed change in the industrial fisheries was that catches of "other" species have increased steadily over the last decade. These consisted mostly of small tuna such as *Auxis thazard* and *Euthynnus alletteratus*, which are fished with purse seine, the same gear used for scads.

Since the first national census survey was carried out in the period 1981 to 1983 (Hanek *et al.*, 1984), the number of artisanal fishers has increased moderately from about 3 300 to 4 280 in 1999. The number of boats involved in the artisanal fisheries has varied between 1 200 and 1 400, which is consistent with the moderate increase in terms of effort (Figure 2.5). At the same time, there has been a steadily increasing motorisation of artisanal boats, which has most probably increased the efficiency of these small wooden boats, ranging from 4 to 6 metres in length (Figure 2.8).

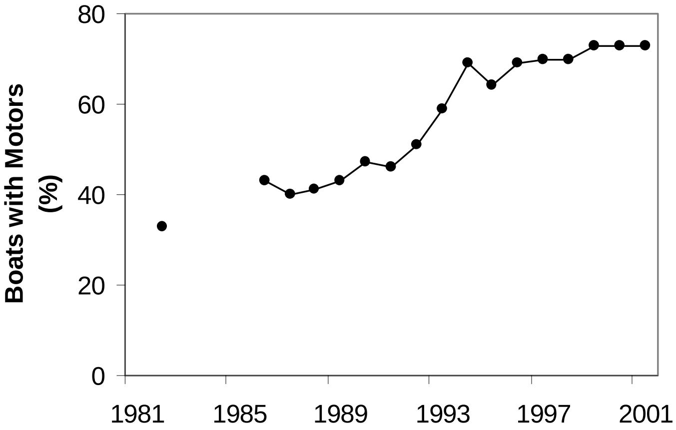

Figure 2.8. Motorisation in the artisanal fisheries.

Previous sections have referred to industrial fisheries, but this can be misleading. Vessels vary in length from 7 to 22 metres and only 60 percent of this fleet can be considered active and in operation. Out of a total of around 60 vessels, there are 8 larger vessels of around 22 metres, most of which are 40 years or even older. The majority of the vessels are relatively small glass fiber boats of around 11 metres, including the 13 vessels that were acquired by the government 12 years ago for the semi-industrial purse seine fishery. The term semi-industrial would be more precise, but as these fisheries are commonly referred to as industrial fisheries, the following sections will maintain the official denominations (INDP, 2003).

Survey data

Table 2.1 shows that there have been a considerable number of surveys, providing a rich source of information. These surveys have covered a wide range of objectives and sampling methodology and gears have varied considerably over time. Thus, although these surveys are a source of valuable information, the available data is fragmentary and not consistent over time. Earlier surveys tended to be exploratory in nature, providing qualitative data, whereas the more recent surveys are quantitative, including objectives such as determining the biomass of important pelagic and demersal fish species. In relation to demersal trawl surveys, one should consider differences in gear and vessel characteristics as well as sampling procedures and intensity when comparing with more recent surveys. In the following, the general results of these surveys are presented according to resource type, giving more attention to quantitative data.

Large pelagic resources

The French vessel "Gérard Trèca" undertook an exploratory survey for tuna in February 1952 (Postel, 1954). Tuna has probably always been an important catch of artisanal fisheries in Cape Verde, but the objective of this survey was to study the feasibility of industrialised fishing for tuna. The tuna fishing companies operating in Cape Verde were mainly small and ill equipped, although the prospects for developing an industrial tuna fishery, using pole and line, were considered good (Frade, 1954; Massuti, 1965a; Castros and Bastos, 1967).

Dr. Postel, who was in charge of the survey, operated under the hypothesis that the waters off the Cape Verde Islands may constitute "epicentres" of high tuna abundance from which a seasonal migration occurs to the other regions in Northwest Africa. Although two dense tuna schools were observed, this can be considered non-conclusive. This survey suffered from bad weather conditions, which is normal for that time of year and is the determining factor for the seasonality of tuna fisheries in Cape Verde (June to November). Interestingly, Postel points out that oceanographic conditions in Senegal are similar to Cape Verde with a time delay of three months. It has since been determined that the Cape Verde Archipelago is in fact on the migratory routes of yellowfin (*Thunnus albacares*) and skipjack tuna (*Katsuwonus pelamis*) as well as being a nursery ground for both species (Bard *et al.*, 1993; ICCAT, 2001-a, 2001-b).

Most of the tuna catches of the "Gérard Trèca" were of yellowfin (*Thunnus albacares*) and skipjack tuna (*Katsuwonus pelamis*), which have always been the most important tuna species in the catches (INDP, 2003). Catches of other species such as bigeye tuna (*Thunnus obesus*) and various billfish species are almost non-significant. However, it is important to point out that industrial or semi-industrial fishing for tuna in Cape Verde has been dominated by pole and line fishing, since the beginning of the 19th Century.

The Portuguese vessel "Baldaque da Silva" visited Cape Verde three times in 1957, 1958, and 1959, undertaking a wide variety of studies on hydrography, tuna, lobster, and demersal fish resources (Costa, 1960, 1962; Franca *et al.*, 1962; Franca and Vasconcelos, 1962; Soares and Vasconcelos, 1962). One of the objectives was to study the feasibility of industrial tuna fishing using pole and line. The results were not encouraging, but the vessel was not suitable for this gear.

Two exploratory surveys have been undertaken using longline (Table 2.1). The Spanish "Playa de Tamaris" used both surface and bottom longline in 1982 to study the prospects of fishing for large pelagics (tuna and billfish) and sharks (Torres, 1982). The results were rather poor and of low value, with sharks dominating both the surface longline catches (about 80%) and bottom longline catches (about 65%) (Table 2.4). Bearing in mind that it was the first survey of its kind, Torres (1982) concluded that industrial longline fisheries did not appear to be feasible in Cape Verde and recommended small-scale exploitation.

Table 2.4. Species composition of longline survey catches (Torres, 1982).

Gear	Species	Proportion (%)
Surface longline	Prionace glauca	25.1
	Carcharhinus brevipinna	21.0
	Carcharhinus obscurus	14.4
	Galeocerdo cuvier	8.8
	Xiphias gladius	16.0
	Thunnus obesus	4.5
Bottom longline	Mustelus mustelus	65.0
	Muraena helena	14.6

Another longline survey was undertaken by the Cape Verdean "Islândia" in 1997 with the objective of exploring longline fishing for tuna, bigeye tuna in particular (Giani *et al.*, 1999). In spite of the objective, no bigeye catches were obtained. Instead, catches were dominated by yellowfin tuna, swordfish (*Xiphias gladius*) and various shark species. Catch rates were considered reasonable (3% of fished hooks), but this was based on a comparison with asian longliners (2%). A direct comparison with the catch rates of the "Playa de Tamaris" was not possible, but the "Islandia" results appear to be poorer. Catches in terms of species composition were clearly different, which may be due to gear deployment. Most of the gear was set at a depth of around 100 to 250 metres in the "Islândia" survey, whereas the "Playa de Tamaris" set the gear closer to the surface or on the bottom, depending on gear type.

Small pelagic resources

Two acoustic surveys have been undertaken to determine the biomass of small pelagics, the mackerel scad (*Decapterus macarellus*) in particular. The Norwegian vessel "Dr. Fridtjof Nansen" undertook the first acoustic assessment in 1981 and the Portuguese vessel "Capricórnio" did the same in 1997 (Table 2.1). The 1981 estimate of total fish biomass estimate was around 100 thousand tonnes, including both pelagic and demersal species (Stroemme *et al.*, 1982). Estimates of the pelagic component ranged from 50 to 65 thousand tonnes comprised primarily by *Decapterus macarellus, Decapterus punctatus, Caranx rhonchus*, and *Sardinella maderensis*. Most of the biomass was concentrated in the narrow shelf areas around the islands, with the more extensive shelf off Boavista accounting for about ¾ of the total fish biomass.

In contrast, the 1997 estimate of total pelagic fish biomass was only around 15 thousand tonnes, comprised primarily by *Decapterus macarellus* and *Decapterus punctatus* (Marques *et al.*, 1997*).* Both vessels were relatively large and had difficulties in prospecting because of the narrow shelves around the islands. Progress in technology and data analysis over the period may be part of the explanation for this large difference in acoustic estimates, which have become more reliable with time. However, fishery exploitation may have played a role in reducing the biomass of small pelagics. Subsequent chapters will attempt to elucidate this point, considering its importance in terms of fisheries and ecosystem functioning.

Demersal fish resources

The first survey using a bottom trawl was undertaken in 1964 with only 6 stations sampled. Since then, bottom trawl surveys have taken place sporadically and they have varied in terms of objectives as well as sampling gear and methodology. For example, the 1964 historic survey in Cape Verde was a detour, the main objective being a survey of the Gulf of Guinea area (Massuti, 1965-b), and the 1976 survey was in fact exploratory in nature (Danke and Koch, 1987). All the surveys included sampling stations in the Boavista-Maio shelf, but sampling of the northwestern islands (S. Vicente and S. Antão) and southern islands (Santiago, Fogo, Brava) was occasional (Figure 2.9). This is related to the fact that bottom trawl surveys are difficult in Cape Verde due to the very narrow shelves and difficult bottom conditions, except for a limited area south of Boavista Island. The "Dr. Fridtjof Nansen" survey in 1981 covered most of the archipelago, but the more recent surveys in 1984, 1985, 1988, and 1994 had more intensive sampling. It was however not possible to recover the data of the 1984 survey (in spite of contacts in Cape Verde and Iceland), which was unfortunate as this was the first truly extensive quantitative demersal fish survey.

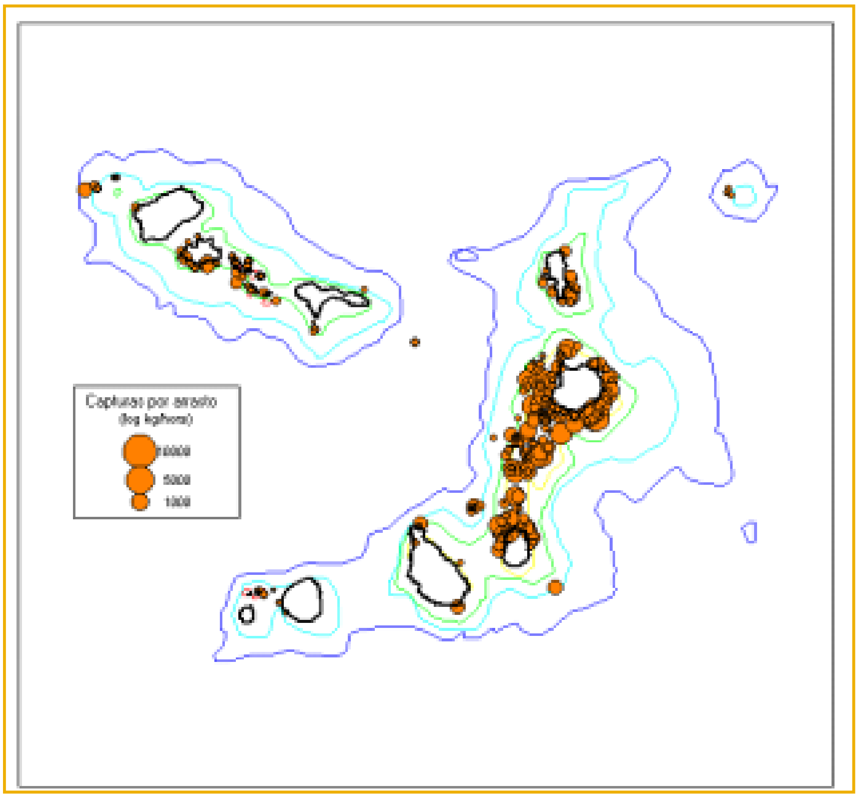

Figure 2.9. Overview of sampling by bottom trawl carried out by research surveys (Table 2.1). Each dot corresponds to a haul and its catch (log kg per hour).

Total biomass of demersal fish was estimated, based on the results of the surveys since 1981. These estimates range from 30 to 35 thousand tonnes in 1981, 43 thousand tonnes in 1984/1985, and around 15 thousand tonnes in 1988 and 1994, where the demersal fish biomass was most abundant in the Boavista-Maio shelf (about 70%).

As the available data was limited, the results of commercial fishing by the "Islândia" were included (available for 1994 to 1997). This vessel belongs to the Fisheries Institute in Cape Verde (INDP) and was undertaking prospective fishing during the referred period. The area of fishing was restricted to the suitable trawling grounds off Boavista and to a lesser degree, some grounds near the homeport of the vessel, Mindelo in S. Vicente Island (Figure 2.9). However, the vessel and gear were the same as in the scientific surveys of the "Islândia", thus making it comparable.

There is a clear trend for decreasing catches over time, which is particularly evident when considering survey data (Table 2.5). This trend is in general consistent with an observed decrease in the biomass of demersal fish stocks (Monteiro, 1999-b). As most of the stations were located in the Boavista-Maio shelf, the mean survey catches are presented specifically for this area also (Table 2.5; Figure 2.10). There is nevertheless some doubt about the data from the "Ernst Haeckel" survey in 1976 as the catches were extreme. Also, 1976 catches were dominated by *Trachurus picturatus*, which has never been the dominant species in successive surveys (Stroemme *et al.*, 1982; Vieira, 1985).

Table 2.5. Catch rates and abundance of demersal fish over time in Cape Verde. Trawl surveys and commercial fishing are included and presented separately for the whole area and the Boavista-Maio shelf system. 1984 survey values were not available by area. (Mean catch: kg per hour; Biomass: t per nm^2).

		Total Area			Boavista-Maio Area		
	Year	Stations	Mean Catch	Biomass	Stations	Mean Catch	Biomass
Surveys	1964	6	812	25	6	812	25
	1976	16	6269	156	16	6269	
	1981	18	1046	28	16	1136	30
	1984	77	517	38			
	1985	37	169	12	5	286	21
	1988	80	488	18	64	527	19
	1994	60	297	11	56	294	11
Commercial fishing	1994				26	464	17
	1995				35	335	12
	1996				75	390	14
	1997				27	375	13

The results of the 1985 survey showed relatively low catch rates, but this appears to have been related to sampling strategy. This survey was the continuation of the 1984 survey, but placing emphasis on sampling the more difficult stations as well as sampling with handline in untrawlable grounds (Magnússon and Magnússon, 1987-a).

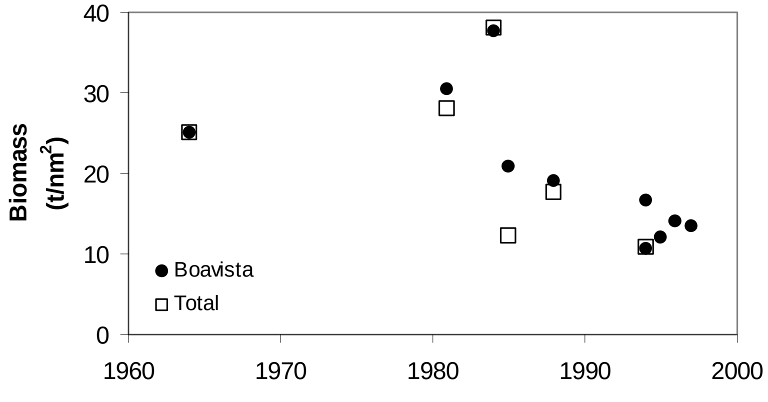

Figure 2.10. Biomass over time in the total survey area and in the Boavista-Maio shelf area (t per nm^2). The 1976 extreme value was not included in the plot (Table 2.5).

Commercial fishing by the "Islândia" showed comparable mean catches with the survey in 1994 (Table 2.5). In fact, the results of commercial fishing appear to indicate that mean catches have stabilised at a lower level, considering the whole period since 1964. On the other hand, one should take care in comparing catch rates of surveys and commercial fishing, as the latter are not random samples. Subsequent chapters will deal with this important point, whether the decreasing trend in catches over time is significant, or not, as well as possible spatial and environmental effects.

The number of species caught varied considerably, showing that species identification was given greater emphasis in more recent surveys. In general, the ten most frequently encountered species constituted about 72 percent of survey catches in weight, while this was about 97 percent in the case of commercial fishing (Appendix B). As expected, the relative species composition in survey and commercial fishing data were different.

It is important to note that biomass estimates from demersal trawl surveys are not considered reliable for rocky grounds, which create difficulties and often lead to the loss or damage of trawl gear. However, various types of rocky grounds are characteristic of Cape Verde and the only ideal ground for trawling is a rather limited area south of Boavista. Several handline surveys were thus undertaken in order to study the demersal small-scale resources or in other words, the resources available to artisanal fisheries (Magnussón and Magnussón, 1987a; Oddsson and Monteiro, 1995, 1996, 1997, 1998, 1999).

The number of species caught ranged between 27 and 35 in each of the four handline surveys and the ten most frequently encountered species constituted between 70 and 90 percent of the catch in weight and numbers (Appendix B). These frequently occurring species are usually caught in low numbers by trawl gear. Catch rates (kg per line per hour) varied according to locality, but there was a clear pattern of lower catch rates in well known fishing grounds and areas closer to major fishing centres. The Beverton and Holt yield per recruit model was applied to seven of the more abundant species in order to give a first preliminary estimate of exploitation rate, which indicated moderately to fully exploited stocks (Oddsson and Monteiro, 1999).

In relation to deep-water resources, the Portuguese vessel "Arquipélago" undertook a survey in 2000 using bottom longline. This survey gave valuable insight on deep-water fish species and a total of ten species were recorded as new occurrences in Cape Verde (Menezes *et al.*, 2001). However, catch rates were relatively low and only a few species appeared to show potential for moderate exploitation. The restricted area of available habitat was considered to be an important determining factor as the shelf areas are limited in size and the slopes are steep (compared to the Azores). Interestingly, this result is in complete agreement with the findings of other surveys, which indicate that fish biomass is concentrated on the shelf and in relatively shallow waters (Stroemme *et al.*, 1982; Magnússon and Magnússon, 1985; Palsson, 1989).

2.4 Concluding Remarks

As seen in the previous sections, tuna are particularly important in the Cape Verde fisheries. All activities concerning tuna such as the compilation of fishery statistics, research coordination and stock assessment as well as developing management advice and procedures for its implementation are undertaken in the context of the International Commission for the Conservation of Atlantic Tunas (ICCAT). Cape Verde is a contracting party of ICCAT (since 1979), which is an inter-governmental fishery organization responsible for the conservation of tunas and tuna-like species in the Atlantic Ocean and its adjacent seas (www.iccat.es).

A vast amount of data and information concerning tuna in the Atlantic are available from ICCAT, but it would be out of context to deal with this in depth. Instead, a short introduction is given on yellowfin and skipjack tuna, focusing on global trends and specifics of tuna fishing in Cape Verde.

Catches of yellowfin and skipjack tuna in the Atlantic increased strongly since the beginning of industrial fishing activities in the 1950s (Figure 2.11). Catches reached a peak in the early 1990s and there appears to be a tendency of a decline in catches over the last decade. These two species were initially not the target species, but have become the dominant species in terms of catch, accounting for about 45 percent of total catches in recent years. This increasing importance followed the development of the industrial purse seine fishery in the 1970s as well as the introduction of fishing on FADs (fish aggregating devices), which became particularly important in the early 1990s (ICCAT, 2004).

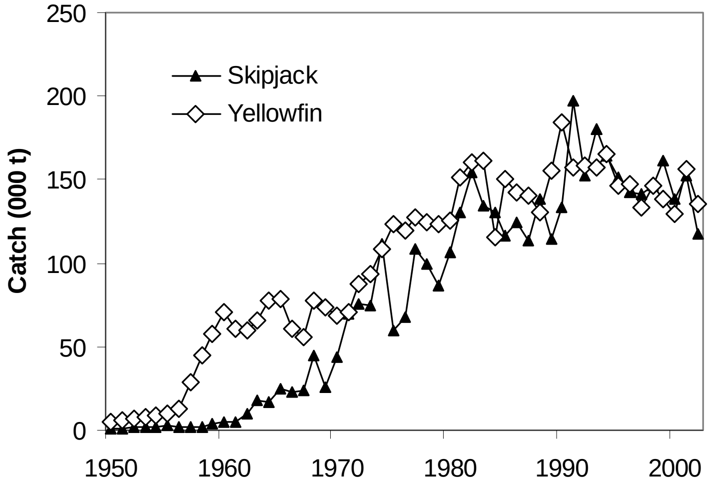

Figure 2.11. Atlantic catches of skipjack and yellowfin tuna (ICCAT, 2004).

A substantial proportion of the catches are taken in the eastern central Atlantic. This is even more evident when considering surface gears such as purse seine and baitboat (pole and line is another term for baitboat) (Figure 2.12). In the case of skipjack tuna, these two surface gears account for most of the catch, while they account for about 80 percent of yellowfin catches. Figure 2.12 shows also that baitboat is a predominant gear in coastal areas only, including the Macaronesian Islands (Azores, Canary Islands, Cape Verde, Madeira). In Cape Verde, there is a peculiarity in that a large proportion of the yellowfin catches are taken by handline gears (artisanal fishery). Note that the Cape Verde Archipelago appears to be outside, or on the outer limits of, the richer fishing grounds off the northwest African continent and the equator (Figure 2.12).

These two species, yellowfin and skipjack tuna, have been the subject of regular assessments by ICCAT. This has involved various stock assessment models or indicators when the data quality is not adequate for modelling purposes. It has been particularly difficult to carry out assessment of skipjack and no management recommendations are given for this species (ICCAT, 2004). Regarding yellowfin, the results vary depending on the model used, but these are more or less consistent in showing a fully exploited or moderately over-exploited stock in the Atlantic Ocean. There is a clear decreasing trend in catch per unit of effort for both species, but a critical level has not been defined for skipjack as yet. A similar trend of decreasing catches and CPUE has been observed for all of the Macaronesian Islands (Bouwsma, 2003-a, 2003-b).

Tuna fishing potential in Cape Verde has been assessed several times over the years (Aubray, 1977; Moal, 1977; Diouf, 1992; Hallier, 1996). The various estimates have ranged from around 25 to 35 thousand tonnes per year for all species combined and including both coastal and offshore waters. In contrast, tuna catches have been decreasing steadily from 5 to 3 thousand tonnes over the last decade.

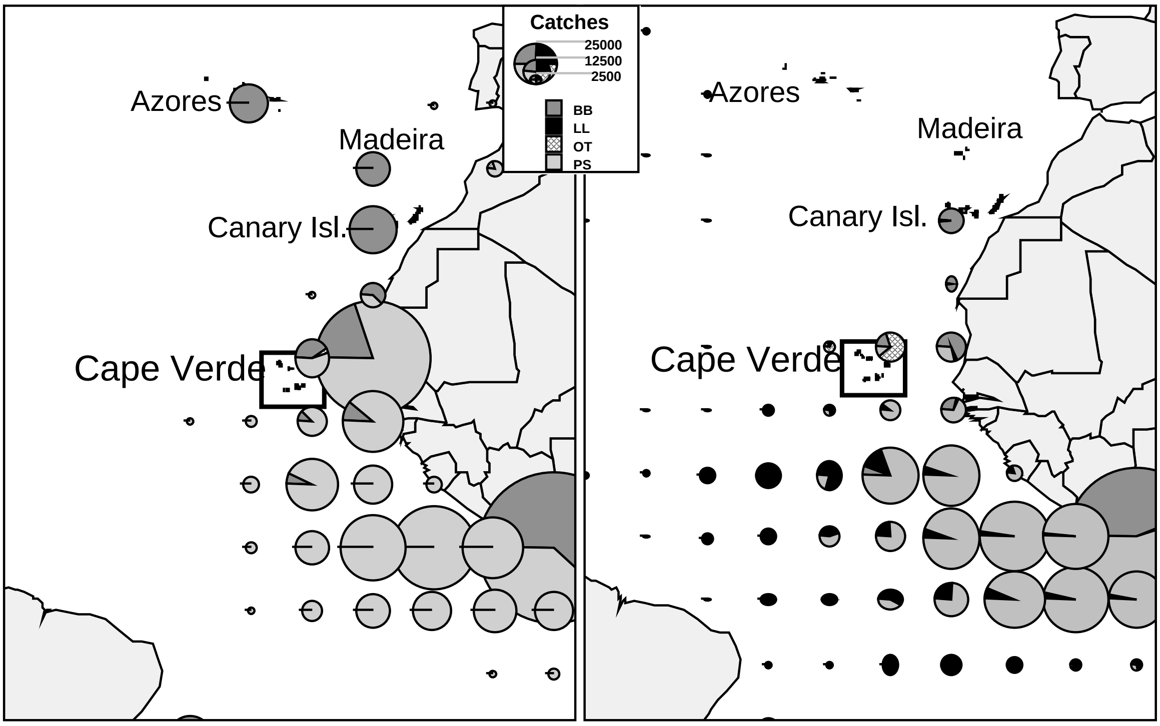

Figure 2.12. Catches of skipjack tuna on the left and yellowfin tuna on the right. Pie chart sizes scaled using square root of catches (sum of 1993 to 2000 catches). (BB: baitboat; LL: longline; OT: other gears; PS: Purse seine)

For the continental shelf, fishing potential was estimated to be a similar 20 to 30 thousand tonnes per year including demersal, small pelagic, and lobster resources (Aubray, 1977; Moal, 1977; Diouf, 1992). These estimates appear to have been overly optimistic for demersal species, in particular, which has become more evident in subsequent assessments based on trawl and acoustic surveys. Again, catches are much lower than these estimates. However, it is interesting to note the reports of high catch rates in the first trawl surveys carried out in Cape Verde (Table 2.5). This suggests that fishery resources appeared to be relatively unexploited or in their virgin state up until the early 1980s (Magnússon and Magnússon, 1985).

The assessments of fishing potential were generally based on production estimates from other areas, such as Senegal, and adjusted to consider the likely lower productivity in the Cape Verde Archipelago. They were overly optimistic and were never revised on the basis of the subsequent trawl and acoustic surveys. Total catches have shown an increasing trend over the last two decades with the introduction of more efficient vessels (industrial fleet) and increasing motorisation of artisanal vessels, but total catches remain low at around 10 thousand tonnes. There are other constraints such as limitations in terms of fishing technology and market conditions, but these factors cannot explain such a modest increase in fisheries production.

Fishery potential was determined by short-term consultants, who had to deliver results in a relatively short time, relying on information produced by previous consultants (Bouwsma, 2003-c). Thus, this led to the general belief or assumption that the fisheries sector had a strong potential for development, which resulted in considerable investments through government and international development funds. However, the fishery goals set out for 2005 in the 1981/1982 food strategy plan will not be met and the development in terms of production over the last two decades can be considered a failure (Bouwsma, 2003-c). Unfortunately, the belief that there is strong fisheries development potential is still widespread in Cape Verdean government and society as well as abroad, making it even more difficult to take the necessary management measures.

INDP is aware of this situation and efforts are being made to communicate a more realistic assessment of the fisheries sector. There is a considerable amount work done in stock assessment studies of economically important species such as lobsters and mackerel scad (*Decapterus macarellus*) (Almada, 1997; Carvalho, *et al.* 1999). However, there is an urgent need to update these assessments as well as the application of more recent methodology and techniques, including the ecosystem approach to fisheries (EAF).

Chapter 3. Model of the Coastal Ecosystem: first attempts with Ecopath[5]

3.1 Introduction

Efforts to improve the current state of affairs in fisheries include the introduction of the ecosystem approach to assess the direct and indirect effects of fishing (Gislason *et al.*, 2000; FAO, 2002-b; Cury and Christensen, 2004). One possible tool for such an approach is Ecopath, using the "Ecopath with Ecosim" modelling software, which was developed using the mass-balance approach (Polovina, 1984; Pauly *et al.*, 2000; Christensen and Walters, 2004). This modelling approach is being used widely as a tool for the analysis of exploited aquatic ecosystems, including numerous studies from both temperate and tropical areas (Christensen and Pauly, 1993, 2004; Pauly *et al.*, 2000).

Ecopath models are relatively straightforward to construct and require limited information as opposed to more data-driven approaches such as MSVPA, making it more feasible to apply in tropical scenarios such as in Cape Verde. Furthermore in connection with Ecopath, a rich theoretical framework exists for the analysis of energy flows or cycling in ecosystems and it is straightforward to compare these flows between different time periods in the same system, or among similarly structured systems (Christensen *et al.*, 2001). Ecopath was initially a deterministic steady-state approach model and it has since developed making it possible to (i) address uncertainty around impact variables for balancing the model and deriving system-level metrics, and (ii) to simulate changes in fishing pattern and intensity through time and space in an ecosystem framework (Walters *et al.*, 1997; Christensen *et al.*, 2001).

This study concerns the coastal ecosystem in Cape Verde, constituted by the narrow and irregular continental shelves around the ten islands of the archipelago. About 70 percent of the demersal fish biomass is found on the more extensive continental shelf system around the eastern islands of Boavista and Maio (Stroemme *et al.*, 1982; Magnússon and Magnússon, 1985). Fishery catches are dominated by pelagic and/or migratory species such as tuna and various small pelagic species, accounting for approximately 80 percent of the catches. Annual catches increased from about 7 000 to 10 000 tonnes over the period 1986 to 2000 (see Chapter 2 for more details), largely due to increasing industrial catches. During this period, there has been a trend of decreasing catches per unit of effort in the industrial fishery and lower biomass estimates for demersal fish stocks (Monteiro, 1999-b).

The period from 1981 to 1985 was chosen for the purpose of creating an Ecopath model, so as to correspond to the initial phase of a more intensive fishery development. This period was

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[5] This chapter is a slightly modified version of the original paper, Stobberup *et al.*, 2002.

characterised by a predominantly artisanal fisheries and low level of motorisation. Demersal fish stocks were considered to be under light exploitation at the time (Magnússon and Magnússon, 1985, 1987-a). It was also during this period that a statistical collection system was established (Shimura, 1980) and the first systematic surveys were undertaken, providing estimates of fisheries catches and biomass for different fish stocks.

By creating an ecosystem model, our aim is to gain a better understanding of system dynamics and consider the effects of fishing in the Cape Verde coastal ecosystem. A previous attempt took the first steps in this direction (Stobberup and Coelho, 2000), but a number of weak points were identified and are dealt with in the present study. These improvements included a reformulation of the ecological groups, making the model much more specific for Cape Verde, and an in-depth bibliographical search for the corresponding diet information. Another important improvement is the handling of migration in the model, considering that Cape Verde is known to be in the migration route of important species such as tuna. The present study may therefore serve as the starting point for a simulation of ecosystem dynamics from the period 1981 to the present.

3.1 Materials and Methods

The Modelling Approach

The Ecopath approach was originally proposed by Polovina (1984) and has since been developed extensively (Christensen and Pauly, 1992; Walters *et al.*, 1997, 1999, 2000). Most importantly, Ecopath no longer assumes steady state, but instead bases parameterisation on an assumption of mass-balance over an arbitrary period, usually one year. Two master equations are defined for parameterisation, one to describe the production term and one for the energy balance of each group defined in the model. The first equation describes how the production term for each group can be split in components, which can be expressed simply as:

1) Production = catches + predation mortality + biomass accumulation + net migration + other mortality

A set of linear equations, one for each group in the model, are solved simultaneously, ensuring balance between energy input and output for all the groups, which can be expressed as (second equation):

2) Consumption = production + respiration + unassimilated food

Ecopath models are relatively straightforward to construct in concept, but the associated software "Ecopath with Ecosim" has become increasingly complex as part of its development. Chirstensen *et al.* (2001) should be consulted for more detailed information of the modelling approach as well as advice on constructing models.

The Cape Verde Model

The present Ecopath model concerns the continental shelf system for the period 1981 to 1985. We assume a homogeneous area of 5 394 km^2, which is the estimated area of the continental shelves around the islands (Bravo de Laguna, 1985). The oceanic waters between islands were not included, as they are considered to be a different ecosystem. However, we assume interaction between the coastal and oceanic ecosystems, particularly through the foraging of oceanic and/or migratory species outside the coastal ecosystem.

Fisheries statistics are available for the period from 1981 to 2000 in Cape Verde (INDP, 2001), but the time series spanning from 1981 to 1985 is not considered reliable. Chapter 2 dealt with the detected errors and the procedure used in the revision of the data. Thus, the present study was based on the revised catch estimates for the period 1981 to 1985 (Table 3.1).

Ecological Groups

31 ecological groups were defined, including small mammals, seabirds, turtles, 19 fish groups, 5 benthic invertebrate groups, zooplankton, 2 primary producers and detritus (Table 3.2). Fish biomass estimates were obtained from survey reports for the more abundant demersal and neritic pelagic fish species as well as basic biological information such as sizes and length/weight relationships (Stroemme *et al.*, 1982; Magnússon and Magnússon, 1985, 1987-a, 1987-b).

Concerning the fish species included in the model, a total of 99 species were specified based on criteria such as their relative importance in trawl, handline, and longline surveys as well as in the fisheries (Palsson, 1989; Oddson, 1995-1999; Menezes, 2001; INDP, 2001). The aggregation into ecological groups was based on characteristics such as diet, habitat, common and maximum length, estimated trophic level and consumption. Table 3.3 indicates which are the dominating species for higher trophic level groups. For a more detailed description, Appendix C and D give a complete listing of fish constituent species and some key parameter estimates as well as the corresponding diet references. Local information on diet composition, in quantitative terms, was available only for *Decapterus macarellus* (Almada, 1997).

Table 3.1. Estimated fisheries catch (t/km^2) by ecological group and fishing fleet. Fishing fleets followed the definition of the fisheries statistics (Artisanal Lines - handlines; Artisanal Nets – seines, gillnets; Industrial – pole and line, seines). Shading indicates that the value was increased to accommodate for unidentified species catches. No discards are assumed for this period from 1981 to 1985.

