

***Walking the bridge  
from single- to multi-species approaches  
in  
southern African fisheries management***

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This thesis is dedicated to the memory of my father

GYULA PLAGÁNYI

who always knew I'd get there some day

AND

my husband

PENN LLOYD

who is my constant great love, partner and friend

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# DECLARATION

I declare that this thesis is my own unaided work, both in concept and execution, and that apart from the normal guidance from my supervisor, I have received no assistance except as stated below:

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Finally, several of the chapters have been submitted for publication in journals and hence have benefited from the comments made by journal reviewers. Published and in press papers based on this thesis include the following:

1. Plagányi, É.E. & D.S. Butterworth 2002. Competition with fisheries. *In Encyclopedia of Marine Mammals*, Perrin, W.F., Würsig, B & H.G.M. Thewissen (eds), Academic Press, 268-273.
2. Plagányi, É.E. & D.S. Butterworth in press. Indirect fishery interactions: assessing the feeding-related interactions between marine mammals and fisheries. *In Looking to the Horizon: Future Directions in Marine Mammal Research*, Reynolds III, J.E., Perrin, W.F., Reeves, R.R., Ragen, T.J. and S. Montgomery. John Hopkins University Press.
3. Plagányi, É.E. & D.S. Butterworth 2004. A critical look at the potential of Ecopath with Ecosim to assist in practical fisheries management. *Afr. J. Mar. Sci.* 26: 261-287
4. Butterworth, D.S. & É.E. Plagányi 2004. A brief introduction to some multi-species/ecosystem modeling approaches in the context of their possible application in the management of South African fisheries. *Afr. J. Mar. Sci.* 26: 53-61
5. Plagányi, É.E., Butterworth, D.S. & A. Brandão. 2001. Towards assessing the South African abalone *Haliotis midae* stock using an age-structured production model. *Journal of Shellfish Research* 20(2): 813-827.
6. Plagányi, É.E. & D.S. Butterworth. In review. Does classic stock assessment have a role in a failed case of reconciliation of fisheries with conservation? Proceedings of World Fisheries Conference, Vancouver, Canada, 2004.
7. Plagányi, É.E. & D.S. Butterworth. In review. Competition between marine mammals and fisheries – can we successfully model this using ECOPATH with ECOSIM? Proceedings of World Fisheries Conference, Vancouver, Canada, 2004

Signed by candidate

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## ABSTRACT

Fisheries management worldwide is in flux with calls for an EAF (Ecosystem Approach to Fisheries) needing to be balanced with the ongoing requirements to provide timely and realistic assessment-based advice for management (often with major economic and social consequences), that is typically based on single-species stock assessment models. This thesis is an attempt to walk the bridge from single- to multi-species approaches to fisheries management by developing a “traditional” single-species stock assessment model that is used for management purposes, assessing possibilities for extending the model to incorporate multi-species effects and evaluating the potential of a range of multi-species approaches to contribute to the furnishing of practical management advice.

The South African abalone *Haliotis midae* fishery is an example of a commercially valuable resource that is currently experiencing a downturn due to a complicated mix of biological, social, political, economic and environmental factors. Core problems include illegal fishing and recent ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part of the range of the abalone. It seems that the lobsters have dramatically reduced sea urchin *Parechinus angulosus* populations, thereby indirectly negatively impacting juvenile abalone, which rely on the urchins for shelter. A spatial and age-structured production model (ASPM) developed as part of this study has provided the basis for management advice for this resource over recent years by projecting abundance trends under alternative future catch levels. The focus is on the main abalone fishery Zones A-D. The model estimates the reduction in juvenile abalone survival due to the ecosystem change extent and estimates the illegal take using a novel fisheries index – the confiscations per unit of policing effort (CPUPE). As a consequence of the recent explosion of poaching activities, the combined Zones A-D model-predicted 2003 poaching estimate of 933 MT (corresponding to the assumption that, on average, 36% of all poached abalone are confiscated) is more than seven times the legal 2003 commercial TAC for these Zones.

Given the complexity of ecosystem processes, there is a need to critically evaluate the tools used to steer this thinking. The focus here is on both the most widely-employed multi-species/ecosystem approach (ECOPATH with ECOSIM or EwE) as well as a scenario in which there is an urgent need (from management) for scientific evaluations to quantify indirect interactions between marine mammals and fisheries. A critical review of EwE highlights some weaknesses related to, for example, the handling of some life history responses such as compensatory changes in natural mortality rates of marine mammals, overcompensatory stock-recruit relationships, inadequate representation of uncertainty, possible problems in extrapolating from the micro-scale to the macro-scale as well as some (not too far-reaching) mathematical inconsistencies in the underlying equations. Strengths include the structured parameterisation framework, the inclusion of a well-balanced level of conceptual realism, a novel representation of predator-prey interaction terms and the inclusion of a Bayes-like approach (ECORANGER) to take account of the uncertainty associated with values for model inputs. The potential of EwE to contribute to five important multi-species management quandaries in the marine environments off southern Africa and Antarctica is assessed, leading to the conclusion that EwE has limited predictive capability in these contexts.

Aspects of the potential application of other multi-species/ecosystem modelling approaches to advise the management of South African fisheries are discussed. In general, reliable predictive ability from such models is likely to be achieved sooner for top predators because relatively fewer links need to be modelled. Accordingly discussion concentrates on the problems of modelling marine mammal-fisheries interactions. Competition is a primary concern, but existing evidence is inconclusive because of the difficulties of substantiating claims that predation by marine mammals is adversely affecting a fishery or *vice versa*. Numerous species have been implicated in such conflicts, and long-term studies are essential to evaluate relationships between rates of predation and types and densities of available prey, i.e., functional responses. More realistic modelling studies are needed to address operational or management issues. Such models should reflect uncertainty in data and model structure, describe the influence of model assumptions, focus on systems where there is the greatest chance of success, incorporate a sufficient array of ecological links, and include appropriate spatial and temporal scaling for data collection and modelling exercises. In general, GADGET (Globally Applicable Area-Disaggregated Generic Ecosystem Evaluation Tool) and Minimum Realistic Models (MRM) are seen to show the most promise for use as tools to assess indirect effects between marine mammals and fisheries. The hake-seal-fishery interactions off the west coast of southern Africa are discussed as an example and the initiatives being pursued to further this modelling work are summarized. An important message derived from this study concerns the need to couple multi-species/ecosystem models with a simulation framework to take explicit account of uncertainty and management issues.

## SUMMARY

Although ecosystem-orientated thinking is increasingly being incorporated into fishery management, the waters separating single-species versus multi-species adherents are still relatively uncharted and difficult to traverse. For example, single-species model analyses used to advise on catch levels concentrate heavily on checking sensitivities and the implications of estimation imprecision. In contrast, these considerations are frequently near ignored in evaluations of the more complex multi-species models. To bridge the gap, it is important to have a good understanding of the *modus operandi* of the two groups of practitioners. This thesis is an attempt to walk the bridge from single- to multi-species approaches to fisheries management by developing a “traditional” single-species stock assessment model that is used for management purposes, assessing possibilities for extending this model to incorporate multi-species effects and evaluating the potential of a range of multi-species approaches to contribute to the furnishment of practical management advice. Given the complexity of ecosystem processes, there is a need to critically evaluate the tools used to steer this thinking. However, this is a vast undertaking so that the efforts reported here represent no more than a small stepping stone towards achieving this goal. The primary focus of this work is on the southern African subregion, although the lessons learnt for the subregion have in several instances been gleaned from a worldwide review of the topic. Thus whereas the development of a single-species model for a local resource is considered in depth, consideration of multi-species issues necessitated a more wide-ranging review of initiatives being pursued worldwide. Attention was focused both on the most widely-employed multi-species/ecosystem approach (ECOPATH with ECOSIM), as well as on a scenario in which there is an urgent need (from management) for scientific evaluations to quantify the nature and magnitude of multi-species interactions in the form of indirect interactions between marine mammals and fisheries.

The South African abalone *Haliotis midae* fishery serves here as an interesting example of a commercially valuable resource that is currently on the verge of collapse due to a complicated mix of biological, social, economic and environmental factors. During the past decade, the resource has been beset with ever-increasing problems

with illegal fishing and an ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part of the range of the abalone. The lobsters have greatly reduced sea urchin *Parechinus capensis* populations, thereby indirectly negatively impacting juvenile abalone, which rely on the urchins for shelter. In response to calls of concern at the time these effects were first identified as potentially problematic, an age- and spatially-structured model was constructed for use in estimating the resource dynamics parameters and projecting biomass trends for the abalone resource in the major fishing zones, Zones A-D. This constituted the first quantitative approach applied to the management of this commercially valuable resource. Complicating factors that have had to be taken into account in the model construction process have included uncertainty regarding the magnitude of the catch taken by the illegal (poaching) sector and regarding the effect on abalone recruitment and survival of an invasion of rock lobsters into Zones C and D during the 1990's.

Novel features of the stock assessment modelling approach employed include the explicit differentiation between inshore and offshore components; the attempts to estimate the magnitude of the poached catch by fitting to CPUE and Fishery-Independent Abalone Survey (FIAS) abundance data as well as several years of catch-at-age (cohort-sliced from catch-at-length) data for the various components of the fishery; and the method of simultaneously fitting to "normal" management Zones A and B and to "lobster-invaded" Zones C and D so as to estimate the magnitude of the recent increase in juvenile abalone mortality rates in Zones C and D, likely as a consequence of the lobster invasion. Temporal changes in the pattern of poaching are reproduced using a new index, the CPUPE (Confiscations Per Unit of Policing Effort), which is based on the annual numbers of confiscations of poached abalone, standardised to take into account changes in policing efficiency in the region. A first look at possible patterns of resource recovery is presented, with some points to be borne in mind in debating the trade-offs associated with embarking on a multi-species management strategy.

Moving from single-species to multi-species and ecosystem approaches, draws attention to ECOPATH with ECOSIM (EwE), an aggregate system modelling package, which is currently the most widely employed approach to assess the ecosystem effects of fishing. The basic equations and assumptions, strengths and

weaknesses, some past and possible future applications and hence the potential of this approach to contribute to practical fisheries management advice are reviewed. Strengths include the structured parameterisation framework, the inclusion of a well-balanced level of conceptual realism, a substantially improved representation of predator-prey interaction terms and the inclusion of a Bayes-like approach (ECORANGER) to take account of the uncertainty associated with values for model inputs. Weaknesses include the constraining nature of the mass-balance assumption (of ECOPATH), the questionable handling of some life history responses such as compensatory changes in natural mortality rates of marine mammals and the paucity of systematic and step-wise investigations into model behaviour and properties. An important limitation related to the predominant use of EwE as a “blackbox” modelling tool is that users fail to consider a range of alternative interaction representations. As with all multi-species approaches, the major limitation in applying the EwE approach lies in the quality and quantity of available data. Current EwE applications generally do not adequately address uncertainty in data inputs and model structure.

The potential of EwE to contribute to five important multi-species management quandaries in the marine environments off southern Africa and Antarctica is assessed, leading to the conclusion that EwE has limited predictive capability in these contexts. Apart from data constraints, this is due, *inter alia*, to the high residual variability in these systems, the important role of abiotic and biotic mesoscale (o(100 km)) processes, life history handling and insufficient flexibility in the preset model structure to simulate hypothesized causative mechanisms adequately. However, prudent EwE applications that utilise good data and are based upon rigorous statistical analyses can complement the quantitative predictions of traditional single species models. They could be particularly useful in some contexts if output in the form of probability distributions encompassing a range of likely ecosystem responses were to be coupled with attempts to extend Operational Management Procedure (OMP) approaches to fisheries management beyond the single-species level, particularly as regards the operating models of the underlying dynamics that are used for computer simulation testing of OMPs.

Aspects of the potential application of other multi-species/ecosystem modelling approaches to advise the management of South African fisheries are discussed. In

general, reliable predictive ability from such models is likely to be achieved sooner for top predators because relatively fewer links need to be modelled. The deliberations of scientific workshops on modelling marine mammal-fisheries interactions held by two international marine mammal Commissions during 2002 are therefore used in part to guide thinking. Five questions are posed, with some responses suggested, relating to the development of a framework for multi-species/ecosystem modelling to contribute to South African fisheries management (should such models be used for testing or making decisions; do they appreciably reduce uncertainties associated with single species models; are whole ecosystem or minimum realistic models more appropriate; what computer software is best suited to implement such approaches; and what are the overall cost implications?). Caution is expressed that general scientific acceptance of predictive reliability for such models (as required for their use for management) is unlikely in the short term, and will probably require considerable data collection and complex analysis at a not insubstantial cost.

Indirect fishery interactions between marine mammals and fisheries are defined as ecological, in contrast to technical, interactions and include:

- 1) Direct competition for commercial species;
- 2) Indirect competition for commercial species;
- 3) Habitat degradation due to trawling and other fishing practices;
- 4) Marine mammals as hosts for parasites to commercial fish species;
- 5) Indirect competition for primary production;
- 6) Changes in shark/killer whale predation rates affecting marine mammals;
- 7) Fishery-induced shifts in the size structure of ecosystems;
- 8) Alterations in foraging strategies in response to fishing effects; and
- 9) Effects of noise from fishing operations on marine mammal foraging activities and effectiveness.

Perceived competitive effects are a primary concern but the evidence for their impact is inconclusive because of the difficulties of substantiating, incontrovertibly, claims that predation by marine mammals is adversely affecting a fishery or *vice versa*. Given the predictions of a global shortfall between supply and demand over the next few decades, coupled with the realisation that food consumption by marine mammals worldwide may be as much as 3-5 times the current commercial fisheries catches,

there is an urgent need for scientific evaluations to estimate the nature and extent of these conflicts.

Some 17 marine mammal species (Common Minke whale, Antarctic Minke whale, Bryde's whale, Humpback whale, Harbor porpoise, Short-beaked common dolphin, Common Bottlenose dolphin, Killer whale, Gray seal, Harbor seal, Harp seal, Hawaiian monk seal, Steller sea lion, Antarctic fur seal, Cape fur seal, Sea otter, Dugong) are identified as having high priority in terms of data acquisition and focused modelling studies. The overriding importance is stressed of prioritising further investigations to collect data (experimental and/or field-based, and on foraging in particular) to assist in resolving issues related to the relationships between the rates of predation and the types and densities of available prey, so as to determine the appropriate forms of the functional responses. Data in the form of time series are needed rather than more conventional snapshot-type estimates.

Additional important considerations for further modelling studies include:

- 1) the need to present results that reflect uncertainty in data and model structure;
- 2) the need to consider operational (i.e. management) issues;
- 3) examination of the extent to which underlying model assumptions predetermine or have implications for the results obtained;
- 4) the need for more realistic modelling of marine mammal populations;
- 5) the need to focus on specific areas/systems where there is the greatest chance of success (either because of the simpler structure of the ecosystem or because there are large signals in the data) e.g. the Antarctic and Barents Sea ecosystems;
- 6) the need for systematic investigations of the numbers of links that have to be included in a non-trivial ecosystem model for reliable predictive ability; and
- 7) the choice of an appropriate spatial and temporal framework for data collection and modelling investigations.

It is suggested that GADGET (Globally applicable Area-Disaggregated General Ecosystem Toolbox) and MRMs (Minimum Realistic Models) are the most appropriate tools to assess competition interactions between marine mammals and fisheries.





# Chapter 1

## Introduction and overview of thesis

The arrival of the 21<sup>st</sup> century has seemingly brought with it a new era in fisheries management in which the prevalent terminology is EAF (an Ecosystem Approach to Fisheries - Garcia *et al.* 2003) in contrast to more “dated” terms such as surplus production and single-species models. This is at least in part attributable to the increasing pressure exerted on species subject to fishing (and interconnected species in the ecosystem), and a growing realization of the need to consider broader socioeconomic effects as well as the ecosystem effects of fishing. Although computational restraints are much less of a problem due to improvements in modern computing power, progress in this field is still (and may always be!) impeded by imprecise parameter estimation given limited and noisy data, and the associated limited understanding of ecosystem functioning.

Nonetheless, as powerful new tools like ECOPATH with ECOSIM (EwE) are further developed and distributed, there is a growing body of scientists being drawn to this challenging new field. Given the relative ease of applying such “black box”-type approaches, there is often a gap between “whole ecosystem” modellers and “single-species” stock assessment modellers used to operating within a more rigorous setting. In practice, single species models are still the dominant tool worldwide for providing timely and reliable scientific advice regarding the management of commercially valuable stocks. As single-species and EAF approaches become increasingly merged in the development of management advice, it is important that modellers have a good understanding of both single-species and ecosystem approaches.

This thesis is an attempt to walk the bridge from single- to multi-species approaches to fisheries management through the following:

1. Collation, standardisation and analysis of raw data for a fishery (Chapters 3 – 5);
2. Development and application of a single-species stock assessment model to assist in the practical management of the species (Chapters 6, 7 and 10);

3. An examination of the potential and advisability of extending the above model to include multi-species considerations (Chapters 8-10 and part of 11);
4. A focus on the EwE approach, given that it is currently the most widely applied ecosystem modelling software worldwide (Chapter 11);
5. A broader overview of the range of available multi-species/ecosystem models and their relative potential in a southern African context (Chapter 12); and
6. Consideration of the indirect interactions between marine mammals and fisheries, a research field in need of urgent attention and also an appropriate starting point for developing and testing multi-species models because of the lesser number of foodweb linkages for apex predators (Butterworth and Punt 2003a). Before delving into the details of models to be applied, some time is spent in reviewing the evidence (and hence the need for models) of competition between marine mammals and fisheries (Chapter 13).

The emphasis throughout is on the provision of practical fisheries management advice with a focus on the southern African subregion, the study area on this candidate's doorstep. To contribute to practical advice, a multi-species modelling approach should provide at least qualitative and ideally defensible quantitative guidance as to the modifications in annual allowable catch levels deemed necessary because of the predicted effects that fishing on a target species will have on other components of the ecosystem. Suggestions in a local context are framed in terms of lessons to be learnt from a worldwide review of these topics. Moreover, the scope of this thesis is on single- and multi-species population dynamics effects, rather than on the full range of ecosystem aspects of fishing encompassing, for example, environmental effects and technical interactions (e.g. bycatch issues).

The South African abalone *Haliotis midae* fishery constitutes a central focus of this thesis for the following reasons:

1. Prior to this study, management advice for this commercially important resource was based solely on a set of *ad hoc* decision rules that used data on trends in the commercial CPUE (catch per unit effort), the average and modal

sizes of abalone in the commercial catches, and the proportion of the catch represented by the smallest legal sizes. This management strategy essentially considered short-term indicators only. Although the indicators provided guidance as to the direction of change required in setting catch limits, they were of little use in quantifying the magnitude of appropriate changes. There was thus clearly a need for the development and application of a more rigorous quantitative approach to assist in the provision of management advice such as on the choice of an appropriate annual TAC (Total Allowable Catch); and

2. Management of this resource is complicated by a number of factors (that link to some extent to a need for an ecosystem approach to fisheries management), including severe poaching, socioeconomic considerations, the need for a spatial framework to adequately describe resource dynamics and a recent abalone - rock lobster (*Jasus lalandii*) – urchin (*Parechinus angulosus*) multi-species effect (as described in Chapter 2 following) that is negatively impacting the abalone resource over a substantial proportion of its range.

The thesis is divided into two sections. Section I deals with the stock assessment of the South African abalone resource and provides some preliminary discussion of the multi-species aspects pertaining to this resource. Section II takes a broad overview of some of the most commonly applied multi-species / ecosystem approaches to fisheries management. The methodology described in Section I is essentially an adaptation and extension of that applied for management purposes for many key South African and Namibian fishery resources, including for the major fisheries for hake (*Merluccius* spp.) in this region (e.g. Butterworth and Geromont 2001, Butterworth *et al.* 2001, Butterworth and Rademeyer in press a). A similar approach has also been applied to a number of international stocks such as southern bluefin tuna *Thunnus maccoyii* (e.g. Butterworth and Plagányi 2000) and the Gulf of Maine cod stock *Gadus morhua* (e.g. Butterworth *et al.* 2003). The stock assessment exercise described in Section I is thus an adapted version of what is currently the most widely-employed single-species approach to fisheries management in the southern African region.

Multi-species considerations are yet to be formally included in the stock assessment approaches for the major fisheries resources in southern Africa. However,

considerable work has been conducted in the region to construct the first so-called “minimally realistic model” (Punt and Butterworth 1995) and, more recently, in implementing EwE (e.g. Shannon *et al.* 2000, Shannon *et al.* 2003), which is currently also the most widely utilized approach world-wide.

The aim here in crossing the bridge from single- to multi-species approaches was originally to critically review a range of multi-species approaches to assist in deciding on the most appropriate way forward. Given the potentially large scope of such a study, the focus has been restricted to the most widely applied approaches and to an example (that of competition between marine mammals and fisheries) for which an application of these approaches is particularly in need. The need for an EAF is well recognised and indeed mandated – locally by South Africa’s Marine Living Resources Act of 1998 (Anon. 1998). However, there is still a need for, on the one hand, many ecosystem modellers to better acquaint themselves with the practical realities of providing reliable management advice and, on the other hand, for single-species modellers to step back from the often frantic process of conducting stock assessments, and to use their expertise to guide the development and implementation of multi-species management tools. This study is about getting to grips with both of these approaches.

## **Overview of Section I – South African abalone stock assessment**

Chapter 2 following is an introduction to and overview of the South African abalone fishery, including discussion of core problems, and of historic and current management measures.

Chapters 3-5 describe the data available to assess the abalone resource and provide details regarding how these data are standardised, analysed and used.

Chapter 6 describes the development, and Chapter 7 an application, of the spatial- and age-structured production model (ASPM) that is currently used to develop management advice for the abalone resource.

Chapters 8 and 9 describe some preliminary attempts to model some multi-species aspects pertaining to the abalone resource.

Chapter 10 summarises some projection results pertaining to a new abalone management policy and provides discussion of some future work that is planned or recommended.

## **Overview of Section II – Multi-species perspectives**

Chapter 11 comprises an in-depth review of the heavily used ECOPATH with ECOSIM approach.

Chapter 12 provides a brief overview of some multi-species and ecosystem modelling approaches in the context of their possible application in the management of South African fisheries.

Chapter 13 tackles the question of the extent to which there is evidence for the interactions/competition that need to be modelled, and how best to attempt to quantify the indirect competition for food, between marine mammals and fisheries.

Chapter 14 synthesises some of the findings of this study, discusses the lessons to be learnt and summarises immediate future research priorities.

## GLOSSARY

Definitions of some terms used throughout this thesis are as follows:

AWG	- Abalone Working Group
BENEFIT	- Benguela Environment Fisheries Interaction and Training Programme
CCAMLR	- Commission for the Conservation of Antarctic Marine Living Resources
CITES	- Convention on International Trade in Endangered Species
CPUE	- Catch per Unit Effort
CPUPE	- Confiscations per Unit Policing Effort
DEAT	- Department of Environmental Affairs and Tourism (South Africa)
EAF	- Ecosystem Approach to Fisheries
EoH	- East of Hangklip (see Fig. 2.1)
EwE	- ECOPATH with ECOSIM
FAO	- Food and Agriculture Organization
FIAS	- Fishery-Independent Abalone Survey
GADGET	- Globally Applicable Area-Disaggregated Generic Ecosystem Evaluation Tool
ICES	- International Council for the Exploration of the Sea
IWC	- International Whaling Commission
MCM	- Marine and Coastal Management
MRM	- Minimal Realistic Model
MSE	- Management Strategy Evaluation
MSY	- Maximum Sustainable Yield
NAMMCO	- North Atlantic Marine Mammal Commission
NMFS	- National Marine Fisheries Service
OMP	- Operational Management Procedure
TAC	- Total Allowable Catch
VPA	- Virtual Population Analysis
WSSD	- World Summit on Sustainable Development

# **Section I**

## **South African abalone stock assessment**





## Chapter 2

### **A gastropod in crisis - an overview of the South African abalone *Haliotis midae* resource and its recent precipitous decline**

#### SUMMARY

Benthic shellfish populations worldwide are under heavy pressure given the inherent characteristics of the resource coupled with the typically high-value of the product (Castilla and Defeo 2001). The South African abalone *Haliotis midae* fishery is one of the oldest in the world but is currently on the verge of collapse. This is due to both ever-increasing problems with illegal fishing and recent ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part of the range of the abalone. It seems that the lobsters have dramatically reduced sea urchin *Parechinus angulosus* populations, thereby indirectly negatively impacting juvenile abalone, which rely on the urchins for shelter.

Total allowable catches (TAC) for abalone are set separately for each of seven fishing Zones, with the typical TAC of some 600t (wet mass) per annum having been reduced recently to some 400t. The recent release in 2003 of a new policy for the allocation of commercial fishing rights in the abalone fishery signals a new era in the management of this resource with the introduction of a system of co-management in an attempt to address the threat of illegal harvesting.

This study presents an age- and spatial-structured production model (ASPM) which has provided the basis for management advice for this resource over recent years by projecting abundance trends under alternative future catch levels.

#### 2.1 INTRODUCTION

The South African abalone fishery dates back to 1949 and is one of the oldest commercial abalone fisheries in the world, with records of commercial catch data since 1953 (Tarr 1992). The Abalone Working Group (AWG) of the Department of Environment and Tourism's (DEAT) Marine and Coastal Management (MCM)<sup>1</sup> branch is responsible for providing recommendations on annual Total Allowable Catches (TACs) for this resource. Four main fishery Zones A-D on the south coast (Fig. 2.1) have typically yielded some 80 - 90% of the total annual TAC (Table 2.1).

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<sup>1</sup> Marine and Coastal Management is hereafter referred to as MCM.

Aspects of the biology, including growth, reproduction and movement, have been well-studied and are summarised in Tarr (1993). The available data, assessment methods and management of key resources in the southern Benguela region are reviewed periodically by an international review panel at annual BENEFIT-NRF(-BCLME) Stock Assessment Workshops (hereafter referred to as BENEFIT Workshops). The terms of reference of the December 2002 workshop included critically reviewing past assessments and making recommendations for future research for the South African abalone (BENEFIT 2002).

## 2.2 BIOLOGY

The South African abalone fishery is reliant on a single gastropod species, *Haliotis midae*, locally termed *perlemoen*, which is restricted to shallow habitats in beds of kelp, *Ecklonia maxima* (Tarr 1993). Abalone are patchily distributed between Cape Columbine on the west coast, and the Transkei region of the Eastern Cape Province on the east coast (Fig. 2.1). A second species, *H. spadicea*, is not as abundant, and is taken by recreational fishers only.

During spawning, eggs and sperm from female and male animals respectively are released directly into the water, where fertilization occurs (broadcast spawning). Abalone larvae are planktonic and typically drift with the currents for about a week (McShane 1992). They settle mostly in shallow inshore waters (< 5 m), where they seek shelter under boulders or under the spines of sea urchins *Parechinus angulosus* (Tarr *et al.* 1996). They are slow growing, requiring a period of about 7 years to attain 100% sexual maturity, and 8-9 years to attain the minimum legal size limit of 114 mm shell breadth. They become emergent only once they have attained a sufficiently large size (~100mm) to be afforded some protection from predation (Tarr 1993). With increasing size, animals gradually disperse into deeper water.

Benthic invertebrates typically have fairly limited dispersal potential (Bradbury and Snelgrove 2001). Strong correlations between the abundances of adult and newly recruited abalone at several sites in South Australia suggests that abalone larvae are not widely dispersed (Prince *et*

*al.* 1988, McShane 1992). For black abalone *H. cracherodii* in southern California, average dispersal ranges are thought to be on the order of only 1- 5 km (Tegner 1993). Studies of genetic structure (e.g. Shepherd and Brown 1993) and simulations of larval transport for southern Australian abalone (Black 1993) provide additional support for the notion of dispersal over spatial scales of a few kilometers only. Although it is presently not known to what extent larval mixing occurs throughout the main fishing area in South Africa, larval and post-larval stages are most likely retained in areas close to the parental population, with some larval interchange between adjacent areas. It is considered unlikely that much inter-change occurs between adjacent fishery zones which each have a long-shore extent of some 30 km. Nonetheless, larval dispersal of South African abalone is likely wider (5 – 30 km) than reported for other species (BENEFIT 2002) based on the observation during recruitment surveys of consistent levels of juvenile (2 – 40 mm) abundance in the Betty's Bay Reserve and the neighbouring Zone D commercial grounds (R. Tarr, pers. commn).

## **2.3 THE FISHERY**

### ***2.3.1 A brief history of the fishery***

Several historic changes in the commencement and closure dates for the commercial fishing season are on record such that a standard Model (fishing) year  $y$  is defined as running from October of year  $y-1$  to September of year  $y$  (hereafter referred to as Model year  $y$ ).

The commercial fishery for abalone began in 1949, and reached a peak annual take of 2 800 MT in 1965 (Fig. 2.2). Regulation of the fishery was introduced in the 1970s (Tarr 1992). Commercial catches were subsequently reduced and then maintained at around 600 MT per annum during the 1970s and 1980s. Heavy poaching started in the early 1990s, necessitating subsequent annual reductions of the total allowable catch (TAC) to 500 MT for Model year 2000, 371 MT for 2001 and 432 MT for 2002. Recommended reductions are typically phased in over a 2-3 year period in the interests of industry stability. The 2004 allocation of 282 MT is the lowest

ever but is tied to a new management system in which abalone fishing rights are allocated for a ten year period (see discussion under 2.5.2.5).

The yield from the South African abalone fishery is less than one-percent by mass of that from the country's major fishery, the hake (*Merluccius* spp.) fishery (Butterworth and Rademeyer in press). However, in terms of economic value, the product is worth twenty or more times the equivalent mass of hake and most other fishery resources. As a rough guide, Gordon and Cook (2001) calculated that on a world-wide basis, when processing and marketing costs are taken into account, the "in shell" price of abalone was about US\$ 32/kilo some three years ago. The South African abalone fishery (i.e. excluding farmed animals) ranks as one of the top 5-10 biggest abalone fisheries in the world, but is nevertheless substantially smaller than the abalone fisheries of Australia and Japan (Gordon and Cook 2001, Tarbath *et al.* 2002).

### ***2.3.2 Commercial fishing for abalone***

The commercial fishery occurs only within the western part of the overall range of abalone, from Cape Columbine to Quoin Point (near Cape Agulhas) (Tarr 2002) (Fig. 2.1). A description of the fishery is presented in Tarr (1992) and Tarr (in press). Briefly, the commercially fished area has been divided into seven fishing zones, with total allowable catches (TACs) set separately for each zone over the last 27 years. The main fishing areas are Zones A-D (Fig. 2.1), and the focus throughout this thesis will be on these four zones which stretch from Cape Hangklip to Quoin Point. This section of coastline is naturally divided into zones because a series of sandy beach areas serve as partitions for areas containing sublittoral rocky seabed and hence suitable habitat for abalone (Tarr 1993). The West Coast Zones E- F (Cape Columbine to Cape Point) are characterized by much lower productivity (primarily due to irregular recruitment) than the 'traditional' commercial grounds east of Cape Hangklip (Tarr 2002). Several initiatives are underway to develop a small Eastern Cape abalone fishery, but this matter will not be discussed further here.

A description of the fishing method is given in Tarr (1992). Briefly, abalone are harvested by divers that utilize the “hookah” diving apparatus, which comprises an on-board compressor that supplies air to the diver by a long hose. Divers use small boats and can only operate in extremely calm sea conditions. The abalone are hand-collected by prising them from rocks with a lever.

There is also a steadily increasing abalone aquaculture industry in South Africa (Cook 1998), but this aspect is not relevant to the central theme of this thesis. Aquaculture may, however, play a small role in boosting recruitment in the wild through the practice of brood-stock transplants, as has been investigated both locally and internationally (see e.g. Tegner 1992).

### ***2.3.3 Sharing the abalone among fishing sectors***

A number of different sectors utilize the South African abalone resource. Tarr (2002) summarises these as follows:

1. The commercial fishery: Full Commercial right holders, in turn comprising two separate components:
  - a) Processors, who process and market their allocation. They may employ qualified divers and may also harvest the allocations of other right-holders.
  - b) Divers, who deliver their catch to processors.

Limited Commercial right-holders were introduced with effect from the 2001/2002 season. These have a maximum allocation of 430 kg which must be self-harvested, and delivered to a recognized abalone processor. These divers are usually compelled to harvest their allocations in the zone nearest their home address.
2. The recreational fishery has been subject to several changes in the legislation in the past few years. Recreational abalone fishers are required to purchase a permit and may only operate from the shore, using a snorkel. Until recently they have been permitted to collect three abalone per day on weekends during the season, the duration of which has been

progressively reduced as resource abundance has declined. Recreational abalone fishing has been suspended indefinitely since October 2003.

3. A subsistence fishery was established in 1998 following the passing of the Marine Living Resources Act by MCM. However, after two years abalone was found to be not suitable for a subsistence fishery, and was converted to a Limited Commercial sector.
4. Illegal fishing by poachers has been particularly serious since 1994.

## 2.4 CORE PROBLEMS

### 2.4.1 *The fight against poaching*

The core of the abalone's many problems resides in its status as a tasty and much sought after aphrodisiac in oriental countries! Its relatively easy accessibility in shallow intertidal waters makes it an easy target. The demand for abalone flesh in oriental countries has led to the setting up of organized Chinese Mafia (Triad) syndicates in South Africa who buy poached abalone (at times with drugs) and smuggle it out the country (Hauck 1999a). Demand for the product is driven by reputed aphrodisiac qualities (China) and a traditional useage as a high status product for important ceremonial events in Japan and China (Hauck 1999a, Clarke 2004). The product can be sold legally in Hong Kong and Taiwan (Gastrow 1999). Drastic declines in abalone populations elsewhere in the world, such as the black abalone *Haliotis cracherodii* fishery off California (Tegner 1993), have further fuelled both the demand for and rising price of abalone. The South African species *H. midae* is one of the most sought after due to its large size and high quality flesh (Tarr 2002, Clarke 2004).

Poaching has undoubtedly occurred throughout the history of the abalone fishery, but only during the last decade has it spiralled out of control (see Chapter 4) (Fig. 2.3). The magnitude of the catch taken by the illegal sector is unknown, and is difficult to estimate in the field because of the evasive behaviour of the illegal fishers (poachers). Spokespersons for the illegal sector claimed

in the late 1990s that their catch is approximately half that of the legal commercial fishery in Zone C. In 1997 Operation Neptune, a joint task force of the South African Police Service and the Department of Environment Affairs and Tourism, was set up to combat the poaching problem. However, efforts were focused in Zone C and resulted in poachers shifting their operations to other zones (see Chapter 4). In 1999, police estimated that the illegal harvesting and trade in abalone was worth approximately 500 million South African rand – approximately the same as the profit from the legal harvesting and sale of abalone (Gastrow 1999).

The years 2002 and 2003 saw the greatest poaching activity ever recorded, with the number of abalone confiscated per year exceeded the total legal take (in terms of numbers) of abalone (Tarr 2002). Given the large scale and lucrative economics of poaching operations, the local fight against poaching has, at times, been akin to a war with gun-toting gangsters set against police, soldiers and environmental officers. A social and criminological study (Hauck 1997, Hauck and Sweijd 1999) has highlighted the need to shift away from the historic reliance on policing to more cooperative management structures.

The minimum legal size limit of abalone is a shell width of 114 mm, which corresponds to an age of 8 years (Tarr 1995). Analysis of confiscated abalone samples recovered from the illegal sector (see Chapter 3) suggests that the minimum age of individuals taken by this sector is approximately 4-5 years.

#### ***2.4.2 The importance of lobsters and urchins to abalone***

In addition to the threat posed by poaching, the South African abalone resource has also, for the past decade, seemingly been the victim of a complex and poorly understood ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part (Zones C and D) of the range of the abalone. During the mid- to late-1990's, MCM data suggested that sea urchins had recently disappeared from sites such as Betty's Bay (Zone D) and Mudge Point (Zone C). As sea urchin abundance declined, so too did the numbers of juvenile abalone. Indeed, the number of juvenile abalone recorded at Mudge Point, for example, has been almost zero since 1995 (R.



Tarr, pers. commn). Juvenile abalone (individuals 3-35 mm in length) depend heavily on a commensal association with urchins *Parechinus angulosus*, because the urchins provide protection against predators and may supplement the diet of juvenile abalone (Day and Branch 2002). Abalone recruits are also thought to be adversely affected by the increased sedimentation that occurs following the removal of urchins (Day and Branch 2002). The observed recent decline in urchins and abalone recruitment failures in Zones C and D are thought to stem from a substantial increase in the abundance of rock lobsters in these zones. Predation of urchins by rock lobsters is blamed for the collapse of urchin populations and dramatic reductions in the numbers of juvenile abalone (Mayfield and Branch 2000).

The eastward movement of rock lobsters in the two westernmost of the abalone Zones east of Hangklip was presumably precipitated by some environmental factor, but this aspect remains open to debate. The affected Zones C and D (Cape Hangklip to Hermanus), are conveniently referred to as the “Lobster Zones” (Tarr 2002). The effect described above is referred to as the “ecosystem-change” effect throughout this thesis. Ironically, it was also in Zone C, adjacent to the coastal community of Hawston, that poaching first escalated and later spread to the other zones.

## **2.5 ASSESSMENT**

### ***2.5.1 Data available***

#### **2.5.1.1 Catch-per-unit-effort (CPUE)**

Several authors have stressed that CPUE is not a reliable index in situations where the stock is highly aggregated, as occurs, for example, for abalone off southeastern Australia (Breen 1992, Keesing & Baker 1998). It is now well established that in areas such as South Australia and Tasmania the spatial variation in density is such that, coupled with the added complexities of diver behaviour, CPUE cannot be considered a reliable indicator of stock abundance (Breen 1986, Prince 1992, Shepherd *et al.* 1992, Keesing & Baker 1998, Worthington & Andrew 1998, Gorfine *et al.* 2001). In these cases, the general pattern of diver behaviour is such that new

aggregations are sought out and then sequentially depleted so that catch rates remain fairly stable despite the resource abundance declining, as has been demonstrated for fisheries off Australia (Shepherd & Baker 1992), California (Tegner *et al.* 1989) and Mexico (Prince & del Proó 1993).

There are, however, a number of reasons why CPUE may be a reasonable index of abalone stock abundance in the South African context but not necessarily elsewhere in the world. The major South African abalone fishery is located in shallow kelp bed areas, relatively close inshore, along a stretch of coastline measuring some 125 kilometres, and with relatively easy access to most of the areas (R. Tarr, pers. commn). Furthermore, the ten or so launching sites along this coastline have been in use consistently since the 1950s, when the fishery began. Even if very large aggregations existed in the past, the resource was so heavily fished during the 1960s that these would since have been reduced substantially, dampening spatial patchiness to a large extent. Moreover, fishing effort is spread fairly evenly over most of the major fishing zones, so that the general pattern of diver behaviour (in recent decades at least) is not one in which new aggregations are sought out and then sequentially depleted.

As proposed by Dichmont *et al.* (2000) one option available to resource managers in a situation where there is some confidence in CPUE as an index of abundance for their fishery, is to use a dynamic model tuned to both CPUE data and a survey index. It is argued here (see Chapter 3) that CPUE data have utility in the South African context, especially when used in conjunction with a fishery-independent index of abundance. There is a long time-series (23+ years) of CPUE data for the abalone resource and these data are suggestive of a resource decline in recent years.

At the time this study was initiated, much attention was focused on how appropriate the use of CPUE data as an index of South African abalone stock abundance was because of, *inter alia*, the apparent contradiction of observed increases in CPUE estimates and overall catch levels during the 1980s and early 1990s. The present form of analysis (which started in 1997) was the first attempt to quantitatively assess the effect of historic catch levels on the abalone resource. This showed that total catches in the 1960s were substantially greater than those taken in the 1970s and 1980s (Fig. 2.2) and hence that it was not too surprising that the CPUE trend showed an

increase towards the end of the 1980s (see Chapter 5). This is to be expected from a relatively slow-growing long-lived resource afforded a respite as high and unsustainable historic catch levels are reduced substantially to below the then current sustainable yields. Despite previous reservations regarding the usefulness of the CPUE data, it is nonetheless considered to have some utility for assessment purposes after all (see Chapters 3 and 5).

#### **2.5.1.2 Length frequency data and conversion to age**

Commercial length frequency sampling is conducted throughout the year by a contracted agent. A general impediment to applying standard stock assessment models to abalone populations has been the lack of adequate data on ageing. Tarr (1995) used a tagging study to estimate growth rates of *Haliotis midae*. This in turn allowed the age structure of catches to be estimated, and consequently it has been possible to construct the first fully age-structured population model to assess the status and productivity of the South African resource. Issues of possible growth variation in time and space are discussed in Chapter 10.

#### **2.5.1.3 Fishery-independent surveys**

During the 1980s, fairly extensive diving surveys of the resource were conducted. Strip transects, stratified in three depth ranges (0-5, 5-10 and 10-15 m) were placed within the major kelp (*Ecklonia maxima*) beds (Tarr 1993), providing data on abalone density and size distribution. Almost the entire abalone resource occurs within kelp forests that are visible from the air because the fronds float on the surface. In an attempt to estimate the total abalone biomass, estimates of the total habitat area were obtained from infrared aerial photographic surveys of the kelp (Tarr 1993). This was negatively biased to some (unknown) extent because the fronds of deeper kelp do not always break the surface. Moreover, these early biomass surveys were characterized by very large coefficients of variation as a result of the aggregatory behaviour of abalone and the diverse bottom topography (Tarr 1993, Dichmont *et al.* 2000). They therefore had very limited power to detect increases or decreases in abalone abundance (Dichmont *et al.* 2000).

New fishery-independent abalone surveys (FIAS) were initiated in 1995 to provide an index of *relative* abundance with a variance low enough to have management utility (Dichmont *et al.* 2000). The chosen survey design involves twenty evenly-spaced GPS-located diving stations which are surveyed once annually per fishing zone. Each station comprises a 30 x 2m transect oriented perpendicular to the coastline. The design of the surveys anticipated a coefficient of variation for each zone of approximately 25%. Some time was required before a time series became available that was sufficiently long to provide an estimate of trend and hence could be useful for management purposes. It was therefore only in late 1999 that the FIAS data were input to resource assessment models, and they have since become an integral component of the resource assessment.

The FIAS data series not only provide a valuable independent index of abundance, but are also useful because they provide information on abalone from about age 5 upwards, compared with the CPUE data (see Chapter 3) which capture information on animals aged approximately 8 years or more. This means that the FIAS abundance index is more sensitive than the CPUE index to changes in the younger age classes. This is particularly important because extensive poaching of sub-legal abalone individuals as well as the ecosystem change effect have caused a rapid decline in the numbers of small individuals. Consequently, the FIAS data have detected the resultant negative trend earlier than have the CPUE data, thereby also providing valuable clues as regards important factors to consider in modelling the resource dynamics.

#### **2.5.1.4 Recreational sector surveys**

Annual telephonic interviews of recreational divers are conducted by a contracted specialist survey company to estimate the total magnitude of the recreational catch taken from each zone (Tarr 2002). In addition, length frequency samples from recreational catches are measured for each zone.

### **2.5.1.5 Monitoring the poachers**

Since 1994, records have been kept of all known abalone confiscations (Fig. 2.3), as well as, when known, the zone from which these illegal catches were removed (see Chapter 4 for further details). Sub-samples of confiscated abalone are also measured to determine length frequency for this sector.

## **2.5.2 Models and Management**

### **2.5.2.1 History of management and management controls: 1949 - 1994**

From the inception of the abalone fishery in 1949 until 1972, divers were not required to submit catch returns and there were few management controls (Tarr 1992). Maximum annual production quotas were imposed for the resource as a whole with effect from 1968, with several reductions being implemented over the following period in response to concerns regarding falling catch rates (Tarr 1992). In 1983, a policy limiting catches per area was first implemented voluntarily for an area near Hermanus (Tarr 1992). This was followed by the formal adoption in 1986 of an annual Total Allowable Catch (TAC) as the sum of individual quotas in each of seven discrete fishing zones (Tarr 1992). This was motivated by the need to balance fishing effort between the different fishing grounds (Tarr 1992). TAC estimates over this period were based on visual inspection of indices such as CPUE for each zone (Tarr 1992).

In addition, a number of management measures were applied historically to the abalone fishery and remain enforced today. These are described in detail in Tarr (2002) and include a minimum legal size restriction, closed season, gear restrictions, slipway controls, zonal partitioning and Marine Protected Areas (MPAs). These measures are described in more detail in Appendix 2.1.

The use of MPAs as a management tool has been particularly problematic as the Betty's Bay Marine Reserve area located within Zone D historically served as the largest protected area for abalone. However, both poaching and the ecosystem-change effect have impacted this reserve to

the extent that it no longer functions effectively as a reserve area. Dyer Island has therefore been proposed as a reserve area to function in its place (Tarr 2003). The important role of marine harvest refugia in coastal fisheries management has been recognised for some time (Dugan and Davis 1993, Tegner 1993, Hockey and Branch 1997). An important consideration in selecting such areas relates to the extent of larval dispersal because refugia need to reseed not only the local population but also a sizeable proportion of the rest of the population i.e. the metapopulation (Shepherd and Brown 1993). It is not currently known to what extent Dyer Island may be able to reseed neighbouring areas, but it is thought that the direction of currents flowing from Dyer Island may be able to facilitate larval dispersal (R. Tarr, pers comm).

#### **2.5.2.2 Guidelines for setting TACs in Zones A-D: 1995 – 1999 and in Zones E-G: 1995 - 2002**

Throughout the history of the abalone fishery, the aim of management has been to maintain constant catches and CPUE from year to year, in the interest of stability in the fishery (Tarr 1993). During the mid-1990's, a set of guidelines were developed by the AWG to ensure consistency in this approach. Management advice was based on trends in the commercial CPUE data, the average and modal sizes of abalone in the commercial catches, and the proportion of the catch represented by the smallest legal sizes. Management advice for all zones over the period 1995 to 1999 was based on the following set of decision rules:

1. TACs are set for separate fishing zones.
2. Each zonal TAC may not be increased or decreased for two years since the last TAC change, unless a major change of resource status becomes apparent.
3. TACs may not be increased or decreased by more than 10% unless special circumstances arise.
4. If application of rule 2 allows for change, an increase or decrease of the zonal TAC may occur only if no indicators of resource trends (CPUE, % of catch in first 5% above the minimum size and mean/modal catch size) are in conflict.

5. In Zone A, a conservative/precautionary approach is applied because this is the distribution limit of the high density Western Cape populations.
6. In Zone C, a conservative/precautionary approach is applied due to illegal harvests.

This management strategy essentially considered short-term indicators only. Although the indicators provided guidance as to the qualitative direction of change required in setting catch limits, they were of little use in quantifying the magnitude of appropriate catches. There was thus a clear need (addressed by this study) for the development and application of a more rigorous quantitative approach to assist in the furnishing of management advice such as the choice of an appropriate annual TAC.

The decision rules above have continued to serve as the chief basis for making management recommendations for Zones E, F and G because there are insufficient commercial catch and effort data with appropriate contrast available at present to apply modelling analyses.

For Zones A-D, over the period 1997 to 1999, the decision rules were used in combination with an assessment model (as described below) for making recommendations. After 1999, modeling analyses formed the basis for the formulation of management advice in Zones A-D because the decision rule restrictions regarding the frequency and magnitude of decreases in TACs were over-ridden by the alarming negative results concerning resource status in these zones.

### **2.5.2.3 Current assessment methodology: 1997 - 2003**

Several different stock assessment approaches have been used to advise abalone fishery management (reviewed in Breen (1992)). One of the greatest impediments in the application of standard stock assessment models to abalone fisheries has been that such models usually rely on catch per unit effort (CPUE) data as an index of abundance. However, as discussed above, for the South African abalone fishery CPUE is considered a reasonable index of stock abundance.

In 1997, an attempt was first made to extend this management strategy from consideration of short-term indicators only, to analyses that incorporated all past information and allowed long-term projections of future trends. An age-structured model was applied to the resource in Zone C in 1997, using commercial catch per unit effort data. In 1998, the commercial CPUE data were standardized by applying a generalised linear model (GLM), and the modelling work was extended. Modelling efforts have concentrated on the Zones A-D component of the stock. A particular advantage of using an age-structured (or length-structured) model to assess the dynamics of the resource is that proper account is taken of the inclusion of animals of sub-legal size in the illegal catch component, which can have important consequences for resource trends and estimates of overall sustainable yields.

The age-structured production model developed in this study provides the basis which is used at present to develop management advice for the abalone resource by projecting abundance trends under alternative future catch levels. The model was applied in the first instance to Zone C because of the severe stock depletion thought to have occurred in that zone and because the data for Zone C exhibited more contrast than the data for the other zones so that it seemed sensible to apply the model first to this area as this contrast suggested better potential for precise parameter estimation. Such estimated parameter values were then input to applications of the model to other zones where data contrast is insufficient to allow their independent estimation.

Some of the early analyses in this study showed that it was difficult to interpret the abalone dynamics in Zone C without consideration of the fact that the western and eastern areas of this zone differed with respect to both density and size composition (Figs. 2.4 a,b). For assessment purposes, Zone C has therefore been split into a “poached” subarea (CP) to the west (Hawston/Mudge Point areas) and a “nonpoached” subarea (CNP) to the east (Hermanus vicinity) (Fig. 2.1). Although some poaching is still thought to occur in the “nonpoached” subarea, this is thought to be considerably less than that in the “poached” subarea.



#### 2.5.2.4 Management strategy: 1994 - 2003

The South African abalone fishery has in the past been managed using a limited access system in combination with the setting of an annual TAC. Political changes in South Africa during the past decade have driven a number of changes in the process of allocation of rights in the abalone industry. Prior to 1994, the abalone TAC remained relatively stable and poaching was thought to be localised and not substantial. The management system was essentially one of long-term rights but these were denied to citizens who were not white. Following the abolishment of apartheid and under the leadership of Nelson Mandela, the South African government undertook a major transformation or restructuring of its fishing industry with effect from 1994. South Africa's new Marine Living Resources Act of 1998 recognised the need to "restructure the fishing industry to address historical imbalances and to achieve equity within all branches of the fishing industry" (Anon. 1998). However, the expectations from citizens previously denied access to the resource far exceeded the quota available to be allocated. To make matters worse, there was insufficient capacity to carefully process the thousands of applications received. Dissatisfaction with the current system was widespread with concerns expressed such as the following: "*Around 70-80 per cent of the ownership of the access rights remains concentrated in the hands of the five largest players. Racial apartheid has been replaced by economic and social apartheid, with coastal communities and local economies still effectively excluded from the fishery*" (O' Riordan 1999).

Abalone poachers argued that they were forced to poach because they were unjustly denied access to abalone diving rights, and hence were unable to become legitimate members of the industry (Hauck and Sweijd 1999). Spokespersons for MCM countered that the process by which permits were awarded had been substantially reformed, rendering the allocation of rights more just. For example, one initiative involved the awarding of 173 limited fishing rights worth a substantial sum of money to fishers from previously disadvantaged communities.

Ongoing illegal activity was attributed to a combination of need, greed and politics, and was further fuelled by organised crime (Hauck and Sweijd 1999). The period 1994 to 2003 was

characterised by a lack of security attached to successful quota applications such that long-term interests were forfeited in favour of lucrative short-term gains. This translated into few incentives for quota holders to “invest” in the resource. Bitter conflicts were evident between the various stakeholders, leading to the local coining of the term “abalone wars”. Industry representatives argued against cutting their legal quota in sync with increases in poaching, perceiving such cuts as merely translating into increased numbers of abalone becoming available to the poachers. Given that total takes of abalone by all sectors has been unsustainable during the past few years, the abalone stock assessment model primarily played the role of documenting the decline of the resource and quantifying the seriousness of the current declines in resource abundance (see Chapter 10).

Indeed, as highlighted by the review panel of the 2002 BENEFIT Stock Assessment Workshop (relevant extract given as Appendix 2.2), the modelling methods were critical in highlighting the extent of the problem and evaluating possible changes in harvest regime, but did not represent the solution. The actual solution was seen to reside in the development of alternative management strategies that include local communities. In response, a new policy to define the process of allocating commercial abalone fishing rights was announced by MCM in October 2003.

#### **2.5.2.5 A new management strategy: 2004 -**

A variety of different fisheries management models are in place worldwide, ranging from open access systems with TAC's to Individually Transferable Quotas (ITQ) and Territorial User Rights in Fisheries (TURF) systems (Stephenson & Lane 1995, Christy 1996, Caddy 1999). Modern thinking in fisheries management foresees that the most successful strategies are likely to be those involving the award of long-term property rights (Stephenson & Lane 1995). The severe overexploitation of many Latin American benthic shellfish stocks has been attributed largely to the absence of co-management practices (Castilla and Defeo 2001). The allocation of TURF's has achieved some success in improving both the quality of the regulatory process and the status

of the shell-fisheries in these regions (Castilla *et al.* 1998, Castilla and Defeo 2001, Parma *et al.* 2003).

Given the failure of more traditional top-down management strategies for South African abalone, a new policy to underpin the process of allocating commercial abalone fishing rights was announced by MCM in October 2003. The adoption of this new policy introduces a radical shift in the way South Africa manages its abalone resource, commercially and recreationally.

As outlined in the new policy document (DEAT 2003), the objectives of adopting a new management regime are to:

- Instill a culture of “ownership” amongst right holders and members of the respective coastal communities;
- Encourage right holders and members of the respective coastal communities to co-manage the abalone resource, together with the Department (i.e. MCM);
- Ensure the long-term viability of the South African abalone fishery; and
- Sustain the level of employment in the abalone fishery.

Each right holder will be allocated a secondary zone. It is intended that each secondary zone will comprise a relatively small number of right holders who will be entitled to harvest abalone only in that secondary zone. This secondary zone will effectively be “owned” by the right holders. The community adjacent to each secondary zone will play an important oversight role, ensuring that only “their” right holders dive in that secondary zone. This policy intends to introduce a new system of co-management and to effectively address the threat of the illegal harvesting and over catching of abalone.

Ten-year abalone fishing rights have therefore been allocated in approximately equal quantities to five abalone fish processors and to 200 divers, with a somewhat smaller amount having been allocated to 24 “legal entities”. Some three-quarters of the successful diver applicants are historically disadvantaged persons (HDP) and the successful legal entities are 66% or more HDP owned and managed. The allocation of secondary zones in which the divers and legal entities will be permitted to operate is to be decided in consultation with the relevant stakeholders and right holders.

### **2.5.2.6 Linking the management of abalone and rock lobsters: 2003 -**

Given that the movement of rock lobsters into the East of Hangklip (EoH) area has negatively impacted the abalone populations in the area, a number of proposals have been put forth (see Chapter 8) to reduce the rock lobster population so as to allow the abalone population a chance to recover. The BENEFIT 2000 workshop agreed that there were no biological reasons to support continued closure of the EoH area to lobster fishing and recommended that the entire EoH catch should be allocated to as small an area (or areas) as possible so as to derive the maximum scientific benefit (in terms of the potential for approximating a large-scale experiment to examine the impacts of reduced numbers of lobsters EoH) (BENEFIT 2002). Moreover, the panel noted that consideration might be given to using these allocations to alleviate some of the problems associated with abalone poaching (see Appendix 2.2).

The new abalone policy described above was intended to complement the policy that has been adopted for the management of the (commercially valuable) west coast rock lobster fishery east of Cape Hangklip. In an effort to reduce lobster numbers and to simultaneously reduce the pressures on the abalone resource, an allocation of some 100 tons of commercial west coast rock lobster was made on a limited basis to commercial applicants with effect from the last quarter of 2003. Long-term abalone rights holder applicants were allowed to simultaneously hold abalone and rock lobster fishing permits.

## **2.6 OUTLINE OF MATERIAL TO FOLLOW**

The work reported here chronicles what has been done (primarily by the author) over a period of 7 years (on a part-time basis) in an attempt to:

- collate all available data for the South African abalone resource;
- standardise the CPUE data;
- develop a stock assessment model; and

- explore the potential of multi-species approaches / extensions of the current model to contribute further insights to the management of this stock.

Given the applied nature of this work, it has been extensively discussed and reviewed at MCM Abalone Working Group (AWG) meetings. Working group papers for these meetings are often put together at short notice and typically are constructed within a framework dictated by Working Group decisions and agreements. Rather than being afforded the freedom to explore theoretical aspects of this work in greater detail, the driving force has been the need to furnish practical advice with immediate management consequences. Some of the decisions made in the assessment process (e.g. the GLM model and terms to be included in the GLM, what policing efficiency fraction to assume – see Chapter 4) were agreed by consensus by the AWG at the time and hence there was limited flexibility to change these decisions.

The model developed here is used as the primary basis for recommending catch limits in the form of an annual TAC per Zones A-D. The Abalone Working Group is a scientific working group and as such is not mandated to advise on management issues such as quota allocations among different sectors. Thus although there is typically a very high correlation between TAC recommendations furnished by the Working Group and those approved by the Minister, this is not the case regarding recommendations on relative quota allocations to different sectors. Section I of this thesis is essentially a consolidation of a large number of unpublished Abalone Working Group documents (see list at end of thesis) authored/co-authored by the writer of this thesis.

Chapter 3 following describes the data available to assess the abalone resource in each of Zones A-D and gives details regarding how these data are analysed and used.

Chapter 4 describes the way in which the data on poaching confiscations are used to derive indices of the relative amount of poaching per unit of policing efficiency.

Chapter 5 describes the application of a General Linear Model (GLM) to standardise the catch-per-unit-effort data input to the model.

Chapter 6 describes the development of the spatial- and age-structured production model (ASPM) that is currently used to develop management advice for the abalone resource. The model is fit simultaneously to Zones A-D.

Chapter 7 provides selected results from applying the 2003 base-case assessment model described in Chapter 6.

Chapter 8 describes a first attempt to model a possible experimental take of rock lobsters from East of Hangklip (EoH) to enhance recovery of the abalone population there.

Chapter 9 describes a preliminary abalone-urchin-lobster multi-species model to explore proposed multi-species management scenarios in the EoH area.

Chapter 10 summarises some projection results pertaining to the new abalone policy and to the 2004 TAC recommendations. Discussion is also provided of some future work that is planned or recommended.

**Appendix 2.1. Extract from Tarr (2002) describing management measures for South African abalone currently in place.**

Minimum legal size: 114 mm shell breadth, usually applied using a circle of inner diameter 114 mm, through which sublegal abalone will pass. This measure has been in place since the 1950's

Closed Season: The season is closed for the months August to September, inclusive. The closure is not imposed for any biological reasons, since abalone do not change behaviour or become more vulnerable to fishing during the reproductive season.

Gear: Divers are compelled to use Hookah diving equipment, which always physically links the diver to the boat. A special smooth-edged collecting tool must be used to remove abalone, because abalone have no blood clotting mechanism, therefore cuts from a sharp collecting tool, on sublegal, returned abalone, attract whelks and other predators, resulting in high mortality.

Slipway controls: divers must notify fishery officers before departing the slipway, must fill in log books on returning, can only operate from certain designated slipways, and the catch is weighed and sealed at the slipway prior to being driven to the factory.

Zones: Each fishery management zone is independently assessed and managed. Catches per zone are monitored daily, and a zone is closed to fishing once the TAC has been filled.

Marine protected areas: A number of marine protected areas have been designated in order to retain near-pristine abalone populations. However the main MPA at Betty's Bay no longer functions adequately due to the breakdown in the reproductive cycle caused by lobster predation. In addition poachers are now specifically targeting the accumulated stocks of abalone in this area, and rapidly depleting it. For this reason Dyer Island (presently unaffected by lobster incursions) has been adopted as a future abalone sanctuary. It too, however, is being targeted by poachers.

**Appendix 2.2. Comments by the review panel attending the 2002 BENEFIT Stock Assessment Workshop.**

*“While the panel supported the recommendations for improving the assessment as detailed in Appendix 5, the prime concern of the panel was that the utility of the assessment process may be severely limited by the poaching crisis affecting the stock. It is clear that the current models indicate a general collapse of the resource and industry TAC under scenarios of continued or increasing poaching. While improvements of modeling methods should continue, the main usefulness of the assessment currently lies in documenting and communicating the unsustainable nature of the status quo harvest regime as it exists alongside uncontrolled illegal exploitation. This documentation is seen as essential in highlighting the extent of the problem and evaluating possible changes in harvest regime, but does not represent the solution.*

*In order to maintain sustainable harvests from the resource, exploitation, including poaching, needs to be controlled. Controlling poaching may be approached either through improved enforcement, or by creating an environment that motivates increased compliance with the regulations, or some combination of these. As an example of the latter, uncontrolled exploitation of benthic invertebrates in Chile led, after a closure due to collapse, to replacing a TAC system with a system of local territorial rights. The actual solution must be locally developed, and the panel was of the opinion that, without significant resources being directed toward the development of alternative management strategies that include local communities, the stock assessment process would only serve to document the demise of the resource as a whole.*

*Interaction between rock lobsters, sea urchins, and abalone along with the increase in rock lobster abundance has played a part in recent abalone recruitment failure east of Cape Hangklip. The panel noted that if quota of West Coast rock lobster is to be allocated east of Cape Hangklip to new fishers, consideration might be given to using these allocations to alleviate some of the problems associated with abalone poaching. For new rock lobster allocations, the panel recommends that consideration be given to allocating to local communities or groups historically or currently dependent on abalone in a manner that encourages compliance with national legislation through local assistance.*

*Moreover, limiting the new fishing area for West Coast rock lobster based on a formal experimental design would aid in determining the efficacy or potential influence of a fishery for West Coast rock lobster on abalone productivity. However, controlling the size of the rock lobster population east of Cape Hangklip would not, in itself, reduce the need for controlling illegal fishing for abalone.”*



Table 2.1. The commercial abalone Total Allowable Catch (TAC) for each of fishery Zones A – G (see Fig. 2.1) and Dyer Island for Model years 1987 to 2003, where a Model (fishing) year  $y$  is defined as running from October of year  $y-1$  to September of year  $y$ . Dyer Island has been closed to commercial abalone fishing with effect from 2004. The last column shows the percentage of the total commercial TAC taken from the main fishery Zones A-D.

Model year	ZONE								Total annual commercial TAC	Zones A-D as % of total TAC
	A	B	C	D	E	F	G	Dyer		
1987	180	160	160	40	20	50	30		640	0.84
1988	190	140	160	50	20	50	30		640	0.84
1989	190	140	160	50	20	50	30		640	0.84
1990	195	140	170	50	20	50	0		625	0.89
1991	195	140	170	50	10	30	0		595	0.93
1992	195	145	150	55	0	30	0	25	600	0.91
1993	195	150	150	55	0	30	0	25	605	0.91
1994	195	150	140	90	0	15	0	25	615	0.93
1995	205	150	130	90	0	15	0	25	615	0.93
1996	205	150	130	90	0	15	0	25	615	0.93
1997	205	150	65	90	0	15	0	25	550	0.93
1998	185	150	30	105	5	15	15	25	530	0.89
1999	185	150	15	105	5	15	15	25	515	0.88
2000	185	145	5	105	5	15	15	25	500	0.88
2001	158	113	0	35	5	20	15	25	371	0.82
2002	136	110	52	50	13	20	25.5	25	432	0.81
2003	170	100	30	40	18	20	27	25	430	0.79

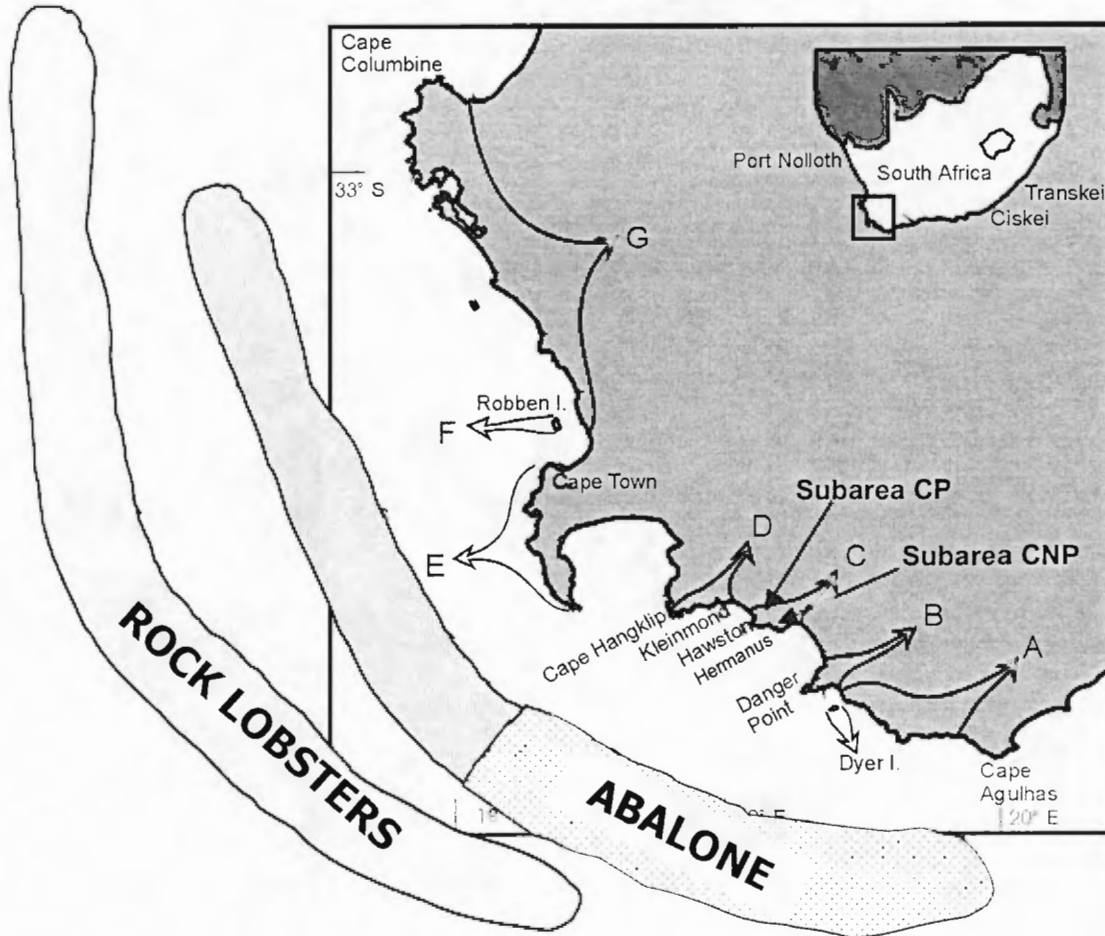


Fig. 2.1. Map (modified from Tarr and MacKenzie 2002) showing the commercial fishing zones defined for abalone *Haliotis midae* in South Africa. This study focuses on the four main fishing Zones (A-D) (the main abalone fishing Zones are shown as the shaded portion of the abalone distribution). Zone C has been split into a “poached” subarea (CP) and a “nonpoached” subarea (CNP). The figure also shows the distribution of the west coast rock lobster *Jasus lalandii*, which has been extending eastwards since the 1990’s and so overlaps with abalone in Zones D and C.

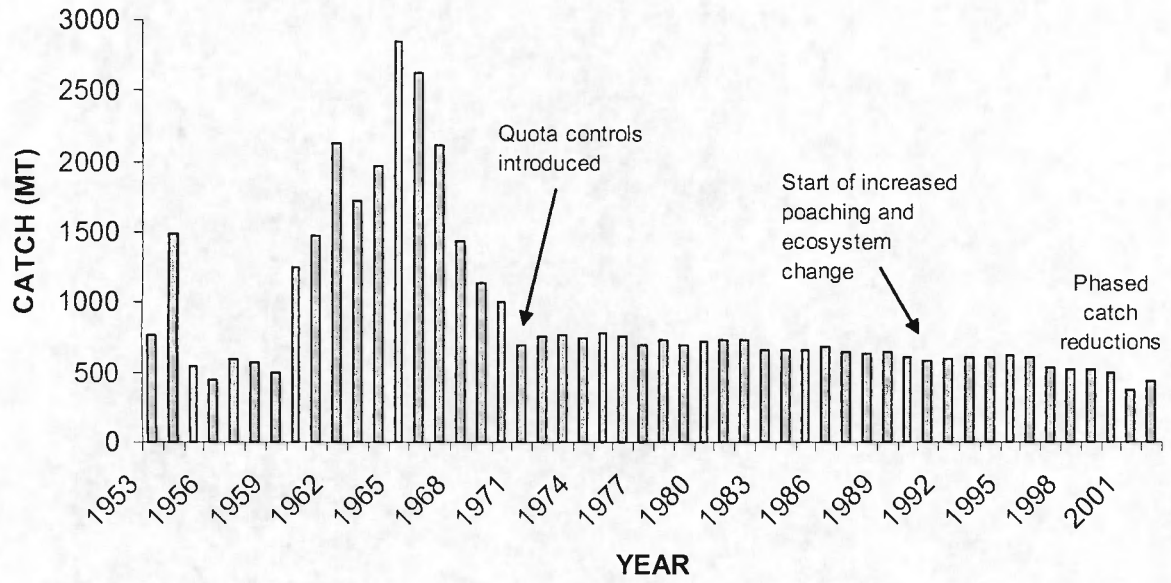


Fig. 2.2. Total commercial catches (whole wet mass in MT) of abalone *Haliotis midae* from 1953 to present.

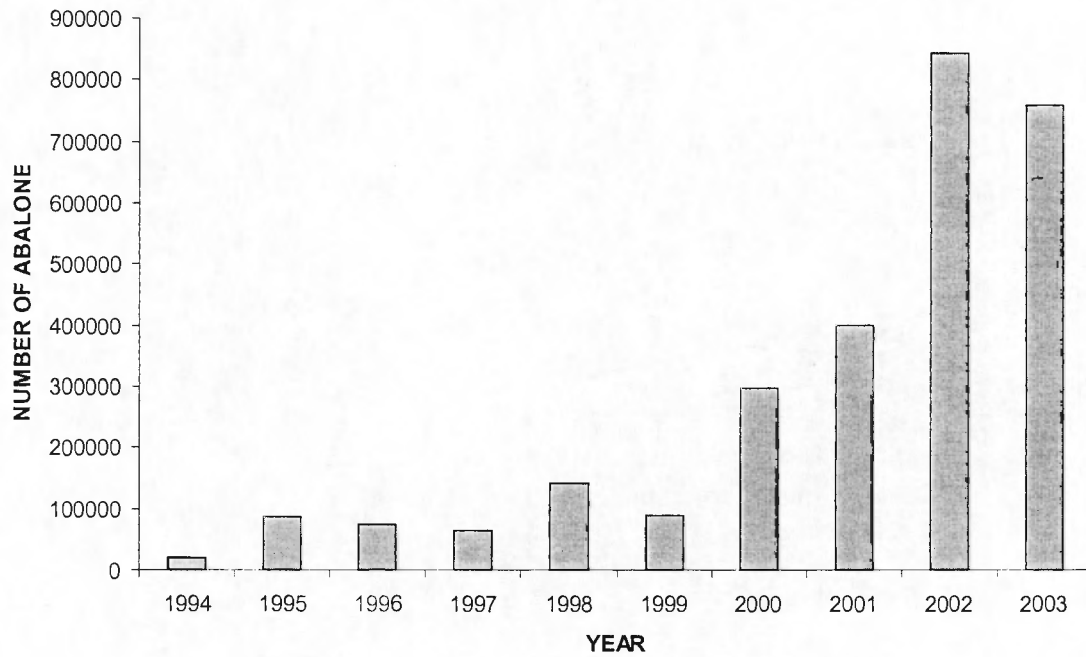


Fig. 2.3. The total number of poached abalone confiscated by law enforcement officers in South Africa from 1994 to present (data from A. MacKenzie, MCM, pers. commn).