	Model definition of Fishing Fleets			
Group Name	**Art Lines**	**Art Nets**	**Industrial**	**Total**
Pelagic sharks	0.000	0	0.001	0.001
Mammals	0	0	0.001	0.001
Billfish	0.001	0	0.001	0.002
Pelagic predators	0.195	0	0.002	0.197
Moray eels	0.008	0	0.003	0.011
Sea birds	0.001	0	0.000	0.001
Bathydemersal	0.010	0	0.001	0.011
Demersal sharks	0.001	0	0.001	0.002
Demersal predators	0.067	0	0.020	0.087
Large Tuna	0.396	0.002	0.434	0.832
Rays	0.000	0	0.000	0
Demersal fish	0.011	0.006	0.010	0.027
Reef feeders	0.005	0	0.001	0.006
Small tuna	0.002	0.005	0.01	0.017
Jacks	0.022	0.008	0.004	0.034
Flatfish	0.001	0	0	0.001
Other Demersal fish	0.005	0	0.001	0.006
Small pelagics	0.053	0.308	0.064	0.425
Flyingfish	0.005	0	0.001	0.006
Sparids	0.012	0.001	0.004	0.017
Turtles	0.004	0	0.001	0.005
Crustaceans	0.001	0	0.006	0.007
Molluscs/Worms	0.000	0.005	0	0.005
Herbivores	0.005	0	0.002	0.007
Heterotrophic benthos	0.005	0	0	0.005
Sum	0.810	0.335	0.568	1.713

Table 3.2. Basic input and estimated parameters of the Ecopath model for Cape Verde. Biomass (B), Production/Biomass (P/B), and Consumption/Biomass (Q/B) estimates are given (see text and footnotes for explanation). Trophic level (TL) and shaded values were estimated by the model.

Group name (unit)	**TL**	**B (t/km^2)**	**P/B (per year)**	**Q/B (per year)**	**EE**	**P/Q (per year)**
Pelagic sharks	4.6	0.045	0.100[A]	2.668	0.50	0.037
Mammals	4.3	0.034	0.100[B]	11.790[B]	0.50	0.008
Billfish	4.2	0.083	0.300[A]	6.400	0.80	0.047
Pelagic predators	4	0.335	0.997	9.971	0.80	0.100
Moray eels	4	0.160	0.280	2.800	0.90	0.100
Sea birds	3.8	0.020	0.250[B]	109.500[C]	0.50	0.002
Bathydemersal	3.8	0.255	0.384	3.844	0.90	0.100
Demersal sharks	3.8	0.141	0.260	2.600	0.90	0.100
Demersal predators	3.7	0.216	0.719	3.806	0.93	0.189
Large Tuna	3.6	2.225[D]	0.900[E]	11.569	0.82	0.078
Rays	3.5	0.019	0.423	4.233	0.90	0.100
Demersal fish	3.5	1.456	0.562	5.759	0.93	0.098
Reef feeders	3.4	0.396	0.584	5.844	0.90	0.100
Small tuna	3.3	0.712	0.987	9.872	0.90	0.100
Jacks	3.3	0.659	0.655	6.550	0.90	0.100
Flatfish	3.3	0.002	0.718	7.175	0.90	0.100
Other Demersal fish	3.1	1.893	0.727	5.142	0.89	0.141
Small pelagics	3	12.050	1.137	11.366	0.95	0.100
Flyingfish	3	0.646	1.430	14.300	0.90	0.100
Sparids	2.8	1.841	0.683	16.247	0.87	0.042
Turtles	2.8	0.126	0.150[F]	3.500[F]	0.50	0.043
Crustaceans	2.8	13.048	1.600[G]	10.000[G]	0.95	0.160
Echinoderms	2.4	15.042	1.200[G]	4.000[G]	0.95	0.300
Molluscs/Worms	2.3	25.842	2.500[G]	7.000[G]	0.95	0.357
Herbivores	2	0.968	0.587	15.863	0.81	0.037
Heterotrophic benthos	2	12.843	3.000[H]	12.500[H]	0.95	0.240
Microfauna	2	0.439	100.000[G]	215.000[G]	0.95	0.465
Zooplankton	2	2.848	63.440[I]	280.000[G]	0.95	0.227
Phytoplankton	1	2.700[J]	285.000[J]	-	0.96	-
Benthic autotrophs	1	20.464	13.250[G]	-	0.50	-
Detritus	1	300.800[K]	-	-	0.79	-

Footnotes:
A: Kitchell *et al.* 1999
B: assumed value based on model for Canada (Bundy *et al.* 2000)
C: based on regression by Nagy (1987)
D: based on Hallier 1999 (see text for further explanation)
E: based on ICCAT reports (ICCAT, 2001-a, 2001-b, 2003, 2004) assuming P/B = Z
F: Polovina 1984
G: Opitz 1993
H: Silvestre *et al.* 1993
I: based on Braun *et al.* 1982 for the Canary Islands
J: based on information given Longhurst (1995, 1998) (see text for further explanation)
K: based on empirical relationship given in Pauly *et al.* 1993

Table 3.3. Dominating species by "higher" group and the estimated trophic levels (TL). For a more detailed description, Appendix C gives a complete listing of fish constituent species, including parameter estimates for L_∞ , W_∞ , and Q/B as well as the diet references used.

Group name	**TL**	**Dominating species**
Pelagic sharks	4.6	Carcharhinus sp.; Sphyrna sp.; Galeocerdo cuvier
Mammals	4.3	*Tursiops sp.; Stenella sp.; Delphinus sp.*
Billfish	4.2	*Xiphias gladius; Istiophoridae*
Pelagic predators	4	*Acanthocybium solandri*
Moray eels	4	*Gymnothorax vicinus; Muraena helena*
Sea birds	3.8	*(Not well known!)*
Bathydemersal	3.8	*Beryx spp.; Scorpaena spp.*
Demersal sharks	3.8	*Mustelus mustelus; Rhizoprionodon acutus*
Demersal predators	3.7	*Serranidae*
Large Tuna	3.6	*Thunnus albacares; Katsuwonus pelamis*
Rays	3.5	*Raja sp.; Dasyatis sp.; Rhinobatos sp.*
Demersal fish	3.5	*Pomadasys sp.; Priacanthus sp.; Pseudupeneus sp.*
Reef feeders	3.4	*Myripristis sp.; Sargocentron; Bodianus sp.*
Small tuna	3.3	*Euthynnus alletteratus*
Jacks	3.3	*Caranx sp.; Seriola sp.; Selene sp.*
Flatfish	3.3	*Bothus sp.; Dicloglossa sp.*
Other Demersal fish	3.1	*Antigonia capros; Dactylopterus volitans*
Small pelagics	3	*Decapterus macarellus; Selar crumenophthalmus*
Flyingfish	3	*Fodiator acutus*
Sparids	2.8	*Lithognathus mormyrus; Diplodus spp.*
Turtles	2.8	*Caretta caretta*
Herbivores	2	*Acanthurus sp; Scarus sp.; Sparisoma sp.*

For non-fish groups, local information is very limited and mostly of a qualitative nature. In order to tackle this problem, we assume that the ecosystem characteristics in Cape Verde are similar to other tropical areas in terms of trophic functioning and community structure. From Opitz (1993), we adopted the definition of benthic invertebrate functional groups such as crustaceans, echinoderms, mollusc/worms, heterotrophic benthos, and microfauna. This was also the source of parameter estimates for benthic invertebrates as well as their corresponding diets (Tables 3.2 & Appendix E).

Numerous bird studies have been undertaken in the Cape Verde Islands, but these have been generally descriptive (*e.g.* Le-Grand 1986), making it difficult to assess in quantitative terms. We therefore assume a similar production as given in Bundy *et al.* (2000) and use an empirical equation to estimate consumption (Nagy, 1987).

In the case of marine mammals, the study by Reiner *et al.* (1996) gives a good indication of dominating dolphin species. However, in the absence of quantitative information, we assume similar production and consumption estimates to those given in Bundy *et al.* (2000). Pauly *et al.* (1998b) was a valuable source of information on diet for the group, which was calculated as the simple average diet composition for the four dominant dolphin species (*Stenella frontalis, Stenella longirostris, Tursiops truncates*, and *Delphinus delphis*).

For other non-fish groups, Table 3.2 indicates the sources of parameter estimates.

Migration

Considering that there is strong oceanic influence on the Cape Verde coastal ecosystem, where migratory species such as tuna constitute a major resource, this could not be ignored. We followed the recommendation by Christensen *et al.* (2001), treating the migratory groups as part of the system, but obtaining part of their food outside the system. This was specified as an "import" of food in the diet matrix (Appendix E), which was estimated by assuming the proportion of oceanic species in the diet as import.

Parameter Estimation

Local information is generally lacking for even basic growth parameters, not to mention production and consumption estimates. However, Magnússon and Magnússon (1987-b) was a valuable source of information on length-weight relationships for the more abundant fish species. Fishbase 2000 (Froese and Pauly, 2000) was used as the source of basic information for fish species such as L_∞, W_∞, and diet, including the empirical relations given therein (Keyfacts), when these were not available from local studies (Appendix C & D).

Fish Consumption/Biomass Ratios (Q/B) were estimated using the empirical equation of Palomares and Pauly (1999):

3) $\log Q/B = 7.964 - 0.204 \log W_\infty - 1.965T' + 0.083A + 0.532h + 0.398d$

where W_∞ (or asymptotic weight) is the mean weight that an individual would reach if it were to grow indefinitely, T′ is the mean environmental temperature expressed as 1000 / (°C + 273.15), A is the aspect ratio of the caudal fin indicative of metabolic activity and expressed as the ratio of the square of the height of the caudal fin and its surface area, 'h' and 'd' are dummy variables indicating herbivores (h=1, d=0), detritivores (h=0, d=1) and carnivores (h=0, d=0).

In the case of demersal fish, which are relatively well sampled by trawl gears, the Production/Biomass Ratio (P/B) was estimated by assuming P/B = Z and Z = M + F, which apply under steady-state conditions (Allen, 1971) assumed for the construction of this model. Estimates of the von Bertalanffy growth parameter were (K) often lacking making it difficult to apply the Pauly's well-known empirical equation (Pauly, 1980). Instead, we used the empirical equation proposed by Froese and Binohlan (2000) to estimate natural mortality (M):

4) $M = 10^{(0.566 - 0.718 * \log(L_\infty) + 0.02 * T)}$

F was estimated from F = C/B, also under a steady-state assumption, when fish biomass estimates were available.

An average temperature of 25ºC for pelagic, 20ºC for demersal, and 15ºC for bathydemersal species was used in the estimation of M and Q/B, based on CTD data from the acoustic survey by R/V *Capricórnio* in 1997 (Marques *et al.*, 1997).

Considering the relative dominance of specific fish species in each group, the parameters Q/B and M as well as diet composition were estimated as weighted averages based on biomass estimates for the demersal fish groups, in particular. In the case of migratory/pelagic species, catch was used as a relative indicator of species abundance in each group and weighted accordingly. For other groups, a simple average was calculated, as the relative abundance of the constituent species is poorly known (Table 3.4).

Table 3.4. Type of weighting procedure adopted for parameter estimates Q/B (Consumption/Biomass) and M (Natural Mortality) as well as diet composition. "None" indicates that the estimate was based on a simple average by ecological group.

Groups	Q/B & M	Diet
Billfish	none	none
Large tuna	by catch	by catch
Pelagic predators	by catch	by catch
Small tuna	by catch	by catch
Jacks	by catch	by catch
Moray eels	by catch	none
Pelagic sharks	none	none
Bathydemersal	none	none
Demersal sharks	none	by biomass
Demersal predators	by catch	by biomass
Rays	none	none
Other Demersal fish	by biomass	by biomass
Demersal fish	by biomass	by biomass
Flatfish	none	none
Small pelagics	by catch	by catch
Reef feeders	by catch	none
Sparids	by biomass	by biomass
Flyingfish	none	none
Herbivores	by biomass	by biomass

As can be seen in Table 3.2, we did not attempt to estimate P/B and B except for a few demersal fish groups that could be considered relatively well sampled. Instead, we allow the model to estimate these values, taking into account predation and fishing mortality as well as other mortality. The procedure used was to assume a production/consumption (P/Q) ratio of 0.1 and fix an ecotrophic efficiency (EE), varying from 0.5 for top predators (and benthic algae), 0.8 for highly migratory groups, and 0.9 for other groups, which were considered reasonable values in light of other studies (*e.g.* Polovina, 1984; Christensen and Pauly, 1993; Kitchell *et al.*, 1999).

3.3 Results and Discussion

A main objective was to create a model, covering the initial stages of a more intensive fisheries development (1981-1985), which can serve as the starting point for simulation studies of ecosystem dynamics. Considering the location of the Cape Verde Archipelago, it is evident that the coastal areas have a strong oceanic influence. The presence of many oceanic species is an example of this (Reiner, 1996) as well as the fact that Cape Verde is in the migratory route of several tuna species. Migratory species such yellowfin and skipjack tuna (called "large tuna" in the model) constitute a major resource for the fisheries in Cape Verde, making it extremely important to deal with this issue in the model. We followed the recommendation by Christensen *et al.* (2001), treating the migratory groups as part of the system, but obtaining part of their food outside the system (and/or the time spent outside the system). The other option given is to estimate dispersal rates (immigration/emigration), which has not been studied in Cape Verde. Another problem in connection with the incorporation of dispersal rates in the model is that Ecopath will deal with flows as "density independent," a constant proportion of immigration/emigration irrespective of the biomass, which is not realistic (Christensen *et al.*, 2001). Thus, we delimit the model to the coastal ecosystem, considering the distinction between oceanic and insular ecosystems, but incorporating interaction through the import term specified in the diet matrix for migratory and/or oceanic species.

Table 3.2 presents basic input parameters of the model as well as the values estimated by the model. As can be seen, biomass estimates were available only for demersal fish groups[6], small pelagics, and large tuna, the dominant groups in terms of biomass, catches, and trophic interactions. This is important as it serves to limit the biomass estimates in general, based on the underlying diet matrix and the fisheries catches.

It would have been desirable to split the large tuna group into yellowfin and skipjack tuna, as these are important for the local fisheries. However, available data on potential catches and annual production are presented only as global estimates for tuna (Diouf, 1992; Hallier, 1999). A tuna biomass estimate of 2.225 tonnes per km^2 was used in the model, considering a coastal production estimate of 1 200 tonnes per year per 1° square (Hallier, 1999) and a fishing mortality of 0.37 (ICCAT, 2001-a, 2001-b, 2003, 2004). There is considerable uncertainty attached to this tuna biomass estimate, but it is crucial to put limits on this parameter, as tuna are very dominant in the system.

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[6] The dominant groups are Demersal fish and Other demersal fish, consisting of commercial and non-commercial species, respectively (see Table 3.3 for species).

Another dominant group is the small pelagics group composed of mackerel scad (*Decapterus macarellus*), primarily. The available biomass estimate of 65 000 tonnes (12.050 t/km²) in the coastal waters of Cape Verde, which is rather high, does not distinguish quantitatively between the different species of small pelagics (Stroemme *et al.*, 1982). Mackerel scad should form its own distinct group in the model, considering its diet, its importance in the pelagic food web, and its economical importance, but this was not possible with the available information at hand.

For the benthic invertebrate groups, Q/B and P/B parameter estimates were adapted from Opitz (1993) and the resulting biomass estimates should be considered as what is necessary to sustain the system without putting much emphasis on specific groups. A comparison of total biomass for these groups (67.2 t/km²) is similar to the aggregate group "heterotrophic benthos" (77 t/km²) in Polovina (1984), but much lower than the 1 200 t/km² (sessile animals such as corals not included) reported in Opitz (1993). In temperate islands such as the Faeroe Islands (Zeller and Freire, 2001), the Azores (Guénette and Morato, 2001), and Iceland (Mendy and Buchary, 2001), total biomass estimate for benthos groups varies from approximately 10 to 40 t/km². As there is a considerable source of variation on biomass estimates and diet composition of fish that prey on benthos groups, the results seem reasonable for a tropical setting. However, there appears to be an error in Opitz (1993), possibly in relation to the unit used.

Diet compositions expressed as prey/predator proportions are presented in Appendix E. For fish groups, the diet composition was estimated by considering information by species available in the literature, preferably for similarly tropical areas (Appendix C & D). Data obtained from the literature were standardised into major groups (*e.g.* fish, zooplankton, phytoplankton, polychaeta, etc.) and resulted in a base matrix with major food items. This could be integrated directly for the lower trophic levels, as the model definition at this level maintains a similar separation into major groups. For the other components in the model, educated guesses of the proportion of a specific fish group preyed upon by the other fish groups had to be applied, as it was not possible to provide better estimates in the absence of local studies on food composition.

One way of determining whether the specified diet composition is reasonable is to analyse the resulting mortalities by predation and fishing. Table 3.5 gives the resulting mortalities, given the specified diet composition. These predation mortalities appear reasonable, but it must be pointed out that this is the end result after a number of modifications to the diet matrix. As mentioned before, the proportion of fish in the diet of several groups had to be split up among the various fish groups defined in the model. Table 3.6 gives total mortality by predation and fishing and one aspect that stands out very clearly is the relatively light effect of fishing on the system as seen in the low values of fishing mortality. The only exceptions are the fishing mortality values for groups such as pelagic predators and demersal predators, as well as tuna to some extent. This is especially evident in the case of small pelagics where fishing pressure is so low that it can almost be disregarded.

A revision of the fisheries statistics was undertaken for the period 1981 to 1985, specified for the model. This resulted in lower total catches for the initial years, but this adjustment had little impact on the estimation of fishing mortalities. It is also evident that an extrapolation error occurred in the processing of fisheries statistics in these years and further revision may yield even lower catch estimates.

In the interest of placing a limit on the primary productivity in the waters of the Cape Verde Archipelago, an attempt was made at estimating production and biomass. Various references indicate that primary production ranges from 150 to 500 mgC/m^2/day (Carvalho, 1994), which is in good agreement with the estimate of 350 mgC/m^2/day for the Canary Islands (Braun *et al.*, 1982). However, satellite images indicate higher values, an estimated average of 939.7 mgC/m^2/day (Villy Christensen, *pers. comm.*). The estimate based on satellite data was considered more reliable and in agreement with the fact that Cape Verde is situated in the transition between the Eastern Canary Coastal Province and North Atlantic Tropical Gyral Province with production estimates of approximately 2 000 mgC/m^2/day and 290 mgC/m^2/day, respectively (Longhurst, 1995). In relation to biomass, the profiles given by province in Longhurst (1998) indicate that biomass may range from 2700 to 240 mgC/m^2 for the two provinces, respectively. Based on the IPIMAR survey in July 1997 (Marques *et al.,* 1997), an integration of chlorophyll *a* profile concentrations yielded a biomass estimate of 859 mgC/m^2, which is lower than 1200 mgC/m^2 estimated for the Canary Islands (Braun *et al.* 1982) and much lower than for other islands (Polovina, 1984; Opitz, 1993; Guénette and Morato, 2001; Mendy and Buachary, 2001; Zeller and Freire, 2001). Thus, we decided to use the upper limit of 2 700 mgC/m^2 given by Longhurst as the phytoplankton biomass estimate, resulting in a P/B ratio of 285 per year.

Mixed trophic impact analysis, as defined by Christensen *et al.* (2001), was undertaken in order to determine the direct and indirect effects of a change in biomass of one group on the other groups of the system (Figure 3.1). The resulting pattern reinforces many of the points already discussed.

Table 3.5. The resulting predation mortalities for the higher trophic levels of the model, given the specified diet composition. Predators are arranged along the horizontal axis and prey items down the vertical axis.

Group		**Predator group**																		
no.	**Prey**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**
1	Sea birds	-	-	-	-	-	-	-	-	0.076	-	-	-	-	-	-	-	-	-	-
2	Mammals	-	-	-	-	-	-	-	-	0.021	-	-	-	-	-	-	-	-	-	-
3	Billfish	-	-	-	-	-	-	-	-	0.216	-	-	-	-	-	-	-	-	-	-
4	Large Tuna	0.006	0.018	0.036	0.046	0.255	-	-	-	0.003	-	-	-	-	-	-	-	-	-	-
5	Pelagic predators	0.020	0.085	0.100	-	-	-	-	-	0.004	-	-	-	-	-	-	-	-	-	-
6	Small tuna	0.095	0.035	0.075	0.470	0.188	-	-	-	0.002	-	-	-	-	-	-	-	-	-	-
7	Jacks	0.034	0.037	-	0.196	0.147	0.107	-	-	0.011	-	-	0.006	-	-	-	-	-	-	-
8	Moray eels	-	-	-	-	-	-	-	-	-	-	0.183	-	-	-	-	-	-	-	-
9	Pelagic sharks	-	-	-	-	-	-	-	-	0.028	-	-	-	-	-	-	-	-	-	-
10	Bathydemersal	-	-	-	-	-	-	-	0.176	-	-	0.043	0.084	-	-	-	-	-	-	-
11	Demersal sharks	-	-	-	-	-	-	-	-	0.220	-	-	-	-	-	-	-	-	-	-
12	Demersal predators	0.052	0.019	-	-	-	-	-	0.088	0.022	-	0.085	-	-	-	-	-	-	-	-
13	Rays	-	-	-	-	-	-	-	-	-	-	0.381	-	-	-	-	-	-	-	-
14	Other Demersal fish	-	-	-	0.136	0.071	-	-	0.031	-	0.111	0.010	0.061	0.003	-	0.222	0.000	-	-	-
15	Demersal fish	0.015	0.017	-	0.177	0.046	-	-	0.041	0.002	0.094	0.013	0.096	0.003	-	-	0.000	-	-	-
16	Flatfish	-	-	-	-	-	-	-	-	-	-	0.163	-	0.036	-	-	-	-	-	-
17	Small pelagics	0.070	0.009	0.011	0.662	0.100	0.103	0.073	-	0.000	-	-	-	-	-	0.021	-	-	-	-
18	Reef feeders	-	-	-	-	-	-	0.109	0.057	-	0.030	0.009	0.104	-	-	0.127	-	-	-	0.076
19	Sparids	0.079	0.000	-	0.224	0.056	-	0.047	0.017	-	-	0.002	0.078	-	-	0.032	-	-	-	0.049

Table 3.6. Estimated total mortality for the higher trophic levels of the model. Total mortality (Z) is decomposed into Fishing Moratlity (F), Predation mortality (M2), and Other Mortality (M1).

Groups	P/B = Z	F	M2	M1
Sea birds	0.250	0.049	0.076	0.125
Mammals	0.100	0.029	0.021	0.050
Billfish	0.300	0.024	0.216	0.060
Large Tuna	0.900	0.374	0.364	0.162
Pelagic predators	0.997	0.589	0.209	0.199
Small tuna	0.987	0.024	0.865	0.099
Jacks	0.655	0.052	0.538	0.066
Moray eels	0.280	0.069	0.183	0.028
Pelagic sharks	0.100	0.022	0.028	0.050
Bathydemersal	0.384	0.043	0.303	0.038
Demersal sharks	0.260	0.014	0.220	0.026
Demersal predators	0.719	0.403	0.265	0.051
Rays	0.423	0.000	0.381	0.042
Other Demersal fish	0.727	0.003	0.644	0.080
Demersal fish	0.562	0.019	0.503	0.040
Flatfish	0.718	0.446	0.200	0.072
Small pelagics	1.137	0.035	1.048	0.054
Reef feeders	0.584	0.015	0.511	0.058
Sparids	0.683	0.009	0.584	0.090
Flyingfish	1.430	0.009	1.278	0.143
Herbivores	0.587	0.007	0.470	0.109
Turtles	0.150	0.040	0.035	0.075
Crustaceans	1.600	0.001	1.519	0.080

Industrial and artisanal net fishing generally had a very small effect on the different groups, which is in agreement with the artisanal nature (mostly handlines) of the fisheries in the early 1980's (Figure 3.1). Artisanal handline fishing had negative effects on target species such as pelagic and demersal predators. However, the artisanal handline fishing effects on seabirds and flatfish are uncertain as these catches were assumed (by-catch and unreported catches). The positive effect of small pelagics on the top pelagic predators as well as in the industrial and artisanal net fisheries further reinforces the importance of mackerel scad (*D. macarellus*) in the pelagic food web and the fisheries. Large tuna generally had a negative effect on its prey, illustrating its importance as the dominant predator, and a positive effect in fishing. However, the negative effect of large tuna on artisanal net fishing is the result of tuna preying on small pelagics that are targeted by the net fishery. Pelagic sharks had a negative effect on other large predators, including demersal sharks. In contrast, the positive effects of pelagic sharks on moray eels and rays is the result of pelagic sharks feeding on their predator, demersal sharks.

Groups such as seabirds, small mammals, and rays had little effect on the system, very much a result of their small biomasses. The small effects of jacks and small tuna, which are important groups in the system, appear to be related to their feeding at lower trophic levels.

The effects of groups at lower trophic levels were straightforward (not shown). Phytoplankton had positive effects on zooplankton, which in turn resulted in positive effects on small pelagics and all its predators. The benthic invertebrate groups had a positive effect on their predators as expected.

Cape Verde can be considered similar to other tropical areas in terms of trophic functioning and community structure. Demersal and reef fish fauna in Cape Verde were found to be similar to other tropical areas, but there appear to be some essential differences in that corals and seaweed are of minor importance and that benthic primary productivity depends mainly on calcareous algae (van der Land, 1993). The similarities observed in terms of dominant species in Cape Verde, Jamaica (Munro, 1983), and the Virgin Islands (Opitz, 1993), at the genus or even species level, supports the use of parameters estimated for other tropical areas when local information is lacking. However, we must stress the point that the resulting model is specific for Cape Verde, for higher trophic levels in particular, which were based on survey information and the national fisheries statistics collection system.

The biomass estimates of the model, as well as the biomass estimates based on survey results, indicate that previous assessments of potential harvest (Aubray, 1977; Moal, 1977; Diouf 1992), which range from 25 000 to 56 000 tonnes, appear to be overly optimistic, especially for demersal species. Total harvest shows an increasing trend over the period from 1981 to 2001 with the introduction of more efficient vessels (industrial fleet) and increasing motorisation of artisanal vessels, but total harvest remains around 10 000 tonnes. These assessments were based on production estimates from other areas, such as Senegal, and adjusted to consider the likely lower productivity in the Cape Verde Archipelago.

A simple calculation based on the model results would be a good way of assessing previous estimates of potential harvest. Assuming the model estimate of 124 000 tonnes standing fish stock biomass, a harvest of 25 000 to 56 000 tonnes is equivalent to approximately 20 to 50% of the standing stock biomass. But this simple calculation does not take into account the biomass of species that are of no commercial value. Although an upper limit of 56 000 tonnes of potential harvest is too high, a doubling of harvest appears to be feasible. However, the ecosystem effects of such an increase in harvest has to be studied, considering the effects on each of the ecosystem components, including direct and indirect effects as well as the effects on the more vulnerable groups.

An important next step is to simulate the effects of increasing fishing pressure in the Cape Verde coastal ecosystem. Time series data on catches and fishing effort are essential for such a study covering the period 1981 to 2000 and may help to determine whether the model components have been correctly specified or if adjustments are necessary, in other words a calibration process. Thus, we hope to contribute to the process of introducing the ecosystem approach to fisheries assessment in Cape Verde, which can lead to useful indicators for management purposes.

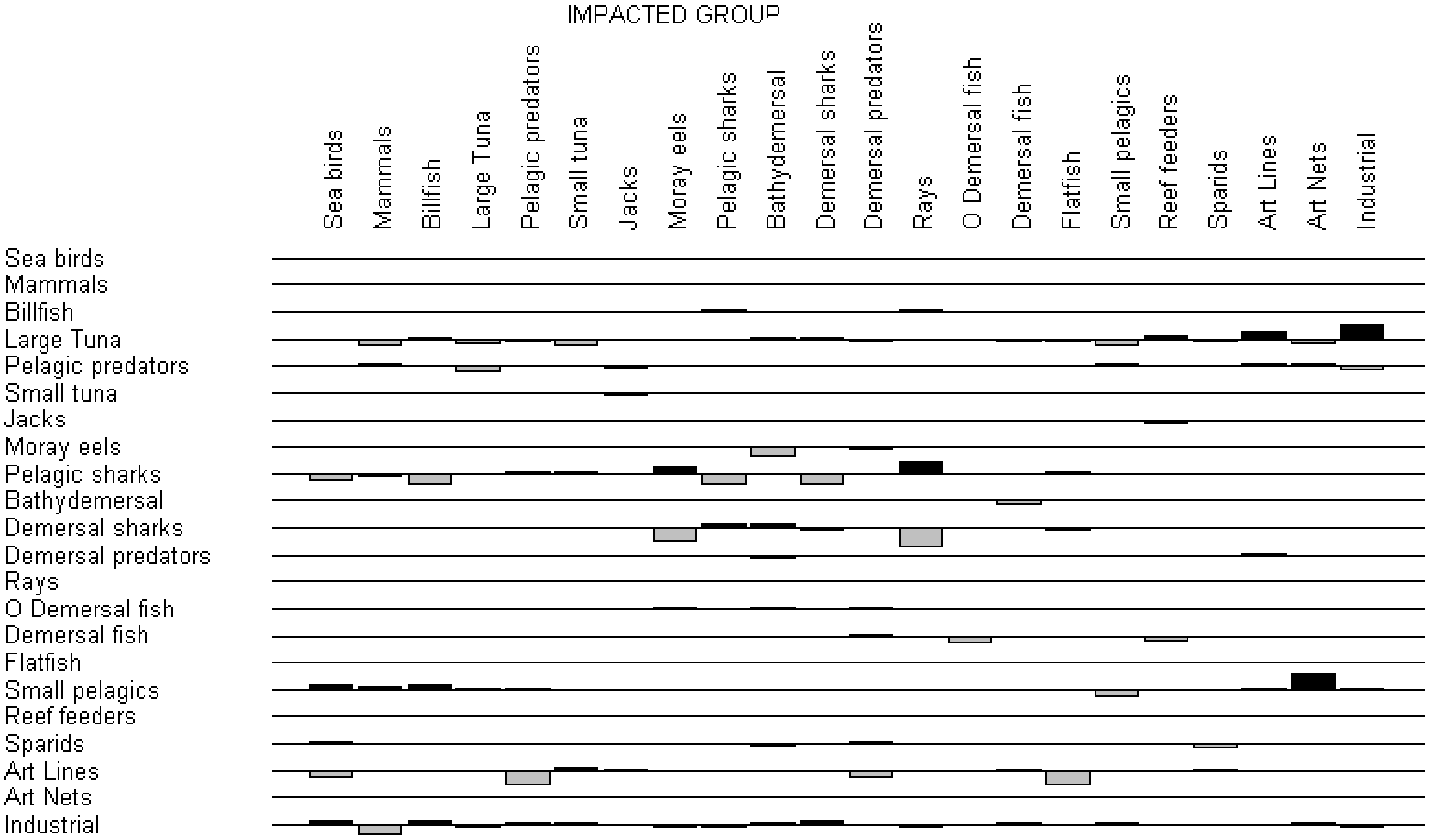

Figure 3.1. Mixed trophic impacts of model groups in the Cape Verde coastal ecosystem for the period 1981 to 1985. The bars indicate relative net impact (negative/positive). Impacted groups are arranged along the horizontal axis and impacting groups down the vertical axis. Groups at lower trophic have been omitted in order to simplify the figure (see text on these groups).

Chapter 4. Changes in the Coastal Ecosystem over the Last Two Decades: a simulation study using Ecosim[7]

4.1 Introduction

The current state of knowledge on the marine environment in Cape Verde includes numerous studies of a generally descriptive nature. In relation to fisheries, many studies and characterisations have been undertaken, including single species stock assessments of economically important species such as lobsters and mackerel scad (*Decapterus macarellus*) (Almada, 1997; Carvalho *et al.*, 1999). On the other hand, recent studies include the first attempts to adopt the ecosystem approach to assess the direct and indirect effects of fishing (Coelho and Stobberup, 2000; Stobberup and Coelho, 2000; Stobberup *et al.*, 2002). These studies have used the Ecopath modelling approach in order to gain a better understanding of trophic relationships and system dynamics as well as the effects of fishing in the Cape Verde coastal ecosystem.

The Ecopath modelling approach is being widely used as a tool for the analysis of exploited ecosystems, including numerous studies from both temperate and tropical areas (Christensen and Pauly, 1993, 2004). In tropical environments, this modelling approach is particularly appealing because of the multi-gear and multi-specific nature of many tropical artisanal fisheries. Furthermore, tropical fisheries are often data-poor situations, which seriously limit the number of approaches possible such as the application of various data-driven multi-species approaches (*e.g.* MSVPA, GADGET, MULTSPEC, BORMICON) or even single-species age-structured models. There has been a tendency for advocates of the Ecopath approach to be over-zealous, which has had the unfortunate appearance of a crusade against the single-species approaches (*e.g.* Cook, 2001). This led in hand to overly harsh criticisms against Ecopath and a resistance to consider this as yet another tool in fisheries science. It is important to note however that the developers behind Ecopath consider it a complementary tool to single-species approaches (Christensen and Walters, 2004; Christensen *et al.*, 2001; Walters *et al.*, 1997) and warn against pitfalls and tunnel vision associated with the use of a single modelling approach (Christensen and Pauly, 2004).