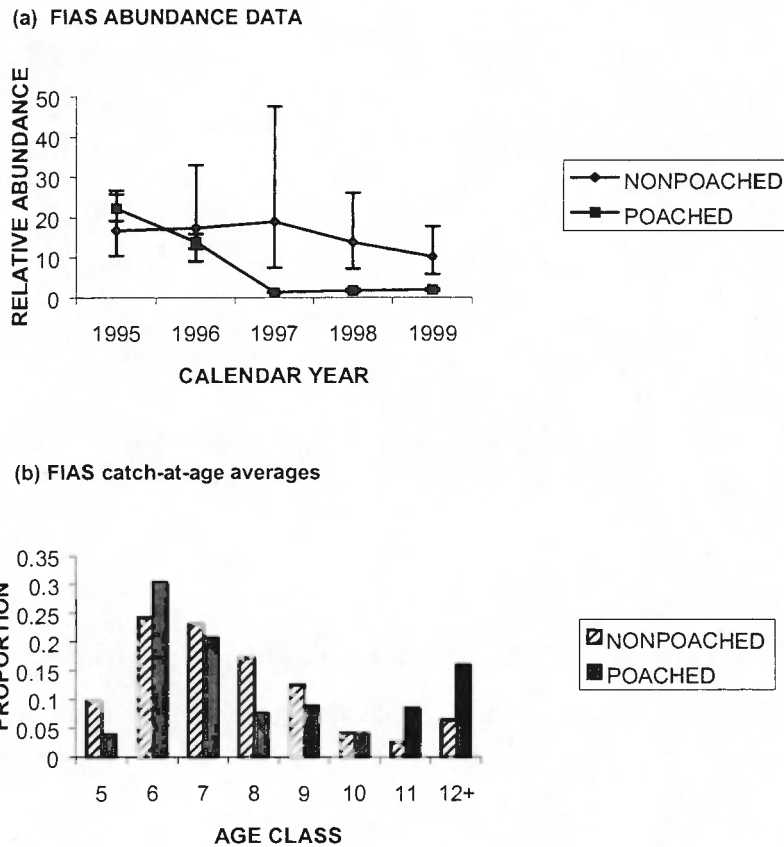


Fig. 2.4. Comparisons between FIAS data obtained for the “poached” (CP) and “nonpoached” (CNP) subareas of Zone C. (a) shows the trend in the relative abundance index (average number  $\bar{x}$  per 60 m<sup>2</sup>) in each subarea. The error bars indicate the 95% confidence intervals calculated as  $\bar{x} * e^{\pm 1.96 \cdot CV}$ , i.e. assuming a lognormal error distribution. (b) compares the catch-at-age data averaged over the period 1995-1999. The catch-at-age data are derived from length distribution data (from A. McKenzie, MCM) that is cohort-sliced using a von Bertalanffy growth curve (Tarr 1995).

## Chapter 3

### Collating available data for the South African abalone *Haliotis midae* stock assessment in Zones A, B, C and D

#### SUMMARY

A large amount of data are available for each of the main abalone fishery Zones A-D. Zone C has been split into two subareas, a “poached” subarea (CP) to the west and a “nonpoached” subarea (CNP) to the east, because of substantial differences in the extent of poaching in these subareas. All available data have been reworked in terms of a standard Model year  $y$  that is taken to run from October of year  $y-1$  to September of year  $y$ . Commercial catch data (in terms of tonnes) are available from 1953 whereas recreational catch data (in terms of numbers) have been collected since 1992. Available indices of abundance for each Zone include a commercial CPUE series (from 1980 to present) and a relative index of abundance from research surveys (FIAS) conducted from 1995 to present. There is a very large amount of catch-at-age data available per Zone from the various fishery sectors and research surveys. An important additional data source derives from an industry and MCM co-operative diving survey in 2002 that was carried out *inter alia* to provide information on recruitment strength and population structure in key fishing Zones B and C. These full population size composition data were first incorporated in the 2002 abalone stock assessment model. Additional data that are currently used in a diagnostic context relate to the extent of population depletion below pristine levels.

#### 3.1 INTRODUCTION

The commercially fished area for the South African abalone fishery has been divided into seven fishing Zones, with total allowable catches (TACs) set separately for each Zone over the last 27 years. The main fishing areas are Zones A-D (Fig. 2.1). Given that they are the focus of the modelling efforts reported in this study, attention has focused on collating all available data for these zones in particular. Unless otherwise specified, the data described here have been based on, and reworked from, raw data supplied by MCM. These data are typically recorded as corresponding to area names located within the various Zones, and a list of these is given as Appendix 3.1.

Several historic changes in the commencement and closure dates for the commercial fishing season necessitated all available data being reworked in terms of a standard Model (fishing) year  $y$  that is taken to run from October of year  $y-1$  to September of year  $y$  (hereafter referred to as Model year  $y$ ). Furthermore, early analyses in this study showed that it was difficult to interpret the abalone dynamics in Zone C without consideration that the western and eastern areas of this Zone differed with respect to both density and size composition, to a large

degree because of the different levels of poaching in the two. For assessment purposes, Zone C has therefore been split into a “poached” subarea (CP) to the west (Hawston/Mudge Point areas) and a “nonpoached” subarea (CNP) to the east (Hermanus vicinity) (Fig. 2.1). Although some poaching is thought to occur in the “nonpoached” subarea, this is considered to be appreciably less than that in the “poached” subarea. The two areas are approximately equivalent in terms of available habitat for abalone because the kelp bed area estimates for areas CP and CNP are 1.38 and 1.59 million square metres respectively (Tarr 1993). In terms of the length of coastline, CP is approximately half of CNP’s length.

Note that the data for Zone C (the commercial CPUE data in particular) exhibit more contrast than the data for the other Zones so that Zone C has been considered the base-case in all analyses, with the stock assessment model being applied first to this area as the contrast suggests better potential for precise parameter estimation. Such estimated parameter values are then input to applications of the model to other Zones where data contrast is insufficient to allow their independent estimation with great reliability.

## **3.2 CATCH DATA**

### **3.2.1. Commercial catches**

Total allowable catches (TACs) for the commercial fishery were set individually for each of seven fishing Zones A to G from the 1986/7 season onwards (Tarr 1992). Moreover, data on the commercial catch of abalone in Zones A-D are available for the period 1977 to 1985. However, prior to 1972 catch data are available only for all Zones (A-G) combined, and hence zonal catch estimates for the period 1953 to 1976 are assumed to be a fixed proportion  $p_c$  of this total annual catch, where  $p_c$  is taken to be the average of a Zone’s proportional contribution to total catch over the period 1977 to 1981. The same approach is used to apportion the Zone C catch estimates for the period 1953 to 1976 between the two subareas CP and CNP. Catches over 1968-1982 have been increased by 15.2% following the suggestion of Tarr (2002b). Moreover, to account for catches pre-1953, commercial catches are assumed to have increased linearly from zero in 1950 to the existing 1953 values. The resultant time series of total commercial catches for Zones A - D as a whole are listed in Table 3.1.

### 3.2.2. Recreational catches

Table 3.2 shows recreational catch estimates in terms of numbers of abalone caught. These are estimated from telephonic surveys, conducted since 1992, of selected recreational permit-holders. Moreover, although estimated recreational catches (i.t.o numbers) per Zone are available from 1992 to 2002, data on the number of recreational permits are available for the period 1989 to 1991. The number of recreational permits issued in each of the years 1989, 1990 and 1991 were respectively 0.74, 0.80 and 0.93 times the number of permits issued in 1992. It is therefore assumed that recreational catches per Zone over this period were simply the above proportions of the 1992 recreational catch estimates per Zone (see Table 3.2). In contrast to the commercial fishery, where a law in force since 1966 prohibits commercial abalone fishing operations within 185m of the high-water mark (Dichmont *et al.* 2000), the recreational fishery is essentially a shallow-water fishery with divers accessing the resource mostly from the shore. The relative proportions of the Zone C recreational catch taken from the two subareas CP and CNP was therefore assumed to be proportional to the relative lengths of the coastline (CP:CNP = 1:2).

### 3.2.3 Poaching confiscation data

Abalone poaching confiscation data compiled by Marine & Coastal Management are available from 1994 onwards and are used as inputs to the stock assessment exercise as described in the next chapter.

### 3.2.4 Subtracting catches in the model – when are catches landed?

Total annual catch in each of the Zone C subareas is calculated in the model as the sum of the commercial, recreational and model-estimated poached catches. Analysis of the proportion of the total commercial catch caught during a season indicates that the majority of commercial catches are landed during the first two quarters of the Model year (ie. October to March) (Fig. 3.3). The fishery is therefore modelled as a pulse catch after the first quarter of the Model year, i.e. at the start of the calendar year. Prior to 1985, an overall catch limit was allocated for abalone and recorded by calendar year (Dichmont *et al.* 2000) so that the proportion of the total catch taken in the last quarter of a calendar year was relatively minor. It may therefore safely be



assumed that no adjustment is necessary to equate catches recorded for a calendar year prior to 1980 with the catch corresponding to a Model year running from October to September.

The recreational fishing season during the early 1990s ran from November to July, but as a result of large increases in the recreational catch, the recreational fishing season was reduced from 9 months to 6 months (1 November – 30 April) in 1996/97 and then reduced further to 4 months (12 December – 13 April) as from 1997/98 and restricted to weekend and public holidays only from the 1998/99 season. From a modelling perspective, the implications of the recreational fishing seasons are that no adjustments are necessary to equate recreational catch estimates with the catch corresponding to a Model year as defined above. Moreover, because most recreational fishing is likely to take place during the austral summer months, it seems reasonable to model the recreational catch also as a pulse catch after the first quarter of the Model year.

The same approach is used to model the poaching catch, supported by the observation of a slight decrease in poaching confiscations during the winter months (A. MacKenzie, pers. commn).

### **3.3 INDICES OF ABUNDANCE**

#### **3.3.1 Commercial CPUE data**

Standardised commercial CPUE data are available for the period 1980-2002 for each of Zones A, B, D and the two subareas constituting Zone C and are presented in Table 5.5 (see Chapter 5).

#### **3.3.2 Historic index of abundance**

In 2002, it was recommended that the historical factory processor records for the abalone resource be re-examined to determine whether they contain information that could be used to enhance the assessment (BENEFIT 2002). Data from Newman (1973) on whole mass per licensed diver month for Zones A-D combined were therefore sourced to potentially be used as a separate CPUE index for the period 1960-1970 (Table 3.4). These historic data are in the

form of annual production figures from 1953 to 1970, with estimates of whole mass and rough effort values having been computed for some of this period.

Tarr (2002b) extracted the following comments of interest from the

Based on a review of the work of Newman (1973), Tarr (2002b) noted that the low catches pre-1959 were attributed to a lack of export demand rather than declining stocks. Furthermore, the decline in catches post-1965 were attributed to a decline in abalone abundance such that the quota was not filled in 1968 and 1969, but was first filled in 1970. Newman (1973) concluded that abalone initially occurred in very dense concentrations with early catches having to a large extent been dependent on these accumulated stocks.

### 3.3.3 FIAS (Fishery Independent Abalone Surveys)

An important additional data source for the abalone resource is the Fishery Independent Abalone Surveys (FIAS) conducted since 1995 (Table 3.5). These surveys were designed to provide an index of relative abundance with a CV of some 25% (which is substantially more precise than that achieved in earlier surveys) (Dichmont *et al.* 2000). Only animals larger than 100mm shell length are recorded in these surveys so as to reduce uncertainty in the estimates due to the non-emergent/cryptic behaviour of juveniles. This corresponds to a minimum sampling age of approximately 5 years. Because the “poached” subarea CP includes only approximately one-third of Zone C’s coastline, 6 of the 20 FIAS transects are located within subarea CP compared to the 14 transects located within subarea CNP. This obviously has some implications on the CVs attached to the “split” estimates, with CVs in subarea CP ranging between 38 - 67% compared to 24 - 47% for CNP (Table 3.5).

## 3.4 LENGTH DISTRIBUTION DATA

### 3.4.1 Length-breadth-mass conversions

A von Bertalanffy growth equation is used to relate shell length  $\ell$  (mm) to age in years ( $t$ ), and is based on tagging data from Betty's Bay (Tarr 1995):

$$\ell(t) = \ell_{\infty} [1 - e^{-k(t-t_0)}] \quad (3.1)$$

Shell length  $\ell$  is converted to shell breadth  $SB$  (mm) using the relationship:

$$SB = 0.913 * \ell(t) - 11.59 \quad (3.2)$$

The conversion from shell breadth ( $SB$ ) to abalone whole wet mass (g) is based on that given in Tarr (1993):

$$w_{y,a} = w(y, t = a) = c \cdot (SB)^d \quad (3.3)$$

The above relationships were derived based on a total of 517 data points from 1994 for the combined Betty's Bay and Danger Point areas (Tarr 1993). These data were reanalyzed as part of this study because it was deemed possible (and preferable) to directly relate shell length (the measure used in the von Bertalanffy equation) to abalone whole mass. A power curve was fitted ( $r_{0.01,516} = 0.99$ ,  $p < 0.001$ ) using the statistical package SPSS v. 10.1 and it was recommended that in future model applications mass-at-age be computed using a combination of the von Bertalanffy equation and the following equation relating *shell length* (mm) to abalone whole wet mass (g):

$$w_{y,a} = w(y, t = a) = 0.000098 \cdot (\ell)^{3.1549} \quad (3.4)$$

### 3.4.2 Catch-at-age data – commercial and survey data

The catch-at-age data are derived from length distribution data that are cohort-sliced using a von Bertalanffy growth curve (from Tarr 1995) that was fitted to tagging data. For reference purposes, the age-length key from Moloney and MacKenzie (2001) is given as Appendix 3.2 of this document. Although “cohort-slicing” is a widely employed technique in fisheries stock assessment world-wide, it is recognised that this approach is nonetheless not particularly satisfactory (see discussion in Chapter 10). Some progress has been made in fitting directly to size-composition data instead, but this was not included in the 2003 assessment model being described here and hence the presentation of input data in this chapter is restricted to those inputs actually used in 2003.

The catch-at-age data used in the model-fitting process include all available data from the commercial, recreational and poaching sectors (the last from confiscations), as well as from the fishery-independent and industry surveys. In several instances age classes have been lumped together to reduce the number of categories containing a proportional abundance less than 2% in any one year. A summary of the various data series available for Zone C is shown in Table 3.6 with the complete set of catch-at-age data being presented in Tables 3.7 – 3.14.

All of the data have been computed based on length distribution data that are given as shell breadths, except for the FIAS data (Tables 3.9 and 3.10), which are based on length distribution data in the form of shell length.

Confiscated samples from the poaching sector have been analysed by grouping them into mass categories based on the dry shucked mass (minus shell) of the animals (Table 3.11). In this case, shucked mass  $w_{shuck}$  is converted to shell length  $\ell$  using the following relation (Moloney & MacKenzie 2001):

$$w_{shuck} = 2 \times 10^{-5} * \ell^{3.2209} \quad (3.5)$$

An additional source of data is in the form of inshore and offshore catch-at-age data available from the “old surveys” conducted in the 1980’s (Tables 3.13 and 3.14). In the survey, abalone were counted 1.5m on either side of a 200m transect. A separate sample was collected for length frequency investigations. The 0-5 m data are assumed to apply to the inshore area as considered in the model, whereas the 5-10 m and 10-15 m data are assumed to apply to the offshore area (see Chapter 6).

### 3.4.3 Full population survey

In 2002, scientific consultants appointed by the abalone fishing industry conducted a diving survey in cooperation with MCM to provide information on recruitment strength and population structure in the key fishing Zones B and C (OLRAC 2002). These length distribution data (Table 3.15, Fig. 3.3) are particularly useful because:

- a) they sampled the full population of abalone (whereas the minimum sampling age of FIAS is approximately 5 years and that of the commercial and recreational

- sectors is approximately 8-9 years). Unlike other surveys, “destructive” sampling was used (all animals extracted) to guard against size-biased counting underwater;
- b) they sampled both inshore and offshore components of the habitat (whereas other fishery sectors/surveys are concentrated either inshore or offshore); and
  - c) they sampled both a “lobster-invaded” and “normal” Zone for purposes of comparing recruitment patterns in the two Zones.

### 3.5 DEPLETION ESTIMATE COMPARISONS

#### 3.5.1 Data on relative abalone densities

Due to a concern expressed in 2001 that the abalone stock assessment models should be reflecting a greater level of depletion relative to pristine, rough depletion statistics were obtained from available data for purposes of calibrating / cross-comparing with the model. The perception of some divers and scientists with several years' familiarity with the abalone resource is that " the present level of the resource should be reflecting a highly depressed level relative to pristine" (Tarr 2002b). This perception is based to some extent on visual comparisons of the *numbers* (per unit area) of abalone occurring in a region *relative* to the numbers in a *pristine reserve area* such as Betty's Bay. The following is a simple analysis that was done in an attempt to quantify this perception.

Tarr (2002a) summarises the typical abalone density and size range per 600m<sup>2</sup> transect encountered during the “old surveys” of the 1980’s. The old surveys also included counts within Betty's Bay (BB), a marine reserve that is considered to have been representative during the 1980’s (in terms of e.g. suitable habitat for abalone) of the rest of the south coast (R. Tarr, pers commn). It was proclaimed a reserve in 1966 and the poaching there is thought to have escalated only over the last few years. If we thus assume that (as with most of the other areas – see Chapter 3) poaching in BB escalated appreciably only after 1996, then it is reasonable to assume that until that time abalone densities in BB were approximately representative of pristine abalone densities along the whole south coast region.

The "old survey” density estimates are used in a preliminary analysis to provide information on relative density in the early 1980' s in BB compared to each of Zones A-D. Assuming that

BB densities represent pristine densities, the relative densities are then used as a proxy for % depletion in the early 1980's in Zones A-D (Table 3.12). This analysis was repeated using 1990's FIAS data on inshore numbers. FIAS data for 1995, as well as 1995-1996 combined, from the inshore BB region were assumed to represent pristine densities. A measure of % depletion was obtained by comparing a Zone's density to the BB density (pre-poaching).

Note that these density estimates are not considered as comparable in an absolute sense to measures such as FIAS in the 1990's. The density values as summarised in Table 3.12 are used as a possible "sensitivity" for exploration in model fits as well as to make relative comparisons between inshore and offshore densities in a Zone for the sampling year indicated. Absolute comparisons across the decades could be checked in a "diagnostic" context.

**Appendix 3.1. List of area names and corresponding Zones**

Area name	Area no.	Area name	Zones
BUFFELSJACHT	3	BUFFELSJACHT	A
CELT BAY	3	BUFFELSJACHT	A
DE DAM	3	BUFFELSJACHT	A
HOEWALLE	3	BUFFELSJACHT	A
JESSE POINT	3	BUFFELSJACHT	A
OUBAAI (QUOIN POINT)	3	BUFFELSJACHT	A
PLAATJIESKRAAL	3	BUFFELSJACHT	A
QUOIN POINT	3	BUFFELSJACHT	A
QUOIN POINT	3	QUOIN POINT	A
SANDBAAI (QUOIN PNT)	3	BUFFELSJACHT	A
BANTAMSKLIP	4	SOETFONTEIN	A
CELT BAY	4	SOETFONTEIN	A
SOETFONTEIN	4	SOETFONTEIN	A
STEENBRASBAAI	4	SOETFONTEIN	A
CLYDE ISLAND	5	PEARLY BEACH	DYRS
DIE GRUIS	5	PEARLY BEACH	DYRS
DONKERGAT (PEARLY B)	5	PEARLY BEACH	A
DYER ISLAND	5	PEARLY BEACH	A
HOLBAAI (PEARLY BCH)	5	PEARLY BEACH	A
PEARLY BEACH	5	PEARLY BEACH	A
PEARLY BEACH	5	PEARLY BEACH	A
SANDY POINT	5	PEARLY BEACH	A
UILSKRAALRIVIERMOND	5	PEARLY BEACH	A
DANGER POINT	6	GANSBAAI	B
DIE KELDERS	6	OUDEKRAAL	B
FRANSKRAALSTRAND	6	GANSBAAI	B
GANSBAAI	6	GANSBAAI	B
KLEINBAAI	6	GANSBAAI	B
KRUISMANSBAAI	6	GANSBAAI	B
ROMANSBAAI	6	GANSBAAI	B
STANFORD'S COVE	6	GANSBAAI	B
TIERGAT	6	GANSBAAI	B
VAN DYKSBAAI	6	GANSBAAI	B
HERMANUS	7	HERMANUS	C
MELKBOS(ONRUSRIVIER)	7	HERMANUS	C
ONRUSRIVIER	7	HERMANUS	C
SANDBAAI (HERMANUS)	7	HERMANUS	C
SKULPHOEK	7	HERMANUS	C
SWARTDAM	7	HERMANUS	C
HAWSTON	8	HAWSTON	C
MELKBOS(MUDGE POINT)	8	HAWSTON	C
MUDGE POINT	8	HAWSTON	C
NUWE BAAI	8	HAWSTON	C
ONRUS	8	HAWSTON	C
PLANK HUIS	8	HAWSTON	C
VERMONT	8	HAWSTON	C
DAWIDSKRAALBAAI	9	KLEINMOND	D
DE WETSBAAI	9	KLEINMOND	D
DIE MOND (PALMIET)	9	KLEINMOND	D
KLEINMOND	9	KLEINMOND	D
PALMIET	9	KLEINMOND	D

VOELSTEEN(KLEINMOND)	9	KLEINMOND	D
AASBANK	10	BETTY'S BAY	D
BETTY'S BAY	10	BETTY'S BAY	D
BOSKOP	10	BETTY'S BAY	D
DRAADBAAI	10	BETTY'S BAY	D
SILVERSANDS BAY	10	BETTY'S BAY	D
BASBAAI (HANGKLIP)	11	HANGKLIP	D
BLOKBAAI (HANGKLIP)	11	HANGKLIP	D
DORINGBAAI	11	HANGKLIP	D
GROOTBAAI (HANGKLIP)	11	HANGKLIP	D
HOLBAAI (HANGKLIP)	11	HANGKLIP	D
HOLBAAIPUNT	11	HANGKLIP	D
KAAP HANGKLIP	11	HANGKLIP	D
MAASBAAI (HANGKLIP)	11	HANGKLIP	D
OSPUNT	11	HANGKLIP	D
PRINGLE BAY	11	HANGKLIP	D
MELKBAAI	12	ROOIELS	D
ROOIELSBAAI	12	ROOIELS	D
BUFFELSBAAI	13	MILLER'S POINT	E
CAPE POINT	13	MILLER'S POINT	E
MILLER'S POINT	13	MILLER'S POINT	E
ROOIKRANS	13	MILLER'S POINT	E
SMITSWINKELBAAI(C P)	13	MILLER'S POINT	E
CAPE MACLEAR	14	KOMMETJIE	E
CAPE POINT	14	KOMMETJIE	E
CHAPMAN'S POINT	14	KOMMETJIE	E
GIFKOMMETJIE	14	KOMMETJIE	E
KAAP DIE GOEIE HOOP	14	KOMMETJIE	E
KOMMETJIE	14	KOMMETJIE	E
PLATBOOMPUNT	14	KOMMETJIE	E
S.W. RIDGES	14	KOMMETJIE	E
WITSANDBAAI	14	KOMMETJIE	E
DUIKEREILAND	15	TABLE BAY	E
HOUT BAY	15	OUDEKRAAL	E
SANDY BAY	15	OUDEKRAAL	E
SEA POINT	15	OUDEKRAAL	E
MELKBOSSTRAND	16	MELKBOSSTRAND	G
MURRAY'S BAY	16	MELKBOSSTRAND	G
ROBBEN ISLAND	16	MELKBOSSTRAND	F
GANZEKRAAL	17	BOKPUNT	G
HOSPITAL BAY	17	BOKPUNT	G
JACOBSBAAI	17	BOKPUNT	G
BOOMPUNT	18	SALDANHA	G
DANGER BAY	18	SALDANHA	G
DASSEN EILAND	18	SALDANHA	G
HOUSE BAY	18	SALDANHA	G
ICHABO POINT	18	SALDANHA	G
YSTERFONTEINPUNT	18	SALDANHA	G



**Appendix 3.2. Age-length key for abalone.**

shell width (mm)	shell length (mm)	Age classes (y)													
		2	3	4	5	6	7	8	9	10	11	12	13	14	>15
29	14.9	1.00													
30	15.8	1.00													
37	22.2	1.00													
38	23.1	1.00													
39	24.0	1.00													
47	31.3	0.06	0.94												
48	32.2		1.00												
55	38.6		1.00												
56	39.5		1.00												
63	45.9		0.88	0.12											
64	46.8			1.00											
65	47.8			1.00											
75	56.9			1.00											
76	57.8			1.00											
77	97.0			0.84	0.16										
78	98.1				1.0										
79	99.2				1.0										
80	100.3				1.0										
81	101.4				1.0										
82	102.5				1.0										
83	103.6				1.0										
84	104.7				1.0										
85	105.8				1.0										
86	106.9				1.0										
87	108.0				1.0										
88	109.1				1.0										
89	110.2				0.43	0.57									
90	111.3					1.0									
91	112.4					1.0									
92	113.5					1.0									
93	114.6					1.0									
94	115.7					1.0									
95	116.7					1.0									
96	117.8					1.0									
97	118.9					1.0									
98	120.0					1.0									
99	121.1					0.06	0.94								
100	122.2						1.0								
101	123.3						1.0								
102	124.4						1.0								
103	125.5						1.0								
104	126.6						1.0								
105	127.7						1.0								
106	128.8						1.0								
107	129.9						0.05	0.95							
108	131.0							1.0							
109	132.1							1.0							
110	133.2							1.0							
111	134.3							1.0							

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112	135.4			1.0					
113	136.5			0.68	0.32				
114	137.6				1.0				
115	138.7				1.0				
116	139.7				1.0				
117	140.8				1.0				
118	141.9				1.0				
119	143.0			0.19	0.81				
120	144.1				1.0				
121	145.2				1.0				
122	146.3				1.0				
123	147.4				0.76	0.24			
124	148.5					1.0			
125	149.6					1.0			
126	150.7					1.0			
127	151.8					0.56	0.44		
128	152.9						1.0		
129	154.0						1.0		
130	155.1						0.71	0.29	
131	156.2							1.0	
132	157.3							1.0	
133	158.4							0.33	0.67
134	159.5								1.0
135	160.6								0.50
136	161.7								1.0
137	162.7								1.0
138	163.8								1.0
139	164.9								1.0
140	166.0								1.0
141	167.1								1.0
142	168.2								1.0
143	169.3								1.0
144	170.4								1.0
145	171.5								1.0
146	172.6								1.0
147	173.7								1.0
148	174.8								1.0
149	175.9								1.0
150	177.0								1.0
151	178.1								1.0
152	179.2								1.0
153	180.3								1.0
154	181.4								1.0

Table 3.1. Total commercial catches (MT) from 1950 to 2003 for Zones A, B, C (subareas CNP and CP) and D. Values for 1953 to 1976 are estimated as a proportion of total annual commercial catch (for all Zones combined). Commercial catches are assumed to start from zero in 1950 with a linear trend up to existing 1953 values (to account for unknown catches pre-1953). Values shown for 2003 are the current TAC (including the allocations to the small-scale commercials and the amount held for appeals). The 15.2% CF (from Tarr 2002b) has been added to the 1968-82 data.

Year	A	B	CNP	CP	D	Comments
1950	0	0	0	0	0	Linear trend assumed
1951	37	66	17	54	67	
1952	75	132	34	109	133	
1953	112	198	51	163	200	
1954	215	381	99	312	384	
1955	78	139	36	114	140	
1956	65	114	30	94	115	
1957	86	153	40	125	154	
1958	83	148	38	121	149	
1959	72	128	33	105	129	
1960	181	321	83	263	323	
1961	213	378	98	311	381	
1962	309	549	142	450	553	
1963	249	442	114	362	445	
1964	286	507	131	416	511	
1965	413	732	190	601	738	
1966	380	675	175	553	680	
1967	307	545	141	447	550	
1968	238	424	109	348	427	15% CF added
1969	190	336	88	275	339	15% CF added
1970	167	296	76	243	298	15% CF added
1971	116	206	53	169	207	15% CF added
1972	126	223	58	183	226	15% CF added
1973	128	227	59	187	229	15% CF added
1974	124	220	58	181	222	15% CF added
1975	129	229	60	188	232	15% CF added
1976	126	222	58	183	225	15% CF added
1977	127	176	48	153	197	15% CF added
1978	83	258	58	182	227	15% CF added
1979	78	206	65	203	234	15% CF added
1980	165	202	15	181	223	15% CF added
1981	146	221	40	154	188	15% CF added
1982	172	209	28	149	214	15% CF added
1983	162	187	33.9	103	75	
1984	188.9	194.9	67.7	84.4	106.5	
1985	109.7	162.4	40.5	98.7	156.8	
1986	135.1	229.4	57.3	120.6	50.3	
1987	174.9	166.2	29.7	126.4	45.2	
1988	194.9	138.7	26.2	139.5	49.2	
1989	190.9	137.3	27.8	135.6	52.1	
1990	199.2	142.4	40.3	116.6	46.3	
1991	183.8	138	41	119	49.8	
1992	226.6	147.5	30.5	113.9	56.3	
1993	194.8	152.7	31.9	105.2	53.7	
1994	218.3	150	44.1	91.3	92.5	
1995	235.6	152.3	39.1	84.7	90.2	
1996	232.6	146.8	66.8	62.8	90.9	
1997	221.5	145.9	37.2	16.1	93.2	
1998	189.8	148.3	24.6	7.8	108.3	
1999	184.1	146.7	11.8	1.1	104.7	
2000	184.6	144.1	4.8	0	105.2	
2001	163.4	135.2	0	0	67.3	Note small scale added
2002	124.0 (136)	103.3 (110)	52 (52)	0	47.1 (50)	Note Small scale added; TAC amount allocated shown in ()
2003 (TAC)	131.3	53.5	24.9	0	32	Note small scale added

Table 3.2. The estimated recreational catch (in numbers) input into the models for Zones A, B, C and D. Values for 2000 are assumed to be the average of the 1998 and 1999 estimates (note that all estimated quantities are shown in italics in the Table below). Note that, consistent with changes to the historic commercial catches, historic recreational catches are assumed to have commenced in 1951. Actual estimates of recreational catches (i.t.o numbers) per Zone are available from 1992 to 2002 and are neither italicised nor marked in bold in the Table below. The recreational catch per Zone for the years 1989, 1990 and 1991 (shown in bold type) are computed as proportions of the 1992 Zonal recreational catch estimates, based on the proportion of recreational permits issued in each of the these years (0.74, 0.80 and 0.93 respectively) relative to the number of permits issued in 1992. The 1970 value is set as 25% of the 1989 level for each Zone and a linear increase in recreational catches is then assumed from zero in 1950 to the 1970 value and then a second linear increase from the 1970 value to the existing 1989 value for each Zone (linear interpolations shown in italics).

Year	Zone				Comment
	A	B	C	D	
1950	0	0	0	0	
1951	<i>344</i>	<i>705</i>	<i>587</i>	<i>1684</i>	
1952	<i>689</i>	<i>1410</i>	<i>1174</i>	<i>3368</i>	
1953	<i>1033</i>	<i>2114</i>	<i>1761</i>	<i>5052</i>	
1954	<i>1378</i>	<i>2819</i>	<i>2348</i>	<i>6736</i>	
1955	<i>1722</i>	<i>3524</i>	<i>2935</i>	<i>8420</i>	
1956	<i>2066</i>	<i>4229</i>	<i>3522</i>	<i>10104</i>	
1957	<i>2411</i>	<i>4934</i>	<i>4110</i>	<i>11787</i>	
1958	<i>2755</i>	<i>5639</i>	<i>4697</i>	<i>13471</i>	
1959	<i>3100</i>	<i>6343</i>	<i>5284</i>	<i>15155</i>	
1960	<i>3444</i>	<i>7048</i>	<i>5871</i>	<i>16839</i>	
1961	<i>3788</i>	<i>7753</i>	<i>6458</i>	<i>18523</i>	
1962	<i>4133</i>	<i>8458</i>	<i>7045</i>	<i>20207</i>	
1963	<i>4477</i>	<i>9163</i>	<i>7632</i>	<i>21891</i>	
1964	<i>4822</i>	<i>9868</i>	<i>8219</i>	<i>23575</i>	
1965	<i>5166</i>	<i>10572</i>	<i>8806</i>	<i>25259</i>	
1966	<i>5510</i>	<i>11277</i>	<i>9393</i>	<i>26943</i>	
1967	<i>5855</i>	<i>11982</i>	<i>9980</i>	<i>28627</i>	
1968	<i>6199</i>	<i>12687</i>	<i>10567</i>	<i>30311</i>	
1969	<i>6544</i>	<i>13392</i>	<i>11155</i>	<i>31995</i>	
1970	<i>6888</i>	<i>14097</i>	<i>11742</i>	<i>33679</i>	
1971	<i>7976</i>	<i>16322</i>	<i>13596</i>	<i>38996</i>	
1972	<i>9063</i>	<i>18548</i>	<i>15449</i>	<i>44314</i>	
1973	<i>10151</i>	<i>20774</i>	<i>17303</i>	<i>49631</i>	
1974	<i>11238</i>	<i>23000</i>	<i>19157</i>	<i>54949</i>	
1975	<i>12326</i>	<i>25225</i>	<i>21011</i>	<i>60267</i>	
1976	<i>13413</i>	<i>27451</i>	<i>22865</i>	<i>65584</i>	
1977	<i>14501</i>	<i>29677</i>	<i>24719</i>	<i>70902</i>	
1978	<i>15589</i>	<i>31903</i>	<i>26573</i>	<i>76220</i>	
1979	<i>16676</i>	<i>34128</i>	<i>28427</i>	<i>81537</i>	
1980	<i>17764</i>	<i>36354</i>	<i>30281</i>	<i>86855</i>	
1981	<i>18851</i>	<i>38580</i>	<i>32135</i>	<i>92173</i>	
1982	<i>19939</i>	<i>40806</i>	<i>33989</i>	<i>97490</i>	
1983	<i>21026</i>	<i>43032</i>	<i>35843</i>	<i>102808</i>	
1984	<i>22114</i>	<i>45257</i>	<i>37697</i>	<i>108126</i>	
1985	<i>23202</i>	<i>47483</i>	<i>39551</i>	<i>113443</i>	
1986	<i>24289</i>	<i>49709</i>	<i>41405</i>	<i>118761</i>	
1987	<i>25377</i>	<i>51935</i>	<i>43259</i>	<i>124079</i>	
1988	<i>26464</i>	<i>54160</i>	<i>45112</i>	<i>129396</i>	
1989	<b>27552</b>	<b>56386</b>	<b>46966</b>	<b>134714</b>	
1990	<b>29910</b>	<b>61213</b>	<b>50987</b>	<b>146246</b>	
1991	<b>34866</b>	<b>71354</b>	<b>59434</b>	<b>170475</b>	
1992	37427	76596	63800	182998	
1993	81679	73706	119318	172079	
1994	63347	144593	100682	273770	
1995	26871	47013	118093	215814	
1996	17319	133890	99411	180566	
1997	42019	137729	130370	189440	
1998	27273	45652	17736	120077	
1999	13183	56679	20780	50730	
2000	20228	51165	19258	85403	Mean of 1998 & 1999 as season same 4 month
2001	9963	42385	2031	46930	
2002	11987	28947	4747	50443	
2003	10975	35666	3389	48686.5	Mean of 2001 & 2002 as season same 11 days

Table 3.3 The proportion (by mass) of the total commercial catch from 1980 to present caught during a season as shown (for fishing Zones A-D combined). Fishing years  $y$  are taken to run from October of year  $y-1$  to September of year  $y$ .

FISHING YR	SEASON				4&1
	4 (Oct – Dec)	1 (Jan – Mar)	2 (Apr – Jun)	3 (Jul - Sep)	
1980	0.00	0.55	0.19	0.26	0.55
1981	0.02	0.59	0.16	0.23	0.61
1982	0.11	0.51	0.19	0.19	0.62
1983	0.06	0.43	0.22	0.29	0.49
1984	0.00	0.55	0.23	0.22	0.55
1985	0.07	0.53	0.20	0.20	0.60
1986	0.48	0.27	0.22	0.03	0.74
1987	0.50	0.30	0.20	0.00	0.80
1988	0.48	0.29	0.22	0.01	0.77
1989	0.56	0.33	0.11	0.00	0.89
1990	0.45	0.54	0.01	0.00	0.99
1991	0.58	0.29	0.10	0.04	0.86
1992	0.56	0.35	0.01	0.08	0.91
1993	0.85	0.15	0.01	0.00	0.99
1994	0.78	0.19	0.02	0.02	0.97
1995	0.75	0.18	0.06	0.01	0.93
1996	0.14	0.72	0.10	0.04	0.86
1997	0.40	0.16	0.05	0.40	0.55
1998	0.02	0.59	0.27	0.12	0.61

Table 3.4. Historic CPUE index calculated as whole mass per licensed diver month (from Newman 1973) for Zones A-D combined.

Year	Production (thousands of pounds)	Metric tons	Whole mass (24% conversion)	Effort (licensed diver months)	Whole mass (tons) per licensed diver month
1953	387	176	733		
1954	783	356	1483		
1955	285	130	540		
1956	236	107	447		
1957	314	143	595		
1958	304	138	576		
1959	264	120	500		
1960	659	300	1248	328	3.8
1961	778	354	1473	363	4.1
1962	1129	513	2138	628	3.4
1963	908	413	1720	597	2.9
1964	1042	474	1973	737	2.7
1965	1508	685	2856	1393	2.1
1966	1388	631	2629	1399	1.9
1967	1122	510	2125	1246	1.7
1968	757	344	1434	1027	1.4
1969	600	273	1136	947	1.2
1970	528	240	1000	750	1.3

Table 3.5. FIAS-based indices of abundance used in the model versions applied to Zones A, B, C (subareas CNP and CP) and D. Abundance estimates are expressed as the mean number  $\bar{x}$  (and associated standard error) of abalone per 60 m<sup>2</sup>, calculated by averaging over the indicated number of 30m x 2m transects. As only half of the transects in Zone B were sampled in each of the years 1999 and 2000, the average of the density and standard error values (shown in brackets in italics) in these two years are assumed for each of these years, with a 0.5 weighting being assigned in each case. Note that for Zone C, sampling stations 1-6 fall within subarea CP and stations 7-20 are within subarea CNP. Also note that FIAS data for Zones B and D for 2002 were not available and there are currently only data for Zones B and D in 2003.

Year	A	B	CNP	CP	D
<u>Mean density</u>					
1995	-	35.7	16.8	22.3	15.1
1996	30.7	41.1	17.4	14	29.8
1997	29.2	34	19	1.3	23.2
1998	28.3	31	13.9	1.8	19.2
1999	-	15.8 (25.88)	10.3	2	7.7
2000	<sup>a</sup>	36.0 (25.88)	-	-	9.6
2001	29.5	44.9	12.1	4.2	10.7
2002	30.1	-	7.7	1.8	-
2003		23.85			3.3
<u>Standard error</u>					
1995	-	13.4	4	10	3.9
1996	10.9	13	5.7	5.7	8
1997	6.8	12.4	8.9	0.5	6.5
1998	6.7	10.5	4.5	1.2	4.2
1999	-	3.3 (4.18)	2.9	1	1.4
2000	<sup>a</sup>	5.1 (4.18)	-	-	1.9
2001	6.4	17.4	2.8	1.9	2.1
2002	8.7	-	1.8	0.5	-
2003		16.7			0.94
<u>No. of stations sampled</u>					
1995	0	20	14	6	19
1996	20	19	14	6	20
1997	20	20	14	6	20
1998	18	19	14	6	20
1999	0	12	14	6	20
2000	4	10	0	0	20
2001	20	20	14	6	20
2002	20	0	14	6	0
2003		20			20

<sup>a</sup> Point excluded from analysis due to small sample number of stations sampled.

Table 3.6. Summary of Zone C catch-at-age data series included in the 2003 abalone assessment model.

Catch-at-age data series	No. of years of data used	Min. age / Minus group	Plus group age	Compared to inshore/offshore model compartment/s
1. Commercial (CNP)	19	9- (8-9)	15+	Inshore + Offshore
2. Commercial (CP)	18	9- (8-9)	15+	Offshore
3. Recreational (CNP&CP combined)	8	9- (8-9)	15+	Inshore
4. Recreational (CP only)	1	9- (8-9)	15+	Inshore
5. Poaching sector (from confiscated samples) (assumed all from CP)	7	5	12+	Inshore
6. Inshore FIAS (CNP)	7	5	12+	Inshore
7. Inshore FIAS (CP)	4	5	12+	Inshore
8. Inshore 0-5 m "old survey" (CP)	1	2	12+	Inshore
9. Offshore 5-15 m "old survey" (CP)	1	3- (2-3)	15+	Offshore
10. Industry survey (2002) (CNP&CP combined)	1	1	15+	Inshore+Offshore

Table 3.7. Catch at age matrix for commercial catches from Zones A, B, C (NONPOACHED subarea CNP and POACHED subarea CP) and D calculated for standardised model years 1981 - 2002 (hyphens indicate no data are available for the corresponding year/s). Proportions in each age class are calculated using the cohort slicing matrix from Moloney & Mackenzie (2001). The last column shows the total number of animals in the sample of lengths. Note that in the 2002 assessment models, the proportions in ages 8 and 9 were combined into a 9- category

Zone A MODEL YEAR	AGES								TOTAL
	8	9	10	11	12	13	14	15+	
1981	0.009	0.087	0.116	0.171	0.161	0.088	0.042	0.326	231
1982	0.000	0.100	0.108	0.103	0.105	0.079	0.025	0.481	550
1983	0.006	0.102	0.246	0.101	0.137	0.080	0.045	0.283	328
1984	0.012	0.072	0.160	0.139	0.165	0.093	0.031	0.327	241
1985	0.002	0.099	0.235	0.147	0.100	0.088	0.052	0.276	512
1986	0.004	0.159	0.240	0.168	0.097	0.085	0.028	0.219	454
1987	0.002	0.097	0.193	0.149	0.134	0.100	0.043	0.281	1280
1988	0.001	0.044	0.160	0.141	0.168	0.105	0.048	0.332	1248
1989	0.000	0.047	0.129	0.117	0.105	0.104	0.038	0.461	1145
1990	0.000	0.107	0.230	0.170	0.137	0.099	0.037	0.220	926
1991	0.000	0.100	0.189	0.160	0.149	0.112	0.046	0.244	3946
1992	0.001	0.093	0.160	0.145	0.152	0.108	0.049	0.292	4086
1993	0.006	0.084	0.180	0.146	0.153	0.091	0.035	0.306	7134
1994	0.002	0.064	0.128	0.137	0.144	0.098	0.043	0.384	8575
1995	0.003	0.087	0.146	0.144	0.134	0.092	0.053	0.342	8864
1996	0.001	0.054	0.142	0.117	0.150	0.102	0.051	0.383	8417
1997	0.011	0.115	0.151	0.146	0.136	0.085	0.049	0.307	7576
1998	0.007	0.132	0.166	0.123	0.133	0.086	0.039	0.314	6848
1999	0.008	0.132	0.168	0.138	0.123	0.087	0.041	0.303	7830
2000	-	-	-	-	-	-	-	-	-
2001	0.032	0.252	0.207	0.133	0.115	0.068	0.029	0.163	4917
2002	0.025	0.167	0.202	0.144	0.131	0.072	0.035	0.224	10753
Zone B									
	8	9	10	11	12	13	14	15+	total
1981	0.003	0.297	0.223	0.156	0.102	0.040	0.020	0.161	400
1982	0.000	0.086	0.205	0.119	0.099	0.106	0.047	0.337	498
1983	0.005	0.170	0.268	0.167	0.116	0.082	0.026	0.165	1352
1984	0.008	0.109	0.222	0.164	0.116	0.104	0.034	0.244	1169
1985	0.014	0.174	0.249	0.133	0.101	0.082	0.028	0.218	1776
1986	0.004	0.231	0.301	0.140	0.094	0.065	0.027	0.138	3232
1987	0.004	0.136	0.241	0.161	0.120	0.078	0.040	0.220	2699
1988	0.000	0.094	0.213	0.145	0.135	0.104	0.033	0.275	2041
1989	0.002	0.108	0.228	0.132	0.136	0.096	0.038	0.260	2472
1990	0.002	0.131	0.224	0.144	0.108	0.072	0.040	0.279	1639
1991	0.002	0.139	0.249	0.176	0.151	0.081	0.034	0.169	3677
1992	0.001	0.139	0.207	0.164	0.136	0.099	0.042	0.213	5051



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1993	0.015	0.189	0.237	0.172	0.126	0.062	0.027	0.172	5494
1994	0.011	0.197	0.202	0.142	0.116	0.080	0.032	0.219	6349
1995	0.006	0.182	0.216	0.149	0.127	0.079	0.038	0.203	6897
1996	0.006	0.168	0.226	0.153	0.135	0.075	0.038	0.200	6914
1997	0.025	0.233	0.217	0.135	0.103	0.064	0.029	0.193	3667
1998	0.012	0.188	0.217	0.149	0.133	0.078	0.033	0.190	5911
1999	0.013	0.240	0.223	0.132	0.116	0.076	0.032	0.169	6030
2000	-	-	-	-	-	-	-	-	-
2001	0.030	0.266	0.213	0.133	0.112	0.061	0.030	0.155	3679
2002	0.015	0.152	0.202	0.151	0.143	0.073	0.034	0.231	6894
Zone CNP									
	8	9	10	11	12	13	14	15+	total
1981	0.000	0.081	0.190	0.085	0.131	0.096	0.041	0.375	199
1982	0.000	0.052	0.068	0.094	0.117	0.099	0.025	0.545	100
1983	-	-	-	-	-	-	-	-	-
1984	0.037	0.207	0.279	0.147	0.120	0.058	0.016	0.136	488
1985	0.010	0.232	0.299	0.123	0.086	0.069	0.025	0.155	1091
1986	0.000	0.141	0.270	0.186	0.107	0.081	0.020	0.195	691
1987	0.000	0.141	0.264	0.212	0.169	0.095	0.046	0.073	338
1988	0.002	0.085	0.293	0.184	0.142	0.101	0.042	0.153	657
1989	0.000	0.085	0.313	0.162	0.152	0.120	0.020	0.150	151
1990	0.009	0.074	0.208	0.138	0.127	0.095	0.033	0.317	330
1991	0.001	0.102	0.198	0.176	0.147	0.107	0.047	0.222	1591
1992	0.000	0.047	0.111	0.115	0.132	0.101	0.053	0.441	836
1993	0.009	0.165	0.207	0.151	0.126	0.065	0.026	0.251	1738
1994	0.000	0.085	0.171	0.173	0.144	0.092	0.040	0.294	2218
1995	0.000	0.092	0.156	0.129	0.125	0.082	0.043	0.371	2086
1996	0.002	0.045	0.118	0.103	0.121	0.078	0.042	0.491	3496
1997	0.008	0.174	0.208	0.163	0.135	0.061	0.033	0.218	620
1998	0.003	0.152	0.194	0.146	0.136	0.099	0.039	0.232	1596
1999	0.011	0.171	0.178	0.140	0.133	0.092	0.042	0.233	1984
2000	-	-	-	-	-	-	-	-	-
2001	-	-	-	-	-	-	-	-	-
2002	0.003	0.066	0.125	0.144	0.156	0.097	0.053	0.356	3561
Zone CP									
	8	9	10	11	12	13	14	15+	total
1981	0.011	0.292	0.271	0.140	0.099	0.065	0.028	0.094	1080
1982	0.000	0.298	0.254	0.122	0.106	0.051	0.019	0.150	1000
1983	0.018	0.247	0.263	0.139	0.103	0.058	0.030	0.142	1071
1984	0.027	0.190	0.296	0.142	0.111	0.067	0.029	0.138	2147
1985	0.006	0.173	0.246	0.136	0.108	0.072	0.026	0.233	1112
1986	0.004	0.194	0.338	0.193	0.097	0.066	0.016	0.092	1479
1987	0.004	0.144	0.269	0.180	0.135	0.086	0.042	0.141	1980
1988	0.000	0.130	0.250	0.174	0.125	0.102	0.041	0.179	2030
1989	0.000	0.091	0.246	0.168	0.137	0.098	0.043	0.216	2838
1990	0.006	0.142	0.260	0.150	0.139	0.092	0.037	0.174	3641
1991	0.001	0.176	0.261	0.180	0.131	0.076	0.026	0.150	2545
1992	0.002	0.153	0.261	0.159	0.143	0.088	0.037	0.155	3647
1993	0.010	0.167	0.301	0.183	0.124	0.063	0.028	0.123	4101
1994	0.001	0.205	0.257	0.189	0.137	0.070	0.026	0.115	3890
1995	0.002	0.124	0.187	0.170	0.166	0.082	0.039	0.229	3506
1996	0.001	0.041	0.090	0.102	0.118	0.086	0.047	0.515	2094

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1997	0.009	0.149	0.158	0.119	0.099	0.059	0.029	0.379	916
1998	0.006	0.112	0.154	0.111	0.105	0.066	0.027	0.419	526
1999	-	-	-	-	-	-	-	-	-
2000	-	-	-	-	-	-	-	-	-
2001	-	-	-	-	-	-	-	-	-
2002	-	-	-	-	-	-	-	-	-
Zone D									
	8	9	10	11	12	13	14	15+	total
1981	0.000	0.044	0.066	0.079	0.087	0.072	0.025	0.626	1245
1982	0.003	0.239	0.261	0.114	0.105	0.066	0.022	0.191	400
1983	0.000	0.091	0.154	0.118	0.123	0.093	0.039	0.382	393
1984	-	-	-	-	-	-	-	-	-
1985	0.008	0.202	0.297	0.134	0.106	0.073	0.026	0.153	2976
1986	0.000	0.142	0.285	0.185	0.113	0.076	0.033	0.166	1125
1987	0.002	0.159	0.266	0.125	0.115	0.083	0.029	0.221	1045
1988	0.000	0.103	0.275	0.161	0.121	0.092	0.034	0.213	1225
1989	0.000	0.094	0.197	0.130	0.140	0.098	0.050	0.292	1143
1990	0.000	0.073	0.230	0.146	0.137	0.095	0.042	0.277	1319
1991	0.002	0.057	0.163	0.163	0.160	0.104	0.048	0.303	2149
1992	0.000	0.063	0.193	0.136	0.136	0.096	0.036	0.341	2684
1993	0.005	0.047	0.095	0.119	0.131	0.082	0.045	0.475	1821
1994	0.000	0.050	0.124	0.122	0.130	0.101	0.037	0.436	3574
1995	0.002	0.087	0.134	0.147	0.135	0.084	0.046	0.365	3497
1996	0.002	0.043	0.093	0.094	0.136	0.101	0.053	0.478	3396
1997	0.012	0.093	0.121	0.110	0.144	0.098	0.045	0.377	2413
1998	0.005	0.103	0.124	0.109	0.111	0.096	0.050	0.403	3831
1999	0.005	0.087	0.136	0.103	0.126	0.122	0.056	0.365	3907
2000	0.013	0.089	0.124	0.106	0.112	0.082	0.035	0.438	1392
2001	-	-	-	-	-	-	-	-	-
2002	0.002	0.038	0.090	0.085	0.115	0.073	0.044	0.553	2916

Table 3.8. Catch at age matrix for the RECREATIONAL data from zones A, B, C and D. The total number ( $n$ ) of animals in each sample of lengths is shown in the second column. The 2001 Zone A data are all from Pearly Beach. The only data available for 2002 and 2003 are from Zone D and these length data have been converted using the age-slicing method given in Moloney & MacKenzie (2001). Note that the 9- group includes all sampled animals with ages corresponding to 9 or younger.

Zone A	$n$	AGES						
		9-	10	11	12	13	14	15+
1993	836	0.137	0.242	0.180	0.131	0.112	0.039	0.160
1994	510	0.403	0.252	0.153	0.084	0.038	0.012	0.058
1999	926	0.169	0.211	0.193	0.146	0.087	0.035	0.159
2001	1483	0.061	0.168	0.160	0.185	0.118	0.053	0.255
Zone B								
1992	612	0.26	0.207	0.119	0.121	0.072	0.032	0.190
1993	2013	0.209	0.241	0.179	0.123	0.072	0.030	0.145
1994	3950	0.214	0.310	0.167	0.115	0.065	0.020	0.109
1995	2241	0.23	0.302	0.164	0.110	0.071	0.031	0.093
1996	2627	0.196	0.229	0.160	0.130	0.093	0.037	0.155
1997	3654	0.251	0.277	0.162	0.123	0.062	0.025	0.099
1998	2974	0.322	0.319	0.155	0.089	0.043	0.018	0.054
1999	3799	0.237	0.254	0.163	0.145	0.068	0.022	0.111
2001	5585	0.158	0.236	0.216	0.164	0.093	0.036	0.096
Zone C								
1992	1031	0.226	0.231	0.147	0.115	0.086	0.030	0.164
1993	1407	0.230	0.274	0.150	0.109	0.078	0.030	0.129
1994	3096	0.140	0.268	0.189	0.162	0.086	0.026	0.130
1995	4073	0.179	0.290	0.181	0.159	0.073	0.025	0.093
1996	2840	0.198	0.264	0.158	0.127	0.068	0.026	0.159
1997	1707	0.294	0.292	0.143	0.094	0.055	0.021	0.102
1998	902	0.143	0.272	0.170	0.126	0.100	0.030	0.158
1999	616	0.185	0.257	0.170	0.124	0.107	0.033	0.125
2000	-	-	-	-	-	-	-	-
Mudge Pt (subarea CP)								
1997	1769	0.215	0.139	0.090	0.099	0.065	0.036	0.356
Zone D								
1993	1743	0.225	0.225	0.145	0.117	0.067	0.033	0.187
1994	5128	0.184	0.220	0.136	0.113	0.070	0.033	0.243
1995	5973	0.155	0.228	0.170	0.139	0.077	0.039	0.193
1996	4142	0.131	0.248	0.171	0.138	0.095	0.033	0.185
1997	7844	0.199	0.245	0.158	0.113	0.079	0.027	0.179
1998	1492	0.198	0.270	0.159	0.108	0.083	0.030	0.152
1999	3403	0.228	0.247	0.150	0.119	0.074	0.030	0.152
2001	4491	0.219	0.210	0.134	0.114	0.079	0.034	0.211
2002	4457	0.248	0.212	0.133	0.108	0.082	0.030	0.188
2003	3821	0.098	0.130	0.115	0.121	0.092	0.051	0.392

Table 3.9. Catch at age matrix from INSHORE FIAS data for the “poached” and “nonpoached” areas of Zone C for each of the standardised model years indicated. Data are averaged over a total of six equidistant transects (30 x 2 m) for the poached subarea and 14 transects for the nonpoached subarea. The total number ( $n$ ) of animals sampled per year in each of the subareas is given in the second column of the table. Proportions in each age class are calculated using the cohort slicing matrix from Moloney & Mackenzie (2001). The proportions in age classes 12 to 15+ have been lumped together to reduce the number of categories containing a proportional abundance less than 2% in any one year. Note that data from the poached subarea for 1997, 1998 and 2002 have been excluded because  $n < 10$  in these years.

POACHED SUBAREA – CP		Proportion in age class							
	$n$	5	6	7	8	9	10	11	12+
1995	120	0.053	0.257	0.237	0.162	0.107	0.079	0.033	0.073
1996	61	0.072	0.266	0.280	0.169	0.052	0.046	0.049	0.066
1999	11	0.000	0.182	0.182	0.000	0.182	0.091	0.091	0.273
2000	-	-	-	-	-	-	-	-	-
2001	19	0.105	0.116	0.095	0.368	0.105	0.158	0.000	0.053
NONPOACHED SUBAREA – CNP									
1995	234	0.118	0.195	0.193	0.168	0.138	0.049	0.025	0.115
1996	184	0.100	0.245	0.230	0.157	0.135	0.048	0.006	0.079
1997	156	0.122	0.310	0.214	0.227	0.051	0.026	0.030	0.019
1998	160	0.082	0.191	0.246	0.220	0.143	0.048	0.018	0.054
1999	120	0.090	0.257	0.243	0.153	0.118	0.052	0.032	0.055
2000	-	-	-	-	-	-	-	-	-
2001	131	0.023	0.095	0.124	0.141	0.165	0.156	0.070	0.226
2002	90	0.022	0.080	0.119	0.137	0.084	0.153	0.127	0.278

Table 3.10. Catch at age matrix for the INSHORE FIAS data from zones A, B and D (data for 1995-2000 period are from Moloney & Mackenzie 2001). Note that the Zone A data for 2000 are based on the sampling of stations 17-20 only. The Zone B data for 1999 and 2000 are based on only 8 and 10 stations respectively. Data to be input to the stock assessment model in 2003 were only available for Zones B and D.

<u>Zone A</u>	<i>n</i>	AGES							
		5	6	7	8	9	10	11	12+
1995	-	-	-	-	-	-	-	-	-
1996	446	0.131	0.167	0.185	0.138	0.100	0.065	0.062	0.151
1997	502	0.134	0.196	0.195	0.156	0.120	0.056	0.036	0.108
1998	499	0.182	0.216	0.150	0.167	0.093	0.066	0.029	0.098
1999	-	-	-	-	-	-	-	-	-
2000	183	0.267	0.347	0.189	0.103	0.064	0.008	0.008	0.014
2001	444	0.146	0.259	0.156	0.130	0.106	0.055	0.045	0.102
2002	444	0.198	0.240	0.189	0.137	0.070	0.039	0.044	0.083
2003	-	-	-	-	-	-	-	-	-
<u>Zone B</u>									
1995	480	0.131	0.179	0.196	0.152	0.135	0.066	0.045	0.096
1996	479	0.181	0.168	0.220	0.172	0.111	0.050	0.036	0.062
1997	462	0.114	0.194	0.176	0.186	0.132	0.079	0.051	0.069
1998	430	0.130	0.198	0.160	0.145	0.152	0.069	0.049	0.096
1999	178	0.090	0.091	0.188	0.183	0.185	0.079	0.067	0.117
2000	341	0.247	0.227	0.174	0.146	0.075	0.055	0.041	0.035
2001	490	0.142	0.228	0.205	0.146	0.118	0.059	0.029	0.073
2002	-	-	-	-	-	-	-	-	-
2003	208	0.104	0.230	0.174	0.210	0.150	0.072	0.020	0.039
<u>Zone D</u>									
1995	287	0.044	0.083	0.101	0.186	0.146	0.110	0.088	0.241
1996	392	0.026	0.142	0.146	0.209	0.193	0.121	0.073	0.090
1997	265	0.048	0.090	0.148	0.227	0.200	0.149	0.065	0.073
1998	301	0.064	0.117	0.147	0.159	0.141	0.132	0.062	0.178
1999	129	0.013	0.122	0.166	0.242	0.106	0.159	0.057	0.134
2000	171	0.109	0.184	0.156	0.148	0.143	0.088	0.034	0.139
2001	164	0.027	0.115	0.133	0.154	0.115	0.156	0.090	0.211
2002	-	-	-	-	-	-	-	-	-
2003	54	0.050	0.154	0.072	0.187	0.115	0.033	0.115	0.274

Table 3.11. Catch at age matrix for the POACHED data from zones A, B, C and D for the period 1995-2002. No data for 2003 were available at the time the assessment was done. The total number ( $n$ ) of animals in each sample of lengths is shown in the second column. The size data for Zone A (1996 and 2001), Zone B (1995) and Zone D (2001) (shown in italics) were based on relatively small samples of confiscated abalone ( $n = 200$ ) and these data were thus excluded from the model following an AWG decision. The base-case model assumes that the Zone C data are all from the poached subarea CP. Data for the period 1995 - 2000 are from Moloney & Mackenzie (2001) and are shown in terms of proportional abundance per age class per model year.

Zone A	$n$	AGES							
		5	6	7	8	9	10	11	12+
<i>1996</i>	<i>174</i>	<i>0.116</i>	<i>0.391</i>	<i>0.328</i>	<i>0.087</i>	<i>0.039</i>	<i>0.018</i>	<i>0.010</i>	<i>0.011</i>
1997	7409	0.133	0.089	0.136	0.137	0.136	0.083	0.058	0.228
1998	5035	0.048	0.096	0.144	0.146	0.148	0.096	0.066	0.257
1999	6706	0.047	0.097	0.143	0.125	0.118	0.091	0.063	0.316
2000	1793	0.173	0.210	0.225	0.151	0.111	0.057	0.023	0.050
<i>2001</i>	<i>177</i>	<i>0.000</i>	<i>0.000</i>	<i>0.017</i>	<i>0.024</i>	<i>0.072</i>	<i>0.113</i>	<i>0.111</i>	<i>0.663</i>
2002	587	0.212	0.273	0.230	0.133	0.054	0.035	0.030	0.033
<b>Zone B</b>									
<i>1995</i>	<i>112</i>	<i>0.000</i>	<i>0.000</i>	<i>0.025</i>	<i>0.105</i>	<i>0.339</i>	<i>0.213</i>	<i>0.131</i>	<i>0.186</i>
1996	229	0.089	0.195	0.319	0.088	0.008	0.000	0.010	0.291
1997	928	0.034	0.072	0.080	0.151	0.212	0.158	0.102	0.192
1998	632	0.066	0.063	0.131	0.215	0.201	0.152	0.070	0.102
1999	3154	0.104	0.130	0.149	0.125	0.116	0.089	0.071	0.217
2000	1532	0.052	0.110	0.161	0.173	0.161	0.098	0.060	0.185
2001	1846	0.114	0.141	0.152	0.131	0.098	0.095	0.070	0.198
2002	2561	0.109	0.134	0.163	0.164	0.106	0.115	0.057	0.150
<b>Zone CP</b>									
1995	550	0.007	0.033	0.109	0.174	0.190	0.176	0.108	0.203
1996	871	0.060	0.190	0.291	0.206	0.125	0.059	0.019	0.051
1997	2327	0.087	0.103	0.131	0.120	0.109	0.090	0.058	0.301
1998	924	0.069	0.123	0.199	0.197	0.162	0.096	0.053	0.101
1999	1525	0.007	0.028	0.057	0.077	0.107	0.109	0.069	0.547
2000	3500	0.042	0.079	0.123	0.112	0.112	0.077	0.061	0.394
2001	492	0.068	0.177	0.252	0.238	0.105	0.073	0.024	0.063
2002	641	0.102	0.155	0.130	0.098	0.076	0.101	0.053	0.285
<b>Zone D</b>									
1995	397	0.000	0.004	0.046	0.083	0.119	0.072	0.059	0.618
1996	458	0.008	0.021	0.064	0.081	0.083	0.107	0.112	0.524
1997	676	0.238	0.184	0.114	0.057	0.046	0.063	0.045	0.253
1998	672	0.019	0.060	0.085	0.118	0.089	0.078	0.049	0.501
1999	1961	0.003	0.026	0.057	0.078	0.111	0.093	0.082	0.549
2000	474	0.011	0.028	0.085	0.110	0.089	0.081	0.070	0.527
<i>2001</i>	<i>200</i>	<i>0.000</i>	<i>0.005</i>	<i>0.005</i>	<i>0.020</i>	<i>0.012</i>	<i>0.123</i>	<i>0.125</i>	<i>0.710</i>
2002	643	0.030	0.022	0.031	0.049	0.075	0.119	0.106	0.569

Chapter 3 – Collating available data

Table 3.14. Catch at age matrix from OFFSHORE 5-15m depth range OLD SURVEY data for Zones and years as indicated. Data (from Tarr (2002)) are from the following locations in the various Zones: BU - Buffeljags; PB - Pearly Beach; DP - Danger Point; MP - Mudge Point; HK - Hangklip. Note that the data for Zones A and C are from 200m transects conducted in the 5-10m depth range. For Zones B and D, additional data were available from 200m transects conducted in the 10-15m depth range. For these Zones, the 5-10m depth range and 10-15m depth range data have been combined, yielding combined 5-15m depth range size compositions. In the survey, abalone were counted 1.5m on either side of a 200m transect. A separate sample was collected for length frequency investigations. The data in Tarr (2002) are in the form of numbers per size class (in mm shell breadth) which were calculated from the percentage of abalone per size class multiplied by the total number counted on the transect. The total number (*n*) of abalone counted on each 5-10m transect (Zones A and C) or along both the 5-10m and 10-15m transects (Zones B and D) is given in the last column of the Table. The proportions in each age class as given below were calculated using an extended form (because of the younger age classes included in the sample) of the cohort slicing matrix from Moloney & Mackenzie (2001) (reproduced in Appendix 3.2). Note that the ages 2 and 3 categories have been lumped together to reduce the number of categories containing a proportional abundance less than 2% in any one year. The 5-15m depth range data are assumed to apply to the offshore area in the model.

	Ages (yrs)													<i>n</i>
	2-3	4	5	6	7	8	9	10	11	12	13	14	15+	
Zone A BU (86)	0.035	0.063	0.092	0.040	0.067	0.078	0.098	0.071	0.058	0.053	0.047	0.037	0.263	19
Zone A PB (87)	0.034	0.049	0.046	0.044	0.043	0.073	0.081	0.070	0.074	0.068	0.060	0.049	0.310	27
Zone B DP (82)	0.081	0.081	0.084	0.088	0.103	0.129	0.084	0.072	0.052	0.037	0.027	0.022	0.140	91
Zone CP MP (81)	0.033	0.031	0.040	0.083	0.074	0.124	0.173	0.136	0.081	0.050	0.031	0.029	0.115	27
Zone CP MP (85)	0.042	0.050	0.055	0.062	0.120	0.172	0.157	0.082	0.077	0.052	0.033	0.021	0.076	70
Zone D HK (83)	0.015	0.029	0.058	0.051	0.086	0.086	0.088	0.077	0.053	0.054	0.052	0.031	0.322	70

Table 3.15. Catch-at-age data for Zones B and C from a full population survey (Industry and MCM) conducted in March 2002.

	Age class														
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
Zone C	0.01	0.00	0.00	0.01	0.02	0.05	0.07	0.12	0.10	0.10	0.06	0.11	0.09	0.04	0.23
Zone B	0.37	0.11	0.09	0.11	0.05	0.05	0.06	0.05	0.04	0.02	0.02	0.01	0.00	0.00	0.01

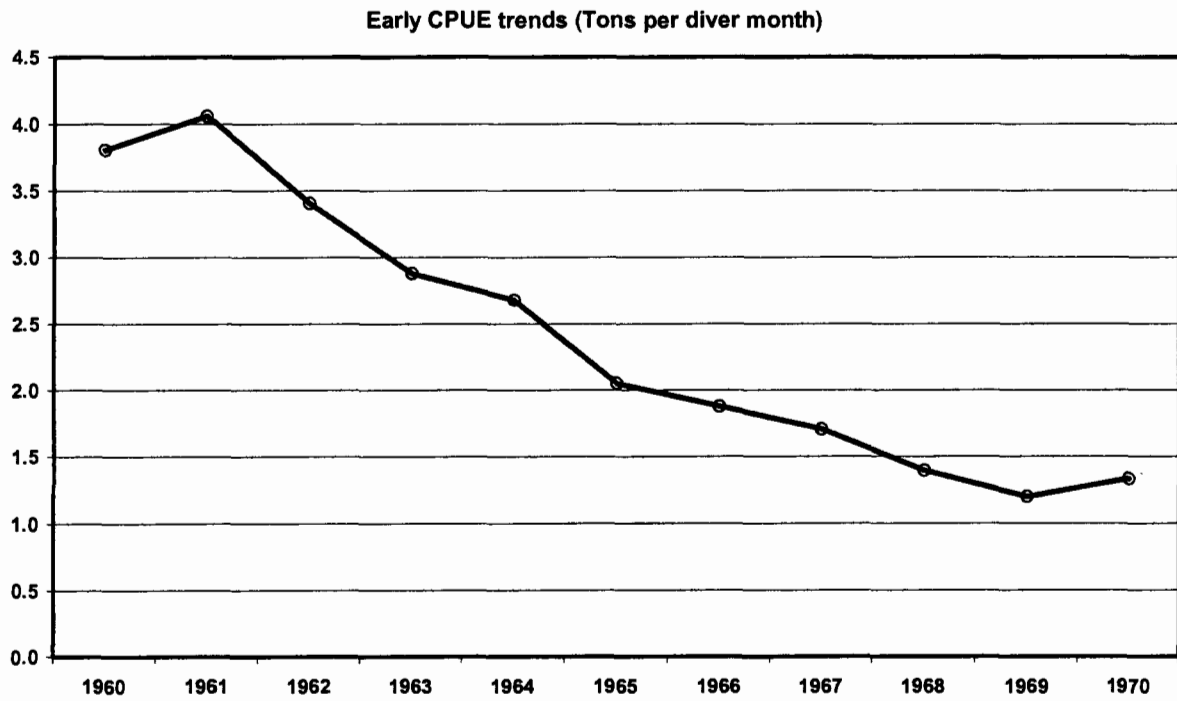


Fig. 3.1. Historic CPUE index calculated as whole mass per licensed diver month (from Newman 1973) for Zones A-D combined.



Poaching data for 2003 are only available for 8 months (up until the end of May) of Model year 2003 and hence (as with last year's assessment) have been linearly extrapolated to make them comparable to the estimates for the previous years. A summary as to how the data in Tables 4.2 and 4.3 were obtained is given hereunder:

1. The raw data show the number of abalone confiscated per Zone (in cases where the Zone is known; otherwise classified in Unknown category) and per month of the year. For each Zone as well as the Unknown category, the preliminary number of confiscations is computed as the number of computations from 1 October 2002 to 31 May 2003.
2. The values above are then linearly extrapolated by multiplying by a factor of  $12/8^*$ .
3. The Zone D estimate is halved (cf. Betty's Bay). The Unknown category confiscations is similarly reduced by an amount corresponding to half the Zone D proportion as computed in (4) below (to account for Betty's Bay).
4. The relative magnitudes of poaching in each of the four Zones is computed as the ratio of the confiscations in a Zone relative to the total Zones A-D confiscations.
5. The total confiscations in the Unknown category are assumed to derive from Zones A-D in the same proportions as described in (4) above. The total number of confiscations per Zone per Model year is then determined as the sum of the Zonal estimate and the "Unknown category" estimate.

\* Note that the scaling factor  $4/3$  is supported further by the fact that it is similar to an alternative scaling factor considered which is based on the average proportion of confiscations (0.20) in the last quarter (July-Sept) of a Model year relative to the total confiscations in the first three-quarters of a Model year (seasons 4, 1 and 2) during each of the last three years.

It was noteworthy that a retrospective analysis showed that for Zone A the scaled up poaching confiscation estimate for Model year 2002 was only about 60% the actual number of confiscations achieved in Model year 2002.

Table 4.2 shows the total number of abalone confiscated (after adjusting for the Zone D effect and scaling the 2003 values linearly upwards) per Zone per Model year.

Table 4.3 shows the TOTAL number of abalone confiscations per Zone if the Zone “Unknown” confiscation component is assumed allocated to the various Zones in the same proportion as the relative number of confiscations per Zone. In the 2002 and 2003 abalone assessments, the total number of abalone assumed poached per Zone in model computations was taken as either double or four times the confiscation estimates (in a form as given in Table 4.3), depending on whether a 50% or 25% confiscation proportion was assumed. The values shown in Table 4.3 are also used as a diagnostic check in scenarios in which the poaching level is estimated within the model: they represent the minimum realistic poaching estimates (i.e. the actual amount poached must be greater than the corresponding confiscation estimates).

#### 4.2 Poaching trend assumptions

The poaching confiscation data are used to obtain base-case estimates of the trend in poaching over time in each of Zones A-D. A better measure of changes in the level of poaching in a Zone is not confiscation *per se*, but confiscations per unit of policing activity. Because of the absence of clear quantitative data on policing intensity, three alternative options have been proposed for consideration in the model implementation process to investigate the sensitivity of model results to this assumption. Scenario I assumes that the level of policing has remained constant "near" to shore, so that location-known confiscations are a consistent index of the amount of poaching in the Zone in question. The Scenario I poaching trend is thus based only on the location-known zonal confiscation data as given in Table 4.2. As in previous years, a linear increase in poaching (from zero in 1990 increasing up to the 1994 level) is assumed for Zone C and for Zone D (from zero in 1991 increasing up to the 1994 level) because confiscation data are available only from 1994 onwards. Poaching is thought to have started earlier in these Zones than in Zones A and B.

Policing efficiency levels corresponding to Scenarios II and III are shown in Table 4.4. The Scenario II policing levels (MCM Scenario) represent the "best guess" of the increase in policing efficiency based on knowledge of police operations (as advised by Marcel Kroese, MCM) while the Scenario III trend (OLRAC Scenario advised in 2002) was proposed to bound the policing effect. Note that, for example, a policing level factor of 2 implies a 100% increase in policing efficiency so that the corresponding confiscation amounts are multiplied by a factor of 0.5 to make them comparable to the other values. The revised poaching trends

used as inputs (in terms of relative numbers poached) into the 2003 model runs are summarised in Table 4.4.

### **4.3. Comparing poaching levels between Zones**

The relative amounts of poaching when compared across the four Zones A-D is simply provided by the ratio of the maximum poaching level of a Zone and the maximum for Zone C. This is used in model scenarios in which the base-case maximum poaching level is estimated in the model for Zone C and then scaled for other Zones based on that estimate, rather than perhaps estimating within the model for those Zones as well. Table 4.4 also gives base-case scaling factors to be used (if and where necessary) to obtain poaching estimates for Zones A, B and D relative to the Zone C base-case maximum poaching estimate for 1997 (Scenario I) or 1995 (Scenarios II and III).

Table 4.1. Summary of method to summarise poaching confiscation data.

	Zone A	Zone B	Zone C	Zone D	Unknown	Total	Total from A-D only
Number confiscated from 1 Oct 2002 to 31 May 2003	26647	15694	2819	6949	466519	518628	
Extrapolated total to Sept. 2003	39971	23541	4229	10424	699779	777942	
Halving estimate for Zone D	39971	23541	4229	5212	699779	772730	72952
Prop. Poaching per Zone	0.55	0.32	0.06	0.07			
TOTAL including Unknown component	423381	249354	44790	55205			

Table 4.2. Summary of the total number of abalone confiscated per Zones A-D and rest of South Africa per standard Model year  $y$  that is taken to run from October of year  $y-1$  to September of year  $y$ . The Zone D data from the beginning of Model year 2001 and onwards have been halved to account for the recent increase in poaching confiscations from Betty's Bay, a marine reserve area located within Zone D. Data for 2003 were available (at the time of analysis) only until May 2003 and have been (linearly) extrapolated to make them comparable to the estimates for the previous years (see text). Actual recorded values are differentiated from adjusted/scaled up values using italics in the Table below.

Model Year	Zone A	Zone B	Zone C	Zone D	Unknown	Total
1994	0	419	10004	1081	0	11504
1995	0	2637	15284	2668	55551	76140
1996	0 <sup>a</sup>	1512	12663	1563	55607	71345
1997	6101	4520	16027	3086	32383	62117
1998	25289	7671	10797	3521	97145	144423
1999	17841	3663	6849	2429	49406	80188
2000	8970	17929	12445	5142	196836	241322
2001	9289	39308	4770	<i>4311</i>	226397	284075
2002	45560	79966	9682	<i>7053</i>	662006	804267
2003	<i>39971</i>	<i>23541</i>	<i>4229</i>	<i>5212</i>	<i>719091</i>	<i>792044</i>

<sup>a</sup> Single sample of 174 confiscated abalone excluded from analysis due to small sample size.

Table 4.3. Total confiscation estimates per Zone after adding contribution from “Unknown” category.

Model Year	Zone A	Zone B	Zone C	Zone D
1994	0	419	10004	1081
1995	0	9752	56522	9867
1996	0	6854	57405	7086
1997	12746	9443	33482	6447
1998	77252	23433	32982	10756
1999	46476	9542	17842	6328
2000	48659	97259	67510	27894
2001	45750	193599	23493	13088
2002	257572	452084	54737	23870
2003	433962	255583	45914	31817

Chapter 4 – Poaching catch per unit of policing activity

Table 4.4. Summary of revised 2003 reference poaching trend scenarios for each of the main abalone fishery Zones A-D. Note that the proportions in each column represent the poaching intensity in that Zone relative to the maximum poaching level observed for that Zone. Scenario I refers to the pattern based purely on the zonal poaching confiscation data given in Table 3. As was the case in 2001, a linear increase in poaching is assumed to have occurred from zero in 1990 in Zone C to the 1994 level, and from zero in 1991 to the 1994 level in Zone D. The "policing efficiency levels" shown in the last two columns were proposed by the Abalone Working Group and have been used to derive modified time series representing confiscations-per-unit-policing. Scenario II assumes a smaller increase in policing efficiency than Scenario III. In conjunction with this, Scenarios II and III assume that for all Zones A-D, all years from 1980 to 1993 are set to 10% of the 1997 value; and for Zone A, years from 1994 to 1996 are similarly set to 10% of the 1997 Zone A value. Other model sensitivities will most likely also be tested. Note that for Zone C, the same pattern of poaching is assumed to apply to subareas CNP and CP, but the base-case assumes that the relative proportions of the poached amount taken are 10% from CNP and 90% from CP. The last row of the Table gives the maximum poaching level of a Zone (based on the confiscation level) as a ratio of the maximum for Zone C. For Scenario I this is based purely on the zonal confiscations, whereas the Scenario II and III values have been adjusted to allow for the assumed stepwise increase in policing effectiveness. These computations assume that the 2003 policing efficiency level is the same as the 2002 level.

	Zone C			Zone A			Zone B			Zone D			Policing efficiency level (MCM Option)	Policing efficiency level (OLRAC 2002 Option)
	I	II	III	I	II	III	I	II	III	I	II	III		
Pre-1980	0	0	0	0	0	0	0	0	0	0	0	0		
1980-1989	0	0.095	0.070	0	0.015	0.018	0	0.006	0.008	0	0.050	0.058		
1990	0	0.095	0.070	0	0.015	0.018	0	0.006	0.008	0	0.050	0.058		
1991	0.156	0.095	0.070	0	0.015	0.018	0	0.006	0.008	0	0.050	0.058		
1992	0.312	0.095	0.070	0	0.015	0.018	0	0.006	0.008	0.054	0.050	0.058		
1993	0.468	0.095	0.070	0	0.015	0.018	0	0.006	0.008	0.107	0.050	0.058		
1994	0.624	0.655	0.655	0	0.015	0.018	0.005	0.007	0.010	0.153	0.192	0.307	1	1
1995	0.954	1	1	0	0.015	0.018	0.033	0.041	0.066	0.378	0.473	0.757	1	1
1996	0.790	0.829	0.829	0	0.015	0.018	0.019	0.024	0.038	0.222	0.277	0.443	1	1
1997	1	0.953	0.699	0.134	0.152	0.179	0.057	0.064	0.075	0.438	0.497	0.583	1.1	1.5
1998	0.674	0.642	0.471	0.555	0.631	0.740	0.096	0.109	0.128	0.499	0.567	0.666	1.1	1.5
1999	0.427	0.358	0.224	0.392	0.392	0.392	0.046	0.046	0.046	0.344	0.344	0.344	1.25	2
2000	0.777	0.651	0.407	0.197	0.197	0.197	0.224	0.224	0.224	0.729	0.729	0.729	1.25	2
2001	0.298	0.250	0.156	0.204	0.204	0.204	0.492	0.492	0.492	0.611	0.611	0.611	1.25	2
2002	0.604	0.507	0.317	1	1	1	1	1	1	1	1	1	1.25	2
2003	0.264	0.221	0.138	0.877	0.877	0.877	0.294	0.294	0.294	0.739	0.739	0.739	1.25	2
Inter-Zone ratio	1	1	1	2.84	2.38	1.49	4.99	4.19	2.62	0.44	0.37	0.23		

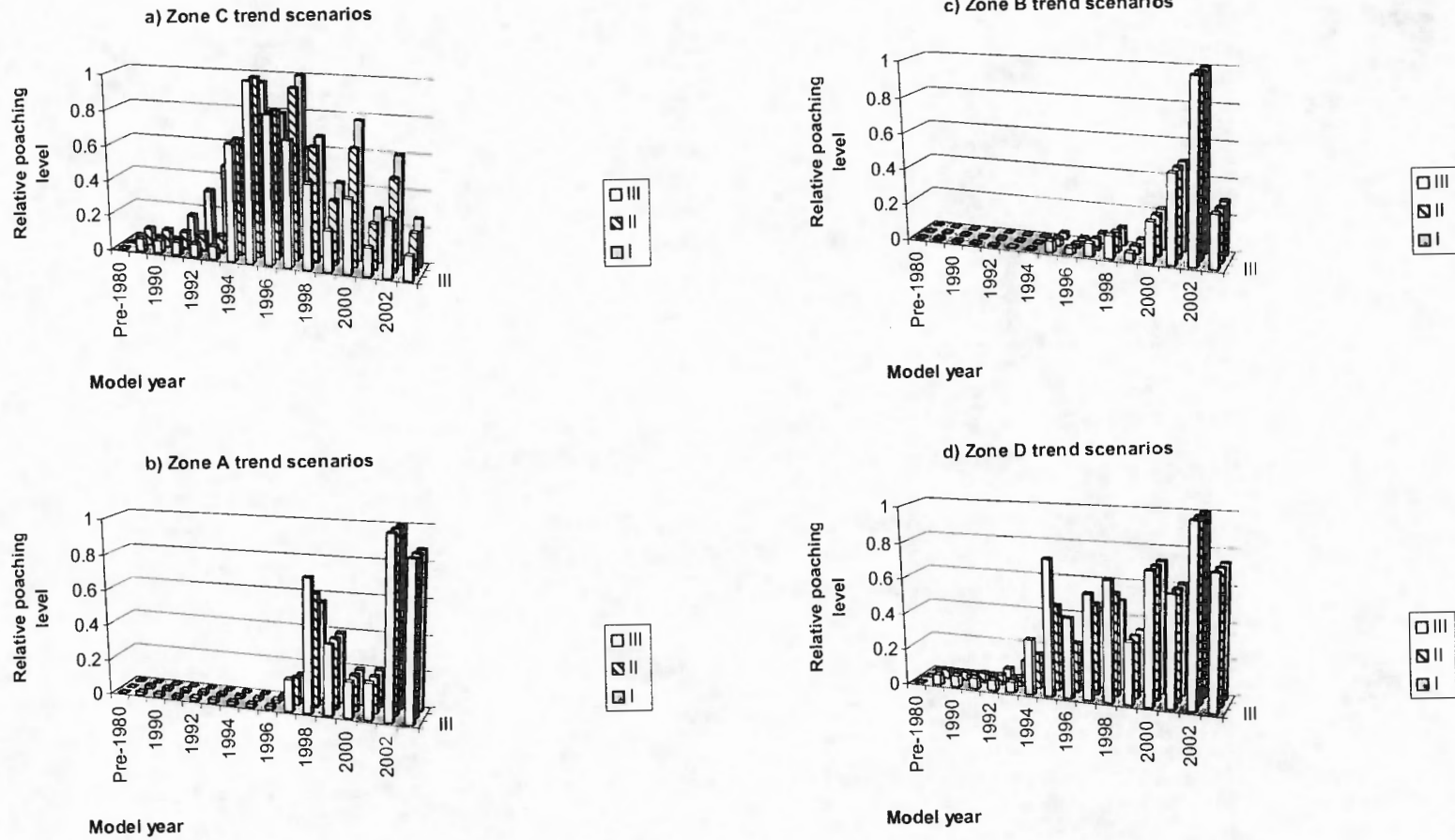


Fig 4.1 a-d. Poaching confiscation trend scenarios I (no adjustment for changes in policing efficiency), II and III for a) Zone C, b) Zone A, c) Zone B and d) Zone D. Note that for all Zones, the 2003 data point has been scaled upwards because the raw data were available only for the first 2/3 of the Model year.

## Chapter 5

### Summary of the General Linear Model used to standardise abalone Catch-Per-Unit-Effort data in Zones A-D

#### SUMMARY

A General Linear Model (GLM) is used to standardise the commercial abalone CPUE data available for Zones A, B, D and subareas CNP (“nonpoached”) and CP (“poached”) of Zone C. The model includes four explanatory variables (year, season, fisher, zone) and two interaction terms (year×season, year×Zone). Following the removal of outliers (1% of total), a total of 36 122 data points remained for the GLM fit. An iterative, inverse-weighting procedure is used to account for heteroscedasticity in the residuals. Approximately 40% of the total variation is explained by the model. Suggested future improvements include standardising on a more representative fisher and downweighting data in cases where the number of contributory data points is small, such as in Zone C for Model years 1999 and 2000 when the TAC was considerably reduced. Further thought needs to be given to complications that have arisen because of differences between the efficiency and variability associated with data from the old entitlement holders (1-84) compared to the newer entrants into the fishery (post 1998) coded as divers 85-241.

#### 5.1 INTRODUCTION

Catch-per-unit-of-effort (CPUE) data are central to the assessment of the abalone resource and indeed to most other fisheries stock assessments. It is typically assumed that CPUE is proportional to abundance although in some instances it has been argued that the assumption of linear proportionality does not necessarily hold (Harley *et al.* 2001). In particular, situations, termed hyperstability (Hilborn and Walters 1992), in which CPUE remains high while abundance declines can lead to overestimation of biomass, stressing the importance of considering the possibility of nonlinearity in the relationship between CPUE and abundance (Harley *et al.* 2001) (see Chapter 7).

Given the primary role of CPUE data in stock assessment modelling, it is important to remove or reduce the influence of factors such as improvements in fishing technologies and spatial shifts in fishing patterns which could bias the CPUE as an indicator of trends in



abundance (Glazer 1999). This process of standardisation is most commonly achieved with the use of Generalized Linear Models (GLM) (Nelder and Wedderburn 1972).

This chapter describes the application of a General Linear Model (GLM) to standardise the commercial abalone CPUE time series of abalone for the influence of other factors on the CPUE apart from resource abundance. The raw data were supplied by A. Mackenzie (MCM) and the GLM was implemented using the General Linear Models procedure of the statistical package SPSS v.10.1.0. The application of a GLM is not a central theme of this thesis and hence will only be dealt with to the extent needed as an adequate input to the modelling exercises described in the following two chapters. It should also be noted that whilst in some instances there was clearly room for improvement in the GLM, the choice of a suitable GLM was decided in part by the AWG.

The approach described here is that used in the 2003 stock assessment modelling exercises and hence involved standardising the commercial abalone CPUE time series over the period 1980 – 2002. The primary mandate from the AWG was to repeat and update the GLM analyses conducted in earlier years (Moloney 1999, Moloney and MacKenzie 2000, Plagányi *et al.* 2001), and if there was sufficient time, to pursue the following recommendations stemming from the 2002 Benefit Stock Assessment workshop (BENEFIT 2002):

A.18 (L, 3.2). Further efforts to standardize the commercial catch and effort data should:

- examine the implications of different error-models (e.g. negative binomial); and
- base the index on the results for a single season, rather than an average over seasons, if sample sizes for some seasons are low relative to those for others.

A.19 (L, 3.2). Consideration should be given to including the nominal CPUE directly into the likelihood function.

The model has recently been rerun using the statistical package *R* v. 1.8.1. (*R* Development Core Team 2003) and future work has commenced in addressing the above recommendations. However, in 2003, there was sufficient time to investigate only one of these, namely to base the CPUE index on a single season only.

The explanatory variables included in the GLM were those selected by Moloney (1999) and Moloney and Mackenzie (2001). The same is true for the interaction terms in the model. This chapter essentially describes the refitting of the GLM to the abalone data to update earlier analyses by including data available for the 2002 Model year (October 2001 – September 2002). Earlier methods used were thus replicated as far as possible. However, whereas Moloney and Mackenzie (2001) included only divers/entitlement holders 1-80 in their analysis, the current GLM has been updated to include the data from not only entitlement holders 1-84 (see following section for clarification re codes 81-84), but also new divers, assigned individual codes from 1999 onwards. The "entitlement holder" variable  $h$  used previously by Moloney and Mackenzie (2000) has thus here been replaced with a variable called FISHER CODE. For the period 1980-1998, FISHER CODE = entitlement holder no. (1-84). From 1999 onwards, FISHER CODE = diver number, where this number is in the range 1-241 (with numbers 1-84 reserved as codes for "old" entitlement holder divers). The main difference between the entitlement holder codes and new diver codes is that (new) "diver" now represents one person only, whereas "entitlement holder" often includes data from several divers.

## 5.2 DATA INCLUDED IN THE MODEL

**DATA:** Catch data (as kg whole mass), and effort data (as total duration of dives in minutes for each day dived) are available for the period 1980 to 2002. Earlier data for the period 1977 to 1979 have yet to be captured electronically. Additional information in the database pertains to the dates, the divers, and the areas and zones that were dived.

**MODEL YEAR:** A standard Model year  $y$  that is taken to run from October of year  $y-1$  to September of year  $y$ . The CPUE data are analysed for Model years 1980 to 2002.

**ZONES/SUBAREAS:** All data for Zones A, B, C and D are included in this analysis. The "nonpoached" and "poached" subareas correspond respectively to area codes 7 and 8 as recorded in the abalone database, except that area names "Onrus" and "Vermont" fall within the "nonpoached" subarea. Specifically, subarea CP includes area codes HWST (077), MLKM (121), MDGP (129), NWB (134) and PLNH (150), and subarea CNP includes area codes HRMN (078), MLKN (122), ONRR (139), SNDH (172), SKLH (188), SWRM (204), ONR (138) and VRMN (214).

**SEASONS:** Because of a strong seasonal effect on the distribution of effort through the year (Moloney and Mackenzie 2000), dates in the database were used to group the 12 months into four seasons, *viz.* 1) January-March, 2) April-June, 3) July-September, 4) October-December.

**FISHER CODE:** Includes both the entitlement holders coded in the database with numbers 1-84 as well as "divers" (after 1998) coded with numbers 85 to 241. Following Moloney (1999), the three old entitlement holders (5, 8 and 37) who passed on their rights to family members have been allocated numbers 82, 83 and 84 respectively. Following Moloney and Mackenzie (2000), the following entitlement holders were excluded from the current analysis: 3, 4, 12, 13, 28, 33, 46, 54, 61, 69, 70, 76 and 81. Note also that 42 new divers operating in 2002 have yet to be assigned diver codes and hence were simply lumped together under diver code 333 in the current analysis.

Following the removal of outliers (1% of total) based upon observations with large residuals (> 6 standard deviations) in an initial GLM fit, a total of 36 860 data points remained for the analysis. Summaries of the data included in the model are given in Tables 5.1 – 5.3, and incorporate a breakdown of the numbers of data entries used in the GLM analysis (Table 5.1), as well as the associated catch (Table 5.2) and effort (Table 5.3) statistics.

### 5.3 THE GLM MODEL

The GLM model applied is given by:

$$\ln(CPUE) = \mu + \alpha_{year} + \beta_{season} + \varphi_{fisher} + \gamma_{zone} + \eta_{year \times season} + \delta_{year \times zone} + \varepsilon \quad (5.1)$$

where:

*CPUE* is in kg/min,

$\mu$  is the intercept,

*year* is a categorical variable associated with the year (i.e. abundance) effect (1980–2002),

*season* is a categorical variable associated with the season effect (1 = Jan-Mar; 2 = Apr-Jun; 3 = Jul-Sep; 4 = Oct-Dec),

*fisher* is a categorical variable associated with the fisher code (1–241, 333),

*zone* is a categorical variable associated with the different zones/subareas (A, B, CP, CNP and D),  
*year × season* is the interaction between years and seasons,  
*year × zone* is the interaction between years and zones/subareas, and  
 $\varepsilon$  is the error term, that is assumed to follow a normal distribution.

The above assumes that the CPUE data are log-normally distributed. This was chosen both in the interests of keeping analyses simple and because it is unlikely that any single error model is ideal anyway given that there are a multitude of processes contributing to the overall error structure (Butterworth 1996). However, depending on the dataset being analysed, other error structure models may be more appropriate, such as gamma; Poisson, negative binomial and delta-lognormal models (e.g. Lo *et al.* 1992, Cooke and Lankester 1995, Pennington 1996).

#### **5.4 MODEL 2 – ONE SEASON ONLY**

The method above was repeated with one exception – only the data for a single season was used in the analysis.

#### **5.5 RESULTS OF THE GLM MODEL IMPLEMENTATION – ALL SEASONS**

Following an initial GLM fit, the residuals were examined to check, in particular, for evidence of heteroscedasticity. Fig. 5.1 shows evidence of larger residuals associated with lower effort. To account for this heteroscedasticity, the model was refit to the data using the iterative, inverse-weighting procedure applied in the year 2002 and in previous years, in which reduced weight is given to the data points with the largest variance in the model. To replicate this method, weighting factors were calculated by grouping the data into 12 effort categories, and estimating the mean variance of the residuals (the  $\varepsilon$ 's) in each category, such that category 1 represents effort of <50 minutes dive duration, category 2 is 50-100 minutes and so on. The weighting factors are simply the inverse of the variance for each category. The residuals were iteratively re-weighted until the model converged. Fig. 5.1 shows the final residual plot per effort category used to investigate the assumption of equal variance. Cook's distance measure was used to identify points that are potentially influential due to their

location in  $x$ -space (Montgomery and Peck 1992). Only three such points (with  $D_i > 1$ ) were identified after the second model iteration.

As an alternative to the method described above to account for heteroscedasticity, Plagányi *et al.* (2001) used the iterative procedure described by Glazer (1999) to develop a weighted GLM, where the weights are the inverse of the variance,  $1/\sigma_i^2$ . In this instance, the assumption is made that  $\sigma_i^2 = a + b/E_i$ , with  $E_i$  the effort (dive time) for observation  $i$ . The resultant maximum likelihood estimates for  $a$  and  $b$  were 0.8 and 20.5 minutes respectively. There was very little difference between the standardised CPUE indices obtained using these two different approaches, such that the former was considered adequate for current purposes.

A summary of the model results is presented in Table 5.4. Full model results are given in document WG/AB/03/06/02 listed in the Appendix at the end of the thesis. Approximately 41% of the total variation is explained by the model. The nominal and standardised CPUE indices are shown in Table 5.5, and Figs. 5.2 a-e show graphical comparisons of the same. Note that in the plots, the nominal and the standardised values have been divided by the mean value for their respective series. This was necessary because the standardised values are slightly lower in absolute terms than the nominal values, this being a reflection of the fact that the former have been standardised in terms of the fisher code no. 333, (the “new” divers lumped into this category being slightly less efficient than average). This is not ideal but cannot be rectified in this instance because the statistical package SPSS standardises on the largest data entry in a category.

## 5.6 GLM MODEL IMPLEMENTATION RESULTS – SEASON 1 ONLY

An analysis of the numbers of data entries for each of the four seasons revealed that considerably more data are available for Season 1 (Jan-Mar) (Table 5.6). However, despite the apparent advantages of basing the CPUE index on a single season’s data only, it became apparent in this process that one clear disadvantage was that this greatly increases the number of missing cells in the database because there are a number of instances in which there are no data for a particular Zone and year in season 1 (Table 5.7). Approximately 40% of the total variation is explained by the model.

## 5.7 DISCUSSION

The principle objective of the GLM analyses conducted here has been to obtain a model that incorporates factors that explain a significant fraction of the variation in the abalone CPUE data and to obtain a standardised CPUE series that indexes abundance. The updated standardised CPUE series described in this manuscript is similar to that used in 2002 except that another season's data have been added and the effect explored of basing the index on a single season's data only. The GLM-standardised CPUE series presented in Table 5.5 was considered adequate for use in the 2003 abalone assessment, but could be improved in the future.

In particular, some complications have arisen because of differences between the efficiency and variability associated with data from the old entitlement holders (1-84) compared to the newer entrants into the fishery (post 1998) coded as divers 85-241. These two categories of fisher are lumped together in the current analysis, but an examination of the model residuals associated with each of these two categories revealed that the latter category show a higher variance than do the former (residual standard deviation of 1.208 compared to 0.982), and hence should be downweighted in the model.

The attempt to base the CPUE index on a single season's data only was not considered satisfactory because there were too few data available for some zones in some years (Table 5.7). The trends obtained were generally similar to those obtained using all the data and are not presented here given that this approach will not be pursued in the future.

The number of contributory data points available for Zone C (and subarea CP in particular) for Model years 1999 and 2000 is very small ( $n < 60$ ) due to the small quota allocated in these years. It may thus be advisable to downweight such points by weighting them as inversely proportional to the standard error on the year effects in the (base-case) model output. Examination of the latter revealed that the average standard error associated with the data for CNP and CP in 1999 and 2000 was 0.23 compared to an average of 0.05 for the rest of the data (with a similar magnitude of difference in the CV's). Note that analysis of the standard errors alone is not entirely satisfactory because SPSS standardises on the last entry in a series and the data for 2002 (and Zone D) are not necessarily representative. Nonetheless, based on the examination of the S.E.'s and CV's, there clearly is a case to be made for

downweighting data based on a few points only. As a quick test as to whether this would lead to any marked changes, an extreme scenario was investigated in which the model was rerun with a zero weighting factor applied to all the 1999 and 2000 CP/CNP data (Table 5.8). There were only extremely small differences between the two sets of CPUE indices. For practical purposes there is thus no difference and hence this aspect will not be pursued further in the immediate future given other priorities.

Finally, as recommended by the BENEFIT review panel, it is advisable to investigate the implications of alternative error structure models such as the negative binomial. Future work will take this into account but it is considered unlikely that it will have any major effect on results. Future work will also give consideration to including the nominal CPUE directly into the likelihood function.

Table 5.1. The number of data entries per zone used in the GLM analysis to standardise the commercial CPUE series. Note the small number of entries in Zone C during the 1999 and 2000 fishing seasons. Zone CNP was closed during the 2001 fishing season and Zone CP during both the 2001 and 2002 fishing seasons as indicated by the absence of data for these years. Model years are defined as the period October to September.

MODEL YEAR	A	B	CNP	CP	D	ABCD
1980	222	463	58	718	502	1963
1981	223	504	124	571	361	1783
1982	336	550	100	583	583	2152
1983	451	662	129	452	296	1990
1984	287	588	249	325	361	1810
1985	323	600	146	371	602	2042
1986	328	751	221	445	205	1950
1987	423	574	105	494	144	1740
1988	442	410	96	498	147	1593
1989	436	400	87	507	187	1617
1990	510	392	138	458	140	1638
1991	428	393	161	544	168	1694
1992	416	301	94	398	142	1351
1993	310	234	105	335	75	1059
1994	436	276	146	286	158	1302
1995	490	235	138	333	169	1365
1996	592	310	424	475	202	2003
1997	921	243	260	116	198	1738
1998	870	504	223	77	327	2001
1999	584	398	57	5	304	1348
2000	459	347	27	2	315	1150
2001	399	293			141	833
2002	295	233	110		100	738



Table 5.2. Summary of abalone catch (MT) per model year for Zones A-D based on the GLM input data set. Note that these figures are underestimates of catch because some of the data have been excluded from the GLM analysis.

MODEL YR	A	B	CNP	CP	D	ABCD
1980	129.9	151.3	13.3	156.0	174.3	624.8
1981	120.2	156.4	34.5	133.5	121.7	566.3
1982	159.7	172.7	24.3	129.7	186.6	673.0
1983	213.3	194.7	33.7	102.7	78.6	623.0
1984	143.0	172.1	67.5	84.6	97.3	564.5
1985	124.5	171.9	38.1	101.1	157.2	592.9
1986	125.7	223.3	57.0	120.6	50.3	576.9
1987	162.5	161.0	29.7	126.4	45.2	524.8
1988	186.7	125.7	26.2	139.5	49.2	527.3
1989	183.4	129.9	27.3	135.6	52.1	528.2
1990	190.5	134.7	40.1	116.6	46.3	528.2
1991	171.6	132.5	41.0	119.0	49.8	513.9
1992	216.7	146.4	28.4	113.9	56.3	561.7
1993	185.5	149.9	30.4	105.2	53.7	524.6
1994	213.0	140.7	41.9	90.3	91.1	577.0
1995	230.2	150.9	39.1	84.3	89.9	594.3
1996	225.8	139.2	64.6	62.2	87.0	578.8
1997	219.3	142.8	36.9	16.1	91.8	507.0
1998	184.7	146.0	23.2	7.7	107.3	468.9
1999	179.4	145.2	11.3	1.1	103.8	440.8
2000	182.4	141.9	4.1	0.2	100.3	428.8
2001	155.8	114.2	0.0	0.0	35.4	305.4
2002	113.6	86.6	31.9	0.0	19.3	251.5

Table 5.3. Summary of total effort (hours dived) per model year (October to September) for Zones A-D based on the GLM input data set. Note that these figures are underestimates of effort because some of the data have been excluded from the GLM analysis.

	A	B	CNP	CP	D	ABCD
1980	952.3	1713.7	178.5	1948.4	1751.2	6544.1
1981	965.7	1709.1	403.2	1651.1	1304.6	6033.6
1982	1408.0	1884.2	307.1	1639.0	2042.9	7281.1
1983	1902.9	2228.1	396.0	1259.6	959.5	6746.1
1984	1171.4	1913.6	784.0	1033.1	1136.3	6038.3
1985	1085.3	1839.1	444.6	1139.6	1760.1	6268.7
1986	997.4	2222.9	597.5	1191.2	599.6	5608.6
1987	1248.9	1674.2	286.5	1327.1	479.7	5016.4
1988	1359.1	1173.0	241.5	1347.6	449.8	4570.9
1989	1430.5	1174.5	255.6	1320.8	544.3	4725.6
1990	1336.4	1012.3	319.3	1047.4	343.7	4059.2
1991	1196.8	969.7	373.8	1197.8	395.3	4133.4
1992	1347.0	1000.7	258.1	1005.5	449.7	4061.0
1993	1123.5	883.6	303.1	849.3	252.8	3412.3
1994	1289.4	962.8	372.0	724.7	481.7	3830.6
1995	1512.3	936.9	350.8	795.4	543.6	4138.9
1996	1481.1	952.7	748.2	759.6	556.5	4498.1
1997	1634.0	890.8	500.0	255.3	569.7	3849.7
1998	1292.4	1024.9	271.6	108.6	642.6	3340.0
1999	1589.3	1129.3	118.3	12.3	927.3	3776.4
2000	1468.0	1136.5	36.7	2.0	985.1	3628.2
2001	1223.5	956.6	0.0	0.0	387.9	2568.0
2002	884.2	701.8	286.3	0.0	239.2	2111.5

Table 5.4. Results of the GLM analysis (using SPSS 10.1) showing the variables included in the model and some corresponding model statistics. A Type III Sum of Squares (equivalent to Yates' weighted-sum-of-means technique) was used. The variables are abbreviated as follows: MODEL\_YR = Model year (Oct to Sept.) (1980-2002), SEASON = season (Jan-Mar = 1; Apr-Jun = 2; Jul-Sep = 3; Oct-Dec =4), SPLTZONE = zone/subarea (A,B,CNP,CP,D), FISHER\_C = entitlement holder (1-84) and diver (85-241,333). Diver code 333 represents 40 recent divers who had not yet been assigned diver codes and hence were lumped into one category.

**Tests of Between-Subjects Effects<sup>a</sup>**

Dependent Variable: LNCPUE

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	3405.159 <sup>a</sup>	392	8.687	63.610	.000
Intercept	90.136	1	90.136	660.050	.000
ZONE * YEAR	235.147	85	2.766	20.258	.000
SEASON * YEAR	137.116	61	2.248	16.460	.000
FISHER	1103.353	217	5.085	37.233	.000
SEASON	40.134	3	13.378	97.963	.000
ZONE	221.557	4	55.389	405.604	.000
YEAR	108.790	22	4.945	36.211	.000
Error	4979.792	36466	.137		
Total	20877.121	36859			
Corrected Total	8384.951	36858			

a. R Squared = .406 (Adjusted R Squared = .400)

b. Weighted Least Squares Regression - Weighted by WT1

Table 5.5. Nominal and standardised commercial CPUE series for abalone for model years (October to September) 1980 to 2002 and Zones A, B, C (shown separately for subareas CNP and CP) and D. Both the nominal and the standardised values have been divided by the mean value of the respective series. Data are plotted in Fig. 5.2 a-e.

	Zones				
	A	B	CNP	CP	D
<u>Nominal CPUE series</u>					
1980	0.976	0.719	0.818	0.832	0.823
1981	0.904	0.729	0.859	0.840	0.769
1982	0.808	0.730	0.822	0.823	0.733
1983	0.817	0.701	0.866	0.858	0.666
1984	0.883	0.737	0.881	0.856	0.720
1985	0.825	0.750	0.864	0.947	0.736
1986	0.923	0.821	0.955	1.084	0.700
1987	0.947	0.802	1.067	1.035	0.789
1988	1.013	0.877	1.106	1.125	0.940
1989	0.922	0.889	1.077	1.094	0.814
1990	1.056	1.081	1.276	1.190	1.132
1991	1.047	1.134	1.140	1.077	1.122
1992	1.215	1.191	1.180	1.205	1.058
1993	1.256	1.418	1.029	1.267	1.735
1994	1.222	1.228	1.156	1.297	1.557
1995	1.151	1.327	1.169	1.108	1.344
1996	1.172	1.261	0.927	0.882	1.324
1997	1.051	1.341	0.804	0.713	1.364
1998	1.128	1.202	0.913	0.734	1.416
1999	0.864	1.052	0.922	0.939	0.933
2000	0.945	1.025	1.036	1.095	0.861
2001	0.932	0.976			0.780
2002	0.941	1.006	1.132		0.686
<u>Standardised CPUE series</u>					
1980	0.991	0.768	0.849	0.914	0.866
1981	0.953	0.796	0.963	0.932	0.845
1982	0.849	0.809	0.946	0.914	0.812
1983	0.841	0.759	0.915	0.928	0.730
1984	0.881	0.755	0.878	0.888	0.738
1985	0.864	0.800	0.907	0.975	0.786
1986	0.909	0.891	0.995	1.155	0.845
1987	0.931	0.812	1.045	1.001	0.884
1988	0.964	0.886	1.031	1.069	1.002
1989	0.983	0.951	1.123	1.132	0.943
1990	1.009	1.037	1.190	1.124	1.141
1991	1.024	0.983	1.121	1.033	0.971
1992	1.205	1.215	1.196	1.174	1.128
1993	1.078	1.264	1.007	1.261	1.546
1994	0.957	0.934	1.037	1.122	1.153
1995	1.077	1.132	1.110	1.106	1.134
1996	1.190	1.222	0.990	0.974	1.239
1997	1.085	1.403	0.782	0.710	1.353
1998	1.126	1.261	0.909	0.686	1.390
1999	0.932	1.126	0.988	1.028	0.974
2000	1.002	1.074	1.009	0.874	0.898
2001	1.083	1.096			0.862
2002	1.065	1.027	1.011		0.761

Table 5.6. Summary of the number of data entries per Model year per season, where seasons are defined as follows (Jan-Mar = 1; Apr-Jun = 2; Jul-Sep = 3; Oct-Dec =4) and a Model year is defined as the period October to September.

	Season 1	Season 2	Season 3	Season 4
1980	1069	446	448	
1981	956	342	444	41
1982	1084	454	440	174
1983	798	521	539	132
1984	968	448	381	13
1985	962	478	417	185
1986	593	474	77	806
1987	489	362	9	880
1988	469	380	20	724
1989	563	187		867
1990	866	16		756
1991	504	269	76	845
1992	524	23	152	652
1993	267	29		763
1994	332	44	56	870
1995	368	151	28	818
1996	1123	498	202	180
1997	287	112	937	402
1998	890	591	415	105
1999	589	506	253	
2000	615	304	42	189
2001	544	193	12	84
2002	370	174	71	123
<b>TOTAL</b>	<b>15230</b>	<b>7002</b>	<b>5019</b>	<b>9609</b>

Table 5.7. Summary of the number of data entries per year per Zone during Season 1.

	Zone				
	A	B	CNP	CP	
1980	154	259	41	332	283
1981	163	298	67	212	216
1982	179	252	70	295	288
1983	236	248	51	176	87
1984	165	272	126	213	192
1985	135	206	81	285	255
1986	61	298	55	123	56
1987	70	169	21	188	41
1988	135	110	22	138	64
1989	157	147	22	182	55
1990	361	130	70	267	38
1991	119	69	60	222	34
1992	228		52	193	50
1993	6		48	212	
1994	164		43	112	13
1995	146		65	157	
1996	533	211	149	85	145
1997	179		48	39	21
1998	213	434	25	5	213
1999	157	269	13	1	149
2000	266	167	15	2	165
2001	294	179			71
2002	139	156	41		34
<b>TOTAL</b>	<b>4260</b>	<b>3876</b>	<b>1185</b>	<b>3439</b>	<b>2470</b>

Table 5.8. Sensitivity results of standardised commercial CPUE series for abalone for model years (October to September) 1980 to 2002 and Zones A, B, C (shown separately for subareas CNP and CP) and D; and obtained after rerunning to exclude points where the number of contributory points per year per Zone is < 60.

Zones:	A	B	CNP	CP	D
<u>After additional weighting</u>					
1980	0.992	0.768	0.849	0.909	0.866
1981	0.953	0.796	0.963	0.927	0.845
1982	0.849	0.809	0.946	0.910	0.812
1983	0.841	0.760	0.915	0.924	0.730
1984	0.882	0.755	0.878	0.884	0.738
1985	0.865	0.800	0.906	0.970	0.786
1986	0.909	0.891	0.995	1.149	0.846
1987	0.931	0.812	1.045	0.996	0.884
1988	0.964	0.886	1.031	1.063	1.002
1989	0.984	0.951	1.123	1.126	0.944
1990	1.010	1.038	1.190	1.118	1.141
1991	1.023	0.984	1.120	1.028	0.971
1992	1.205	1.215	1.196	1.169	1.129
1993	1.079	1.265	1.007	1.254	1.546
1994	0.957	0.935	1.036	1.116	1.153
1995	1.077	1.132	1.110	1.100	1.134
1996	1.190	1.223	0.989	0.969	1.239
1997	1.085	1.403	0.782	0.707	1.354
1998	1.125	1.261	0.909	0.683	1.391
1999	0.931	1.122			0.971
2000	1.003	1.075			0.898
2001	1.082	1.095			0.861
2002	1.064	1.025	1.010		0.759

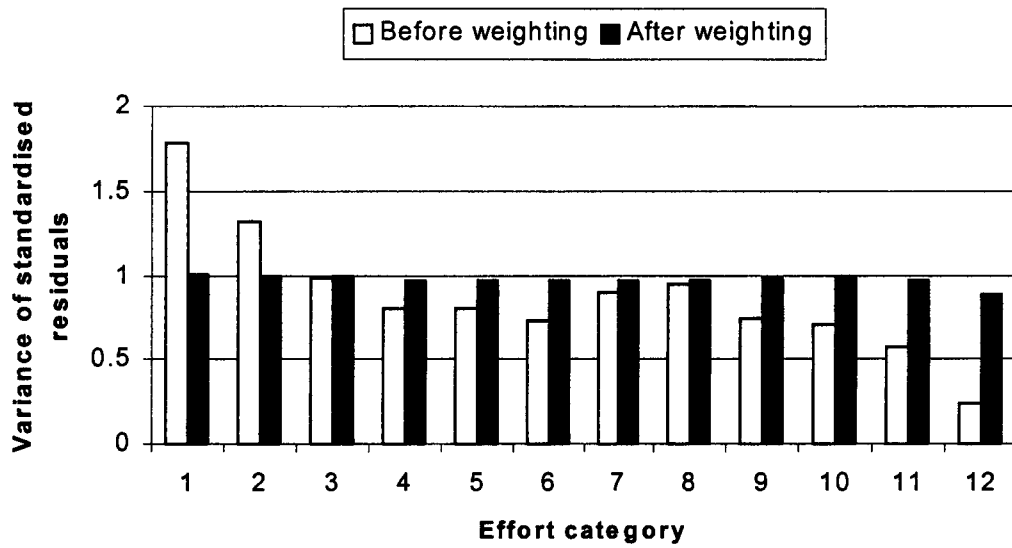


Fig. 5.1 Plot of the average variance of the residuals (corresponding to an initial GLM fit of the data) versus effort category (50 minutes interval). To take account of this heteroscedasticity, a weighted GLM was developed as described in the text, with the final (homoscedastic) pattern of residuals shown alongside.

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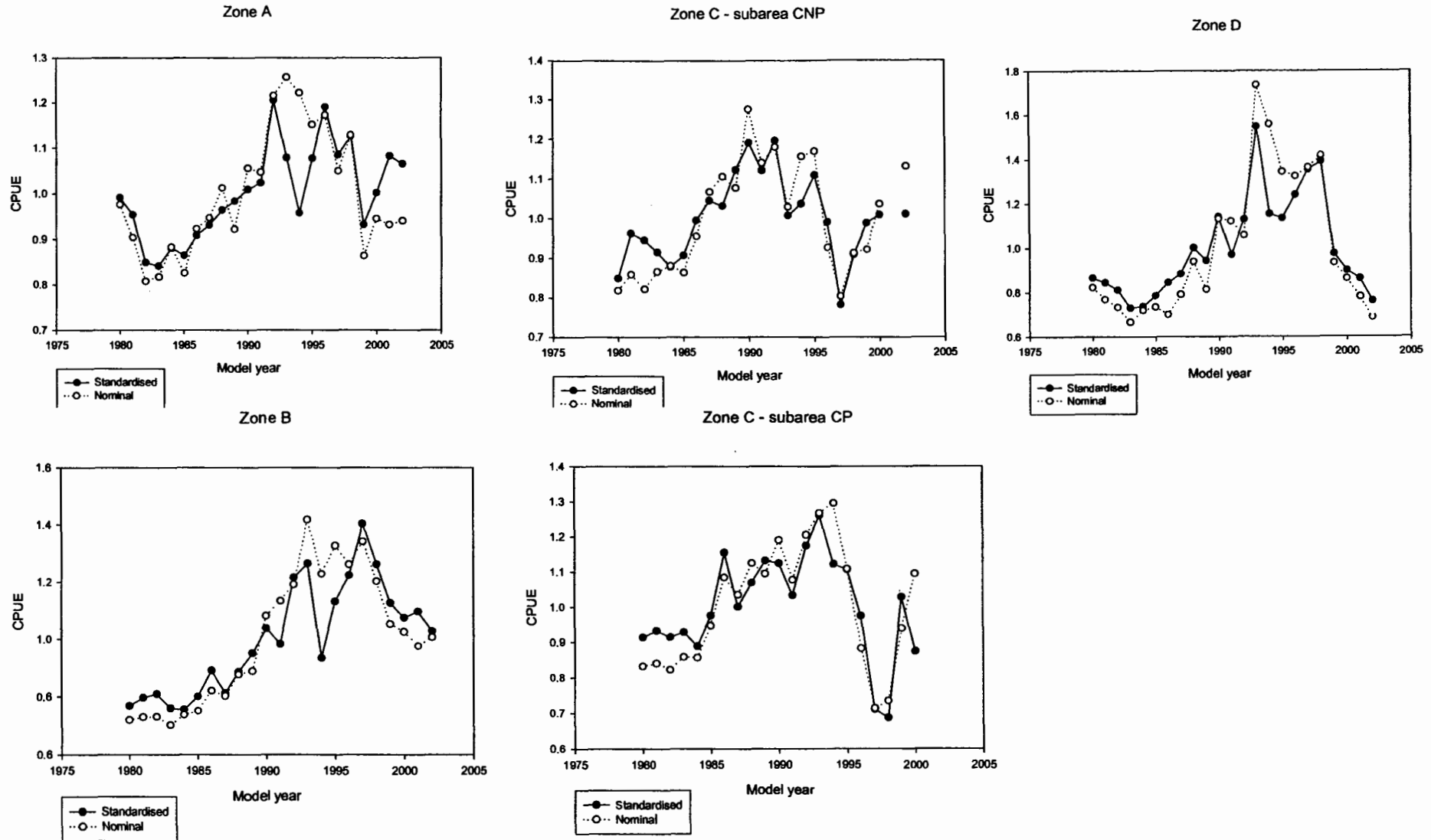


Fig. 5.2 a-e. Comparisons between the nominal and GLM-standardised catch-per-unit-effort (CPUE) trends for zones/subareas A, B, CNP, CP and D. For ease of viewing, both the nominal and the standardised values have been divided by the mean value of the respective series. Values are derived from an iterative effort weighted GLM (see text). Note that the vertical axes have non-zero intercepts for better visual discrimination of trends.

## Chapter 6

# Development of an age- and spatially-structured assessment model to simultaneously represent abalone resource dynamics in "normal" management Zones A and B and in "lobster-invaded" Zones C and D

### SUMMARY

The development of an age- and spatially-structured model for use in estimating the resource dynamics parameters and projecting biomass trends for the South African abalone *Haliotis midae* is described. This constituted the first quantitative approach applied to the management of this commercially valuable resource. Complicating factors that have had to be taken into account in the model construction process have included uncertainty regarding the magnitude of the catch taken by the illegal (poaching) sector and regarding the effect on abalone recruitment and survival of an invasion of rock lobsters into Zones C and D during the 1990's. Novel features of the stock assessment modelling approach employed here include the explicit differentiation between inshore and offshore components; the attempts to estimate the magnitude of the poached catch by fitting to CPUE and Fishery-Independent Abalone Survey (FIAS) abundance data as well as several years of catch-at-age (cohort-sliced from catch-at-length) data for the various components of the fishery; and the method of simultaneously fitting to "normal" management Zone B and "lobster-invaded" Zone C so as to estimate the magnitude of the recent increase in juvenile abalone mortality rates in Zone C, likely as a consequence of the lobster invasion. The most recent version of the model is simultaneously fit to Zone C (which comprises two separate subareas CNP and CP) and Zones A, B and D.

### 6.1 INTRODUCTION

#### *6.1.1 Approaches used elsewhere in the world*

Several different stock assessment approaches have been used in abalone fishery management and a review of these is given in Breen (1992). One of the greatest impediments to the application of stock assessment models to abalone fisheries has been that such models usually rely on commercial catch per unit effort (CPUE) data as an index of abundance. Several authors have stressed that CPUE is not a reliable index in situations where the stock is highly aggregated, as occurs for example off southeastern Australia (Breen 1992, Keesing & Baker 1998). Here it is argued that CPUE data have utility in the South African context, especially when used in conjunction with a fishery-independent index. A second impediment



to applying standard stock assessment models to abalone populations has been the lack of adequate data on age. Tarr (1995) used a tagging study to estimate growth rates of *Haliotis midae*. This in turn allowed the age structure of catches to be estimated (admittedly by the rather coarse method of cohort slicing), and consequently it has been possible to construct a fully age-structured population model to assess the status and productivity of the South African resource.

### **6.1.2 Appropriateness of using CPUE data**

The appropriateness or otherwise of using CPUE data as an index of abalone stock abundance has in the past been questioned both in the South African context and elsewhere. There are, however, a number of reasons why CPUE (available from 1980 onwards) may be a reasonable index of abalone stock abundance in the South African context but not necessarily elsewhere in the world. The major South African abalone fishery is located in shallow kelp bed areas, relatively close inshore, along a stretch of coastline measuring some 150 kilometres and with relatively easy access to most of the areas (R. Tarr, pers. commn). Furthermore, the ten or so launching sites currently in use along this coastline are exactly the same sites that were in use during the 1950s when the fishery began, and hence fishers have always fished fairly evenly throughout the area. Even if very large aggregations existed in the past, the resource was so heavily fished during the 1960s that these would since have been reduced substantially and hence the effect of spatial patchiness dampened to a large extent.

Abalone fishing effort in South Africa is spread over most of the major fishing Zones. The general pattern of diver behaviour is thus not one in which new aggregations are sought out and then sequentially depleted so that catch rates remain fairly stable despite the resource abundance declining, as has been demonstrated for fisheries off Australia (Shepherd & Baker 1992), California (Tegner *et al.* 1989) and Mexico (Prince & del Proó 1993). It is now well established that in areas such as South Australia and Tasmania the spatial variation in density is such that, coupled with the added complexities of diver behaviour, CPUE cannot be considered a reliable indicator of stock abundance in these areas (Breen 1986, Prince 1992, Shepherd *et al.* 1992, Keesing & Baker 1998, Worthington & Andrew 1998). As proposed by Dichmont *et al.* (2000) one option available to resource managers with some confidence in

CPUE as an index of abundance for their fishery, is to use a dynamic model tuned to both CPUE data and also a survey index.

The Fishery Independent Abalone Survey (FIAS) data series (see Chapter 3) not only provide a valuable independent index of abundance but are also useful because they include information on abalone from age 5 upwards compared with the CPUE data, which captures information on animals aged approximately 9 years or more. This means that the FIAS abundance index is more sensitive than the CPUE index to changes in the younger age classes. This is particularly important in Zones C and D where extensive poaching of sublegal-sized abalone individuals as well as a hypothesized multispecies effect (Tarr *et al.* 1996) have caused a rapid decline in the numbers of small individuals. Consequently, the FIAS data have indicated a more negative scenario than have the CPUE data and are thereby providing valuable clues as regards factors to consider in modelling the resource.

### ***6.1.3 The history of building models to model abalone***

#### **6.1.3.1 From decision rules to models**

Before 1997, management advice for abalone was based solely on a set of decision rules that used data on trends in the commercial CPUE (catch per unit effort), the average and modal sizes of abalone in the commercial catches, and the proportion of the catch represented by the smallest legal sizes. Modelling analyses were impeded by arguments as to the appropriateness or otherwise of using CPUE data as an index of South African abalone stock abundance because, *inter alia*, of questionable comparability through the data series and the apparent contradiction of an observed increase in CPUE estimates during the 1980's despite known increases in overall catch levels in the mid to late 1980's. In 1997 the first simple Age-Structured Production Model (ASPM) (e.g. Hilborn 1990) was constructed and demonstrated that the CPUE data had some utility for assessment purposes after all (see Chapter 7).

The abalone ASPM was initially only fitted to (unstandardised) CPUE data and estimated three resource parameters (the pre-exploitation spawning biomass  $B_0^{sp}$ , the present annual poaching take  $CP_{max}$  and the natural mortality  $M$ ). An important model result was that the maximum sustainable yield ( $MSY$ ) is reduced by nearly one half if catches include animals substantially smaller than the minimum legal size limit, as taken by poachers.

An ASPM approach was specifically chosen in preference to, for example, a Virtual Population Analysis (VPA). VPA requires catch-at-age data be available for all years and essentially reconstructs the history of each cohort, assuming that the observed information is known without error. In contrast the ASPM methodology is more flexible, in that it does not require catch-at-age data for all the years considered, and it can accommodate likely errors in such data, by making assumptions about the selectivity-at-age of the catch (Butterworth *et al.* 2003a). An ASPM assessment involves constructing an age-structured model of the population dynamics and fitting it to all available abundance indices by maximising the likelihood function. Available catch and survey abundance index data can thus be formally incorporated in the analysis.

The methodology used here is essentially an adaptation of that applied for management purposes for many key South African and Namibian fishery resources, including for the major fisheries for hake (*Merluccius Spp.*) in this region (e.g. Butterworth and Geromont, 2001, Butterworth *et al.* 2001, Butterworth and Rademeyer, in press a;). The ASPM approach has also been applied to a number of international stocks such as southern bluefin tuna (e.g. Butterworth and Plagányi 2000) and the Gulf of Maine cod stock (e.g. Butterworth *et al.* 2003 a,b). In a local context, this methodology has been peer-reviewed during annual international assessment review workshops held in Cape Town, South Africa, over the last four years (BENEFIT 2000, 2001, 2002 and 2004), by international reviewers. The terms of reference for the December 2002 BENEFIT Stock Assessment workshop included a critical review of past assessments of the South African abalone resource and many of the recommendations stemming from this review have subsequently been included in the model.

### 6.1.3.2 Other abalone fishery zones

For Zones E, F and G (west coast), the data are not as plentiful as for the south coast, and there are large gaps in the time series, corresponding to times when commercial fishing was not allowed. The modelling effort is therefore concentrating on the south coast part (Zones A-D) of the stock, and the simple decision rules referenced earlier remain currently used as the basis for management on the west coast.

The data for Zone A exhibit insufficient contrast to adequately discriminate between alternative models. Hence from 1997 to 2002, management advice for this Zone was largely based on a replacement yield approach.

### 6.1.3.3 Exploring the effects of linked recruitment

The original model treated each Zone in isolation, but this was extended in 1998 to investigate the effects of linking the recruitment of the abalone resources of Zones C and B (Rosenthal *et al.* 1998 – Appendix A1). This was done by assuming a spawning stock-recruit relationship where spawning stock corresponds to the sum of the spawning biomasses in Zones B and C – the motivation being that the eggs and larvae are pelagic and so could easily move between Zones. It is presently not known to what extent larval mixing occurs throughout the main fishing area, but the AWG later decided that larval and post-larval stages are most likely retained in areas close to the parental population with some larval interchange between adjacent areas. Hence in most later formulations of the model, the recruitment levels for each of the Zones have been calculated separately. McShane (1992) presented evidence for localised dispersal of larval *Haliotis rubra*. This aspect should be explored further in future work.

### 6.1.3.4 Development of a spatially-structured model

In 1999, the first attempt was made to standardise the CPUE data (Moloney 1999) and the ASPM being developed as part of this study was fitted for the first time to FIAS data and

some catch-at-age data. This analysis showed that it was difficult to interpret the abalone dynamics in Zone C without giving consideration to the fact that the western and eastern areas of this Zone differed with respect to both density and size composition. Hence in 2000 Zone C was split into a “poached” subarea (CP) to the west (Hawston/Mudge Point areas) and a “nonpoached” subarea (CNP) to the east (Hermanus vicinity). Nonetheless, the stock assessment model failed to fit the steeply declining FIAS data satisfactorily, and several different scenarios were investigated to try to improve the fit to the FIAS data. It was speculated that the inability of the current ASPM to reproduce the steep decline evident in the FIAS data for Zone C was at least partly due to the fact that the model assumes that commercial, recreational and poaching operations occur homogeneously over space. A spatial ASPM was thus developed (see Appendix 6.1) to take into account the fact that inshore (within 185m of the highwater mark) and offshore components of the resource are affected differently by the different sectors of the fishery.

#### **6.1.3.5 The abalone recruitment programme data**

The numbers of recruiting abalone over the size range 5-25 mm SL were recorded at three study sites east of Cape Hangklip (Betty’s Bay, Mudge Point and Danger Point) during the 1990’s (data from Rob Tarr, MCM). The data from Betty’s Bay (Zone D) and Mudge Point (Zone C) indicate a dramatic drop in recruitment with effect from about August 1994, whereas the data from Danger Point (Zone B) indicate a relatively stable recruitment trend over the 1990’s. The AWG agreed in 2001 that whilst the data had associated problems and could therefore not be used as a reliable relative index of recruitment, there was nonetheless sufficient evidence that recruitment levels in Zones C and D had declined to almost zero since at least 1995. A decision was therefore made to incorporate this feature in the base-case model versions applied to Zones C and D in 2001. For these Zones, a knife-edge drop in recruitment (down to 10% of the 1993 level) was assumed to have occurred as from 1994.

### 6.1.3.6 Independent verification of the model

In 2001, the abalone industry carried out independent modelling analyses (an equilibrium analysis) through their scientific consultants (OLRAC). In 2002, a Reference Case was agreed upon for purposes of cross-checking the output from different modellers using the same model coded independently. This was a useful exercise for checking the model and resulted in constructive discussions, for example, as to the best way to model the changes over time in the average age of the inshore and offshore plus group components. Close but not exact agreement has been achieved between the computed output of the models, but they currently differ in representing the abalone recruitment failure effect (see e.g. Bergh 2002).

### 6.1.3.7 Towards finding the global minimum

One of the difficulties encountered throughout the model development process has been the fact that the likelihood surface is relatively flat and hence there are numerical difficulties in maximising the likelihood. This remains a problem even though the model was recoded in 2002 from a program using the AMOEBA (downhill simplex method) minimiser to AD Model Builder (AD Model Builder™, Otter Research, Ltd.). The latter software provides for faster, more reliable and more powerful (in terms of the number of parameters readily estimable) minimisation when fitting nonlinear models to data. ADMB uses quasi-Newton minimisation, a technique which uses information contained in the derivative of a function  $f$  to follow the function downhill, perhaps along a curving path, until the function reaches a local minimum. A particular advantage of ADMB is that it calculates the Hessian at the minimum to determine whether the results correspond to a classical local minimum (i.e. positive definite Hessian).

### 6.1.3.8 The battle to fit the FIAS data satisfactorily

A consistent problem with the model was the inability to fit the FIAS abundance index for subarea CP satisfactorily. Considering that the poaching trends input to the model were updated in 2002 and hence represent the best current knowledge regarding the trends in

poaching, attention was focused instead on addressing some concerns related to the recruitment decrease effect. In particular, the assumption that the recruitment decrease in Zone C commenced before 1994 was questioned. According to Rob Tarr (pers commn.), a pre-1994 recruitment failure " *is certainly possible, if not in fact likely. Reason being that the method we used to monitor recruitment was the targeting of aggregations of urchins. In 1994 we "ran out" of urchins i.e. no more aggregations. But it is quite possible that the overall numbers of urchin aggregations may have been declining in previous years unknown to us, which is a more likely scenario given a probable sliding scale of lobster incursion / predation as opposed to a 1994 step-wise effect.*"

A number of different scenarios for estimating the extent/rate of decline in the recruitment were investigated and many of these proved unsatisfactory for various reasons. For example, Ianelli (2002) proposes including time series effects (simple autocorrelation) in resource productivity determinations in cases where environmental changes may have shifted the production regime of a species. Following Ianelli (2002), a simple time series component was thus added to the stock-recruitment curve within the model and the stock-recruit residuals estimated for each of the years 1990-2001. As there is a problem in estimating both the autocorrelation coefficient and  $\sigma_r$  simultaneously (J. Ianelli, pers. commn), these parameters were either fixed or estimated separately. However, despite some experimenting with this approach, no statistically significant improvements to the fit of the model to the data resulted.

It was then decided that the best way forward might be to include the recruitment failure effect in a more explicit manner. The best current hypothesis to explain the recruitment failure effect proposes that rock lobsters have increased substantially in Zones C and D and are to be blamed for the recorded collapse of urchin populations and dramatic reductions in the numbers of juvenile abalone (Mayfield and Branch 2000). In particular, it is noteworthy that juvenile abalone (individuals 3-35 mm in length) depend heavily on their commensal association with urchins, most likely because the urchins provide protection against predators and may supplement the diet of juvenile abalone with kelp fragments (Day and Branch 2002). Moreover, abalone recruits are also thought to be adversely affected by the removal of urchins due to increased sedimentation following the removal of urchins (Day and Branch 2002).

Based on the above, it was therefore decided to rather incorporate the recruitment failure effect by allowing for an increase in the natural mortality rates of juvenile abalone in the model. A brief description of the method used is given in Appendix 6.2.

#### **6.1.3.9 Combined fitting to Zones B and C**

One of the problems in trying to model the abalone resource has been the confounding of background natural mortality rate, mortality due to unknown levels of illegal take by the poaching sector and increased juvenile mortality due to the “lobster” effect in Zones C and D. This has been further complicated by the fact that the recruitment failure effect (starting in the early 1990’s) and escalated poaching levels (from 1994 onwards) commenced at approximately the same time. Moreover, both processes have a large effect on the juvenile abalone age classes with the former mainly affecting ages 0 – 3 yr and the latter affecting age classes as young as 4 years. As FIAS samples animals only 5 years and older, it was not possible to discriminate clearly between these effects given catch-at-age data with a minimum age category of 5 years. A major breakthrough was therefore achieved with the 2002 industry/MCM full population surveys conducted in Zones B and C. By fitting to Zones B (no recruitment failure effect) and C (with recruitment failure effect) simultaneously, it was possible to estimate the extent of the “lobster” effect.

In 2002, a combined Zone B & C base-case model was therefore adopted by the AWG. Several of the parameter estimates obtained from this model fit were fixed for use in the combined Zones C & D model and Zones A & C model.

#### **6.1.3.10 Combined fitting to Zones A, B, C and D**

In 2003, a combined Zone A, B, C and D model was attempted. This model version simultaneously models the dynamics in each of ten regions (A, B, CNP, CP, D – inshore and offshore regions for each zone/subarea). As Hessian-based standard errors output by the



ADMB package can be unreliable, confidence intervals for model parameters reported in this thesis have been computed using the profile likelihood method.

Given the various problems encountered in the model development process, this has been a step-wise and ongoing process such that the model is only now at a stage where improved estimates of uncertainty could be computed, for example, through the use of the MCMC (Monte Carlo Markov Chain – see, e.g. Gilks *et al.* 1996) approach. The ADMB package used for the abalone model allows for a fairly straightforward extension of assessments from a maximum likelihood estimation (MLE) approach to a fully Bayesian form, by using the MCMC method to perform the Bayesian integrations. However, this is a particularly time-consuming exercise and will not be discussed further here.

## 6.2. THE MODEL

The full details of the spatial age-structured production model (ASPM) are provided in Appendix 6.1. The equations in the Appendix are described for the case of Zone C. In the base-case combined Zone B & C model, the same equations apply to model the resource in Zone B, and the model is fitted to both Zones simultaneously. The base-case combined ABCD model applies the same equations to Zone A and Zone D also, but fixes some parameter estimates to the values obtained from fitting to Zones B and C. A schematic summary of the model area is shown in Fig. 6.1. A summary of the model assumptions and structure is given below:

### 6.2.1 Definition of inshore and offshore regions

A law in force since 1966 prohibits commercial abalone fishing operations within 185m of the high-water mark, which is used roughly as the dividing line between model inshore and offshore regions. Most of the inshore area consists of shallow 0-5 m depth kelp habitat (Tarr 1993).

As the recreational fishery is essentially a shallow-water fishery with divers accessing the resource mostly from the shore, recreational catches are assumed taken from the inshore model region only. Poaching catches are similarly assumed to derive exclusively from the inshore

model region. The AWG agreed that the 0-2 m depth range is the only habitat that is almost never fished by commercial divers encroaching inshore because the shallow depth prevents boats from operating easily in these waters. Inshore encroachment by commercial divers is seen as being particularly common in areas that do not have residential houses along the beachfront. Thus, though thought to be a relatively minor problem in subarea CNP, inshore encroachment by commercial fishers is considered to have been a problem throughout the history of the fishery in subarea CP and in Zones A, B and D. Commercial fishing is therefore assumed to occur in both the inshore and offshore model regions in subarea CP and Zones A, B and D, under the assumption that fishing effort is the same in these two regions. In subarea CNP, commercial fishing is assumed to be confined to the offshore model region only with effect from 1966.

### ***6.2.2 Selectivity assumptions***

Unique selectivity assumptions are assigned to each of the following sectors (where the various research surveys are also termed ‘sectors’ for convenience): commercial, recreational, poaching, FIAS and “old survey” (i.e. pre-FIAS surveys – see Chapter 3). The same selectivity function is assumed to apply to the inshore and offshore components of the commercial, FIAS and “old survey” sectors. In the case of the 2002 industry/MCM survey, a uniform selectivity function ( $S_a = 1$ ) is assumed because of the extractive sampling methodology used.

### ***6.2.3 Inshore-Offshore movements***

Abalone survey data show decreasing density with depth and increasing average size with depth (Tarr 2003). This is because the 0-5 m depth kelp habitat (the inshore region) is the most suitable habitat for abalone larvae to settle, with larger animals known to gradually move offshore. According to Tarr (pers. comm) it is reasonable to assume that only animals with a shell length > 100 mm (“emergent” animals) are likely to migrate offshore. This corresponds to a minimum age of ca. 5 years and hence it is assumed that only animals older than 5 years move offshore in the model.

The inshore-to-offshore migration parameter estimated for subarea CP ( $\rho$  (CP)) in previous model versions was very low (and much lower than that estimated for subarea CNP). One problem related to this was that the model predicted that, for subarea CP during the last decade, the mean age of the INSHORE plus group was greater than the OFFSHORE plus group mean age, which is clearly unrealistic. To constrain this parameter to a more biologically realistic value, it was fixed such that  $\rho$  (CP) = 0.5 \*  $\rho$  (CNP). This very roughly accounts for the fact that from a biological point of view,  $\rho$  should be similar among Zones, but is likely lower in subarea CP because of the greater offshore extent of the shallow “inshore” region in subarea CP (i.e. it would take animals longer to move out of this shallow “inshore” region in CP into the deeper waters). The migration parameter for Zones A, B and D is set equal to the same value.

#### ***6.2.4 Dividing recruitment between inshore and offshore areas***

The 0-5 m depth kelp habitat is the most suitable habitat for abalone larvae to settle and hence spats are expected to settle predominately in the inshore model region (Fig. 6.1). The proportion of spats settling inshore is set at 90% in the base-case model, although fitted values of these (time invariant) proportions are also considered.

#### ***6.2.5 Assuming local recruitment***

Subareas CNP, CP and Zones A, B and D are all assumed independent in terms of recruitment.

#### ***6.2.6 Natural mortality assumptions***

Natural mortality  $M_a$  is modelled as age dependent with common parameters estimated for all zones. Natural mortality is year invariant except in Zone C after 1990 (see Appendix 6.2).

#### ***6.2.7 Estimating the level of poaching***

The maximum poaching level is estimated (in terms of numbers) within the model and the poaching estimates for the remaining years are then computed using the poaching trend

scenarios as given in Table 4.4 of Chapter 4. In the case of Zone C, early model versions assumed that 90% of the poached catch derived from subarea CP with the remaining 10% assumed taken from subarea CNP. The (year-independent) model parameter  $p_{poach}$  (base-case value = 0.1) specified the proportion of the poached catch that originates from the “nonpoached” subarea CNP, i.e.

$$C_y^{poa,CNP} = p_{poach} (C_y^{poa,CNP} + C_y^{poa,CP}) \quad (6.1)$$

This formulation was modified in 2003 following the suggestion to assume that  $p_{poach}$  relates to the exploitation rate for poachers rather than the catch by this sector (BENEFIT 2002). This was effected by assuming that there is a fixed ratio between the proportions of the resource poached each year from CNP and CP i.e. the parameter  $p_{poach}$  relates to the exploitation rate for poachers. Equation (6.1) above was thus modified as follows:

$$C_y^{poa,CNP} = \frac{p_{poach} \cdot \frac{B \exp_y^{poa,CNP}}{B \exp_y^{poa,CP}} (C_y^{poa,CNP} + C_y^{poa,CP})}{1 + p_{poach} \cdot \frac{B \exp_y^{poa,CNP}}{B \exp_y^{poa,CP}}} \quad (6.2)$$

The “old” base-case  $p_{poach}$  value of 0.1 is thus not directly comparable and hence  $p_{poach}$  was estimated in the fitting process, yielding a best fit estimate of 0.35. As this value resulted in a statistically better overall likelihood value, it was adopted as the base-case. It translates into assuming that, on average, the proportion of the poached catch that originates from the “nonpoached” subarea CNP is 0.097. The time-series of the proportion of the poached catch assumed taken from CNP is shown in Fig. 6.2.

A minimum realistic poaching estimate for a zone is given by the total number of confiscations from that Zone. In instances (e.g. for Zone B) where the model estimate of poaching is less than the number of confiscations, a fixed maximum poaching level is input instead, with a minimum realistic value determined as the total number (location-known + proportion of location-unknown) of confiscations (see Chapter 4). Alternative scenarios are also considered in which a lower confiscation proportion (the proportion of the total poached catch that is confiscated) is assumed. Model sensitivity is also tested using, for example, the Zone B : Zone C maximum poaching level ratios presented in Table 4.4.