The Ecopath with Ecosim (EwE) software has become a complex and powerful tool for modelling aquatic ecosystems. Recent applications have gone a step further and made a comparison of ecosystem characteristics between different time periods (*e.g.* Trites *et al.*, 1999; Heymans *et al.*, 2004; Neira *et al.*, 2004). Emphasis is increasingly being placed on the development and application of the dynamic simulation module, Ecosim, in the context of

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[7] This chapter is a slightly modified version of the original paper, Stobberup *et al.*, 2004.

applied research in relation to fisheries management (Walters *et al.*, 1997, 2000; Christensen and Pauly, 2004). Most recent studies have dealt with the simulation of ecosystem perturbations over time by varying fishing effort or mortality in order to study the effects in the system (*e.g.* Kitchell *et al.*, 1999, 2002; Shannon *et al.*, 2000, 2004; Manickchand-Heileman *et al.*, 2004). These simulation studies have in some cases tested hypotheses concerning predator/prey control (*e.g.* Cury *et al.*, 2000; Shannon *et al.*, 2000; Vasconcellos and Gasalla, 2001; Duarte and Garcia, 2004; Pinnegar and Polunin, 2004). The effects of environmental variables can also be incorporated in the exercise by including so-called forcing functions in Ecosim (*e.g.* Shannon *et al.*, 2004). More than 150 EwE models have been published (Christensen and Pauly, 2004), but it is nevertheless surprising that the incorporation of time series data in Ecosim studies has been rare with some notable exceptions (Christensen, 1998; Cox *et al.*, 2002; Martell *et al.*, 2002; Harvey *et al.*, 2003; Christensen *et al.*, 2003, 2004). This is regrettable as it is essential to be able to reproduce historical dynamics in order to determine the credibility of any model (Cox *et al.*, 2002).

The dynamic simulation module, Ecosim, has since its development been capable of incorporating time series data on relative or absolute abundance indices (*e.g.* survey data, catch per unit effort data), catches, fleet/gear effort, fishing rates, and total mortality estimates (Walters *et al.*, 1997, 2000; Christensen *et al.*, 2001). In this study, we present a simulation study based on the Cape Verde model (Stobberup *et al.*, 2002; Chapter 3), incorporating time series data on catch per unit of effort, survey biomass estimates, effort by fleet, and catches in order to study change over time. We attempt to reproduce the observed time series and show that model testing with simulation is essential for identifying possibly inconsistent parameter specification as well as other problems related to model structure. This aspect of model testing and calibration is an important part in any modelling study, but in this specific case, it is particularly important due to the data-poor situation in Cape Verde and the need to test parameter estimates obtained from the literature.

4.2 Materials and Methods

The Base Model

The Ecopath approach was originally proposed by Polovina (1984) and has since been developed extensively (Christensen and Pauly, 1992; Walters *et al.*, 1997, 1999, 2000). Most importantly, Ecopath no longer assumes a steady state, but instead bases parameterisation on an assumption of mass-balance over an arbitrary period, usually one year. Two master equations are defined for parameterisation, one to describe the production term and one for the energy balance of each group defined in the model. The first equation describes the production term for each group, represented by the following expression:

1) $$B_i \cdot (P/B)_i \cdot EE_i = BA_i + Y_i + \sum_{j=1}^{n} B_j \cdot (Q/B)_j \cdot DC_{ji}$$

where B_i is the biomass of group *i*, $(P/B)_i$ is the production/biomass ratio of group *i*, $(Q/B)_j$ is the consumption/biomass ratio of predators *j* that eat group *i*, and DC_{ji} is the average diet fraction of prey *i* for predator *j*. EE_i is the ecotrophic efficiency, or fraction of total mortality that is accounted for by the modelled system, and Y_i are fisheries yields. The biomass accumulation term BA_i accounts for trends in absolute biomass during the period over which biomass estimates were made.

A set of linear equations as defined above, one for each group in the model, are solved simultaneously, ensuring balance between energy input and output for all the groups, which can be expressed as (second equation):

2) Consumption = Production + Respiration + Unassimilated food

The Cape Verde model considers the total area of the continental shelf system, assuming a homogeneous area of 5 394 km^2. The oceanic waters between islands were not included as they were considered a different ecosystem. However, we assume interaction between the coastal and oceanic ecosystems through the foraging of oceanic and/or migratory species, following the recommendation by Christensen *et al.* (2001). This was specified as an "import" of food in the diet matrix, which was estimated by assuming the proportion of oceanic species in the diet as import.

Fisheries statistics were available for the period from 1981 to 2000 (INDP, 2001), but estimates for the period 1981 to 1985 are not considered reliable (Carlos Monteiro[8]; *pers. comm.*). This was the period of implementation of the statistical collection system and the high estimates given are the result of errors in extrapolation. An artisanal and an industrial fishery were defined in the model. Thus, the simulation studies included fisheries catch, effort and catch per unit of effort (CPUE) starting from 1986, which coincides with the creation of the National Fisheries Institute (INDP) in Cape Verde.

Twenty-seven ecological groups were defined, including small mammals, seabirds, turtles, 18 fish groups, 2 benthic invertebrate groups, zooplankton, 2 primary producers and detritus (Table 4.1). The study by Stobberup *et al.* (2002) gives a detailed description of the creation of the Cape Verde Ecopath model, including sources of information as well as procedures in parameter estimation. However, a brief description is included in the following.

Parameter estimation

Concerning the fish species included in the model, a total of 99 species were specified based on criteria such as their relative importance in trawl, handline, and longline surveys as well

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[8] Head of Fisheries Statistics Department, INDP, Cape Verde

as in the fisheries. The aggregation into ecological groups was based on characteristics such as diet, habitat, common and maximum length, estimated trophic level and consumption. Biomass estimates for the more abundant demersal and neritic pelagic fish species were obtained from survey reports (Magnússon and Magnússon, 1985, 1987-a; Marques *et al.*, 1997; Palsson, 1989; Stroemme *et al.*, 1982; Thorsteinsson *et al.*, 1995).

Fish Consumption/Biomass Ratios (Q/B) were estimated using the empirical equation of Palomares and Pauly (1999). In the case of demersal fish, which are relatively well-sampled by trawl gears, the Production/Biomass Ratio (P/B) was estimated by assuming P/B equal to Z (total mortality), which applies under steady-state conditions (Allen, 1971). Natural mortality (M) was estimated using the empirical equation proposed by Froese and Binohlan (2000). Fishing mortality (F) was estimated from F = Catch / Biomass (C/B), also under a steady-state assumption, when fish biomass estimates were available. Thus, total mortality can be calculated as Z = M + F.

Q/B and M were estimated for each fish species, in most cases, and a weighted average, based on biomass, was calculated for demersal fish groups. In the case of migratory/pelagic species, catch was used as a relative indicator of species abundance in each group and the parameters were weighted accordingly. For other groups, a simple average was calculated as the relative abundance of the constituent species is poorly known. The same procedure was used in estimating diet composition by fish group.

Fishbase 2000 (Froese and Pauly, 2000) was an invaluable source of information for most fish species, including habitat, diet compostion, common and maximum length, and von Bertalanffy parameters L_∞ and W_∞, including the empirical relations given therein (Keyfacts). Local information on diet composition, in quantitative terms, was available only for mackerel scad, *Decapterus macarellus* (Almada, 1997).

Table 4.1 shows that P/B and B were estimated by the model in many cases, because information was lacking for these groups. However, biomass was estimated for yellowfin and skipjack tuna, based on production estimates for Cape Verde given in the literature (Hallier, 1999). These tuna species play an important role in the system as abundant predators, migrating through the system. Thus, it is essential to impose reasonable constraints in terms of biomass as this will have bearing on the model results.

When allowing the model to estimate P/B and B, the production/consumption (P/Q) ratio was assumed to be 0.1 and the ecotrophic efficiency (EE) was fixed, varying from 0.5 for top predators (and benthic algae), to 0.8 for highly migratory groups, and 0.9 for other groups, which were considered reasonable values in light of other studies (*e.g.* Polovina, 1984; Christensen and Pauly, 1993; Kitchell *et al.*, 1999).

For non-fish groups, local information is very limited and mostly of a qualitative nature. In order to tackle this problem, we assume that ecosystem characteristics in Cape Verde are similar to other tropical areas in terms of trophic functioning and community structure.

Information for other groups was estimated using a number of references; phytoplankton (Longhurst, 1998), benthic autotrophs (Opitz, 1993), zooplankton (Braun *et al.,* 1982), benthic invertebrates (Opitz, 1993; Silvestre *et al.,* 1993), seabirds (Le-Grand, 1986; Nagy, 1987), and small mammals (Reiner *et al.,* 1996; Pauly *et al.,* 1998-b; Bundy *et al.,* 2000).

Simulation with Ecosim

Simulation with Ecosim consists of biomass dynamics expressed through a series of coupled differential equations, which are based on the first equation (1) defined in the Ecopath base model (Christensen *et al.,* 2001):

$$3)\qquad dB_i/dt = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i) \cdot B_i$$

where dB_i/dt represents the growth rate during the time interval dt of group i in terms of its biomass, B_i, g_i is the net growth efficiency (production/consumption ratio), M_i the non-predation ('other') natural mortality rate, F_i is fishing mortality rate, e_i is emigration rate, I_i is immigration rate. The two summations estimate consumption rates, with the first expressing the total consumption by group i, and the second the predation by all predators on the same group i .

Ecosim was used to simulate the dynamics of the system, incorporating time series information on biomass, catches, and CPUE for a number of groups as well as effort by fishery for the period 1986 to 2000. Ecosim simulates the effects of fishing and predation on each group in the system, including possible changes in food availability and the indirect effects of fishing or predation on other groups in the system. Ecosim provides several options in relation to time series fitting, such as prey vulnerability parameters and forcing functions (Walters *et al.* 1997; Christensen *et al.,* 2001). However, default settings were used for vulnerability parameters (0.3) and forcing functions were not applied as the main objective of the study was to determine whether the Cape Verde model was able to simulate biomass, catch and CPUE time series realistically with as few adjustments as possible.

Table 4.1. Comparison of parameter estimates for the start (1986) and end (2000) of the simulation. The start parameters correspond to the model created for the period between 1981 to 1985, using available information (shaded). See text for explanation on procedure adopted for simulating forward to 2000. (abbreviations: pel – pelagic; dem – demersal; O Demersal fish – other non-commercial demersal fish).

	Trophic level		**Biomass (t/km²)**		**Prod./ biom. (/year)**		**Cons./ biom. (/year)**		**Ecotrophic eff.**		**Fishing mortality**	
Group name	***Start***	***End***	***Start***	***End***	***Start***	***End***	***Start***	***End***	***Start***	***End***	***Start***	***End***
Sea birds	3.7	3.8	0.020	0.021	0.250	0.264	109.500	114.329	0.50	0.59	0.049	0.094
Mammals	4.4	4.4	0.034	0.028	0.100	0.124	11.790	11.912	0.50	0.42	0.029	0.056
Billfish	4.3	4.3	0.083	0.064	0.300	0.313	6.400	6.404	0.80	0.77	0.024	0.047
Yellowfin	3.5	3.6	1.803	0.490	0.800	0.909	11.493	12.607	0.84	0.86	0.350	0.696
Predators pel	3.7	3.8	0.186	0.167	0.991	1.062	9.911	10.449	0.80	0.83	0.253	0.489
Skipjack	3.5	3.5	0.457	0.179	0.970	1.038	12.500	13.576	0.91	0.95	0.350	0.797
Small tuna	3.4	3.4	0.493	0.776	1.498	1.354	14.978	13.653	0.90	0.80	0.059	0.176
Jacks	3.5	3.5	0.250	0.230	0.475	0.476	4.754	4.894	0.90	0.93	0.044	0.132
Moray eels	3.9	4.0	0.149	0.095	0.280	0.346	2.800	3.071	0.90	0.83	0.061	0.181
Sharks pel	4.6	4.6	0.045	0.036	0.100	0.128	2.668	2.604	0.50	0.32	0.022	0.042
Bathydemersal	3.8	3.8	0.224	0.316	0.384	0.314	3.844	3.657	0.90	1.02	0.004	0.013
Sharks dem	3.7	3.6	0.137	0.126	0.260	0.264	2.600	2.602	0.90	0.89	0.007	0.022
Predators dem	3.7	3.8	0.216	0.123	0.561	0.679	4.323	4.863	0.90	0.87	0.153	0.455
O Demersal fish	3.1	3.1	1.893	2.179	0.729	0.713	5.142	4.994	0.87	0.83	0.001	0.003
Demersal fish	3.4	3.4	1.456	1.746	0.561	0.535	5.759	5.565	0.71	0.65	0.009	0.027
Small pelagics	3.0	3.0	4.635	5.260	1.111	1.075	11.108	10.751	0.97	0.96	0.048	0.143
Reef feeders	3.2	3.2	0.811	0.847	0.549	0.534	5.494	5.415	0.90	0.91	0.002	0.007
Sparids	2.7	2.7	1.841	2.178	0.680	0.647	16.247	15.643	0.72	0.65	0.004	0.011
Flyingfish	3.0	3.0	0.758	0.395	1.430	1.535	14.300	15.039	0.90	0.91	0.009	0.028
Herbivores	2.0	2.0	0.968	0.885	0.585	0.598	15.863	15.987	0.85	0.84	0.004	0.012
Turtles	2.8	2.8	0.139	0.089	0.150	0.215	3.500	3.716	0.50	0.46	0.043	0.128
Crabs/Shrimps	2.7	2.8	13.699	12.821	1.600	1.636	10.000	10.200	0.95	0.95	0.000	0.001
Benthos	2.1	2.1	33.280	34.512	3.000	2.966	12.500	12.371	0.95	0.95	-	-
Zooplankton	2.0	2.0	2.186	2.100	63.440	65.923	280.000	290.886	0.95	0.95	-	-
Phytoplankton	1.0	1.0	2.360	2.414	285.000	281.744	-	-	0.95	0.95	-	-
Benthic autotrophs	1.0	1.0	18.879	18.527	13.250	13.376	-	-	0.50	0.51	-	-
Detritus	1.0	1.0	1.000	42.574	-	-	-	-	0.66	0.66	-	-

4.3 Results and Discussion

The period from 1981 to 1985 was chosen so that the model corresponds to the initial phase of a more intensive fishery development. Artisanal fisheries dominated at the time, consisting of small boats with or without outboard motors. Fish stocks were considered to be in their virgin state at the time (Magnússon and Magnússon, 1985, 1987-a). It was also during this period that a statistical collection system was established and the first systematic surveys were undertaken, providing estimates of fisheries catches and biomass for different fish stocks.

Considering the location of the Cape Verde Archipelago, the coastal areas are expected to have strong oceanic influence. The presence of many oceanic species is an example of this (Reiner, 1996) as well as the fact that Cape Verde is in the migratory route of several tuna species (Bard *et al.*, 1993; Hallier, 1999; ICCAT, 2001-a, 2001-b) Migratory species such yellowfin and skipjack tuna are important predators and constitute a major resource for the fisheries in Cape Verde, making it extremely important to deal with this issue in the model. We followed the recommendation by Christensen *et al.* (2001), treating the migratory groups as part of the system, but obtaining part of their food outside the system (and/or the time spent outside the system). Another option would have been to estimate dispersal rates (immigration/emigration), which have not been studied in Cape Verde. Also, a problem in connection with the incorporation of dispersal rates in the model is that Ecopath will deal with flows as "density independent" as a constant proportion of immigration/emigration irrespective of the biomass, which is not realistic (Christensen *et al.*, 2001).

There is considerable uncertainty attached to the specified model parameters. Q/B and M were estimated with empirical equations, which are generally applicable but do not account for local conditions. For biomass of demersal fish, only two estimates were available for the whole period. The results of the trawl surveys undertaken in 1984 and 1985 were used for the initial model, leaving one estimate for 1994 in the simulation study. This was further aggravated by the fact that the trawl survey in 1994 did not cover as large an area as the first survey. For small pelagics, consisting primarily of *Decapterus macarellus*, acoustic biomass estimates were available for 1981 and 1997. However, these estimates gave a decrease from around 12 to 2.5 tonnes per km^{-2}, which is questionable. Also, biomass estimates were not available for yellowfin and skipjack tuna, so these were based on production estimates for Cape Verde given in the literature (Hallier, 1999). It was essential to impose reasonable constraints in terms of biomass, as these tuna species are important predators and are targeted by the fisheries.

The first attempts failed to adequately simulate the observed values for biomass, catch and CPUE. Available estimates of relative fishing rate over time showed modest increases and could not account for the observed catches. Furthermore, if fishing rate was increased, the biomass of tuna tended towards extinction. Thus, these initial simulations were used to identify problems and inconsistencies in the model given in Stobberup *et al.*, (2002) (or Chapter 3). The following gives a summary of the modifications undertaken:

- Previously, a large tuna group was defined, including yellowfin and skipjack. This was split into two groups in order to simulate differences in consumption and mortality as well as catches by the different fisheries. Also, more recent estimates for Q/B were available from Kitchell *et al.* (1999).
- Only two fisheries were considered; artisanal and industrial. Originally, the artisanal fishery was divided into handline and seine/gillnet fisheries, but the quality of effort data for the seine fishery made this distinction senseless.
- The 1981 acoustic biomass estimate (12.05 tonnes per km^{-2}) for small pelagics appeared unreasonable. Instead, a more reasonable estimate of 4.64 tonnes per km^{-2} was obtained from a recent stock assessment study (Almada, 1997).
- This reduction in biomass of small pelagics made it necessary to increase the import of food in the model for pelagic/migratory species in order to provide sufficient food (Appendix F).
- The original model was simplified by reducing the number of groups in an attempt to reduce uncertainty. Information on the original groups rays and flatfish was almost non-existent, so these were joined with demersal sharks and demersal fish, respectively. Also, several benthic invertebrate groups were joined to form one benthos group as the emphasis was placed on higher trophic level groups, fish in particular.

Estimates of diet composition were based on information given in the literature, preferably in other tropical areas. This diet information had to be standardised into major groups (*e.g.* fish, zooplankton, phytoplankton, polychaeta, etc.), which resulted in a base matrix with major food items. In practical terms, this could be integrated directly for the lower trophic levels as the model definition at this level maintains a similar separation into major groups. The balancing of the model and improvements to the simulation results were obtained by minor adjustments to the diet matrix (Appendix F).

A different approach to simulation was thereafter adopted. Emphasis was placed on obtaining fits to the catch time series, in particular, as well as CPUE. This reasoning was based on the fact that statistical sampling programmes tend to provide better estimates of catch (lower coefficient of variation) compared to effort as well as CPUE (Bellemans and Monteiro, 2000). Few biomass estimates were available and less emphasis was placed on fitting simulation results to biomass. The use of available estimates for relative fishing rate (or effort) was abandoned and these were estimated by trial and error.

A reasonable overall fit to the time series of catches was obtained for the 18 fish groups by using three overall trends in relative fishing rate (Figure 4.1). The trend estimated for the industrial fishery was similar to observed values, but increased slightly. This is reasonable as the industrial fishery is well sampled, at least in recent years, and the slight increase adopted can be interpreted as an increase in efficiency, which was not captured by measuring effort as fishing days. A triplication of relative fishing rate was estimated for the artisanal fishery, which was double the effort measured in fishing trips. This appears to indicate that effort is not well sampled and is further aggravated by the fact that efficiency must have increased with the motorization of boats over time. Simulation results improved by assuming a duplication of relative fishing rate for migratory species, including large pelagic fish as well as small mammals. Thus, fishing pressure on coastal species was assumed to have increased more strongly. However, it was necessary to apply adjustments to fishing rate on yellowfin tuna separately in order to fit observed catch and CPUE.

Figure 4.2 shows the simulation results for important migratory species such yellowfin and skipjack tuna as well as pelagic predators. The resulting fits for both CPUE and catches are reasonable for all these groups. The decrease in CPUE of yellowfin and skipjack has been confirmed by ICCAT (2001-a, 2001-b) and was thus incorporated in the model. This decrease in abundance of important predators has important consequences on the modelling results as will be seen in the following.

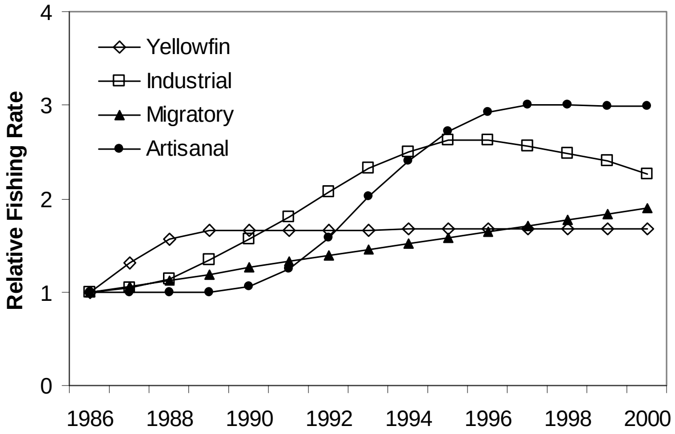

Figure 4.1. Trends in relative fishing rate used for simulating from 1986 to 2000. Apart from the artisanal and industrial fisheries, specific trends were used for yellowfin tuna and other large migratory species (Migratory) such as small mammals, billfish, pelagic predators, and pelagic sharks. See text for explanation.

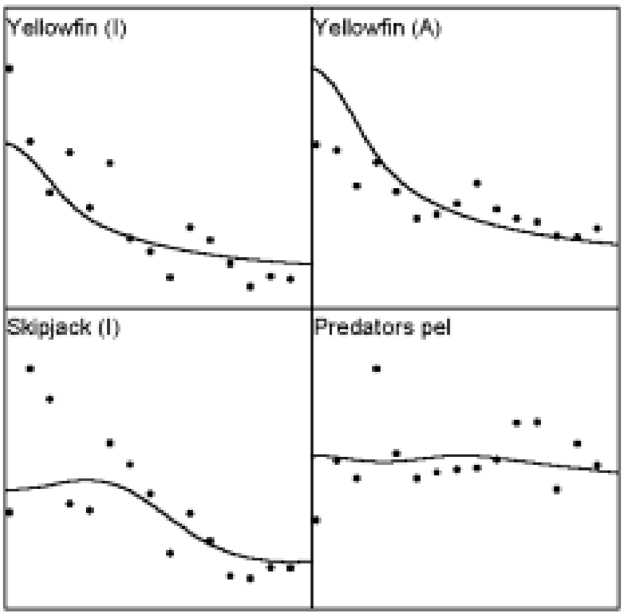

Figure 4.2. Comparison of simulated (lines) and observed (dots) abundance of important pelagic, migratory species such as yellowfin and skipjack tuna as well as pelagic predators (mostly *Acanthocybium solandri*). Catch per unit effort (CPUE) was used as the observed index of abundance. In the case of yellowfin, both the artisanal (A) and industrial (I) indices are given.

Model estimates of biomass were poor (not shown). This is not surprising since these estimates were considered unreliable, with only two points available for the time period. It should also be noted that one estimate was used for the creation of the Ecopath model, leaving only one estimate for the fitting process in Ecosim.

The results of the fit to observed CPUE time series are shown in Figure 4.3. These CPUE estimates were based on the artisanal fishery and they cannot be considered as good indices of abundance. The artisanal fishery is a typical multi-gear and multi-species fishery, which makes it difficult to obtain reliable indices of abundance from fishery data. On the other hand, the CPUE estimates for tuna species were considered more reliable as these are well-defined targets of the fisheries.

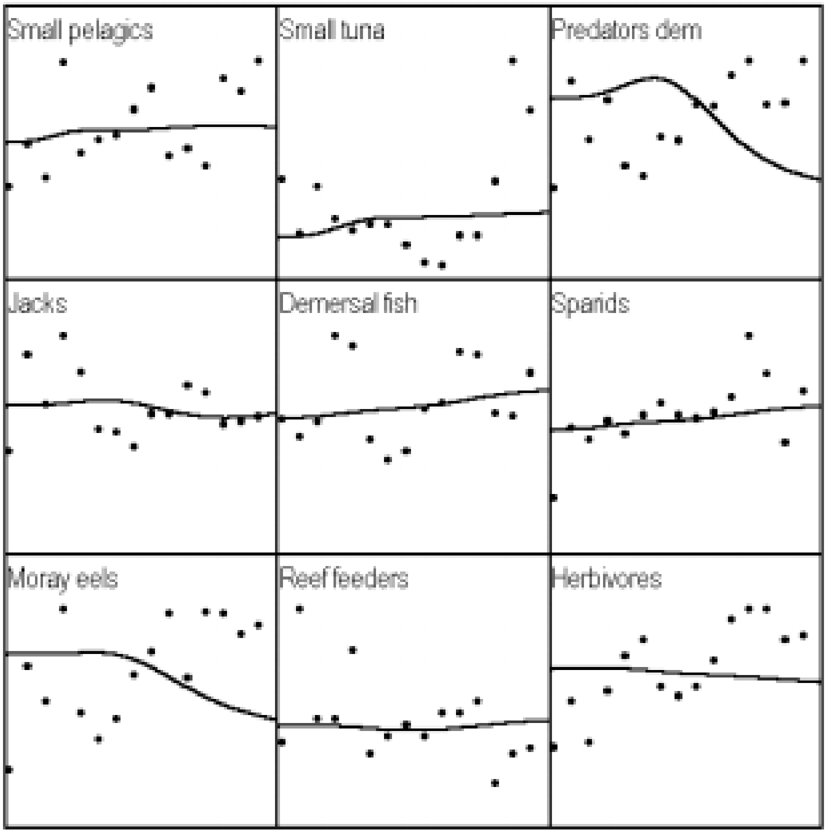

Figure 4.3. Comparison of simulated (lines) and observed (dots) abundance of groups for which survey biomass estimates (not shown) were available. Catch per unit effort (CPUE) was used as the observed index of abundance. See Table 3.3 for the dominant species in each group.

In the case of simulated and observed catch estimates, good fits were obtained for small pelagics, which is an important prey group for tunas and other pelagics (Figure 4.4). The fit for small tuna was less satisfactory, but the trend is correct, showing increases in catch and biomass as a result of less predation from larger tuna (Figure 4.3 & 4.4). For various demersal groups, reasonable good fits were obtained for both catches and CPUE, particularly in recent years (Figure 4.3 & 4.5). The poor fit to observed catches in the early years was a frequent result for many demersal and coastal pelagic groups (Figure 4.5 & 4.6). It is difficult at this stage to determine whether this may have been an environmental effect or due to a relative increase in fishing rate.

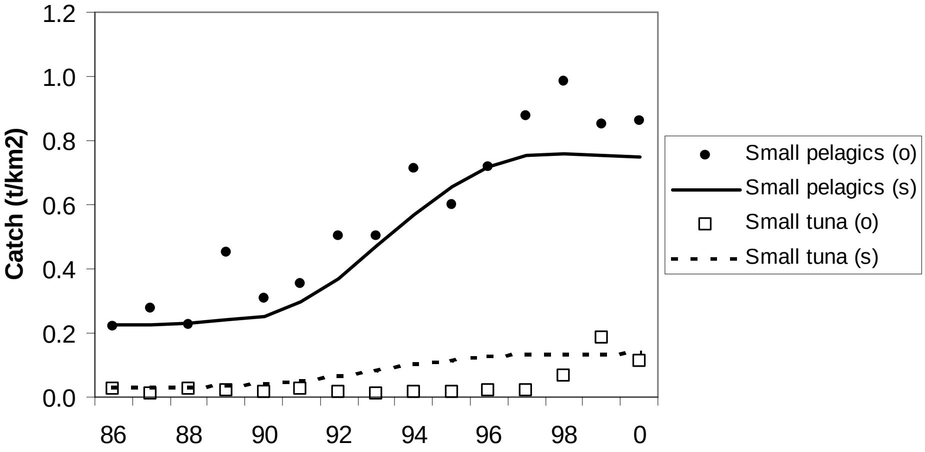

Figure 4.4. Comparison of simulated (lines) and observed (symbols) catches of Small pelagics and Small tuna. Abbreviations: (o) – observed; (s) – simulated.

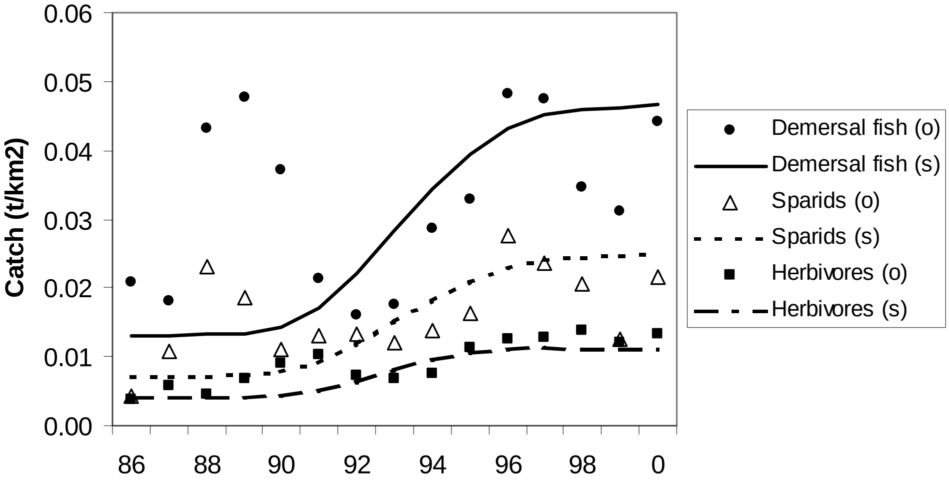

Figure 4.5. Comparison of simulated (lines) and observed (symbols) catches of Demersal fish, Sparids and Herbivores. Abbreviations: (o) – observed; (s) – simulated.

A different pattern was observed for important commercial groups such as demersal predators and moray eels. The fit to observed catches was reasonable in the middle of the period, but tended to become poor in recent years, where observed catches were higher than simulated catches (Figure 4.6). At the same time, a strong decrease in simulated biomass was obtained. This may be an indication that these groups are being over-exploited (Figure 4.3).

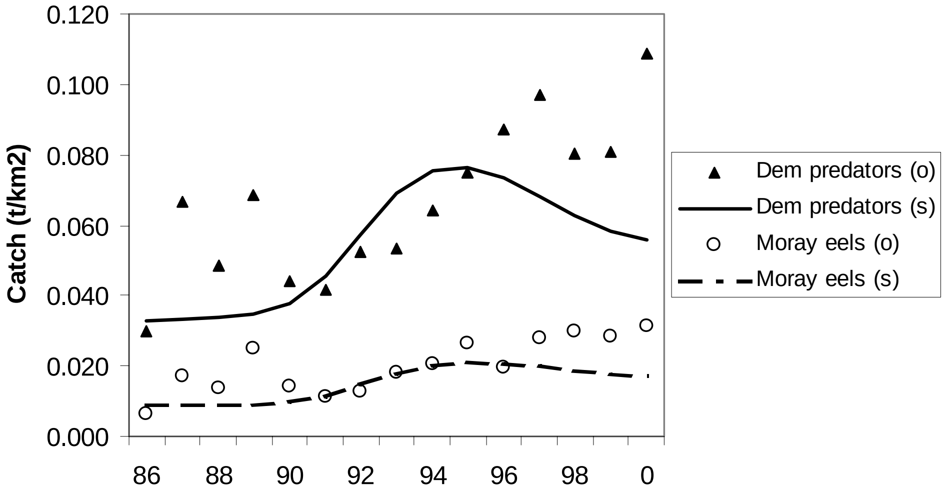

Figure 4.6. Comparison of simulated (lines) and observed (symbols) catches for Demersal predators and Moray eels. Abbreviations: (o) – observed; (s) – simulated.

A reasonable overall fit to time series of catches and CPUE was obtained for 18 fish groups, using only three overall trends in effort and minor adjustments to fishing rate on yellowfin tuna. This approach was intentional in the sense that adjustments to relative fishing rate were kept to a minimum. The fit could easily be improved by modifying fishing rate for each species, but this would be unrealistic as it implies the existence of fisheries with very specific target species. However, fisheries in Cape Verde can be characterised as tropical, multi-gear, multi-species fishery, the artisanal fisheries in particular.

The observed decrease in abundance of important predators such as yellowfin and skipjack tuna resulted in decreased predation on neritic pelagic species and some demersal fish groups. Consequently, the model estimated a 13 percent increase in biomass of coastal (neritic) fish species from 1986 to 2000 (Table 4.1). Simulated biomass of small pelagics, small tuna, and most demersals increased as a result of lesser predation. Most demersal groups increased in biomass a result of less predation from predators such as demersal predators and moray eels, which decreased in biomass. Overall fish biomass, including pelagic migratory species, remained almost constant (- 2%). On the other hand, relative fishing effort was assumed to have doubled or almost tripled over the time period from 1986 to 2000, but this resulted in only a 36 per cent increase in catches, from 6 800 to 9 200 tonnes. In fact, the official statistics indicate a total catch of 10 800 tonnes in 2000, which means that relative fishing rate should be increased slightly in the simulation. These results indicate that previous assessments of potential harvest, which range from 25 000 to 56 000 tonnes (Aubray, 1977; Diouf, 1992; Moal, 1977), appear to have been overly optimistic.

The modifications that were applied to the model, such as revised consumption and production rates for tuna and lower biomass of small pelagics, had profound effects on the results. For example, total standing fish stock decreased from about 124 000 tonnes (Chapter

3) to 88 000 tonnes. This decrease in total fish biomass made the previous estimates of potential harvest appear even more optimistic (equivalent to a removal of about 28 to 64% of fish biomass). Groups such as demersal predators and moray eels, which are high-valued commercial species, appeared to show signs of over-exploitation and should be monitored carefully. An increase in total catch appears nevertheless to be feasible, if managed properly, but a large increase seems unlikely. The simulation results showed that fishing pressure increased considerably over the period from 1986 to 2000 without bringing about the expected increases in harvest. This is also related to the fact that external factors have a major role determining the biomass of tuna, which are important predators in the system.