### Appendix 6.1. The base-case inshore/offshore population model used for estimating resource dynamics parameters and projecting biomass trends

The model applied is basically an age-structured production model (ASPM) with a spatial extension to allow inshore and offshore components of the resource to be modelled separately with exchange between the two. In each of Zones A, B and D, the resource is modelled using two compartments – an inshore and offshore compartment. Zone C is split further into two subareas, one “poached” (CP) and one “nonpoached” (CNP), and hence the model for this Zone has four compartments. The description which follows is for the Zone C base-case model. Although certain parameter values (such as natural mortality  $M$ ) and the commercial and survey selectivity parameters are forced to be the same in each of the Zone C model compartments, the dynamics of the following four compartments are modelled separately (exchange contributions apart):

- i) nonpoached subarea CNP inshore (within 185 m of the highwater mark) compartment;
- ii) CNP offshore component;
- iii) poached subarea CP inshore component; and
- iv) CP offshore component.

#### 6.1.1 Dynamics

For each subarea, the dynamics of the **inshore** component are given by:

$$N_{y+1,0}^I = r_I \cdot R(B_{y+1}^{sp}) \quad (\text{A6.1.1})$$

$$N_{y+1,a+1}^I = \left( N_{y,a}^I e^{-\frac{M_a}{4}} - C_{y,a}^I \right) e^{-\frac{3M_a}{4}} \quad 0 \leq a \leq 4 \quad (\text{A6.1.2})$$

$$N_{y+1,a+1}^I = \left( (1-\rho) \cdot N_{y,a}^I e^{-\frac{M_a}{4}} - C_{y,a}^I \right) e^{-\frac{3M_a}{4}} \quad 5 \leq a \leq z-2 \quad (\text{A6.1.3})$$

$$N_{y+1,z}^I = \left( (1-\rho) \cdot N_{y,z}^I e^{-\frac{M_z}{4}} - C_{y,z}^I \right) e^{-\frac{3M_z}{4}} + \left( (1-\rho) \cdot N_{y,z-1}^I e^{-\frac{M_{z-1}}{4}} - C_{y,z-1}^I \right) e^{-\frac{3M_{z-1}}{4}} \quad (\text{A6.1.4})$$

- where  $N_{y,a}^I$  is the *inshore* number of abalone of age  $a$  at the start of Model year  $y$ ,
- $\rho$  is the proportion of inshore animals of age  $a$  ( $5 \leq a \leq z$ ) that move offshore at the start of Model year  $y$ ,
- $C_{y,a}^I$  is the total number of abalone of age  $a$  taken by recreationals and by poachers in Model year  $y$ , as well as the *inshore* number of abalone taken by the commercial fishery,
- $R(B^{sp})$  is the recruitment vs spawner biomass relationship assumed (see below),
- $r_I$  is the proportion of the recruits which settle inshore,
- $M_a$  is the (time-invariant) natural mortality rate on abalone of age  $a$ , and
- $z$  is the largest age considered (i.e. corresponding to a “plus group”).

Similarly, for each subarea, the dynamics of the **offshore** component are given by:

$$N_{y+1,0}^O = r_O \cdot R(B_{y+1}^{sp}) \quad (\text{A6.1.5})$$

$$N_{y+1,a+1}^O = \left( N_{y,a}^O e^{-\frac{M_a}{4}} - C_{y,a}^O \right) e^{-\frac{3M_a}{4}} \quad 0 \leq a \leq 4 \quad (\text{A6.1.6})$$

$$N_{y+1,a+1}^O = \left( (N_{y,a}^O + \rho \cdot N_{y,a}^I) e^{-\frac{M_a}{4}} - C_{y,a}^O \right) e^{-\frac{3M_a}{4}} \quad 5 \leq a \leq z-2 \quad (\text{A6.1.7})$$

$$N_{y+1,z}^O = \left( (N_{y,z}^O + \rho \cdot N_{y,z}^I) e^{-\frac{M_z}{4}} - C_{y,z}^O \right) e^{-\frac{3M_z}{4}} + \left( (N_{y,z-1}^O + \rho \cdot N_{y,z-1}^I) e^{-\frac{M_{z-1}}{4}} - C_{y,z-1}^O \right) e^{-\frac{3M_{z-1}}{4}} \quad (\text{A6.1.8})$$

where  $N_{y,a}^O$  is the *offshore* number of abalone of age  $a$  at the start of Model year  $y$ ,  
 $r_O$  is the proportion of the recruits which settle offshore ( $= 1-r_I$ ), and  
 $C_{y,a}^O$  is the *offshore* number of abalone of age  $a$  taken by the commercial fishery.

The commercial abalone fishery season currently extends from October to June but several historic changes in the commencement and closure dates for the commercial fishing season are on record (see Chapter 3). For reasons of internal consistency in the assessment process, a standard Model or fishing year  $y$  is thus taken to run from October of year  $y-1$  to September of year  $y$ . The population model used here assumes pulse fishing (Pope's approximation – Pope 1984), rather than the more customary Baranov catch equations which assume continuous fishing through the year (Baranov 1918). Pope's approximation has been used in order to simplify computations. As long as mortality rates are not too high, the differences between the Baranov and Pope formulations will be minimal. The approximation of the fishery as a pulse catch at the start of each calendar year is here considered to be of sufficient accuracy given that most of the catch is made over the October-March period, and because the annual catches from this long lived resource are not that large a fraction of the overall biomass. This last reason also constitutes the justification for treating inshore-to-offshore movement as a pulse at the start of the Model year. The equations reflect the fact that catches are subtracted at the end of the first quarter of the Model year (i.e. in the middle of the October-March period of high catches). As the fishery-independent surveys (FIAS) are conducted only towards the end of the second quarter of the Model year, comparisons with the abundance indices obtained from FIAS are made at time  $y + \frac{1}{2}$  in terms of the model whereas comparisons with the CPUE data are made at time  $y + \frac{1}{4}$  in the model.

Because different sectors of the fishery exhibit different selectivity patterns with age, the following five sectors are explicitly differentiated in the model: the commercial fishery sector (mostly offshore); the recreational sector (mostly inshore); the poaching/illegal sector (mostly inshore), the fishery-independent survey (inshore and offshore) and the "old survey" (inshore and offshore). Note that the small-scale commercial sector of the fishery is not modelled separately in the current analysis but allocations to this sector have simply been added to the commercial catches for 2001 and 2002.

The equations given below are applied separately to each of the inshore and offshore components of the two subareas.

The total number of abalone of age  $a$  caught each year ( $C_{y,a}$ ) is given by:

$$C_{y,a} = \sum_s C_{y,a}^s \quad (\text{A6.1.9})$$

where  $s$  indicates the sector of the fishery (e.g. commercial, recreational, poaching).

The annual catch by mass ( $C_y^s$ ) for sector  $s$  is given by:

$$C_y^s = \sum_{a=4}^z w_{y,a+\frac{1}{4}} C_{y,a}^s \quad (\text{A6.1.10})$$

where  $w_{y,a+\frac{1}{4}}$  is the mass of an abalone of age  $a$  at the end of the first quarter of Model year  $y$  (note however that only the plus group mass  $w_{y,z}$  is year-dependent in the model formulation pursued and that the plus group mass is modelled separately for the inshore and offshore components). The summation is taken from age  $a = 4$  as no abalone of a size corresponding to ages below 4 are taken by any of the fishing sectors.

A von Bertalanffy growth equation is used to relate shell length  $\ell$  (mm) to age in years ( $t$ ), and is based on tagging data from Betty's Bay (Tarr 1995):

$$\ell(t) = \ell_\infty [1 - e^{-k(t-t_0)}] \quad (\text{A6.1.11})$$

The relationship between *shell length* (mm) and abalone whole wet mass (g) is based on data from the Betty's Bay and Danger Point areas and is determined using the following power relationship that was revised from the original during the course of this study:

$$w_{y,a} = w(y, t = a) = c \cdot (\ell)^d \quad (\text{A6.1.12})$$

Note that mass-at-age is year-independent for abalone of age  $a < z$  and that  $w_{y,a+\frac{1}{4}} = w(y, t = a + \frac{1}{4})$  is computed for use in calculating the sector-specific exploitable biomasses after the first quarter of each year (see below). However, the mass-at-age for the plus group varies over time, depending on the average age of the inshore and offshore plus group components in year  $y$ ,  $\bar{z}_y^I$  and  $\bar{z}_y^O$  respectively, which are calculated as:

$$\bar{z}_y^I = \frac{(\bar{z}_{y-1}^I + 1)((1 - \rho)N_{y,z}^I - C_{y,z}^I)e^{-M_z} + z \cdot ((1 - \rho)N_{y,z-1}^I - C_{y,z-1}^I)e^{-M_{z-1}}}{N_{y,z}^I} \quad (\text{A6.1.13})$$

$$\bar{z}_y^O = \frac{\left(\bar{z}_{y-1}^O + 1\right)\left(N_{y,z}^O - C_{y,z}^O\right) + \left(\bar{z}_{y-1}^I + 1\right)\rho N_{y,z}^I e^{-M_z} + z \cdot \left(N_{y,z-1}^O + \rho N_{y,z-1}^I - C_{y,z-1}^O\right) e^{-M_{z-1}}}{N_{y,z}^O} \quad (\text{A6.1.14})$$

The above is an approximation only (as it ignores, e.g., the fact that catches are subtracted not at the start of the year but at the end of the first quarter of each year) but is considered sufficiently accurate for present purposes.

The recreational catch by mass in year  $y$  is given by:

$$C_y^s = \sum_{a=8}^{z-1} w_{a+\frac{1}{4}} N_{y,a}^I (1-\rho) e^{-M_{a/4}} S_a^s F_y^s + w_{y,\bar{z}_y+\frac{1}{4}}^I N_{y,z}^I (1-\rho) e^{-M_{z/4}} S_z^s F_y^s \quad (\text{A6.1.15})$$

and the poaching catch by mass in year  $y$  by:

$$C_y^s = w_{4+\frac{1}{4}} N_{y,4}^I e^{-M_{4/4}} S_4^s F_y^s + \sum_{a=5}^{z-1} w_{a+\frac{1}{4}} N_{y,a}^I (1-\rho) e^{-M_{a/4}} S_a^s F_y^s + w_{y,\bar{z}_y+\frac{1}{4}}^I (1-\rho) N_{y,z}^I e^{-M_{z/4}} S_z^s F_y^s \quad (\text{A6.1.16})$$

where  $S_a^s$  is the fishing selectivity-at-age for sector  $s$  (this pattern is assumed not to change over time),  $w_{y,\bar{z}_y+\frac{1}{4}}^I$  is the mean mass of the inshore plus group with average age  $\bar{z}_y + \frac{1}{4}$  after the first quarter of Model year  $y$ , and  $F_y^s$  is the fishing “mortality” (strictly here that proportion of the numbers present after the first quarter of the Model year which are caught) at a reference age, set for these computations to be  $a = 11$  for all sectors. Based on data from A. Mackenzie (Marine & Coastal Management, pers commn), the minimum age of animals assumed caught by the poaching sector is 4 years, so that for this sector  $S_a^s = 0$  for  $a < 4$ . Note also (cf. Eqn. A6.1.16) that there is no inshore-offshore movement of animals aged four and younger. The commercial and recreational sectors are both assumed not to catch animals below the legal size limit, so that for these sectors  $S_a^s = 0$  for  $a < 8$ .

In the case of the recreational sector (which reports in terms of numbers rather than mass), estimates of the annual catch by mass are computed using equation (A6.1.15) but it is necessary to first compute the fishing “mortality”  $F_y^s$ , using the following relation for the numbers caught in year  $y$ :

$$N_y^s = \sum_{a=8}^z N_{y,a}^I (1-\rho) e^{-M_{a/4}} S_a^s F_y^s \quad (\text{A6.1.17})$$

The relative proportions of the Zone C recreational catch (i.t.o. numbers) taken from the two subareas CP and CNP is assumed to be proportional to the relative lengths of the coastline (CP:CNP = 1:2).



The amount of poached abalone is estimated in terms of numbers and hence the following relation is used to compute the fishing “mortality”  $F_y^s$  for the poaching sector in year  $y$ :

$$N_y^s = \sum_{a=5}^z N_{y,a}^I (1-\rho) e^{-M/4} S_a^s F_y^s + N_{y,4}^I e^{-M/4} S_4^s F_y^s \quad (\text{A6.1.18})$$

Equations (A6.1.15) to (A6.1.18) assume that poaching and recreational activities occur exclusively in the inshore region. In the case of the commercial sector, the 0-2 m depth range is thought to be the only habitat that is almost never fished by commercial divers encroaching inshore because the shallow depth prevents boats from operating easily in these waters. Inshore encroachment by commercial divers is seen as being particularly common in areas that do not have residential houses along the beachfront. Thus, whereas this is thought to be a relatively minor problem in subarea CNP, inshore encroachment by commercial fishers is considered to have been a problem throughout the history of the fishery in subarea CP.

Thus, whereas the commercial catch by mass in year  $y$  in subarea CP is given by:

$$C_y^s = \sum_{a=8}^{z-1} w_{a+1/4} (N_{y,a}^I + N_{y,a}^O) e^{-M/4} S_a^s F_y^s + \left( w_{y,\bar{z}_y+1/4}^I N_{y,z}^I + w_{y,\bar{z}_y+1/4}^O N_{y,z}^O \right) e^{-M/4} S_z^s F_y^s \quad (\text{A6.1.19})$$

in subarea CNP, the commercial catch by mass in year  $y$  is given by equation (A6.1.19) above for years prior to 1967, and by equation (A6.1.20) for years 1967 onwards:

$$C_y^s = \sum_{a=8}^{z-1} w_{a+1/4} (N_{y,a}^O + \rho N_{y,a}^I) e^{-M/4} S_a^s F_y^s + \left( w_{y,\bar{z}_y+1/4}^O N_{y,z}^O + \rho w_{y,\bar{z}_y+1/4}^I N_{y,z}^I \right) e^{-M/4} S_z^s F_y^s \quad (\text{A6.1.20})$$

where  $w_{y,\bar{z}_y+1/4}^O$  is the mean mass of the offshore plus group with average age  $\bar{z}_y + 1/4$  after the first quarter of Model year  $y$ .

The exploitable (“available”) components of abundance for the recreational and poaching sectors are both expressed in terms of population numbers and are computed using Eqn. (A6.1.21) below for the recreational sector and Eqn. (A6.1.22) for the poaching sector:

$$B_y^{exp,s} = \sum_{a=8}^{z-1} S_a^s (1-\rho) N_{y,a}^I e^{-M/4} \quad (\text{A6.1.21})$$

$$B_y^{exp,s} = \sum_{a=5}^z S_a^s (1-\rho) N_{y,a}^I e^{-M/4} + S_4^s N_{y,4}^I e^{-M/4} \quad (\text{A6.1.22})$$

On the other hand, the exploitable components of abundance for the commercial sector operating in subareas CP (all years) and CNP (years prior to 1967) are computed as:

$$B_y^{exp,s} = \sum_{a=8}^{z-1} S_a^s w_{a+1/4} (N_{y,a}^I + N_{y,a}^O) e^{-M/4} + S_z^s \left( w_{y,\bar{z}_y+1/4}^I N_{y,z}^I + w_{y,\bar{z}_y+1/4}^O N_{y,z}^O \right) e^{-M/4} \quad (\text{A6.1.23})$$

and in the case of subarea CNP, exploitable biomass for years from 1967 onwards is computed as:

$$B_y^{exp,s} = \sum_{a=8}^{z-1} S_a^s w_{a+\frac{1}{4}} (N_{y,a}^O + \rho N_{y,a}^I) e^{-M_a/4} + S_z^s (w_{y,\bar{z}_y+\frac{1}{4}}^O N_{y,z}^O + \rho w_{y,\bar{z}_y+\frac{1}{4}}^I N_{y,z}^I) e^{-M_z/4} \quad (A6.1.24)$$

In the case of FLAS, which for these purposes can be considered as another fishery sector  $s$ , “available” population numbers are given by:

$$N_y^{exp,s} = \sum_{a=5}^z S_a^s \left( (1-\rho) N_{y,a}^I e^{-M_a/4} - C_{y,a}^I \right) e^{-M_a/4} \quad (A6.1.25)$$

The summation is from age  $a = 5$  as only animals larger than 100mm shell length are recorded so as to reduce uncertainty in the estimates due to the non-emergent/cryptic behaviour of juveniles. This corresponds to a minimum sampling age of approximately 5 years, so that for this sector  $S_a^s = 0$  for  $a < 5$ .

The proportion of the resource harvested each year ( $F_y^s$ ) by sector  $s$  is given by:

$$F_y^s = C_y^s / B_y^{exp,s} \quad (A6.1.26)$$

so that numbers-at-age removed each year by the poaching and recreational sectors can be computed from:

$$C_{y,a}^s = S_a^s F_y^s (1-\rho) N_{y,a}^I e^{-M_a/4} \quad \text{for } a \geq 5 \quad (A6.1.27)$$

and 
$$C_{y,a}^s = S_a^s F_y^s N_{y,a}^I e^{-M_a/4} \quad \text{for } a = 4 \text{ (poaching catches)} \quad (A6.1.28)$$

In the case of the commercial sector, the numbers-at-age removed each year from subarea CP is given by:

$$C_{y,a}^s = S_a^s F_y^s (N_{y,a}^I + N_{y,a}^O) e^{-M_a/4} \quad (A6.1.29)$$

The commercial numbers-at-age removed from subarea CNP for each of the years prior to 1967 is given by equation (A6.1.29) above, and then by equation (A6.1.30) below as from 1967:

$$C_{y,a}^s = S_a^s F_y^s (N_{y,a}^O + \rho \cdot N_{y,a}^I) e^{-M_a/4} \quad (A6.1.30)$$

The annual Zone C catch by the illegal sector  $C_y^{poach}$  is modelled by estimating the numbers poached in the year with the greatest level of poaching and then extrapolating this figure to other years using the poaching trend scenarios given in Table 4.4. Three poaching trend scenarios (i.t.o. numbers poached) are considered and are based on poaching confiscation data and a correction factor used to account for changes in policing efficiency (see Chapter 4). A separate model parameter  $p_{poach}$  sets a fixed ratio between the proportions of the resource poached each year from CNP and CP by relating to the exploitation rate for poachers rather than the catch by this sector (see equation 6.2).

### 6.1.2 Spawning biomass - recruitment relationship

The spawning biomass for each subarea in year  $y$  is given by:

$$B_y^{sp} = \sum_{a=1}^{z-1} f_a w_a (N_{y,a}^I + N_{y,a}^O) + f_z (w_{y,\bar{z}_y}^I N_{y,z}^I + w_{y,\bar{z}_y}^O N_{y,z}^O) \quad (\text{A6.1.31})$$

where  $f_a$  is the proportion of abalone of age  $a$  that are mature. Note that this formulation assumes independence of subareas in terms of recruitment, *viz.* the recruitment in one subarea depends only on the spawning biomass in that subarea and not on the biomass in adjoining subareas.

The number of recruits in each of the two subareas at the start of Model year  $y$  is related to the spawner stock size by a stock-recruitment relationship. A Beverton-Holt form (Beverton and Holt, 1957) is assumed, i.e. :

$$R(B_y^{sp}) = \frac{\alpha B_y^{sp}}{\beta + B_y^{sp}} \quad (\text{A6.1.32})$$

Note from equations (A6.1.1) and (A6.1.5) that the relative proportion of recruits settling inshore versus offshore in each subarea is determined by parameter  $r_I$ .

In order to work with estimable parameters that are more meaningful biologically, the stock-recruit relationship is re-parameterised in terms of the pre-exploitation equilibrium spawning biomass,  $B_0^{sp}$ , and the “steepness” of the stock-recruit relationship, where “steepness” is the fraction of pristine recruitment that results when spawning biomass drops to 20% of its pristine level, i.e.

$$hR_0 = R(0.2B_0^{sp}) \quad (\text{A6.1.33})$$

from which it follows that:

$$h = 0.2[\beta + B_0^{sp}] / [\beta + 0.2B_0^{sp}] \quad (\text{A6.1.34})$$

and hence:

$$\alpha = \frac{4hR_0}{5h-1} \quad (\text{A6.1.35})$$

and: 
$$\beta = \frac{B_0^{sp}(1-h)}{5h-1} \quad (\text{A6.1.36})$$

### 6.1.3 Starting values for biomass trajectories

The resource is assumed to be at the deterministic equilibrium (corresponding to an absence of harvesting) at the start of 1951, the initial year considered here. Given a value for the pre-exploitation spawning biomass  $B_0^{sp}$  of abalone, together with the assumption of an initial equilibrium age structure, it follows that on a subarea basis:

$$B_0^{sp} = R_0 \cdot \left[ \sum_{a=1}^{z-1} f_a w_a \exp\left(-\sum_{a'=0}^{a-1} M_{a'}\right) + f_z w_{0,\bar{z}_0} \frac{\exp\left(-\sum_{a'=0}^{z-1} M_{a'}\right)}{1 - \exp(-M_z)} \right] \quad (\text{A6.1.37})$$

which can be solved for  $R_0$ . Note that here  $w_{0,\bar{z}_0}$  means the equilibrium value of this quantity prior to exploitation, computed using the equilibrium plus group mean age  $\bar{z}_0$ , where:

$$\bar{z}_0 = z + \frac{e^{-M_{z-1}}}{1 - e^{-M_z}} \quad (\text{A6.1.38})$$

The initial **inshore** numbers at age for the projections, corresponding to the deterministic equilibrium, are:

$$\begin{aligned} N_{0,0}^I &= r_I R_0 \\ N_{0,a+1}^I &= N_{0,a}^I e^{-M_a} & 0 \leq a \leq 4 \\ N_{0,a+1}^I &= N_{0,a}^I (1-\rho) e^{-M_a} & 5 \leq a \leq z-2 \\ N_{0,z}^I &= \frac{N_{z-1}^I (1-\rho) e^{-M_{z-1}}}{1 - (1-\rho) e^{-M_z}} \end{aligned} \quad (\text{A6.1.39})$$

Similarly, the initial **offshore** numbers at age, corresponding to the deterministic equilibrium, are:

$$\begin{aligned}
 N_{0,0}^O &= (1-r_f)R_0 \\
 N_{0,a+1}^O &= N_{0,a}^O e^{-M_a} & 0 \leq a \leq 4 \\
 N_{0,a+1}^O &= N_{0,a}^O e^{-M_a} + N_{0,a}^I \rho e^{-M_a} & 5 \leq a \leq z-2 \\
 N_{0,z}^O &= \frac{N_{z-1}^O e^{-M_{z-1}} + \rho(N_{0,z}^I e^{-M_z} + N_{0,z-1}^I e^{-M_{z-1}})}{1 - e^{-M_z}} & a = z
 \end{aligned} \tag{A6.1.40}$$

It follows from the steady-state solutions to these equations that the inshore and offshore equilibrium plus group mean ages are as follows:

$$\begin{aligned}
 \bar{z}_0^I &= z + \frac{(1-\rho)e^{-M_{z-1}}}{1 - (1-\rho)e^{-M_z}} \\
 \bar{z}_0^O &= z + \frac{e^{-M_{z-1}}}{1 - e^{-M_z}} + \left( \frac{\rho e^{-M_{z-1}}}{(1 - e^{-M_z})(1 - (1-\rho)e^{-M_z})} \right) \cdot \frac{N_{0,z}^I}{N_{0,z}^O}
 \end{aligned} \tag{A6.1.41}$$

Numbers-at-age for subsequent years are then computed by means of equations (A6.1.1)-(A6.1.36).

#### 6.1.4 Parameter Values

##### **Input parameters:**

The following fixed parameter values are used in the model. The three von Bertalanffy parameters are from Tarr (1995) and the two mass-length relationship parameters were computed in this study:

$$l_\infty = 172.76 \text{ mm}$$

$$\kappa = 0.186 \text{ yr}^{-1}$$

$$t_0 = 0 \text{ yr (and is assumed to correspond to October because Tarr (1995) tagged animals *in situ* in October and November)}$$

$$c = 0.000098 \text{ gm/mm}^{3.155}$$

$$d = 3.1549$$

with the computations assuming a plus group at age  $z = 15$  yrs.

The proportion of abalone of age  $a$  that are mature is approximated by  $f_4 = 0.25, f_5 = 0.5, f_6 = 0.75$  and  $f_a = 1$  for  $a \geq 7$  (Tarr 1995).

Moreover, the base-case assumes that  $h = 0.7$ . The base-case value of the steepness parameter  $h$  corresponds roughly to the median ( $h = 0.74$ ) of a distribution of  $h$  values for stock-recruit functions fitted to the fisheries stock recruitment database developed by R.A. Myers and colleagues (Myers *et al.* 1995a), as advised by J. Ianelli (pers. commn).

***Estimable parameters:***

The sector-specific fishing selectivities  $S_a^s$  (including those for FIAS) are assumed to follow the functional form:

$$S_a^s = \frac{P \cdot e^{-\mu a}}{1 + e^{-\delta(a-\tilde{a})}} \quad (\text{A6.1.42})$$

where  $\mu$ ,  $\delta$  and  $\tilde{a}$  are three estimable parameters that control the shape of the function and  $P$  is simply a scalar fixed at a value such that  $S_{11}^s = 1.00$ . In essence,  $\mu$  controls the slope of the right hand limb of the function,  $\delta$  controls the steepness of the ascending left hand limb, and  $\tilde{a}$  shifts the function to the left or right, all in relation to age  $a$ .

The assumption that commercial selectivity parameters are the same for the inshore and offshore compartments might seem severe, given the greatly different age profiles of abalone in the inshore and offshore areas. Note however that only a small component of the commercial fishing takes place in the inshore region (the numbers of commercially exploitable size in that region being small), so that even if the assumption is in error, the impact on results should not be substantial.

Under the assumption that the sampling methodology is the same inshore and offshore, the same selectivity parameters are used for the inshore and offshore FIAS sectors. A separate selectivity function is used to compute model-predicted catch-at-age when fitting to the "old survey" data and it is again assumed that the same parameters apply to the inshore and offshore regions.

**6.1.5 The likelihood function**

The likelihood function which is maximised in the parameter estimation process is based on equations developed by Geromont and Butterworth (1999). The model is fitted to CPUE and FIAS abundance and catch-at-age data from all sectors (commercial, recreational, poaching, old survey, inshore FIAS and deep (offshore) FIAS) and the contributions by each of these to the negative of the log-likelihood ( $-\ln L$ ) calculated as described below.

**Abundance data:**

The likelihood contribution is calculated assuming that the observed abundance index is log-normally distributed about its expected value:

$$I_y^s = \hat{I}_y^s e^{\varepsilon_y^s} \quad \text{or} \quad \varepsilon_y^s = \ln(I_y^s) - \ln(\hat{I}_y^s) \quad (\text{A6.1.43})$$

where  $I_y^s$  is the abundance index for year  $y$  and sector  $s$ ,

$\hat{I}_y^s = q^s B_y^{exp,s}$  is the corresponding model estimated value, where  $B_y^{exp,s}$  is the model value for exploitable resource biomass corresponding to sector  $s$ , given by equations (A6.1.21- A6.1.24) (if the index refers to numbers,  $B_y^{exp,s}$  is replaced by  $N_y^{exp,s}$  - see equation (A6.1.25)).

$q^s$  is the constant of proportionality for abundance series corresponding to sector  $s$ , and

$$\varepsilon_y^s \text{ from } N(0, (\sigma_y^s)^2).$$

The contribution of the abundance data to the negative of the log-likelihood function (after removal of constants) is given then by:

$$-\ln L = \sum_s \left[ \sum_y \ln \sigma_y^s + (\varepsilon_y^s)^2 / 2(\sigma_y^s)^2 \right] \quad (\text{A6.1.44})$$

*Variance unspecified:* (CPUE abundance series)

In this case the standard deviation of the residuals for the logarithms of abundance series  $s$  is assumed to be independent of  $y$ , and is estimated in the fitting procedure by its maximum likelihood value:

$$\hat{\sigma}^s = \sqrt{\frac{1}{n_s} \sum_y (\ln I_y^s - \ln \hat{I}_y^s)^2} \quad (\text{A6.1.45})$$

where  $n_s$  is the number of data points for the abundance series corresponding to sector  $s$ .

The catchability coefficient  $q^s$  for sector  $s$ 's abundance index is estimated by its maximum likelihood value:

$$\ln \hat{q}^s = \frac{1}{n_s} \sum_y (\ln I_y^s - \ln \hat{B}_y^{exp,s}) \quad (\text{A6.1.46})$$

*Variance specified: (FIAS data)*

The catchability coefficient  $q^s$  for such a sector's abundance index is estimated by its maximum likelihood value which, for the case of a log-normal error distribution, is given by:

$$\ln \hat{q}^s = \frac{\sum_y 1/(\sigma_y^s)^2 (\ln I_y^s - \ln \hat{B}_y^{exp,s})}{\sum_y 1/(\sigma_y^s)^2} \quad (\text{A6.1.47})$$

where  $(\sigma_y^s)^2 = \ln(1 + (CV_y)^2)$  and the coefficient of variation ( $CV_y$ ) of the resource abundance estimate for year  $y$  is input.

**Catches-at-age:**

The likelihood contribution is calculated assuming a log-normal error distribution and by making an adjustment (suggested by A. Punt, pers. commn) to weight in relation to the observed proportions so that undue importance is not attached to poorly represented age classes:

$$-\ln L = \sum_s \sum_y \sum_a \left[ \ln \left( \sigma_c^s / \sqrt{p_{y,a}^s} \right) + p_{y,a}^s \left( \ln(\delta + p_{y,a}^s) - \ln(\delta + \hat{p}_{y,a}^s) \right)^2 / 2(\sigma_c^s)^2 \right] \quad (\text{A6.1.48})$$

where  $p_{y,a}^s = C_{y,a}^s / \sum_{a'} C_{y,a'}^s$  is the observed proportion of abalone caught/sampled by sector  $s$  in year  $y$  that are of age  $a$ ,

$\delta = 0.05$  is a constant included because not all of the  $p_{y,a}^s$  values are nonzero,

$\sigma_c^s$  is the standard deviation associated with the catch-at-age data for sector  $s$ , estimated in the fitting procedure by:

$$\sigma_c^s = \sqrt{\sum_y \sum_a p_{y,a}^s \left( \ln(\delta + p_{y,a}^s) - \ln(\delta + \hat{p}_{y,a}^s) \right)^2 / \sum_y \sum_a 1} \quad (\text{A6.1.49})$$



and  $\hat{p}_{y,a}^s = \hat{C}_{y,a}^s / \sum_{a'} \hat{C}_{y,a'}^s$ , is the model-predicted proportion of abalone caught/sampled by sector  $s$  in year  $y$  that are of age  $a$ .

For subarea CNP, the earliest catch-at-age data are from 1980 and hence correspond to the period during which all commercial catches are assumed taken from the offshore region, so that  $\hat{C}_{y,a}^s$  is given by:

$$\hat{C}_{y,a}^s = (N_{y,a}^O + \rho N_{y,a}^I) e^{-\frac{M_a}{4}} S_a^s F_y^s \quad (\text{A6.1.50})$$

whereas for subarea CP,  $\hat{C}_{y,a}^s$  is determined as follows:

$$\hat{C}_{y,a}^s = (N_{y,a}^I + N_{y,a}^O) e^{-\frac{M_a}{4}} S_a^s F_y^s \quad (\text{A6.1.51})$$

The model-predicted recreational catch-at-age data is based on abalone assumed caught from both the CNP and CP subareas, such that for this sector:

$$\hat{C}_{y,a}^s = ((1 - \rho_{CNP}) N_{y,a}^{I_{CNP}} + (1 - \rho_{CP}) N_{y,a}^{I_{CP}}) e^{-\frac{M_a}{4}} S_a^s F_y^s \quad (\text{A6.1.52})$$

except in the case of the single year's (1997) recreational catch-at-age data from subarea CP, for which  $\hat{C}_{y,a}^s$  is computed as:

$$\hat{C}_{y,a}^s = (1 - \rho) N_{y,a}^{I_{CP}} e^{-\frac{M_a}{4}} S_a^s F_y^s \quad (\text{A6.1.53})$$

The poached catch is taken primarily from the inshore region of subarea CP and hence Eqn. (A6.1.53) above is used to calculate  $\hat{C}_{y,a}^s$  for the poaching sector.

The FIAS, "old survey" and industry survey catches-at-age are similarly incorporated into the negative of the log-likelihood, except that comparisons with observed proportions are made at mid-year rather than after the first quarter of each Model year. Data from the inshore FIAS stations is assumed to correspond to the inshore model region whereas data from the deep FIAS stations is assumed to correspond to the offshore model region. The 0-5 m and 5-15 m "old survey" data are assumed to respectively correspond to the inshore and offshore model regions. Thus, for each subarea, the inshore FIAS and inshore "old survey" model-predicted numbers of abalone of age  $a$  sampled are computed as:

$$\begin{aligned} \hat{C}_{y,a}^s &= (N_{y,a}^I e^{-\frac{M_a}{4}} - C_{y,a}^I) e^{-\frac{M_a}{4}} S_a^s F_y^s & a < 5 \\ \hat{C}_{y,a}^s &= ((1 - \rho) N_{y,a}^I e^{-\frac{M_a}{4}} - C_{y,a}^I) e^{-\frac{M_a}{4}} S_a^s F_y^s & a \geq 5 \end{aligned} \quad (\text{A6.1.54})$$

and  $\hat{C}_{y,a}^s$  for the deep FIAS and offshore "old survey" are given by:

$$\begin{aligned}\hat{C}_{y,a}^s &= \left( N_{y,a}^O e^{-\frac{M_a}{4}} - C_{y,a}^O \right) e^{-\frac{M_a}{4}} S_a^s F_y^s & a < 5 \\ \hat{C}_{y,a}^s &= \left( \left( N_{y,a}^O + \rho N_{y,a}^I \right) e^{-\frac{M_a}{4}} - C_{y,a}^O \right) e^{-\frac{M_a}{4}} S_a^s F_y^s & a \geq 5\end{aligned}\tag{A6.1.55}$$

Data from the 2002 industry “total population size composition” survey are assumed representative of the entire Zone C area and hence  $\hat{C}_{y,a}^s$  for the industry survey is computed by summing over mid-year inshore and offshore regions for both CNP and CP.

The summation of the above quantities over age  $a$  is taken from the minimum age (or minus group where applicable) to the plus group for each catch-at-age series as summarised in Table 3.6. This was necessary because of small sample sizes outside these age ranges.

Inspection of the various  $-\ln L$  contributions has revealed that the catch-at-age  $-\ln L$  contributions are substantially larger than those for CPUE and the FIAS series, in part because they include many more data points as a result of summation over age as well as year. This is questionable as the  $p_{y,a}^s$  values for a given  $y$  and  $s$  are not likely to be independent of each other (as implicitly assumed by equation (A6.1.48)), because the cohort-slicing method used to provide the catch-at-age information from length composition data likely introduces positive correlation. The catch-at-age  $-\ln L$  contributions are thus downweighted by a multiplicative factor of 0.1, thereby downscaling these contributions to a similar order of magnitude as the CPUE and FIAS contributions.

### 6.1.6 Estimation of precision

Where quoted, 95% confidence interval estimates have been evaluated using the likelihood profile method because the automated likelihood profile routine in ADMB can misbehave when the minimisation has convergence difficulties.

**Appendix 6.2 – Incorporating the “ecosystem-change” effect****Method for modelling increased juvenile mortality**

1. The following formulation was used to model age-dependent natural mortality rates  $M_a$  :

$$M_a = \mu + \frac{\lambda}{a+1} \quad (\text{A6.2.1})$$

where parameter  $\mu$  was estimated in the model-fitting process and  $\lambda$  was either estimated or set equal to a constant (e.g. 0.2). Note as an example, if  $\mu = 0.14$  and  $\lambda = 0.2$ , then  $M_0 = 0.34$ ,  $M_1 = 0.24$ ,  $M_2 = 0.207$ , ....  $M_{13} = 0.154$ ,  $M_{14} = 0.153$  and  $M_{15+} = 0.153$ .

2. The number of new recruits to the population from 1994 onwards is no longer reduced to 10% of the 1993 level as in previous model versions, but is instead determined in the same way as for the earlier years, i.e. by using the Beverton – Holt stock-recruit function.
3. To model the rate and extent of the “recruitment failure” effect, two new parameters were introduced: a steepness of recruitment failure parameter  $\nu$  and a maximum increase in mortality parameter  $M_{max}$ . An exponential increase in the  $M_0$  mortality rate is assumed to have occurred as from year  $y$ , where different values of the starting year  $y$  were tried and the rate of increase in  $M_0$  is determined by parameter  $\nu$ .  $M_0$  is assumed to increase continuously up to a maximum value  $M_{max}$  and then remains constant at this value from years  $y_{M_{max}}$  forwards. For example, Combined B&C Model I in 2002 was as follows:  $\mu = 0.138$  (estimated),  $\lambda = 0.2$  (fixed), first year with increase  $M_0$  is 1990,  $\nu = 0.227$  (estimated) and  $M_{max} = 3.856$  (estimated).

As  $M$  values are more easily understandable when converted to survival rates  $S$  (= the proportion of that age-class surviving from one year to the next),  $M_0$  values will be discussed in terms of  $S_0$  instead. The above parameter values thus translate into a situation where currently only 2.1% of abalone recruits survive into the second year compared to 71% in the absence of this “recruitment failure” effect.

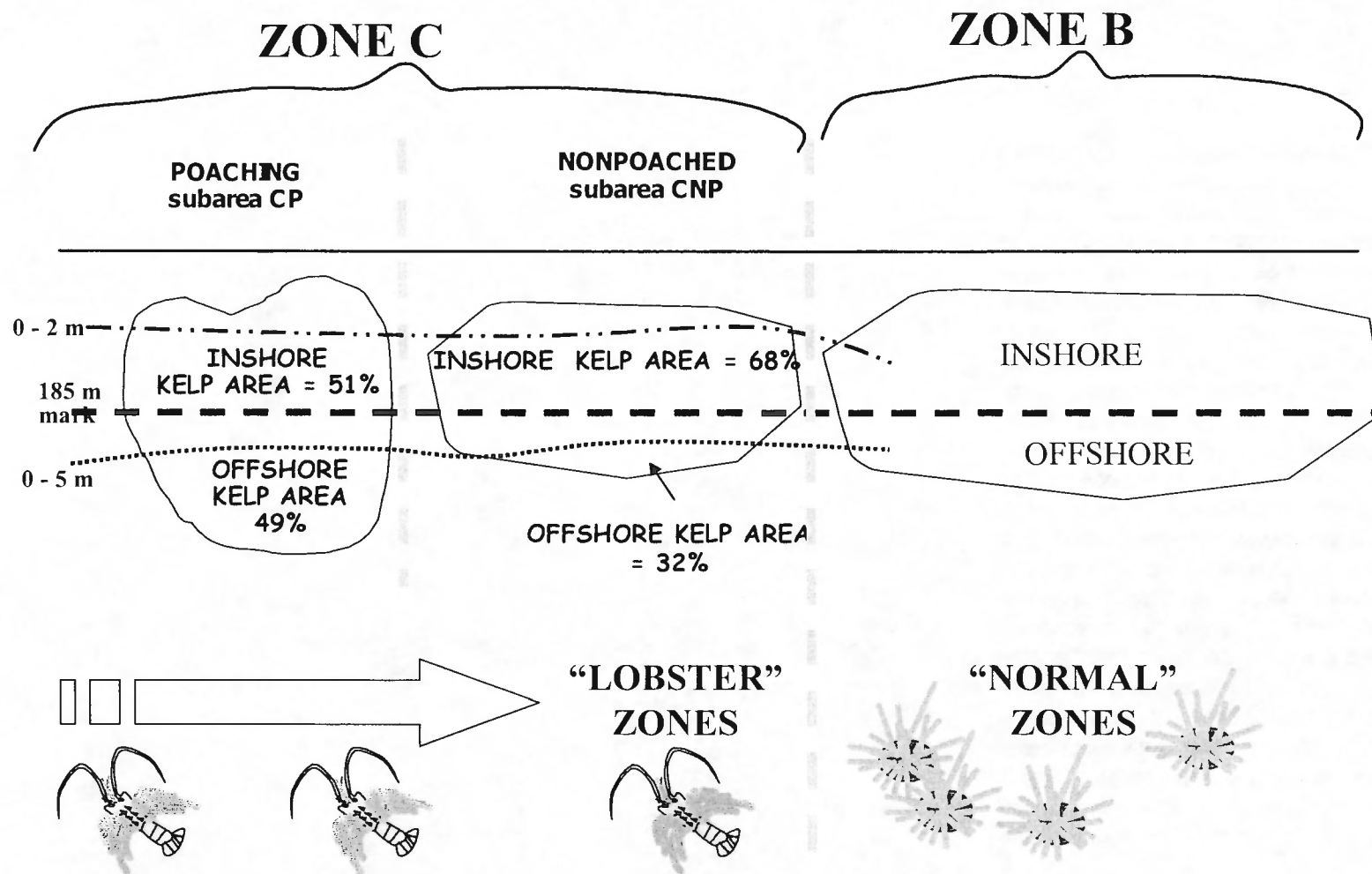


Fig. 6.1. Schematic summary of the structure of the combined Zones B&C abalone model. This model has subsequently been extended to a combined zones ABCD model by including “lobster”zone D to the west of Zone C and “normal”zone A to the east of Zone B, and simultaneously fitting to all these zones.

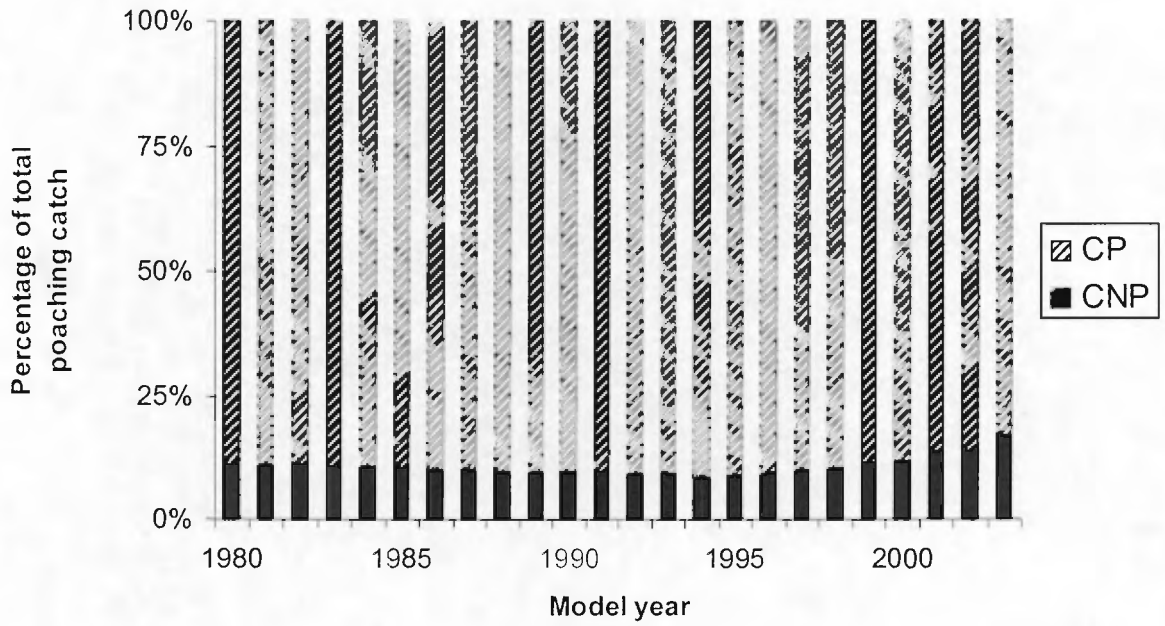


Fig. 6.2. Plot showing the percentage of the total Zone C poaching catch taken from subarea CP compared to subarea CNP when assuming that the split between the subareas relates to the exploitation rate for poachers rather than being set as a fixed percentage of the total poaching catch.

## Chapter 7

### Selected results from the spatial- and age-structured production model used to simultaneously represent abalone resource dynamics in Zones A, B, C and D

#### SUMMARY

A summary is presented of the results obtained from the 2003 base-case model that was fit to Zones A, B, CNP, CP and D in combination (hereafter referred to as the “combined ABCD model”). The full details of the spatial- and age-structured production model (ASPM) are provided in Chapter 6. The complete set of data input to the model are provided in Chapter 3, Chapter 4 (confiscation data and poaching trend assumptions) and Chapter 5 (GLM-standardised CPUE indices). A large number of different sensitivities and model formulations have been investigated but only a selected few key results are presented here in the interest of brevity.

Model results estimate a pristine spawning biomass,  $B_0^{sp}$  (in tonnes), of 8030 [95% C.I.: 7030 ; 12 800], 5870 [5450 ; 6300], 6740 [6160 ; 7700] and 7460 [6800 ; 8950] for Zones A, B, C and D respectively. The current spawning biomasses of abalone in the “nonpoached” CNP and “poached” CP areas of Zone C are estimated at ca. 35 % and 24 % respectively of their pre-exploitation levels, but the inshore region is particularly depleted, with, for example, a current estimate for CP of only 12 %. Natural mortality is reasonably estimated and in Zones C and D, the additional mortality estimated for 0-yr old abalone (due to the ecosystem-change effect) corresponds to near zero current annual survival rates. Poaching is severely impacting the resource with recent annual proportions of biomass extracted by the poaching sector alone topping 0.6 (Zone B), which is substantially larger than even the historic maximum commercial fishing proportion estimates of some 0.4. As a consequence of the recent explosion of poaching activities, the combined Zones A-D model-predicted 2003 poaching estimate of 933 MT (corresponding to the assumption that, on average, 36% of all poached abalone are confiscated) is more than seven times the legal 2003 commercial TAC for these zones. As a result, assuming a policy to maintain *status quo*, the recommended TAC for the 2003/2004 fishing season was some 24% less than the previous season’s TAC.

#### 7.1 INTRODUCTION

The development, extension and implementation of the abalone spatial- and age-structured production model (ASPM) has been a complex and ongoing process. A great many different model formulations and sensitivity tests have been conducted during this process, to the extent that it is not possible to report all of these results in this thesis. Instead, this Chapter focuses on the results from the most recent implementation of the model – that used to provide advice in August 2003 regarding recommended abalone TACs for the following

fishing season (October 2003 – September 2004). This was the first occasion that the model was simultaneously fitted to data for Zones/Subareas A, B, CNP, CP and D (hereafter referred to as the “combined ABCD model”) and hence attention is focused on these model results, notwithstanding that some of them are preliminary. Throughout this thesis, “reference case” refers to a convenient benchmark for comparison for a set of sensitivity tests, whereas “base-case” refers to the “best assessment”, taking account of all pertinent factors. A summary description of parameters and selected abbreviations referenced in the text and tables is given in Table 7.1.

As explained in Chapter 6, a major breakthrough was achieved in 2002 because the data obtained from two industry/MCM full population surveys conducted in Zones B and C enabled estimation of the extent of the “lobster” effect through fitting to Zones B (no recruitment failure effect) and C (with recruitment failure effect) simultaneously (hereafter referred to as the “combined B&C model”). The current assessment model, the “combined ABCD model” was developed based on the “combined B&C model” (the model version adopted by the Abalone Working Group [AWG] in 2002), and hence more detailed discussion of the results and relevant sensitivity tests pertaining to the combined B&C model are given as background information in Appendix 7.1. For a full set of model results and sensitivities covering the range of model versions considered during this study, the reader is referred to the AWG documents listed in Appendix A1 at the end of this thesis. As motivated in Chapter 6, of the four main fishing zones, Zone C has been treated as a base-case throughout the model development process and hence will be discussed in relatively greater detail in what follows. Some of the parameter estimates obtained from the Zone C model fit are fixed for use in the combined ABCD model.

## **7.2 BASE-CASE COMBINED ABCD MODEL SPECIFICATIONS**

The base-case combined ABCD model simultaneously describes the population dynamics in each of ten regions (A, B, CNP, CP and D, with inshore and offshore compartment for each zone).

Although prior attempts have been made to model the Zone A component of the abalone resource, parameter estimation has generally proved problematic because the CPUE data for this zone do not show much contrast. Specifically, model estimates of the pre-exploitation

spawning biomass  $B_0^{sp}$  (or  $K$ ) for this zone tend to get very large. The most pragmatic approach for handling this problem seems to be to fix as many parameters as possible and then to estimate either  $K$  or natural mortality  $M$ . The preferred approach adopted here is to estimate  $K$  and fix  $M$  as it seems reasonable to assume a common  $M$  for the abalone resource in Zones A-D. Following discussions by the AWG, "a reasonable estimate of  $K$ " was defined as a value of  $K < 15\,000$  MT.

Earlier model results indicated that the Zone A CPUE data show a slight increasing trend over the 1980's that was not reflected by the model. Another concern is that the two sets of 1980's depletion estimates for Zone A (obtained by comparing Zone A abalone densities to those in the Betty's Bay sanctuary) are substantially lower than the corresponding model estimates. The above suggest that the Zone A historic catches input to the model may be underestimates. This could be due both to under-reporting of catches during the highpoint of the commercial fishery in the 1960's as well as to the fact that there are no pre-1977 data detailing the proportions of the commercial catch taken from the different zones and hence these are assumed to be fixed over time to their average values over 1977 - 1981 (and the resultant proportion for Zone A may be too low). As a simple method to address this problem, an additional "catch multiplier" parameter  $C_{mult}$  was introduced (as agreed by the AWG) into the model to scale the assumed level of historic catches for Zone A for the period 1951 to 1976.

A summary of the main features of the base-case combined ABCD model is given below and summarised further in Table 7.1. Further motivation for this choice of model is summarised in Appendix 7.1.

- i) Fix the inshore recruitment proportion for all zones equal to 0.9.

These parameter values were generally close to 0.9 when estimated in model sensitivity tests, and this choice of parameter value is consistent with the observation (Tarr 1993) that most of the recruitment occurs in inshore waters. It seems sensible to fix this parameter to the same value for all zones, and this approach also overcomes the problem of correlation between estimates of this parameter and the inshore-offshore movement parameters.



- ii) Estimate all selectivity  $\mu$  parameters (i.e. don't fix  $\mu(\text{CS}) = 0$ ).  
 Setting  $\mu(\text{CS}) = 0$  corresponds to assuming flat selectivity for the older abalone in the commercial catch. This was considered a reasonable assumption in some model runs and hence was attempted as a method to reduce the correlation between estimates of the natural mortality parameter and the selectivity parameter  $\mu$ . However, after considerable debate in the AWG, it was agreed that the possibility of reduced commercial selectivity for older abalone (e.g. in the form of animals afforded a “cryptic” refuge at depth) could not be excluded, and hence it was decided to rather free this parameter in the base-case model version. Estimation of  $\mu(\text{CS})$  resulted in a significantly better model fit (log likelihood difference = 2.21 when comparing the base-case model fit to a model version setting  $\mu(\text{CS}) = 0$ ).
- iii) Estimate selectivity parameters by fitting simultaneously to Zones A, B, CNP, CP and D.  
 The same selectivity parameters were assumed for all zones, as the habitats are similar and there are no obvious operational differences of any of the fishery sectors between the zones.
- iv) Fix selectivity for the industry survey to 1 for all ages.  
 A uniform selectivity function was assumed because of the extractive sampling methodology used.
- v) Poaching levels constrained to be not less than twice the number of confiscations.  
 The best available estimates from policing and MCM inspectors (as advised to the AWG) are that the proportion of poached abalone that is actually confiscated is between 20-50%.
- vi) Use poaching trend scenario II (MCM option) (see Chapter 4).  
 This trend is based on the best available information.

- vii) For Zone A, set catch multiplier parameter  $C_{mult} = 1.5$ .  
 This *ad hoc* adjustment was deemed necessary to satisfactorily fit the Zone A data. The value was accepted by the AWG on the basis that it is not unrealistic (see discussion in text below). Inspection of the correlation matrix for model versions with  $C_{mult}$  estimated as a free parameter suggested that this parameter is highly positively correlated (correlation coefficient = 0.95 – higher than for any other parameter pair) with the corresponding pristine spawning biomass estimate for Zone A (see Table A7.3).
- viii) Downweight all catch-at-age data by a multiplicative factor of 0.1 in the negative log likelihood.  
 See Appendix 6.1.
- ix) Include “old” survey and 2002 industry survey catch-at-age data also downweighted by a multiplicative factor of 0.1.  
 The “old” survey and industry survey catch-at-age data are the most recent to have become available for inclusion in the model.
- x) Fix the inshore-offshore migration parameter for subarea CP equal to HALF that for CNP.  
 Subarea CP encompasses a large shallow area that extends further offshore than in other zones. The greater offshore relative to its longshore extent of this shallow “inshore” region in subarea CP means that it would take animals longer to move out of this “inshore” region into the deeper waters.
- xi) Fix the inshore-offshore migration parameter for Zones A, B, CNP and D equal to the same value.  
 Given the similar size and shape of these zones, from a biological point of view,  $\rho$  should be similar among zones.
- xii) Allow  $M$  to be age-dependent (see Equation A6.2.1).  
 $M$  is most likely age-dependent in a long-lived animal such as abalone. This is also a useful assumption for purposes of estimating the “ecosystem-change”

effect (which impacts juvenile mortality rates). See sensitivity analyses in Appendix 7.1.

- xiii) Estimate the rate/extent of decline in recruitment in Zones C and D (cf. Appendix 6.2).

These were estimated predominantly through having the industry/survey data available for Zones C and B, and these same values were assumed to apply to Zone D given that these two zones are adjacent to one another and subject to exactly the same “ecosystem-change” effect.

- xiv) Modify the likelihood function for the age-composition data by replacing the predicted proportions by the observed proportions when defining the variance of the proportions-at-age (see equation A6.48).

Change effected in response to a recommendation by the 2002 BENEFIT Workshop. The results of simulation-estimation trials have indicated that the use of estimated proportions leads to biases (A. Parma, Centro Nacional Patagónico, Argentina, pers. commn), hence the change to weighting by observed rather than predicted proportions.

- xv) Assume that  $p_{\text{poach}}$  (see Table 7.1) relates to the relative exploitation rate effected by poachers rather than the relative catches by this sector.

Change effected (see Appendix 6.1) in response to a recommendation by the 2002 BENEFIT Workshop on the basis that the difference in the poaching takes from subareas CNP and CP is attributable to differences in exploitation rates between the two subareas.

- xvi) Ignore the FIAS age-composition data for the deepwater strata.

These data were included in earlier model versions but are too sparse, with associated variances that are unacceptably high, and hence they have been excluded from the base-case analysis.

### 7.3 PARAMETERS

The base-case combined ABCD model estimates the following 30 parameters:

- 1) Pre-exploitation spawning biomass  $B_0^{sp}$  for A, B, CNP, CP and D [5 parameters]
- 2) Inshore-offshore migration parameter  $\rho$  (CP) [1 parameter]
- 3) Inshore-offshore migration parameter  $\rho$  (A, B, CNP, D) [1 parameter]
- 4) Poaching estimate for the year with the assumed highest level of poaching for the zone in question:  $CP_{max}$  estimated for A, B, C (combined) and D. However, estimates for A and B hit the constraint  $v$ ) above [4 parameters]
- 5)  $p_{poach}$  (cf. xv above) [1 parameter] – equates roughly to old assumption that 10% of the Zone C poaching take is from CNP
- 6)  $M_a : \mu$  ( $\lambda = 0.2$ ) where the formulation to model age-dependent mortality rates is 
$$M_a = \mu + \frac{\lambda}{a+1}$$
. Natural mortality parameters  $\mu$  and  $\lambda$  assumed common to all zones [1 parameter]
- 7) Two “recruitment failure” effect parameters common to CNP, CP and D: a steepness of recruitment failure parameter  $v$  and a maximum increase in natural mortality parameter  $M_{max}$  [2 parameters]
- 8) Three parameters ( $\mu$ ,  $\delta$  and  $\tilde{a}$ ) for each of five selectivity functions (assumed common to all zones) [15 parameters]

## 7.4 SENSITIVITY TESTS

A number of sensitivity tests were conducted. The following are the selected few sensitivity investigations that are reported in detail in section 7.6:

1. Sensitivity to fixing the selectivity parameters for Zones A and D to those estimated for Zones B and C, because the data for the latter pair are more informative (Model v. 2);
2. Sensitivity to estimating the industry survey selectivity parameters (Model v. 3);
3. Sensitivity to reducing the value of the parameter  $h$ , which determines the “steepness” of the stock-recruit relationship, from 0.7 to 0.6. This direction of change was chosen because it effectively reduces the fraction of pristine recruitment that results when spawning biomass drops, i.e. it increases the possibility of recruitment overfishing (Model v. 4); and

4. Sensitivity to changing the downweighting of the catch-at-age  $-\ln L$  contributions (as detailed in Chapter 6) from 0.1 to 0.5 (Model v. 5).

As outlined in Appendix 7.1, sensitivity to the three poaching scenarios developed (I-III – see Chapter 4) was investigated and found to not make much difference. An additional important aspect of sensitivity relates to the confiscation percentage constraint, and the summarised results obtained under the two scenarios (max. 25% or 50%) are presented in Table 7.6. This Table also gives results describing sensitivity to the assumption that the “ecosystem-change” effect is not as drastic as indicated by the best fit model estimates (see text under Projections section below).

Brief mention is made of the results of the following additional sensitivity tests:

1. Sensitivity to the assumption that mortality is age-dependent;
2. Sensitivity to the assumed linear relationship between CPUE and exploitable biomass;  
and
3. Sensitivity to the assumption that for subarea CNP, commercial catches post-1966 are assumed taken from the offshore region only.

## 7.5 RESULTS AND DISCUSSION

A large number of different runs were conducted but to assist in achieving focus here, only the results of a selected few deemed to be of particular importance are given in Tables 7.3, 7.4 and 7.6 and Figs 7.1-7.22. Model parameter estimates as well as log-likelihood contributions for the base-case combined ABCD model adopted by the AWG in 2003 are summarised in Table 7.3. The base-case model fit to the abundance indices and catch-at-age data are presented in Figs. 7.1 to 7.13. Where quoted (in square brackets after an estimate), 95% confidence interval estimates have been evaluated using the likelihood profile method.

### *7.5.1 Comparing observed values and corresponding model predictions*

The same general pattern of an increase in standardised commercial catch rates during the 1980s, followed by a decrease during the 1990s is evident in all zones (Figs. 7.1, 7.3). The observed decline in both catch rates and the fishery independent abundance index (Figs. 7.1 – 7.3) in recent years is particularly steep and it has been recognised for some time that

continued depletion of the resource is inevitable unless the combined catch by all sectors can be drastically reduced. Considering that total catches in the 1960s were substantially greater than those taken in the 1970s and 1980s (Fig. 7.16) (this hardly seems likely not to have been the case, even though the details of the zonal-partitioning of catches assumed above could be in error to some extent), it is not too surprising that the CPUE trend shows an increase towards the end of the 1980s (Fig. 7.1, 7.3). This is the obvious explanation for the observed increase in CPUE estimates during the 1980s despite increases in overall catch levels over this period (Tarr 1993). It is to be expected from a relatively slow-growing long-lived resource afforded a respite as high and unsustainable historic catch levels are reduced substantially to below the then current sustainable yields.

In Zone C for example, if one assumes that the CPUE trend is a reasonable index of stock abundance, this suggests that whereas the Zone C commercial take during at least some of the 1970s and 1980s was below sustainable yield (SY) levels, total catches during the 1990s and early 2000's have again exceeded such yields (Fig. 7.17), resulting in a concomitant decline in CPUE values over this most recent period (Fig. 7.1). This recent decline in CPUE is fully consistent with the catch data only if the latter are considered to include the considerable poaching component of the overall catches over recent years as estimated by the model. The estimated poaching catches are substantial in all of the scenarios investigated in this analysis (Tables 7.3, 7.4, 7.6). Moreover, fitting simultaneously to full population survey data for Zones B (with poaching but without ecosystem-change effect) and C (with both poaching and the ecosystem-change effect) has suggested dramatic increases in juvenile mortality rates in Zone C (and Zone D by extrapolation) that has further exacerbated the recent observed declines in both CPUE and FIAS in these regions. In particular, earlier modelling attempts demonstrated an inability of the model to successfully simulate the trends in the Zone C FIAS data without taking into account the “ecosystem-change” effect.

The model shows generally reasonable fits to all indices with the possible exception of the fit to the CPUE trend for Zone B over the most recent period (Fig. 7.3). In all model variations considered in this study, the best fits for Zone B were obtained when poaching catches were freed and estimated to be either very low or zero. Given that considerable poaching has taken place in this zone (as evident from the confiscation data), these “best fits” are clearly unrealistic and hence the poaching estimates in the model have been constrained by a pre-set lower limit. Viewed in the shorter recent term, the Zone B CPUE data does show an

appreciable drop, reflecting resource reduction and hence implying poaching. The reasons for the poor fit to the Zone B CPUE data are currently being investigated but no clear reasons have emerged at the time of writing this thesis. The work reported here is not definitive but rather forms part of an ongoing investigation.

Inspection of plots comparing model-predicted and observed (standardised) CPUE values (Figs. 7.1 and 7.4) suggests that the model is in particular failing to capture some of the large CPUE values observed for Zone D during the 1990's. This is not surprising because these CPUE values reflect the fact that the overall extent of Zone D has changed twice since 1977 due to the addition and removal of sectors of coastline (Dichmont *et al.* 2000). Thus whereas access to a large kelp bed to the west of this zone was prohibited with effect from the 1986 Model year, permission to fish a part of that area was again granted from 1993 (Dichmont *et al.* 2000), resulting in the unusually high CPUE value for this and the following few years (Fig. 7.1). The focus of the current study has been on Zones B and C, but it has been recognised for some time that to more accurately assess Zone D, future work needs to take account of issues such as this which are discussed in Chapter 10.

### 7.5.2 Additional diagnostics

The historic production data from the 1960's constitute a form of "CPUE" index and hence are compared to the combined (A+B+CNP+CP+D) commercially exploitable biomass trajectory (Fig. 7.5). They compare reasonably well suggesting that the model is adequately capturing the resource decline that took place during the 1960's.

An additional useful diagnostic is provided by the pattern and scale of the fishing proportion  $F_y^s$  (Fig. 7.6). Note that the fishing proportion underlying these analyses represents the fished proportion of a fully selected age class rather than the more familiar annual fishing mortality rate referred to in fisheries stock assessment literature. The  $F_y^s$  values in Fig. 7.6 have been plotted on approximately the same scale for ease of comparison, and suggest that historically Zones B and CP have been the most heavily fished. Subarea CP is known as a favourite abalone diving spot because of the extended shallow regions (see Chapter 6). As expected, Zone A appears to have been the least heavily fished of the zones presumably because it is the zone furthest away from the main town in the region, Hermanus (Dichmont *et al.* 2000). This is reinforced by the relatively low recreational fishing proportion evident in Zone A compared

to the substantial  $F_y^{rec}$  values for subarea CNP, which encompasses locations such as Hermanus and Vermont (Fig. 2.1). These areas are favoured holiday locations and hence it is to be expected that  $F_y^{rec}$  values are higher in these regions. In response to an ever-growing recreational catch during the 1990's, increasingly stringent limitations were imposed on recreational fishers (R. Tarr, MCM, pers. commn) and hence the contribution of the recreational sector to the total catch has gradually decreased and is currently zero by legislation (Fig. 7.6).

Comparisons of time-trajectories of exploitation proportion by sector predicted by the model suggest that efforts to control the illegal poaching sector have largely been fruitless as  $F_y^{poa}$  has either increased or remained steady in all zones during the last few years (Fig. 7.6). In Zones A, B and D, the total fishing proportions as estimated by the model for recent years have exceeded even the initially high  $F$  values corresponding to the initial "mining out" of the abalone resource. Thus whereas average commercial sector  $F$  values range between 0.05 and 0.16 (as could be considered reasonable for a fairly productive long-lived resource), recent estimates of  $F_y^{poa}$  are 0.2 or higher, with a maximum of 0.64 in Zone B in 2002 (Fig. 7.6), which is clearly not sustainable in the medium to long term. Note however that the  $F$  values for the different sectors are not precisely comparable – although they all refer to a common age 11 (see Appendix 6.1), they spread differently over age-classes and apply differentially to inshore and offshore areas (see Fig. 7.14).

Although there are no data available to directly validate the split of the commercial catches between the inshore and offshore areas, it has been suggested that this could nonetheless serve as a coarse diagnostic of possible unreasonable model behaviour (BENEFIT 2002). Model predictions seem reasonable for all zones (although results are only shown here for two zones), with a general pattern evident of a slight increasing trend in the proportion of the total commercial catch taken from the offshore region during the 1970's and 1980's and a marked increase in this proportion in recent years (Fig. 7.7). As inshore areas have become progressively more denuded due to poachers (all zones) and the ecosystem-change effect (in Zones C and D), it seems reasonable to expect commercial fishers to increasingly obtain heir catches from the offshore regions.



Additional diagnostics checked for the abalone model include, for example, the mean mass of the inshore and offshore plus-groups, as well as the mean mass of abalone in the commercial catches. These diagnostics are not reported here as they were all considered reasonable and are discussed in greater detail in some of the working group documents listed at the end of the thesis.

### 7.5.3 Catch-at-age comparisons

To assist in identifying potential yearly patterns in the catch-at-age residuals, the standardized residuals ( $\varepsilon_{y,a} \rightarrow \frac{(\ln p_{y,a}^i - \ln \hat{p}_{y,a}^i)}{\sigma^i / \sqrt{\hat{p}_{y,a}^i}}$ ) have been plotted (Figs. 7.8 – 7.11). Some indications of

systematic effects in the residuals are evident. For example, the model systematically predicts too many age 14-15 abalone caught during the 1980's (subarea CNP) and mid-1980's to mid-1990's (subarea CP) (Fig. 7.8) and too many age 14 abalone corresponding to the commercial catch-at-age data for Zones A (Fig. 7.9), B (Fig. 7.10) and D (Fig. 7.11). This may reflect errors with the cohort slicing or that the model overestimates the number of older abalone. The fits to the Zone A catch-at-age data corresponding to the FIAS sector are particularly poor, with the model consistently tending to over-estimate the proportion of animals in the larger age classes and *vice versa* for the smaller age classes.

The preponderance of positive residuals (i.e. the catches exceed model predictions) for the 12+ age class corresponding to the poached sector in Zone D (Fig. 7.11) differs somewhat from the patterns evident for the other zones and suggests possibly relatively greater selectivity for the older age classes by poachers operating in this zone. This is an interesting result because it can be ascribed to some of the Zone D confiscated samples having originated from Betty's Bay, the reserve area situated within Zone D, which has a relatively greater proportion of mature abalone than other areas (due to it having functioned as a reserve until relatively recently).

In general, the patterns of residuals do not indicate any very obvious model-misspecification. However, the selectivity functions may warrant some further exploration to see whether it is possible to improve the residuals for the fits to the proportions-at-age data to reflect better randomness and homoscedasticity. Although the poaching sector is thought to have possibly changed its mode of fishing during recent years by moving into deeper waters (A.

MacKenzie, MCM, pers. commn), there are no obvious indications from the residuals of any changes in selectivity over time for the poaching or any of the other sectors.

Comparisons between observed and model-predicted catch-at-age proportions corresponding to the “old” 1980’s survey data (Fig. 7.12) and 2002 Industry/MCM survey (Fig. 7.13) are particularly instructive because they cover the full range of age classes. The fits obtained using the base-case combined ABCD model are reasonably good and represent substantial improvements on earlier attempts at fitting these data (cf. Figs A7.2 – A7.3).

#### 7.5.4 Parameter estimates

Model results suggest a pristine spawning biomass,  $B_0^{sp}$ , of 2012 [95% C.I.: 1780 ; 2400] and 4724 [4380 ; 5300] tonnes respectively for subareas CNP and CP, and hence a total Zone C spawning biomass of ca. 6740 tonnes. The difference in the pristine spawning biomass estimates  $B_0^{sp}[CNP]$  and  $B_0^{sp}[CP]$  are in the main due to the partitioning of the historic zone C catch data between the two subareas. The pristine spawning biomass estimates for the other zones are on a similar scale to the Zone C estimates, with 8030 [7030 ; 12 800], 5870 [5450 ; 6300] and 7460 [6800 ; 8950] tonnes estimated for Zones A, B and D respectively (Table 7.3).

The 2002 BENEFIT review panel (BENEFIT 2002) queried whether earlier attempts to apply an ASPM model to Zone A represented any improvement on setting *TACs* using a replacement yield approach, given that the data are clearly inadequate to estimate the values for the model parameters directly. In response, it was noted that a particular advantage of the ASPM modelling compared to a replacement yield approach is that the former can provide some information on current depletion levels (and hence provide additional information as to the level of risk associated with *TAC* recommendations).

It is encouraging that a reasonable estimate of the pre-exploitation spawning biomass for Zone A was obtained as previous attempts (fitting a Zone A model in isolation) to estimate this parameter yielded unrealistically high values due to the uninformative nature of the data. The confidence interval for the Zone A pristine biomass estimate is much wider than for the other zones. The Zone A data provide a classical example of a so-called “one-way-trip-trajectory” (Hilborn and Walters 1992) – the CPUE data could derive from either a relatively

unproductive population with a high pristine biomass or from a more productive population with a lower pristine biomass. One approach to refine these estimates would be to incorporate prior information on these parameters ( $K$  and  $M$ ) to reduce the space of likely parameter combinations. The approach adopted here is analogous to some extent because the natural mortality parameter for Zone A is constrained to be the same as for the other zones (considered reasonable on biological grounds), thereby restricting the range of likely pristine biomass values. Thus although the Zone A fit is still viewed with some scepticism, it at least represents a step forward.

The broad similarities in the catchability coefficient estimates  $\hat{q}^{CPUE}$  (Table 7.3) for the Zones A, B, CP and D are as expected given their approximately equal habitat areas and the similarities in the standardised catch rates over much of the 1980s (Fig. 7.1, 7.3). The reason for the much higher  $\hat{q}^{CPUE}$  estimate for subarea CNP is presumably because of the assumption that commercial fishing was confined to the offshore model region only (in contrast to inshore and offshore regions as for the other zones) with effect from 1966 (see Chapter 6).

The base-case selectivity estimates are illustrated in Fig. 7.14. The estimated commercial and recreational selectivity functions reflect the fact that the minimum legal size corresponds to an age of approximately 9 years, whereas the estimated poaching selectivity function reflects the fact that sub-legal-size animals are caught. Because the FIAS transects are situated inshore, the estimated FIAS selectivity function (Fig. 7.14) concurs with the observation of Tarr (1993) that the mean size of *H. midae* increases with depth.

#### **7.5.5 Mortality estimates and the “ecosystem-change” effect**

A wide range of estimates of natural mortality ( $M$ ) for abalone in the wild have been reported in the literature, ranging from about 0.05 to more than 1.00 yr<sup>-1</sup> (Shepherd and Breen 1992, McShane and Naylor 1997). There is some (weak) evidence that  $M$  decreases with increasing age (Shepherd and Breen 1992). The base-case estimate of (age-dependent)  $M$  as ranging from 0.33 for 0-yr old abalone (when not subject to the “ecosystem-change” effect) to 0.14 for age 11 and older individuals is considered reasonable for a relatively long-lived species such as this. The likelihood profile estimate of the 95% CI for the mortality parameter  $\mu$  (with

estimate 0.127) is relatively tight [0.126 ; 0.142], suggesting that natural mortality is well estimated in the model fit.

In earlier model versions, estimates of mortality rates were confounded because, given the limited data available, it was not possible to discriminate between background mortality rates, mortality due to unknown levels of illegal take by the poaching sector and increased juvenile mortality due to the “lobster” effect in Zones C and D (see Chapter 6 for further explanation). This has been resolved to some extent in the current model by fitting to the 2002 industry/MCM full population surveys conducted in Zones B and C. By simultaneously fitting to Zones B (no recruitment failure effect) and C (with recruitment failure effect), it has been possible to estimate the extent of the “lobster” effect.

For “lobster” Zones C and D, the model parameter  $M_{max}$  sets the maximum mortality rate of 0-yr old abalone that is assumed to have occurred due to the ecosystem-change effect. The base-case model estimate of this parameter (Table 7.3) tended to hit whatever maximum constraint was set for the parameter (the base-case maximum constraint was 10 which corresponds to a near zero annual survival rate of 0-yr old abalone in Zones C and D). Based on the likelihood profile method, the lower 95% confidence limit for  $M_{max}$  is approximately  $M_{max} = 1.6$ , which implies an annual 0-yr old survival rate of 0.2, compared to the pre-1990 0-yr survival rate of 0.72. Given that the value of this parameter has a critical impact on projections, the AWG adopted the additional use of an alternative somewhat less pessimistic value (of  $M_{max} = 2$ ) for use in a sensitivity test in projections (see Section 7.7).

### 7.5.6 Poaching estimates

The estimates of  $CP_{max}$  (in terms of numbers) are given also in terms of mass (Table 7.3). It should be noted that the composition (in terms of age distribution) of “poached” catches in the model varies among zones such that the numbers-to-mass conversion is not the same across zones. The Zone C poaching estimate (for 1995) furnished by the base-case model is 556 000 [95% CI: 510 000 ; 695 000] abalone or 319 [267 ; 360] tonnes (Table 7.3), which is some three times greater than the average Zone C commercial take during the 1990s and 20% greater than the current commercial TAC for Zones A-D combined. Whereas towards the end of the 1990s poaching estimates were equal to or slightly less than the commercial TAC, the recent explosion of poaching activities has resulted in a total catch for Zones A-D combined

which is more than seven times the legal 2003 commercial TAC for these zones. These estimates are phenomenally high but remain quite plausible considering that they correspond to the assumption that, on average, 36% of all poached abalone are confiscated (Table 7.3). If the confiscation success rate is less than this, these estimates may be under-estimates. The highest current poaching estimate is for Zone B (810 tonnes) but the base-case model hit the constraint that the number of abalone poached should be at least twice the corresponding number of confiscated animals – the AWG considered it highly unlikely that law enforcement officers were able to successfully confiscate more than half of all abalone poached.

One potential problem with the poaching confiscation data (see Chapter 4) is that in recent years very large numbers of confiscated abalone have been assigned to the “unknown zone” category, and these abalone were then divided among the various zones based on the proportion of “zone-known” confiscations per zone. Because of the possibility that some of these “zone-unknown” confiscations were ascribed to the wrong zone or were actually from a region other than Zones A-D, the 50% confiscation proportion constraint is considered the more realistic. It is important to note that the “50% confiscation proportion” scenario is unfortunately a misnomer (which will be rectified in future) because the base-case model assumes on average, a confiscation percentage success rate over the past 5 years in Zones A-D of 20% (A: 27%; B: 27%; C: 18%; D: 7%) (Table 7.3). This is similar to the confiscation percentage success rate estimated by policing operations (Marcel Kroese, MCM, pers. comm) and to estimates from an attempt to estimate compliance confiscation success rates from data from the NGO Traffic Hong Kong office (Mackenzie and Burgener 2004).

### **7.5.7 Biomass trajectories**

Fig. 7.15 illustrates the differences between the predicted inshore and offshore spawning biomass trajectories per zone whereas Fig. 7.16 shows spawning and exploitable biomass trajectories relative to total catches in each zone. Total catches include recorded or model-estimated takes by the commercial, recreational and poaching sectors combined. Note that the “exploitable” biomass trajectories shown are those corresponding to the resource component harvested by the commercial sector.

The model estimates an initial steep decline in the spawning biomass of abalone in all zones (Figs. 7.15 - 7.16), as a result of the high historic exploitation levels in the 1960s. A slight

recovery of the stock level is estimated to have occurred during the 1980s, followed by a relatively stable period and then an appreciable downward trend in recent years (Fig. 7.16). Consistent with the incidence of poaching having commenced earliest (around 1994) in Zones C and D (also subject to the ecosystem-change effect), the recent downward trends in these zones are particularly marked. Given that the CPUE index effectively relates only to animals of ages nine and older, there is a time lag before the negative effects of poaching become evident. Inspection of the trend in the inshore region (which determines the numbers available to the inshore FLAS survey) (Fig. 7.15) highlights more clearly the substantial recent decreases in the numbers of small abalone in all zones. Given the long-lived nature of abalone, these decreases are particularly evident when projecting forward in time as shown in the next section.

Based on the results of the base-case model, the current spawning biomasses of abalone in the “nonpoached” CNP and “poached” CP areas of Zone C are estimated at ca. 35 % and 24 % (Table 7.3) respectively of their pre-exploitation levels. The inshore region is particularly depleted, with a current estimate for CP of only 12 % (Table 7.3, Fig. 7.15). Zones B and D are estimated to have similar current levels of depletion (Table 7.3), with the current depletion for Zone A (59 %) being estimated as somewhat higher. It is to be expected that abundance in Zone A should be slightly less depleted than the other zones as it is the least accessible (in terms of being furthest from human settlements) of the main fishing zones and poaching is thought to have commenced later in this zone than in the other zones. Nonetheless, the depletion estimates for Zone A need to be seen as preliminary given the problems in fitting the model to the data for this zone. Moreover, there is reasonably good agreement between the 1980’s “observed” and model-predicted depletion estimates for all zones except Zone A (Table 7.3), for which the predicted values are markedly higher than the “observed” values. This suggests that the model may not be accurately reflecting the population level relative to pristine during the 1980’s and hence that the current depletion estimates may be inflated. The “observed” 1980’s depletion values are useful as an additional diagnostic, but not too much confidence can be attached to them because they are based on the assumption that the abalone densities in the Betty’s Bay Reserve during the 1980’s are representative of the pristine abalone densities that existed in Zone A during the 1950’s, and it is unclear how reliable such an assumption may be.

### 7.5.8 Reference point considerations

The estimated maximum sustainable yield  $MSY$  for commercial selectivity and when considering inshore and offshore regions combined for abalone in Zone B is 474 tonnes per annum with an associated  $MSYL^{sp}$  (maximum sustainable yield level – the spawning biomass level relative to pristine that yields the  $MSY$ ) of 0.41. However, if sub-legal animals are caught, sustainable yields will be less than this. The estimated  $MSY$  for poaching selectivity (i.e. zero commercial and recreational catches) for abalone in Zone B is 318 tonnes per annum – approximately one-third less. Estimates of  $F_{MSY}$  are substantially less when assuming poaching selectivity (0.20) than when assuming commercial selectivity (0.64). Estimates for the other zones are similar. For example, for subarea CP, the  $MSY$  for commercial selectivity is 381 tonnes per annum. As expected, estimates of  $MSY$  are sensitive to the choice of the steepness parameter  $h$ , with, for example, the corresponding estimate of  $MSY$  (commercial selectivity) for Zone B decreasing to 432 MT.yr<sup>-1</sup> when  $h = 0.6$ , rather than the base-case assumed value of 0.7.

Under poaching selectivity, comparison of the best estimate of  $MSY$  with total catch estimates in subarea CP and Zone B (Fig. 7.17), suggests that catches probably exceeded replacement yields from about 1995 and 2000 respectively, resulting in a decrease in stock biomass. Model results suggest that the abalone stock remained at a level initially close to then increasing slightly above the  $MSYL^{sp}$  throughout the 1970's and 1980's, but has fallen below  $MSYL^{sp}$  during the last few years (Fig. 7.17), such that, for example, the current spawning biomass in subarea CP is estimated to be about half the  $MSYL^{sp}$ , with substantial further decreases predicted in model projections (see section 7.7).

One fairly general target reference point consideration which has been used by the AWG seeks to ensure depletion levels do not fall below 0.4  $K$  but it is recognised that there are a number of reasons why reference point targets related to historic  $MSYL$  are problematic when considering management advice for the abalone stock. Most notably, the “ecosystem-change effect” in Zones C and D means that more recent carrying capacity estimates for these zones may need to be revised downwards. The “historic”  $MSY$  and  $MSYL$  estimates discussed above are thus likely no longer realistic for these zones.

## 7.6 SENSITIVITY TESTS

Sensitivity to input values for model parameters, assumptions and model structure have been examined throughout the model development process, and changes and improvements made accordingly. However, it has not always been possible to rely on strict model selection criteria such as Likelihood Ratio Tests (*LRT*) and Akaike Information Criterion scores (*AIC*) (Burnham and Anderson 2002) because of the following:

- i) Constraints are imposed on certain parameters such as the model poaching estimates. This was necessary because in some cases the best fit model estimates were clearly unrealistic. For example, in cases where the model estimate of the number of abalone poached from a zone was less than the known number of illegal abalone catches confiscated for that zone.
- ii) The best fit estimate (using the base-case combined ABCD model) of the historic catch multiplier parameter  $C_{mult}$  (Zone A) was 3.78 but this value was deemed unrealistically high by the AWG. This is because part of the motivation for introducing Zonal TAC's in 1986/7 was to better balance effort across the fishing grounds by, for example, setting the TAC in Zone A higher than the historic commercial catch from that part of the coast (Dichmont *et al.* 2000). There were compensating decreases in Zone B and the extremely popular Zone D (Dichmont *et al.* 2000), suggesting that historic catches in Zone A are not likely to have been as high as indicated by the model estimate. Thus even though freeing the  $C_{mult}$  parameter resulted in a significantly improved fit to the model overall, and to the Zone A CPUE data in particular, this parameter was fixed at what was considered a more realistic value of 1.5 in the base-case model version. This aspect will be examined further in future work.
- iii) The choice of the most appropriate model version for use as a reference case or base-case model in performing projections was determined to some extent through a process of negotiation and compromise by the stake-holders attending the meetings of the AWG. Thus, for example, the choice of Poaching Scenario II was essentially motivated on the basis that the AWG considered it to be based on the best interpretation of available information rather than being chosen on the grounds of formal model selection criteria.



The most recent model selection debates pertaining to the base-case combined ABCD model have concerned the most appropriate way in which to model the selectivity functions, given that the base-case model assumes a common selectivity pattern per fishery sector for each of the five zones A, B, CNP, CP and D. In terms of the AIC model selection criterion, the base-case model (which simultaneously estimates common selectivity parameters for all zones) is markedly preferred to the model version with selectivities for Zones A and D fixed to the values fitted for Zones B and C. Estimation of a selectivity pattern for the Industry/MCM full population survey did not improve the model (in terms of AIC scores) compared to the base-case which assumed uniform selectivity (Table 7.4).

The value of the steepness parameter is important, as the overall potential yield for an ASPM depends primarily on the steepness of the stock-recruitment curve and on the natural mortality rate. However, reducing the steepness parameter (scenario (4)), which would reflect a less productive resource, resulted in a significantly worse fit ( $-\ln L = -313.216$  vs  $-\ln L = -319.492$  with  $h = 0.7$ ) – in particular, the fit to the Zone B CPUE data deteriorated (Table 7.4). The base-case value of  $h$  was close to the best fit value obtained and hence it was considered important to test sensitivity to reducing  $h$  but not necessarily to increasing  $h$ , given that the former approach is the more precautionary.

Changing the downweighting of the catch-at-age contributions to the overall likelihood from 0.1 to 0.5 (scenario (5)) resulted in some trade-offs between the quality of fits to the CPUE data and age data, with worse fits to the CPUE data (i.e. an increase in the log-likelihood and  $\hat{\sigma}$  values) resulting for subarea CNP, Zone D and (to a lesser extent) Zone A. The main conflict in the model fits is between the fit to the Zone D CPUE data and the catch-at-age data for the poaching sector. The reason for the CPUE(Zone D) and poaching sector age data pushing in different directions is most likely explained by the biased nature of the poaching catch-at-age data for Zone D (see discussion under 7.5.3). Another effect of upweighting the likelihood contribution corresponding to the catch-at-age data was to increase the pre-exploitation spawning biomass estimate for Zone D (Table 7.4). However, the associated decrease in the maximum predicted confiscation percentage from 0.11 to what seems a less realistic value of 0.06 suggests that resource abundance is over-estimated under this assumption.

In contrast, the upweighting scenario yields a slightly lower estimate (7220 MT compared to base-case value of 8030 MT) of pre-exploitation spawning biomass in Zone A. Previous analyses produced a similar result when upweighting the likelihood contribution corresponding to the catch-at-age data. It may thus be worth pursuing this further in future work, but only once the Zone D poaching catch-at-age data have been re-examined for potential biases.

Other than the aspects mentioned above, none of the sensitivity scenarios examined resulted in dramatically different estimates of basic model parameters such as natural mortality. In terms of depletion statistics, estimates of the ratios  $B^{sp}(2003)/B_0^{sp}(1951)$  for Zone A varied the most across the various model scenarios (from 0.50 to 0.61).

The consequences were explored of assuming CPUE proportional to the square root of biomass, which implicitly incorporates CPUE saturation effect possibilities (e.g. see Dichmont *et al.* 2000) but no satisfactory model fits resulted. Sensitivity to the assumption that mortality is age-dependent and to the assumption that for subarea CNP, commercial catches post-1966 are taken from the offshore region only, both resulted in appreciably worse fits and hence no further results for these scenarios are given in this document.

## 7.7 PROJECTIONS

Selected projection results from fitting the abalone spatial- and age-structured production model (ASPM) to Zones/Subareas A, B, CNP, CP and D in combination and projecting forwards under six future catch scenarios as specified by the AWG in 2003 are summarised in Table 7.6. The projection scenarios shown in Table 7.5 were chosen by the AWG for illustrative purposes so as to broadly include a fairly wide range of possible future catch scenarios. Five- and twenty-year replacement yields (the annual catch per zone that keeps the spawning biomass approximately at the current level at the end of the period considered) were also computed and other future catch scenarios considered, but these are not presented in detail here in the interests of brevity. However, some further projection results are given in Chapter 10 and the final TAC recommendations made in August 2003 are summarised in the next section. Note that although abalone are fairly long-lived animals and hence it is important to consider projections over the medium- to long-term, the projections shown here are for five years only because of an AWG decision to this effect. Reasons for focusing on

the short-term in this instance were mainly because it was argued that there is considerable uncertainty regarding both future poaching levels and the future consequences of the ecosystem-change effect. Selected longer-term projections are shown in Chapters 8-10.

Projection results are given for the following four model versions:

- I. Base-case combined ABCD model, which has a max. 50% confiscation level constraint and no constraint on the maximum mortality rate of 0-yr old abalone following the onset of the “recruitment failure” effect in Zones C and D (model parameter  $M_{max}$ );
- II. As for I but with a max. 25% confiscation level constraint;
- III. Higher recruitment scenario that is same as I. above (i.e. 50% confiscation level constraint) except that the maximum permitted value of  $M_{max}$  is set as 2 (this is equivalent to assuming a current annual survival rate of about 14% for 0-yr old abalone in Zones C and D – this is about 20% of the pre-1990 (and Zone A and B) 0-yr old abalone survival rate);
- IV. Higher recruitment scenario identical to III. above but includes a 25% confiscation level constraint.

Summaries of projection specifications and results are given in Tables 7.5 and 7.6 respectively and in Figs 7.18 – 7.22. For Zones A and B, there is a clear separation between model projections assuming *status quo* compared to reduced future poaching catches, with the former assumption resulting in a further reductions (relative to current levels) in spawning biomass under almost all scenarios considered (Fig. 7.18 – 7.19). In contrast, for Zones C and D, the model results were less sensitive to assumptions regarding future poaching levels because the recruitment declines in these regions are such that even under a scenario of zero take from these zones, the resource is predicted to show a steep decline (Figs. 7.20 – 7.22).

In all cases, imposing a maximum 25% confiscation level constraint (i.e. increasing the minimum permitted poaching estimate furnished by the model) increased the model estimate of the pre-exploitation biomass  $B_0^{sp}$  (Table 7.6). This was particularly noticeable for Zone B where the model attempted to account for annual poaching estimates as high as 1700 MT by increasing the estimates of both  $B_0^{sp}$  and natural mortality  $M$  (Table 7.6). The Zone D

estimates were the least affected because the model estimates of poaching for this zone corresponded to confiscation percentages of approximately 10%.

Imposing an upper bound on the estimate of  $M_{max}$  did not result in an appreciably different fit under the 50% confiscation proportion scenarios, but did result in a better fit in the case of the 25% confiscation proportion scenarios (Table 7.6). For a particular confiscation proportion scenario, changing the  $M_{max}$  value produced almost no differences in the model fit and parameter estimates, but had a noticeable effect on model projections with the lower  $M_{max}$  value translating into slightly less pessimistic projection results (see Figs. 7.20 – 7.22 for Zones C and D).

## 7.8 TAC RECOMMENDATIONS

Due to uncertainty regarding the future direction to be taken by the new Abalone Policy which was proposed at approximately the same time (August 2003) as scientific advice concerning TACs for the 2003/4 season had to be given, two scenarios were offered in August 2003:

Scenario 1 was based on the same rationale as used previously. It assumed a policy *status quo*, with existing right holders remaining in the fishery, and in certain zones allowed for further depletion of the resource in the interest of maintaining existing commercial structures. The objective was to phase in the severity of negative economic impacts on the existing fishery participants.

Scenario 2 was based on the assumption that 10-year rights would be issued to a new range of participants, and that therefore no reduction in the spawning biomass should be considered (see Chapter 10 for detailed projection results). This view was motivated by both the desire for sustainable 10 year rights as well as the possibility that good custodianship should allow the resource to grow during this term. Two variants of this approach were provided, based (a) on projections that poaching continues at present levels, and (b) that poaching is reduced by 50% and remains constant at that level for future years.

In summary, the reasons furnished by the AWG for the recommended allocations as outlined in Table 7.7 were as follows:

Zone A :

General comments: Poaching continues to impact this zone severely. However, this Zone is estimated to be the least depleted of the zones with a current level of depletion of some 59%.

Scenario 1: allows for a sequential phasing down of the catch to zero (to compensate for poaching), over a four year period, assuming that poaching remains at present levels. Under this option, the TAC for 2003/04 would be 125 MT, and would result in approximately a 15% decrease in the spawning biomass level (relative to the current level) in this zone over the next four years. This is not considered an overly large decrease given that the current estimated depletion level is much higher than for the other zones.

Scenario 2: indicates that (a) at present poaching levels, there is no sustainable commercial harvest available and (b) should poaching levels be immediately reduced by 50%, and continue at this level, a commercial harvest of 35 MT should be sustainable.

Zone B:

General comments: Poaching in this zone declined during 2003.

Scenario 1: a TAC of 120 MT was recommended (in part to compensate for reductions in other zones). This would likely result in a reduction of spawner biomass (relative to the current level) of 10% over the next four years if poaching continues at the current level.

Scenario 2: projections indicated that (a) at present levels of poaching, the sustainable harvest was 25 MT. Should poaching be immediately reduced by 50% (b), the capacity for a sustainable commercial harvest increased to 165 MT.

Zone C:

General comments: Continued zero take was recommended for subarea CP. In the interest of economic considerations, a two-year phase-down to zero had been advocated in 2002 for CNP, requiring a TAC of 30 tons for 2002/03, followed by zero for 2003/4. This recommendation was re-iterated.

Scenario 1: There is no sustainable TAC for this zone.

Scenario 2: Zero sustainable TAC.

Zone D:

General comments: This zone is not as depleted as Zone C, and hence, for economic reasons, a 3-year phase-down approach had been recommended in 2002 (40-20-0 MT).

Scenario 1: The TAC recommendation of 20 MT was recognised as being a non-sustainable compromise solution.

Scenario 2: There is zero sustainable TAC available.

The reasons for the Scenario 2 allocations are explained in Chapter 10. Chapter 10 also explains the rationale for managers of the abalone resource opting for Scenario 2b. The final TAC values adopted are given in Table 7.7. The 2003/04 TAC for Zones A to D combined was only some 60% of the previous year's TAC.

Table 7.1 Summary description of model parameters and definitions of other abbreviated terms utilised in the text. The parameters listed in the Table are defined in more detail in Appendices 6.1 and 6.2.

Parameter	Description	Units
$B_0^{sp} = K$	Pre-exploitation (assumed to be 1951) spawning biomass	MT
$B^{sp}, B_{insh}^{sp}, B_{offsh}^{sp}$	Spawning biomass (total per zone), Inshore spawning biomass, Offshore spawning biomass	MT
$\rho$	Rate at which inshore animals move offshore at the start of each Model year	yr <sup>-1</sup>
$r_I$	Proportion of the recruits which settle inshore	-
$CP_{max}$ (number) (zone)	The total number of abalone poached in the year corresponding to the poaching maximum for the zone under consideration	no.
$CP_{max}$ (MT) (zone)	The poaching maximum in terms of mass	MT
$C_{mult}$	Historic catch multiplier for Zone A	-
$P_{poach}$	Parameter that specifies the relative exploitation rate effected by poachers in subareas CP and CNP	-
$M_a : \mu$ ( $\lambda = 0.2$ ) ( $M_a = \mu + \frac{\lambda}{a+1}$ )	Age-dependent mortality rate parameters; $M_0$ is the mortality rate of 0-yr old animals; $M_{15}$ is the plus group mortality rate etc.	yr <sup>-1</sup>
$\nu$	Parameter that controls the steepness of the function describing an increase in 0-yr old mortality due to the ecosystem-change effect	-
$M_{max}$	Maximum increase in 0-yr old mortality rate due to the ecosystem-change effect	yr <sup>-1</sup>
$\hat{a}$ (sector)	Selectivity parameter for sector as indicated; shifts the selectivity function to the left or right	-
$\mu$ (sector)	Selectivity parameter that controls the slope of the right hand limb of the function	-
$\delta$ (sector)	Selectivity parameter that controls the steepness of the ascending left hand limb of the selectivity function.	-
<b>Other definitions</b>		
Zone	Fishery area / management unit: Zones A-G	
CNP, CP	Two subareas comprising Zone C, with CNP subject to less poaching historically than CP	
FIAS	Fishery Independent Abalone Survey	
FIAS $N_{2003}/N_{1951}$	FIAS depletion statistics expressing depletion in terms of <i>number</i> rather than mass	
CS	Commercial sector	
RS	Recreational sector	
PS	Poaching sector (corresponding to illegal catches)	
FS	Parameters pertaining to FIAS	
OS	Parameters pertaining to the Old Surveys conducted during the 1980's	
IS	Industry/MCM joint full population surveys conducted in 2002	
Co/Po <sub>yr</sub>	Confiscations (i.t.o. number) as a proportion of the model-estimated number of animals poached in year <i>yr</i> .	
CI	Confidence Interval (typically 95% CI) determined by likelihood profile method	
LRT	Likelihood Ratio Test	
MSY	Maximum Sustainable Yield	
MSYL	Maximum Sustainable Yield Level	
TAC	Total Allowable Catch (annual catch allocation)	

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Table 7.2. Summary of model assumptions and structure pertaining to the base-case combined ABCD model that simultaneously fits the model to the data for all the zones. All zones are assumed to comprise an inshore and offshore compartment and Zone C is further subdivided into a “poached” subarea CP and “nonpoached” subarea CNP. Parameter values are either fixed externally, “fixed” based on the Zone C values estimated in the model fit, or are estimated simultaneously for all zones.

	Zone A	Zone B	Zone C – subareas CNP & CP	Zone D
Number of parameters estimated in base-case	2	3	23	2
$B_0^{sp}$	Estimate	Estimate	Estimate (2 par)	Estimate
$M$			Estimate (1 par)	
Estimate $p_{poach}$	n/a	n/a	Estimate (1 par)	n/a
Include catch multiplier parameter $C_{mult}$ ?	Yes – fix = 1.5	No	No	No
Poaching amount fixed or estimated in model-fitting process? Does it hit the constraint?	Estimate (hits 50% constraint)	Estimate (hits 50% constraint)	Estimate (1 par)	Estimate
Maximum poached amount relative to Zone C maximum	0.65	0.37	1	0.15
Year poaching assumed to start	1997	1995	1991	1991
First year poaching at maximum level	1997	1995	1994	1995
Migration parameter $\rho$ fixed or estimated in model-fitting process?		Estimate 1 common par for A, B, CNP and D	Estimate (1 par) $\rho_{CP} = 0.5 \rho_{CNP}$	
Proportion of recruitment occurring inshore fixed or estimated?	Fix = 0.9	Fix = 0.9	Fix = 0.9	Fix = 0.9
Downweight catch-at-age data?	Yes	Yes	Yes	Yes
Model ecosystem-change effect?	No	No	Yes (2 pars)	Yes – fix using Zone C parameter estimates
Selectivity parameters			Estimate 15 parameters simultaneously for all zones	



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Table 7.3 Base-case results from model version fitting Zones A, B, CNP, CP and D simultaneously. Best fit estimates of the pre-exploitation spawning biomass  $B_0^{SP}$  (or  $K$ ) for the “poached” CP and “nonpoached” CNP areas of Zone C, and for each of Zones A, B and D, the estimated natural mortality estimates  $M_a$ , the inshore-offshore migration parameters  $\rho$  ( $yr^{-1}$ ), the proportions of recruitment in each subarea that occur inshore versus offshore  $r_i$ , and the poaching maximum  $CP_{max}$  (i.t.o. NUMBERS), which is marked with an asterisk in cases where these values hit constraint boundaries. The  $CP_{max}$  estimates are also shown in terms of biomass and the years to which these estimates apply are given in the row below. Minimum values of the negative of the log-likelihood function are also shown. The estimated selectivity parameters are shown for the commercial sector (CS), recreational sector (RS), poaching sector (PS), FIAS (FS) and the old 1980's survey (OS). Note that for the 2002 industry survey (IS),  $S_a^{IS} = 1$ . Note also that all  $-lnL$  contributions from catch-at-age data have been multiplied by 0.1 as an *ad hoc* adjustment to compensate for likely positive correlation in these data.

Model No. parameters	1) ABCD BASE-CASE 2003 MODEL					LIKELIHOOD	CNP	CP	B	A	D
	CNP	CP	B	A	D						
Zone	II	II	II	II	II	-ln L CPUE	-44.163	-38.349	-35.258	-34.427	-32.089
Poaching Scenario						-ln L FIAS	-7.777	9.246	-7.303	-6.415	-2.228
Confiscation prop. Constrains	50%	50%	50%	50%	50%	-ln L age CS	-7.861	-10.429	-15.508	-15.577	-7.851
$B(0)^{SP}$	2012	4724	5868	8032	7460	-ln L age RS	-6.961	-0.028	-8.034	-1.478	-7.115
$\rho$	0.000935	0.000488	0.000499	0.000499	0.000499	-ln L age PS		-1.230	-3.368	-2.381	1.673
$r^I$	0.9	0.9	0.9	0.9	0.9	-ln L age FIAS	-6.031	-0.468	-6.578	-2.846	-5.088
$CP_{max}$ (no.)		555960	1738660*	989653*	402100	-ln L age OS inshore		-1.742	-1.332	-2.351	-1.142
$CP_{max}$ (MT)		319	810	518	266	-ln L age OS offsh.		-1.386	-0.956	-2.290	-0.316
Year $CP_{max}$ applies to		1995	2002	2002	2002	-ln L age IS insh+offsh.	-0.795		-1.263		
$C_{mult}$ (Zone A)				1.50		-ln L zone subtotal	-117.972		-79.600	-67.765	-54.155
$P_{poach}$	0.29					-ln L TOTAL	-319.492				
$M_0$	0.327					$\sigma$ CPUE	0.081	0.100	0.134	0.139	0.154
$M_1$	0.227					$\sigma$ age CS	0.114	0.090	0.073	0.073	0.116
$M_2$	0.194					$\sigma$ age RS	0.060	0.195	0.057	0.121	0.076
$M_3$	0.177					$\sigma$ age PS		0.155	0.103	0.108	0.208
$M_4$	0.167					$\sigma$ age FIAS	0.064	0.127	0.071	0.105	0.090
$M_5$	0.160					$\sigma$ OS insh.		0.035	0.050	0.060	0.061
$M_6$	0.156					$\sigma$ OS offsh.		0.053	0.076	0.064	0.115
$M_7$	0.152					$\sigma$ IS	0.066		0.026		
$M_8$	0.149					$q$ CPUE	0.0022	0.000747	0.000596	0.000281	0.000365
$M_9$	0.147					Confiscation proportion					
$M_{10}$	0.145					Zone C					
$M_{11}$	0.144					Co/Po <sub>1994</sub>	0.02		0.03		0.01
$M_{12}$	0.143					Co/Po <sub>1995</sub>	0.09		0.14		0.05
$M_{13}$	0.141					Co/Po <sub>1996</sub>	0.11		0.16		0.06
$M_{14}$	0.140					Co/Po <sub>1997</sub>	0.06		0.08	0.08	0.03
$M_{15}$	0.140					Co/Po <sub>1998</sub>	0.08		0.12	0.12	0.05
$v$ (steepness of recruitment fit)	0.249					Co/Po <sub>1999</sub>	0.08		0.12	0.12	0.05
$M_{max}$ (Recruitment failure)	10					Co/Po <sub>2000</sub>	0.17		0.25	0.25	0.10
$h$	0.7	0.7	0.7	0.7	0.7	Co/Po <sub>2001</sub>	0.15		0.23	0.23	0.05
$a(CS)$	8.999					Co/Po <sub>2002</sub>	0.18		0.26	0.26	0.06
$a(RS)$	8.999					Co/Po <sub>2003</sub>	0.34		0.50	0.50	0.11
$a(PS)$	9.905					Mean CS Fishing mortality	0.11	0.13	0.16	0.12	0.05
$a(FS)$	5.300					Catches					
$a(OS)$	6.648					Ccomm(2003)	24.9	0	53.5	131.3	32
$a(IS)$	-					Crec(2003)	1.7	0.8	25.4	8.5	38.2
$\mu(CS)$	0.00045					Cpoa(2003)	14.7	71.8	199.7	441.6	205.0
$\mu(RS)$	0.00044					Catch total (2003) MT	41.4	72.7	278.6	581.3	275.2
$\mu(PS)$	0.00098					CNP		CP	B	A	D
$\mu(FS)$	0.00005					Depletion comp. yr		1981	1982	1986/87	1983
$\mu(OS)$	0.00130					Insh OBS		0.33	0.67	0.33	0.36
$\mu(IS)$	-					Insh PRED		0.57	0.58	0.81	0.65
$\delta(CS)$	498.517					Offsh OBS		0.24	0.54	0.20	0.50
$\delta(RS)$	499.232					Offsh PRED		0.35	0.36	0.66	0.48
$\delta(PS)$	0.369					Depletion statistics					
$\delta(FS)$	1.711					$B^{SP}$ (2003)/ $K$ (Insh. + Offsh)	0.35	0.24	0.34	0.59	0.33
$\delta(OS)$	0.715					$B^{SP}$ (2003)/ $K$ (Insh.)	0.27	0.12	0.25	0.57	0.22
$\delta(IS)$	-					$B^{SP}$ (2003)/ $K$ (Offsh.)	0.42	0.43	0.47	0.61	0.49
						$B^{total}$ (2003)/ $K$	0.32	0.22	0.39	0.62	0.30
						$B^{commercial}$ (2003)/ $K$	0.30	0.27	0.27	0.52	0.39
						FIAS $N_{2003}/N_{1951}$	0.23	0.10	0.27	0.55	0.16

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Table 7.4 Sensitivity results for model version fitting Zones A, B, CNP, CP and D simultaneously.

Model Scenario No. parameters	2) ABCD when fixing selectivities for A&D to values fitted for B&C					3) Estimate industry selectivity parameters					4) $h = 0.6$					5) Catch-at-age likelihood contribution downweighting changed from 0.1 to 0.5				
	30					33					30					30				
Zone	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D
B(0) *	2024	4736	6068	8015	7100	2012	4724	5868	8032	7460	2012	4724	5868	8032	7460	1969	4794	8073	7220	9981
p	0.000932323	0.00046616	0.00069799	0.00069799	0.00069799	0.000935214	0.00046761	0.00049897	0.00049897	0.00049897	0.000935213	0.00046761	0.00049896	0.00049896	0.00049896	0.00073772	0.00036886	0.00041677	0.00041677	0.00041677
r'	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Cpmax (no.)		538057	1.74E+06	989655	333248		555960	1.74E+06	989651	402100		553035	1.74E+06	989653	402083		535696	1.74E+06	991750	680118
Cpmax (MT)		290	805				299	810	518	266		291	783	516	263		288	798	499	466
Cmull (Zone A)				1.50					1.50					1.50					1.50	
Ppoch						0.29					0.29					0.29				
M <sub>0</sub>	0.337					0.327					0.327					0.314				
M <sub>1</sub>	0.237					0.227					0.227					0.214				
M <sub>2</sub>	0.203					0.194					0.194					0.181				
M <sub>3</sub>	0.187					0.177					0.177					0.164				
M <sub>4</sub>	0.177					0.167					0.167					0.154				
M <sub>5</sub>	0.170					0.160					0.160					0.148				
M <sub>6</sub>	0.165					0.156					0.156					0.143				
M <sub>7</sub>	0.162					0.152					0.152					0.139				
M <sub>8</sub>	0.159					0.149					0.149					0.137				
M <sub>9</sub>	0.157					0.147					0.147					0.134				
M <sub>10</sub>	0.155					0.145					0.145					0.133				
M <sub>11</sub>	0.153					0.144					0.144					0.131				
M <sub>12</sub>	0.152					0.143					0.143					0.130				
M <sub>13</sub>	0.151					0.141					0.141					0.129				
M <sub>14</sub>	0.150					0.140					0.140					0.128				
M <sub>15</sub>	0.149					0.140					0.140					0.127				
v	0.2500					0.2491					0.2491					0.2491				
Mmax	10					10					10					10				
h	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.7	0.6	0.6	0.6	0.6	0.6	0.7	0.7	0.7	0.7	0.7
a(CS)	8.9990					8.9983					8.9995					8.9996				
a(RS)	8.9995					8.9992					8.9993					8.9996				
a(PS)	9.7711					9.9046					9.9045					4.7421				
a(FS)	5.4201					5.3000					5.3001					5.3553				
a(OS)	6.6446					6.6476					6.6476					5.7764				
a(IS)	-					5.8109					-					-				
m(CS)	0.0005					0.0005					0.0005					0.0007				
m(RS)	0.0004					0.0004					0.0004					0.0011				
m(PS)	0.0010					0.0010					0.0010					0.0000				
m(FS)	0.0000					0.0001					0.0001					0.0009				
m(OS)	0.0013					0.0013					0.0013					0.0005				
m(IS)	-					0.0018					-					-				
d(CS)	498.52					498.52					498.52					575.32				
d(RS)	499.23					499.23					499.23					596.79				
d(PS)	0.4227					0.3687					0.3686					0.3435				
d(FS)	1.6165					1.7114					1.7113					1.4137				
d(OS)	0.7129					0.7151					0.7152					0.5698				
d(IS)	-					0.6321					-					-				

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Table 7.4 (contd) Sensitivity results for model version fitting Zones A, B, CNP, CP and D simultaneously.

Model Scenario	2) ABCD when fixing selectivities for A&D to values fitted for B&C					3) Estimate industry selectivity parameters					4) $h = 0.6$					5) Catch-at-age likelihood contribution downweighting changed from 0.1 to 0.5				
	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D
<b>Likelihood contributions</b>																				
-ln L CPUE	-44.239	-38.387	-34.329	-34.794	-32.700	-44.163	-38.349	-35.258	-34.427	-32.089	-44.614	-38.384	-32.644	-33.815	-31.601	-41.557	-38.118	-34.326	-32.526	-28.690
-ln L FIAS	-7.798	9.581	-7.351	-6.399	-2.379	-7.777	9.246	-7.303	-6.415	-2.228	-7.701	9.157	-7.092	-6.383	-2.378	-7.713	9.331	-7.314	-6.303	-2.680
-ln L age CS	-8.207	-10.249	-16.192	-15.429	-6.417	-7.861	-10.429	-15.508	-15.577	-7.851	-8.051	-10.362	-14.824	-15.856	-7.718	-41.569	-52.745	-79.754	-78.228	-50.012
-ln L age RS	-7.125	-0.027	-8.100	-1.533	-4.537	-6.960	-0.028	-8.035	-1.478	-7.115	-6.478	0.046	-8.009	-1.470	-6.537	-36.262	-0.010	-40.410	-7.823	-41.567
-ln L age PS		-1.304	-3.369	-2.496	2.754		-1.230	-3.368	-2.381	1.673		-0.753	-3.102	-2.406	1.965		-7.900	-15.091	-12.814	0.970
-ln L age FIAS	-5.994	-0.521	-6.659	-3.509	-4.584	-6.031	-0.468	-6.577	-2.846	-5.088	-5.903	-0.396	-6.431	-2.915	-5.090	-27.306	-1.972	-35.156	-23.232	-23.705
-ln L age OS inshore		-1.741	-1.368	-1.752	-1.324		-1.742	-1.332	-2.351	-1.142		-1.827	-1.384	-2.323	-1.149		-5.937	-5.570	-17.304	-4.613
-ln L age OS offsh.		-1.342	-0.914	-1.893	-0.114		-1.386	-0.956	-2.290	-0.316		-1.329	-0.926	-2.272	-0.300		-6.759	-7.358	-17.036	-10.321
-ln L age IS in-offsh.	-0.852		-1.181			-0.621		-0.262			-0.799		-1.364			-3.855		-5.517		
-ln L zone subtotal	-118.206		-79.463	-67.804	-49.300	-117.797		-78.599	-67.765	-54.155	-117.392		-75.777	-67.240	-52.607	-262.370		-230.498	-195.264	-160.617
-ln L TOTAL	-314.773					-318.317					-313.216					-448.740				
<b>s CPUE</b>	0.081	0.100	0.139	0.137	0.150	0.081	0.100	0.134	0.139	0.154	0.080	0.100	0.150	0.142	0.157	0.092	0.101	0.139	0.151	0.178
s age CS	0.111	0.091	0.070	0.073	0.129	0.114	0.090	0.073	0.073	0.116	0.113	0.090	0.077	0.072	0.117	0.110	0.089	0.071	0.072	0.100
s age RS	0.058	0.195	0.057	0.119	0.110	0.060	0.195	0.057	0.121	0.076	0.065	0.217	0.057	0.121	0.083	0.057	0.202	0.057	0.117	0.064
s age PS		0.153	0.103	0.105	0.252		0.155	0.103	0.108	0.208		0.167	0.108	0.107	0.219		0.147	0.110	0.103	0.160
s age FIAS	0.064	0.125	0.070	0.091	0.088	0.064	0.127	0.071	0.105	0.090	0.065	0.130	0.073	0.103	0.090	0.071	0.130	0.066	0.072	0.085
s OS insh.		0.035	0.048	0.079	0.052		0.035	0.050	0.060	0.061		0.032	0.047	0.061	0.061		0.057	0.060	0.036	0.075
s OS offsh.		0.055	0.078	0.074	0.135		0.053	0.076	0.064	0.115		0.055	0.077	0.064	0.117		0.054	0.051	0.042	0.030
s IS	0.061		0.029			0.082		0.091			0.066		0.023			0.088		0.032		
q CPUE	0.00216	0.00070	0.00052	0.00028	0.00039	0.00220	0.00075	0.00060	0.00028	0.00036	0.00236927	0.0006656	0.00069737	0.00029276	0.00039462	0.00310	0.00082	0.00062	0.00036	0.00024
<b>Confiscation prop.</b>		Zone C					Zone C					Zone C					Zone C			
CoPo <sub>2004</sub>	0.03		0.03		0.02	0.02		0.03		0.01	0.02		0.03		0.01	0.03		0.03		0.01
CoPo <sub>2005</sub>	0.10		0.14		0.06	0.09		0.14		0.05	0.09		0.14		0.05	0.09		0.14		0.03
CoPo <sub>2006</sub>	0.12		0.16		0.08	0.11		0.16		0.06	0.11		0.16		0.06	0.12		0.16		0.04
CoPo <sub>2007</sub>	0.06		0.08	0.08	0.04	0.06		0.08	0.08	0.03	0.06		0.08	0.08	0.03	0.06		0.08	0.08	0.02
CoPo <sub>2008</sub>	0.09		0.12	0.12	0.06	0.08		0.12	0.12	0.05	0.08		0.12	0.12	0.05	0.08		0.12	0.12	0.03
CoPo <sub>2009</sub>	0.08		0.12	0.12	0.06	0.08		0.12	0.12	0.05	0.08		0.12	0.12	0.05	0.08		0.12	0.12	0.03
CoPo <sub>2010</sub>	0.17		0.25	0.25	0.11	0.17		0.25	0.25	0.10	0.17		0.25	0.25	0.10	0.17		0.25	0.25	0.06
CoPo <sub>2011</sub>	0.16		0.23	0.23	0.06	0.15		0.23	0.23	0.05	0.15		0.23	0.23	0.05	0.16		0.23	0.23	0.08
CoPo <sub>2012</sub>	0.18		0.26	0.26	0.07	0.18		0.26	0.26	0.06	0.18		0.26	0.26	0.06	0.18		0.26	0.26	0.04
CoPo <sub>2013</sub>	0.35		0.50	0.50	0.13	0.34		0.50	0.50	0.11	0.34		0.50	0.50	0.11	0.35		0.50	0.50	0.06
Mean CS F mort.	0.11	0.12	0.10	0.07	0.15	0.11	0.13	0.16	0.07	0.05	0.11	0.13	0.25	0.07	0.06	0.15	0.13	0.19	0.13	0.03
<b>Catches</b>																				
Ccomm(2003)	24.9	0	53.5	131.3	32	24.9	0	53.5	131.3	32	24.9	0	53.5	131.3	32	24.9	0	53.5	131.3	32
Crec(2003)	1.8	0.9	24.9	8.3	37.2	1.7	0.8	25.4	8.5	38.2	1.7	0.8	24.4	8.5	37.9	1.8	0.9	25.8	8.4	38.9
Cpoc(2003)	13.1	73.4	197.4	429.1	164.6	14.7	71.8	199.7	441.6	205.0	17.1	66.2	184.2	439.4	202.2	14.7	71.0	200.8	423.5	359.7
Catch total (2003) MT	39.8	74.3	275.8	568.7	233.9	41.4	72.7	278.6	581.3	275.2	43.8	67.0	262.1	579.1	272.1	41.4	71.9	280.0	563.1	430.6
	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D	CNP	CP	B	A	D
<b>Depletion comp. yr</b>		1981	1982	1986/87	1983		1981	1982	1986/87	1983		1981	1982	1986/87	1983		1981	1982	1986/87	1983
Insh OBS		0.33	0.67	0.33	0.36		0.33	0.67	0.33	0.36		0.33	0.67	0.33	0.36		0.33	0.67	0.33	0.36
Insh PRED		0.60	0.85	0.84	0.67		0.57	0.58	0.81	0.65		0.52	0.52	0.78	0.61		0.51	0.53	0.75	0.71
Offsh OBS		0.24	0.54	0.20	0.50		0.24	0.54	0.20	0.50		0.24	0.54	0.20	0.50		0.24	0.54	0.20	0.50
Offsh PRED		0.39	0.42	0.69	0.49		0.35	0.36	0.66	0.48		0.31	0.32	0.64	0.45		0.31	0.33	0.59	0.58
<b>Depletion statistics</b>																				
B*(2003)K (Zone)	0.37	0.26	0.41	0.61	0.34	0.35	0.24	0.34	0.59	0.33	0.32	0.20	0.28	0.56	0.30	0.32	0.22	0.31	0.50	0.36
B*(2003)K (Insh.)	0.29	0.14	0.30	0.60	0.21	0.27	0.12	0.25	0.57	0.22	0.24	0.03	0.19	0.55	0.20	0.29	0.12	0.23	0.49	0.23
B*(2003)K (Offsh.)	0.43	0.45	0.53	0.63	0.49	0.42	0.43	0.47	0.61	0.49	0.38	0.37	0.40	0.58	0.45	0.36	0.40	0.43	0.52	0.56
B <sup>mean</sup> (2003)K	0.33	0.24	0.46	0.64	0.31	0.32	0.22	0.39	0.62	0.30	0.29	0.18	0.33	0.59	0.28	0.30	0.21	0.35	0.53	0.33
B <sup>weighted</sup> (2003)K	0.31	0.30	0.34	0.54	0.41	0.30	0.27	0.27	0.52	0.39	0.28	0.22	0.21	0.50	0.35	0.24	0.25	0.26	0.44	0.42
FIAS N <sub>2002/N 1981</sub>	0.24	0.12	0.31	0.56	0.15	0.23	0.10	0.27	0.55	0.16	0.20	0.06	0.20	0.53	0.13	0.23	0.10	0.28	0.52	0.15

Table 7.5. Modified projection scenarios as agreed by the AWG for use in determining predicted changes in spawning biomass in each of the zones/subareas as indicated. For all scenarios, H is the estimated 2003 poaching level (assumed to remain at this level for all future years) and M is 50% of the 2003 poaching level (i.e. projections assume poaching is halved relative to the 2003 level in all future years). The projected catches below refer to commercial + recreational catches in tonnes.

	Year=2004	2005	2006	2007	2008	Poaching
<b>Zone A</b>						
1.	100	100	100	100	100	M
2.	100	100	100	100	100	H
3.	170	170	170	170	170	M
4.	170	170	170	170	170	H
5.	200	200	200	200	200	M
6.	200	200	200	200	200	H
<b>Zone B</b>						
1.	80	80	80	80	80	M
2.	80	80	80	80	80	H
3.	100	80	60	40	20	M
4.	100	80	60	40	20	H
5.	120	120	120	120	120	M
6.	120	120	120	120	120	H
<b>Zone CNP</b>						
1.	0	0	0	0	0	M
2.	0	0	0	0	0	H
3.	30	0	0	0	0	M
4.	30	0	0	0	0	H
5.	40	20	0	0	0	M
6.	40	20	0	0	0	H
<b>Zone CP</b>						
1.	0	0	0	0	0	M
2.	0	0	0	0	0	H
<b>Zone D</b>						
1.	0	0	0	0	0	M
2.	0	0	0	0	0	H
3.	30	0	0	0	0	M
4.	30	0	0	0	0	H
5.	40	20	0	0	0	M
6.	40	20	0	0	0	H

Table 7.6 Summary of four model versions used in projection scenarios, and summarised future depletion estimates corresponding to the six projection scenarios detailed in Table 7.5.

Model	I) ABCD BASE-CASE 2003 MODEL Co/Po = 0.50					II) ABCD with 25% confiscation proportion scenario Co/Po = 0.25				
	CNP	CP	B	A	D	CNP	CP	B	A	D
<b>Zone</b>	PII	PII	PII	PII	PII	PII	PII	PII	PII	PII
<b>Poaching Scenario</b>	50%	50%	50%	50%	50%	25%	25%	25%	25%	25%
<b>Confiscation / Poaching %</b>										
$B(t)^{SP}$	2012	4724	5868	8032	7460	7164	5029	10965	10928	8000
$C_{pmax}$ (no.)		555960	1.74E+06	989653	402100		607643	3.48E+06	1979310	373031
$C_{pmax}$ (MT)		319	810	518	266		417	1679	967	247
$M_{15}$	0.140					0.188				
-ln L zone subtotal	-117.972		-79.600	-67.765	-54.155	-107.785		-71.273	-66.349	-46.823
<b>-ln L TOTAL</b>	<b>-319.482</b>					<b>-282.230</b>				
$Co/Po_{2002}$	0.18		0.26	0.26	0.06	0.13		0.13	0.13	0.06
$Co/Po_{2003}$	0.34		0.50	0.50	0.11	0.25		0.25	0.25	0.12
<b>Depletion comp. yr</b>		1981	1982	1986/87	1983		1981	1982	1986/87	1983
<b>OBSERVED</b>		<b>0.29</b>	<b>0.61</b>	<b>0.27</b>	<b>0.43</b>		<b>0.29</b>	<b>0.61</b>	<b>0.27</b>	<b>0.43</b>
<b>MODEL</b>		0.46	0.47	0.74	0.56		0.64	0.80	0.86	0.71
<b>Depletion statistics</b>										
$B^{SP}$ (2003)/ $K^{SP}$	0.35	0.24	0.34	0.59	0.33	0.49	0.29	0.60	0.68	0.36
<b>Projections for different Scenarios</b>										
1. $B^{SP}$ (2008)/ $K$ for Scenario no. 1	0.20	0.11	0.39	0.54	0.14	0.23	0.12	0.65	0.62	0.13
2. $B^{SP}$ (2008)/ $K$ for Scenario no. 2	0.19	0.11	0.33	0.45	0.13	0.22	0.10	0.58	0.51	0.12
3. $B^{SP}$ (2008)/ $K$ for Scenario no. 3	0.19		0.40	0.51	0.14	0.23		0.65	0.60	0.13
4. $B^{SP}$ (2008)/ $K$ for Scenario no. 4	0.17		0.33	0.42	0.13	0.22		0.58	0.49	0.11
5. $B^{SP}$ (2008)/ $K$ for Scenario no. 5	0.17		0.37	0.50	0.13	0.22		0.64	0.59	0.13
6. $B^{SP}$ (2008)/ $K$ for Scenario no. 6	0.16		0.31	0.41	0.13	0.21		0.57	0.48	0.11
<b>Model</b>	<b>HIGHER RECRUITMENT SCENARIOS</b>					<b>IV) ABCD with upper bound on Mmax and confis % = 25%</b>				
	<b>III) ABCD with upper bound on Mmax</b>					<b>Co/Po = 0.25</b>				
	<b>Co/Po = 0.50</b>					<b>Co/Po = 0.25</b>				
<b>Zone</b>	CNP	CP	B	A	D	CNP	CP	B	A	D
<b>Poaching Scenario</b>	PII	PII	PII	As above	PII	PII	PII	PII	As above	PII
<b>Confiscation / Poaching %</b>	50%	50%	50%		50%	25%	25%	25%		25%
$B(t)^{SP}$	2012	4724	5868		7460	7132	5027	10771		8178
$C_{pmax}$ (no.)		555961	1.74E+06		402104		599421	3.47E+06		277458
$C_{pmax}$ (MT)		319	810		256		405	1647		174
$M_{15}$	0.140					0.185				
-ln L zone subtotal	-117.380		-79.600		-53.829	-106.676		-70.264		-47.420
<b>-ln L TOTAL</b>	<b>-318.573</b>					<b>-284.896</b>				
$Co/Po_{2002}$	0.18		0.26		0.06	0.13		0.13		0.09
$Co/Po_{2003}$	0.34		0.50		0.11	0.25		0.25		0.16
<b>Depletion comp. yr</b>		1981	1982		1983		1981	1982		1983
<b>OBSERVED</b>		<b>0.29</b>	<b>0.61</b>		<b>0.43</b>		<b>0.29</b>	<b>0.61</b>		<b>0.43</b>
<b>MODEL</b>		0.46	0.47		0.56		0.63	0.80		0.71
<b>Depletion statistics</b>										
$B^{SP}$ (2003)/ $K^{SP}$	0.36	0.25	0.34		0.34	0.51	0.31	0.60		0.41
<b>Projections for different Scenarios</b>										
1. $B^{SP}$ (2008)/ $K$ for Scenario no. 1	0.26	0.17	0.39		0.21	0.32	0.20	0.64		0.25
2. $B^{SP}$ (2008)/ $K$ for Scenario no. 2	0.25	0.15	0.33		0.17	0.32	0.18	0.57		0.23
3. $B^{SP}$ (2008)/ $K$ for Scenario no. 3	0.25		0.40		0.21	0.32		0.64		0.25
4. $B^{SP}$ (2008)/ $K$ for Scenario no. 4	0.24		0.33		0.17	0.31		0.57		0.23
5. $B^{SP}$ (2008)/ $K$ for Scenario no. 5	0.23		0.37		0.20	0.32		0.63		0.25
6. $B^{SP}$ (2008)/ $K$ for Scenario no. 6	0.22		0.31		0.17	0.31		0.56		0.22

Table 7.7. Abalone TAC recommendations as made by the AWG in 2003 for total catches (in tonnes) from each zone for the 2003/04 season. Given uncertainty regarding the proposed future changes in management, two scenarios were presented for consideration. Scenario 1 assumed a policy *status quo* with existing right holders remaining in the fishery, and allowed for further depletion of the resource in certain zones in the interest of retaining existing economic structures. Scenario 2 assumed adoption of a new policy, and new right holders with 10-year rights. Recommendations related to this latter Scenario allowed for no further depletion of the resource, in the interests of maintaining (or improving) TACs over a 10-year term. Two options were presented for this latter scenario: (a) poaching continues at present levels, and (b) poaching continues at 50% of present levels. TAC values for Zones E-G are shown here for completeness and to enable comparison with the preceding year's TAC, but the motivations for the choice of the values shown are not discussed in this thesis. The preceding year's TAC is shown in italics and includes allowance for the recreational catches. The actual recommendation adopted is shown in the last row of the Table – the reasons for the Zones C and D allocations (marked with an asterisk) are explained in Chapter 10.

Zone	A	B	C	D	E	F	G	Total
<u>Scenario 1</u>								
<i>Last TAC (2002/03)</i>	<i>170</i>	<i>100</i>	<i>30</i>	<i>40</i>	<i>18</i>	<i>20</i>	<i>27</i>	<i>430</i>
TAC for 2003/04	125	120	0	20	15	20	27	327
<u>Scenario 2</u>								
2a	0	25	0	0	15	20	27	87
2b	35	165	0	0	15	20	27	262
<u>Adopted 2003/04 recommendation</u>	35	165	10*	10*	15	20	27	282

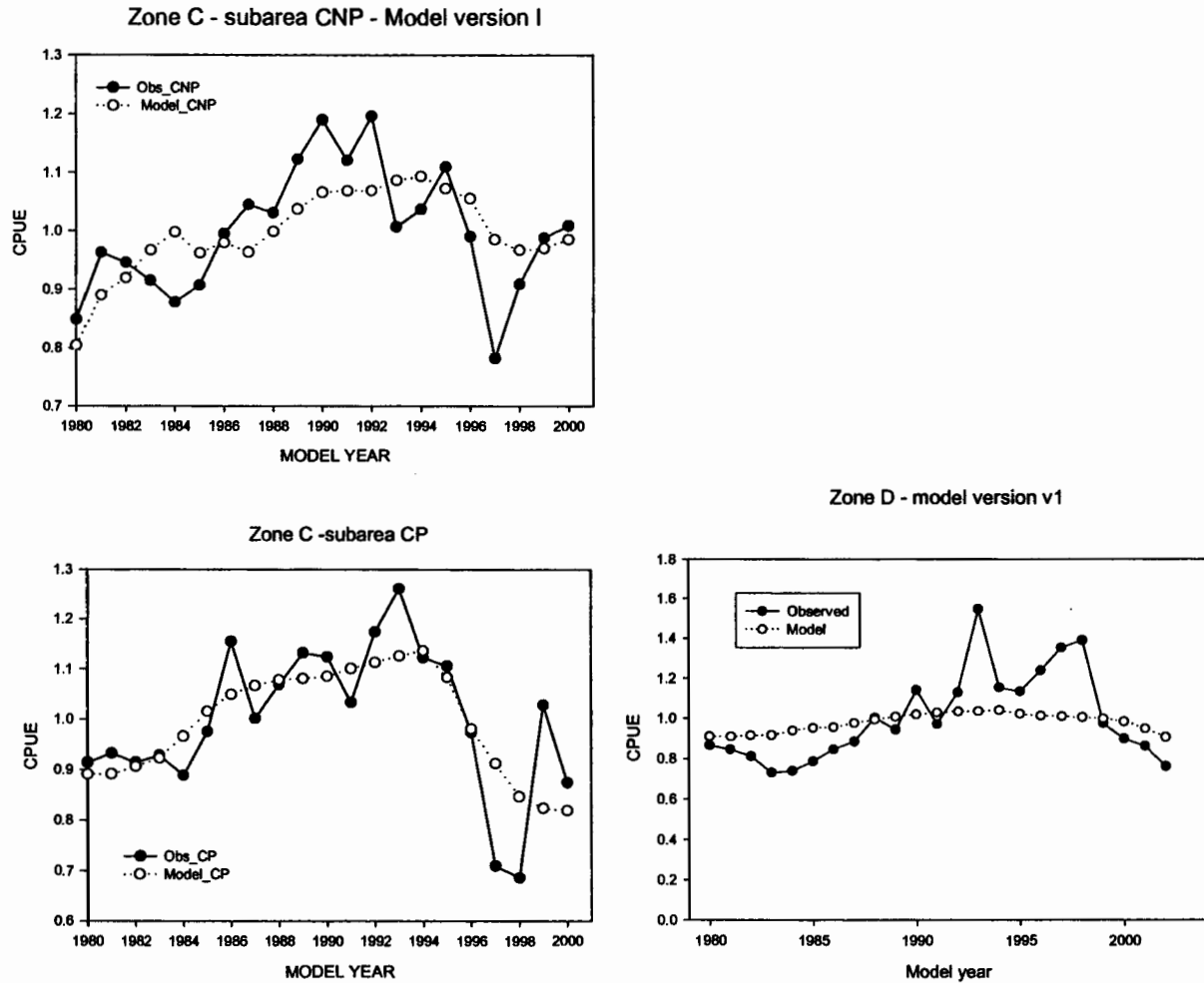


Fig. 7.1. Comparisons between the standardised CPUE and model-predicted CPUE values (for the base-case combined ABCD model) for each of subareas CNP and CP in Zone C and for Zone D. Note that the vertical axes have non-zero intercepts for better visual discrimination of trends.

### Appendix 7.1. Background to model results and sensitivities.

The 2003 base-case ABCD abalone model was developed based on the combined Zones B and C model adopted by the AWG in 2002. This Appendix details some of the motivations for the choice of the Zones B and C models described below because these made a critical contribution to the development of the current base-case combined ABCD model. The original combined B and C model differed from the current model in the following respects:

- i) Only Zone B and subareas CNP and CP were included;
- ii) No constraints were applied when estimating poaching levels;
- iii) Poaching trend scenarios I, II and III were used (see Table 4.4; Table A7.1);
- iv) Deep FIAS catch-at-age data were included and were downweighted by a factor of 0.1;
- v) The inshore-offshore migration parameter for Zone B was set equal to the CNP migration parameter.

Following a number of sensitivity tests, three model versions were selected in 2002 by the Abalone Working Group (AWG) for further consideration. Model v.1 used Poaching Scenario II and was the reference-case model; hence summarised results for this model version are given in Table A7.2. Model v.2 (the best fit model) used Poaching Scenario III whereas model v.3 used Poaching Scenario II, but fixed the Zone B maximum poaching estimate at a higher level.

The differences between the three Zones B&C model versions were therefore as follows:

- v1: Poaching Scenario II + fixes Zone B maximum poaching level = 504460 abalone;
- v2: Poaching Scenario III + fixes Zone B maximum poaching level = 504460 abalone;
- v3: Poaching Scenario II + fixes Zone B maximum poaching level = 1008920 abalone.

The following is a brief summary motivating the choice of these model versions:

- If the Zone B poaching estimate was freed to be estimated by the model, the resulting poaching estimates obtained under a range of model scenarios were close to zero or less than the total number of abalone known to have been confiscated from Zone B (Table A7.1). Similar problems were encountered in trying to fix the Zone B poaching estimate relative to the Zone C estimate. Hence the Zone B maximum poaching estimate (corresponding to the year 2002) was set at a minimum realistic value equal to the TOTAL number (allocation known confiscations + a proportion of the allocation-unknown confiscations) of Zone B confiscations (i.e. confiscation proportion assumed = 100%). Results were also presented for a case assuming a 50% confiscation proportion. The 100% confiscation proportion scenario corresponds to a Zone B (2002) poaching estimate of 272 MT, which is less than the Zone C (1995) maximum poaching estimate of 397 MT. Given that the Zone B : Zone C maximum poaching level ratio should be approximately 5 (see Table 4.4), the Zone B poaching estimate used here is likely to be an underestimate; hence an example is also given which incorporates a doubling of the Zone B poaching estimate.
- Sensitivities were examined using Poaching Scenarios II ( $-\ln L = -205.51$ ) and III. Poaching Scenario III ( $-\ln L = -207.41$ ) gave the best fit overall (as well as the best fit to the industry survey catch-at-age data). Poaching Scenario I produced the worst fit ( $-\ln L = -199.75$ ).
- Sensitivities were examined for different starting years for the recruitment failure effect (see Fig. A7.1 for example). The effect of assuming a later (versus earlier) start year for this effect was to increase both the steepness of recruitment failure parameter  $\nu$  and the



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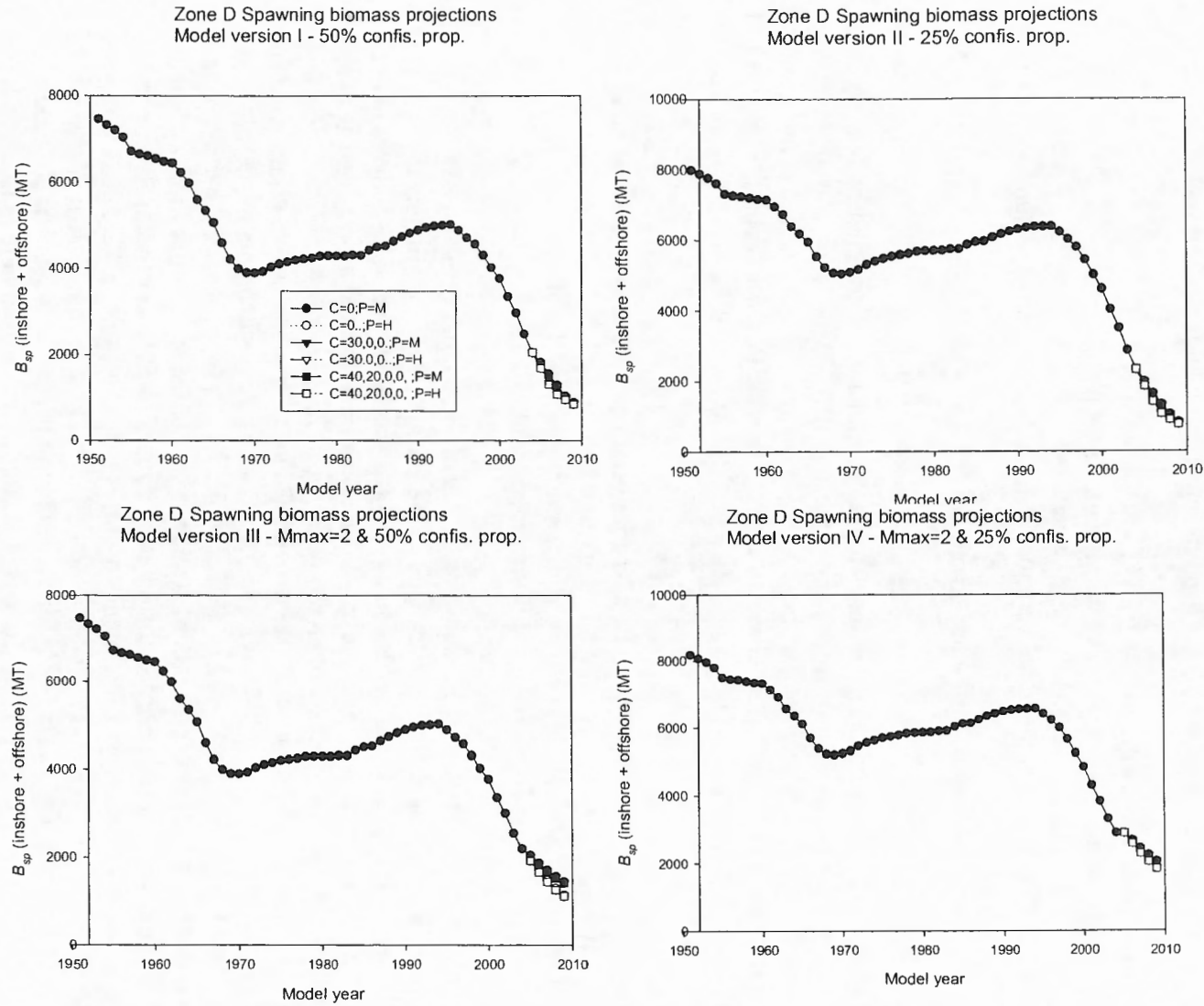


Fig. 7.22 Projected spawning biomass trajectories for Zone D under six alternative future catch scenarios as summarized in Table 7.5 and for four different model scenarios as detailed in the text.

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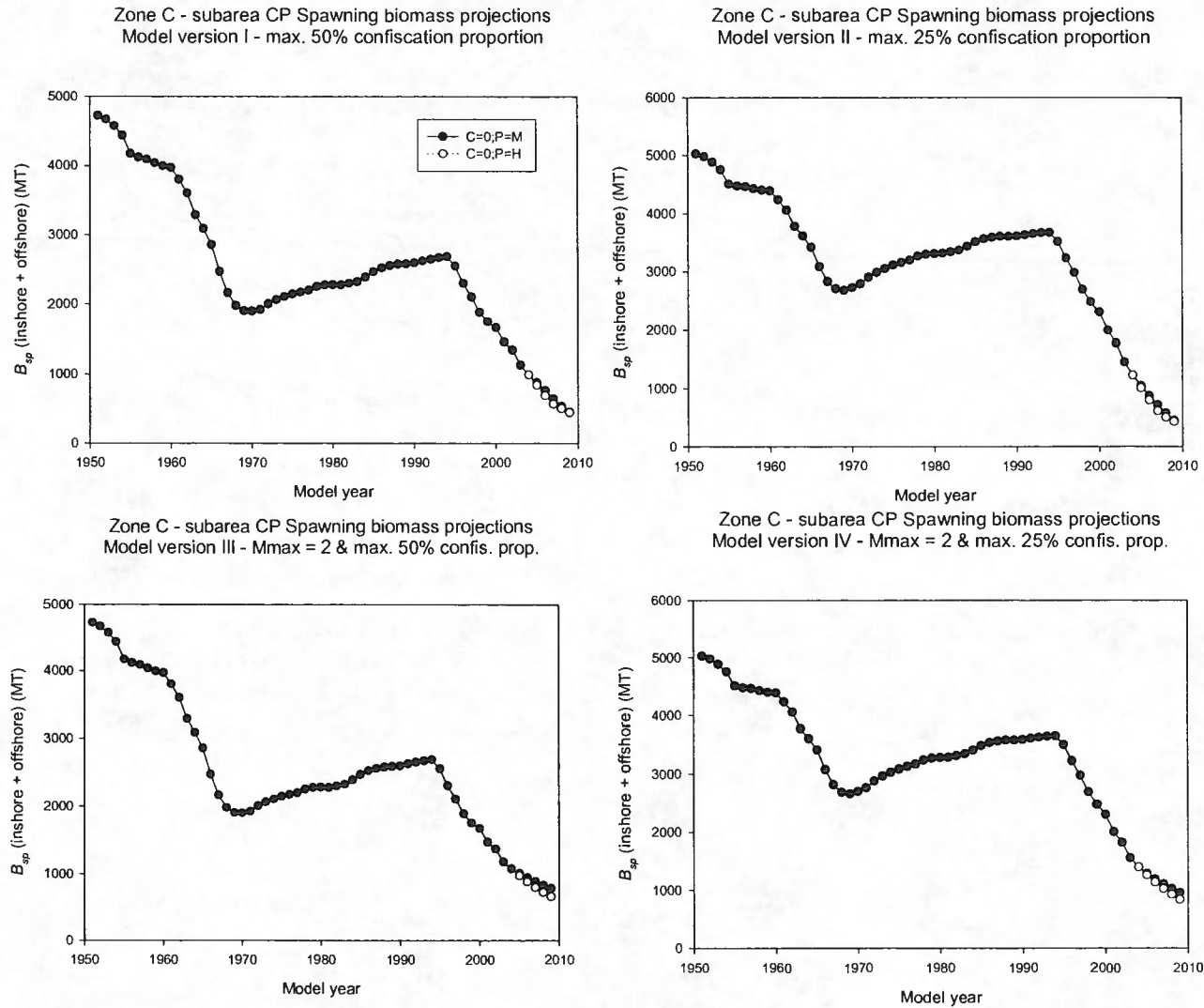


Fig. 7.21. Projected spawning biomass trajectories for subarea CP of Zone C under two alternative future catch scenarios as summarized in Table 7.5 and for four different model scenarios as detailed in the text.