Further research is necessary in order to gather supporting evidence for the simulation results. Previous studies have indicated a decrease in biomass of commercial demersal fish (Monteiro, 1999-b), which corresponds to the group of the same name in this study, but this is in conflict with our results. This issue has to be resolved, including an in-depth revision of the fishery statistics in Cape Verde. One possible way forward would be to incorporate tuna completely by creating an oceanic model and linking this to the Cape Verde model. In this way, the tuna groups can be modelled explicitly without making assumptions on the amount of food imported or the time spent in the system.

Chapter 5. Assessing small pelagic resources in Cape Verde: a comparison of the "frequentist" and Bayesian approaches to biomass dynamic modelling[9].

5.1 Introduction

Small pelagic resources are particularly important in Cape Verde, constituting around 45 percent of total catches in recent years. This large proportion corresponds to around 4 200 tonnes annually. In terms of species, the mackerel scad (*Decapterus macarellus*) has always been a major part of the catches (Figure 5.1). Two other species, *Spicara melanurus* and *Selar crumenophthalmus*, have recently gained importance in artisanal and industrial catches, respectively. Most of the catches are taken with seine gears, both in the industrial and artisanal fisheries, although gillnets are increasingly playing an important role in the catches of *Spicara melanurus* around Santiago Island, in particular. Other species such as *Decapterus punctatus* and *Sardinella maderensis* are caught in minor quantities in the small pelagic fisheries.

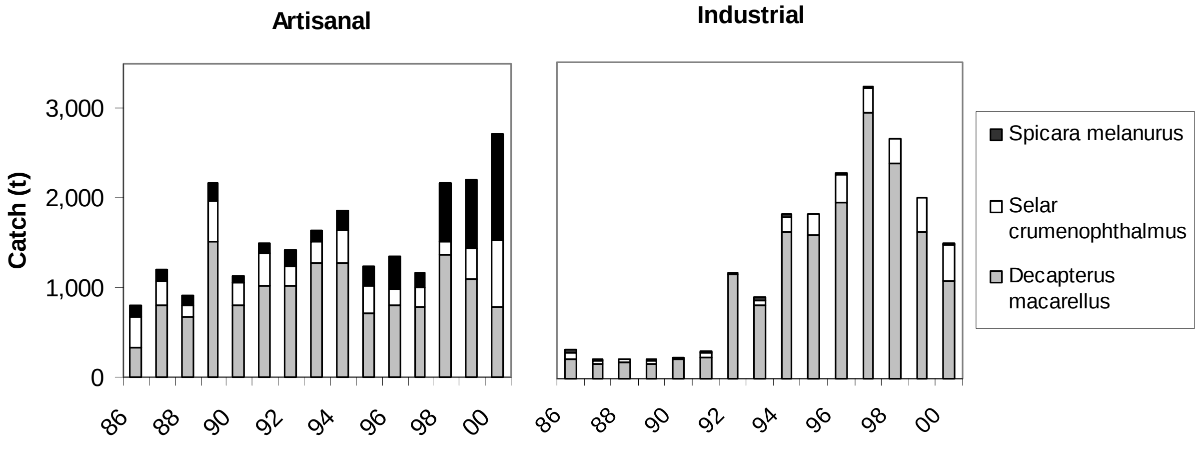

Figure 5.1. Artisanal and industrial catches of small pelagics in Cape Verde (INDP, 2001).

The mackerel scad stock has been assessed on several occasions (Carvalho, 1993; Almada, 1997; Carvalho and Caramelo, 1999; Jardim, 1999), but these assessments need updating, including the application of more recent methodology. This is particularly urgent in the case of mackerel scad due to its importance in terms of catch weight and value in the national market.

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[9] A paper is under preparation for submission to Fisheries Research, based on this chapter.

Biomass dynamic models are one of the simplest analytical methods available that provide for a full fish stock assessment. Catch estimates and an abundance index for a number of years are the minimum requirements for this method, although ancillary data may be incorporated to improve parameter estimation. The logistic formulation in terms of biomass became known as the surplus production model (also called the Schaefer model; Schaefer, 1954, 1957), while the more recent term biomass dynamic models implies a dynamic fitting process (Hilborn and Walters, 1992; Haddon, 2001). The original method of assuming equilibrium conditions has been criticised for providing overly optimistic estimates of optimum effort and Maximum Sustainable Yield (MSY) and has been associated with many failures in management (Botsford *et al.*, 1997; Mace, 2001; Pauly *et al.*, 2002). Equilibrium fitting methods, which assume a linear relationship between catch per unit of effort (CPUE) and effort, are now known to produce biased and unreliable parameter estimates and their use is not recommended (Hilborn and Walters, 1992; Haddon, 2001; Williams and Prager, 2002). On the other hand, the use of dynamic fitting methods and modern computers for parameter estimation has brought about a revival of biomass dynamic models, including estimates of uncertainty around parameters and risk assessment (Prager, 1994, 2002; McAllister and Kirkwood, 1998; McAllister *et al.*, 2000, 2001; Abuanza *et al.*, 2003; Carbonell and Azevedo, 2003; Mendoza and Larez, 2004).

One of the major difficulties in modelling fisheries data is the lack of contrast between fishing effort and stock abundance, which has been termed the "one-way trip" (Hilborn and Walters, 1992). This is observed as a steady decline of CPUE over time, which is very common in fisheries data sets. In the absence of ancillary information, an infinite number of plausible solutions exist in such a case, ranging from a large (biomass) stock with a low growth rate to a small stock with a high growth rate. All methods are expected to fail in this type of situation due to data failure (not model failure), including models using equilibrium and non-equilibrium fitting methods as well as age-structured models (Hilborn and Walters, 1992; Haddon, 2001).

Biomass dynamic models are usually considered a poor substitute to age-structured models, but they may be the only method available for carrying out a full stock assessment in data-poor situations such as in many tropical fisheries. Comparative studies of biomass dynamic and age-structured models have nevertheless shown that both types of models perform equally well, giving the same answers for management, and that in some cases, biomass dynamic models may provide better estimates of management parameters and are thus more robust (Hilborn and Walters, 1992; Punt, 1995; Abuanza *et al.*, 2003). Age-structured analysis is often not practical in tropical fisheries and the more simple requirements of biomass dynamic models make them a cost-effective solution. Furthermore, it may be difficult to obtain species specific catches and CPUE in tropical fisheries, making it necessary to treat the entire catch (or part) as a biomass dynamic pool. Conventional age-structured analysis would not be appropriate in such cases.

We carried out an assessment of small pelagic resources (5 species introduced above) considered as “bulk” biomass, using biomass dynamic models. Mackerel scad could not be assessed specifically due to difficulties in obtaining a reliable abundance estimate for this species alone. Non-equilibrium approaches to modelling were applied on a comparative basis using the bootstrap (a type of “frequentist” approach) and Bayesian approach. In order to tackle the “one-way trip” problem, the intrinsic rate of increase, r, was estimated separately, inspired by recent developments such as the meta-analytic, comparative and demographic approaches to estimate specific parameters externally in order to reduce uncertainty in stock assessment (Cortés, 1998; Hilborn and Liermann, 1998; McAllister *et al.*, 2001; Myers, 2001; MacKenzie *et al.*, 2003). In relation to Bayesian statistics, this is termed the prior knowledge of r specified as a probability density function. Uncertainty in parameter estimates and risk assessment of projections was addressed by using bootstrap and Bayesian techniques.

5.2 Materials and Methods

Fisheries data on catch and effort are available starting in 1981 with the implementation of a national statistical collection system for fisheries. As catch estimates are not considered reliable for the early years (see Chapter 2 for more details), we consider the time period from 1987 to 2001 in the present study. These catch and effort data were available at different levels of aggregation.

- Aggregated and raised yearly estimates for the period 1987 to 2001; available for artisanal and industrial fisheries separately.
- Aggregated and raised monthly estimates for the period 1996 to 2000; available by gear and island for artisanal and industrial fisheries.

Preliminary analysis was carried out with non-parametric Multi-Dimensional Scaling (MDS) to determine whether there are important spatio-temporal effects to consider (see Chapter 6 for more details on this type of analysis). MDS was applied to a similarity matrix calculated with the Bray-Curtis coefficient based on monthly CPUE data by species and gear (1996-2000). The Bray-Curtis similarity coefficient was used as it has the desirable property of not being sensitive to joint absences or double zeros, which was common in the data (zero catches for a specific species). The resulting ordination plot showed that there was a clear pattern in target species according to the gear used, but most importantly, that yearly, seasonal and island effects could be ignored (not shown).

Standardised Abundance Index

CPUE is often used as an index of abundance in the absence of fishery-independent data such as the results of resource surveys. In such cases, a standardized CPUE index is modelled, typically with Generalized Linear Models (GLM), in order to remove the effects of different fleets and/or gears (*e.g.* Hilborn and Waters 1992). As the data available for this study was highly aggregated, the approach used is similar in concept but highly simplified. There were two CPUE time series available from the artisanal and industrial fisheries, which are based on effort measured in fishing trips (typically a day) and fishing days at sea, respectively (Table 5.1). These effort units were assumed to be approximately the same (fishing days) and the artisanal CPUE was scaled up in relation to the indutrial CPUE (Figure 5.2). The standardized CPUE was estimated as the average of these two. The industrial CPUE estimates for period 1987 to 1991 were not considered in this calculation, as there was a significant shift in catch rates due to the introduction of new more efficient vessels (Table 5.1).

Table 5.1. Catch (t) and CPUE (kg per fishing day) of small pelagics in the artisanal and industrial fishery. The shaded area indicates that these industrial CPUE values were omitted from the estimation of standardized CPUE (see further explanation in text).

Year	Catch (t)	% of total	Artisanal CPUE	Artisanal Scaled	Industrial CPUE	Std. CPUE
1987	1453	18.0	493	1064	96	1064
1988	1180	18.7	477	1030	133	1030
1989	2436	28.5	291	628	93	628
1990	1556	23.7	249	538	121	538
1991	1834	26.6	268	578	150	578
1992	2622	40.0	195	422	904	663
1993	2562	36.2	285	616	455	536
1994	3717	45.0	317	684	645	665
1995	3103	36.5	233	503	590	546
1996	3693	40.4	196	424	672	548
1997	4484	46.1	256	553	643	598
1998	5026	51.6	228	492	513	502
1999	4276	41.4	164	355	342	349
2000	4329	42.7	185	401	320	361
2001	3307	37.5	166	358	475	417

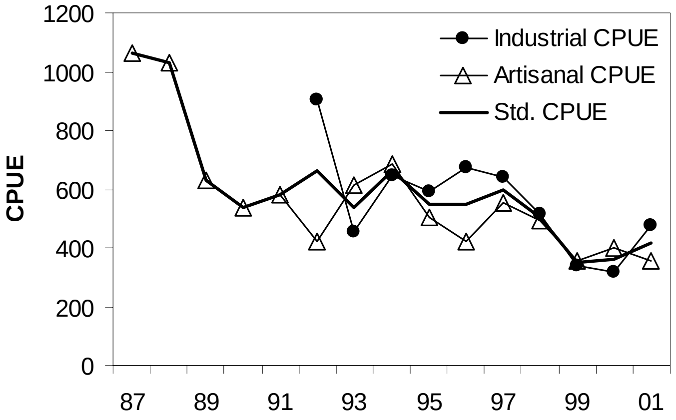

Figure 5.2. CPUE (kg per fishing day) of small pelagics.

Biomass dynamic model

The formulation of the Schaefer surplus production model (Hilborn and Walters, 1992; Haddon, 2001) in its continuous and discrete form is:

(1) $$\frac{dB_t}{dt} = rB_t - \frac{r}{K}B_t^2 - C_t \qquad \text{or} \qquad B_{t+1} = B_t + rB - \frac{r}{K}B^2 - C_t$$

where *r* is the intrinsic rate of increase, *K* is the carrying capacity, B_t is the abundance (biomass), and C_t is the catch at time *t*. In the Schaefer model, the biomass level that sustains the maximum sustainable yield (*Bmsy*) is at one half of *K*. Maximum sustainable yield is defined as:

(2) $$Y_{MSY} = \frac{rK}{4}$$

and the effort that sustains the maximum sustainable yield is:

(3) $$f_{MSY} = \frac{r}{2q}$$

where *q* is the catchability coefficient, also called a nuisance or scaling parameter (Haddon, 2001).

Specifying r

Preliminary trials at modelling biomass dynamics of small pelagics in Cape Verde were unsatisfactory. Parameters of the Schaefer model were ill-defined, resulting in large uncertainty around parameter estimates and reference points for management. This was due to lack of contrast in the CPUE time series or in other words, "the one-way trip". To overcome this limitation, *r* was estimated using the equation proposed by Sullivan (1991) for non-gadoid species. In this formulation *r* is a function of the von Bertalanffy growth coefficient (*k*) and of asymptotic weight (W_∞):

(4) $r = 0.947 + 1.189k - 0.095 \ln(W_\infty)$

Growth parameter estimates for small pelagics were calculated as the mean of estimates for constituent species weighted by the catch ($k = 0.386$ and $W_\infty = 1347$ gr.), assuming that abundance is proportional to catch (see Appendix C for further details). This gave a point estimate of 0.721 per year for *r* and consequently, a lognormal probability density function (pdf) was defined (lognormal median = 0.721, lognormal mean = 0.767, lognormal SD = 0.350; Figure 5.3), corresponding to the uncertainty around this estimate. The value for standard deviation (SD) was chosen arbitrarily so as to correspond to a CV (coefficient of variation) of about 50 % (46% in this case). This follows the recommendation by several authors to avoid an overly precise estimate that may be biased (Punt and Hilborn, 1997; McAllister and Kirkwood, 1998). It is important to point out that this estimate of *r* and its probability density function (pdf) was used in both the "frequentist" and Bayesian approaches, where *r* is considered known or defined as a prior, respectively.

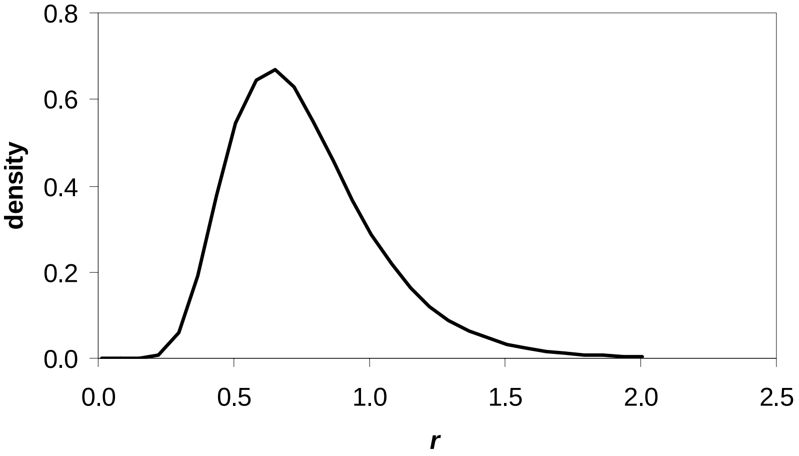

Figure 5.3. Estimated probability density function for r.

Dynamic fitting process

An observation error model was used following Haddon (2001), where equation 1 describes stock dynamics deterministically (no process error) and all residual errors are assumed to occur in the relationship between stock biomass and the index of relative abundance. Biomass can therefore be estimated by projecting forward the initial biomass, B_0, at the start of the time series, using the historical annual catches. The discrete version of equation 1 is used to produce the predicted series of biomass, and these are used to produce a predicted series of CPUE (I) values:

$$\hat{I} = q\frac{(B_{t+1} + B_t)}{2}e^{\varepsilon} \tag{5}$$

where e^{ε} indicates that the residual errors are assumed to be log-normally distributed. Expected catch rates are related to mid-year biomass (average of start and end values of biomass in year t). Observed and predicted catch rates are log-transformed to normalise the residual errors. Estimates of the model parameters are obtained by maximizing the appropriate likelihood function:

$$L(data \mid B_0, r, K, q) = \frac{1}{\sqrt{2\pi}\hat{\sigma}}\prod_t e^{\frac{-(LnI_t - Ln\hat{I}_t)^2}{2\hat{\sigma}^2}} \tag{6}$$

where $L(data \mid B_0,r,K,q)$ is the likelihood of the data given the parameters, the product is over all years (t) for which CPUE data are available and where:

$$\hat{\sigma}^2 = \sum_t \frac{(LnI_t - Ln\hat{I}_t)^2}{n} \tag{7}$$

An estimate of q, which maximises equation 7 is given by the geometric mean of the time series of individual q estimates:

$$\hat{q} = e^{\frac{1}{n}\sum Ln(\frac{I_t}{\hat{B}_t})} \tag{8}$$

Equation 6 can be converted to a log-likelihood and greatly simplified (Haddon, 2001):

$$LL = -\frac{n}{2}(Ln(2\pi) + 2Ln(\hat{\sigma}) + 1) \tag{9}$$

where LL is log-likelihood, n is the number of observed catch rates, and σ is defined in equation 7.

In order to assess the precision of parameter estimates and reference points, confidence intervals were calculated by using bootstrap (Haddon, 2001). Bootstrapping was carried out as follows: (i) Generate a new set of CPUE data by randomly sampling and replacing residuals from the original fitted model. Assuming a log-normal error distribution, residuals are calculated as the ratio of CPUE values (observed/expected) and multiplied with the expected CPUE values to get the bootstrap sample; (ii) Randomly draw a value from the log-normal pdf of r; (iii) Re-fit the model, obtaining new estimates of B_0, K, and q; (iv) Calculate new management reference points (ex. Y_{MSY}, f_{MSY}); (v) Repeat procedure 2000 times; (vi)

Calculate 95% bias-corrected confidence intervals for parameters and reference points based on the percentiles of the bootstrap distributions.

Additionally, 15-year biomass projections were estimated under different TAC (Total Allowable Catch) settings; 2000t, 3000t, 4000t, and 5000t. The procedure was similar to the bootstrap procedure except that biomass was projected forward 15 years, using equation 1 and assuming constant catches. The difference here is that various performance indices are calculated for each sample (1000). Risk is defined as the probability of an undesirable event (*e.g.* stock collapse), which was calculated as the sum of such events divided by 1000 for each management scenario.

Bayesian modelling

Bayesian surplus production (BSP) modelling followed McAllister and Babcock (2004). The BSP model uses a Bayesian parameter estimation method, in which the joint probability distribution of the parameters given the data (called the posterior distribution) is calculated from the likelihood of the data and the prior probability distribution of the data using Bayes theorem (McAllister *et al.*, 1994; McAllister and Ianelli, 1997; McAllister and Kirkwood, 1998; McAllister *et al.* 2001):

$$(10) \qquad P(\theta_i \mid y) = \frac{L(y \mid \theta_i) p(\theta_i)}{\int L(y \mid \theta_i) p(\theta_i) d\theta}$$

where $P(\theta_i \mid y)$ is the posterior distribution of the parameter vector θ_i given the data y, $p(\theta_i)$ is the prior distribution of the parameters and $L(y \mid \theta_i)$ is the likelihood of data given the parameters. When data contain little information, the posterior pdf tends to reflect the prior pdf. However, as data become more informative, $L(y \mid \theta_i)$ predominates and priors lose their influence over the posterior (McAllister *et al.*, 1994).

The fitting process in the BSP model is in fact the same as described in the previous section, which refers to likelihood in both cases. The approach differs completely thereafter with the application of the Bayes theorem (Equation 10) in contrast to bootstrapping.

The BSP model fits the Schaefer or Fletcher/Schaefer model to one or more time series of CPUE data. The CPUE data are assumed to be log-normally distributed about the model predicted values, so that the log likelihood function of the CPUE data is:

$$(11) \qquad \ln(L(y \mid \theta_i)) = -\sum_{j=1}^{nind} \sum_{t=iyr}^{icur} J_t \frac{0.5}{\sigma_j^2} \left[\log\left(\frac{I_{j,t}}{q_j B_t} \right) \right]^2 - \log(\sigma_j) - 0.5\log(2\pi) - \log(I_{j,t})$$

where $I_{j,t}$ is the CPUE value for index j in year t, q_j is the constant of proportionality or catchability coefficient for CPUE series j and σ_j^2 is the variance in the lognormal likelihood function of data series j, $J_t = 1$ if $I_{j,t} > 0$, otherwise $J_t = 0$. The variance σ_j^2 can vary depending on the weighting method selected. In the present study, production modelling involves only one CPUE time series and equation 11 can be simplified to equation 9.

The BSP program approximates the posterior distribution using a numerical integration method called the sampling-importance resampling (SIR) algorithm. The SIR algorithm is based on the idea that the posterior distribution of a parameter of interest $g(\theta)$ can by reformulated so that it is defined in terms of its expected value ($E^{h(\theta)}$) with respect to an arbitrary probability density function $h(\theta)$, called the importance function (McAllister *et al.*, 1994). The expected value can then be approximated by taking a large number (m) of draws from the importance function, so that the marginal posterior of $g(\theta)$ becomes:

$$(12) \qquad E^{P(\theta|y)}\left(g(\theta)\right) \cong \frac{\sum_{k=1}^{m} g(\theta_k) w(\theta_k)}{\sum_{k=1}^{m} w(\theta_k)}$$

where:

$$(13) \qquad w(\theta_k) = \frac{L(y \mid \theta_k) p(\theta_k)}{h(\theta_k)}$$

is called the importance ratio or the weight of draw k.

In other words, the importance ratios form an approximation of the posterior distribution, placing on each θ_k the following mass:

$$(14) \qquad F(\theta_k \mid y) = \frac{w(\theta_k)}{\sum_{k=1}^{m} w(\theta_k)}$$

The marginal distributions can be calculated directly from the importance draws, or a sub-sample can be taken to facilitate computation (the re-sampling part of the SIR).

For a sufficiently large number of draws equation 12 converges on the posterior distribution of $g(\theta)$ for any importance function $h(\theta)$, which allows a non-zero probability of sampling each point in the posterior distribution. In general, an importance function that closely approximates the posterior distribution will be more efficient, meaning that it will converge more rapidly to the posterior distribution. Also, an importance function that places too little weight in the tails of the posterior distribution will be inefficient in approximating the posterior distribution. Details on diagnostics of convergence are available from McAllister and Ianelli (1997) and McAllister and Babcock (2004).

BSP implementation

The pdf of *r* was defined previously and used as the prior of *r*. The ratio B_0/K was estimated, assuming a prior with a mean of 1 and a standard deviation of 0.2. A non-informative prior was defined for *K* (uniform on log *K)*, following the recommendations by Punt and Hilborn (1997).

Risk analysis involved a biomass projection of 15 years, using equation 1, under different management scenarios (TACs of 2000t, 3000t, 4000t, 5000t and 6000t). This included the following steps: (i) Randomly draw values for parameters from the posterior probability distribution (*e.g. r* and *K* in the logistic model); (ii) Using the drawn parameter values, project from the current year the model into the future and apply the policy of interest in each future year to predict its consequences; (iii) Calculate the performance indices for each policy; (iv) Repeat steps (i) to (iii) 5000 times; (v) Produce a distribution of performance indices for the policy.

5.3 Results

A plot of catch against effort appears to show a classical example of a developing fishery, which has reached and overshot Y_{MSY} in recent years (Figure 5.4). CPUE in the most recent year (2001) has dropped below the control curve, which is the theoretical yield curve fitted under equilibrium assumptions (linear fitting process). The equilibrium Y_{MSY} was estimated at 4 146 tonnes, but fishery catches exceeded this value during the period 1997 to 2000 (Table 5.1). This control curve shows a relatively good fit to the data, but it was included merely for comparative purposes.

Three models were specified in the dynamic fitting process, where bootstrap techniques were used to estimate uncertainty around model parameters (Table 5.2). These were:

- Model 1: all parameters were estimated, including r. The constraint $K/B_0 \geq 1$ was imposed, which states that the biomass (B_0) level at the beginning of the time series cannot exceed carrying capacity (K). This constraint was necessary in order to obtain reasonable results in biological terms. Another constraint assumed positive values for all parameters.
- Model 2: r is assumed known and described through its pdf (Figure 5.3). The constraint $K/B_0 \geq 1$ was imposed. No further constraints were necessary as r was assumed to follow a lognormal distribution (non-negative values).
- Model 3: r is assumed known and described through its pdf. No further constraints were applied.

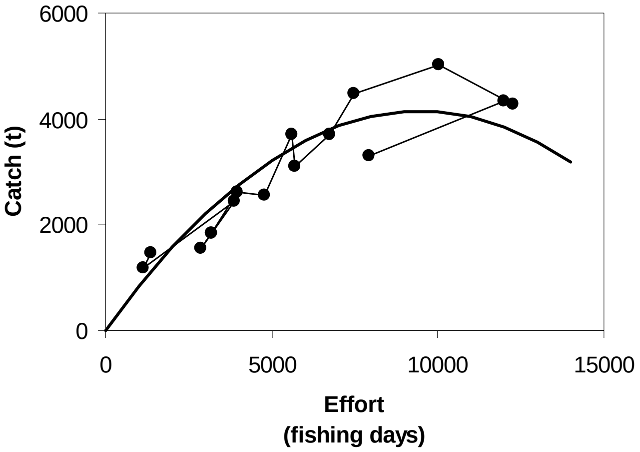

Figure 5.4. Plot of catch against standardised effort. The theoretical yield curve was also included, fitted under equilibrium assumptions.

Table 5.2. Parameter and management reference estimates of the three models specified in the dynamic fitting process (see text for further details). Bias-corrected 95% confidence intervals and bias (%) were estimated using bootstrap techniques (2000 samples). Units for B, K, and Y are tonnes and r is per year.

		r	K	B_0	q	K/B_0	B_{2001}	B_{2001}/K	Y_{MSY}
Model 1	Mean	1.026	53374	31882	0.0548	1.64	15642	0.44	3089
($K/B_0 \geq 1$)	Median	0.475	27352	25316	0.0292	1.00	12279	0.45	3432
	U95%	3.930	339177	92326	0.1985	5.97	49152	0.62	4811
	L95%	0.000	4608	3000	0.0079	1.00	1912	0.06	0
	Bias%	90.7	114.7	28.3	74.3	63.8	37.0	-4.2	-7.6
Model 2	Mean	0.771	20754	20749	0.0445	1.00	9519	0.45	4002
($K/ B_0 \geq 1$)	Median	0.731	20330	20322	0.0377	1.00	10276	0.50	3767
	U95%	1.355	22764	22768	0.1341	1.00	17783	0.81	7109
	L95%	0.353	15746	15746	0.0284	1.00	100	0.00	1673
	Bias%	6.9	3.8	3.8	13.4	0.0	3.6	-2.0	11.1
Model 3	Mean	0.774	20876	20760	0.0436	1.01	9662	0.46	4048
	Median	0.719	20501	20342	0.0374	1.00	10371	0.50	3734
	U95%	1.514	21749	21920	0.1409	1.18	17590	0.81	7425
	L95%	0.355	14703	15595	0.0293	0.85	100	0.00	1612
	Bias%	7.4	4.4	3.9	11.0	0.9	5.1	-0.7	12.3

The parameter estimates of Model 1 are highly uncertain as shown by the broad confidence intervals and strong bias (Table 5.2). Median parameter estimates appear to be reasonable, but associated with great uncertainty. Biomass estimates in 2001 ranged between 2 000 and 50 000 tonnes (95% CI). It is interesting to note that reference points for management such as Y_{MSY}, B_{2001}, and K/B_0 are relatively well defined in spite of large uncertainty around model parameters. The median estimate of Y_{MSY} was 3 432 tonnes and a ratio B_{2001}/K of 0.45, which indicates that exploitation was at or close to maximum sustainable levels. We generally refer to median estimate values as these present less bias when referring to skewed distributions.

The results for Model 2 and 3 were radically different, compared to Model 1, in that parameter values were more precisely defined. The results for the parameter r followed closely the defined pdf (lognormal median = 0.721, lognormal mean = 0.767), which improved considerably the precision of the other model parameters such as K and B_0 as well as the scaling parameter q (catchability). Note that the estimate of Y_{MSY} at about 3 760 tonnes was slightly more optimistic than in Model 1, while a B_{2001}/K of 0.50 indicates that exploitation level should be reduced to avoid biomass ratios below 0.5 ($B_{MSY} = K/2$). This would be the precautionary course of action. The improvement in the precision of reference points such as Y_{MSY}, B_{2001}, and K/B_0 did not improve much in relation to Model 1. However, Model 1 gave a lower 95% confidence limit of 0 for Y_{MSY}, which would imply the necessity to make very conservative decisions, if this model was used as the basis for management advice. The precision of biomass estimates improved considerably using Models 2 and 3, but there was a problem of growing uncertainty over time (Figure 5.5).

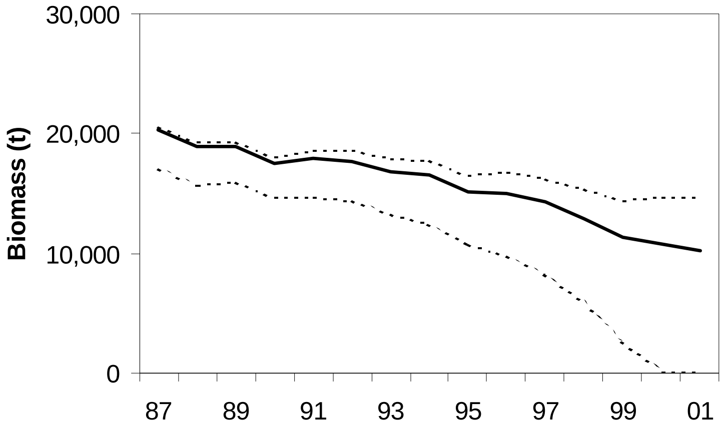

Figure 5.5. Estimated biomass levels (Model 3) for the time period 1987 to 2001, showing median values and 80% confidence intervals.

The results of Model 2 and 3 are practically identical, whether or not applying the constraint $K/B_0 \geq 1$, which indicates that "bulk" biomass of small pelagics appears to have been close to virgin levels (or carrying capacity) at the beginning of the series in 1987. Model 3 (no constraints) was used as the basis for the following comparison with the Bayesian modelling approach.

Comparison with the Bayesian modelling approach

The results of the Bayesian surplus production model gave more optimistic results (Table 5.3) compared to the bootstrap model (Model 3). Parameter and reference point estimates are presented as means in both models for the purpose of comparison, considering that the posterior mean is the Bayes estimator of the parameter. The median would be a better estimator for the bootstrap model, but the differences between mean and median are generally small in this case (Table 5.2). Parameter estimates were generally higher in the

Bayesian model, for K and B_0 in particular, which led to higher biomass and Y_{MSY} estimates. B_{2001} in the Bayesian model was estimated at almost double the level in the bootstrap model, which together with Y_{MSY} appears to show a more sustainable pattern of exploitation. The uncertainty associated with the estimates, expressed as the coefficient of variation, was roughly similar except for the notable cases of K and B_0. Note that these two parameters are estimated much more precisely in the bootstrap model. Both models estimated that B_0 was close to K at the beginning of the study period, although more so in the bootstrap model.

Table 5.3. Parameter and management reference estimates of the bootstrap (Model 3) and Bayesian models (marginal posterior distributions). The quantities given are mean, standard deviation (SD), and coefficient of variation (CV). Units for B, K, and Y are tonnes and r is per year.

		r	K	B_0	B_{2001}	B_{2001}/K	Y_{MSY}
Bootstrap	Mean	0.774	20876	20760	9662	0.456	4048
	SD	0.294	1909	1940	5694	0.256	1624
	CV	0.379	0.091	0.093	0.589	0.562	0.401
Bayesian	Mean	0.845	25624	23766	17211	0.638	5028
	SD	0.309	9117	9058	9602	0.194	1841
	CV	0.366	0.356	0.381	0.558	0.304	0.366

The Bayesian and bootstrap models were similar in their parameter estimation. For example, the best point estimates of the parameters at the mode of the posterior distribution in the Bayesian model were:

- $r = 0.699$
- $K = 19\,636$
- $B_0/K = 0.96$
- $q = 0.04242$

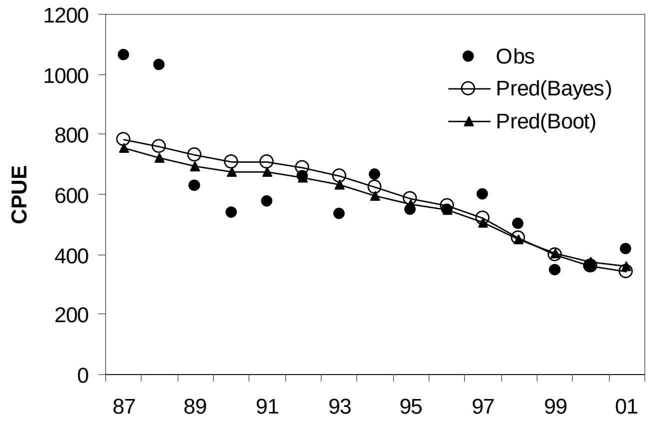

Figure 5.6. Observed and predicted CPUE values (kg per fishing day) using the dynamic fitting process in the Bayesian and bootstrap models.

A comparison with Table 5.3 shows that these parameter estimates were similar to the results obtained in the bootstrap models. Figure 5.6 shows that predicted CPUE values were in fact similar in parameter estimation between the two approaches. The minor differences may be attributed to differences in the technical details of iteration and search algorithms. Parameter estimates of r were also similar, although there was a widening of the r posterior in relation to the r prior (Figure 5.7).

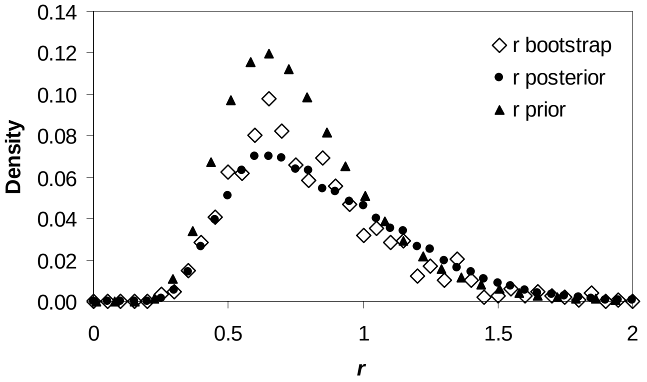

Figure 5.7. The marginal posterior distribution and prior of r from the Bayesian model and the bootstrap sample of r.