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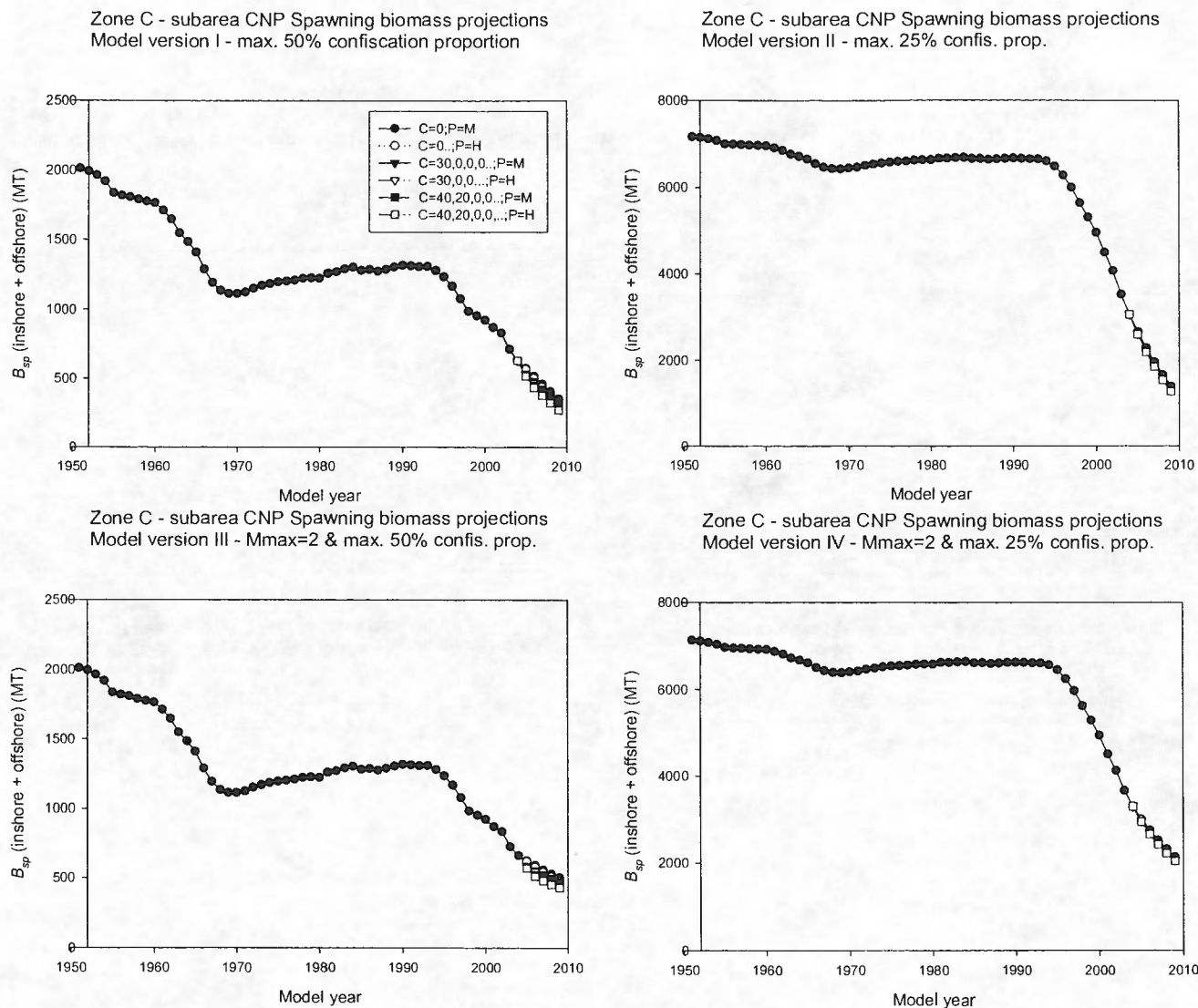


Fig. 7.20 Projected spawning biomass trajectories for subarea CNP of Zone C under six alternative future catch scenarios as summarized in Table 7.5 and for four different model scenarios as detailed in the text.

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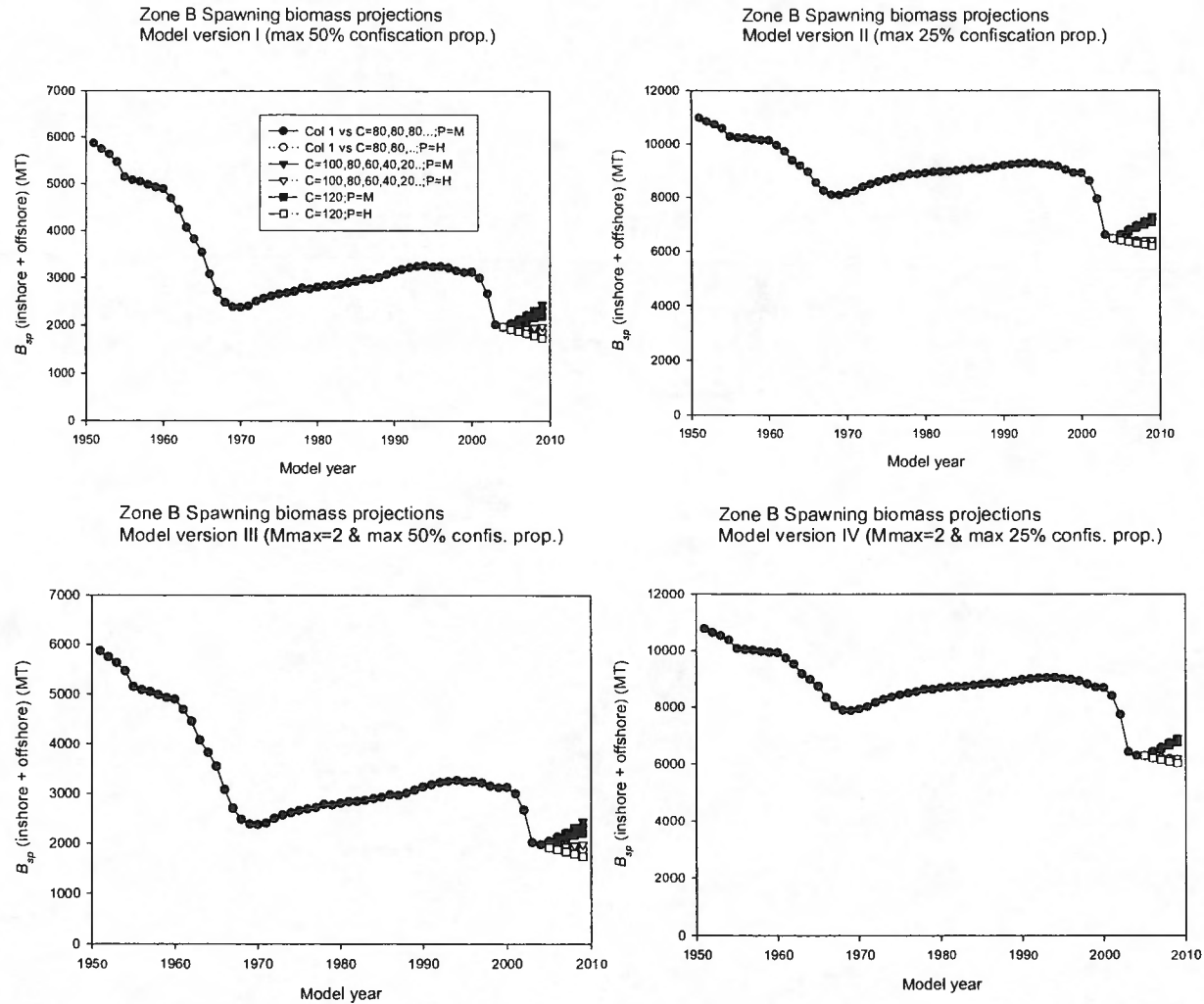


Fig. 7.19 Projected spawning biomass trajectories for Zone B under six alternative future catch scenarios as summarized in Table 7.5 and for four different model scenarios as detailed in the text. The top left plot is the base-case combined ABCD model. The left-hand plots assume a 50% confiscation level constraint and the right-hand plots a 25% confiscation level constraint. The model is refitted when fixing  $M_{max} = 2$  (lower plots). Although this parameter pertains only to Zones C and D, it is estimated through combined fitting to Zones B and C and hence the best fit model corresponding to this scenario needs to be shown for Zone B also.

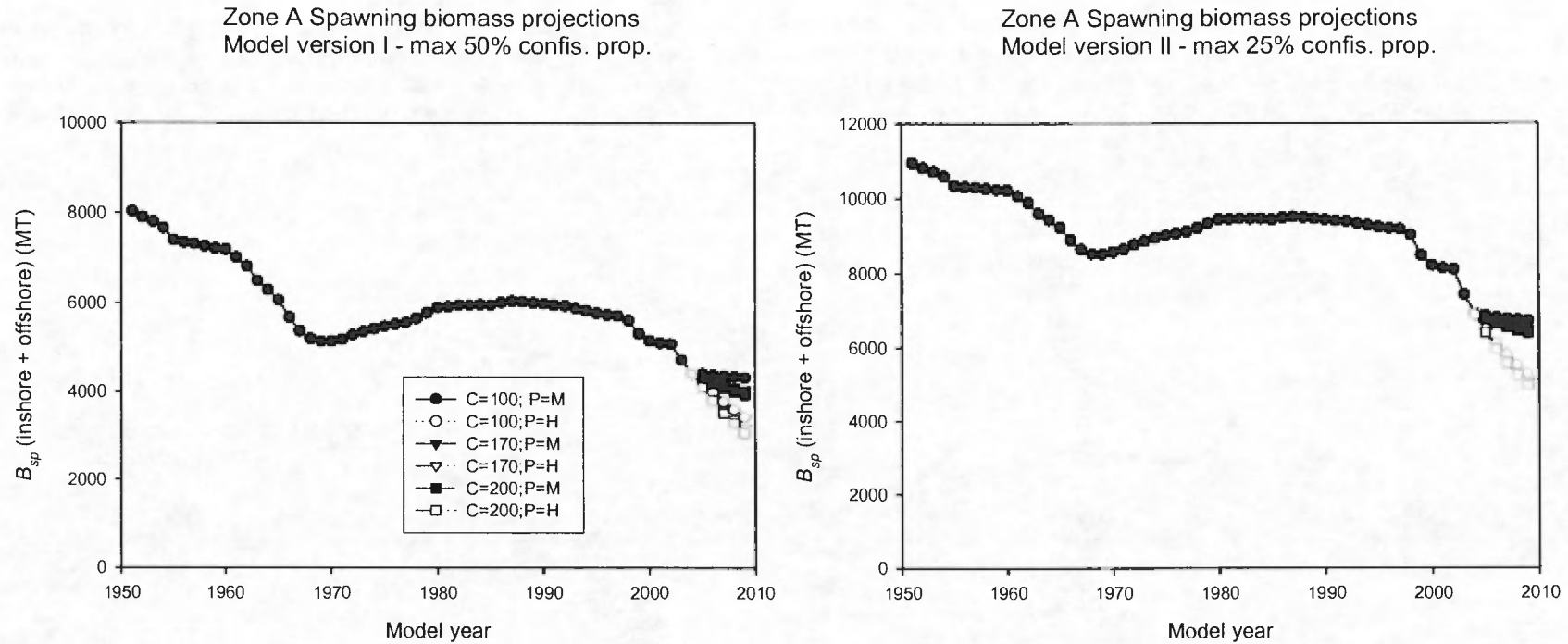


Fig. 7.18 Projected spawning biomass trajectories for Zone A under six alternative future catch scenarios as summarized in Table 7.5 and for the base-case combined ABCD model (LHS), which has a 50% confiscation level constraint, and a sensitivity scenario (RHS) which has a maximum 25% confiscation level constraint. Changing the  $M_{max}$  parameter which applies to Zones C and D has no effect on the predictions for Zone A. Note that projections assuming future poaching catches are half the current estimates are indicated by solid symbols compared to the open circles which represent scenarios under which future poaching levels are assumed to remain at the current estimated level.

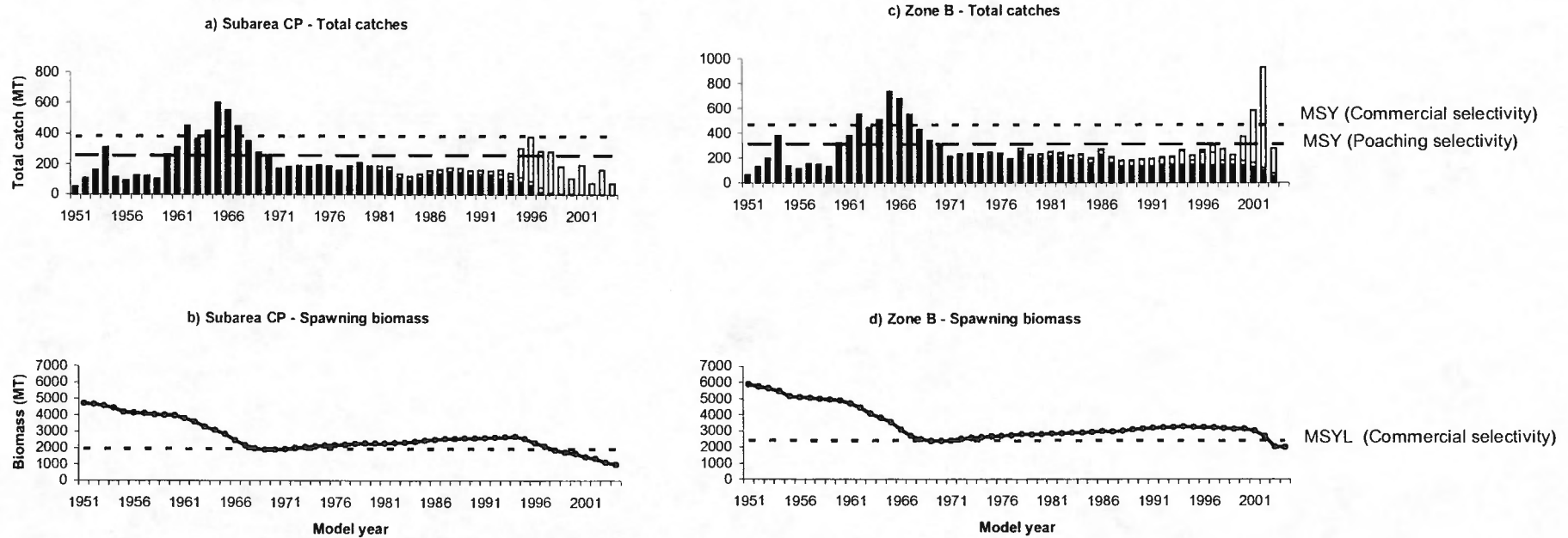


Fig. 7.17 Comparison between the base-case combined ABCD model maximum sustainable yield (*MSY*) (tonnes) estimates (assuming either a commercial or a poaching selectivity function (see Fig. 7.14)) and total annual abalone catches (tonnes) over the period 1951 to 2003 for subarea CP of Zone C and for Zone B. The total catches shown include both the commercial and recreational catches, as well as the total poaching catch estimated in the base-case model. The lower panels show model-predicted spawning biomass trajectories for CP and Zone B compared with the corresponding estimates of the spawning biomass level corresponding to the *MSY*.

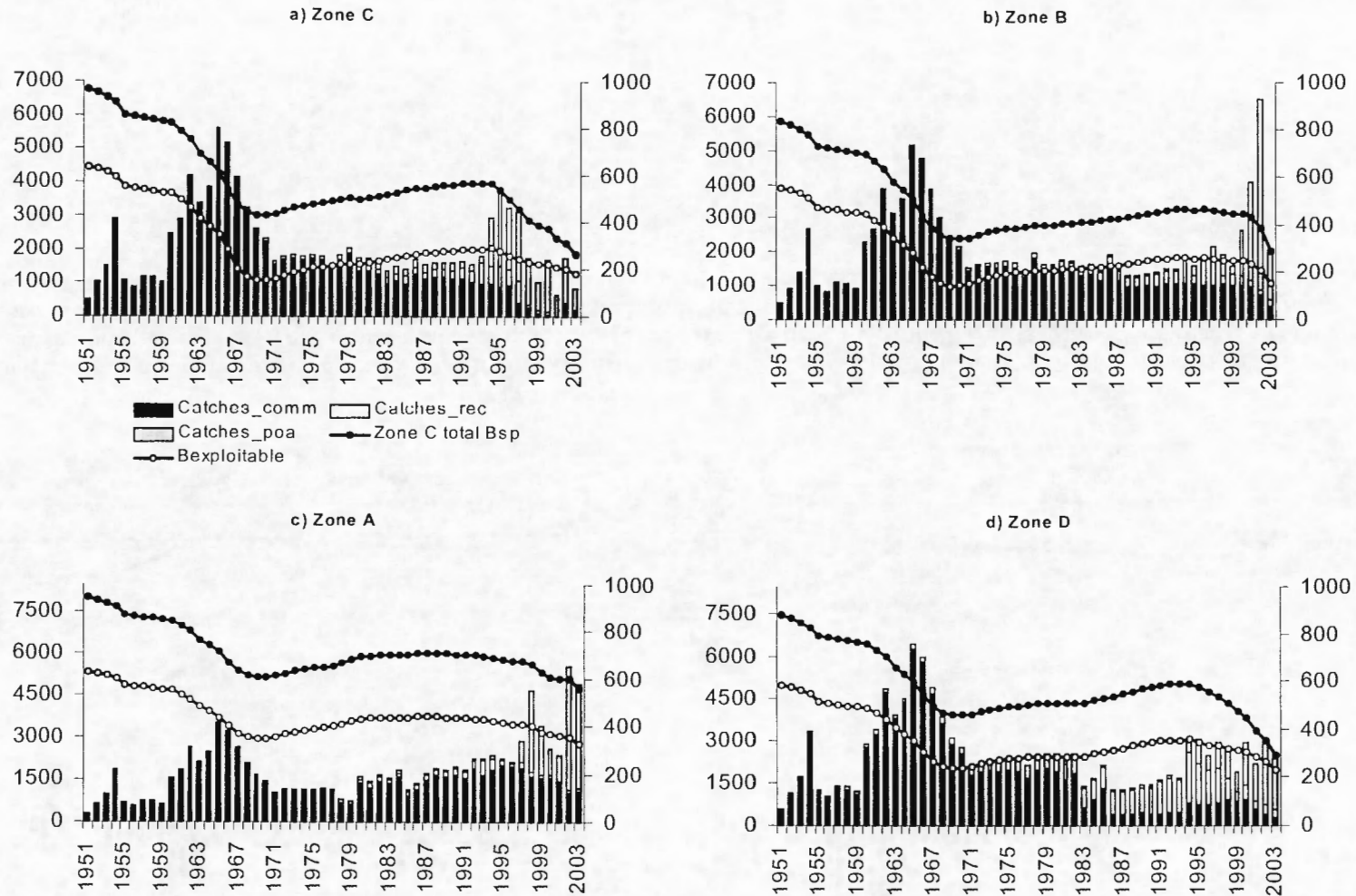


Fig. 7.16. Spawning biomass and commercial exploitable biomass on the left hand axis (as estimated for the base-case combined ABCD model) are shown relative to total catches (commercial + poaching + recreational) as plotted on the right hand vertical axis, for each of a) Zone C, b) Zone B, c) Zone A and d) Zone D.



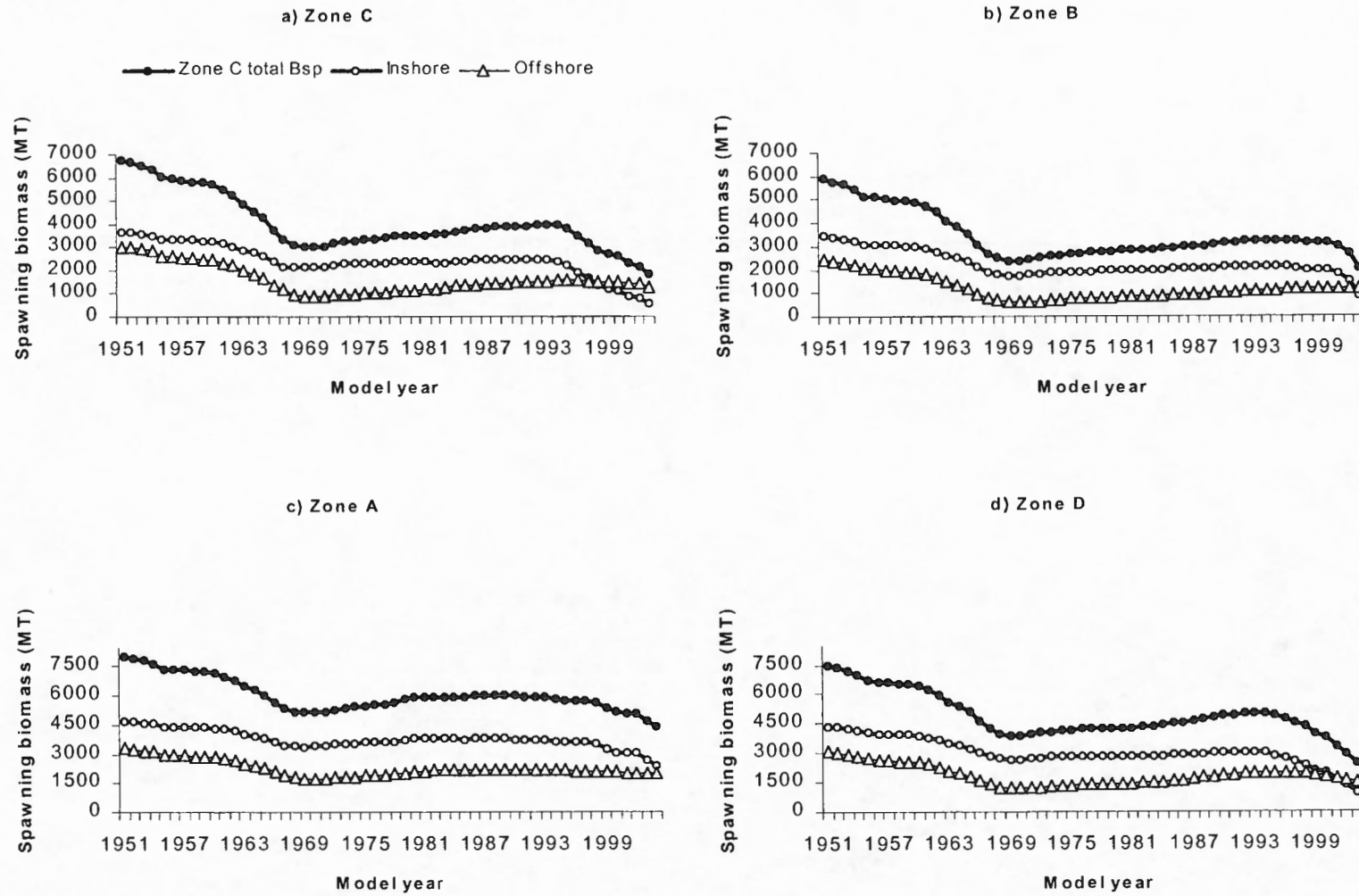


Fig. 7.15 Base-case combined ABCD model inshore, offshore and total spawning biomass trajectories shown for a) Zone C (subareas CNP and CP combined), b) Zone B, c) Zone A and d) Zone D.



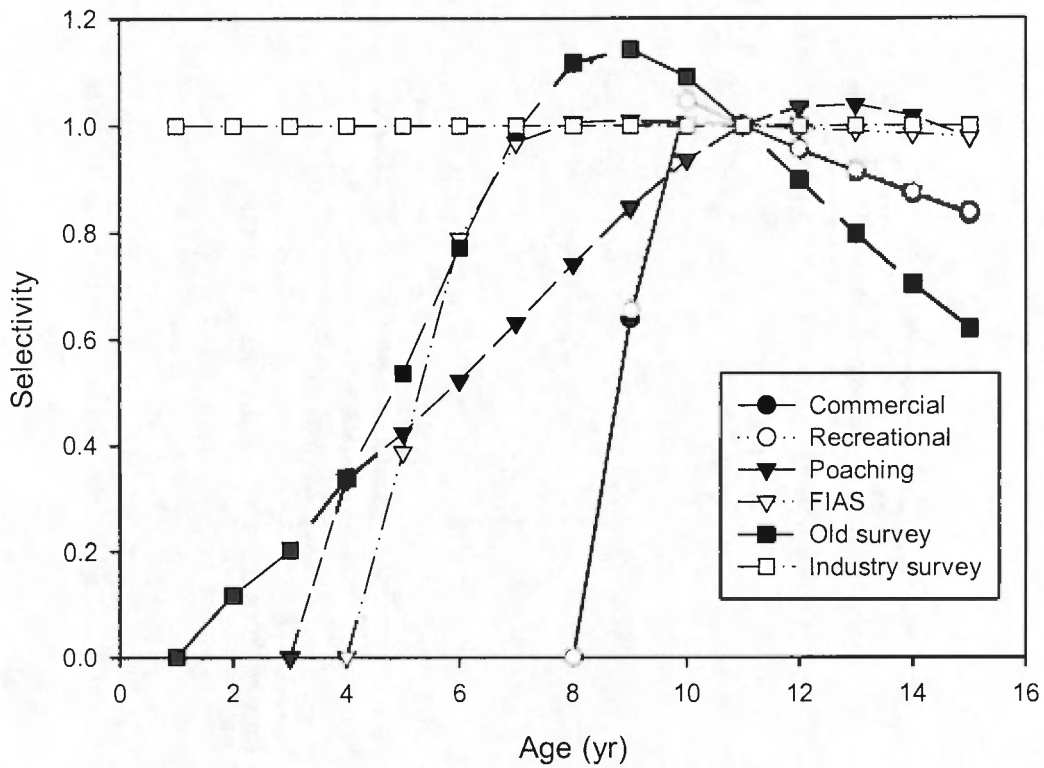


Fig. 7.14. Plots of the base-case combined ABCD model selectivity functions estimated for the commercial, recreational and poaching fishery sectors, and for FIAS and the old 1980's surveys. A description of the general functional form used is given in Appendix 6.1 and the fitted parameter values are listed in Table 7.3. A uniform value is assumed for the industry/MCM survey because of the extractive nature of the sampling methodology used. Note that the estimates for commercial and recreational are very similar, with the plot over-printing the recreational estimates.

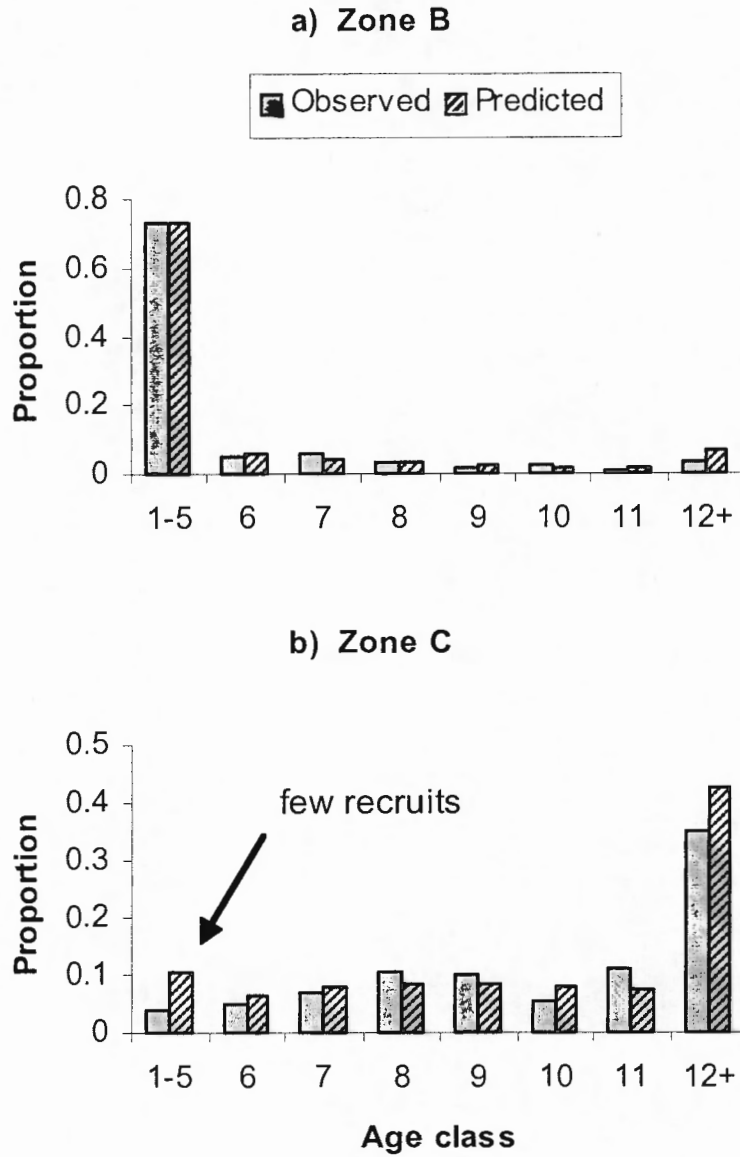


Fig. 7.13. Comparisons between (a) Zone B and (b) Zone C observed and model-predicted (base-case combined ABCD model) catch-at-age proportions (based on the number of abalone in each age class) corresponding to the Industry/MCM 2002 full population survey. The ages 1-5 were lumped together to reduce the number of categories containing a proportional abundance less than 2%. Note that a uniform selectivity function was assumed because of the extractive sampling methodology used.

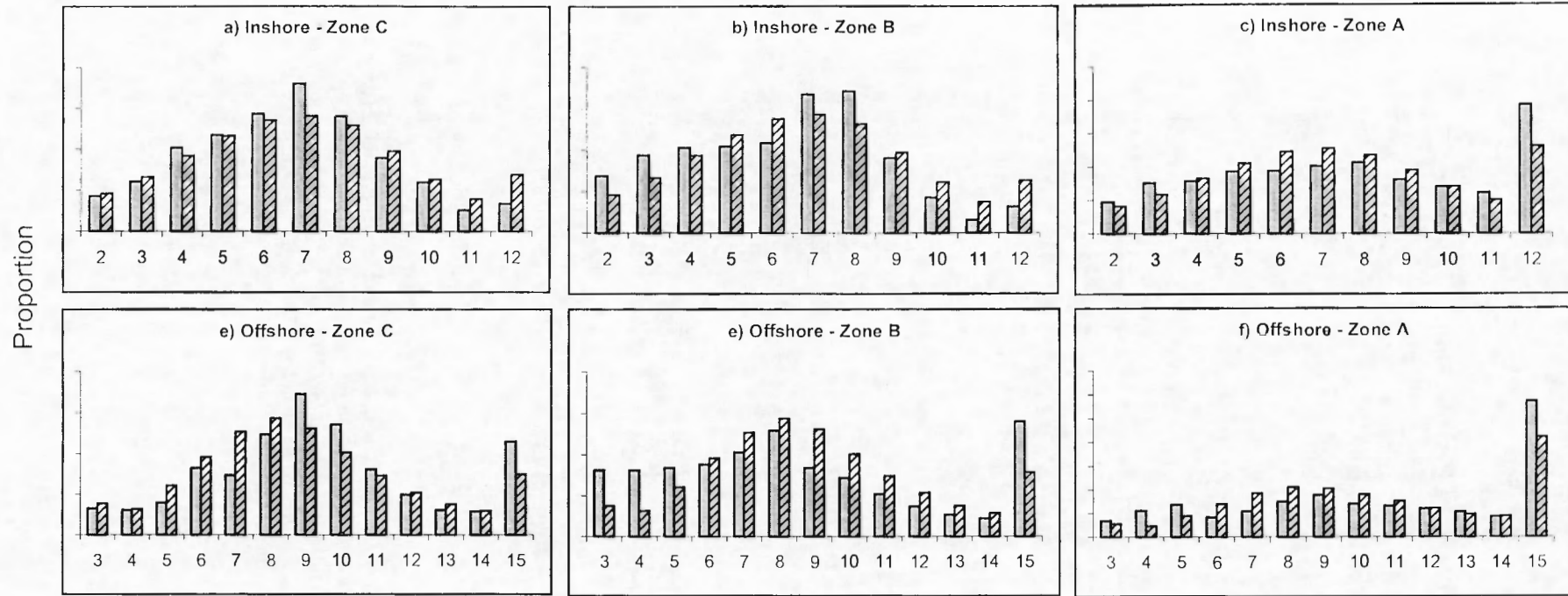


Fig.7.12 Comparisons between observed and model-predicted catch-at-age proportions for the “old” 1980’s survey data for the base-case combined ABCD model. The data for Zone C are from subarea CP. The data for Zone A have been averaged given that they are available for the two years 1986 and 1987.

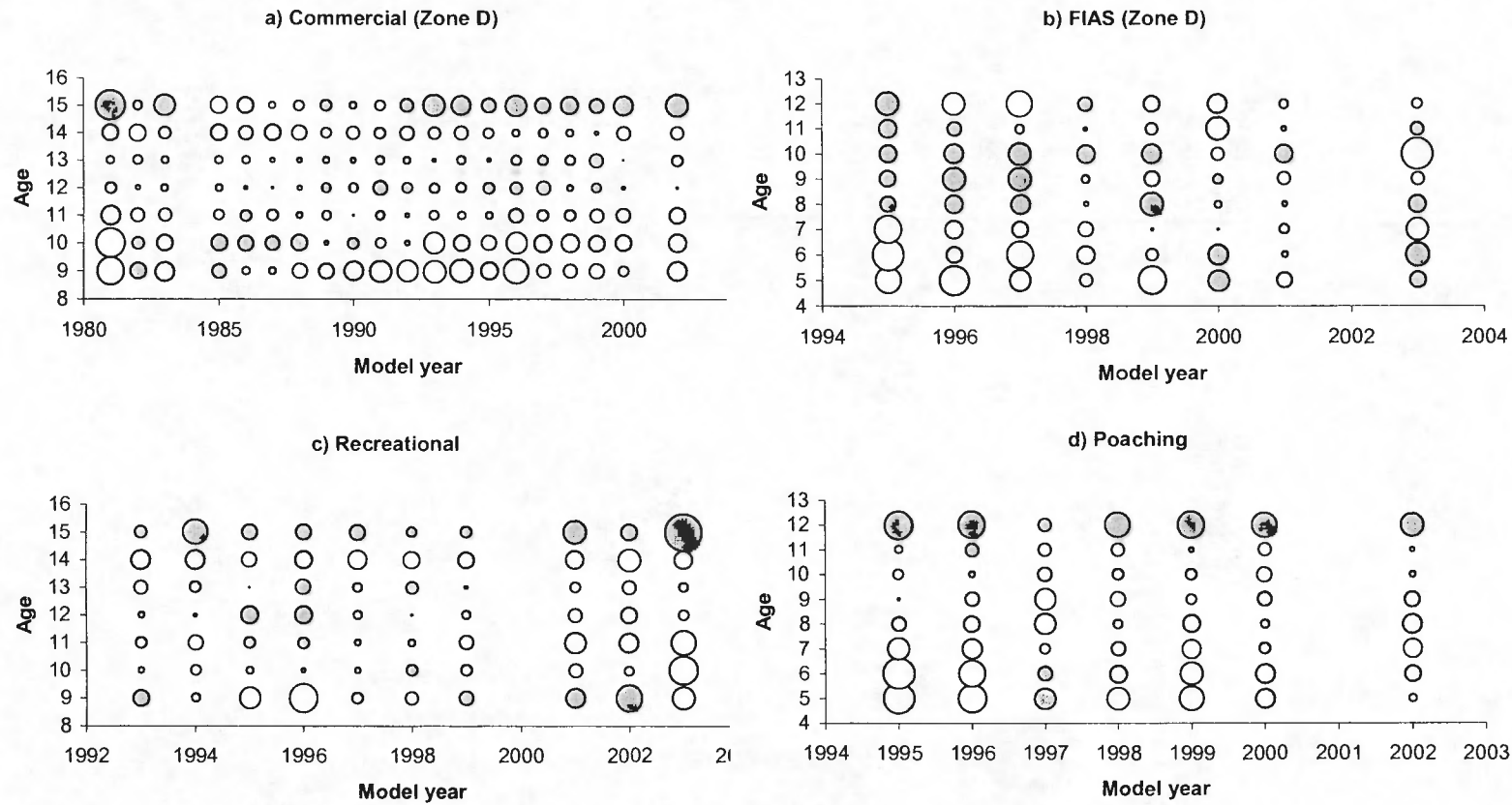


Fig 7.11. Catch-at-age residuals for Zone D for a) the commercial data, b) the FIAS data, c) the recreational data and d) the poached sector (based on confiscation data) for the base-case combined ABCD model. The size (radius) of the “bubble” in the plots is proportional to the corresponding standardized residual  $((\ln(\text{obs}) - \ln(\text{pred})) / (\sigma / \sqrt{\text{pred}}))$ . White bubbles represent negative residuals and grey bubbles represent positive residuals.

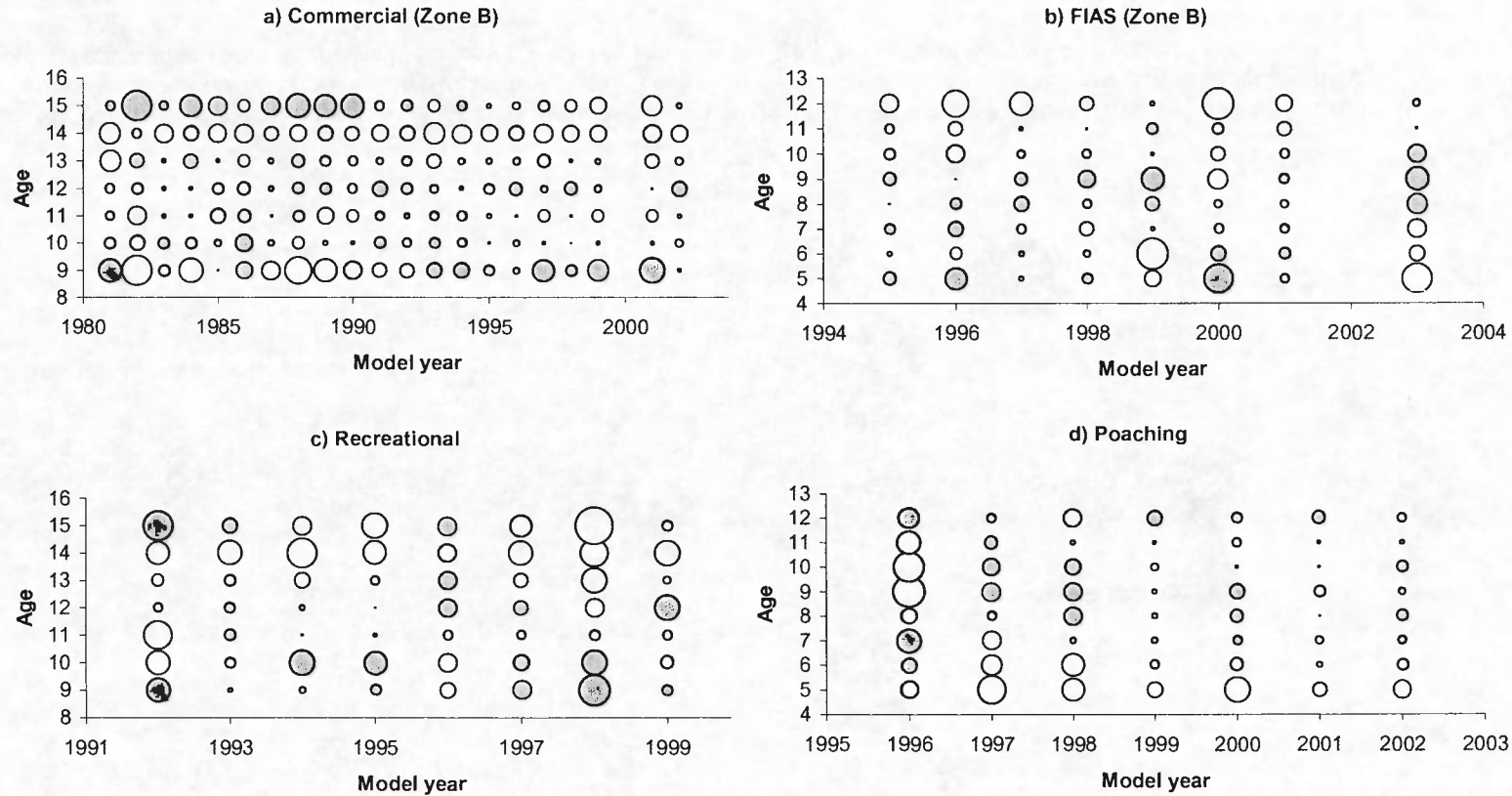


Fig 7.10. Catch-at-age residuals for Zone B for a) the commercial data, b) the FIAS data, c) the recreational data and d) the poached sector (based on confiscation data) for the base-case combined ABCD model. The size (radius) of the “bubble” in the plots is proportional to the corresponding standardized residual  $((\ln(\text{obs}) - \ln(\text{pred})) / (\sigma / \sqrt{\text{pred}}))$ . White bubbles represent negative residuals and grey bubbles represent positive residuals.

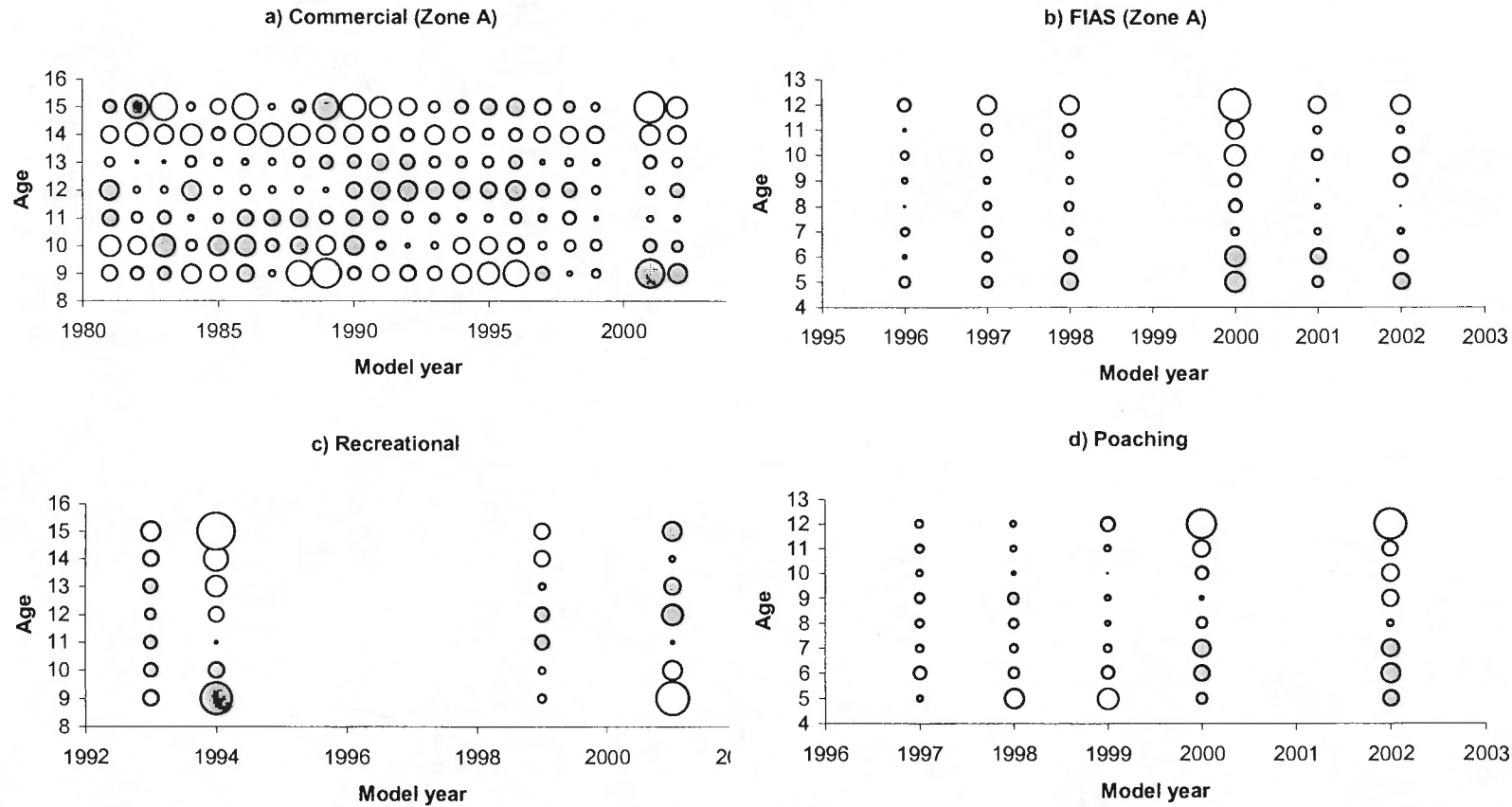


Fig 7.9. Catch-at-age residuals for Zone A for a) the commercial data, b) the FIAS data, c) the recreational data and d) the poached sector (based on confiscation data) for the base-case combined ABCD model. The size (radius) of the “bubble” in the plots is proportional to the corresponding standardized residual  $((\ln(\text{obs}) - \ln(\text{pred})) / (\sigma / \sqrt{\text{pred}}))$ . White bubbles represent negative residuals and grey bubbles represent positive residuals.

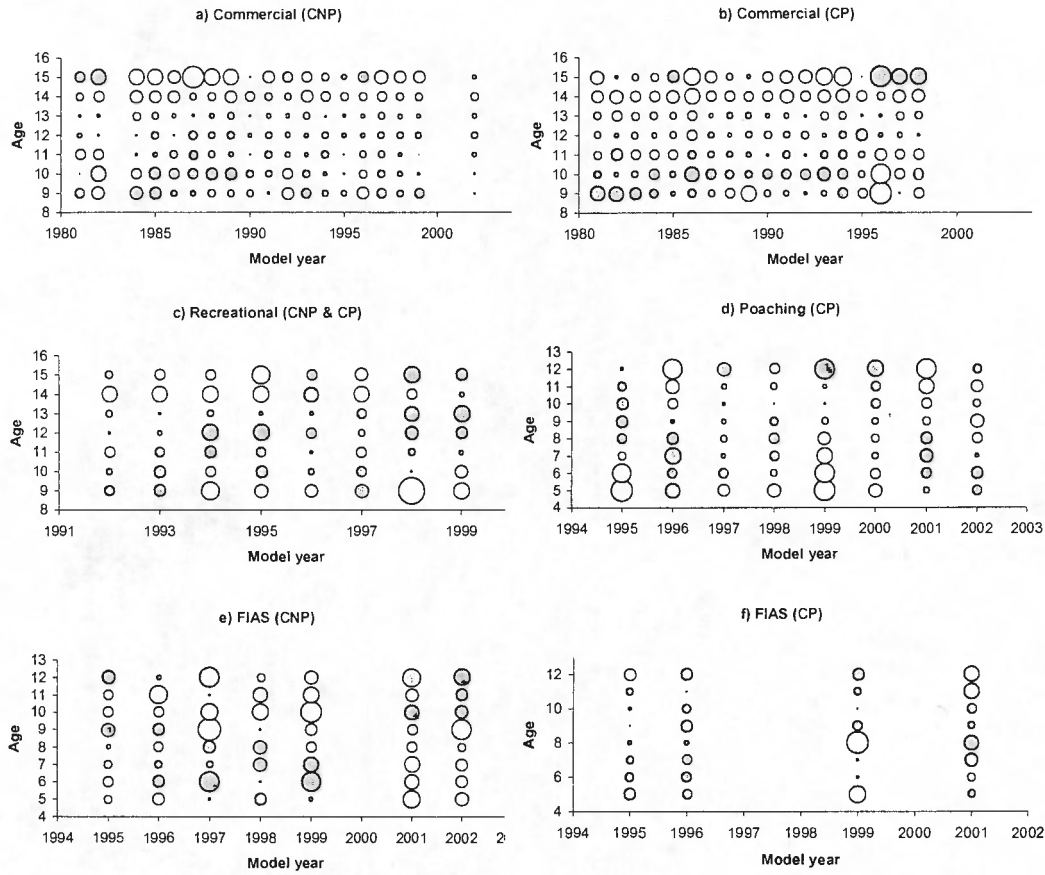


Fig. 7.8. Catch-at-age residuals for Zone C for a) the commercial data for subarea CNP, b) the commercial data for subarea CP, c) the recreational data (fitted to subareas CNP and CP combined), d) the poaching data (assumed to derive from subarea CP), e) the FIAS data for subarea CNP and f) the FIAS data for subarea CP for the base-case combined ABCD model. The size (radius) of the “bubble” in the plots is proportional to the corresponding standardized residual  $((\ln(\text{obs}) - \ln(\text{pred})) / (\sigma / \sqrt{\text{pred}}))$ . White bubbles represent negative residuals and grey bubbles represent positive residuals.



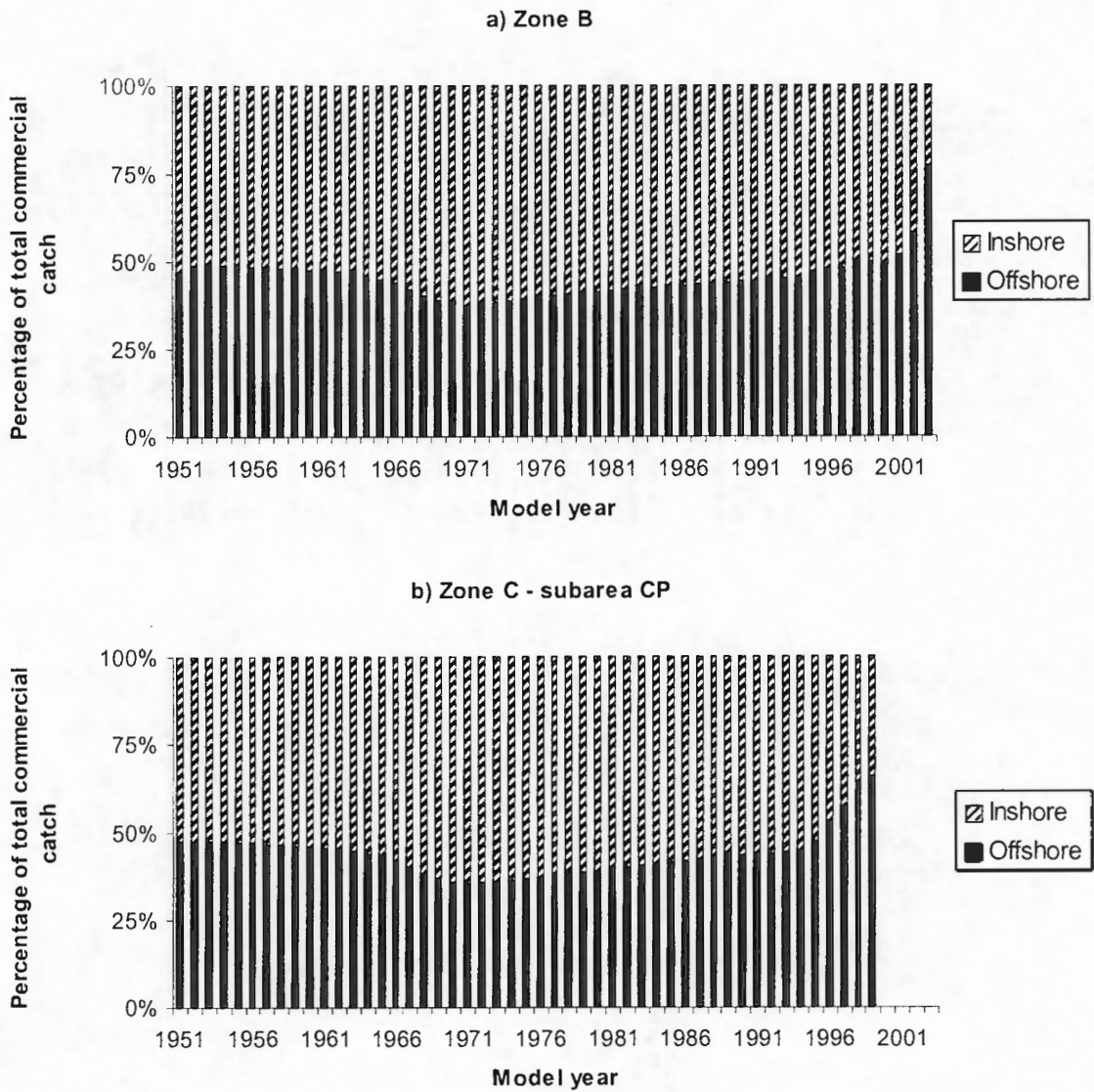


Fig. 7.7. Plots of the split of the commercial catches between the inshore and offshore areas predicted by the base-case combined ABCD model for a) Zone B and b) subarea CP of Zone C. For subarea CP, zero commercial catches have been allocated since 2000.



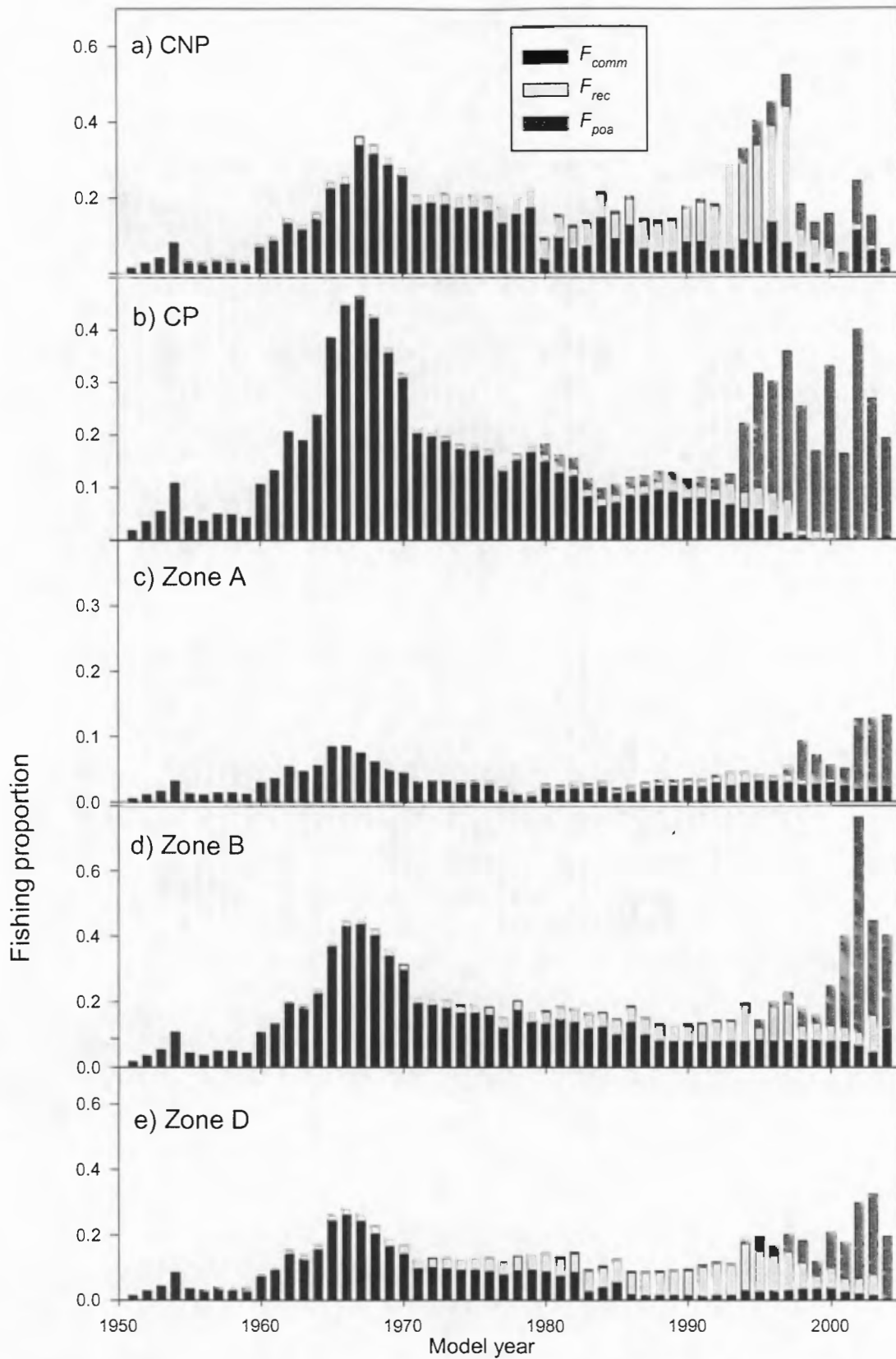


Fig. 7.6 Changes in the fishing proportion  $F$  over time for each of a) subarea CNP, b) subarea CP, c) Zone A, d) Zone B and e) Zone D. The figures show the contribution to the total fishing proportion by the three sectors: commercial (comm.), recreational (rec) and poaching (poa) for the base-case combined ABCD model.

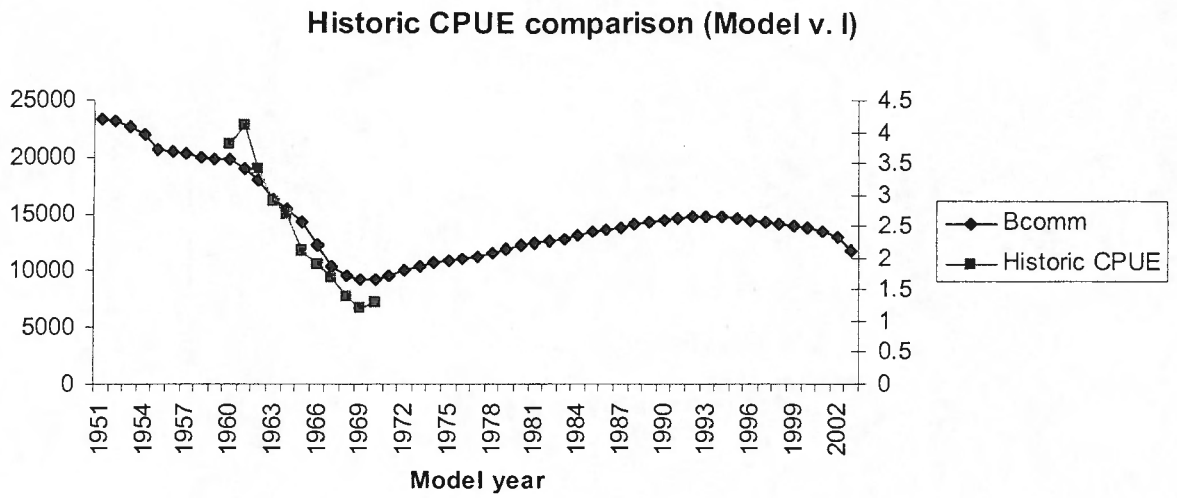


Fig. 7.5. Comparison of commercial exploitable biomass for Zones A, B, CNP, CP and D combined with historic “CPUE” data for the base-case combined ABCD model.

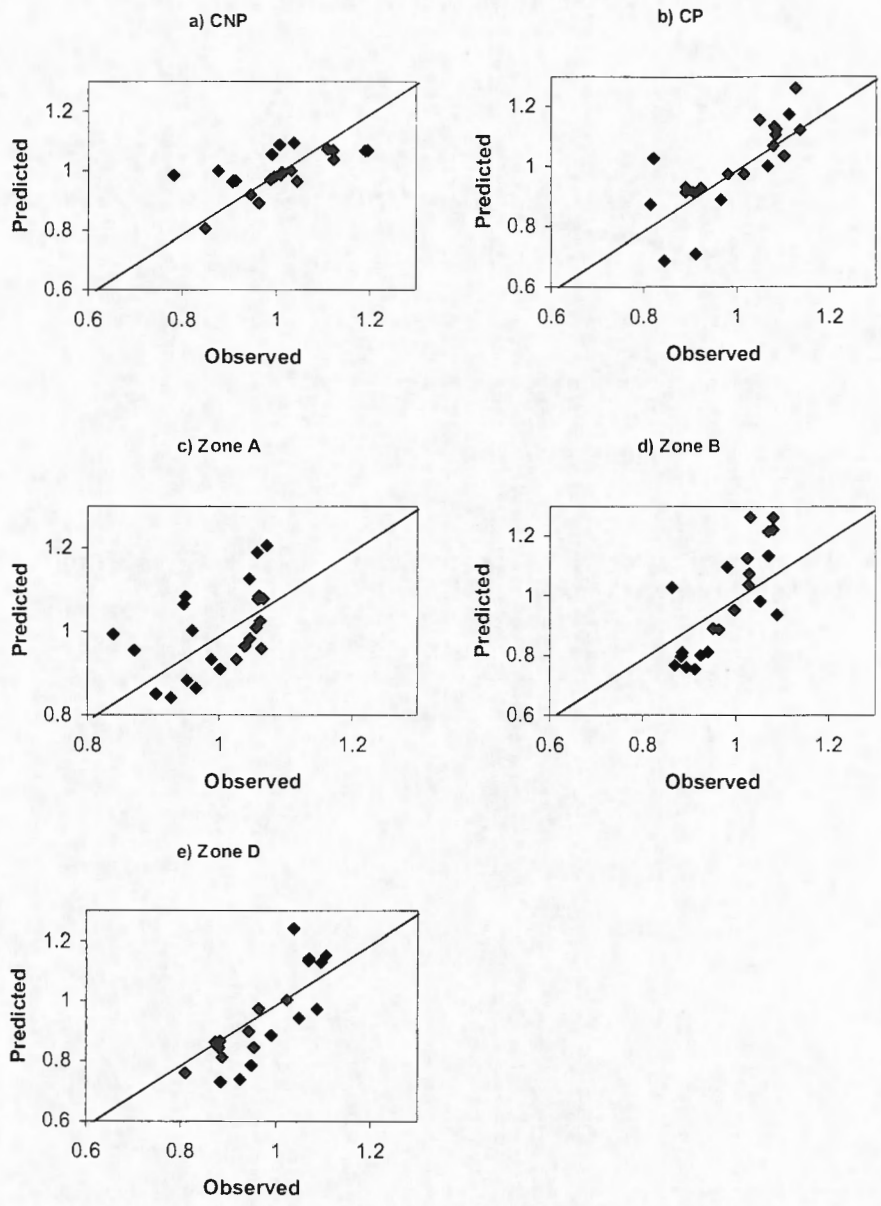


Fig. 7.4 Plots of model-predicted CPUE values (for the base-case combined ABCD model) against observed (standardised) CPUE for each of subareas CNP and CP, and for Zones A, B and D.

Chapter 7 – model results and projections

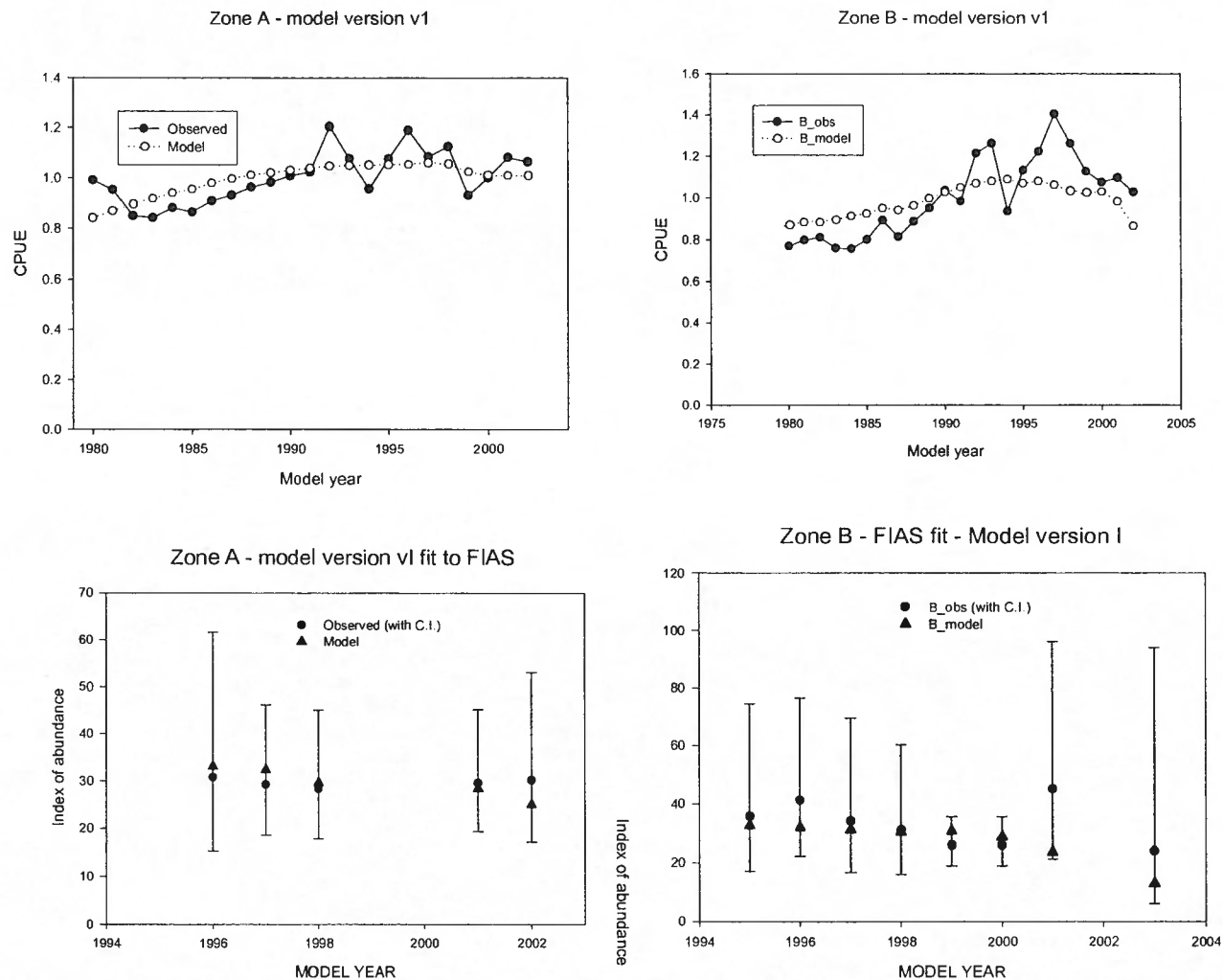


Fig.7.3 Comparisons between the standardised CPUE and model-predicted CPUE values (for the base-case combined ABCD model) for each of Zones A and B. The base-case model fit to the FIAS data is also shown. The 95% confidence intervals shown for the observed FIAS data were calculated as  $estimate * e^{\pm 1.96 * CV}$ .

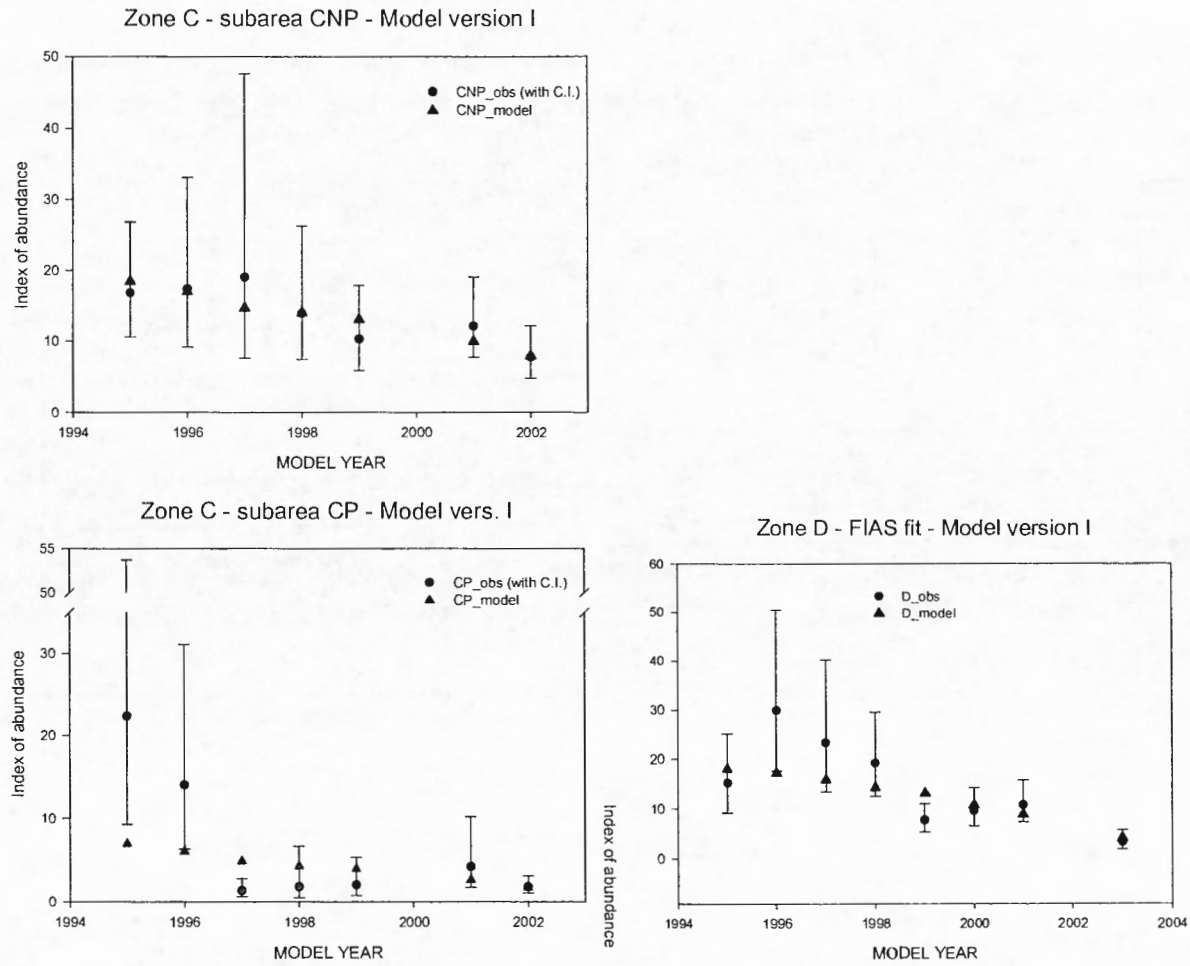


Fig. 7.2 Comparison of model-predicted (base-case combined ABCD model) and observed FIAS trends for each of subareas CNP and CP in Zone C and for the adjacent Zone D. Note that 95% confidence intervals have been computed as estimate\*exp( $\pm 1.96 * CV$ ). Note the break inserted on the y-axis for subarea CP for ease of viewing purposes (because it allows amplification of the rest of the figure).

maximum increase in mortality parameter  $M_{max}$  : for example, if  $y_{start} = 1985$ , then  $\nu = 0.134$  and  $M_{max} = 3.22$  ( $-\ln L = -204.98$ ); if  $y_{start} = 1990$ , then  $\nu = 0.225$  and  $M_{max} = 3.827$  ( $-\ln L = -205.58$ ); and if  $y_{start} = 1993$ , then  $\nu = 0.586$  and  $M_{max} = 4.07$  ( $-\ln L = -202.78$ ).  $y_{start}$  was thus set equal to 1990.

- For Zone B, no statistically significant improvements in fit resulted if the inshore-offshore migration parameter or recruitment proportion parameter were estimated instead of fixed as described.
- The effect of estimating the natural mortality parameter  $\mu$  (which essentially determines how different mortality rates are among age classes) was investigated. Under Poaching Scenario II, the best fit estimate of  $\mu$  was 0.000151 ( $-\ln L = -205.78$ ) and under Poaching Scenario III, the best fit estimate of  $\mu$  was 0.1294 ( $-\ln L = -207.36$ ).
- The  $M_{max}$  values estimated by the model are important because they are also used in projections of resource dynamics. Note that Combined B & C Model I estimated  $M_{max} = 3.86 \Rightarrow S_0 = 0.021$ . Based on the likelihood profile method, the lower 95% confidence limit for  $M_{max}$  is approximately  $M_{max} = 1.203 \Rightarrow S_0 = 0.3$  (cf pre-1990  $S_0 = 0.713$ ). An "optimistic" scenario (not presented in this thesis) was thus also considered in the 2002 assessment process, whereby consideration was given to projections effected assuming  $M_{max} = 1.203$ .

The "ecosystem-change" effect as determined from parameters estimated in the Combined B&C model was input to the combined Zones C and D model in 2002.

The following Model version v1 fits to the data were considered particularly unsatisfactory:

- i. The fit to the subarea CP FIAS catch-at-age data.
- ii. The fit to the subarea CP poaching sector, deep FIAS sector and "old survey" offshore sector catch-at-age data. In particular, the model underestimated the proportion of older animals represented in the inshore "poaching sector" catch-at-age samples and in the offshore 1980's "old survey" catch-at-age data. The poor fit to the CP deep FIAS data is not a concern as these data derive from a single sample of 16 animals surveyed in 1996.
- iii. For Zone B, the worst fit was to the "old survey" offshore sector catch-at-age data where the model underestimated the proportions of animals in both the youngest and oldest age classes.
- iv. The fits to the industry population structure survey data (Fig. A7.2) represented considerable improvements to those obtained using previous model versions, but were still not satisfactory. In Zone C, the model actually overestimated the proportions of animals in the youngest age classes because these animals were almost entirely absent from the survey sample. In Zone B the disproportionately large number of 1-yr old abalone was not reflected to the same extent by the model. Note that Model version v2 resulted in an improved fit to these data.

Reasons why the fits to the catch-at-age data (Fig. A7.3) were worse in subarea CP than in subarea CNP may be related to the following:

- i. Post-1966, commercial catches in CNP are assumed taken from the offshore model region only, whereas in CP they are assumed taken from both the inshore and offshore model regions (see Chapter 6 for details).
- ii. The method of fixing the CP inshore-to-offshore migration parameter relative to the CNP value needs further investigation. Note in particular that their ratio was selected on the basis that CP has a greater offshore extent of the shallow

“inshore” region than the other zones (see Chapter 6) and hence the proportion of recruitment that occurs inshore is also likely to differ from that assumed for the other zones.

Following discussions by the AWG and further investigations, the assumption that the same selectivity function applies to the inshore and offshore components of the FIAS and “old survey” sectors was considered adequate.

Table A7.1. Total numbers (including proportion of unknown confiscations) of confiscated abalone per year for Zones B and C. The Zone C numbers are compared to model-derived estimates obtained using combined Zones B and C Model version v1 (Poaching Scenario II). Model-derived poaching estimates are also shown in terms of mass and the last column gives the “total” Zone C numbers confiscated as a proportion of the corresponding model estimate of the total number poached.

**COMBINED B & C MODEL v1 (POACHING SCENARIO II)**

a) Zone C

Model year	Model estimated numbers poached from Zone C	Model estimated poached mass (MT) from Zone C	Confiscations by number KNOWN to be from Zone C	Confiscation proportions based on allocation-known confiscations	Zone C TOTAL observed confiscations by number	Confiscation proportions based on total number confiscated
1994	468070	263.0	10004	0.02	10004	0.02
1995	714611	397.0	15284	0.02	56522	0.08
1996	592412	319.5	12663	0.02	57405	0.10
1997	681024	358.7	16027	0.02	33482	0.05
1998	458780	233.6	10797	0.02	32982	0.07
1999	255831	130.6	6849	0.03	17842	0.07
2000	465212	244.2	12445	0.03	67510	0.15
2001	178652	94.2	4770	0.03	24193	0.14
2002	363737	199.9	9716	0.03	54300	0.15

b) Zone B

Model year	Model estimated numbers poached from Zone B	Model estimated poached mass (MT) from Zone B	Confiscations (by number) KNOWN to be from Zone B	Confiscation proportions based on allocation-known confiscations	Zone B TOTAL observed confiscations by number	Confiscation proportions based on total no. confiscated
1994	2926	1.6	419	0.14	419	0.14
1995	18423	10.1	2637	0.14	9752	0.53
1996	10563	5.8	1512	0.14	6854	0.65
1997	28704	15.6	4520	0.16	9443	0.33
1998	48711	26.3	7671	0.16	23433	0.48
1999	20471	11.1	3663	0.18	9542	0.47
2000	100199	54.7	17929	0.18	97259	0.97
2001	219678	119.9	39308	0.18	199366	0.91
2002	504460	274.3	90265	0.18	504460	1.00



Table A7.2. Model version fitting Zones B and C simultaneously. Best fit estimates of the pre-exploitation spawning biomass  $B_0^{sp}$  for Zone B and for the “poached” CP and “nonpoached” CNP areas of Zone C, the estimated natural mortality estimate  $M$ , the inshore-offshore migration parameters  $\rho$  ( $yr^{-1}$ ), the proportions of recruitment in each subarea that occur inshore versus offshore  $r_j$ , and the poaching maximum  $CP_{max}$  (i.t.o. NUMBERS), where the proportion of  $CP_{max}$  taken from area CNP is fixed at 0.1 in the base-case. Minimum values of the negative of the log-likelihood function are also shown. The estimated selectivity parameters are shown for the commercial sector (CS), recreational sector (RS), poaching sector (PS), FIAS (FS) and the old 1980's survey (OS). Note that for the 2002 industry survey (IS),  $S_a^{IS} = 1$ . Note also that all  $-lnL$  contributions from catch-at-age data have been multiplied by 0.1 as an *ad hoc* adjustment to compensate for likely positive correlation in these data.

## COMBINED B &amp; C MODEL VERSION v1

Number of parameters Poaching Scenario	FITTING B & C TOGETHER			
	23 II			
<u>Parameters</u>	<u>Estimate</u>	<u>Likelihood contributions - Zone B</u>	<u>-lnL</u>	<u>Sigma</u>
<b>Zone B</b>				
$B_0^{sp}$ (Zone B)	5425.15	-lnL CPUE [ $\hat{\sigma}^{CS}$ ] -lnL FIAS	-37.238 -7.240	0.112
fix $\rho$ (Zone B) = $\rho[CNP]$	-	-lnL age CS [ $\hat{\sigma}_c^{CS}$ ]	-14.368	0.078
fix $r_j$ (Zone B)=0.9	fix 0.9	-lnL age RS [ $\hat{\sigma}$ ]	-8.194	0.058
fix $CP_{max}$ (Zone B) (2002)	fix 504460 (272 MT)	-lnL age PS [ $\hat{\sigma}$ ]	-1.464	0.161
		-lnL age FIAS [ $\hat{\sigma}$ ]	-6.914	0.059
<b>Zone C</b>				
$B_0^{sp}$ [CNP]	2066.063	-lnL age DEEP FIAS [ $\hat{\sigma}$ ]	-1.070	0.171
$B_0^{sp}$ [CP]	4390.507	-lnL age OS inshore [ $\hat{\sigma}$ ]	-1.559	0.041
$\rho[CNP]$	0.0795	-lnL age OS offsh. [ $\hat{\sigma}$ ]	-0.807	0.080
$\rho[CP]$ - FIX = 0.5* $\rho[CNP]$	-	-lnL age IS insh+offsh. [ $\hat{\sigma}$ ]	-0.795	0.076
fix	fix 0.9	<b>Likelihood contributions - Zone C</b>		
$r_j[CNP]=r_j[CP]=0.9$		-lnL CPUE (CNP) [ $\hat{\sigma}^{CS}$ ]	-42.614	0.082
$CP_{max}$ (1995)	714610 (397 MT)	-lnL CPUE (CP) [ $\hat{\sigma}^{CS}$ ]	-45.439	0.057
Steepness of recruitment failure $v$	0.227	-lnL FIAS (CNP)	-7.877	
Rec_scal $M_{max}$	3.856	-lnL FIAS (CP)	8.562	



Table A7.2 continued

Number of parameters Poaching Scenario	<b>FITTING B &amp; C TOGETHER</b>			
	23 II			
<b>Common parameters</b>	<b>Estimate</b>	<b>Likelihood contributions - Zone B</b>	<b>-lnL</b>	<b>Sigma</b>
$M_a : \mu$ ( $\lambda = 0.2$ )	0.1384	-lnL age CS (CNP) [ $\hat{\sigma}_c^{CS}$ ]	-6.505	0.125
$\hat{a}(CS)$	9.00	-lnL age CS (CP) [ $\hat{\sigma}_c^{CS}$ ]	-12.384	0.082
$\hat{a}(RS)$	9.00	-lnL age RS [ $\hat{\sigma}$ ] (combined)	-6.547	0.066
$\hat{a}(PS)$	6.97	-lnL age RS (CP) [ $\hat{\sigma}$ ]	-0.279	0.144
$\hat{a}(FS)$	6.02	-lnL age PS [ $\hat{\sigma}$ ]	-1.437	0.156
$\hat{a}(OS)$	7.17	-lnL age FIAS (CNP) [ $\hat{\sigma}$ ]	-5.459	0.077
$\hat{a}(IS)$	-	-lnL age FIAS (CP) [ $\hat{\sigma}$ ]	-0.070	0.190
$\mu(CS)$	0.031	-lnL age DEEP FIAS (CNP) [ $\hat{\sigma}$ ]	-1.936	0.142
$\mu(RS)$	0.034	-lnL age DEEP FIAS (CP) [ $\hat{\sigma}$ ]	0.231	0.305
$\mu(PS)$	0.076	-lnL age OS inshore [ $\hat{\sigma}$ ]	-2.339	0.021
$\mu(FS)$	0.084	-lnL age OS offshore [ $\hat{\sigma}$ ]	-0.878	0.077
$\mu(OS)$	0.212	-lnL age IS insh+offsh. [ $\hat{\sigma}$ ]	-0.894	0.081
$\mu(IS)$	-	<b>-lnL (TOTAL)</b>	<b>-205.512</b>	
$\delta(CS)$	369.930			
$\delta(RS)$	606.190			
$\delta(PS)$	1.055	<i>q</i> values		
$\delta(FS)$	1.080	$\hat{q}^{CPUE}$ (Zone B)	0.000667	
$\delta(OS)$	0.778	$\hat{q}^{CPUE}$ [CNP]	0.002292	
$\delta(IS)$	-	$\hat{q}^{CPUE}$ [CP]	0.000884	
<b>DEPLETION STATISTICS - Zone B</b>	<b>Zone B</b>	<b>DEPLETION STATISTICS - Zone C</b>	<b>CNP</b>	<b>CP</b>
$B^{sp}(2002)/K$ (INSHORE+OFFSHORE)	0.56	$B^{sp}(2002)/K$ (INSHORE+OFFSHORE)	0.445	0.240
$B^{sp}(2002)/K$ (inshore)	0.61	$B^{sp}(2002)/K$ (inshore)	0.391	0.137
$B^{sp}(2002)/K$ (offshore)	0.49	$B^{sp}(2002)/K$ (offshore)	0.498	0.430
$B^{total}(2002)/K$	0.60	$B^{total}(2002)/K$	0.410	0.226
$B^{commercial}(2002)/K$	0.49	$B^{commercial}(2002)/K$	0.327	0.235
$B8+(2002)/K$	0.45	$B8+(2002)/K$	0.480	0.242
FIAS $N_{2002}/N_{1951}$	0.62	FIAS $N_{2002}/N_{1951}$	0.352	0.101
<b>1980's depletion diagnostic comparison - B</b>	1981/1982 observed Model		Observed	Model
Bsp Inshore (1982)/K	0.67	0.58	0.33	0.55
Bsp Offshore (1982)/K	0.54	0.35	0.24	0.34
	<b>1980's depletion diagnostic comparison - CP</b>			
	Zone C: Bsp Inshore (1981)/K			
	Zone C: Bsp Offshore (1981)/K			

Appendix – background model results

Table A7.3. Correlation matrix (output using AD Model Builder software) for a preliminary model constructed to simultaneously fit to Zones A, B, CNP, CP and D. Values greater than or equal to |0.70| are highlighted in bold.

index	name	value	std dev	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	
1	lnK	8.734	0.075	1.00																												
2	rho_scal	4.086	1.802	0.20	1.00																											
3	Mpar	0.120	0.016	-0.61	-0.05	1.00																										
4	lnK_CNP	7.691	0.117	0.04	-0.42	0.13	1.00																									
5	lnK_CP	8.467	0.059	0.47	0.03	-0.49	-0.11	1.00																								
6	rho_scal_CNP	7.365	2.302	-0.02	0.49	-0.11	<b>-0.73</b>	0.15	1.00																							
7	CPnum_scal_C	0.707	0.105	-0.38	-0.28	0.49	0.32	-0.21	-0.47	1.00																						
8	a_cs	9.000	0.060	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00																					
9	a_rs	8.999	0.079	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00																				
10	a_ps	15.028	0.828	0.03	-0.13	-0.20	0.04	0.15	-0.06	0.05	0.00	0.00	1.00																			
11	a_fs	5.691	0.732	0.07	-0.24	-0.01	0.16	0.04	-0.18	0.14	0.00	0.00	0.00	1.00																		
12	a_ss	4.915	1.778	0.24	0.10	-0.16	-0.01	0.14	0.03	-0.11	0.00	0.00	0.02	0.03	1.00																	
13	mus_cs	7.669	4.066	0.79	0.11	<b>-0.70</b>	0.03	0.56	0.00	-0.40	0.00	0.00	0.10	0.09	0.25	1.00																
14	mus_rs	11.541	6.036	0.35	-0.63	-0.30	0.45	0.28	-0.51	0.09	0.00	0.00	0.13	0.27	0.09	0.40	1.00															
15	mus_ps	376.6	3815.2	-0.01	0.04	0.01	-0.03	-0.01	0.04	0.00	0.00	0.00	-0.59	-0.01	-0.01	-0.02	-0.05	1.00														
16	mus_fs	13.594	8.087	0.20	-0.54	-0.14	0.44	0.13	-0.46	0.13	0.00	0.00	0.15	<b>0.79</b>	0.05	0.23	0.60	-0.04	1.00													
17	mus_ss	2.902	6.256	0.51	0.08	-0.46	0.02	0.36	0.00	-0.24	0.00	0.00	0.09	0.08	0.85	0.56	0.29	-0.02	0.17	1.00												
18	d_cs	563.5	68130.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00											
19	d_rs	541.0	74169.0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00										
20	d_ps	3.861	38.154	-0.01	0.04	0.01	-0.03	-0.01	0.04	0.00	0.00	0.00	-0.59	-0.01	-0.01	-0.02	-0.05	1.00	-0.04	-0.02	0.00	0.00	1.00									
21	d_fs	1.199	0.622	-0.04	0.17	0.01	-0.08	-0.02	0.10	-0.11	0.00	0.00	0.02	<b>-0.87</b>	-0.02	-0.06	-0.17	0.01	-0.64	-0.05	0.00	0.00	0.01	1.00								
22	d_ss	0.669	0.268	-0.06	0.02	0.06	-0.01	-0.05	0.00	0.03	0.00	0.00	-0.01	-0.02	<b>-0.81</b>	-0.07	-0.06	0.00	-0.04	-0.60	0.00	0.00	0.00	0.01	1.00							
23	Msteep	0.273	0.020	0.19	0.06	-0.26	-0.08	0.18	0.08	-0.38	0.00	0.00	0.04	-0.14	0.03	0.18	0.05	-0.01	-0.03	0.10	0.00	0.00	-0.01	0.14	-0.01	1.00						
24	Mmax	4.997	0.782	-0.01	0.00	0.01	0.00	-0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	-0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.02	1.00				
25	poaprop	0.296	0.234	-0.02	-0.19	0.07	0.28	-0.05	-0.63	0.54	0.00	0.00	0.00	0.12	-0.03	-0.04	0.21	-0.02	0.15	-0.02	0.00	0.00	-0.02	-0.10	0.01	-0.12	0.01	1.00				
26	lnK_A	9.591	0.332	0.27	0.08	0.13	0.14	0.14	-0.10	0.02	0.00	0.00	-0.08	0.13	0.18	0.32	0.16	-0.01	0.15	0.31	0.00	0.00	-0.01	-0.08	-0.05	-0.04	0.00	0.03	1.00			
27	Catmul	3.782	0.925	0.18	0.06	0.22	0.15	0.07	-0.11	0.06	0.00	0.00	-0.10	0.12	0.13	0.22	0.12	-0.01	0.12	0.21	0.00	0.00	-0.01	-0.07	-0.03	-0.06	0.00	0.04	<b>0.95</b>	1.00		
28	lnK_D	9.299	0.283	0.35	0.18	0.09	0.18	0.17	-0.15	0.04	0.00	0.00	0.03	0.14	0.22	0.35	0.26	-0.03	0.17	0.36	0.00	0.00	-0.03	-0.08	-0.03	-0.12	0.00	0.07	0.46	0.41	1.00	

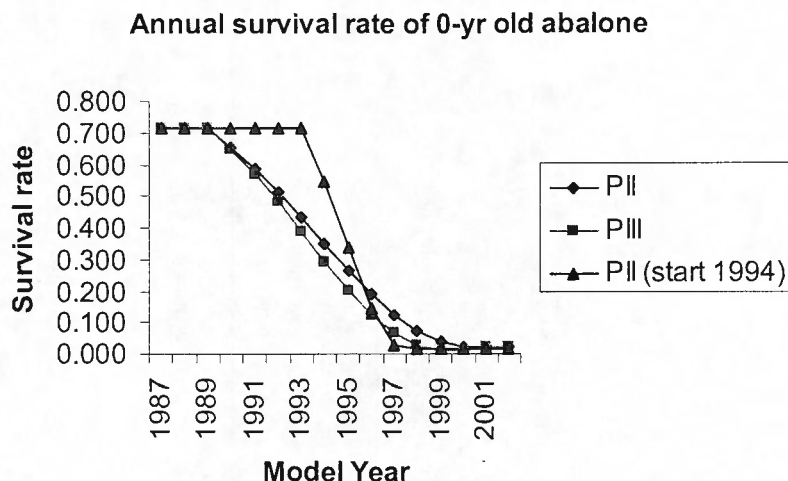


Fig. A7.1. Plot of 0-yr old abalone survival rate using base-case combined B&C model v1 to estimate the rate and extent of decline in survival rates after 1989 under Poaching Scenarios II and III. The estimated trend is also shown for a model scenario in which survival rates are allowed to decrease only post-1993 (and Poaching Scenario II is used). Note that in the latter case the decline is steeper and the current estimated survival rate is lower.

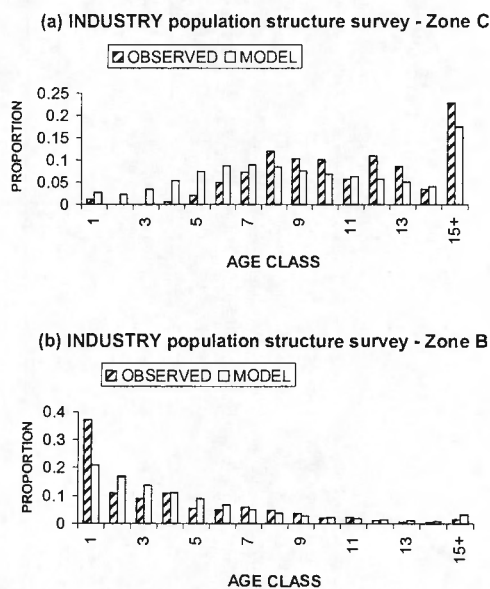
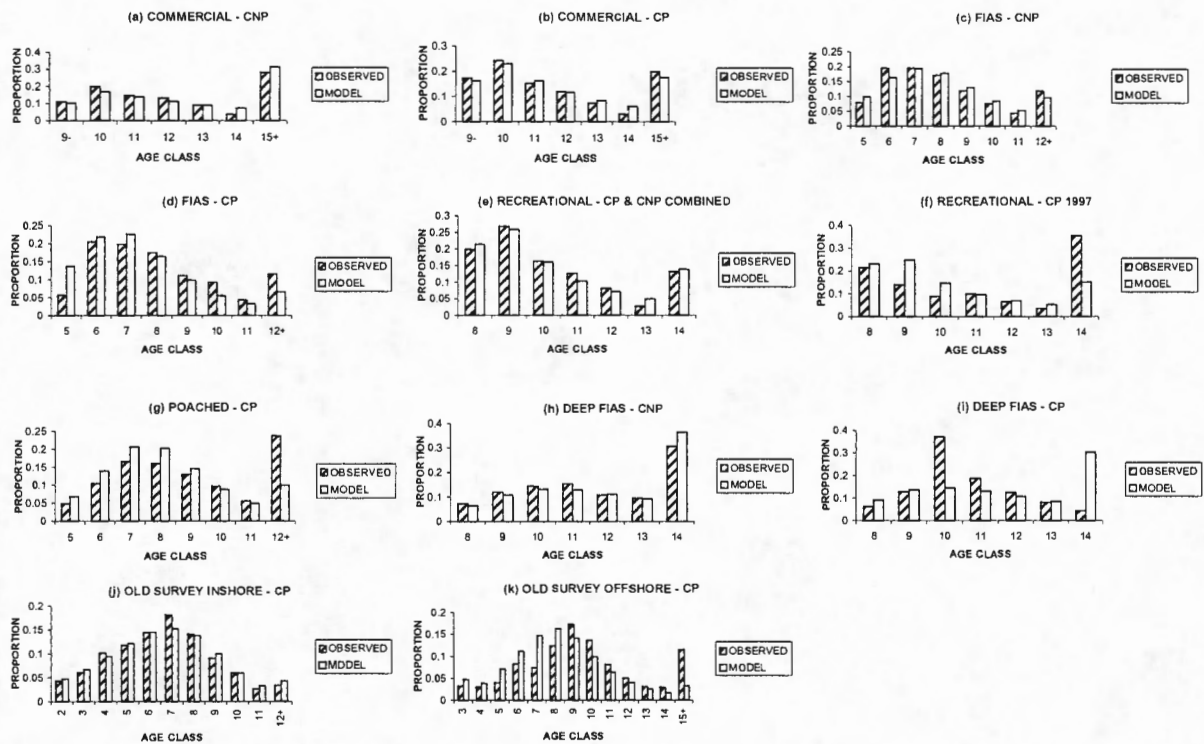


Fig. A7.2. Comparisons between (a) Zone C and (b) Zone B observed and model-predicted (model version v3) catch-at-age proportions corresponding to the Industry/MCM 2002 full population survey. Note that a uniform selectivity function was assumed because of the extractive sampling methodology used.

(a) Comparisons between Zone C observed and model-predicted catch-at-age proportions averaged over all years for which data was available



(b) Comparisons between Zone B observed and model-predicted catch-at-age proportions averaged over all years for which data was available

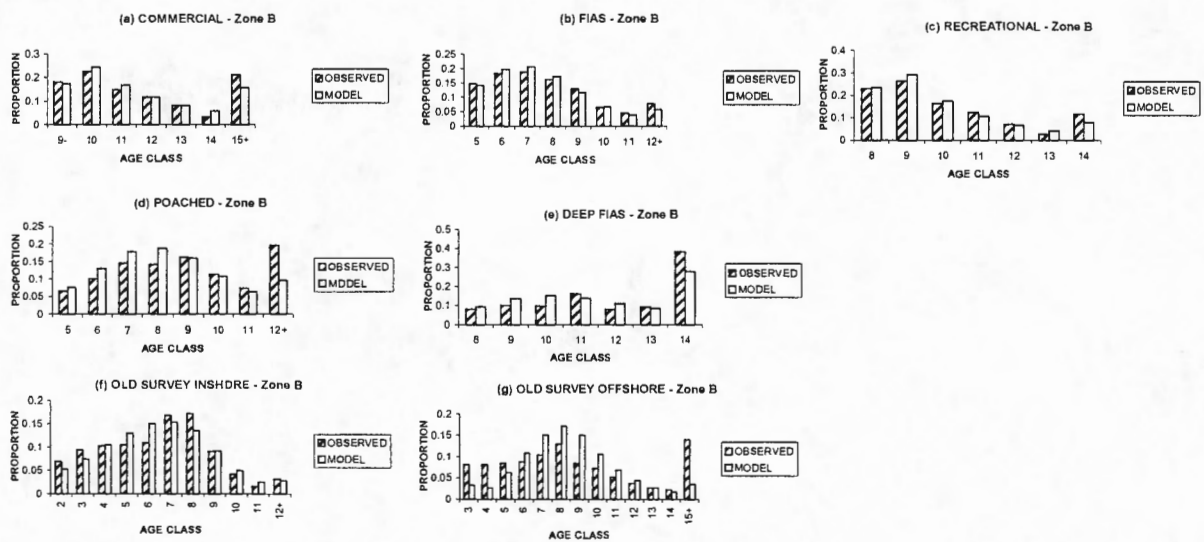


Fig. A7.3 Comparisons between a) Zone C and b) Zone B observed and model-predicted proportions averaged over all years for which data were available. Results shown are for the 2002 combined B&C model.



## Chapter 8

### **A first look at the potential pattern of recovery of abalone in Zone C in response to the removal of west coast rock lobsters from the East of Hangklip (EoH) region**

#### **8.1 INTRODUCTION**

This chapter describes some preliminary analyses that were done in 2002 in response to proposals regarding a possible experimental take of rock lobsters from East of Hangklip (EoH) to advantage abalone. Johnston (2002a) projected the EoH rock lobster population forward under three future constant catch scenarios. Her analysis suggested, for example, that a future annual commercial catch of 500 MT (i.e. total removals of 900 MT) would have the effect of reducing the biomass of lobsters above 75 mm down to 18% of the 2002 estimate by 2008. The extent to which this level of removal of lobsters would advantage the abalone resource is not known and is difficult to predict.

In weighing the relative advantages and disadvantages of embarking on a multi-species management approach, it is important to try to assess what the effects on the EoH abalone resource may be. As abalone are a relatively long-lived species, the potential benefits need to be assessed over the long rather than the short term. The age-structured production model (ASPM) currently used to model the abalone resource in Zones A-D is therefore useful for exploring various resource recovery scenarios. This document presents a first look at possible patterns of resource recovery and summarises some points to be borne in mind in debating the trade-offs of embarking on a multi-species management strategy.

#### **8.2 BACKGROUND**

The best current hypothesis to explain the recruitment failure effect in Zones C and D proposes that rock lobsters have increased substantially in these zones and are to be blamed for the recorded collapse of urchin populations and dramatic reductions in the numbers of juvenile abalone (Mayfield and Branch 2000). Based on this, the recruitment failure effect has been incorporated in the ASPM by allowing for an increase in the natural mortality rates of juvenile abalone (specifically 0-year old abalone recruits) in the model as follows:



1. Natural mortality  $M_a$  rates ( $a$  = age) are modelled as age-dependent. The 2002 base-case estimate of  $M_0$  was 0.338 (which corresponds to an annual survival rate of 0.713).
2. To model the rate and extent of the “recruitment failure” effect, two parameters are introduced: a rapidity of recruitment failure parameter  $\nu$  and a maximum increase in mortality parameter  $M_{max}$ . In the base-case model, an exponential increase in the  $M_0$  mortality rate is assumed to have occurred from 1990 (approximately the time at which substantial increases in lobster east of Hangklip were noted), with the rate of increase in  $M_0$  determined by parameter  $\nu$ .  $M_0$  increases continuously up to a maximum value  $M_{max}$  and then remains constant at this value from years  $y_{Mmax}$  forwards. In the 2002 base-case model,  $\nu = 0.227$  (estimated),  $y_{Mmax} = 2000$  and  $M_{max} = 3.856$  (estimated). This suggests that currently only 2.1% of 0-yr old abalone survive from one year to the next.
3. The mortality parameters and time trend in the 0-yr old mortality rate are estimated through fitting to both research survey and fishery-based indices of abundance and catch-at-age data. One potential problem in trying to model abalone population dynamics involves the confounding of natural mortality, the recruitment failure effect and the impact of high levels of poaching (whose magnitude is not well known) on the resource. This has been resolved to some extent by simultaneously fitting the model to Zone B (no recruitment failure effect) and Zone C (recruitment failure effect) (see Chapter 7). This was possible because of a 2002 directed MCM and Industry co-operative full population survey conducted in Zones B and C.

### 8.3 METHOD

As a first look at the potential recovery of abalone in response to fairly large reductions in rock lobster biomass having been effected by 2008, two scenarios were simulated:

- I. A linear increase from 2002 to 2008 in juvenile abalone survival rate in Zone C is assumed to occur from the current estimated value of 2.1% up to 50% of the pre-1990 survival rate of 71.3%. From 2008 onwards, the juvenile survival rate remains constant at 35.6% per annum;
- II. As above, except that a 100% recovery in juvenile survival rate is assumed as from 2008 (although unlikely, this extreme scenario is included to give some sense of the maximum resource recovery that could be seen). Note that lobster removals

*might* be differentially *more* effective if lobsters in the shallower water are targeted (*and* lobsters don't move around too fast to compensate!)

Results are presented for Zone C (subareas CNP and CP combined). In projecting forwards, it is assumed that the 2002/2003 commercial take from CNP is 30 MT, and is reduced to zero for later years. Subarea CP is assumed closed to fishing over the entire 20-yr projection period and zero recreational catches are assumed taken from Zone C over the entire projection period. Zone C poaching catches are assumed to remain constant (i.t.o. numbers) at the current estimated level.

#### 8.4 RESULTS

Table 8.1 summarises projected depletion statistics in terms of abalone spawning biomass (the statistic of most interest in a biological context) and numbers available to the FIAS sector (because the survey index would be most sensitive to changes in recruitment).

Graphic representations of the projected abalone population under the two scenarios are shown in terms of total spawning biomass (Fig. 8.1) and inshore numbers of abalone (Fig. 8.2). Assuming a constant future poaching catch of 369500 abalone/yr from Zone C, Fig. 8.1 shows the projected poaching catch in terms of biomass.

Table 8.2 summarises projected depletion statistics in terms of abalone spawning biomass under three different future catch scenarios and assuming the 50% recovery scenario.

#### 8.5 DISCUSSION

The projections are useful in highlighting the timescale necessary before any measurable recovery in the abalone resource occurs. Under the assumption of a linear increase in survival rates, there is approximately a 10-yr time lag before the current projected demise of the resource is reversed (Table 8.1 and Figs. 8.1 & 8.2). Under the 50% recovery scenario, the resource is predicted (assuming future catches remain at current levels) to stabilise at approximately 37% of the current level in terms of spawning biomass.

Given the argument by some sectors that the decline of the abalone resource is inevitable (due to the rock lobster effect) so that catches from the affected zones should be increased



rather than decreased, an initial investigation was undertaken as to what the potential effects on stock recovery may be of further depressing stock levels through continued harvesting. Preliminary results (Table 8.2) suggest that if takes from Zone C by all sectors were reduced to zero, under the 50% survival recovery scenario the stock is predicted to recover to the 2002 level over a twenty-year period. At the other extreme, if poaching in Zone C continues at the current estimated level in combination with a fixed future commercial catch of 30 MT/yr from subarea CNP, the resource is predicted to decrease to 20% of the 2002 level over the next twenty years. This corresponds to a spawning biomass of approximately 5% of pristine levels. It is not unreasonable to assume that an Allee effect will almost certainly operate at such low stock levels of a broadcast spawning species, especially given the relatively sessile nature of these organisms. These results highlight the importance of guarding against further stock reductions if there is some hope that the stock decline can be reversed by removing rock lobsters.

The following needs to be borne in mind in interpreting these results:

1. The assumption of a linear increase in survival rates was considered the most prudent approach given that there are so many other unknown factors. A sigmoidal curve would probably be more biologically realistic, but would require the estimation of another parameter.
2. The extent to which the current situation is reversible is not known. Removal of rock lobsters will almost certainly reduce any direct predation effects on the abalone, but there will be a lag before urchin populations are able to recover to sufficient numbers to provide cover for juvenile abalone. The recovery of the urchin populations is presumably in turn impeded by the increased sedimentation that has occurred in their absence. Moreover, the increased sedimentation may not be easily reversed, and this may have both a direct and indirect effect on abalone recruitment success.
3. Both inshore-offshore and longshore movements of rock lobsters will influence the success of a rock lobster removal project.
4. From a purely biological perspective, it seems unlikely that a complete reversal (such as is implied by the 100% recovery scenario simulated here) of the rock lobster effect could be achieved. Before deciding to “wipe out” the EoH rock lobster resource, some indication is needed as to the relative economic gains from the two resources as well as the extent to which the abalone resource is likely to recover in the long term. This also needs to take into account the fact that Zones A and B may similarly be threatened by

rock lobster invasions, particularly if the abundance of rock lobster EoH increases further.

From a modelling perspective, these multi-species interactions are difficult to incorporate explicitly in a model because some of the underlying processes are still speculative and there are insufficient data to quantify the interactions. As a first step, data would need to be collected to shed light on inshore and longshore EoH rock lobster migration rates. Marine and Coastal Management (MCM) are currently co-ordinating a tagging program that may provide the necessary data. Furthermore, the model itself could be used to identify which types of data are likely to be most informative, and especially to explore the potential for spatial adaptive management (e.g. Walters 1986, Hilborn and Walters 1992, Sainsbury *et al.* 1997).

A final point that was reiterated in 2002 was that abalone populations in Zones C and D are currently depressed relative to pristine levels and are continuing to decline. If management action is going to be taken to reverse this process, this needs to happen fast before the spawning stock biomass is reduced to such a low level that there is a danger of an Allee effect kicking in. The extent to which reseedling of these areas could occur due to larval drift from Zones A and B is not currently known. On the other hand, if it seems likely that the decline in the abalone population is irreversible, then effort should be focused instead on managing the rock lobster EoH on a sustainable basis.

The following chapter describes a simple multi-species model that could be used as an exploratory tool.

Table 8.1. Zone C depletion statistics for the base-case model scenario and for two scenarios assuming a recovery in juvenile abalone survival rates.

	Base-case (no recovery)	50% recovery by 2008 in juvenile abalone survival rate	100% recovery by 2008 in juvenile abalone survival rate
$B_{2005}^{sp} / B_{2002}^{sp}$	0.62	0.62	0.62
$B_{2008}^{sp} / B_{2002}^{sp}$	0.40	0.41	0.42
$B_{2022}^{sp} / B_{2002}^{sp}$	0.06	0.37	0.88
FIAS $N_{2005} / N_{2002}$	0.30	0.30	0.30
FIAS $N_{2008} / N_{2002}$	0.05	0.09	0.10
FIAS $N_{2012} / N_{2002}$	0.03	0.41	0.85
FIAS $N_{2021} / N_{2002}$	0.01	0.74	2.22

Table 8.2. Zone C depletion statistics for the 50% survival recovery scenario under different future catch scenarios. Scenario I assumes that poaching remains at the current level (referred to as H = High poaching scenario) and that a commercial catch of 30 MT is taken from CNP in the 2003 model year, but that this is reduced to zero thereafter. Scenario II looks at the effect on the resource of setting a constant commercial catch of 30 MT/yr (from CNP) over the entire projection period. Scenario III assumes zero future commercial and poaching catches.

	I 50% recovery by 2008 in juvenile abalone survival rate	II Future commercial catch (CNP) = 30 MT/yr ; Poaching = H	III Future commercial catch (CNP) = 0 MT/yr ; Poaching = 0
$B_{2005}^{sp} / B_{2002}^{sp}$	0.62	0.58	0.81
$B_{2008}^{sp} / B_{2002}^{sp}$	0.41	0.33	0.62
$B_{2022}^{sp} / B_{2002}^{sp}$	0.37	0.20	0.99

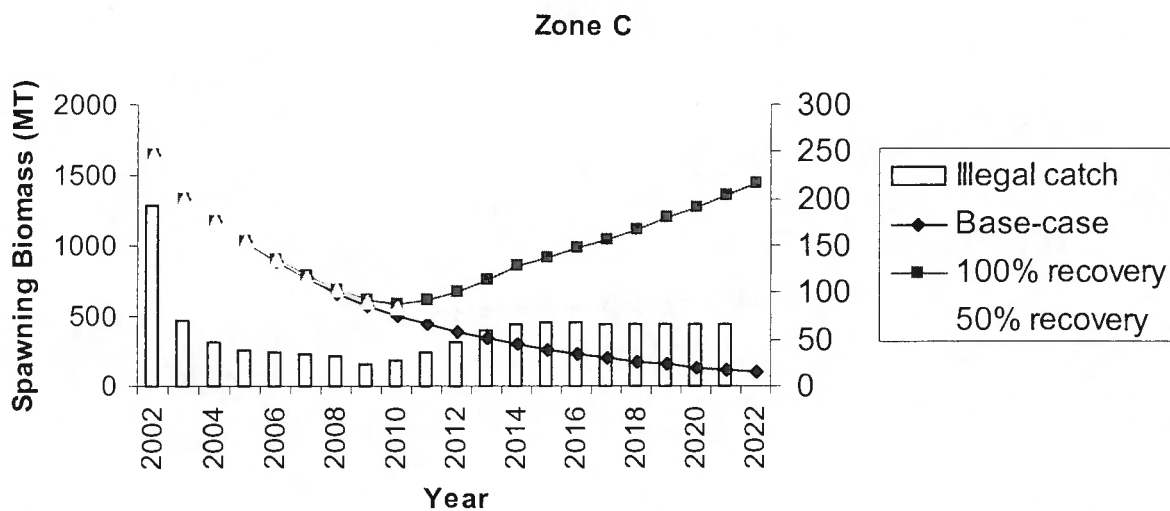


Fig. 8.1. Spawning biomass projections under three different future recruitment scenarios. The second axis shows the estimated future poaching catch of 369500 abalone/yr when converted to biomass using the 50% recovery scenario.

### Zone C model-predicted FIAS

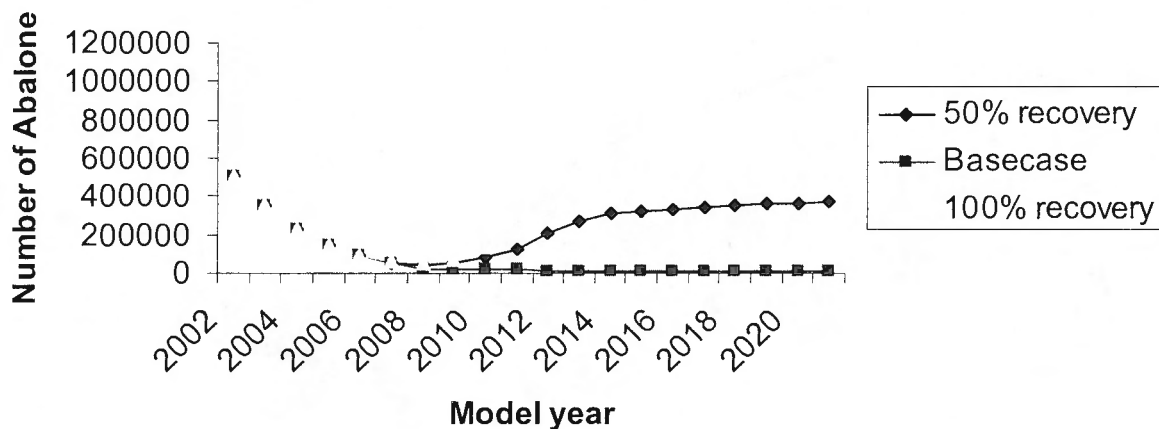


Fig. 8.2. Model-predicted numbers of abalone “available” to FIAS in Zone C under three different future harvesting scenarios.



## Chapter 9

### Preliminary development of an exploratory abalone-urchin-lobster multi-species model

#### SUMMARY

A preliminary abalone-urchin-lobster multi-species model has been constructed as a tool to explore proposed multi-species management scenarios in the East of Hangklip (EoH) region. The model extends on the base-case abalone age- and spatial-structured population model as follows:

- i) A discrete logistic equation is added to model the dynamics of urchins, under the assumption that the biomass of urchins consumed by rock lobsters can be described using a Holling type II interaction term;
- ii) Two discrete logistic equations with added inshore/offshore and longshore migration terms are used to model the inshore and offshore components of the EoH rock lobster resource;
- iii) Below a threshold urchin density, the mortality rate of 0-yr old juvenile abalone is assumed to increase exponentially with decreasing urchin biomass; and
- iv) The consumption of abalone by lobsters is accounted for by including rock lobsters as an additional “fishing fleet” in the abalone population model.

The model highlighted the difficulties in parameterising even a simple multi-species model. A preliminary sensitivity analysis for abalone spawning biomass indicated that model predictions are particularly sensitive to a) the urchin growth rate parameter, b) the urchin-lobster interaction parameter, c) the long-shore lobster migration rate and d) the initial (1990) lobster biomass level assumed. An illustrative application of the model suggested that the predicted recovery of the abalone resource may be slower than that predicted using a single species modelling approach only.

#### 9.1 INTRODUCTION

As outlined in the previous chapter, there are a number of difficulties associated with trying to construct a model that explicitly incorporates the various multi-species interactions thought to be operating in the East of Hangklip (EoH) area. Moreover, in Chapter 11 it is argued that approaches such as ECOSIM (Walters *et al.* 1997 – see Section 2 of thesis) have limited utility in trying to address questions related to the multi-species interactions in this system, because *inter alia*, the equilibrium assumptions of ECOPATH are too constraining in this context, there is insufficient flexibility in the ECOSIM inbuilt species interaction terms to account for, in particular, the urchin-abalone interaction, and there is insufficient data for



accurate specification of the large number of parameters required. Nonetheless, given discussions in 2002 regarding an experimental take of EoH lobsters to advantage abalone, there is a need for a simple multi-species model to assist in understanding the complexity of the dynamics involved. This chapter briefly summarises one such possible model. The principle aims of this analysis are as follows:

1. to construct the simplest possible multi-species model that adequately captures the main features of the EoH abalone-urchin-lobster interactions; and
2. to construct the model in a “minimally realistic” manner, given the data limitations, and thereby to explore what the minimum number of parameters is that need to be estimated, either within the model or by direct measurement from field studies.

The main processes that need to be captured by a model include the following:

1. the negative effect of current and possibly increasing rock lobster (*Jasus lalandii*) numbers on urchin biomass;
2. the (possible) negative effect of such rock lobster numbers on juvenile abalone (due to the direct predation effect);
3. the negative effect of decreasing urchin biomass on juvenile abalone survival rates; and
4. the effect of lobster removals (by the fishery) on lobster numbers in the inshore habitat area where they have the greatest effect (both direct and indirect) on abalone survival rates.

Day and Branch (2002) explored the interactions between the urchin *Parechinus angulosus* and both recruits and juveniles of the abalone *Haliotis midae*, in the light of the fact that juvenile abalone shelter beneath the urchins. They conducted field experiments at a number of sites and made comparisons between undisturbed and disturbance-control plots in which urchins were eliminated. They found that juvenile abalone declined dramatically in urchin-removal plots. Contrary to their expectations, they also found that abalone recruits (recent settlers <3 mm in length) disappeared from the urchin-removal plots, presumably in response to the fact that the cover of sediment almost doubled there. The effect of increased sedimentation rates in possibly impeding the recovery of urchin populations and /or abalone recruitment success is beyond the scope of this analysis and is not explicitly included in this model.

The model developed here is based on the assumption that the underlying causal mechanisms responsible for the observed abalone recruitment failure effect in Zones C and D are as described in Tarr *et al.* (1996), Mayfield and Branch (2000) and Day and Branch (2002), *viz* that juvenile abalone gain protection from predators by sheltering beneath urchins and may also benefit from an enhanced food supply due to bottom-up subsidization of urchins by drift kelp and that a dramatic decrease in urchin numbers was observed in the EoH area during the 1990's due to a movement of west coast rock lobsters (which prey on urchins and abalone) into the region.

## 9.2 MODEL

The model is applied in the first instance to Zone C (including both subareas CNP and CP) and is basically an extension of the 2002 base-case Zone C model (with specifications as given in Appendix 6.1).

### 9.2.1 Modelling urchin dynamics

A discrete logistic equation is applied to model the dynamics of the sea urchin *Parechinus angulosus* as follows:

$$u_{y+1} = u_y + r_u u_y \left( 1 - \frac{u_y}{K_u} \right) - \frac{\alpha_{u-l} u_y J_y^I}{1 + \beta_{u-l} u_y} \quad (9.1)$$

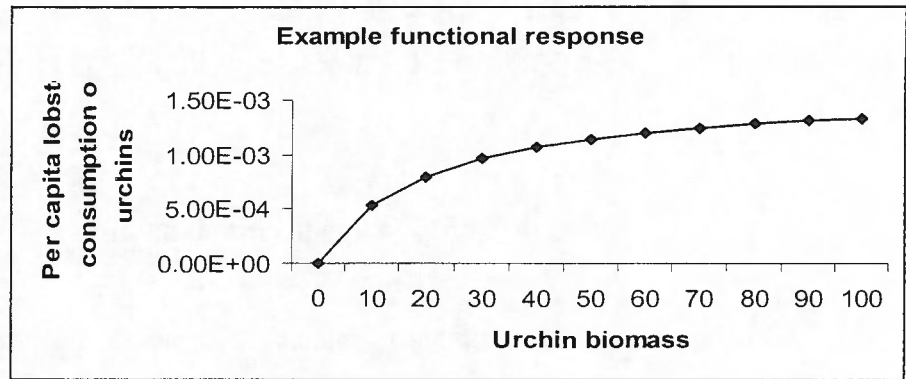
- where
- $u_y$  is the biomass of urchins at the start of Model year  $y$ ;
  - $r_u$  is the intrinsic growth rate of urchins;
  - $K_u$  is the urchin carrying capacity in Zone C;
  - $\alpha_{u-l}$  is an urchin-lobster interaction parameter;
  - $\beta_{u-l}$  is a second urchin-lobster interaction parameter; and
  - $J_y^I$  is the total inshore biomass of rock lobsters in Zone C in Model year  $y$ .



A logistic growth term was considered most appropriate given that the urchin population needs to be modelled as declining from a relatively high level relative to pristine down to a very low level followed by a possible recovery in biomass in response to lobster removals.

The sea urchin *P. angulosus* is an important component of the diet of both juvenile and adult west coast rock lobsters (Mayfield *et al.* 2000). However, Mayfield *et al.* (2000) found that despite very large differences in prey availability at different sites on the West Coast of South Africa, rock lobsters consume similar prey species in roughly equal proportions and amounts. Based on this, the biomass of urchins removed by rock lobsters is modelled using a simplified form of a Holling type II functional response. This formulation was chosen because it approximates a scenario in which the *per capita* consumption of urchins by rock lobsters remains

approximately constant at high urchin densities, but declines with decreasing urchin density. Sensitivity to alternative



interaction formulations can easily be considered. An example of the functional response formulation with parameter values  $\alpha_{u_l} = 8 \times 10^{-5}$ ;  $\beta_{u_l} = 0.05$  is shown alongside.

### 9.2.2 Modelling rock lobster dynamics

The current *Jasus lalandii* stock assessment model (Johnston 2002a) is rather complex and not suitable for inclusion in the model proposed here. Instead, the following equations are used to respectively model the inshore and offshore Zone C rock lobster populations:

$$\begin{aligned}
 J_{y+1}^I &= J_y^I + r_J (J_y^I + J_y^O) \left( 1 - \frac{(J_y^I + J_y^O)}{K_J} \right) - C_y^{J^I} + m_J^I + \rho_J J_i^O - \rho_J^* J_i^I \\
 J_{y+1}^O &= J_y^O + r_J (J_y^I + J_y^O) \left( 1 - \frac{(J_y^I + J_y^O)}{K_J} \right) - C_y^{J^O} + m_J^O - \rho_J J_i^O + \rho_J^* J_i^I
 \end{aligned}
 \tag{9.2}$$

where  $J_y^o$  is the total offshore biomass of lobsters in Zone C in Model year  $y$ ;  
 $r_J$  is the intrinsic growth rate ( $\text{yr}^{-1}$ ) of EoH lobsters;  
 $K_J$  is the lobster carrying capacity in the Zone C region;  
 $C_y^{J^i}$  and  $C_y^{J^o}$  are respectively the total inshore and offshore annual catches of rock lobsters in Model year  $y$ ;  
 $m_J^i$  and  $m_J^o$  are constants depicting the annual net longshore immigration of lobsters (biomass per year) into the inshore and offshore areas of Zone C respectively;  
 $\rho_J$  is a constant representing the annual proportion of lobsters in the offshore region that migrate inshore; and  
 $\rho_J^*$  is a constant representing the annual proportion of lobsters in the inshore region that migrate offshore.

A logistic rather than simpler exponential growth term is used in this instance because the EoH lobster population is currently considered to be close to carrying capacity (S. Brouwer, Marine & Coastal Management, pers. commn.)

Note that whilst the lobsters are modelled as impacting both the urchin and abalone (see below) populations negatively, the lobster dynamics are modelled as independent of these two populations. The justifications for choosing a one-way interaction representation are first because it is likely that the feedback loops are relatively small, and secondly to avoid getting into much more complicated details. For example, if the contribution of abalone and urchins to the diet of lobsters is to be explicitly considered, it has to be done in the context of the availabilities of the full spectrum of other prey items selected by lobsters. However, this is still open to debate, given the assertion of Mayfield *et al.* (2000) that the abundance of “preferred” prey may influence the growth of lobsters because they are highly selective foragers, “targeting” certain food groups. The work of Mayfield *et al.* (2000) suggested that urchins are a notable but not substantial (<10%) prey source for adult lobsters, but may be a more important component of the diet of juveniles.

Rock lobsters can occur in deeper water than abalone, and it is particularly important to note that they will have the largest effect on abalone populations in the shallow inshore waters

where the majority of abalone recruits and juveniles are to be found. It was therefore considered necessary to model the inshore and offshore proportions of the resource separately. This was also considered useful to be able to investigate the effects of removing lobsters differentially from the inshore and offshore regions.

Migration/immigration parameters are difficult to estimate but are considered critical in this context where it is necessary to quantify the biomass of lobsters that overlaps directly with the primary distribution of juvenile abalone. This preliminary model assumes that the longshore migration rate is constant, but this could be modified, given sufficient evidence, to include a time trend in this migration rate. Given that the inshore region may be a preferred habitat for lobsters, it is important to consider the possible “reseeding” of the inshore region with lobsters from the offshore region under a scenario where lobsters are preferentially removed from the inshore region. Further thought is required as regards the validity of assuming a constant offshore to inshore migration rate compared, for example, to assuming that this rate is a function of inshore (and perhaps offshore) lobster biomass. Tagging studies currently underway (S. Brouwer, Marine & Coastal Management, pers. commn) may assist in quantifying some of these parameters.

### 9.2.3 Modelling abalone dynamics

The dynamics of the Zone C abalone resource are described using the 2002 age-structured production model except for two changes.

#### *a) Explicitly including the negative effect of decreasing urchin biomass on the juvenile abalone survival rate*

This is done in a similar fashion to the method described in Appendix 6.2, except that increases in 0-yr old survival rate are explicitly linked to urchin biomass using the relations:

$$\begin{aligned}
 M_0(y) &= M_{\max} e^{-\nu u_i} && \text{for } u_i < K_u / \eta_u \\
 M_0(y) &= M_0^{\text{base}} && \text{for } u_i \geq K_u / \eta_u
 \end{aligned}
 \tag{9.3}$$

- where  $M_0(y)$  is the (time-variant post 1989) natural mortality rate in Model year  $y$  on abalone of age  $a < 1$ ;
- $M_0^{base}$  is the (time-invariant) natural mortality rate on abalone of age  $a < 1$ , as estimated to apply over the pre-1990 period (it is also the minimum 0-yr old natural mortality rate);
- $M_{max}$  is the maximum 0-yr old natural mortality rate (given very low urchin biomass);
- $\nu$  is a constant controlling the rate at which  $M_0$  increases in response to decreasing urchin biomass; and
- $\eta_u$  is a constant determining the threshold urchin biomass (relative to pristine  $K_u$ ), below which the survival rate of juvenile abalone is negatively impacted.

Mayfield and Branch (2000) indicated that there is a threshold urchin density below which the survival of juvenile abalone is affected.

***b) Explicitly accounting for the predation mortality on abalone due to rock lobsters***

The following is based on the assumption that rock lobsters predate on young abalone as proposed by some. The natural mortality parameter  $M$  used in single-species models implicitly takes into account the predation effects of a suite of “background” predator species and should be a reasonable approximation in all but situations where there is a dramatic sustained increase or decrease in the biomass of and associated predation mortality caused by one of the predators. Given the large recent increases in the EoH rock lobster population, it is thus desirable to explicitly include the associated increase in abalone mortality rates due to predation by lobsters in the model. The simplest method to introduce a biological interaction into a model appears to be to include the predator as a fishing fleet. The EoH inshore rock lobster population has therefore been included in the model as an “additional fishing fleet” operating since 1990, with the proportion of the abalone resource consumed by lobsters each year ( $F_y^J$ ) given by:

$$F_y^J = C_y^J / B_y^{prey} \quad (9.4)$$

where the biomass of abalone prey available for consumption by rock lobsters is computed as:

$$B_y^{prey} = \sum_{a=0}^{z-1} S_a^J w_{a+1/4} (N_{y,a}^I + N_{y,a}^O) e^{-M_a/4} \quad (9.5)$$

where  $S_a^J$ , the fishing selectivity-at-age for the lobster “fishing fleet”, can be interpreted as the predator feeding preferences for abalone of age  $a$  (assumed not to change over time). The summation is shown over ages 0 to  $z-1$ , where  $z$  is the plus group age, as it is unlikely that plus group animals are preyed upon by lobsters.

The consumption of abalone by lobsters is given by a simple Lotka-Volterra interaction term:

$$C_y^J = \alpha_{a-1} J_y^I B_y^{prey} \quad (9.6)$$

where  $\alpha_{a-1}$  is an abalone-lobster interaction parameter that in essence captures the “availability” of prey to the predator. The choice of a Lotka-Volterra interaction term is not ideal in this instance as it will result in a very strong interaction effect compared to other interaction formulations (see Chapter 12). However, the formulation above can easily be replaced with more sophisticated interaction terms if required. The purpose of the present exercise was simply to get an idea of the scale of the problem and to assess whether, given other reservations, it would be worth pursuing a data gathering exercise to estimate some of the parameters and pursue this modelling exercise.

It follows that the abalone numbers-at-age consumed by lobsters each year is given by:

$$C_{y,a}^J = S_a^J F_y^J (N_{y,a}^I + N_{y,a}^O) e^{-M_a/4} \quad (9.7)$$

Note that to take spatial effects into account crudely, only the inshore component of the lobster resource is assumed to prey on the abalone. Both the inshore and offshore components of the abalone resource are impacted by lobster predation, but because the majority of juvenile abalone occur inshore, the greatest impact is on the inshore component of the abalone population. Although simple, equation (9.6) is a logical first step in constructing this model and can easily be extended later to a more complicated functional form.

The approximation of lobster predation as a discrete pulse after the first quarter of the year is considered adequate given that predation is a relatively small fraction only of abalone biomass.

#### 9.2.4 Parameterising the model

##### Urchin component

$r_u$ : A number of models of urchin population dynamics (although not necessarily *Parechinus*) exist in the literature and these can be reviewed to provide some guidance as regards a specific growth rate for urchins. An initial rough estimate of this parameter has been obtained from population size census data available for the echinoid *Paracentrotus lividus* at Lough Hyne, Ireland (Barnes *et al.* 2002). Their database is one of the longest running censuses of a discrete echinoid population and shows spectacular population fluctuations for this urchin. The population increased steadily over the period 1975-1979 so that these data were used to compute a (presumably maximal) increase rate for an echinoid. Laboratory growth rate studies on *P. angulosus* have been done in Cape Town and hence these data could be analysed and used in this study if required (G. Branch, UCT, pers. commn).

$K_u$ : Data on (pristine/maximum) urchin densities are available for the EoH area and could be converted to a rough estimate of carrying capacity by multiplying by the total Zone C kelp habitat area (Tarr 1993). Accurate estimates of  $K_u$  are not critical for applying this model – it is more important to simulate the relative changes in urchin biomass (and in any case the interaction constants scale upwards or downwards depending on the actual value of  $K_u$  chosen).

$\alpha_{u,l}$ : This is the most difficult parameter to estimate, but some idea of this parameter can be obtained by attempting to fit the steep decline observed in urchin numbers. The dynamic regression approach proposed by Laska and Wootton (1998) should be explored as a potentially useful method to quantify interaction strength. The approach involves using multiple regression techniques to evaluate the relationship between empirically observed



rates of change and species abundances (and hence works best for situations far from equilibrium such as the EoH ecosystem).

### Rock lobster component

$r_j$ : This parameter needs to reflect both somatic and reproductive growth processes. Although the EoH increase in lobster biomass was initially attributed solely to the movement of large lobsters into the area, the lobsters are now thought to be completing their full life cycle in this area as small lobsters are increasingly being seen (S. Brouwer, MCM, pers. commn). To obtain an initial estimate of this parameter, S. Johnston projected the west coast rock lobster assessment model forward under the assumption of zero future catches. An average lobster population growth rate (somatic + reproductive) was then computed simply from the change in biomass from 2001 to 2030. This process was also repeated under an assumption of both zero future catches and zero future recruitment (to get an estimate for somatic growth rate only) but the population could not sustain itself under this scenario because of the low reference somatic growth rate. The above assumes that the growth rate is similar both west and east of Hangklip and in the inshore and offshore regions. The sensitivity to this assumption should be tested. Also, it may be necessary to allow at least one time-dependent change in the growth rate parameter to capture the increasingly important role of reproductive versus somatic growth processes.

$K_j$ : The EoH lobster population is currently thought to be at or close to the carrying capacity level with lobsters starting to “overflow” into the areas further east (abalone management Zones A and B) (S. Brouwer, MCM, pers. commn). The current biomass estimates (possibly scaled according to the relative kelp habitat areas in Zones C and D) could thus be used as estimates of  $K_j$ .

$C_i^{j'}$  and  $C_i^{j^o}$ : The total annual lobster catches need to be split between Zones C and D and between the inshore and offshore regions based on the relative proportions of the catch allocated to different sectors (including poaching). Most of the recreational and poaching takes are from the inshore region.

$m_J^I$  and  $m_J^O$ : The simplest assumption is that the net longshore immigration rate is the same in the inshore and offshore regions. This is a difficult parameter to estimate, particularly if it is not approximately constant (apart from the initial large influx of lobsters) from one year to the next. It may be possible to obtain an estimate from tagging data, a field experiment or by treating this as an estimable parameter in a population model fit to available data.

$\rho_J$  and  $\rho_J^*$ : Similarly still to investigate whether it is possible to obtain an estimate from tagging data, a field experiment or by fitting to available data.

Note that the initial values assumed and parameter values chosen need to be consistent with an observed trend of an increase in lobster biomass over the 1990's up to a maximum level in 2002 (under the assumption that the EoH lobster population – at least in Zones C and D - is currently at carrying capacity level).

### **Abalone component**

#### *a) Urchin – abalone interaction*

$M_{max}$ : Estimated within the abalone population model by fitting to data (as before, this is achieved by simultaneously fitting to industry survey and other data available for Zones B and C – see Chapters 6 and 7).

$\nu$ : As above, estimated within the model by fitting to data (indices of abundance and catch at age data).

$\eta_u$ : Either by fitting to data or it may be possible to select a value based on the work presented by Mayfield and Branch (2000).

#### *b) Lobster-abalone interaction*

$S_a^J$  This needs to be based on data/laboratory studies examining the preferences of rock lobsters for different sizes (to be converted to age) of abalone. The simplest possible approach involves ascertaining what age of abalone is too large to be susceptible to



predation by lobsters and then to assume  $S_a^J = 1$  (knife-edge selectivity) for all younger ages.

$\alpha_{a,l}$  Estimates could be obtained from field/laboratory studies or by fitting to data. This needs further thought.

### 9.3 EXAMPLE MODEL APPLICATION

As alluded to above, much more work is necessary before this model might be considered useful in a management context. However, there is some utility in experimenting with the model in the interim, and hence a single example of a model run is presented here. The parameter settings used in this initial run are shown in Table 9.1. Example output is presented in Figs 9.1 – 9.4.

An important management aim of a multi-species model such as this involves predicting the likely nature and scale of an increase in the abalone resource in response to lobster removals. The base-case demonstration model run predicted that in ten years' time the abalone resource would be at 31% of the current level whereas in 30 years' time it would be at 36% of the current estimated level. A preliminary sensitivity analysis was conducted (Table 9.2) to test the robustness of these predictions to relatively large uncertainties in the input parameter values and in the choice of initial (1990) values for the urchin and lobster biomass. Parameter values were either halved or doubled one at a time, and the model was rerun with the two mortality rate parameters as the only free (estimable) parameters to determine the change in the depletion statistics (Table 9.2).

### 9.4 DISCUSSION

Assuming that the EoH ecological shift is at least partly reversible, the model developed here may be useful as a tool to explore proposed multi-species management scenarios. It is not impossible that this model could be reasonably parameterised – however, the difficulties in parameterising it caution against the use of any more complex multi-species model to address this issue.

The model was most sensitive to the following parameters (Table 9.2):

- i) the urchin growth rate parameter  $r_u$  ;
- ii) the urchin-lobster interaction parameter  $\alpha_{u_l}$ ; and
- iii) the longshore migration parameter  $m_l$ .

Moreover, the model was highly sensitive to the initial (1990) lobster biomass level, with a halving of this value resulting in dramatically different abalone resource projections (Table 9.2). Given the levels of uncertainty associated with estimating the 1990 biomass level of lobsters, this illustrates the difficulties of trying to model these multi-species interactions reliably.

The basic aim of the demonstration simulation was to roughly capture the observed changes in EoH lobster and urchin biomass, so as to see what the effects on the projected abalone biomass might be. Note that the rock lobster predation effect on abalone may be included in the model to a greater or lesser degree following further investigations. Little time has been spent at this stage in trying to fine tune parameters, but nonetheless preliminary investigations were useful in highlighting the following.

- To simulate the crash in the Zone C urchin population as having occurred over a relatively short timespan, it was necessary to use a relatively large urchin-lobster interaction term. The result of this was that urchin populations are extremely slow to recover (and hence there was no obvious immediate benefit to abalone of removing lobsters) unless a high urchin growth rate was assumed. Once a more accurate estimate of urchin growth rate is input to the model, the model will potentially be very useful in predicting the timescale of a recovery of the abalone resource (assuming this is possible!). The sensitivity of model results to the urchin growth rate is positive in the sense that this is likely one of the easier parameters to measure in the field or laboratory and that (raw) data are potentially available to quantify this parameter.
- The sensitivity of model results to the lobster migration rates highlight the importance of trying to quantify these parameters.
- Although no great reliability should be accorded to the parameter values chosen for this illustrative application of the model, there were nonetheless indications from preliminary investigations that the predicted recovery of the abalone resource may be

slower than that predicted by a model that does not explicitly take the various interactions into account.

The 2002 BENEFIT Stock Assessment Workshop (BENEFIT 2002) agreed that the modelling exercise as presented here was useful as a first step to explore and bound the problem (in the sense of an upper limit in particular), to investigate the data needs and to test the potential sensitivity of model results to the choice of parameter values. Specific recommendations for improving the model included:

- Taking account of the strong size effect in interactions between lobsters and urchins;
- Giving due consideration to explicitly incorporating the important role of associated changes in the substrate and more specifically in the availability of coralline algae; and
- Re-examining and refining the interaction terms, in particular to test sensitivity to having used the strongest possible interaction terms and to consider reflecting the role of the habitat in mediating the urchin – abalone interaction.

The Workshop agreed that no biological reasons were presented that justified closing the EoH area to lobster fishing. Moreover, there was general consensus that lobsters do appear to impact abalone but that there is considerable uncertainty as to whether the current situation is potentially reversible. Part of this uncertainty resides in the possibility that the observed changes are due in part to an environmental or regime shift. The Workshop supported the use of both short-term experimental and modelling studies to assist in elucidating the nature and extent of the underlying causal mechanisms. There is already a considerable amount of data available to assist in parameterising the model (G.M. Branch, UCT, pers. commn) and experimental demonstrations (using caging) have already successfully been applied to confirm that both sediment composition and urchin populations can show some recovery when protected from predation effects (G.M. Branch, UCT, pers. commn). Nonetheless, the possibility should not be excluded that ecosystem change may occur in a manner contrary to expectations, as occurred at Malgas and Marcus Islands off the west coast of South Africa, where rock lobster removals resulted in an explosion in the abundance of a number of benthic species but not abalone (Barkai and Branch 1998).

Conditional on some of the recommended increase in lobster *TAC* for the 2002/3 season being allocated to the EoH region, the Workshop recommended that lobster fishing effort in the EoH region be concentrated in a small area so as to derive the maximum scientific benefit (in terms of the potential for duelling as a large-scale experiment to examine the impacts of reduced numbers of lobsters EoH) whilst simultaneously providing an associated opportunity to offset some of the socio-economic issues pertaining to abalone. This aspect is discussed further in the following chapter.

Table 9.1. Preliminary parameter estimates and initial values used in the demonstration model run.

Parameter	Value	Derivation
$r_u$	0.77	Barnes <i>et al.</i> 2002
$K_u$	100	i.t.o. arbitrary units
$\alpha_{u\_l}$	$8 \times 10^{-5}$	Tuning to achieve sensible urchin trajectory
$\beta_{u\_l}$	$1 \times 10^{-3}$	Tuning to achieve sensible urchin trajectory
$r_J$	0.006	Johnston (pers comm)
$m_J^I$	100	Arbitrary
$m_J^O$	100	Arbitrary
$\rho_J$	0.008	Arbitrary
$\rho_J^*$	0.001	Arbitrary
$M_{max}$	2.725	Estimated within model (purely illustrative)
$v$	0.0687	Estimated within model (purely illustrative)
$\eta_u$	2.5	By tuning to get sensible trajectory
$S_a^J$	$S_a^J = 1$ for $a = 1, 2, 3$ ; zero otherwise	Based roughly on practical assessment of likely feeding relationship
$\alpha_{a\_l}$	$1 \times 10^{-5}$	By tuning to get reasonable $F_y^J$ (ca. 5%)
<b>Initial values</b>		
$u_{1990}$	$50 (= \frac{K_u}{2})$	Arbitrary
$J_{1990}^I$	10000	Johnston (2002a)
$J_{1990}^O$	4000	Johnston (2002a)
<b>Annual lobster catches by all sectors of the fishery</b>		
$C_J^I$	50 MT/yr for yrs 1990 to 2000; C(2001) = 150; C(2002) = C(2003) = 350 600 MT/yr for yrs thereafter	
$C_J^O$	10 MT/yr for yrs 1990 to 2000; C(2001) = 20; C(2002) = 30; C(2003) = 50; 300 MT/yr for yrs thereafter	

Table 9.2. Results of preliminary sensitivity analysis for abalone spawning biomass. Parameters were either doubled or halved as indicated and the model was refit to determine the effect of the change in a parameter on the Zone C abalone spawning biomass in 2012 and 2032, relative to the 2002 level. The base-case depletion statistics are shown in bold and parameter perturbations to which the model was particularly sensitive are indicated in bold italics.

a) Parameters:												
	<b>Base</b>	$r_u = 1.54$ (base*2)	$K_u = 200$ (base*2)	$\alpha_{u\_l}$ (base * ½)	$\beta_{u\_l}$ (base * ½)	$r_J$ (base * 2)	$K_J$ (base * 2)	$m_J = 200$ (base * 2)	$\rho_J$ (base*2)	$\eta_u = 5$	$S_a^J = 1$ for $a =$ 1,2,...6	$\alpha_{a\_l}$ (base*2)
$B_{2012/2002}^{sp}$	<b>0.31</b>	<b><i>0.63</i></b>	0.30	<b><i>0.61</i></b>	0.32	0.31	0.31	<b><i>0.30</i></b>	0.31	0.22	0.22	0.22
$B_{2032/2002}^{sp}$	<b>0.36</b>	<b><i>1.32</i></b>	0.34	<b><i>1.29</i></b>	0.39	0.27	0.22	<b><i>0.06</i></b>	0.25	0.21	0.25	0.24
b) Initial values:												
	<b>Base</b>	$u_{1990}$ (base*1/2)	$J_{1990}$ (base*1/2)									
$B_{2012/2002}^{sp}$	<b>0.31</b>	0.22	<b><i>0.70</i></b>									
$B_{2032/2002}^{sp}$	<b>0.36</b>	0.28	<b><i>1.34</i></b>									

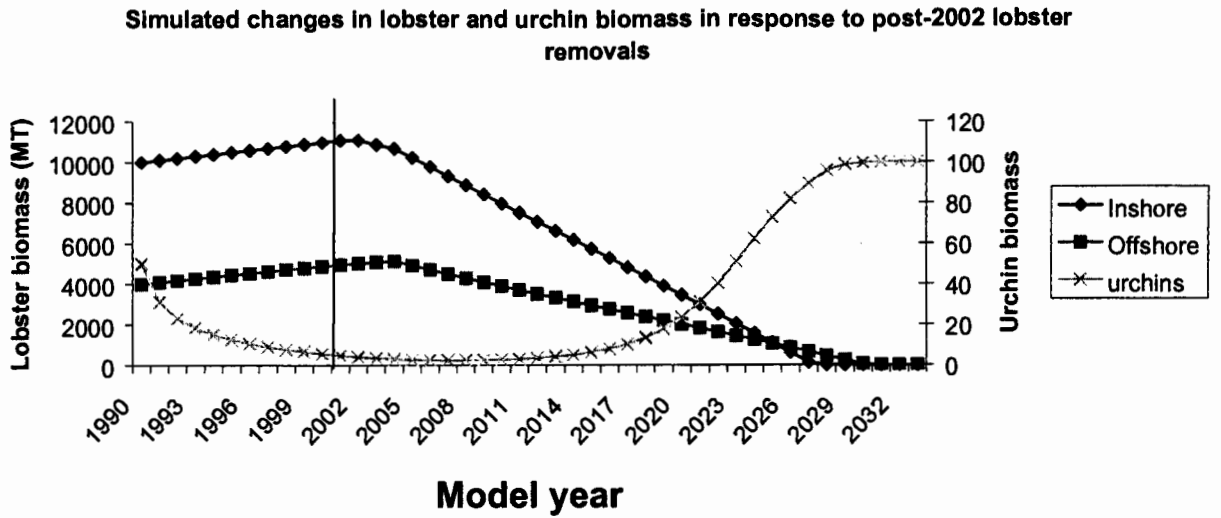


Fig. 9.1 Example inshore and offshore lobster and urchin biomass trajectories obtained using input parameters as detailed in Table 9.1 and under a scenario in which lobster catches are increased substantially post-2002.