On the other hand, the Bayesian and bootstrap models gave very different results, when considering K (Figure 5.8). The bootstrap estimate of K was considerably more precise than in the Bayesian model. These differences in modelling approach carried over to the estimation of biomass and Y_{MSY}, resulting in higher estimates in the Bayesian model (Figure 5.9 and 5.10).

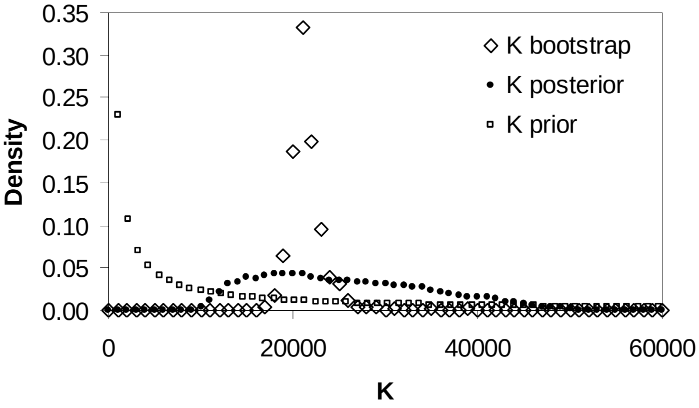

Figure 5.8. The marginal posterior and prior of K from the Bayesian model and the K estimate from the bootstrap model.

Three performance measures were chosen for the purpose of decision analysis using the two modelling approaches, Bayesian and frequentist (Table 5.4). $B_{Fin}/K < 0.2$ is the probability that biomass at the end of the period (projections of 5, 10 or 15 year periods) is below 20 % of K or, in other words, has collapsed; $B_{Fin} > B_{MSY}$ is the probability that biomass at the end of the period is above estimated biomass at Y_{MSY} or, in other words, that biomass is at healthy levels; $B_{Fin} > B_{2001}$ is the probability that biomass at the end of the period is greater than biomass in 2001.

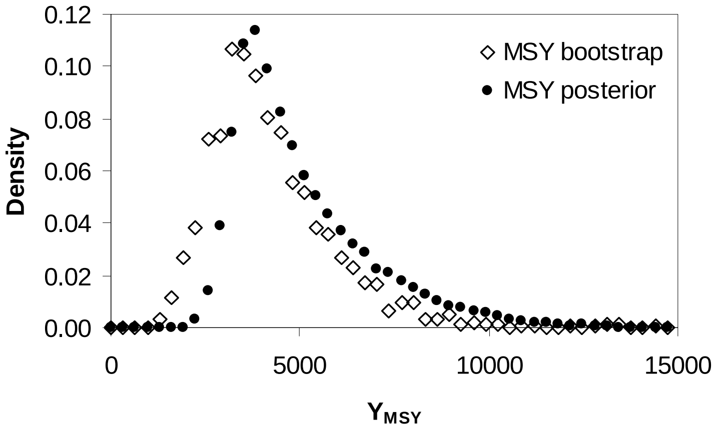

Figure 5.9. The marginal posterior distribution of Y_{MSY} estimated by the Bayesian and the Y_{MSY} estimates from the bootstrap model.

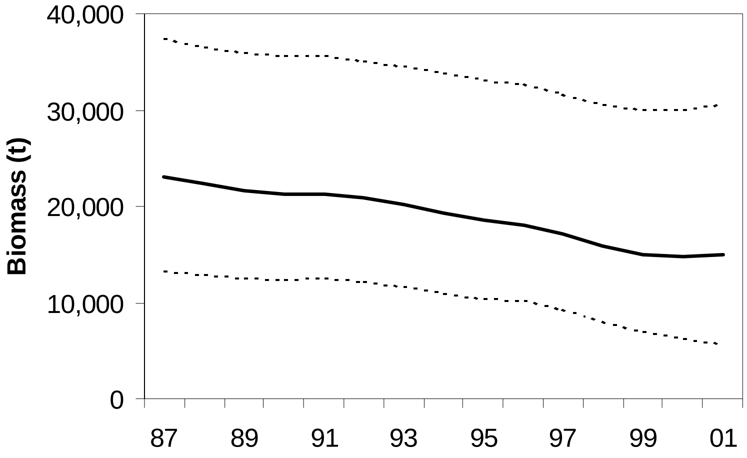

Figure 5.10. Estimated biomass levels using the Bayesian model, showing median values and 80% confidence intervals.

Decision analysis will depend on management objectives, but the Bayesian model will invariably lead to more optimistic decisions due to the higher biomass and Y_{MSY} estimates. If the objective is simply to avoid the collapse of the fishery, a TAC of 5 000 or 6 000 tonnes may be chosen, using the frequentist and Bayesian models, respectively. If the objective is to keep biomass above B_{MSY}, then the frequentist model indicates that this can be achieved by setting a TAC between 3 000 and 4 000 tonnes (using 50 % probability). In the Bayesian model, this can be achieved by setting a higher TAC of 5 000 tonnes. Thus, the frequentist model gave generally more conservative recommendations.

Table 5.4. Consequences of alternative TAC policies for the stocks of small pelagics in Cape Verde, projected from 2001 to 2016 using the frequentist and Bayesian model. Three performance measures are compared for these two modelling approaches: $B_{Fin}/K<0.2$ is the probability that biomass at the end of the period is below 20 % of K; $B_{Fin}>B_{MSY}$ is the probability that biomass at the end of the period is above estimated biomass at MSY; $Bfin>B_{2001}$ is the probability that biomass at the end of the period is greater than biomass in 2001.

Horizon	Model	TAC	$B_{Fin}/K<0.2$	$B_{Fin}>B_{MSY}$	$Bfin>B_{2001}$
5 -year	Frequentist	2000	0.26	0.68	0.73
		3000	0.30	0.59	0.64
		4000	0.37	0.48	0.47
		5000	0.47	0.35	0.00
		6000	0.60	0.20	0.00
	Bayesian	2000	0.02	0.94	0.97
		3000	0.08	0.85	0.88
		4000	0.17	0.70	0.05
		5000	0.31	0.52	0.00
		6000	0.46	0.36	0.00
10 -year	Frequentist	2000	0.26	0.73	0.74
		3000	0.33	0.62	0.65
		4000	0.45	0.47	0.44
		5000	0.63	0.27	0.00
		6000	0.80	0.13	0.00
	Bayesian	2000	0.03	0.96	0.97
		3000	0.10	0.87	0.88
		4000	0.24	0.68	0.05
		5000	0.44	0.45	0.00
		6000	0.61	0.28	0.00
15 -year	Frequentist	2000	0.26	0.74	0.74
		3000	0.35	0.64	0.65
		4000	0.49	0.46	0.42
		5000	0.69	0.25	0.00
		6000	0.85	0.11	0.00
	Bayesian	2000	0.03	0.97	0.97
		3000	0.10	0.88	0.88
		4000	0.27	0.67	0.05
		5000	0.50	0.42	0.00
		6000	0.68	0.26	0.00

5.4 Discussion

Previous assessments have been undertaken for the mackerel scad stock specifically (Carvalho, 1993; Almada, 1997; Carvalho and Caramelo, 1999; Jardim, 1999). These are not directly comparable with the present study, but a review of these results is nevertheless elucidative. Table 5.5 shows the various methodologies applied, most of them commonly used in tropical fisheries at the time (Sparre and Venema, 1992). None of these studies deal with uncertainty in parameter or reference point estimates, although the study by Almada (1997) included risk assessment by projecting biomass forward with stochastic variability. It is important to note that all of these studies included data available for the period 1981 to 1986, which was excluded from this study as these data have been found to be unreliable (see Chapter 2). As a rule of thumb to assist in the comparison of results, mackerel scad constitutes about 70 percent of total small pelagic catches as defined in the introduction. If we assume that catch composition reflects relative abundance in terms of species, the Y_{MSY} of mackerel scad is around 70 percent of the Y_{MSY} estimates given in Table 5.2 and 5.3 (bootstrap ≈ 2800 t; Bayesian ≈ 3500 t). Of course, the data indicates that this may not be the case, at least that relative abundance changes over time, but the rule of thumb simplifies for the sake of discussing previous assessment results.

Table 5.5. Results of previous assessments on mackerel scad, *Decapterus macarellus*, in Cape Verde.

Reference	Method	Time Period	Parameters and Reference points
Carvalho, 1993	Z equations (Powell-Wetherall)	1983 – 1989	Z/K = 4.63 Z = 2.770 F = 2.021
Carvalho and Caramelo, 1999	Jones Length-based Analysis	1981 – 1994	$F=F_{MAX}$
Jardim, 1999	Z equations (Beverton and Holt)	1981 –1991	Z/K = 4.79 Z = 1.17
	Jones Length-based Analysis	same as above	Y = 1555; F = 0.28 $Y_{MSY} = 1868$ $B_{MSY} = 3084$
	Fox production model	(industrial data)	r = 2.39; K = 6085 $Y_{MSY} = 5358$ $B_{MSY} = 2239$
	Fox production model	(artisanal data)	r = 1.97; K = 2125 $Y_{MSY} = 1548$ $B_{MSY} = 782$
Almada, 1997	Yield per recruit & stock recruitment relation	1983 – 1995	$Y_{MSY} = 5500$ $B_{MSY} = 24000$

The results of Carvalho (1993) show a very high exploitation, which was considered to be above sustainable levels. However, this study relied heavily on data that is now considered unreliable, from 1981 to 1983 in particular. The extremely high catch estimates in the beginning of the period may have resulted, when analysing the length frequency data, in what seemed like heavy exploitation.

Jones length-based cohort analysis (Sparre and Venema, 1992) was applied on two occasions (Table 5.5). This method assumes equilibrium conditions and its implementation involves the averaging of length frequency data (by length class) over a period of several years. The application of such a method that assumes equilibrium conditions would be criticised by the scientific community nowadays, but we have seen in this specific case that the stocks do not appear to be far from equilibrium condition in Cape Verde. In fact, the small pelagics fishery was characterised by moderately increasing effort in 80s and early 90s, supporting the notion that stocks may be near the equilibrium state. If the results of these equilibrium methods are used as a control curve (Figure 5.4) or as a limit reference point, they can be useful in data-poor situations, bearing in mind that they may be highly uncertain and due considerations should be given on the evolution of the fishery (*e.g.* Caddy, 1996, 2002).

Using the Jones length-based method, Carvalho and Caramelo (1999) found that fishing was at maximum sustainable level, while Jardim (1999) showed that there was some room for expanding the fishery. The main difference between these two studies was the time period in question. Scad catches in 1994 were 2 900 tonnes, which in hindsight supports the findings of Carvalho and Caramelo (1999). On the other hand, catches in 1991 were 1 260 tonnes, which is also consistent with the findings of Jardim (1999) that there was room for increasing catches. Note however that the reference points for management given by Jardim (1999) appear to be unreliable.

Jardim (1999) also applied the Fox production model, which is the exponential formulation of the surplus production model. Estimates of Y_{MSY} were not unreasonable if the associated uncertainty is considered, but the parameters r and K of the Fox model appear to be unreasonable. This application suffered from methodological weaknesses such as not standardising the abundance index (CPUE) and not considering the shift towards targeting mackerel scad in the industrial fishery. Note that Y_{MSY} is greater than B_{MSY} in both attempts with the Fox model, which indicates that a mathematical solution was reached, but it does not make much biological sense (Table 5.5). The evolution of industrial catches of small pelagics in Figure 5.1 shows the underlying pattern of increasing and then decreasing targeting of mackerel scad, which is confounded with relative abundance. This was the reason for assessing small pelagics as “bulk biomass” in the present study in an attempt to minimise this problem (catchability q cannot be assumed constant). In these circumstances, we assume that environmental change is not driving the CPUE, a reasonable assumption for the time period, and that q is constant. However, the increasing motorisation of the artisanal fishery is expected to have had an effect on q, although rather small, but it was not possible to estimate the increment in catchability due to the non-informative nature of the data.

The approach used by Almada (1997) included a much more thorough review and analysis of data and the use of methods that are not too sensitive to the quality of effort data available (*e.g.* length frequency data). Due consideration was given to the shift in target species of the industrial fishery. The stock recruitment relationship used by Almada (1997) was highly uncertain and fitted using a certain dose of scientific judgement, which implies that the estimates were also highly uncertain. It is nevertheless comforting that the results from this structurally different model are comparable to this study in terms of Y_{MSY} and biomass, when considering the uncertainty involved. The Beverton and Holt yield per recruit model assumes equilibrium conditions, but the coupling of a stock recruitment relationship to the model by Almada (1997) deals with the major limitation of the yield per recruit model.

It is evident from this study that considerable care should be taken in assessing the quality of data available and that knowledge of the fishery is essential. Structurally different models should in principle give the same answers, given that the requisites are not violated. Surplus production models have been heavily criticised, most notably by Larkin (1977 in Mace, 2001), but a revival has occurred under the new name of biomass dynamic models. The application of the surplus production model on non-informative data was the main reason for these failures (Hilborn and Walters, 1992). This was also the case in Cape Verde, where an unsuccessful attempt was undertaken in the past (the Fox model). The data continues to be largely the same in the present study, a typical "one-way trip", but with the notable difference that we expect the value of r to be within a definite range (prior knowledge). This r parameter was estimated using an empirical equation and assuming a variation based on other studies, which gave reasonable estimates, considering that these are small, fast-growing species in tropical waters. The approach used was an over-simplification, but this built on other studies and more data intensive methods such as the demographic or meta-analytical approaches can easily be implemented in a formal assessment setting. We consider an in-depth analysis of available non-aggregated data a more important issue in order to study possible changes in catchability (q), feasibility of modelling on a species basis, and better standardisation procedures for CPUE. Nevertheless, the estimates of Y_{MSY} were relatively precise and the implications of various TAC policies were presented, using the bootstrap and Bayesian approach. The latest acoustic survey was carried out in 1997 and gave an estimate of about 15 thousand tonnes of small pelagics, mostly *Decapterus macarellus* and *Decapterus punctatus* (Marques *et al.*, 1997). Although inconclusive, this appears to give more support to the findings of the bootstrap approach (Figure 5.5), while Bayesian biomass estimates may be biased (Figure 5.10). Regarding the other acoustic survey undertaken in Cape Verde in 1981, it appears most likely that the estimate of 50 to 65 thousand tonnes of small pelagics biomass was exaggerated and inaccurate and may have caused considerable harm in subsequent fisheries development plans.

To be Bayesian or to bootstrap?

In order to explain the results of the bootstrap and Bayesian approaches, it is important to mention the differences in concept. Bootstrapping attempts to characterise uncertainty by relating alternative data sets (generated on the basis of model residuals) and the optimal model fit. Bayesian analyses characterise uncertainty by investigating how the quality of fit to the sample data alters as the parameter set (hypothesis) selected is altered. Bootstrapping approaches the problem by modifying the data and determining the implications, while Bayesian analysis modifies the parameter sets (hypotheses) and determines the implications (Haddon, 2001). It is important to note that bootstrapping involves calculating likelihood of data given the hypothesis, but not the probability of the hypothesis given the data. To calculate the probability of the hypothesis ones needs to apply Bayes theorem, which is mathematically proven and not contested by anyone (Punt and Hilborn, 2001). This difference has important theoretical consequences in the application of decision theory, which requires probabilities. The sampling distributions of estimates generated by bootstrapping do not represent probabilities, but they have often been applied and interpreted as if they were probability distributions for alternative hypotheses (such as in this study for the sake of comparison). Nonetheless, the use of bootstrap distributions in the evaluation of alternative harvest policy has provoked controversy (McAllister and Kirkwood, 1998; Punt and Hilborn, 2001).

This study differs from common bootstrapping procedures in that r is assumed known and defined by its pdf. An extra step was introduced, which was to randomly sample from the pdf of r. The steps involved were to generate a new bootstrap sample, draw from the pdf of r, fit the model, and repeat these many times. Consequently, the estimate of parameter K was relatively precise (Table 5.3; Figure 5.8), because of the constraint imposed on r. Normal bootstrapping procedure, based on conventional frequentist statistical theory, would assume that nothing is known about r and let the data give an estimate of this, which would result in highly uncertain results (Model 1 in Table 5.2) because of the non-informative data. However, there are many examples of assessments that fix some parameters (such as natural mortality) for the analysis, which is against conventional theory. In a Bayesian context, this implies that the parameter is known exactly (a very informative prior!), which is far from the case. Thus, we consider our approach more sensible in spite of it being a blend of conventional frequentist and Bayesian school of thought. Yet another point is an advantage of the bootstrapping approach, with its emphasis on the data, if the parameter estimation is in any way biased then bootstrapping can provide an estimate of that bias, but this is not the case with the Bayesian approach (Haddon, 2001).

The Bayesian approach determines the relative quality of fit produced by different combinations of parameter estimates (hypotheses) along with the prior probabilities of each parameter set. This relative fit is described by the posterior probability distribution for the parameter set and it does not involve fitting the model after each iteration, which is why the K parameter continues to be poorly defined in the present study (Figure 5.8). The relatively imprecise estimate of K is the result of the non-informative data and the lack of a model-fitting step as in the bootstrap procedure defined above. This imprecision carries over in the estimation of Y_{MSY} and biomass, leading to the differences seen in Table 5.3.

There is much controversy on the issue of priors, which are considered subjective by frequentists, thus seriously compromising the results. If the data is poor and non-informative, the posterior distribution will follow closely the prior distribution, which emphasises that care should be taken in defining priors. This was the case in the present study, where the posterior of r followed closely the prior (Figure 5.7), thus the danger of defining overly precise estimates that may be highly biased. We increased the coefficient of variability from 46 to 61 percent of the pdf of r to study its effect, which were minor, indicating that the Bayesian estimators were not sensitive to this and that the prior was properly defined. A non-informative prior was chosen for K, because this is highly uncertain and acoustic surveys have given conflicting results (Stroemme *et al.*, 1982; Marques *et al.*, 1997). An informative prior for B_0/K was based on survey reports, indicating that stocks were at virgin levels in the beginning of the 1980s (Stroemme *et al.*, 1982; Magnússon and Magnússon, 1985). Much has been written about informative and non-informative priors as well as procedures in their definition, which should be consulted for further detail (*e.g.* McAllister *et al.*, 1994, 2000, 2001; McAllister and Ianelli, 1997; Punt and Hilborn, 1997, 2001; McAllister and Kirkwood 1998). The crucial issue is related to the fact that so-called non-informative priors appear to be very difficult or impossible to generate, which would require them to be non-informative over all parameters and model outputs. However, this problem becomes relevant when considering complex models and high dimensional parameter space, but not the simple case of the two-parameter Schaefer model as in this case. An interesting point put forward by Haddon (2001) is that truly non-informative priors are the same as not using priors at all or to omit them from analysis altogether, but this would go against Bayesian thinking.

One of main weaknesses in the management process has been termed the ratchet effect (Botsford *et al.*, 1997). When under political pressure, managers will allow harvests to increase for their short-term benefits to society (jobs and profits) when fishery scientists cannot specify with certainty that the next increase will lead to over-fishing and collapse. Until recently, policy advice from stock assessments has typically been based on point estimates of management reference points, not explicitly accounting for uncertainty. When facing conflicting interests such as high biological yields and high rates of employment and profits, it appears that these reference points were often not taken seriously. Thus, much emphasis has been placed on uncertainty and risk analysis in recent developments of stock assessment methods (Punt and Hilborn, 1997; Sainsbury, 1998).

The Bayesian statistical approach offers an elegant and theoretically consistent framework within which to provide policy advice (McAllister and Kirkwood, 1998). For the relatively simple case of the two-parameter Schaefer model, we developed a frequentist approach that considers what would be called prior knowledge (pdf of r) in a Bayesian context. This proved to be a more efficient method in terms of estimating the parameter K and it resulted in a more conservative estimate of Y_{MSY}, thus giving further support to the notion that frequentist approaches are more efficient parameter estimation methods. However, taking the next step of calculating probabilities for the implications of various management options is moving into fragile theoretical ground with the bootstrap approach. We are assuming that the bootstrap sampling distributions of the various estimates, when projecting forward, can be interpreted as probability distributions (Haddon, 2001). A comparison of the bootstrap and Bayesian "probabilities" in Table 5.4 show discrepancies, which support the view that bootstrap methods cannot be used for decision analysis, whether in theory or in practical applications. The difference in the Y_{MSY} estimate cannot explain these discrepancies as the results indicate distinct distributions between Bayesian and bootstrap methods. Note that the time horizon does not imply much change in the Bayesian approach, but more so in the bootstrap approach. This was again related to the emphasis on data in bootstrapping, which includes more variability in forward projections. Thus, the advantages of Bayesian statistics for the purpose of providing policy advice appear to be evident, but it also provides a theoretical framework that permits the consideration of structurally different models as alternative hypotheses. These advantages become increasingly obvious when dealing with increasingly complex models as well as structurally different models for the purpose of providing quantitative management advice.

Chapter 6. Quantitative Ecosystem Indicators: applications in Cape Verde

6.1 Introduction

Quantitative ecosystem indicators have been proposed in recent years to determine the effects of fishing on marine ecosystems from a global or broad perspective (Gislason and Rice, 1998; Jennings and Kaiser, 1998; Pauly *et al.,* 1998-a; Bianchi *et al.,* 2000; Gislason *et al.,* 2000; Rice, 2000; Jennings *et al.,* 2002; Link *et al.,* 2002; Rochet and Trenkel, 2003; Trenkel and Rochet, 2003; Nicholson and Jennings, 2004). Considerable efforts have gone into the development and evaluation of these indicators as part of the process of incorporating a broader ecosystem view of the impacts of exploitation; the ecosystem approach to fisheries (FAO, 2002-c, 2003; Garcia *et al.*, 2003; Cury and Christensen, 2004). These new complementary management tools may prove useful for improving on the current state of fisheries resources and their application would be particularly convenient in data-limited situations typical of many tropical fisheries.

The Cape Verde ecosystem has strong links to the open oceanic ecosystem of the central Atlantic and is characterised by relatively low productivity (Longhurst, 1998) and very limited shelves around the islands. Trawl surveys have been undertaken sporadically and therefore, a consistent time series of survey data was not available. On the other hand, Cape Verde has a reasonably well-functioning statistical collection system since 1986. Catch and effort have increased since 1986 by a factor of about two and the total estimated catches are around 10 thousand tonnes at the present (INDP, 2003). Fisheries are dominated by pelagic neritic or migratory species including carangids and tunas. Artisanal fisheries account for a major proportion of the catches and industrial fisheries are characterised by relatively small vessels primarily exploiting pelagic resources (see Chapter 2 for more details). Industrial trawling is prohibited in Cape Verde.

Considering the limitations in terms of data, the application of ecosystem indicators may prove useful for management purposes in Cape Verde. Thus, various indicators and techniques were used to study their usefulness in determining the effects of fishing in Cape Verde, following up on previous efforts (Stobberup *et al.*, 2002, 2004, 2005, in press; Christensen *et al.*, 2004; Thiam *et al.*, 2004). In the present study, this included indicators such as the abundance of commercial and non-commercial demersal fish, species composition and abundance over time as well as time series analysis of catch data.

6.2 Materials and Methods

Data from a number of trawl surveys undertaken in Cape Verde were available for the present study (Table 6.1). The surveys in 1964 and 1976 were exploratory in nature with sampling of a few stations around and south of Boavista Island. Later surveys were more systematic in covering the Boavista-Maio shelf, but sampling of the northwestern islands (S. Vicente and S. Antão) and southern islands (Santiago, Fogo, Brava) was occasional (see Chapter 2 for further details; Figure 2.9).

As the available data was limited, the results of commercial fishing by the trawler vessel "Islândia" were included (available for 1994 to 1997). This vessel belongs to the Fisheries Institute in Cape Verde (INDP) and was undertaking prospective fishing in the Boavista and S.Vicente area. The vessel and gear were the same as in the 1994 scientific survey (Table 6.1), thus making it comparable.

Trawl survey catches were standardised (kg/nm^2) using the swept-area method taking into consideration haul duration, average trawling speed, and a fixed value for the horizontal net opening. This standardisation was not possible on a haul-by-haul basis, as data on speed was often lacking and the net opening was not measured (see Chapter 2; Table 2.2).

Table 6.1. Trawl surveys considered in the study, including the number of haul stations, depth coverage, year and month for each survey. Data selection and omission followed several criteria, which resulted in a specific number of stations selected for analysis (see text for further explanation).

Year	Survey Context	Stations Total No.	Stations Included No.	Reference
1964	German (FDR) Cooperation	6	6	Massuti, 1965
1976	German (DDR) Cooperation	16	16	Danke and Koch, 1987
1981	Fridtjof Nansen Programme	18	15	Stroemme *et al*., 1981
1985	Icelandic Cooperation	37	19	Magnússon and Magnússon, 1985
1988	Icelandic Cooperation	80	79	Palsson, 1989
1994	Cape Verde / Iceland	60	55	Thorsteinsson *et al*., 1994
1994 -1997	Commercial fishing	163	163	INDP

A revision of the identified fish species was undertaken using FishBase 2000 (Froese and Pauly, 2000), which attempted to detect problems such as species misidentifications, inconsistencies or contradictions in the observations. Habitat was designated for all fish including elasmobranches, using the following categories; pelagic, demersal, benthopelagic, reef-associated, bathydemersal, and bathypelagic. For the subsequent data analysis, pelagic and bathypelagic species were omitted, since trawl gears are not considered suitable for sampling pelagic species. Standardised catches by species were aggregated by family in order to avoid problems due to species misidentification.

Trawl stations were grouped according to year, depth (≤ 30m; 30 - 80m; 80-200m) and area. These depth strata were chosen so as to divide into depths above the thermocline, depths including the thermocline, and depths below until 200m. All trawl stations with depths over 200 metres were omitted as these depths were sporadically sampled. The area was divided according to continental shelf; shelf around the northeastern island Sal, the northwest shelf system, and the more extensive shelf around Boavista and Maio Islands (eastern islands).

Multivariate analysis

The multivariate ordination method non-parametric Multi-Dimensional Scaling (MDS) was used to study changes over time and space in the species composition and abundance of demersal fish assemblages. This method (MDS) is generally applicable and no assumptions are made about the distribution of the data (Clarke and Warwick, 2001). Thus, when considering biological survey data, it is not necessary to deal with problems related to the skewed distributions of the data and all species can be included, which is important in studies concerning species assemblages.

It is important to point out that only research survey data were used in the MDS analysis. Commercial catches can be biased because of non-accounted catches (discards) and were therefore not included. Species data was aggregated to the family level so as to avoid problems of misidentifications, particularly concerning the 1976 survey. Thus, a total of 190 trawl stations (samples) and 60 families (variables) were considered. MDS was applied to a similarity matrix calculated with the Bray-Curtis coefficient on square-root transformed data. The Bray-Curtis similarity coefficient has the desirable property of not being sensitive to joint absences or double zeros (Clarke and Warwick, 2001), which is common in species occurrence or abundance data.

MDS is simple in concept, creating a sample map or ordination plot where the distances between points have the same rank order as the corresponding similarities (or dissimilarities) between samples (or haul stations in this case) (Clarke and Warwick, 2001). MDS plots can be arbitrarily scaled or rotated, while maintaining the relative distance between points in two or three dimensions. This is in contrast to a classical Principal Components Analysis (PCA) plot, where the PC axes explain a certain amount of variance based on linear combinations of species data (Euclidean distances). A measure of goodness-of-fit is termed the stress value in MDS:

$$Stress = \sqrt{\sum_j \sum_k \left(d_{jk} - \hat{d}_{jk} \right)^2 \Big/ \sum_j \sum_k d_{jk}^2}$$

where $\hat{d}_{jk}$ is the predicted distance between the jth and kth sample points and the fitted non-parametric regression line corresponding to dissimilarity δ_{jk}. Stress values larger than 0.3

indicate an almost random position of points; values lower than 0.1 correspond to a good ordination with clear interpretation; and values between 0.1 and 0.2 indicate a potentially useful ordination but interpretation should be handled with care (Clarke and Warwick, 2001). The PRIMER software was used for multivariate analysis (Clarke and Gorley, 2001).

The procedure Analysis of Similarities (ANOSIM) in PRIMER was used to test for the effect of pre-defined factors such as year, depth and area. ANOSIM is a non-parametric analogue to ANOVA and its multivariate extension MANOVA. It is built on a non-parametric permutation procedure applied to the rank similarity matrix underlying the ordination of samples and a general randomisation approach to the generation of significance levels (Monte Carlo tests) (Clarke and Warwick, 2001). The resulting R-values are a measure of variation between samples, ranging from -1 to 1. Values close to zero indicate that there is little difference between stations in terms of their species composition, while values close to unit demonstrate that the species compositions between groups is different.

Another procedure in Primer, BVSTEP, was used to compare two similarity matrices based on the rank correlation coefficient, using the Spearman coefficient (Clarke and Warwick, 2001):

$$\rho_s = 1 - \frac{6}{N\left(N^2 - 1\right)} \sum_{i=1}^{N} \left(r_i - s_i\right)^2$$

where r_i and s_i are the elements of the respective rank similarity matrices and n is the number of samples ($N = n(n-1)/2$). The Spearman rank correlation coefficient, ρ, lies in the range between –1 and 1, and can be used to assess the similarity of similarity matrices. It is however not possible to determine the significance (Pr: $\rho > 0$) as the ranks of r_i and s_i are not mutually independent variables. Instead, an arbitrary limit was chosen ($\rho = 0.95$) to indicate an almost complete agreement between two similarity indices.

Characteristic species assemblages were defined using "similarity percentages" (procedure SIMPER in Primer), which are the percentage contribution of each species to the similarity (typical species) and dissimilarity (discriminator species) between groups of samples (Clarke and Warwick, 2001). These calculations were based on square root transformed data and the pre-defined factors.

Generalized linear models

One possible indicator is mean abundance of demersal fish over time, which was modelled using generalized linear models (GLM) (McCullagh and Nelder, 1992; Dalgaard, 2002; Zuur, 2003). This is a common procedure to obtain standardised abundance indices in stock assessment, based on fisheries catch and effort data including fleet characteristics (*e.g.* Hilborn and Walters, 1992). The approach used here differs in that we are not standardising according to fleets, but according to other factors such as depth and area strata. Standardised catches (swept-area method) were used in an attempt to remove vessel effects. Furthermore, catches (kg/nm^2) were aggregated according to whether the species were of commercial value or no value. This is considered another possible indicator, where the biomass/abundance of commercial species are expected to decrease over time a result of fishing and increase in the case of non-commercial species (Trenkel and Rochet, 2003). Aggregated catches were log-transformed in order to obtain approximate normal distribution of model residuals and homogenous variance. The model fitted had the following general form:

1) log(std. catch) = a(intercept) + c(year) + c(season) + c(depth) + c(area) + c(group) + c(c.value) + c(type)

where a is the intercept and c refers to coefficients of the factor terms. The term "group" refers to species groups fish and elasmobranches; "c.value" refers to commercial or non-commercial species; "type" refers to research survey or fishing data. GLM analysis was carried out with the R software (www.r-project.org).

Time series analysis

Min/Max Autocorrelation Factor Analysis (MAFA) was applied to the time series of fisheries catch statistics (1986 to 2000), which is a recently developed technique for analysing short (at least 15 years), non-stationary, multivariate data sets (Solow, 1994; Zuur, 2003).

MAFA is a type of principal components analysis (PCA) where the axes represent autocorrelations and give an indication of the amount of association between variable Y_t and Y_{t+k} where k is the time lag with values 1, 2, 3, etc. Unlike PCA where the first axis explains most of the variance, the first MAFA axis has the highest auto correlation and since trends are associated with high autocorrelation it therefore represents the main trend in the data (Erzini *et al.,* 2005; Erzini, in press). MAFA can be used to extract trends from multiple time series, estimate index functions from time series and for smoothing (Zuur, 2003). The significant MAFA axes can be considered quantitative indicators for multispecies fisheries time series data (Erzini *et al.,* 2005).

As in PCA, loadings can be estimated and used to determine the relationship of individual response variables to particular MAFA axes. Cross-correlations between MAFA axes and the response variables, also known as canonical correlations, are a measure of the relationship between Y_t and X_{t-k} and can also be calculated for the same purpose. This allows significant relationships between trends and explanatory variables to be identified. The software Brodgar (www.brodgar.com) was used to carry out MAFA on catches in Cape Verde.

Catch statistics were available at the species level (58 species and species groups), but these were grouped into main ecological functional groups for MAFA analysis (using the same groups as in Chapter 4); small pelagics, yellowfin tuna, skipjack tuna, small tuna, pelagic predators, demersal predators, demersal fish, jacks, moray eels and sparids. Variables such as fishing effort, yearly precipitation, an SST index (yearly differences between October and March mean values) and the North Atlantic Oscillation (NAO) winter index (http://www.cgd.ucar.edu/) (Hurrell, 1995) were included for canonical correlation analysis.

Sea surface temperature (SST) monthly data was available from the NASA Physical Oceanography Distributed Active Archive Center at the Jet Propulsion Laboratory, California Institute of Technology (http://podaac.jpl.nasa.gov/sst/). Monthly precipitation data was available through the web (http://www.cru.uea.ac.uk/cru/data/), courtesy of Dr. Mike Hulme at the Climate Research Unit, University of East Anglia, Norwich, UK.

6.3 Results

Structure of demersal fish assemblages

MDS results indicated that there was no clear pattern in the structure of demersal fish assemblages (Figure 6.1). Various plots considering the effect of time, area, or depth showed that these effects were not apparent (shown for area in Figure 6.1). This lack of pattern is also evident from the relatively high stress value (0.23). Such a stress value, between 0.2 and 0.3, indicates that the plot is difficult to interpret and the results are inconclusive. Other attempts with various data transformation did not alter the overall result of an uninformative MDS plot.

Considering the uninformative MDS plots, Analysis of Similarities (ANOSIM) was used in an exploratory manner in order to decide whether to take the analysis a step further. The 2-way crossed ANOSIM allows for the testing of two factors simultaneously as well as pairwise tests for each factor level. But as there were three pre-defined factors Year, Area and Depth, these were tested using all the possible combinations of two (capital letters refer to specific factors) (Table 6.2). The differences between Year and Depth group averages (square-root transformed standardised catches) appear to be highly significant ($P<0.001$ or $P<0.1\%$).

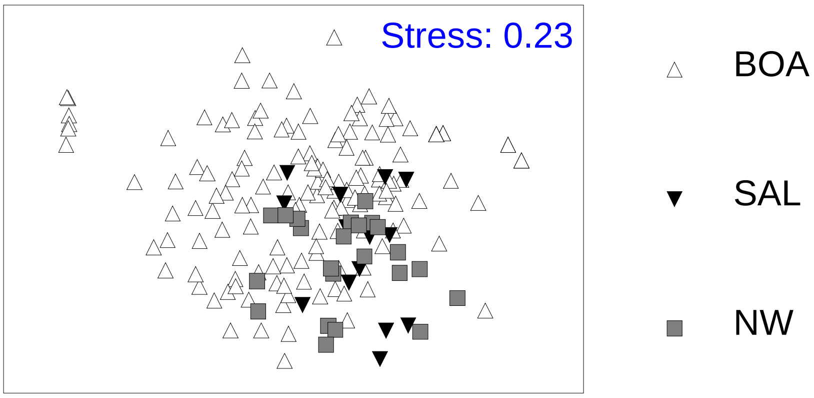

Figure 6.1. Two-dimensional MDS plot based on the similarity of haul stations in terms of species composition and abundance (aggregated by family and square-root transformed) by area. The symbols refer to area; SAL: shelf around the northeastern island Sal, NW: northwest shelf system, BOA: Boavista-Maio shelf (eastern islands). Note that MDS plots can be rotated without affecting the result (the relative distance between points) and the axes represent two arbitrarily chosen dimensions.

Table 6.2. Results of the "stepwise" 2-way crossed ANOSIM for the various combinations of three factors Year, Area and Depth.

Step	**Factors**	**R**	**Significance (%)**
1	Depth	0.205	0.1
	Area	0.040	22.1
2	Area	0.074	13.3
	Year	0.300	0.1
3	Year	0.330	0.1
	Depth	0.237	0.1

MDS plots based on averaged data yielded completely different results, which were consistent with the ANOSIM results (Figure 6.2: left-hand plot). These were based on standardised catches averaged over Year and Depth. A stress value of 0.09 indicates a good ordination result with three clearly defined groups. These groups are the greater depths (upper left), the 1976 survey (upper right), and the rest. Cluster analysis yielded a similar result with these three groups at the 50 percent similarity level (not shown). Depth strata 1 and 2 appear to be similar, but quite different from strata 3. The change over time appears to be mainly due to the anomalous results of the 1976 survey as there is no clear time gradient in the MDS plot. It is also worth noting that area did not have any clear effect on the species composition and abundance of demersal fish communities.

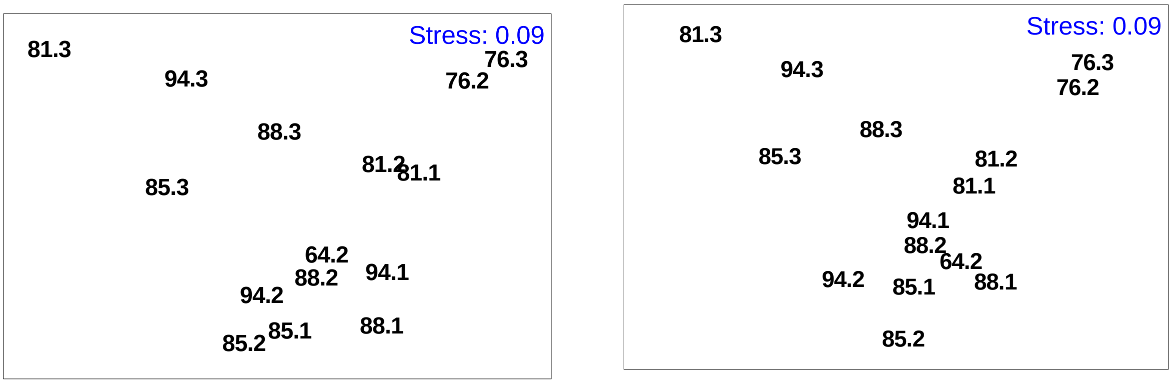

Figure 6.2. MDS plots based on averaged catches by Year and Depth, showing the similarity in terms of species composition and abundance (aggregated by family and square-root transformed). Left-hand plot was based on the full set of 60 variables (families) and the right-hand plot was based on 8 variables (families) selected by BVSTEP (190 samples or stations in both cases). For given codes, the first two digits represent Year and the last digit represents Depth strata (Depth1: ≤30m; Depth2: 30-80m; Depth3: 80-200m).

An interesting question is to determine which species (or rather families) are contributing most to the observed pattern over time and space. The procedure used was to compare the original similarity matrix with a second similarity based on a subset of variables (families) from the original data set. The BVSTEP procedure was used to iteratively choose a subset of variables, the "best result" being the smallest number of variables with the highest Spearman rank correlation coefficient, ρ (criteria $\rho > 0.95$). In this way a subset of 8 variables were found to explain most of the observed variation over time and space (Year and Depth) ($\rho = 0.959$). The right-hand plot in Figure 6.2 shows that an MDS plot based on these 8 variables only is almost undistinguishable from the MDS plot based on the original data.

The variation in the abundance of these eight families that accounted for about 90 percent of all catches over time and space is shown (Figure 6.3). Caproidae (*Antigonia capros*) dominates in terms of abundance at greater depths (Depth 3: 80-200m). The results of the 1976 survey stand out clearly with relatively high catches dominated by Sparidae and Carangidae (Appendix G). In the other cases, all of these eight families are present but variable over time and space. Thus, there is a clear grouping in three distinct groups as in the MDS results (deeper waters, 1976, and the rest; Figure 6.2), and the lack of a consistent trend over time.

Finally, characteristic species assemblages were defined using the SIMPER procedure, which calculates the percentage contribution of each species to the similarity (typical species) and dissimilarity (discriminator species) between depth groups. Depth strata 1 and 2 were very similar in terms of typical species and the results were contradictory in that discriminator species were also typical species (not shown). Thus, these two depth strata (1 & 2) were pooled as a separation did not seem justified. The effects of Year were removed by omitting 1976 survey data. Appendix H presents the typical assemblages of shallow and deep waters and the discriminator species that distinguish them. The general conclusion is that shallow and deep assemblages do not differ in terms of species composition, but rather in abundance. The deep-water assemblage appears nevertheless to be less diverse in composition. Caproidae appeared to be the only species that clearly distinguished these two assemblages, being very abundant in deeper waters and rare in shallow waters. In general, many of the same species were found to be both typical and discriminator species and the main distinguishing factor was higher abundance in shallow waters and the dominance of Caproidae (*Antigonia capros*) in deeper waters.

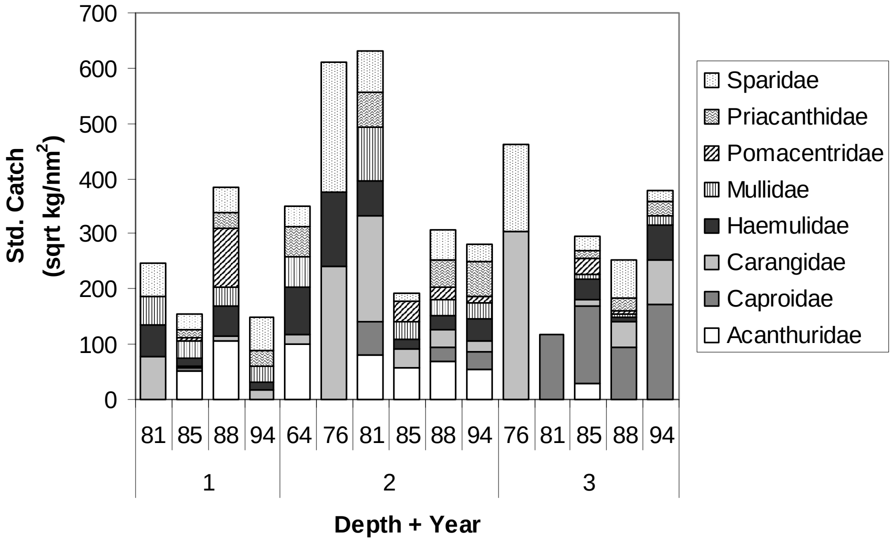

Figure 6.3. Mean abundance of eight families over Year and Depth (square-root transformed standardised catches). Depth strata (Depth1: ≤30m; Depth2: 30-80m; Depth3: 80-200m).

Demersal fish abundance over time

In relation to GLM analysis, the first step was to analyse the full time series including data from commercial fishing. The time period in question is from 1964 to 1997, but some limitations were imposed at the cost of including commercial fishing data (1994 – 1997; Table 6.1). For example, depth information was not available by haul and we assume that most fishing occurred at depths between 20 and 80 metres, which corresponds to the first two depth strata. Also, the catches of non-commercial species were suspiciously low, so this category was not considered in the analysis. On the other hand, commercial fishing took place during the whole year, making it possible to test for seasonal effects (cold: December to June, warm: July to November)

The GLM model was highly significant ($p<0.001$; Table 6.3) with significant effects of factors such as Year, Season and Type (research survey or commercial fishing) on catches. Capital letters will be used in the following when referring to specific factors. Area did not have a significant effect and was omitted from the model. The effect of Season was observed as significantly lower catches during the warm season ($p<0.001$). Likewise, catches of research surveys were significantly lower ($p=0.025$). Of particular relevance are the Year effects, which represent a standardized abundance index. Abundance was significantly higher in 1976 ($p<0.001$) and lower in 1996 ($p<0.039$), although the latter was only mildly significant. Figure 6.4 shows mean abundance of demersal fish over time based on the GLM model, which appears to show lower abundance in recent years but this was not significant except for 1996.

Table 6.3. GLM results for Cape Verde, including model specification and results as well as regression coefficients for each factor level. Only the significant factors Year, Season and Type (survey or fishing data) were included in the model. Coefficients express the difference between each level of the factors and the first level. Abundance of demersal fish expressed as kg/nm^2 and transformed as Log (standard catch + 100).

Model: Log (std. catch + 100) ~ Year + Season + Type

Coefficients:	**Estimate**	**Std. Error**	**t value**	**Pr(>|t|)**
(Intercept)	4.571	0.277	16.517	0.000
Year_1976	1.124	0.300	3.754	0.000
Year_1981	0.322	0.302	1.068	0.286
Year_1985	-0.227	0.294	-0.771	0.441
Year_1988	-0.032	0.270	-0.120	0.905
Year_1994	-0.199	0.273	-0.728	0.467
Year_1995	-0.365	0.314	-1.164	0.245
Year_1996	-0.603	0.291	-2.072	0.039
Year_1997	-0.162	0.328	-0.495	0.621
Season_warm	-0.398	0.110	-3.633	0.000
Type_survey	-0.320	0.142	-2.259	0.025

Residual standard error: 0.5823 on 341 degrees of freedom
Multiple R-Squared: 0.2152, Adjusted R-squared: 0.1922
F-statistic: 9.352 on 10 and 341 DF, p-value: 9.872e-14

A second GLM analysis was undertaken on trawl survey data only, omitting the commercial fishing. The logic behind this was that a comparison of these two types of data should be handled with care. Thus, the results should be similar and consistent after omitting fishing data and this allowed also for a testing of depth effects. Again the GLM model was highly significant ($p<0.001$; Table 6.4) and showed that the factors Year, Depth and Group had significant effects on standardised catches. The effect of Group shows significantly higher catches of fish in relation to elasmobranchs ($p<0.001$). Catches at greater depths were significantly lower (Depth2: $p<0.001$, Depth3: $p=0.046$). Commercial value did not have a significant effect ($p=0.463$), which indicates that the proportion of catches of commercial and non-commercial species has been stable over time and space. Note that Area did not have a significant effect as in the first GLM model (Table 6.3 & 6.4).

In relation to the standardized abundance index over time, the results were similar to the first GLM model. Abundance was significantly higher in 1976 ($p<0.001$). However, abundance was significantly lower in 1985 ($p=0.014$) and 1994 ($p=0.004$), which does not coincide with the results of the first model (Table 6.3; significantly lower abundance in 1996 only). These results appear to show that abundance has decreased in recent years, but there is no apparent trend in the data.

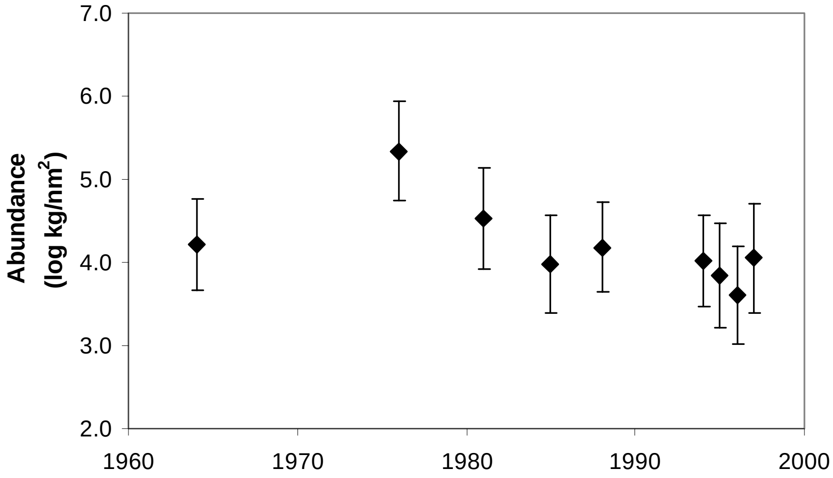

Figure 6.4. Abundance of demersal fish over time. Estimated by GLM as the intercept plus the appropriate year coefficient plus the mean of Season and Survey coefficients (Table 6.2). Error bars represent approximate 95 % confidence intervals estimated as ±2 x SE (Standard Error).

Table 6.4. GLM results for trawl survey data only, including the effects of factors such as Year, Depth (Depth1: ≤30m; Depth2: 30-80m; Depth3: 80-200m), Area (SAL: Sal Island, NW: northwest shelf system, BOA: Boavista-Maio shelf), Group (fish or elasmobranches), and Commercial value (c.value: com and ncom representing species of value and of no value, respectively). Coefficients express the difference between each level of the factor and the first level. Mean abundance of demersal fish expressed as kg/nm^2 and transformed as Log (standard catch).

Model: Log (std. Catch) ~ Year + Depth + Area + Group + C.Value

Coefficients:	Estimate	Std. Error	t value	Pr(>|t|)
(Intercept)	3.455	0.210	16.440	0.000
Year_76	1.291	0.269	4.807	0.000
Year_81	0.101	0.214	0.473	0.636
Year_85	-0.603	0.244	-2.465	0.014
Year_88	-0.250	0.181	-1.386	0.166
Year_94	-0.528	0.185	-2.859	0.004
Depth_2	-0.384	0.109	-3.516	0.000
Depth_3	-0.259	0.130	-1.999	0.046
Area_NW	0.062	0.152	0.410	0.682
Area_SAL	0.094	0.144	0.652	0.515
Group_fish	0.520	0.098	5.314	0.000
C.value_ncom	-0.057	0.078	-0.735	0.463

Residual standard error: 0.7479 on 399 degrees of freedom
Multiple R-Squared: 0.2706, Adjusted R-squared: 0.2505
F-statistic: 13.46 on 11 and 399 DF, p-value: < 2.2e-16

An interaction term between Year and Depth was included in the model to determine whether this had bearing on the results, the Year effects in particular. This created problems because the data is unbalanced and not all coefficients could be determined. However, the results were similar in showing significantly lower abundance in recent years but with no apparent trend.

Time series analysis

The first MAFA axis is shown in Figure 6.5, which has the highest auto-correlation with time lag 1. The underlying idea is that a trend is associated with high auto-correlation. Therefore, the first MAFA axis represents the significant trend or the main underlying pattern in the data; a trend of steadily increasing catches. Only the first axis was significant (auto-correlation with time lag 1=0.97; p=0.048). As expected, fishing effort was correlated significantly with this increasing trend in catches (industrial effort = 0.79 and artisanal effort = 0.93; critical level for a significant correlation was ± 0.52 (p=0.05). The correlations to environmental variables such as yearly precipitation, NAO, and the SST index were not significant.

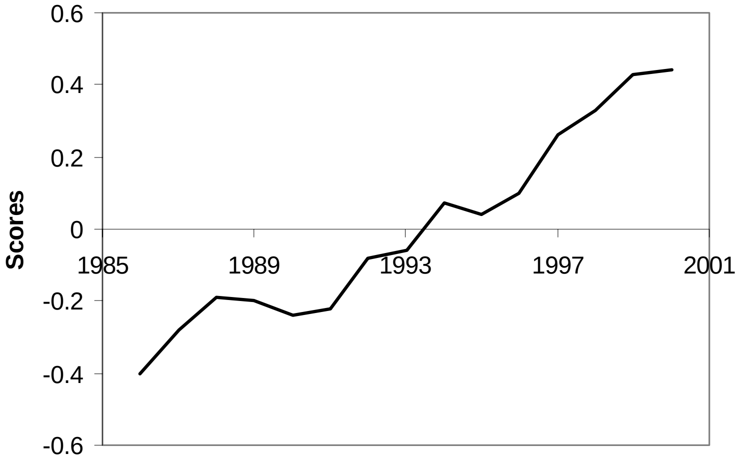

Figure 6.5. Plot of the first MAFA axis showing the trend over time or the main underlying pattern in the data. MAFA scores on the y-axis are analogous to scores in PCA analysis, a measure of distance of points from the axis. The first axis is chosen so as to minimize the sum of squared distances of the points from the line and maximising the variance of points projected on the axis. In MAFA however, the points are projected on an axis that represents a trend.

Figure 6.6 shows the canonical correlations between species group catches and the first MAFA axis. The significant negative correlation for yellowfin tuna shows a decreasing trend in catches (the same for skipjack but not significant). Except for demersal fish and jacks, a significant positive correlation was obtained for the other groups, indicating increasing catches over time. For demersal fish and jacks, catches have followed a different pattern with two peaks during the study period.

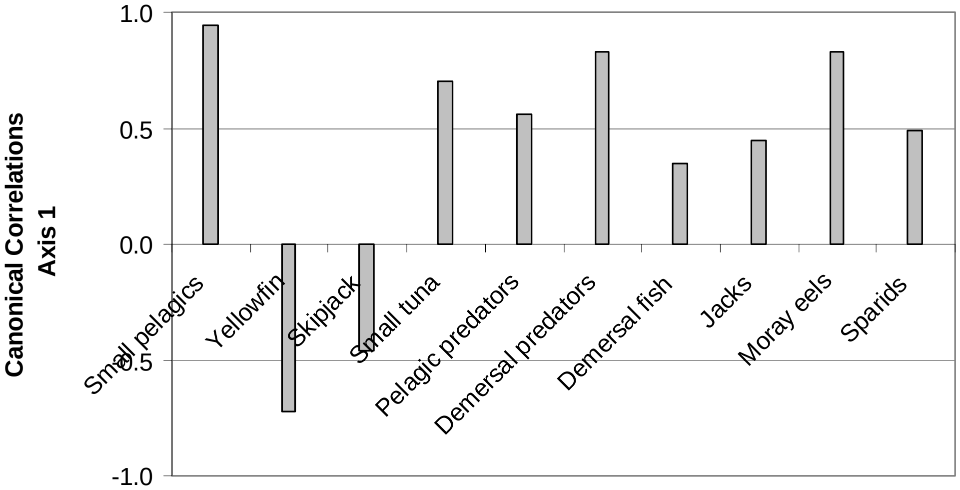

Figure 6.6. Plot of the canonical correlations between species groups and the first MAFA axis. Critical level for a significant correlation was ± 0.52 ($p=0.05$).

6.4 Discussion

Fishing is expected to bring about change in the structure of species assemblages because of direct effects such as on size and biomass of target species as well as indirect effects due to the complex interrelationships of species. Fariña *et al.* (1997) found that the variability in assemblage structure in the Galician continental shelf was determined mainly by temporal changes over a 10 year period and, to a lesser extent, by depth and geographical location. However, spatial structure is generally considered important in the definition of demersal fish assemblages in the northwest African region (Fager and Longhurst, 1968; Longhurst and Pauly, 1987; Domain *et al.*, 1999; Amorim *et al.*, 2004), as well as in other tropical and subtropical areas (Bianchi, 1992; Garcia *et al.*, 1998; Rodríguez *et al.*, 1998).

Continental shelves of the tropical eastern Atlantic are characterized by the presence of a sharp thermocline separating the upper mixed layer from much cooler and deeper waters, which explains the presence below or in association with the thermocline, of species typical of the Mediterranean, South Africa, and the more temperate coast of West Africa (Longhurst and Pauly, 1987; Saetersdal *et al.*, 1999). This temperature difference may be the principle cause for distinct assemblages according to depth strata. Sediment type also appears to be useful in defining distinct habitats and the associated demersal fish assemblages (Longhurst and Pauly, 1987; Domain *et al.*, 1999; Amorim *et al.*, 2004).

In the case of Cape Verde, a recent study did not find any clear pattern in the structure of demersal fish assemblages when considering factors such as time, area and depth (Stobberup *et al.*, in press). This study presents the same results, but with more detail concerning the assemblages and including exploratory analysis concerning the effects of time, depth and area.

Exploratory analysis of similarities (ANOSIM) showed that the effects of Year and Depth were significant. This was not the ideal approach to testing as all three factors should have been tested simultaneously including Year, Depth and Area, but this was not possible using the Primer software. Surveys did not have a consistent coverage of depth and area, in particular, which creates the danger of confounded effects (Table 6.5). However, the effects of Year and Depth were consistent and it was shown that this was largely the result of variation in the abundance of eight families, which dominate in terms of biomass. Further analysis showed that the effect of Year was largely the result of the 1976 survey (Figure 6.3). The study by Danke and Koch (1987) was the only source of information for the 1976 survey as the raw data was not available for analysis. There is however some doubt as the catches were extremely high and these were dominated by *Trachurus picturatus*, which has never been the dominant species in successive surveys (Stroemme *et al.*, 1982; Vieira, 1985). Standardised catches were aggregated by family to avoid possible problems in relation to species identification, but this did not mask the anomalous results of the 1976 survey. The effects of fishing are expected to be clear trends such as decreasing abundance or proportion of commercial target species, but this was not observed (Figure 6.3).

Table 6.5. Number of haul stations by Area and Depth strata over time (Year). Area - BOA: Boavista-Maio shelf, NW: northwest shelf system, SAL: Sal Island; Depth - Depth1: ≤30m, Depth2: 30-80m, Depth3: 80-200m.

Year	**BOA**			**NW**			**SAL**		**Total**
	1	**2**	**3**	**1**	**2**	**3**	**1**	**2**	
64		6							6
76		4	12						16
81	1	11	2				1		15
85			3	5	5	6			19
88	9	4	14		4	3	3	6	79
94	7	34	1					4	55
Total	17	95	41	5	9	9	4	10	190

Depth had a clearer effect on the structure of demersal fish assemblages, but it was shown that the dominance of *Antigonia capros* at greater depths was the sole distinguishing characteristic (Appendix G & H). Shallow and deep assemblages did not differ in terms of species composition, but rather in relative abundance. In fact many of the same species were found to be both typical and discriminator species which indicates a "fuzzy" distinction. This lack of distinction may be related to the limited space available to demersal fish assemblages on the continental shelves around the islands, which also seems to be a characteristic limitation on the deep-water assemblages in the archipelago (Menezes *et al.*, 2001). On the other hand, Area did not have any clear effect on the abundance and species composition of demersal fish assemblages, which was unexpected. This unexpected result may be related to the quality of the historical surveys in terms of species identification as well as the difficult conditions for adequately sampling the shelves around the islands. However, the similarity among areas is likely as it is the same habitat that is being sampled in several islands and shelves, which are the trawlable grounds with sediments of the sandy-bottom type. It is worth noting that this has important implications and it supports the approach used in modelling studies (Stobberup *et al.*, 2002, 2004), which disregard differences between areas. In relation to fisheries management, there seems to be no evidence for the existence of separate independent populations and the biomass appears to depend on the available habitat, the continental shelf around each island. Thus, each island / shelf system should be considered separately in terms of productivity, but not as independent units for management purposes.

The present study did not consider depths greater than 200 metres and the southern islands of Brava, Fogo and Santiago are not represented. There may be some important differences between the southern and northern islands, considering the frontal system that divides the archipelago in warmer southern waters and colder northern waters during part of the year (Almada, 1993, 1994). The survey carried out in 1985 did in fact cover greater depths and the southern islands, but these stations were omitted in an attempt to achieve greater balance in the data (Table 6.1). In 1985, distinct assemblages were observed at greater depths, bearing in mind that this refers to the continental slope, which cannot be compared to the results of the other trawl surveys. But more importantly, there were no differences between areas including northern and southern islands. These results for 1985 were not included, as they did not shed any new light on the main objective, which was to study change over time and determine the possible effects of fishing. Also, various other transformations were applied to the data to determine whether this had any bearing on the results. The effect of the different transformations was surprisingly small (presence-absence, log), but this is related to the fact that most of the variability in the data is contained in the abundance of only eight families.

Abundance and CPUE over time

Stobberup *et al.* (in press) found a decreasing trend in demersal fish abundance in Cape Verde, considering the same data and a similar analysis. The approach used here was to prolong the time series by including data from a commercial fishing operation. This was possible because the vessel belongs to the Fisheries Institute (INDP) in Cape Verde. However, significant differences were found between survey and commercial fishing data, and as expected, commercial catches were significantly higher than survey catches. However, there was some doubt about the species composition of commercial catches, indicating that discards were not taken into account and that catch estimates may be underestimates.

Generally, GLM analysis results indicated lower abundance in recent years. The two approaches, with and without commercial data, gave similar but not entirely consistent results. A GLM model on survey data with an interaction term between Year and Depth was attempted, but this was associated with problems because of the unbalanced data (Table 6.5). These inconsistent results as to which years had a significantly lower abundance may be related to the confounded effects of Year and Depth. However, the tendency for lower abundance in recent years was clear, which is consistent with an increase in fishing pressure over time. Fishing effort has increased gradually by a factor 2 to 3 over the last twenty years, but the fishery for demersal fish is dominated by small artisanal boats with outboard engines using handlines. Thus, this fleet is expected to exert a moderate effect of lowering demersal fish abundance gradually. The significantly higher abundance in 1976 should be interpreted with care, since there are some doubts about the quality of the data. Area did not have a significant effect on abundance in both GLM models, which is also consistent with the multivariate results. Abundance was significantly lower at greater depths, which is generally the case (*e.g.* Longhurst and Pauly, 1987). It is also worth noting that Commercial value did not have a significant effect (Table 6.4), which indicates that the proportion of catches of commercial and non-commercial species has been stable over time and space.

Standardised catches were log-transformed in an attempt to normalise the observations as well as stabilise variance, but the resulting distributions of the GLM model residuals were slightly skewed. GLM is nevertheless robust to this type of slight deviations (Dalgaard, 2002; Zuur, 2003)

The available survey data made it possible to span a period of more than 30 years from 1964 to 1997, but the sporadic frequency of these surveys and the changing methodology imposed some limitations in terms of data analysis as well as making it difficult to reach conclusive results. A reliable time series of catch per unit of effort (CPUE) was available since 1986, making it possible to apply MAFA, which is a type of time series analysis. MAFA is a recently developed technique for analysing short (at least 15 years), non-stationary, multivariate data sets, which are typically found in fisheries data (Solow, 1994; Zuur, 2003; Erzini *et al.,* 2005; Erzini, in press). It is important to point out that all major commercial species groups were analysed, including pelagic and migratory species.

A general trend of increasing catches was observed with MAFA (Figure 6.5). This was related to increasing catches of small pelagics and neritic tuna, primarily, while catches (and CPUE) of important species such as yellowfin and skipjack tuna have decreased (negatively correlated with MAFA axis). There was no significant trend in the catches of demersal fish, which is comparable with the results for demersal fish in the GLM and multivariate analysis. MAFA results showed that there was an important shift in the composition of fishery catches over the period from 1986 to 2000.

The MAFA results were useful in providing a general pattern over time for the multispecies fisheries data in Cape Verde. Also important was the fact that the correlation coefficients in relation to environmental variables were not significant, implying that fishing was driving the changes. However, it is important to point out that species such as yellowfin and skipjack tuna have ocean-wide distributions and the observed decreases in catch and CPUE cannot be interpreted as the local effects of fishing in Cape Verde (*e.g.* ICCAT, 2004). A recent modelling study showed that the decrease in abundance of important tuna predators led to less predation on small pelagics and neritic tuna, in particular, which resulted in biomass increases (Stobberup *et al.*, 2004). The same study also questions a decrease in biomass of demersal fish estimated by trawl surveys (Monteiro, 1999-b), as it was not compatible with observed catches. Further research is necessary to deal with these conflicting results.

6.5 Conclusions

Two recent studies have studied size spectra of demersal fish communities in Northwest Africa, including Cape Verde (Thiam *et al.*, 2004; Stobberup *et al.*, 2005). Size spectra as well as other size related indicators (mean size, mean size by class, etc.) have been suggested as potentially useful for detecting the effects of fishing upon fish communities and they appear to have a suitable theoretical foundation (Pope *et al.*, 1988; Haedrich and Barnes, 1997; Murawski and Idoine, 1992; Rice and Gislason, 1996; Bianchi *et al.*, 2000; Zwanenberg, 2000; Rochet and Trenkel, 2003). An increasingly negative slope of size spectra over time is interpreted as being consistent with the expected effects of fishing on community structure. Apart from changes over time, little attention has been given to spatial variability in the study of size spectra, but these appear to be important and should not be disregarded (ICES, 2003; Thiam *et al.*, 2004). Also, the problem of defining reference levels and separating the effects of fishing from the effects of the environment have been central themes in a recent symposium (Cury and Christensen, 2004).

Stobberup *et al.* (2005) found that size spectra intercepts and slopes were relatively stable for a period spanning typically 10 to 15 years in Guinea and Mauritania, respectively, in spite of a strong increase in fishing pressure. When separating demersal fish of commercial and non-commercial value, the expected result of increasing intercept and steeper decreasing slope was observed in Mauritania only. In the case of Cape Verde, only two years of data were available for size spectra analysis and this lack of a consistent time series gave unexpected results such as important change in structure over time, which was not compatible with a relatively moderate increase in fishing pressure over the same period (Thiam *et al.*, 2004; Stobberup *et al.*, 2005). This appeared however to have been a problem of data (extreme and influential points). Generally, the slopes of the size spectra in these three cases (Cape Verde, Guinea and Mauritania) appear to be less sensitive to changes in fishing intensity over time, compared to the findings in higher latitudes. Considering this, Stobberup *et al.* (2005) concluded that change in size structure over time, expressed as the intercepts and slopes of size spectra, does not appear to be a suitable indicator for determining the effects of fishing in tropical areas, if it is used as the sole indicator. The analysis of size spectra in other tropical areas has also shown a similar lack of trend (Bianchi *et al.*, 2000; Rochet and Trenkel, 2003). Fishing effects taking place may be masked when adopting highly aggregative indicators, particularly when considering tropical areas that are characterized by higher growth rates, smaller sizes, high species diversity and complex inter-relationships (Bianchi *et al.*, 2000; Thiam *et al.*, 2004; Stobberup *et al.,* 2005).

In Cape Verde, fishing effects were particularly clear through MAFA, which showed that an important shift has occurred in the proportions of pelagic and demersal species in the catches. The limitations concerning sampling with trawl gears become particularly evident in an area such as the Cape Verde Archipelago, where there are very restricted areas of trawlable sandy-bottom grounds. Instead, pelagic and/or migratory species dominate this type of ecosystem, making it essential to consider these species. On the other hand, demersal fish resources are particularly vulnerable in Cape Verde, because of limited habitat space and relatively low productivity of the system. The decreases in catches of tuna species have to be considered on the regional or ocean scale, but the effects of fishing on demersal species has to be monitored closely in order to make proper assessments of exploitation level (*e.g.* Monteiro, 1999-b; Stobberup *et al.*, 2004). The present study showed several approaches that can be used to study fishing effects, which become clearer when using a comparative approach. However, the usefulness of ecosystem indicators in fisheries management is still a developing field of research and we hope to have contributed for tropical areas, in particular.

Chapter 7. General Conclusions and Discussion

In this last chapter, we will present an overview of the most important results including a comparison of results using alternative methods. The present study placed emphasis on studying the dynamics of the Cape Verde coastal ecosystem, including the possible changes that have occurred over a period of almost four decades. Considerable efforts went into identifying, compiling and standardising trawl survey data in order to span this period. These included the first quantitative trawl survey, carried out in 1964, and the more extensive surveys in the 1980s and 1990s. Trawl surveys were carried out sporadically over this period and this lack of a consistent trawl survey time series was a serious limitation. These data were nevertheless valuable in giving indications concerning demersal fish resources in trawlable grounds.

However, demersal fish resources account for less than 10 percent of the catches in Cape Verde and the coastal ecosystem is dominated by pelagic fish species, including migratory and neritic species. The Ecopath approach was used to study the dynamics of these pelagic groups as well as other major ecological groups, but with emphasis on fish. It is nevertheless important to point out that migratory species are ocean-wide in distribution and should be considered in a much wider context. These species, tuna in particular, were included as important fishery resources and predators, but treated as external factors and beyond the control of the national fisheries. The ecosystem model was based on data from 1981 to 1985, which was a time period characterised by predominantly artisanal fisheries and a low level of motorisation; the initial phase of a more intensive fishery development. Fisheries statistics became available in this period also. Subsequently, the simulation study using Ecopath with Ecosim covered the period from 1986 to 2000, thus shortening the study period in comparison with the analysis of trawl survey data. On the other hand, reports on surveys carried out in the early 1980s indicate that stocks were at virgin levels, thus indicating a very moderate exploitation.

Various techniques were used to analyse the available data for the purpose of comparison. This was useful as each technique was particularly suited for answering different aspects of the same question such as changes in biomass, species composition, size structure or CPUE over time as well as common trends for multispecies fisheries data (Chapter 6). Acoustic survey estimates of small pelagic biomass were questionable, indicating a very high 12 tonnes per km^2 in 1981 and a sharp decrease to 2.5 tonnes per km^2 in 1997. Conventional biomass dynamic modelling was applied to analyse data on small pelagics as a complementary tool to the ecosystem model (Chapter 5). Being a stock assessment method, the results of biomass dynamic modelling were particularly useful for management purposes. Moreover, these results were also useful to evaluate the simulation results for small pelagics from the Ecopath model.

7.1 Working Hypotheses

Based on the results, there appears to be no doubt that we can reject the null hypothesis concerning change in the coastal ecosystem. Change has occurred and has manifested itself in various ways, which we will summarise by answering the research questions given in the introduction (Chapter 1). These were:

a) Is the change observed due to fishing or environmental effects or possibly a combination of both?

Environmental effects are expected to play an important role in marine ecosystems, but the question here is its relative importance compared to fishing.

Environmental variables were generally disregarded in the present study except for the time series analysis with MAFA (Chapter 6). In this case, the correlation between catches and indices of SST, NAO, and precipitation were not significant. Simulations with Ecosim gave a reasonable overall fit to the time series of catches for 18 fish groups by using three overall trends in relative fishing rate (Chapter 4). In addition, biomass of demersal fish showed relative stability over the period 1964 to 1997, although there was a tendency for lower biomass in recent years (Chapter 6). Thus, this indicates that fishing was the main driving force behind the observed changes over a 15-year period (1986 to 2000).

Considerable efforts have gone into the study of the effect of climate change in marine ecosystems in order to understand the effect of human impacts in a varying system (Steele, 1998; Barange and Harris, 2003). It has been possible to relate climate change and fluctuations in pelagic forage fish, mostly in upwelling ecosystems (Binet, 1998; Klyashtorin, 1998, 2001; Roy and Reason, 2001; Jacobson *et al.*, 2001). At the ecosystem level, there are a few examples of the consequences of changing climate or regime shifts in terms of fisheries catches and composition, but these cases refer exclusively to temperate, sub-arctic, boreal or upwelling ecosystems (Francis *et al.*, 1998; Livingston and Tjelmeland, 2000; Walther *et al.*, 2002; Barange and Harris, 2003). On the other hand, tropical oceans are often characterised as biological deserts that are characteristically stable (Bakun, 1996). For example, the recently observed global warming of the world’s oceans shows that relatively small increases in temperature are found in tropical oceans compared to higher latitudes (Walther *et al.*, 2002). Considering the period between 1985 and 1998, coinciding with the simulation study, sea surface temperatures have been relatively stable in the area around Cape Verde (Figure 7.1).

Precipitation is considered as one of the most important environmental effects in Cape Verde. Drought has brought about disaster repeatedly in the history of Cape Verde (Almeida, 1997) and the problem is expressed in a local proverb that says:

> *If the rains come, we die of drowning. If the rains don't come, we die of thirst.*

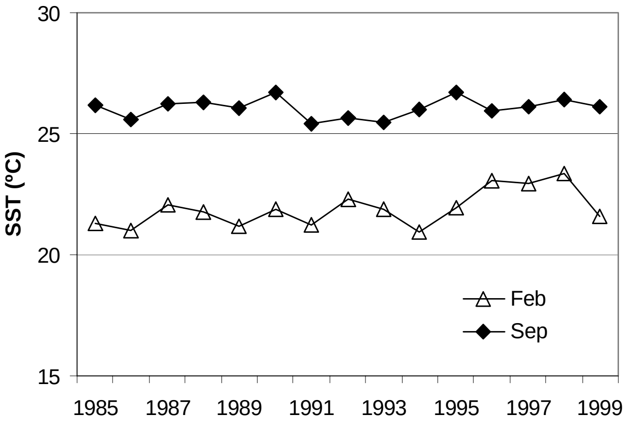

Figure 7.1. Sea surface temperatures (SST) during the cold (February) and warm (September) seasons in Cape Verde. SST monthly data was available from NASA (http://podaac.jpl.nasa.gov/sst/). (Upper left and lower right coordinates of the defined area are: 17º42'N 25º73'W & 14º57'N 22º50'W).

As this proverb indicates, when rainfall does occur, it often brings about problems related with excess water such as flooding and strong erosion. Anecdotal evidence on the importance of precipitation in enriching coastal waters is found among fishermen (*e.g.* Watanabe, 1981). Data on precipitation showed two peaks during the period 1980 to 1998, which is similar to the pattern observed in some CPUE artisanal time series (Figure 7.2; Figure 4.3). The MAFA results indicated that this relationship was not significant, but this possible enrichment may play a role for specific fish groups. However, the short time period of the simulation study (15 years) and the inherent variability of the CPUE data make it impossible to reach firm conclusions in this respect.

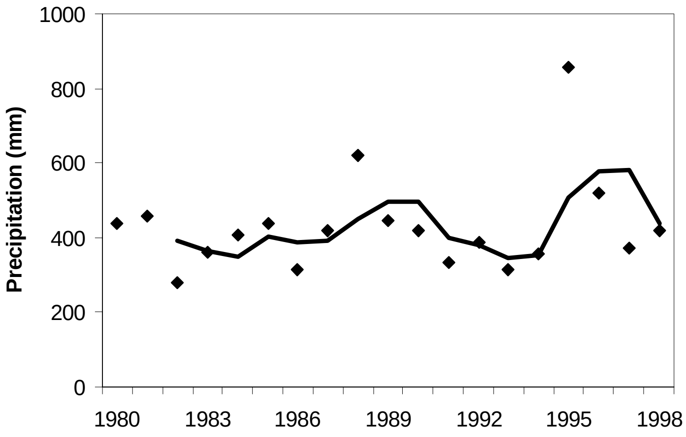

Figure 7.2. Precipitation in Cape Verde, showing yearly estimates and the 3-point running average as a trend line. Monthly precipitation data was available, courtesy of Dr. Mike Hulme, from the Climate Research Unit, University of East Anglia (http://www.cru.uea.ac.uk/cru/data/).

b) Has there been a change (decrease or increase) in biomass for specific species or groups?

The migration of tuna through the system, where they feed on coastal species and are exploited by national fisheries, can be regarded as an external factor. An interesting question in this context is whether the observed decrease in abundance of tuna, yellowfin tuna in particular, is a result of fishing. A recent study has shown that industrialised fisheries have typically reduced community biomass by 80 percent within 15 years of exploitation, considering a period of 50 years (Myers and Worm, 2003). This study included an analysis of the fisheries in the tropical and subtropical Atlantic, which are dominated by tuna fisheries. There has been debate on the use of Japanese longline data to analyse tuna abundance over time (see www.spc.int/OceanFish/) and the effect of climate and the environment were not considered. However, it is commonly accepted that fishing has been the main driving force behind the decline of tuna biomass and CPUE in the Atlantic over recent decades, but the question is how much.

The lack of a continuous time series of fishery-independent data on the abundance of fish species was a serious limitation, making it difficult to determine changes in biomass in the coastal ecosystem of Cape Verde. The exception was demersal fish on sandy, trawlable grounds, but these constitute a minor proportion of overall biomass and trawl surveys were sporadic. CPUE time series were used as a substitute, but this was problematic as these were based on sampling from multigear, multispecies fisheries. The pattern of variability or trend in CPUE was very dependent on gear and scientific judgement had to be used to choose the appropriate gear for specific species groups, based on knowledge of the fisheries. However, there may be considerable bias as a result of this procedure. Results from the simulations with Ecosim showed that available effort estimates could not account for the observed catches, which indicates the use of CPUE as an index of abundance should be regarded as highly uncertain (Chapter 4).

Bearing in mind these uncertainties, we can only answer whether there may have been an increase or decrease in biomass of neritic species groups over the period from 1986 to 2000. Simulations indicated an increase of demersal groups and relative stability in small pelagics (Chapter 4). However, biomass dynamic modelling showed a strong decrease of small pelagic biomass, but again, this had to do with the choice of gear and fisheries in estimating CPUE (Chapter 5). In the case of demersal fish biomass, GLM analysis of standardised survey catches indicated that these were relatively stable over the period from 1964 to 1997 with a tendency for lower abundance in recent years, but this trend was not significant (Chapter 6). Note however that there is considerable variability around the abundance estimates of demersal fish (Figure 6.4).

c) What have been the consequences of these changes in terms of trophic interactions?

The answer to this question is related to the predicted changes in biomass in the simulation study. A decrease in tuna biomass led to a decrease in predation on pelagic neritic species such as small pelagics and small tuna as well as some demersal groups to a lesser extent. As a result, the simulated biomass of small tuna increased considerably, which is confirmed by fisheries data. Catches of small tuna have increased considerably, constituting an increasing proportion of total artisanal and industrial catches, which indicates a relative shift towards small tuna by both fisheries (Figure 7.3).

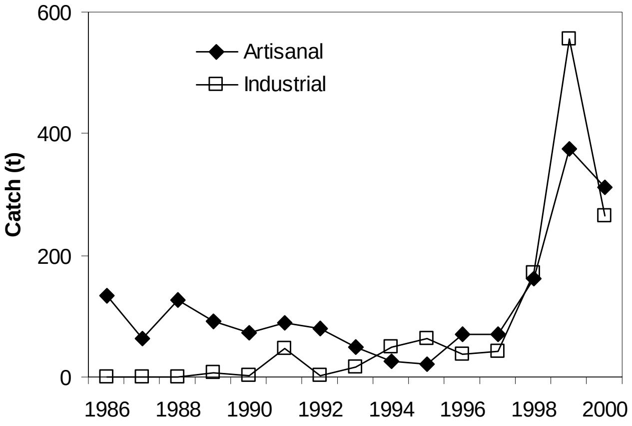

Figure 7.3. Artisanal and industrial catches of small tuna, consisting of *Euthynnus alletteratus* and *Auxis thazard thazard*.

In the case of small pelagics, the decrease in predation by tuna was compensated to a certain extent by increased fishing mortality, which led to a predicted slight increase in biomass (Figure 7.4). As referred above, the results of the biomass dynamic model showed instead a strong decrease in biomass (Chapter 5). The choice of catch and effort data was undertaken with more care in this latter case, as Chapter 5 dealt exclusively with only one fish group, consisting of a few species. Small pelagic CPUE were estimated by selecting data pertaining to both the industrial and the artisanal net (seines and gillnets) fisheries, as well as estimating an overall standardised CPUE. On the other hand, overall artisanal catch and effort were used for estimating small pelagic CPUE in Chapter 4, which is dominated by the handline fisheries. This may not have been the best choice for this particular fish group and adjustments to the ecosystem model should be made to incorporate the results of biomass dynamic modelling.

We have already touched on the difficulty of determining whether demersal fish biomass has increased or decreased. A previous study indicated that biomasses of some important commercial species have decreased (Monteiro, 1999-b). Much of the same survey data was used in this study, but we extended the time period by including historical surveys and standardising trawl catches using approximate trawl net dimensions. GLM analysis indicated a tendency for lower survey catches in recent years, but there was no significant decreasing trend and there were important seasonal and spatial effects (depth) (Table 6.3 & 6.4). This indicates that these significant effects may easily be confounded for time effects if not considered in the analysis. Also, the possible differences between commercial and non-commercial species was considered in the GLM analysis, but this was not significant.

Moreover, multivariate analysis of species composition and abundance showed that this has been relatively stable for an extensive time period since 1964, which covers the initial stages of more intensive exploitation. It is however important to point out that these survey data do not adequately sample groups such as moray eels and demersal predators (Serranidae) as well as reef-associated species. On the other hand, a plot of catch against effort for demersal species shows an overall increasing trend with no indication of a peak (Figure 7.5). This trend is driven by primarily by species such as *Cephalopholis taeniops* (demersal predator), *Pseudupeneus prayensis* (demersal fish), *Priacanthus arenatus* (demersal fish), and *Muraena helena* (moray eels), which dominate their respective ecological group in terms of catches and presumably biomass. Based on these indications, it is difficult to reach firm conclusions on whether there has been an increase or decrease. It appears that whatever change has occurred, this has not been sufficiently strong to stand out from the inherent variability of the data (C.V. ≈ 15 % for log-transformed data).

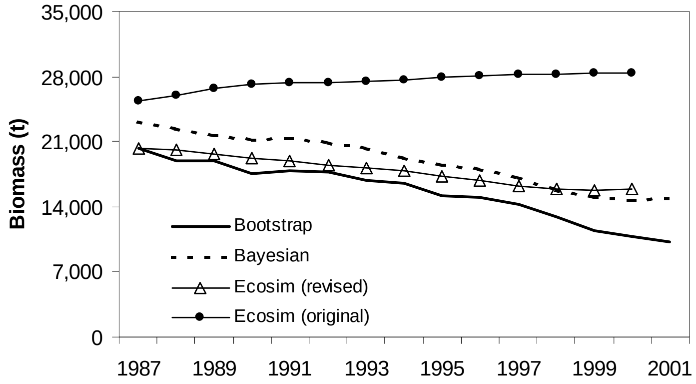

Figure 7.4. Estimated biomass of small pelagics using various methods. The Bayesian and bootstrap estimates come from Chapter 5. Ecosim (original) simulated biomass was estimated in Chapter 4 and the Ecosim (revised) estimates are explained in this chapter (7).

d) What is the current level of fishing pressure on the ecosystem and the different fishery resources and what are the implications for potential harvest?

Simulation with Ecosim showed that overall fish biomass, including pelagic migratory species, remained almost constant (- 2%) from 1986 to 2000 in spite of doubling or more of relative fishing rate (Chapter 4). This strong increase in fishing pressure resulted in only a 36 per cent increase in catches from about 7 to 10 thousand tonnes. There appears to be no doubt that previous assessments of potential harvest, ranging from 25 to 56 thousand tonnes, were overly optimistic. An increase in total catch appears nevertheless to be feasible, if managed properly, but a large increase seems highly unlikely. This depends also on the biomass and migration of tuna, which are important predators in the system.

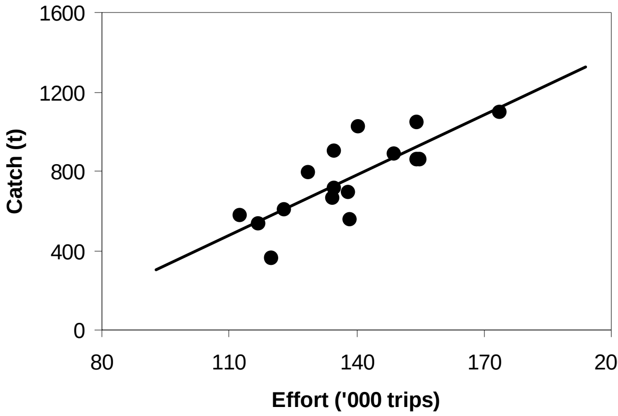

Figure 7.5. Catch (t) of demersal fish groups, considering total catch and effort in fishing trips of the artisanal handline fishery. The increasing trend is shown as a simple regression line.

A general result of ecosystem modelling is the relatively low fishing mortalities relative to predation mortalities. This is a common result from multispecies models, leading to the realisation that predation plays a dominant role in terms of mortality and may be important for recruitment success (Pope, 1991; Bax, 1998; Hollowed *et al.*, 2000; Overholtz, 2000). Similar results were obtained in Cape Verde, since the only groups that showed relatively high fishing mortality were demersal predators and moray eels (Table 4.1). These are high-valued commercial species in the fisheries of Cape Verde, which appear to show signs of over-exploitation and should be monitored carefully. Note however that the level of exploitation is highly uncertain, as there are no reliable estimates of biomass for these two groups. Fishing mortality was also relatively high for some migratory species such as tuna, but these estimates should not be considered reliable as explained in the following.

7.2 The Ecosystem Model: improvements and projections

The ecosystem model can be projected forward for different management scenarios, analogous to projections with stock assessment models. This is particularly useful in order to study the implications of changing effort by fisheries or gears and give further indications concerning potential harvest. Although straightforward with the "Ecopath with Ecosim" software, it was considered important to first make some improvements to the model regarding the inconsistencies identified for small pelagics (Figure 7.4). In this way, we attempt to show that this is an iterative process, whereby the model is improved as new data or results are incorporated. Moreover, we regard multispecies models and conventional stock assessment as complementary tools, which become more powerful when used in conjunction.

Improvements to the Model

The following modifications were applied to the ecosystem model:

a) The basic parameters for small pelagics were changed (Table 7.1). Biomass was decreased to correspond with the bootstrap estimate for B_0 (Table 5.2). Production was increased and consumption decreased, which made small pelagics more susceptible to mortality, leading to a simulated decrease in biomass (Figure 7.4).

Table 7.1. Basic parameters used for small pelagics in the original Ecopath model (Chapter 4) and in the revised model, following the adjustments explained in the text.

	Biomass (t/km²)	Prod./ biom. (per year)	Cons./ biom. (per year)	EE
Original	4.635	1.111	11.108	0.974
Revised	3.700	1.400	10.000	0.953

b) These changes in the basic parameters, leading to a smaller and less productive group of small pelagics, made it necessary to adjust the diets of its main predator, yellowfin tuna (Table 7.2). Cannibalism in yellowfin tuna was removed, predation on neritic species was lowered, and import (the food from outside the system) was increased considerably.

Table 7.2. Changes applied to the diet matrix of the original Ecopath model (Chapter 4). Values under the headings of Yellowfin and Small tuna are proportions of total diet (sums to 1) and differences (Diff.) indicate the change applied. (see Appendix F for further detail)

		Predator			
No.	Prey	Yellowfin	Diff.	Small tuna	Diff.
4	Yellowfin	0.021	-0.021		
14	O Demersal fish	0.015	-0.004		
16	Small pelagics	0.095	-0.075	0.17	0.20
23	Benthos	0.030	-0.030	0.10	-0.10
24	Zooplankton	0.120	-0.020	0.35	0.00
28	Import	0.620	0.150	0.15	-0.10

c) Some adjustments to the diet of small tuna were also made to increase its predation on small pelagics and decrease its reliance on benthos and imported food. This increased the predation on small pelagics as small tuna were increasing due to less predation from tuna.

d) In Ecosim, the fishing mortalities of tuna were adjusted upwards in order to improve the fit to catches. For yellowfin tuna, fishing mortality was increased from 0.350 to 0.476. For skipjack tuna, this was increased from 0.350 to 0.594.

e) Again in Ecosim, the standardised CPUE time series for small pelagics, estimated in chapter 5, was incorporated in the fitting process. This was the only time series data that was changed out of a total of 23 time series on catches and CPUE, not considering effort data.

f) Following the same strategy adopted in Chapter 4, emphasis was placed on obtaining reasonable fits to catches. Two overall trends in fishing effort for industrial and artisanal fisheries were used as in Chapter 4. However, specific trends in relative fishing rate (or mortality) were used for yellowfin and skipjack tuna as well as for small pelagics in order to improve further the respective fits to catch series. It is important to point out that these were relatively small adjustments except for small pelagics, where fishing mortality was increased from 0.06 to 0.33 at its peak, thus increasing by a factor of six (Figure 7.6). Again, this was based on the results of biomass dynamic modelling of small pelagics (Chapter 5).

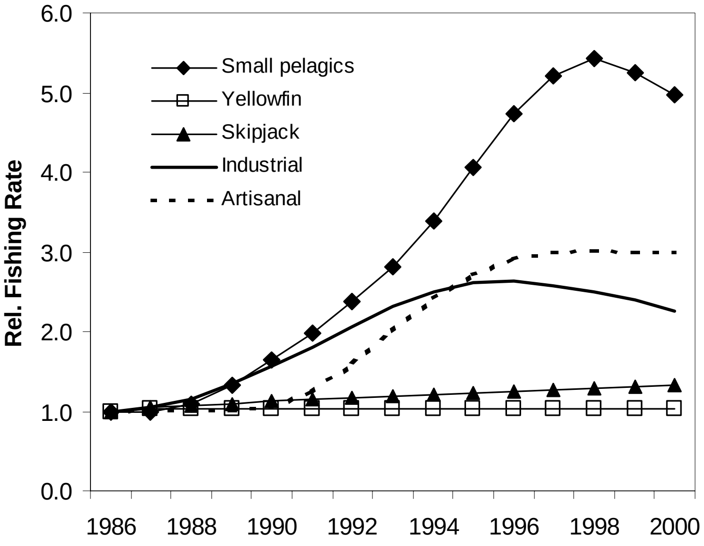

Figure 7.6. Trends in fishing effort, expressed as relative fishing rate, applied in the simulation study using the revised model. Relative fishing rate expresses the increase in fishing mortality, relative to the base model, applied to specific species groups or to the species targeted by the whole fishery.

We adopted a strategy of applying as few changes as possible in the model developed in Chapter 4. The objective was to obtain a decrease in small pelagic biomass and to adjust fishing mortality in conformity with the results of Chapter 5. The diets of its predators were adjusted in accordance, including what appears to be a more reasonable diet for small tuna. Secondary objectives were to improve the fit to catches, which was obtained adjusting relative fishing rate (mortality) of yellowfin and skipjack tuna. However, these estimated fishing mortalities for tuna should not be considered generally applicable values, as they are part of the approach used in treating tuna as external factors. These fishing mortality values are simply consequences of the other assumptions on biomass, production and consumption in order to obtain a fit to observed catches. The overall fit to catches improved as a result of these relatively few adjustments (Figure 7.7). Note the higher variability of observed catches in the beginning of the period, showing what appears to be increasing precision in estimating catches over time. This has to do with the continuous improvement of the sampling programme in terms of area coverage and sampling intensity (Chapter 2).

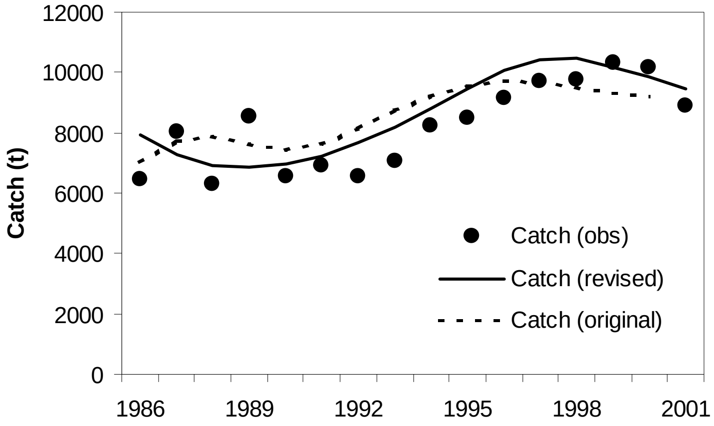

Figure 7.7. Observed and simulated catches using Ecosim. Original refers to the simulation study in Chapter 4 and revised refers to the simulation after the adjustments explained in the text.

These relatively few adjustments that were applied to the simulation model had a wide range of implications for specific species groups (Table 7.3). For example, total fish biomass decreased by 10 percent after the above adjustments compared to 2 percent in the original model (Chapter 4). Catches increased by 19 percent compared to 35 percent. Neritic fish biomass decreased by 1 percent compared to an increase of 13 percent. Small tuna biomass increased by 16 percent compared to 58 percent. However, the wide-ranging implications should not be considered surprising as we are modelling a complex food web. Also, the adjustments were made to key species in terms of their importance as predators, prey and/or fishery catches as well as in terms of biomass. The adjustments were relatively few, obtaining the desired changes, precisely because these were applied to key species.

Table 7.3. Summary of results of the simulation in terms of biomass and catch changes over the period from 1986 to 2000. The biomass (B) and catch (C) values are given at the start (S; t/km^2) of the simulation and as the ratio end/start (E/S). Original model refers to the simulation study in Chapter 4 and revised model refers to the simulation results after the adjustments explained in this chapter. (see Appendix I and Table 4.1 for more detail)

	Revised Model				**Original Model**			
	B (S)	**B (E/S)**	**C (S)**	**C (E/S)**	**B (S)**	**B (E/S)**	**C (S)**	**C (E/S)**
Total fish	15.39	0.90	1.51	1.19	16.39	0.98	1.24	1.35
Tuna	2.17	0.44	1.11	0.46	2.25	0.30	0.85	0.57
Wahoo	0.19	0.65	0.05	1.30	0.19	0.90	0.05	1.71
Neritic fish	12.14	0.99	0.34	3.51	13.07	1.13	0.34	3.27
Small pelagics	3.70	0.80	0.23	3.89	4.64	1.13	0.22	3.37
Small tuna	0.49	1.16	0.03	3.45	0.49	1.58	0.03	4.69
All demersal	7.70	1.07	0.07	2.36	7.70	1.10	0.07	2.44
Demersal predators	0.22	0.54	0.03	1.61	0.22	0.56	0.03	1.70
Moray eels	0.15	0.66	0.01	2.00	0.15	0.64	0.01	1.89

Major Sources of Uncertainty

We have placed emphasis on model fitting using observed time series data, catches in particular. Discussions on the choice of basic parameters and the ecological basis for the definition of groups and their diet have generally been avoided in this study. Other issues that have been completely ignored are feeding type responses, predatory/prey control (*e.g.* top-down, bottom-up, and wasp-waist control), and prey switching. This comes from the recognition of the fact that these effects are not well known even in well-studies areas (Bax, 1998; Cury *et al.*, 2000; Hollowed *et al.*, 2000; Rochet and Trenkel, 2003; Christensen and Walters, 2004). An emphasis on model fitting was considered more constructive, making adjustments that seem reasonable and identifying key research issues. Ecosystem models in general tend to be over-parameterised in the sense that many parameter combinations explain the observed data equally well (Cox *et al.*, 2002). This is also the case here and we do not pretend to have developed a perfect model in the sense that it can be used for specific yield predictions.

One of the major sources of uncertainty in the ecosystem model is related to migratory species, tuna in particular. Ideally, the model of Cape Verde should be linked to an Atlantic Ocean model, so that tuna groups can be modelled explicitly. This has been done in a recent study, where Ecopath models were created for oceanic ecosystems and linked to coastal models, including Cape Verde (Christensen *et al.*, 2004). Biomass, catches and effort were simulated over time from 1950 to 1999, using available time series data in the fitting process, and considering spatial effects. The approach used in spatial modelling was to disaggregate the area off Northwest Africa into cells (½ degree spatial grid) and distribute fish biomass (except small pelagics) estimates from available models according to habitat-defining factors; year, depth, primary production, distance from coast, upwelling index, fish catch and composition. Most ecopath models assume homogeneous distribution of biomass within the modelled area, which makes it essential to distribute biomass more realistically when considering large areas. The factors used are considered to be relatively simple but effective in determining habitat for specific groups (*e.g.* demersals, large pelagics, flatfish, sharks/rays, etc.), as well as temporal variability (year). This study was particularly useful in providing global indications on the evolution of biomass, catches, and fishing rate in the region, which showed strong over-exploitation of demersal fish resources, in particular.

On the other hand, this study by Christensen *et al.* (2004) was not particularly useful with regards to the Cape Verde coastal ecosystem. First of all, the Cape Verde model included was preliminary (Chapter 3) and has since been improved and modified considerably. But most importantly, Christensen *et al.* (2004) did not attempt to model tuna movement, using instead a rule-based distribution of fish biomass. This approach appears to be a reasonable approximation at the larger scale, but it may be highly biased at the smaller scale as it does not consider the dynamics of movement, concentration and residence of tuna. Such studies on the movement and distribution of tuna in the Atlantic are in course, following examples from the Pacific (Lehodey, 2001; Lehodey *et al.*, 2003), which may become available in the near future.

This study included preliminary attempts to model the Cape Verde coastal ecosystem by creating separate models for each island. This was in fact a similar approach to Christensen *et al.* (2004) at a finer spatial resolution. However, these attempts were abandoned because of the quality of fisheries data. Fishermen from Santiago and Sal Islands are known to exploit the more extensive Boavista-Maio shelf system, but these catches are not classified as coming from this shelf system. This is a weak point in the sampling programme, the location of catches is not specified, which makes spatial modelling difficult and adds more uncertainty. Moreover, a large proportion of neritic fish biomass is found on the Boavista-Maio shelf because of its dimension (≈ 70%; Chapter 2), but the lack of reliable fisheries data makes the fitting and validation highly uncertain.

We treated tuna as external fators, obtaining relatively good fits to both CPUE and catch series from 1986 to 2000. This may appear to be sufficient, but the amount of tuna in the system (biomass) and the time spent in the system will have overriding consequences on predation in the system. Equally good fits could also have been obtained by decreasing biomass of tuna and decreasing their import of food (or vice versa), but the consequences in terms of predation in the system are different.

Another major source of uncertainty is the diet composition assumed with particular relevance in the case of important predators such as tuna. A local food composition study was available only for mackerel scad (*Decapterus macarellus*) (Almada, 1997), but this species is not an important predator on other fish. Christensen and Walters (2004) define one of the major pitfalls in ecosystem modelling as, "incorrect assessments of predation impacts for prey that are rare in predator diets." Local studies on the prey of yellowfin and skipjack tuna in Cape Verde are essential to assess this possible bias. The simulated increase of biomass of demersal groups was a response to lesser predation by tuna, even though they were assumed to constitute a minor (or rare) prey in tuna diet (Appendix F).

The role of small pelagics, especially mackerel scad (*Decapterus macarellus*), as relatively abundant forage fish raises the question of its role in the system. A possible analogy of a tropical "wasp-waist" ecosystem (Rice, 1995 in Bakun, 1996; Cury *et al.*, 2000) may exist. The predation on small pelagics by demersal groups may have been underestimated in this study and again, this should be addressed with studies of food composition.

Projections with the Ecosystem Model

Bearing in mind the uncertainty involved, the revised Ecopath model was projected forward to determine the possible direct and indirect effects of increasing fishing pressure. The chosen scenario involved gradually increasing relative fishing rate over a 15-year period from 2000 to 2015. Artisanal relative fishing rate was increased gradually by 50 percent from the level in 2000 over 15 years, simulating a steadily increasing efficiency of the artisanal fleet (cf. Figure 7.6). Industrial relative fishing rate in 2000 was doubled over the same 15-year period, considering the recent acquisition of modern, larger fishing vessels (26 m) by the Government of Cape Verde. There are plans of acquiring 10 fishing vessels, capable of fishing with pole and line, purse seine, and lobster traps, five of which are already operational in Cape Verde (www.indp.cv). Fishing rates on all migratory species were maintained constant, thus assuming that external factors are maintained constant (tuna) over the period. Details on the results in terms of biomass and catch of important groups are presented in Appendix J, which gave a 9 percent increase in artisanal catches and 30 percent increase in industrial catches. Total catch increased by only 16 percent in spite of the strong increase in fishing pressure. This should not be regarded as a prediction, but it indicates that a strong increase in fishing pressure will most likely result in a much lower corresponding increase in catches. Simulation results, both in Chapter 4 and in the present chapter, indicate that this is most likely. These results support the notion that fisheries in Cape Verde have passed a developing phase and that management is now essential, instead of increasing catches substantially (*e.g.* Bowsma, 2003-c). It is in this context that the Ecopath model may prove to be particularly useful for policy exploration in conjunction with adaptive management (*e.g.* Sainsbury, 1998; Christensen and Pauly, 2004)

7.3 Priorities for Future Research

We believe that this study has been successful in taking the first steps towards an ecosystem approach to assess the direct and indirect effects of fishing in Cape Verde. Several possible improvements have been mentioned in the previous section. Many important shortcomings have been identified, regarding theoretical aspects and practical implementation, but it is important to point out that these are associated with ecosystem and multispecies models in general. The Ecopath tool can nevertheless serve as an important strategic tool for policy exploration and assess the likely effects of specific management actions in an ecosystem context. The Ecopath approach is a sound methodology for assembling and exploring data on aquatic food webs, thus identifying possible inconsistencies and key issues for research. Efforts should be made at improving the existing model, as new data become available, including local studies and theoretical developments. Valuable insight on the structure and functioning of the ecosystem can be gained. This tool should however be considered a complementary tool to conventional assessment approaches and other alternative methods. An interesting alternative method is the simplified approach of the multispecies production model (Sullivan, 1991; Horbowy, 1996; Collie and Delong, 1999), which can be made more efficient in parameter estimation by using meta-analytic or demographic approaches to reduce the number of unknown parameters.

A considerable part of this study consisted in compiling existing information on the coastal ecosystem in Cape Verde. This includes data on biomass estimates from trawl surveys, trophic relations, and biological parameters for important fish species, which have been made available and can be used for further research. We have contributed to a better understanding of ecosystem dynamics in Cape Verde, including the first attempts of quantifying change in an ecosystem context as well as providing indications for management. Special attention was given to assessing small pelagics because this group constitutes around 75 percent of total neritic catches. Tuna in contrast should be assessed on much a larger scale in the context of ICCAT.

Biomass dynamic modelling was applied to assess small pelagic resources. This method has been criticised for a number of reasons, but present day computing power and developments in theory have resulted in a "come-back" for this method. Note that the effects of growth, mortality and reproduction are pooled, which may lead to problems in some cases. For example, this type of model cannot capture fluctuations in recruitment success, since the effects of standing stock biomass and recruitment are confounded (Haddon, 2001; Pope *et al.*, 2001). However, many of the past failures are now considered to have been an indiscriminate application of method, which did not consider possible data failures such as the lack of contrast in catch and effort data. Recent theoretical advances have shown several ways of dealing with this, including the approach used in this study of a modified bootstrap method incorporating a known probability distribution of r, the intrinsic rate of increase. There are other approaches ranging from simple to elaborate methods that can be used to assess resources or assist in determining key biological parameters in data-poor situations. A few examples are; the relative rate of catch increase for catch data without effort data (Gaertner *et al.*, 2001); yield indicators from biomass (Die and Caddy, 1997); assuming equilibrium conditions as an approximation (Caddy, 1996); demographic analysis (Cortés, 1998; McAllister *et al.*, 2001); and meta-analytic approaches (Hilborn and Liermann, 1998; Myers and Mertz, 1998). Of course, these methods can be combined with other assessment methods, thus becoming powerful modern tools. Priority should be given to assess demersal species such as the grouper, *Cephalopholis taeniops*, and moray eels (bulk biomass) in Cape Verde, using such approaches or a combination thereof on available data (*e.g.* specific sampling programmes of the INDP).

Complex modelling efforts, involving intensive data sampling, do not necessarily bring about greater predictive power. For example, despite the huge efforts put into the development of the MSVPA model for the North Sea, it has become clear that there is no particular need to take multispecies effects into account (except by adjusting average natural mortality rates) when short-term management advice is provided (Pope, 1991; Magnússon, 1995; Rochet and Trenkel, 2003). Instead, an effective monitoring system may provide sufficient data on which to base a robust management system. Research on ecosystem indicators that are useful for monitoring and assessing fishing effects appear to be of special relevance for countries such as Cape Verde as these methods are relatively cost-effective. Most likely, it will be necessary to define and use a set of variables or conjugate indicators as no single indicator is expected to perform sufficiently well to monitor whole ecosystems (*e.g.* Link *et al.*, 2002; Rochet and Trenkel, 2003), but also because tropical ecosystems appear to be more resilient to exploitation under certain circumstances (Stobberup *et al.*, 2005), making the choice of indicators crucial.

Ecosystem and multispecies models may not be necessary for defining management measures and these tools are often criticised for not being able to do so within reasonable ranges of uncertainty. However, these tools are particularly relevant when dealing with broader issues such as the management of biodiversity (Pope *et al.*, 2001). Another issue of growing importance is the establishment of marine protected areas (MPA) as tools for fisheries management with calls for protecting 20 to 30 percent of the oceans (Hilborn *et al.*, 2004). These issues that are becoming important in Cape Verde as well with the recent start of a large "biodiversity" Dutch-funded project that also deals with the possible definition of MPAs as part of an integrated coastal zone management approach. In this setting, the conventional single-species approach will invariably fail, as the available tools were not designed to evaluate these types of issues.

Several research priorities have been suggested and recommended, but highest priority should be given to the re-analysis of raw catch and effort data. The Fisheries Institute (INDP) is making efforts to recover these data in raw form, which began in context of the "Fisheries Information and Analysis System" (2000 - 2002). One of the benefits may be the definition of fleet/gear components using multivariate techniques, which will be useful for obtaining more reliable CPUE time series (*e.g.* Silva *et al.*, 2002). Robust methods such biomass dynamic models or others can then be applied to specific species or species groups, based on these improved CPUE series. Other important issues are the estimation of increases in efficiency in the fisheries and ways of improving effort measurements. The INDP is also attempting to identify the location of fishing grounds in order to determine the location of fishery catches, which will be an important step towards spatial considerations. Location of catches has been in many cases recorded as place names, but this information has not been utilised (or registered in the databases) previously because their location was not known.

We believe that this study has been an important contribution to improve on the present understanding of the Cape Verde coastal ecosystem, its form and functioning. In the process, special attention has been given to the sustainable management of marine resources, bearing in mind the crucial importance of fisheries in Cape Verde. However, we are well aware of the fact that these are the first few steps in the long process of introducing and implementing the Ecosystem Approach to Fisheries (EAF). This may well result in a system of simple control and decision rules in a robust management system based on effort control primarily (*e.g.* Punt, 1995; Beverton, 1998; Butterworth and Punt, 1999; Sainsbury *et al.*, 1999; Caddy, 2002; Hilborn, 2003), but in order to get there, we believe that contributions such as ours are important. Moreover, EAF should also consider a wide range of social and economical aspects, including the effects to food security, revenue, employment and local development, which are essential for defining management strategies and policy. Thus, the importance of concerted action involving a wide range of disciplines and interest groups.

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9. Appendices

Appendix A. Catches (tonnes) by species in Cape Verde

	2000	**1990-2000 Average**
Decapterus macarellus	1864	2411
Thunnus albacares	1775	1770
Spicara melanurus	1195	386
Selar crumenophthalmus	1152	505
Katsuwonus pelamis	785	946
Diversos	681	428
Acanthocybium solandri	501	450
Euthynnus alletteratus	495	183
Cephalopholis taeniops	361	221
Muraena helena	157	112
Pseudupeneus prayensis	131	86
Diplodus prayensis	109	84
Seriola dumerili	87	57
Auxis thazard thazard	82	56
Coryphaena hippurus	66	59
Fodiator acutus	65	53
Sardinella maderensis	65	33
Lethrinus atlanticus	57	31
Priacanthus arenatus	54	42
Decapterus punctatus	52	41
Lutjanus agennes	50	37
Scarus hoefleri	46	40
Apsilus fuscus	45	33
Caranx crysos	41	33
Lobsters	37	58
Galeoides decadactylus	33	18
Caranx lugubris	26	31
Bodianus scrofa	23	23
Acanthurus monroviae	21	14
Selene dorsalis	19	17
Virididentex acromegalus	14	14
Mycteroperca fusca	11	15
Serranus cabrilla	10	11
Scorpaena sp.	10	30
Demersal sharks	9	6
Sphyraena guachancho	8	11
Pomadasys incisus	4	15
Thunnus obesus	2	63
Grand Total	10145	8448

Appendix B. Dominant species determined from surveys

Ten most important species in trawl and handline surveys, given in proportion of trawl catches.

Trawl surveys (1964 – 1994)	**Prop.**	**Commercial fishing (1994 – 1997)**	**Prop.**	**Handline surveys (1995 - 1997)**
Antigonia capros	0.15	*Diplodus sp.*	0.26	*Spondyliosoma cantharus*
Decapterus punctatus	0.14	*Pomadasys incisus*	0.22	*Cephalopholis taeniops*
Pagellus acarne	0.09	*Lithognathus mormyrus*	0.13	*Seriola dumerili*
Lithognathus mormyrus	0.08	*Galeoides decadactylus*	0.09	*Parapristipoma humile*
Priacanthus arenatus	0.07	*Pseudupeneus prayensis*	0.09	*Apsilus fuscus*
Acanthurus monroviae	0.06	*Selar crumenophthalmus*	0.08	*Lutjanus agennes*
Pseudupeneus prayensis	0.05	*Priacanthus arenatus*	0.06	*Serranus cabrilla*
Mustelus mustelus	0.03	*Albula vulpes*	0.02	*Mustelus sp.*
Chromis sp.	0.03	*Boops boops*	0.02	*Epinephelus aenus*
Boops boops	0.03	*Selene dorsalis*	0.01	*Muraenidae*

Appendix C. List of fish species by ecological group

List of fish species by ecological group defined in the model, including estimates for trophic level (TL), L_∞ , W_∞ , Q/B, and the diet references used, as explained in the text. An empty cell indicates that no information was available. Diet references with an asterisk indicate that information referring to a different species, but the same Genus, was used. (see Appendix D for complete fish diet references)

Groups	Family	Species	TL	Linf	Winf	Q/B	Diet references
Bathydemersal	Berycidae	*Beryx decadactylus*	4.2	103	15143	2.500	Morato *et al.* 1998
	Berycidae	*Beryx splendens*	4.2	69	7522	2.900	Dubochkin and Kotlyar 1989; Morato *et al.* 1998
	Polymixiidae	*Polymixia nobilis*		50	1243	4.200	
	Scorpaenidae	*Neomerinthe folgori*		42	725	4.700	
	Scorpaenidae	*Pontinus kuhlii*	4.1	54	1575	4.000	
	Scorpaenidae	*Scorpaena scrofa*	4.3	52	2303	3.700	Bradai and Bouain 1990; Harmelin-Vivien *et al.* 1989
	Scorpaenidae	*Scorpaena sp.*	3.9	52	2303	3.700	Randall 1967*
	Sebastidae	*Helicolenus dactylopterus dactylopterus*	4.4	42	1128	5.600	Meyer and Smale 1991-a; Rountree 1999
	Trachichthyidae	*Gephyroberyx darwinii*	3.8	62	4235	3.300	
Billfish	Xiphiidae	*Xiphias gladius*	4.6	213	130000	6.400	Scott and Tibbo 1968; Stillwell and Kohler 1985
Demersal fish	Balistidae	*Balistes carolinensis*	3.5	60	4174	4.800	
	Haemulidae	*Parapristipoma humile*		39	901	7.200	
	Haemulidae	*Pomadasys incisus*	3.5	52	1716	5.900	Caverivière 1989
	Mullidae	*Pseudupeneus prayensis*	3.5	57	3370	5.100	Randall 1967*
	Polynemidae	*Galeoides decadactylus*	3.6	48	1938	6.200	Caverivière 1989
	Priacanthidae	*Priacanthus arenatus*	3.7	42	999	6.500	Randall 1967; Rountree 1999
	Sparidae	*Dentex macrophthalmus*	3.4	59	4248	5.100	Domanevskaya and Patokina 1984
	Sparidae	*Pagellus acarne*	3.3	30	581	6.900	Domanevskaya and Patokina 1984; Morato *et al.* 1998
	Sparidae	*Pagellus bellottii bellottii*	3.6	40	851	6.400	Caverivière 1989
	Sparidae	*Pagrus pagrus*	3.9	65	5420	4.400	Papaconstantinou and Caragitsou 1989
Demersal predators	Congridae	*Paraconger notialis*		65	2746	3.600	
	Fistulariidae	*Fistularia petimba*	3.7	203	7309	3.800	Rountree 1999
	Lethrinidae	*Lethrinus atlanticus*	3.5	52	2292	5.100	
	Lutjanidae	*Apsilus fuscus*	4.2	64	3869	8.600	
	Lutjanidae	*Lutjanus agennes*		78	8737	4.200	Randall 1967*
	Lutjanidae	*Lutjanus fulgens*	4	62	6643	4.100	Randall 1967*
	Serranidae	*Cephalopholis taeniops*	4	72	6524	3.600	Randall 1967*
	Serranidae	*Epinephelus goreensis*		143	48000	5.500	
	Serranidae	*Epinephelus marginatus*	3.9	114	30000	2.700	Derbal and Kara 1996
	Serranidae	*Mycteroperca fusca*	4.5	103	10864	3.700	Randall 1967*

Groups	Family	Species	TL	Linf	Winf	Q/B	Diet references
Demersal predators	Serranidae	*Serranus atricauda*		37	490	7.100	Morato *et al.* 2000
	Serranidae	*Serranus cabrilla*	4.3	42	550	6.900	Labropoulou and Eleftheriou 1997
	Sparidae	*Virididentex acromegalus*		54	3055	5.500	
Demersal sharks	Carcharhinidae	*Rhizoprionodon acutus*	4.1	178	45000	2.800	Cortés 1999; Salini *et al.* 1994
	Centrophoridae	*Centrophorus granulosus*	4.1	163	43547	2.800	Cortés 1999
	Centrophoridae	*Centrophorus uyato*	4.5	113	14429	3.500	
	Ginglymostomatidae	*Ginglymostoma cirratum*	3.8	432	594000	1.400	Cortés 1999
	Triakidae	*Mustelus mustelus*	3.8	203	84150	2.500	Caverivière 1989; Cortés 1999; Smale and Compagno 1997
Flatfish	Bothidae	*Bothus podas*	3.4	47	1151	5.600	Schintu *et al.* 1994
	Citharidae	*Citharus linguatula*	4	26	177	8.200	Belghyti *et al.* 1993
	Paralichthyidae	*Syacium micrurum*	3.3	42	725	6.100	Longhurst 1960
	Soleidae	*Dicologlossa cuneata*	3.3	25	123	8.800	Claude 1979
Flyingfish	Exocoetidae	*Fodiator acutus*	2.9	16	40	14.300	
Herbivores	Acanthuridae	*Acanthurus monroviae*	2.4	47	2744	15.900	Randall 1967*
	Monacanthidae	*Aluterus schoepfii*	2	63	1742	17.400	Randall 1967
	Pomacentridae	*Abudefduf saxatilis*		24	316	20.500	Randall 1967
	Pomacentridae	*Chromis sp.*		13	21	16.200	Randall 1967*
	Scaridae	*Scarus hoefleri*		62	2406	16.300	Randall 1967*
	Scaridae	*Sparisoma rubripinne*	2	51	2498	15.300	Randall 1967
Jacks	Carangidae	*Alectis ciliaris*	4.2	153	90000	3.300	Caverivière 1989
	Carangidae	*Caranx crysos*	4.4	72	6414	9.800	Randall 1967; Rountree 1999; Sierra *et al.* 1994
	Carangidae	*Caranx lugubris*	4.5	98	18636	7.900	Randall 1967
	Carangidae	*Lichia amia*	4.5	203	93000	5.700	Bennett 1989
	Carangidae	*Selene dorsalis*	4	39	823	11.700	
	Carangidae	*Seriola carpenteri*	4.5	57	1862	8.600	
	Carangidae	*Seriola dumerili*	4.5	158	57000	4.300	Randall 1967; Rountree 1999; Sierra *et al.* 1994
	Phycidae	*Phycis phycis*	4.1	65	2784	4.700	Morato *et al.* 1998; Papaconstantinou and Caragitsou 1989
	Sphyraenidae	*Sphyraena guachancho*	3.9	203	65000	3.500	Sierra *et al.* 1994
Large tuna	Scombridae	*Katsuwonus pelamis*	4.4	79	10468	12.500	Roger 1994; Sierra *et al.* 1994
	Scombridae	*Thunnus albacares*	4.5	207	267000	11.640	Fonteneau and Marcille 1993; Maldeniya 1996; Roger 1994
	Scombridae	*Thunnus obesus*	4.5	236	284000	5.900	Fonteneau and Marcille 1993
Moray eels	Muraenidae	*Gymnothorax polygonius*		72	1410	5.300	
	Muraenidae	*Gymnothorax vicinus*	4.3	125	6885	3.900	Randall 1967
	Muraenidae	*Muraena helena*	4.2	153	36027	2.800	
O Demersal fish	Caproidae	*Antigonia capros*	3.6	32	1285	5.300	Rountree 1999
	Dactylopteridae	*Dactylopterus volitans*	3.6	93	10674	3.700	Randall 1967
Pelagic predators	Coryphaenidae	*Coryphaena hippurus*	4.4	234	215000	8.480	Palko *et al.* 1982

Groups	Family	Species	TL	Linf	Winf	Q/B	Diet references
	Scombridae	*Acanthocybium solandri*	4.5	158	26168	10.100	Manooch III and Hogarth 1981
Pelagic sharks	Alopiidae	*Alopias superciliosus*	4.2	472	1767000	1.300	Cortés 1999
	Carcharhinidae	*Carcharhinus brevipinna*	4.2	253	122000	2.300	Cortés 1999
	Carcharhinidae	*Carcharhinus obscurus*	4.2	449	467000	1.700	Cortés 1999; Rountree 1999; Smale 1991
	Carcharhinidae	*Galeocerdo cuvier*	3.8	737	2101000	1.300	Cortés 1999
	Carcharhinidae	*Prionace glauca*	4.1	343	1699000	1.300	Clarke and Stevens 1974; Cortés 1999; Harvey 1989
	Lamnidae	*Isurus oxyrinchus*	4.3	373	489000	9.640	Cortés 1999; Rountree 1999; Stillwell and Kohler 1982
	Sphyrnidae	*Sphyrna lewini*	4.1	321	165000	2.100	Caverivière 1989; Cortés 1999; Rountree 1999
	Sphyrnidae	*Sphyrna zygaena*	4.2	402	551000	1.700	Cortés 1999; Rountree 1999; Smale 1991
Rays	Dasyatidae	*Dasyatis margarita*	3.4	103	10864	3.700	Caverivière 1989
	Rajidae	*Raja miraletus*	3.9	62	1495	5.300	Caverivière 1989; Smale and Cowley 1992
	Rhinobatidae	*Rhinobatos rhinobatos*	3.6	103	10864	3.700	Caverivière 1989
Reef feeders	Balistidae	*Balistes vetula*	3.4	60	5340	4.500	Randall 1967
	Holocentridae	*Myripristis jacobus*	3.2	26	757	6.100	Randall 1967*
	Holocentridae	*Sargocentron hastatus*		26	379	7.000	Randall 1967*
	Labridae	*Bodianus scrofa*		45	899	5.700	Randall 1967*
	Labridae	*Bodianus speciosus*		52	3178	4.400	Randall 1967
Small pelagics	Carangidae	*Caranx rhonchus*	3.6	49	1688	12.200	Caverivière 1989
	Carangidae	*Decapterus macarellus*	3.4	42	1529	11.400	Almada 1997; Randall 1967
	Carangidae	*Decapterus punctatus*	3.1	32	331	15.500	Hales 1987; Randall 1967; Rountree 1999
	Carangidae	*Selar crumenophthalmus*	3.9	37	1189	11.600	Randall 1967; Rountree 1999; Sierra *et al.* 1994;Yamashita *et al.* 1987
	Carangidae	*Trachurus sp.*	3.3	62	2982	8.400	
	Centracanthidae	*Spicara melanurus*	3	32	335	8.100	Longhurst 1960*; Meyer and Smale 1991-b*
	Clupeidae	*Sardinella maderensis*	3.1	30	424	12.600	
Small tuna	Scombridae	*Auxis thazard thazard*	4.4	52	2718	16.200	Blaber *et al.* 1990
	Scombridae	*Euthynnus alletteratus*	4.4	115	19771	8.000	Randall 1967
Sparids	Sparidae	*Boops boops*	3	36	453	16.500	Anato and Ktari 1983
	Sparidae	*Diplodus fasciatus*		42	1417	15.200	Randall 1967*
	Sparidae	*Diplodus prayensis*	2.7	29	579	18.200	Randall 1967*
	Sparidae	*Diplodus sargus lineatus*	2.8	29	660	17.800	Randall 1967*
	Sparidae	*Lithognathus mormyrus*	3.4	40	986	16.000	
	Sparidae	*Spondyliosoma cantharus*	3.2	52	1399	15.200	Gonçalves and Erzini 1998

Appendix D. Diet References

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Appendix E. Diet matrix referring to Chapter 3

Diet matrix showing proportions by prey item (Chapter 3). Predators are arranged along the horizontal axis and prey items down the vertical axis. Import indicates the proportion of the diet that is a result of foraging outside the system.

Group		**Predator group**														
No.	**Prey**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**
1	Sea birds									0.013						
2	Mammals									0.006						
3	Billfish									0.150						
4	Large Tuna	0.006	0.100	0.150	0.004	0.170				0.052						
5	Pelagic predators	0.003	0.070	0.063						0.010						
6	Small tuna	0.030	0.062	0.100	0.013	0.040				0.010						
7	Jacks	0.010	0.060		0.005	0.029	0.010			0.060			0.005			
8	Moray eels											0.080				
9	Pelagic sharks									0.010						
10	Bathydemersal								0.100			0.030	0.026			
11	Demersal sharks									0.258						
12	Demersal predators	0.005	0.010						0.042	0.040		0.050				
13	Rays											0.020				
14	Other Demersal fish				0.010	0.040			0.131		0.215	0.050	0.140	0.070		0.050
15	Demersal fish	0.010	0.060		0.010	0.020			0.132	0.020	0.140	0.050	0.170	0.050		
16	Flatfish											0.001		0.001		
17	Small pelagics	0.375	0.263	0.247	0.310	0.360	0.176	0.203		0.002						0.030
18	Reef feeders							0.010	0.050		0.012	0.010	0.050			0.006
19	Sparids	0.065	0.002		0.016	0.031		0.020	0.069			0.010	0.175			0.007
20	Flyingfish	0.010	0.011	0.050	0.005	0.020	0.082									
21	Herbivores	0.005	0.002						0.100			0.030	0.030			0.029
22	Turtles									0.037						
23	Crustaceans				0.065		0.027	0.025	0.250	0.041	0.443	0.345	0.154	0.403	0.316	0.417
24	Echinoderms										0.001	0.130	0.026	0.180		0.063
25	Molluscs/Worms				0.042		0.038	0.155	0.125	0.001	0.016	0.017	0.118	0.172	0.270	0.218
26	Heterotrophic benthos				0.010		0.100	0.011		0.001	0.038	0.141	0.010	0.003	0.170	0.024
27	Microfauna													0.020	0.003	0.005
28	Zooplankton	0.141		0.050	0.180	0.140	0.447	0.550		0.042	0.135	0.003	0.002	0.051	0.041	0.102
29	Phytoplankton															
30	Benthic autotrophs											0.001	0.094			0.048
31	Detritus											0.032		0.050	0.200	
	Import	0.340	0.360	0.340	0.330	0.150	0.120	0.026		0.246						
	Sum	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

Appendix E (continued). Diet matrix showing proportions by prey item (Chapter 3). Predators are arranged along the horizontal axis and prey items down the vertical axis. Import indicates the proportion of the diet that is a result of foraging outside the system.

Group		Predator group												
No.	**Prey**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**
1	Sea birds													
2	Mammals													
3	Billfish													
4	Large Tuna													
5	Pelagic predators													
6	Small tuna													
7	Jacks													
8	Moray eels													
9	Pelagic sharks													
10	Bathydemersal													
11	Demersal sharks													
12	Demersal predators													
13	Rays													
14	Other Demersal fish	0.004												
15	Demersal fish	0.040												
16	Flatfish													
17	Small pelagics													
18	Reef feeders				0.001									
19	Sparids				0.003									
20	Flyingfish													
21	Herbivores				0.004									
22	Turtles													
23	Crustaceans	0.294	0.008	0.284	0.288		0.001	0.125						
24	Echinoderms	0.014		0.371				0.040	0.120					
25	Molluscs/Worms	0.318	0.001	0.117	0.225		0.001	0.045	0.320	0.096				
26	Heterotrophic benthos	0.004	0.003	0.001				0.400	0.070	0.220	0.059			
27	Microfauna	0.017		0.009					0.050	0.022	0.170	0.019		
28	Zooplankton	0.259	0.938	0.216		0.500			0.101	0.003	0.046	0.025		
29	Phytoplankton								0.005	0.003	0.121	0.011		0.900
30	Benthic autotrophs			0.002	0.479		0.996	0.390	0.200	0.554	0.179	0.084		
31	Detritus	0.050	0.050				0.002		0.134	0.102	0.425	0.861	1.000	0.100
	Import					0.500								
	Sum	1	1	1	1	1	1	1	1	1	1	1	1	1

Appendix F. Diet matrix referring to Chapter 4

Diet matrix showing proportions in the diet for each prey item (Chapter 4). Predators are arranged along the horizontal axis and prey items down the vertical axis. Import indicates the proportion of the diet that was assumed to be imported, the result of foraging outside the system. Only the 20 higher trophic level groups are shown in order to simplify the table.

	Prey \ Predator	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
1	Sea birds										0.013										
2	Mammals										0.006										
3	Billfish										0.150										
4	Yellowfin	0.006	0.100	0.150	0.021						0.052										
5	Predators pel	0.003	0.070	0.120							0.010										
6	Skipjack		0.040	0.040	0.010																
7	Small tuna	0.030	0.062	0.100	0.020	0.040					0.010										
8	Jacks	0.010	0.020			0.029					0.060			0.005							
9	Moray eels												0.080								
10	Sharks pel										0.010										
11	Bathydemersal									0.100			0.030	0.026							
12	Sharks dem										0.258										
13	Predators dem	0.005	0.010		0.001					0.042	0.040		0.050								
14	O Demersal fish				0.015	0.040			0.003	0.131		0.215	0.050	0.140		0.050					
15	Demersal fish	0.010	0.060		0.006	0.020			0.002	0.132	0.020	0.140	0.050	0.170							
16	Small pelagics	0.200	0.063	0.050	0.095	0.100	0.100	0.170	0.050		0.002					0.030					
17	Reef feeders								0.200	0.050		0.012	0.010	0.050		0.006			0.001		
18	Sparids	0.065	0.002		0.010	0.031			0.120	0.065			0.010	0.175		0.007			0.003		
19	Flyingfish	0.010	0.011	0.050	0.005	0.020		0.105													
20	Herbivores	0.005	0.002		0.001					0.100			0.030	0.030		0.029			0.004		
21	Turtles										0.037										
22	Crabs/Shrimps				0.046		0.150	0.130	0.125	0.250	0.041	0.443	0.345	0.154	0.320	0.417	0.008	0.284	0.292		0.002
23	Benthos				0.030		0.150	0.100	0.150	0.130	0.002	0.055	0.300	0.200	0.350	0.311	0.004	0.350	0.200		
24	Zooplankton	0.141		0.050	0.120	0.140	0.100	0.345	0.350		0.042	0.135	0.003		0.130	0.102	0.938	0.316		0.500	
25	Phytoplankton																				
26	Benthic autotrophs												0.010	0.050		0.048		0.050	0.500		0.996
27	Detritus												0.032		0.200		0.050				0.002
	Import	0.515	0.560	0.440	0.620	0.580	0.500	0.150			0.246									0.500	

Appendix G. Species considered in each of the eight families (BVSTEP)

Family	Species
Acanthuridae	*Acanthurus monroviae*
Caproidae	*Antigonia capros*
Carangidae	*Carangidae* *Pseudocaranx dentex* *Selene dorsalis* *Seriola carpenteri* *Seriola dumerili* *Decapterus punctatus* *Caranx rhonchus* *Trachurus sp.* *Seriola sp.*
Haemulidae	*Parapristipoma humile* *Pomadasys incisus* *Pomadasys sp.* *Pomadasys rogerii*
Mullidae	*Pseudupeneus prayensis*
Pomacentridae	*Abudefduf sp.* *Chromis sp.* *Pomacentridae*
Priacanthidae	*Priacanthus arenatus*
Sparidae	*Boops boops* *Dentex macrophthalmus* *Dentex sp.* *Diplodus fasciatus* *Diplodus prayensis* *Diplodus sargus lineatus* *Diplodus vulgaris* *Lithognathus mormyrus* *Pagellus acarne* *Spondyliosoma cantharus* *Viridentex acromegalus* *Pagellus bellottii bellottii* *Diplodus bellottii* *Diplodus sargus sargus* *Diplodus sp.* *Pagrus auriga* *Diplodus puntazzo*

Appendix H. SIMPER results

SIMPER results showing typical and discriminator families among shallow and deep assemblages. Average similarity is based on the Bray-Curtis coefficient computed between samples, while dissimilarity is computed between depth groups (Shallow and Deep) (square-root transformed standardised catches). Also given by family are average abundance (non-transformed), average similarity/dissimilarity (contribution to overall average), contribution in %, and cumulative %.

Group Shallow	**Average similarity: 21.26**					
Typical	**Species**	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**	
	Sparidae	4494	5.17	24.3	24.3	
	Mullidae	1028	3.91	18.4	42.7	
	Priacanthidae	1498	2.38	11.2	53.9	
	Dactylopteridae	506	1.97	9.3	63.2	
	Triakidae	678	1.85	8.7	71.9	
	Haemulidae	942	1.31	6.2	78.1	
	Fistulariidae	600	1.23	5.8	83.9	
	Carangidae	2467	0.71	3.3	87.2	
	Monacanthidae	184	0.50	2.4	89.6	
	Balistidae	141	0.49	2.3	91.9	
Group Deep	**Average similarity: 22.26**					
Typical	**Species**	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**	
	Caproidae	11731	11.34	50.9	50.9	
	Sparidae	3866	6.40	28.7	79.7	
	Priacanthidae	252	1.15	5.2	84.8	
	Dactylopteridae	120	0.67	3.0	87.8	
	Haemulidae	621	0.49	2.2	90.0	
Groups Shallow & Deep	**Average dissimilarity = 86.98**					
		Shallow	**Deep**			
Discriminator	**Species**	**Av.Abund**	**Av.Abund**	**Av.Diss**	**Contrib%**	**Cum.%**
	Caproidae	106	11731	17.47	20.1	20.1
	Sparidae	4494	3866	12.50	14.4	34.5
	Mullidae	1028	31	5.23	6.0	40.5
	Priacanthidae	1498	252	5.17	5.9	46.4
	Carangidae	2467	817	5.10	5.9	52.3
	Haemulidae	942	621	4.80	5.5	57.8
	Triakidae	678	207	4.64	5.3	63.1
	Dactylopteridae	506	120	3.88	4.5	67.6
	Fistulariidae	600	1	3.39	3.9	71.5
	Acanthuridae	1234	20	2.20	2.5	74.0
	Monacanthidae	184	15	1.87	2.2	76.2
	Gempylidae	23	183	1.56	1.8	78.0
	Balistidae	141	0	1.51	1.7	79.7
	Polynemidae	243	0	1.23	1.4	81.1
	Pomacentridae	401	44	1.20	1.4	82.5
	Lethrinidae	177	0	1.06	1.2	83.7
	Scaridae	184	0	1.06	1.2	84.9
	Centriscidae	1	208	0.92	1.1	86.0
	Holocentridae	40	11	0.86	1.0	87.0
	Tetraodontidae	8	27	0.79	0.9	87.9
	Lutjanidae	93	13	0.76	0.9	88.8
	Serranidae	8	43	0.72	0.8	89.6
	Labridae	30	15	0.70	0.8	90.4

Appendix I. Ecosim simulation results

Results of the simulation in terms of biomass and catch changes over the period from 1986 to 2000, using the revised model as explained in chapter 7. All biomass and catch values are given in t/km^2 except for the ratio end/start (E/S).

Group	**Biomass (Start)**	**Biomass (End)**	**Biomass (E/S)**	**Catch (Start)**	**Catch (End)**	**Catch (E/S)**
Sea birds	0.02	0.012	0.59	0.001	0.002	1.77
Mammals	0.034	0.025	0.73	0.001	0.001	1.46
Billfish	0.083	0.048	0.58	0.002	0.002	1.16
Yellowfin	1.744	0.827	0.47	0.847	0.402	0.47
Predators pel	0.186	0.121	0.65	0.047	0.061	1.29
Skipjack	0.43	0.134	0.31	0.262	0.106	0.4
Small tuna	0.494	0.572	1.16	0.029	0.1	3.43
Jacks	0.25	0.261	1.05	0.011	0.034	3.12
Moray eels	0.149	0.098	0.66	0.009	0.018	1.96
Sharks pel	0.045	0.036	0.8	0.001	0.002	1.59
Bathydemersal	0.224	0.306	1.37	0.001	0.004	3.55
Sharks dem	0.137	0.122	0.89	0.001	0.003	2.45
Predators dem	0.216	0.116	0.54	0.033	0.053	1.6
O Demersal fish	1.894	2.046	1.08	0.002	0.006	3.24
Demersal fish	1.456	1.66	1.14	0.013	0.045	3.42
Small pelagics	3.702	2.953	0.8	0.23	0.895	3.9
Reef feeders	0.811	0.849	1.05	0.002	0.006	3.66
Sparids	1.841	2.16	1.17	0.007	0.024	3.23
Flyingfish	0.758	0.671	0.88	0.007	0.019	2.75
Herbivores	0.968	0.913	0.94	0.004	0.011	2.83

Appendix J. Model projections

Results of the projection with the Ecopath model from 2000 to 2015. Simulation results are given in terms of biomass and catch changes, using the revised model as explained in chapter 7. All biomass and catch values are given in t/km^2 except for the ratio end/start (E/S).

	Biomass (Start)	**Biomass (End)**	**Biomass (E/S)**	**Catch (Start)**	**Catch (End)**	**Catch (E/S)**
Billfish	0.047	0.051	1.08	0.002	0.002	1.08
Yellowfin	0.818	0.736	0.9	0.397	0.363	0.91
Predators pel	0.12	0.14	1.16	0.06	0.07	1.16
Skipjack	0.125	0.063	0.5	0.1	0.051	0.51
Small tuna	0.581	0.546	0.94	0.102	0.152	1.49
Jacks	0.269	0.267	0.99	0.036	0.054	1.52
Moray eels	0.094	0.04	0.42	0.017	0.011	0.63
Sharks dem	0.121	0.151	1.24	0.003	0.005	1.84
Predators dem	0.11	0.022	0.2	0.051	0.015	0.3
O Demersal fish	2.035	1.969	0.97	0.006	0.011	1.73
Demersal fish	1.671	1.729	1.04	0.045	0.071	1.58
Small pelagics	2.966	2.757	0.93	0.89	1.152	1.29
Reef feeders	0.844	0.863	1.02	0.006	0.01	1.51
Sparids	2.177	2.354	1.08	0.025	0.042	1.69
Herbivores	0.907	0.88	0.97	0.011	0.016	1.44
Total	12.886	12.567	0.98	1.752	2.026	1.16
Artisanal				1.24	1.358	1.09
Industrial				0.512	0.668	1.3
Total	12.886	12.567	0.98	1.752	2.026	1.16