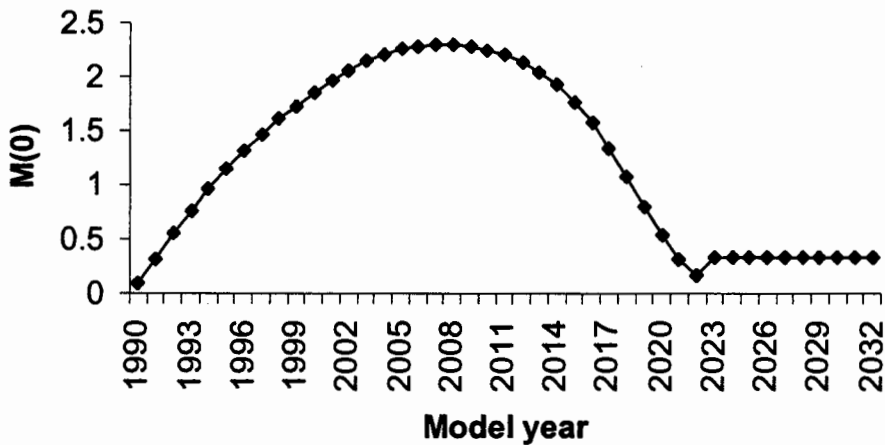


Fig. 9.2. Temporal pattern in 0-yr old mortality rate of abalone for the base-case demonstration model.

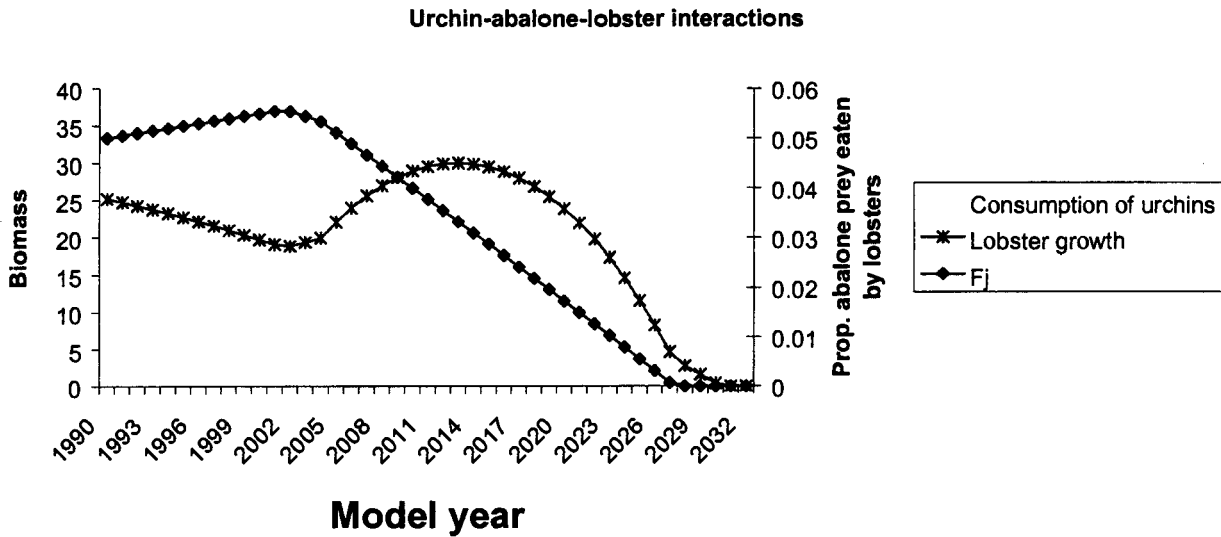


Fig. 9.3. The first y axis shows the temporal trend in lobster growth ( $rJ(1 - J/K)$ ) as determined from the logistic term in the lobster equation used in the model. The line with triangles shows the total biomass of urchins consumed by lobsters each year. The second y axis shows the modelled annual proportions ( $F_y^J$ ) of abalone prey (1-3 yr old abalone in inshore and offshore areas combined) consumed by the inshore component of the rock lobster population in Zone C (cf. Fig. 9.1).

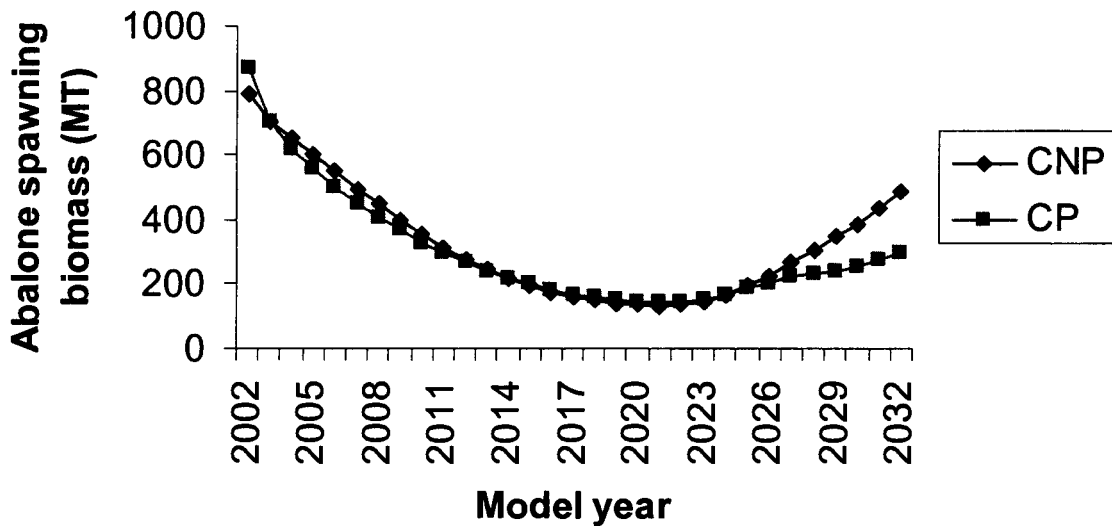


Fig. 9.4. Abalone spawning biomass (inshore + offshore combined) projections in subareas CNP and CP of Zone C given associated lobster and urchin biomass trajectories as shown in Fig. 9.1.





## Chapter 10

### Summary suggestions for further South African abalone research

#### 10.1 INTRODUCTION

A classic stock assessment approach (see Chapters 6 and 7) that attempts to estimate the extent of the illegal take and the reduction in juvenile abalone *Haliotis midae* survival due to the ecosystem shift has been used to set annual TACs for the South African abalone resource. However, as outlined in Chapters 2 to 9, this resource is subject to ever-increasing problems with illegal fishing and recent ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part of the range of the abalone. This begs the question as to whether a reasonably sound stock assessment exercise as applied in these circumstances has played any role other than that of accurately documenting the decline of the resource (see Chapter 14). Indeed, as highlighted by the review panel of the 2002 BENEFIT Stock Assessment Workshop (see Appendix 2.2), the current modelling methods are critical in highlighting the extent of the problem and evaluating possible changes in harvest regime, but do not represent the solution.

The actual solution was seen to reside in the development of alternative management strategies that include local communities. In response, a new policy to define the process of allocating commercial abalone fishing rights was announced by MCM in October 2003 (see Chapter 14). The new policy intends to “effectively address the threat of the illegal harvesting of abalone and introduce a new system of co-management” (DEAT 2003). The new system is based upon the Territorial User Rights in Fisheries (TURF) (Stephenson & Lane 1995, Christy 1996, Caddy 1999) concept and as such proposes dividing the primary fishing zones into smaller secondary zones (Fig. 10.1) with 10-year rights being issued to participants in each region.

At the time of writing this thesis, the management of the abalone resource is therefore in transition and the future remains uncertain. Given the mandate of the Abalone Working Group (AWG) to provide advice on the sustainable utilisation of the abalone resource, the assessment model described in this study has contributed as much as possible, under

challenging circumstances, over the past seven years. But the advent of a new management system requires co-evolution of the assessment method. Although the focus of this thesis (and of the AWG) is on assessment rather than management, broader questions have nonetheless been raised such as the extent to which legal commercial fishers should be penalised because of the actions of illegal fishers and whether classic stock assessment guidelines should still apply in non-classic situations. This aspect is discussed further in Chapter 14 where it is argued that, despite being associated with a management regime that has failed to reconcile fisheries with conservation in this instance, in retrospect there appear to have been clear economic and biological advantages to having continued these classic stock assessment exercises.

This chapter presents a summary of anticipated future work and suggestions for improving the current assessments and management of the South African abalone resource.

## **10.2 OUTSTANDING KEY ASSESSMENT QUESTIONS**

The following is a brief summary of some key assessment issues that need to be addressed in future work:

1. With respect to the CPUE data, as recommended by the BENEFIT review panel, future work will investigate the implications of using alternative error structure models such as the negative binomial. This work will also give consideration to including the nominal CPUE directly into the likelihood function.
2. Although cohort slicing is widely in use and has been adequate for current purposes, an improved method would be to fit the population model directly to length data assuming a fixed (time independent) growth curve with time-independent but length-dependent variance in length-at-age. The parameters of this variance function will be estimated in the fit. An outline of the proposed method being implemented is given in Appendix 10.1. It may also be advisable to change from age-specific to length-specific selectivity.
3. The assumption that expected growth is independent of location and time needs revisiting, recognising nevertheless that there may not be any other options. The von

Bertalanffy growth relationship currently used in the abalone population model is based on tag recapture data from Betty's Bay (Tarr 1995), a reserve area within Zone D. Growth studies were conducted similarly at a number of different sites (Tarr 1995) and suggested that growth relationships may vary on a spatial basis along the south coast. However, these data were not considered appropriate for inclusion in the model because of small associated sample sizes. An additional problem in using a single growth curve is that the growth curve obtained (as a result of tagging studies) may not be representative if it corresponds to, for example, a low productivity period. The BENEFIT Workshop recommended that simulation testing should be used to evaluate the impact of assuming an incorrect functional form and error-structure when modelling growth.

4. Following a recommendation stemming from the 2002 BENEFIT Workshop (BENEFIT 2002), future work will give consideration to allowing for additional variance when fitting to the FIAS data (given that these variance estimates are currently fixed, based on estimates of sampling variance obtained from the data) and to placing minimum bounds (of approximately 0.15) on the residual standard deviations for the CPUE data. The model-estimated  $\sigma$  values associated with the CPUE fit are in the range 0.08 – 0.15 for the base-case model and are thus considered unrealistically low. This can result in the model overemphasizing the need to fit (for example) the GLM CPUE data at the expense of other indices of abundance (such as FIAS).
5. It may be worthwhile to experiment with fitting to depletion estimates for the early 1980's based upon comparisons to Betty's Bay.
6. In the near future it is intended to extend the current approach to a full Bayesian analysis of Zones ABCD together with some common parameters, so as to allow for covariance in some uncertainties (e.g. in the allocation of historic catches between Zones). The extension from a (penalised) MLE to a fully Bayesian form is straightforwardly accomplished using the Markov Chain Monte Carlo (MCMC) method (a feature of the ADMB package currently used) to perform the Bayesian integrations. The MLE penalty functions essentially transform to Bayesian priors in this formulation (Lewy and Nielsen 2003), with the advantage that the Bayesian

formulation has sounder theoretical underpinnings than the penalised MLE approach (Butterworth *et al.* 2003). Moreover, the MCMC methodology can overcome the convergence problems that arise in some circumstances with the penalised MLE computations because of the flatness in many dimensions of the likelihood surface near its maximum.

The focus of the current study has been on Zones B and C, but it has been recognised for some time that to more accurately assess Zone D consideration needs to be given to:

7. The need to in some way take account of the changes in fishable area that have occurred in this zone. In 1986, access to a large kelp bed to the west of Zone D was prohibited, but then permission to fish a part of that area was again granted with effect from 1993 (Dichmont *et al.* 2000).
8. The need to take account of the fact that there is a marine reserve area (Betty's Bay), with a different density and age-structure to the surrounding fished areas, situated in the middle of this zone. Although (because of poaching and the impact of rock lobster) this reserve has not been functioning as such since 1994, the 2002 BENEFIT Workshop (BENEFIT 2002) recommended that it should be included in future assessments as a sub-area of Zone D and that the size-composition data from the Betty's Bay reserve should be included when fitting the model to the data for Zone D. Care then needs to be taken to match the data available with the inshore/offshore structure of the assessment model.

### 10.3 MANAGEMENT QUESTIONS

1. Although the approach to managing the abalone resource is currently in flux, there is room for further discussion related to the following broad policies underlying TAC recommendations:
  - a) Phased reductions: if poaching is estimated to have increased, decrease commercial TAC to compensate, but phase the decrease over a number of years in the interests of greater industry stability (especially if the resource is estimated to be not too depleted).

- b) Seek to ensure depletion levels do not fall below 0.4  $K$ : The choice of this reference point is based on model estimates of the  $MSYL^{sp}$  (maximum sustainable yield level – the spawning biomass level relative to pristine that yields the  $MSY$ ) (Chapter 7), but this reference point could be further refined.
2. In cases of recruitment reduction (currently Zones C and D), a decision needs to be made on whether it is more appropriate for  $K$  for 1b) to refer to the original or revised  $K$ .

## 10.4 CONSIDERATIONS IN THE LIGHT OF THE CHANGE IN POLICY

### 10.4.1 Background and TAC recommendations

Uncertainty regarding the future direction to be taken following the recently proposed Abalone Policy affected the nature of the scientific advice to be given during 2003. As a result, two scenarios were offered:

Scenario 1 was based on the same rationale as in previous years. It assumed a policy *status quo* with existing right holders remaining in the fishery, and in certain zones allowed for further depletion of the resource in the interest of retaining existing economic structures.

Scenario 2 took a longer-term view, and was based on the assumption that 10-year rights would be issued to a new range of participants, and that therefore no reduction in the spawning biomass should be considered. This view was motivated by both the desire for sustainable 10 year rights as well as the possibility that good custodianship should allow the resource to grow during this term.

To provide advice (at short notice) with respect to Scenario 2 above, selected projections (using the base-case model) were effected over a 20-year projection period (Fig. 10.2), in line with the change to rights being granted for a period of 10 years. Given that future poaching catches seem likely to remain at the current estimated levels or at most decrease by half, results were presented for scenarios H and M as before, i.e. H is the estimated 2003 poaching level (assumed to remain at this level for all future years) and M is 50% of the 2003 poaching level (i.e. projections assume poaching is halved relative to the estimated 2003 level in all

future years). The projected catches in Table 10.1 refer to commercial catches in tonnes, given that future recreational catches are set at zero. In each instance depletion statistics are given for one future catch scenario and for one scenario that sets future catches to zero (for illustrative purposes) (Table 10.1). For Zone A, model results indicated that (a) at present poaching levels, there is no sustainable commercial harvest available (Fig. 10.2) and (b) should poaching levels be immediately reduced by 50%, and continue at this level, a commercial harvest of 35 MT should be sustainable. For Zone B, projections indicated that (a) at present levels of poaching, the sustainable harvest was 25 MT. Should poaching be immediately reduced by 50% (b), the potential for a sustainable commercial harvest increased to 165 MT. Management opted for quotas of 35 and 165 MT for Zones A and B respectively under the assumption that future poaching levels will decrease.

Given that commercial abalone rights may be granted for a period of 10 years, on a purely scientific basis, the AWG recommended a zero TAC for each of Zones C and D. However, the AWG were requested to consider some of the following points that arose in response to the draft policy document:

1. Under the assumption that co-management of the abalone resource will result in a reduction in the amount of poaching, managers may wish to consider an option whereby non-zero rights are granted in Zones C and D;
2. The recreational catches from Zones C and D have been about 1.5 and 35 MT/yr respectively for each of the last three years (2000 – 2002). Given the suspension of the abalone recreational fishery, it was argued that this catch could perhaps be allocated to the new commercial rights holders instead;
3. Although open to debate, the removal of rock lobsters from Zones C and D may result in some improvement in the survival of juvenile abalone in these areas, such that it has been argued that the implications for management of an improvement in abalone recruitment success should be presented.

Accordingly, the following projections were effected using the base-case abalone age-structured production model (see Chapter 6 for model details):

1. Future recreational catches are set to zero;

2. Results are presented for scenarios with the future commercial catch in each of Zones C and D set at either:
  - i) 20 MT/yr for all future years;
  - ii) 20 MT/yr for the first 5 years and zero thereafter;
  - iii) 10 MT/yr for all future years.

The catches above are assumed to be equally divided between two secondary zones comprising Zone D, and between the two subareas CNP and CP comprising Zone C. Note that results have been combined here for ease of presentation, but it should be noted that subarea CP is currently substantially more depleted than subarea CNP.
3. Some scenarios are presented in which the survival rate of juvenile abalone in Zones C and D is assumed to increase to 50% of the pre-1990 level over the next five years. Note that this is a particularly optimistic scenario, intended to give some sense of the maximum recruitment recovery that could occur but it is probably unlikely that the resource could recover this rapidly in reality.
4. Poaching catches are assumed to either remain constant (i.t.o. numbers) at the current estimated level or to immediately drop and remain at half (i.t.o. numbers) the current estimated levels.

### ***10.3.2 Additional long-term projection results for Zones C and D***

Table 10.2 summarises projected depletion statistics in terms of abalone spawning biomass (the statistic of most interest in a biological context) for each of seven future catch scenarios as indicated.

Scenario I can be considered the base-case scenario – with zero future commercial catches from Zones C and D, poaching is assumed to continue at the current level, such that the total predicted catches (commercial + poaching) for 2004 are 92 MT (Zone C) and 214 MT (Zone D). If a commercial TAC of 20 MT per Zone is allocated and poaching decreases by half, the total predicted catches (commercial + poaching) for 2004 are reduced by one-third or more to 66 MT (Zone C) and 127 MT (Zone D). At a first glance it thus seems that it would be preferable to grant a TAC in these Zones. However, longer term projections reveal that the poaching catch is not sustainable as even if poachers continue to catch the same *number* of animals per year, the resource is depleted to such an extent that after only a few years this



translates into a catch of only 1 MT or less (Fig. 10.3). Note that this result assumes that poachers continue to fish in their current fashion.

A useful index to consider is the numbers available to the FIAS sector (because the survey index would be most sensitive to changes in recruitment and poaching) – note that this is an index of the numbers of animals in the inshore region. Graphic representations of the numbers available to the FIAS sector (Fig. 10.4) are useful in highlighting the timescale necessary before any detectable recovery in the abalone resource occurs. Under the assumption of a linear increase in survival rates (up to 50% of previous), there is approximately a 10-yr time lag before the current projected decrease of the resource is reversed (Fig. 10.4). It is thus important to bear in mind that even if the current recruitment failure effect is dramatically reversed, current sampling methods (the FIAS survey) are inadequate to detect this. A dedicated recruitment survey would be required. However, even if such a survey detected an appreciable improvement in recruitment (and again this could be verified only after at least a 3-5 year survey period), projected total spawning biomass trajectories for both Zones C (Figs. 10.5, 10.7) and D (Figs. 10.6, 10.8) are such that not only would it be difficult to argue for any increase in TAC in response to good management, but it may be impossible for the resource to sustain even a small annual TAC. *Under all scenarios considered, the abalone spawning biomass in Zones C and D is predicted to decrease to at least 5 - 7% of pristine levels in 10 years time.* It is not unreasonable to assume that an Allee effect (see e.g. Myers *et al.* 1995b) might operate at such low stock levels of a broadcast spawning species, especially given the relatively sessile nature of these organisms. These results highlight the importance of guarding against further stock reductions if some hope is to remain that the stock decline can be reversed.

As noted in Chapter 8, the following needs to be borne in mind in interpreting these results:

1. The assumption of a linear increase in survival rates was considered the most prudent approach given that there are so many other unknown factors. A sigmoidal curve would probably be more biologically realistic, but would require the estimation of another parameter.
2. The extent to which the current situation is reversible is not known. Removal of rock lobsters would almost certainly reduce any direct predation effects on the abalone, but there will be a lag before urchin *Parechinus angulosus* populations are able to recover to sufficient numbers to provide cover for juvenile abalone. The recovery of the urchin

populations would presumably in turn be impeded by the increased sedimentation that has occurred in their absence. Moreover, the increased sedimentation may not be easily reversed, and this may have both a direct and indirect effect on abalone recruitment success.

3. Both inshore-offshore and longshore movements of rock lobsters would influence the success of a rock lobster removal project.
4. From a purely biological perspective, it seems unlikely that a complete reversal or even major reversal (such as is implied by the 50% recovery scenario simulated here) of the rock lobster effect could be achieved. Before deciding to “wipe out” the EoH rock lobster resource, some indication is needed of the relative economic gains and losses from the two resources as well as the extent to which the abalone resource is likely to recover in the long term. This also needs to take into account the fact that Zones A and B may similarly be threatened by rock lobster invasions, particularly if the abundance of rock lobster EoH increases further.

Future work will likely explore further the use of both modelling studies, such as that described in Chapter 9, to assist in elucidating the nature and extent of the underlying causal mechanisms related to the abalone-urchin-lobster interaction. A data gathering exercise will need to be conducted to collate the data available to assist in parameterising the model (G.M. Branch, UCT, pers. comm) and experimental studies may be undertaken to further increase understanding. Indications from models such as that described in Chapter 9 of a short-term enhancement or reduction of the “rock lobster” effect could be fed into a decision model, provided such multi-species / ecosystem models are carefully parameterised and have demonstrated sufficient robustness of their conclusions to uncertainty in the data as well as to a range of plausible alternative hypotheses. In the case of abalone, the development of a tactical ecosystem model as the basis for computing harvest limits within an Operational Management Procedure (OMP) itself would seem to be a very long way off (see also Chapter 14).

A final point worth reiterating is that abalone populations in Zones C and D are currently depressed relative to pristine levels and are continuing to decline. If management action is going to be taken to reverse this process, this needs to happen fast before the spawning stock biomass is reduced to such a low level that there is a danger of an Allee effect coming into play. The extent to which reseedling of these areas could occur due to larval drift from Zones

A and B is not currently known. On the other hand, if it seems likely that the decline in the abalone population is irreversible, then effort should be focused instead on managing the rock lobster EoH on a sustainable basis.

### *10.3.3 Future management considerations*

There is some support for the notion that in the short-term it is preferable to grant a small TAC in each of Zones C and D if it is considered that this will assist in reducing poaching to at least one half of the current level. However, the results presented here underscore the fact that the actual TAC granted for Zones C and D should be as small as possible (without comprising the need to involve enough rights holders to have some effect on reducing poaching). Based on model results, it was recommended that 20 MT per Zone seems too large and a TAC of 10 MT per Zone C (5 MT from subarea CNP and 5 MT from subarea CP) and 10 MT from Zone D is the maximum that should be permitted. This recommendation was adopted by management (see Table 7.7). Moreover, it was suggested that consideration be given to the following possible provisos:

1. There is no guarantee that these TAC's will remain constant over the full 10 yr allocation period or that rights holders will be able to attain the TAC in any year;
2. Continued granting of an annual TAC would depend in part on a demonstration that poaching levels have decreased (to at least half the current level); some indication of this could be determined by examination of poaching confiscation data.
3. A dedicated recruitment survey is required to try and detect any changes in the current recruitment failure effect. Given that a 3 to 5 yr period is required to detect any change in the current trend, this suggests, for example, that catches be reduced to zero after 5 yrs if no improvement is evident;
4. The current survey methods and stock assessment methodology are inadequate to assess the performance of right holders under the new policy for at least the next 3-5 years. There are currently no defined guidelines or definitions as to what constitutes good or poor performance and how this could in turn be linked to the TAC (see also discussion in Chapter 14). However, it would be important to assess both whether there has been a reduction in poaching in a particular zone/sub-zone (based on confiscation data) as well as modelling the dynamics using information from research surveys and CPUE data. If the current stock assessment approach is continued unchanged, it should be possible to differentiate to some extent between good and

poor management scenarios after about 5 years. Ideally more intensive sampling methodology is required; and

5. Under the new management approach, the nature of the CPUE is likely to change and hence care should be taken to evaluate the comparability of CPUE before and after this change.

### Appendix 10.1 Method for fitting directly to length data instead of to catch-at-age information

The approach to be used to take catch-at-length information from the fishery into account in estimating the population parameters is described below (modified from Brandão *et al.* 2002) for a simple case with no separation into inshore and offshore compartments (i.e. no inshore-offshore movement term).

In a general ASPM, the predicted annual catches-at-age (by number) and per sector ( $\hat{C}_{y,a}^s$ ) are given by:

$$\hat{C}_{y,a}^s = N_{y,a} e^{-M_a/4} S_a^s F_y^s \quad (10.1.1)$$

where

$N_{y,a}$  is the number of abalone of age  $a$  at the start of year  $y$ ,

$M_a$  denotes the natural mortality rate on abalone of age  $a$ ,

$S_a^s$  is the fishing selectivity at age  $a$  for sector  $s$ , and

$F_y^s$  is the fished proportion of a fully selected age class, for sector  $s$ .

These are converted into predicted proportions of catch of age  $a$  as follows:

$$\hat{p}_{y,a}^s = \hat{C}_{y,a}^s / \sum_{a'} \hat{C}_{y,a'}^s \quad (10.1.2)$$

The proportions-at-age are then converted into proportions-at-length using the von Bertalanffy growth equation, assuming that the length-at-age distribution remains constant over time:

$$\hat{p}_{y,l}^s = \sum_a \hat{p}_{y,a}^s A_{a,l}^s \quad (10.1.3)$$

where  $A_{a,l}^s$  is the proportion of abalone of age  $a$  that fall in the length group  $l$  for sector  $s$  (thus  $\sum_l A_{a,l}^s = 1$  for all ages  $a$  for sector  $s$ ).

The matrix  $A$  is calculated under the assumption that length-at-age is normally distributed about a mean given by the von Bertalanffy equation, i.e.:

$$L_a \sim N[L_\infty(1 - e^{-\kappa(a-t_0)}); \theta_a^2] \quad (10.1.4)$$

where

$N$  is the normal distribution, and

$\theta_a$  is the standard deviation of length-at-age  $a$ , which is modelled to be proportional to the expected length-at-age  $a$ , i.e.:

$$\theta_a = \beta L_\infty(1 - e^{-\kappa(a-t_0)}) \quad (10.1.5)$$

where  $\beta$  is a parameter estimated in the model fitting process.

The somatic growth curve and the extent of variability about it would be assumed not to change over time for this approach.

Note that since the model of the population's dynamics is based upon a one-year time step, the value of  $\beta$  and hence the  $\theta_a$ 's estimated will reflect the real variability of the length-at-age as well as the 'spread' that arises from the fact that animals in the same annual cohort are not all spawned at exactly the same time, and that catching takes place throughout the fishing season so that there are differences in the age (in terms of fractions of a year) of abalone allocated to the same cohort.

The following term is then added to the negative log-likelihood (replacing Equation 6.48 in Chapter 6):

$$-\ln L^{\text{length}} = w_{\text{len}} \sum_s \sum_y \sum_l \left[ \ln \left( \sigma_{\text{len}}^s / \sqrt{p_{y,l}^s} \right) + p_{y,l}^s \left( \ln p_{y,l}^s - \ln \hat{p}_{y,l}^s \right)^2 / 2 \left( \sigma_{\text{len}}^s \right)^2 \right] \quad (10.1.6)$$

where

$p_{y,l}^s$  is the observed proportion (by number) in length group  $l$  in the catch in year  $y$  for sector  $s$ , and

$\sigma_{len}^s$  is the standard deviation (c.f. Equation 6.49) associated with the length-at-age data for sector  $s$ , estimated in the fitting procedure by:

$$\hat{\sigma}_{len}^s = \sqrt{\sum_y \sum_l p_{y,l}^s (\ln p_{y,l}^s - \ln \hat{p}_{y,l}^s)^2 / \sum_y \sum_l 1} \quad (10.1.7)$$

Equation (10.1.6) assumes that the proportion-at-length data are log-normally distributed about their model-predicted values. The associated variance is taken to be inversely proportional to the observed proportions  $p_{y,l}^s$  to downweight contributions from expected small proportions which will correspond to small observed sample sizes.

The  $w_{len}$  weighting factor may be set at a value less than 1 to downweight the contribution of the catch-at-length data to the overall negative log-likelihood compared to that of the CPUE and survey data. As previously when the  $-\ln L$  contributions were downweighted by a multiplicative factor of 0.1, this downscaling is necessary because the  $p_{y,l}^s$  data for a given year frequently show evidence of strong positive correlation, and so are not as informative as the independence assumption underlying the form of equation (10.1.6) would otherwise suggest.

Table 10.1. Modified projection scenarios. For all scenarios, H is the estimated 2003 poaching level (assumed to remain at this level for all future years) and M is 50% of the 2003 poaching level (i.e. projections assume poaching is halved relative to the estimated 2003 level in all future years). The projected catches below refer to commercial + recreational catches in tonnes.

	<b>A</b>	<b>B</b>	<b>CNP</b>	<b>CP</b>	<b>D</b>
<b>a) Future poaching = half current level</b>					
<i>i) Zero future commercial catch</i>					
<i>Bsp(2008)/K</i>	0.58	0.44	0.20	0.11	0.14
<i>Bsp(2023)/K</i>	0.62	0.61	0.02	0.01	0.02
<i>Bsp(2008)/Bsp(2003)</i>	0.99	1.26	0.57	0.48	0.42
<i>Bsp(2023)/Bsp(2003)</i>	1.05	1.78	0.07	0.06	0.05
<i>ii) Future commercial catch</i>					
	<b>170</b>	<b>80</b>	<b>30</b>	<b>20</b>	<b>20</b>
<i>Bsp(2008)/K</i>	0.51	0.39	0.14	0.10	0.13
<i>Bsp(2023)/K</i>	0.43	0.48	0.00	0.00	0.00
<i>Bsp(2008)/Bsp(2003)</i>	0.87	1.14	0.40	0.42	0.39
<i>Bsp(2023)/Bsp(2003)</i>	0.74	1.42	0.00	0.00	0.00
	<b>A</b>	<b>B</b>	<b>CNP</b>	<b>CP</b>	<b>D</b>
<b>b) Future poaching = current level</b>					
<i>i) Zero future commercial catch</i>					
<i>Bsp(2008)/K</i>	0.49	0.37	0.19	0.11	0.13
<i>Bsp(2023)/K</i>	0.36	0.39	0.02	0.01	0.02
<i>Bsp(2008)/Bsp(2003)</i>	0.83	1.07	0.53	0.45	0.40
<i>Bsp(2023)/Bsp(2003)</i>	0.61	1.12	0.07	0.06	0.05
<i>ii) Future commercial catch</i>					
	<b>170</b>	<b>80</b>	<b>30</b>	<b>20</b>	<b>20</b>
<i>Bsp(2008)/K</i>	0.42	0.33	0.13	0.10	0.12
<i>Bsp(2023)/K</i>	0.16	0.25	0.00	0.00	0.00
<i>Bsp(2008)/Bsp(2003)</i>	0.72	0.95	0.36	0.40	0.37
<i>Bsp(2023)/Bsp(2003)</i>	0.27	0.74	0.00	0.00	0.00



Table 10.2. Zone C and Zone D depletion statistics for the base-case model scenario and for scenarios assuming a recovery in juvenile abalone survival rates (Scenarios 4, 5 and 7). Scenario 1 is the base-case which assumes future poaching continues at the current estimated level, whereas the other Scenarios all assume that future poaching is half the current estimated level. Alternative future commercial catch scenarios are as indicated by the future commercial catches (*Ccomm*) shown for both the next five years (2004-2008) and the following fifteen years (2009-2023). Further details are given in Section 10.3.2 of the text.

<u>Projections</u>	<b>a) Zone C (CNP and CP combined)</b>						
	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6	Scenario 7
Recruitment recovery	0%	0%	0%	50%	50%	0%	50%
Future poaching	Current	Half	Half	Half	Half	Half	Half
Ccomm(2004-2008) MT/yr	0	20	20	20	20	10	10
Ccomm(2009-2023) MT/yr	0	20	0	0	20	10	10
Cpoa(2004) (NUMBERS)	135988	67994	67994	67994	67994	67994	67994
Cpoa(2004) (MT)	91.9	46	46.0	46.0	46.0	46.0	46.0
Catch total (2004) MT	91.9	66	66.0	66.0	66.0	56.0	56.0
<b><i>Bsp(2003)/K (CURRENT DEPLETION)</i></b>	0.27	0.27	0.27	0.27	0.27	0.27	0.27
<i>Bsp(2008)/K</i>	0.13	0.13	0.13	0.13	0.13	0.14	0.14
<b><i>Bsp(2013)/K (AFTER 10 YRS)</i></b>	0.07	0.05	0.06	0.07	0.07	0.06	0.07
<i>Bsp(2023)/K</i>	0.02	0.00	0.02	0.10	0.10	0.01	0.12
<u>Projections</u>	<b>b) Zone D</b>						
	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6	Scenario 7
Recruitment recovery	0%	0%	0%	50%	50%	0%	50%
Future poaching	Current	Half	Half	Half	Half	Half	Half
Ccomm(2004-2008) MT/yr	0	20	20	20	20	10	10
Ccomm(2009-2023) MT/yr	0	20	0	0	20	10	10
Cpoa(2004) (NUMBERS)	297152	148576	148576	148576	148576	148576	148576
Cpoa(2004) (MT)	213.8	106.9	106.9	106.9	106.9	106.9	106.9
Catch total (2004) MT	213.8	126.9	126.9	126.9	126.9	116.9	116.9
<b><i>Bsp(2003)/K (CURRENT DEPLETION)</i></b>	0.33	0.33	0.33	0.33	0.33	0.33	0.33
<i>Bsp(2008)/K</i>	0.13	0.13	0.13	0.13	0.13	0.14	0.14
<b><i>Bsp(2013)/K (AFTER 10 YRS)</i></b>	0.07	0.06	0.07	0.08	0.07	0.07	0.08
<i>Bsp(2023)/K</i>	0.02	0.00	0.02	0.10	0.07	0.02	0.10

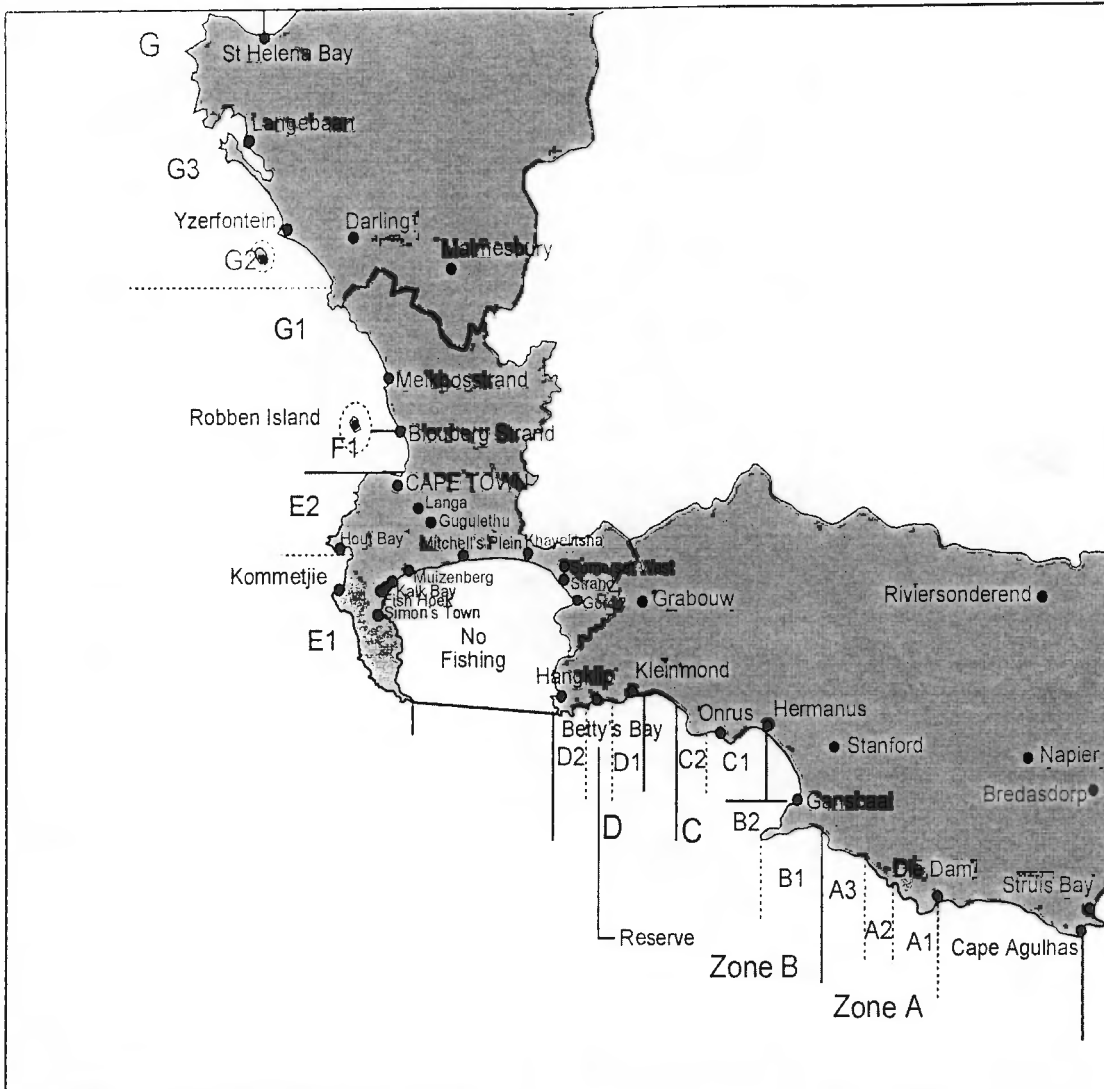


Fig. 10.1. Map (from DEAT 2003) showing the possible division of primary abalone fishing zones into secondary zones as part of the implementation of the new management policy. Zones A, B and D are currently assessed at the level of primary zone only. The secondary zones C1 and C2 of Zone C correspond to subareas CNP and CP which are differentiated in the current models.

Chapter 10 – Future work

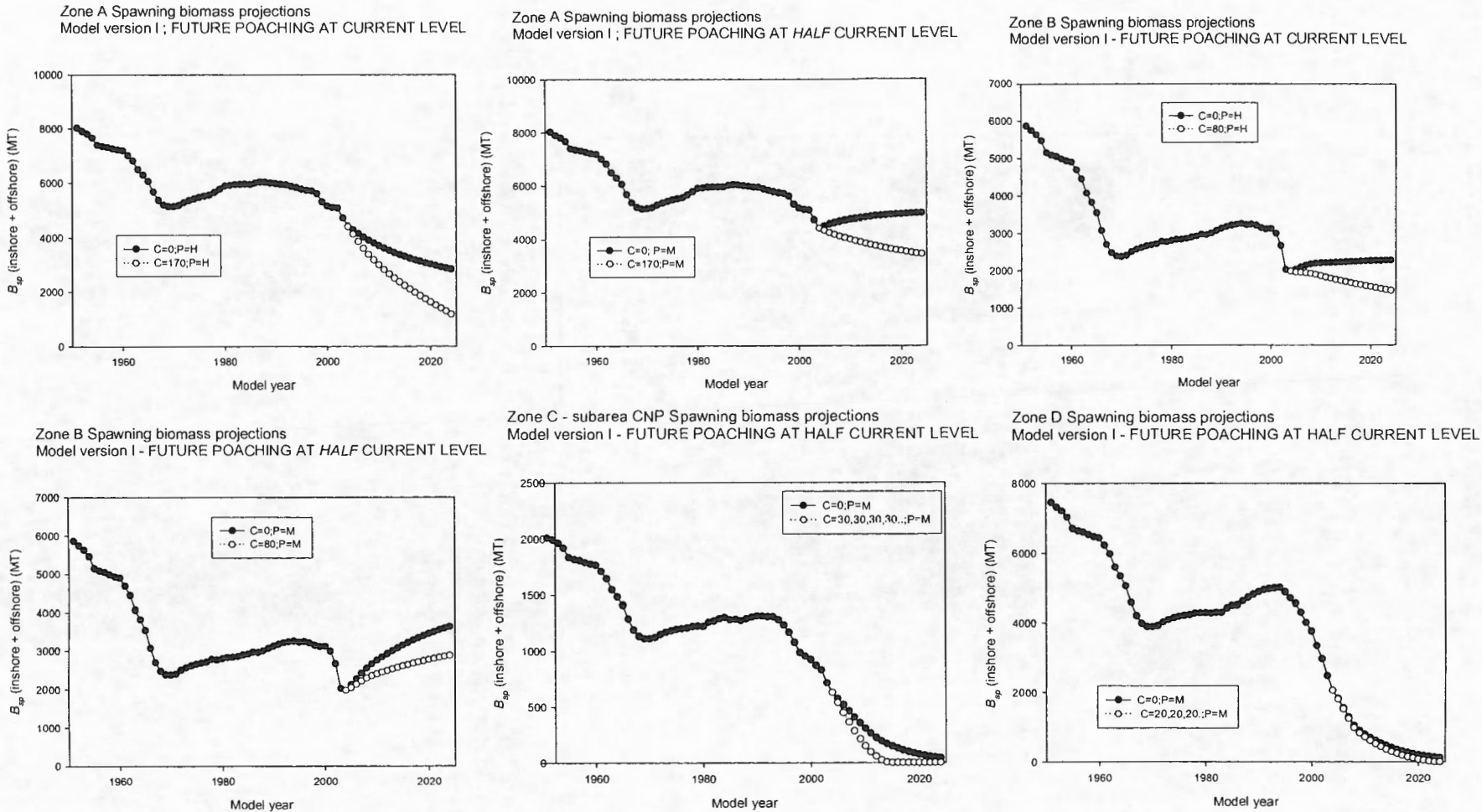
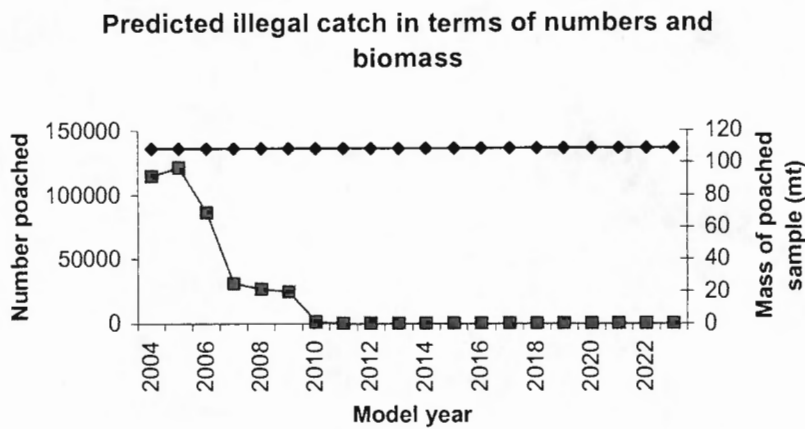


Fig. 10.2 Selected 20-year spawning biomass projections for Zones A-D shown for one future catch scenario and another that sets future commercial catches (but not poaching) to zero (for illustrative purposes). Results are presented for scenarios H and M as before, i.e. H is the estimated 2003 poaching level (assumed to remain at this level for all future years) and M is 50% of the 2003 poaching level (i.e. projections assume poaching is halved relative to the estimated 2003 level in all future years). The projected commercial (C) and poaching (P) catches are in tonnes. Future recreational catches are assumed to be zero. Only a single example is shown for each of Zones C and D because in these instances the resource is predicted to continue to decrease irrespective of future catch levels.

a) Option 1 - Zero future commercial catch & future poaching at current level



b) Option 2 - 20 mt future commercial catch and future poaching halved

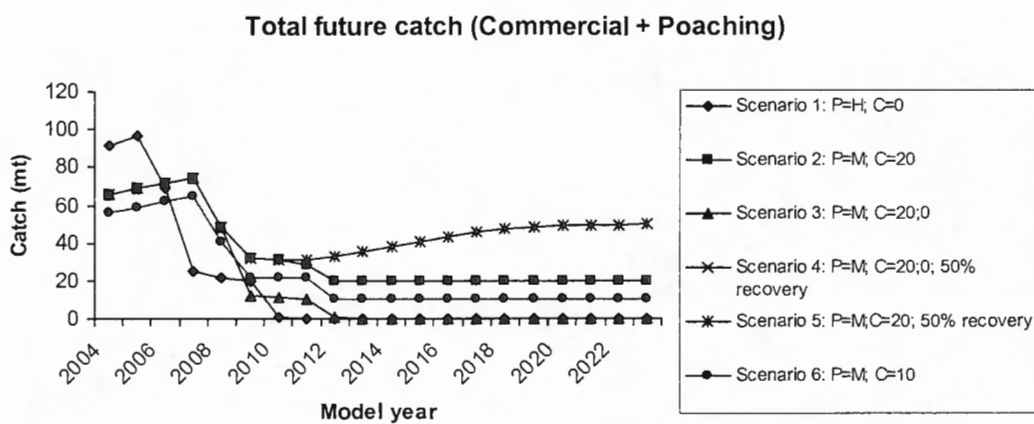
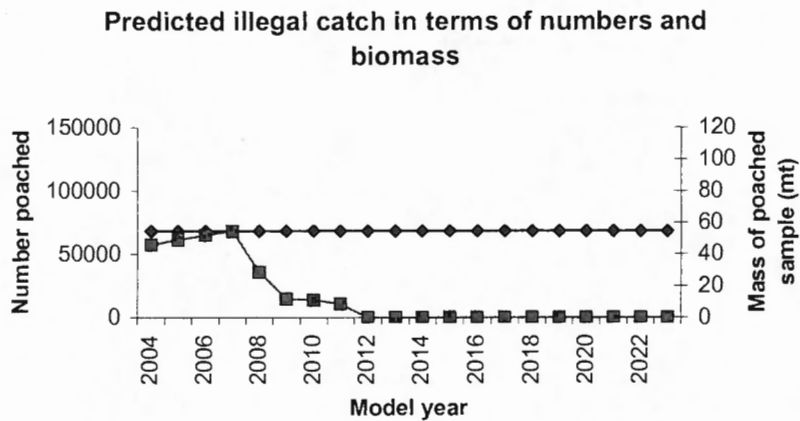


Fig. 10.3. Comparison of future scenarios 1 and 2 highlighting the rapid decrease in the mass of poached catches to be expected even when the number of animals poached stays constant over time. This explains why there are no large differences in resource status predicted after 10 years even when future poaching levels are assumed to decrease by half ( $P=M$ =Medium poaching scenario compared with  $P=H$ =High (equivalent to current) future poaching scenario). The scenarios considered are specified in Table 10.2. The future commercial catch ( $C$ ) key shows the constant catch over the next five years, followed by the catch over the following fifteen years.

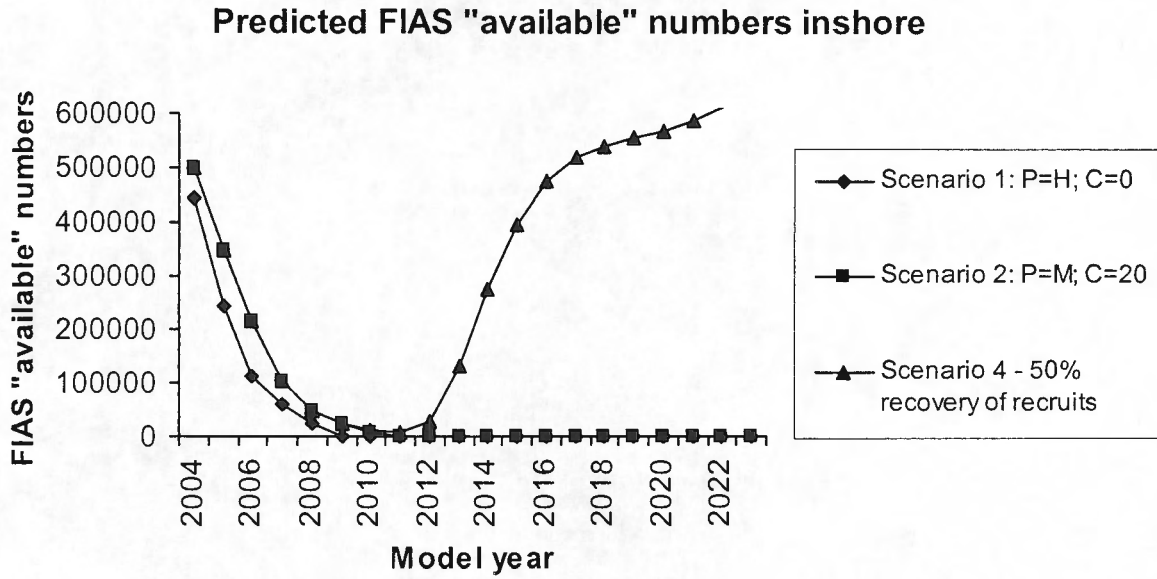


Fig. 10.4. Predicted FIAS “available numbers” in Zone C when compared between scenarios assuming no resource recovery and a 50% recruitment recovery over the next 5 years. Note that the three different scenarios shown are almost indistinguishable for the first 10 years.

### Spawning biomass trajectories - Zone C

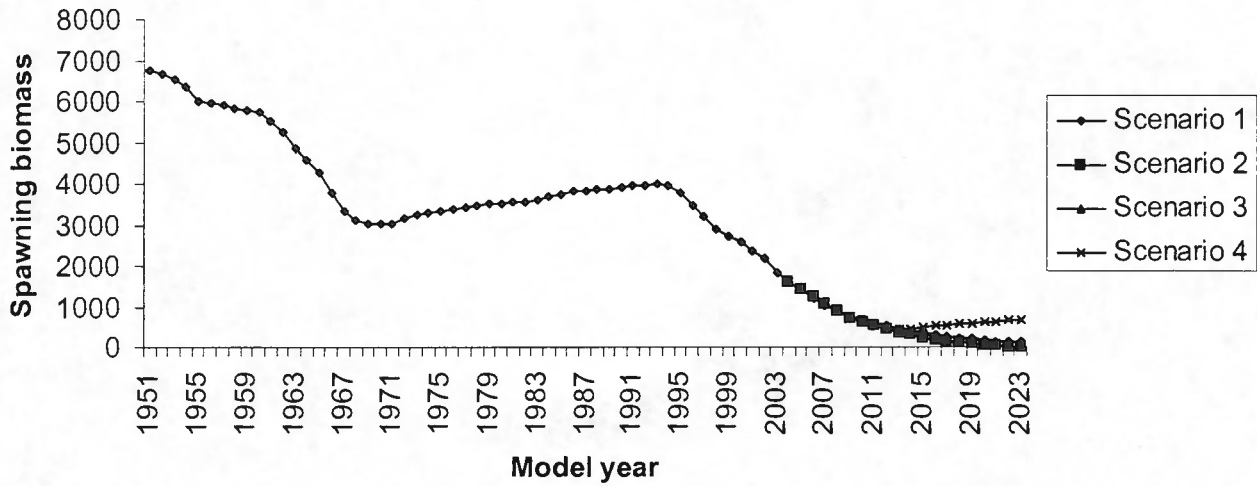


Fig. 10.5. Spawning biomass trajectory for Zone C highlighting the fact that the resource is predicted to decline to extremely low levels over the next 10 years relative to the pristine spawning biomass level, irrespective of the scenario considered.

### Spawning biomass trajectories - Zone D

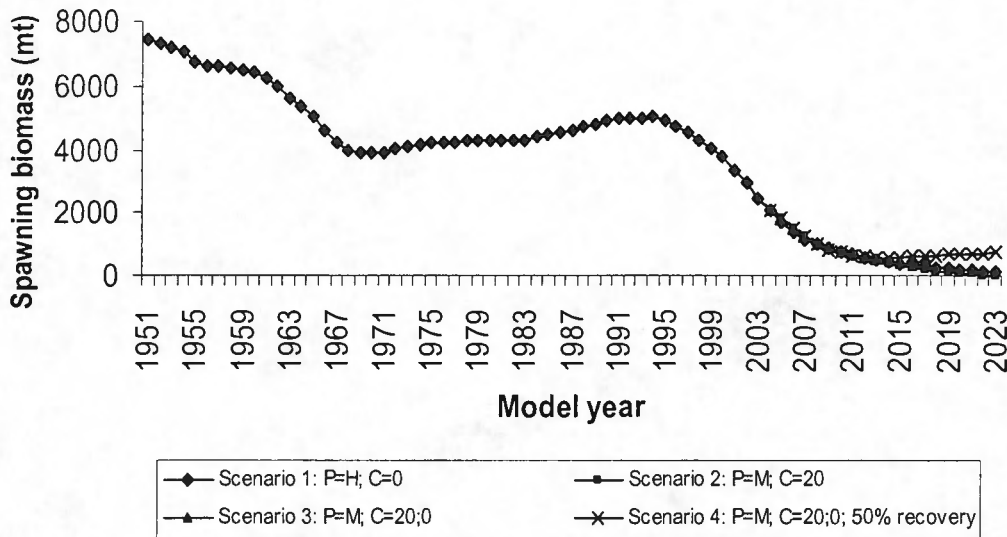


Fig. 10.6. Spawning biomass trajectory for Zone D highlighting the fact that, as for Zone C, the resource is predicted to decline to extremely low levels over the next 10 years relative to the pristine spawning biomass level, irrespective of the scenario considered.

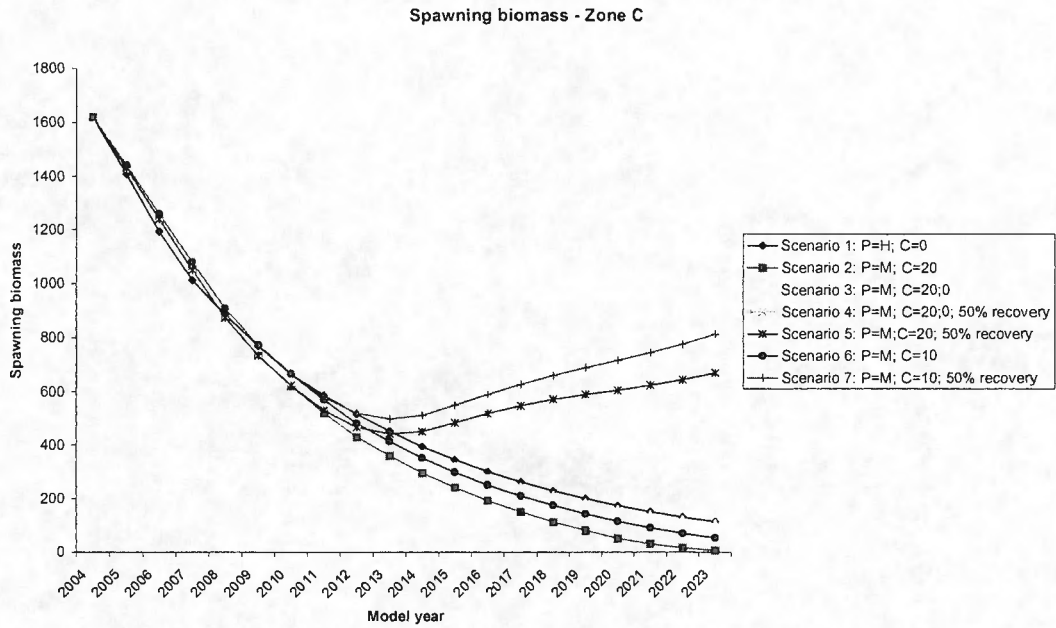


Fig. 10.7. Projected spawning biomass trajectories for Zone C (CNP and CP combined) under seven alternative scenarios as summarized in Table 10.2.

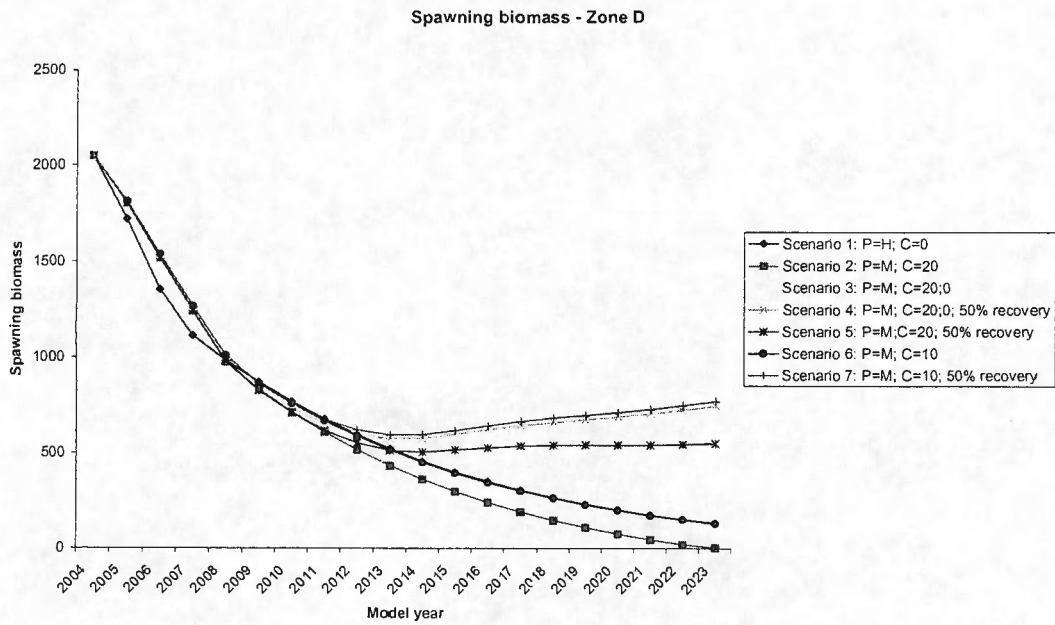


Fig. 10.8. Projected spawning biomass trajectories for Zone D under seven alternative scenarios as summarized in Table 10.2.

# **Section II**

## **Multi-species perspectives**





## Chapter 11

# The global eco-modelling epidemic: a critical look at what ECOPATH with ECOSIM can and cannot achieve in practical fisheries management

### SUMMARY

Ecosystem-orientated thinking is increasingly incorporated into fishery management. Given the complexity of ecosystem processes, there is a need to evaluate the tools used to steer this thinking critically. ECOPATH with ECOSIM (EwE), an aggregate system modelling package, is currently the most widely employed approach to assess the ecosystem effects of fishing and hence is considered in greater detail than other multi-species approaches in this thesis. The basic equations and assumptions, strengths and weaknesses, some past and possible future applications and hence the potential of this approach to contribute to practical fisheries management advice are reviewed. Strengths include the structured parameterisation framework, the inclusion of a well-balanced level of conceptual realism, a novel representation of predator-prey interaction terms and the inclusion of a Bayes-like approach (ECORANGER) to take account of the uncertainty associated with values for model inputs. Weaknesses in model structure include the constraining nature of the mass-balance assumption (of ECOPATH) for initiating projections, the questionable handling of life history responses such as compensatory changes in the natural mortality rates of marine mammals, overcompensatory stock-recruit relationships that result from default parameter settings, possible problems in extrapolating from the microscale to the macroscale, as well as some (not too far-reaching) mathematical inconsistencies in the underlying equations. There is a paucity of systematic and stepwise investigations into model behaviour and properties, and users are cautioned against non-critical use of the default settings. An important limitation related to the predominant use of EwE as a “black-box” modelling tool is that some users fail to consider a range of alternative interaction representations. As with all multi-species approaches, the major limitation in applying the EwE approach lies in the quality and quantity of available data. Current EwE applications generally do not adequately address uncertainty in data inputs and model structure. The potential of EwE to contribute to five important multi-species management quandaries in the marine environments off southern Africa and Antarctica is assessed, leading to the conclusion that EwE has limited predictive capability in these contexts. Apart from data constraints, this is due, *inter alia*, to the high residual variability in these systems, the important role of abiotic and biotic mesoscale (o(100 km)) processes, life history handling and insufficient flexibility in the preset model structure to simulate hypothesized causative mechanisms adequately. However, prudent EwE applications that utilise good data and are based upon rigorous statistical analyses can complement the quantitative predictions of traditional single-species models. They could be particularly useful in some contexts if output in the form of probability distributions encompassing a range of likely ecosystem responses were to be coupled with attempts to extend Operational Management Procedure (OMP) approaches to fisheries management beyond the single-species level. In particular, such applications could serve as the operating models of the underlying dynamics that are used for computer simulation testing of OMPs.

## 11.1 INTRODUCTION

World-wide concerns relating to over-exploited fish stocks are complicated by the demonstrated inadequacies of single-species models to address some of these concerns (e.g. to what extent will exploitation of forage fish impact predators of those fish, themselves perhaps subject to fishing). However, have multi-species modelling approaches matured sufficiently to provide information that can usefully be incorporated into *practical* fishery management advice? The focus here is on biological interactions, in contrast to operational or technical interactions, such as by-catch issues and problems related to animals becoming entangled in fishing gear. To contribute to practical advice, a multi-species modelling approach should provide at least qualitative, and ideally defensible quantitative guidance as to the modifications in annual catch levels deemed necessary because of the predicted effects that fishing on a target species will have on other components of the ecosystem. Figure 11.1 provides a simple framework summarising a range of multi-species models constructed to inform the management of commercially important stocks. In most cases, movement to ecosystem management will involve unprecedented, untested policy changes so that modelling is unavoidable and the choice is only about which one(s) to use (C. Walters, pers. commn).

A key prerequisite for multi-species management advice involves taking explicit account of the major sources of uncertainty and their consequences. Multi-species modelling approaches differ greatly in this regard in addition to differences in their representations of the three fundamental processes structuring ecological systems, namely predation, competition and environmental disturbances (Hollowed *et al.* 2000). In general, as models are made more complex (Fig. 11.1) to take better account of biological realism, there is an inevitable increase in scientific uncertainty, as a result both of a lack of knowledge of functional relationships and of imprecision in estimates of the associated parameter values. Single species model analyses used to advise on catch limits concentrate heavily on checking sensitivities and the implications of estimation imprecision. In contrast, these considerations are frequently near ignored in evaluations of the more complex multi-species models.

The ECOPATH (Polovina 1984, Christensen and Pauly 1992), ECOSIM (Walters *et al.* 1997) and ECOSPACE (Walters *et al.* 2000a) suite is currently dominating attempts worldwide to provide information on how ecosystems are likely to respond to changes in fishery management practices. The model structure is described in various formats by the model developers (see e.g. Walters *et al.* 1997, Christensen and Walters 2000, Pauly *et al.* 2000) and is not exhaustively reviewed here. The development and ongoing evolution of ECOPATH with ECOSIM (EwE) is undoubtedly a major driving force behind the pursuance of an ecosystem approach to fisheries management. First appearances by ECOPATH and then ECOSIM were readily criticised by practising stock assessment scientists. However, it is this author's opinion that the approach and model structure have now matured to the point where they merit serious attention in assessment circles and that, when properly used, EwE can complement (though not replace) traditional stock assessment methods and assist in policy decisions through an increased understanding of ecosystem structure and functioning.

This document is an attempt to assess independently some potential successes and failures of applying EwE in a management context. This analysis is presented in the form of a walk through EwE (see Fig. 11.2), with various detours to highlight selected aspects of model structure and use. In particular, the usefulness of the EwE modelling approach as a predictive multi-species management tool in the southern Benguela upwelling system and the Antarctic ecosystem is addressed. This review examines both aspects of the actual EwE model structure as well as the way in which the model has been used, acknowledging that poor or misuse of EwE and misunderstanding of its limitations, should not be interpreted as a criticism of the model structure itself. Comments in the text are not separated under "structure" and "use" headings in the interests of presentational flow, but a summary is given under each heading in the final discussion section.

Whilst recognising that many of the problems raised could apply also to other multi-species modelling approaches (see e.g. Hollowed *et al.* 2000, Fulton *et al.* 2003), the focus of this chapter is restricted to EwE as conventionally applied to marine ecosystems. Some discussion of other multi-species methods such as multi-species virtual population analysis MSVPA/MSFOR (Pope 1991, Sparre 1991, Magnússon 1995, Vinther 2001), BORMICON (Stefánsson and Pálsson 1998) and GADGET (Globally applicable Area-Disaggregated General Ecosystem Toolbox) (see e.g. webpage <http://www.hafro.is/dst2/report2/> ; coordinator G. Stefánsson) is presented in the next chapter so that detailed comparisons of the

various multi-species approaches are not discussed in this document. Chapter 12 provides more general discussion regarding how ecosystem/multi-species modelling might best be developed to contribute to the management of South African fisheries. Chapter 13 compares these different modelling approaches with respect to their appropriateness as tools to assess competition interactions between marine mammals and fisheries.

This review is based in the main on model formulations and results as given in the available literature. However ECOSIM is constantly being improved, patched and extended, so that some criticisms made may no longer apply to the most recent version/s of ECOSIM. Given the difficulties in defining what exactly constitutes “ECOSIM” (because the package is constantly evolving), together with typical delays before work is published, the focus here is on EwE studies published before the end of 2002, and on software versions up to EwE 4.0, as described in the software manual (Christensen *et al.* 2000) available on the website ([www.ecopath.org](http://www.ecopath.org)) and in other publications referenced in the text. “ECOSIM” may have changed since then, but then this might also bring the results of published studies based on earlier versions of ECOSIM into question.

## 11.2 BACKGROUND TO MODEL EQUATIONS

Full details of the EwE modelling approach, as well as the associated software, are obtainable from the web site [www.ecopath.org](http://www.ecopath.org). Briefly, the fundamental ECOPATH mass balance equation is based on that originally proposed by Polovina (1984). This balance for each functional group  $i$  in an ecosystem (detritus excepted) is described by:

$$B_i \cdot (P/B)_i \cdot EE_i = Y_i + \sum_j B_j \cdot (Q/B)_j \cdot DC_{ji} + BA_i \quad (11.1)$$

where  $B_i$  and  $B_j$  are the biomasses of  $i$  and the consumers ( $j$ ) of  $i$  respectively;

$(P/B)_i$  is the production/biomass ratio for  $i$ ;

$EE_i$  is the fraction of production of  $i$  that is consumed within, or caught from the system (the balance being assumed to contribute to detritus);

$Y_i$  is the fisheries catch ( $Y = FB$ ;  $F$  is the proportion fished);

$(Q/B)_j$  is the food consumption per unit biomass of  $j$ <sup>1</sup>;

$DC_{ji}$  is the fractional contribution by mass of  $i$  to the diet of  $j$ ; and

<sup>1</sup> Note that throughout the discussions in this chapter, the term *per capita* is used equivalently to the term *per unit biomass* because the EwE notation includes references to both numbers and biomass of species groups.

$BA_i$  is a biomass accumulation term that describes a change in biomass over the time period studied and/or net immigration (Christensen 1995).

The ECOSIM models convert the above “steady-state”<sup>2</sup> trophic flows into dynamic, time-dependent predictions. For prey  $i$  and predator  $j$ , Walters and Kitchell (2001) model the dynamics of the vulnerable ( $V_{ij}$ ) and non-vulnerable ( $N_i - V_{ij}$ ) components of the prey abundance (by number) of  $i$  as:

$$\frac{d(N_i - V_{ij})}{dt} = -v_{ij}(N_i - V_{ij}) + v'_{ij} V_{ij} \quad (11.2)$$

$$\frac{dV_{ij}}{dt} = +v_{ij}(N_i - V_{ij}) - v'_{ij} V_{ij} - a_{ij} V_{ij} N_j \quad (11.3)$$

where the total consumption rate  $Q_{ij}$  of prey  $i$  by predator  $j$  is  $a_{ij} V_{ij} N_j$ , and  $N_j$  represents the number of predator group  $j$ .

Under the assumption that the dynamics of the  $V_{ij}$  are much faster than those of the  $N_i$ ,  $dV_{ij}/dt$  is set to zero, yielding:

$$V_{ij} = v_{ij} N_i / (v_{ij} + v'_{ij} + a_{ij} N_j) \quad (11.4)$$

and hence (taking biomass to be proportional to numbers) the standard ECOSIM interaction term form describing trophic flows  $Q_{ij}$  between prey group  $i$  and predator group  $j$ :

$$Q_{ij} = a_{ij} v_{ij} B_i B_j / (v_{ij} + v'_{ij} + a_{ij} B_j) \quad (11.5)$$

where  $a_{ij}$  is the rate of effective search for prey  $i$  by predator  $j$ , and  $v_{ij}$ ,  $v'_{ij}$  are prey vulnerability parameters, with  $v_{ij} = v'_{ij}$  as the default setting (Walters *et al.* 2000b).

As in the classic Lotka-Volterra formulation ( $Q_{ij} = a_{ij}B_iB_j$ ), flows are determined by both prey and predator biomasses, but equation (11.5) incorporates an important modification in that it encompasses a framework for limiting the vulnerability of a prey species to a predator, thereby including the concept of prey refugia and also tending to dampen the unrealistically large population fluctuations usually predicted by the Lotka-Volterra formulation. Note that the vulnerability  $v_{ij}^*$  as input to ECOSIM is a rescaling of the vulnerability parameter  $v_{ij}$  above so that this is expressed relative to the rate at which an individual predator of species  $j$  consumes species  $i$ ; specifically this is such that the recommended ECOSIM input default  $v_{ij}^* = 0.3$  corresponds to (K. Aydin, pers. commn):

$$R_{ij} = \frac{2v_{ij}}{a_{ij}B_j} = 1 \quad (11.6)$$

or 
$$a_{ij}B_j = 2v_{ij} \quad (11.7)$$

Some more technical points regarding the derivation of the form of the ECOSIM functional response are given in Appendix 11.1. In particular, when using the default vulnerability settings, the above equations are shown to be inconsistent with the notion of only a small proportion of prey being vulnerable at any time. The reader is referred to Appendix 11.1 for discussion of the implications of some mathematical inconsistencies underlying the ECOSIM interaction term, which fortunately are not far-reaching (C. Walters, pers. commn).

The early ECOSIM versions (e.g. ECOSIM II) modelled consumers as follows (Walters *et al.* 2000b):

$$\frac{dB_i}{dt} = g_i \sum_j Q_{ji} - \sum_j Q_{ij} + I_i - (M_i + F_i + e_i)B_i \quad (11.8)$$

where  $g_i$  represents net growth efficiency per unit of consumption by  $i$ ;

$I_i$  is the biomass immigration rate for  $i$ ;

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<sup>2</sup> Strictly in applications where some  $BA$  term is non-zero, the ECOPATH approach does not reflect “steady-state”/“equilibrium”. However, in the interests of parsimony of expression in what follows, this terminology is retained (in quotes for this reason) as the spirit of the approach, even with this adjustment, is to represent balances in a “steady” (possibly steadily changing) situation, in contrast to modelling the dynamics fully.

- $M_i$  is the mortality rate on  $i$  not accounted for by predation within the system;  
 $F_i$  is the fishing mortality on  $i$ ; and  
 $e_i$  is the emigration rate of  $i$ .

To overcome the limitations of a biomass dynamics framework, where relevant, juvenile and adult pools in ECOSIM II are linked using a delay-differential equation system that keeps track of flows in terms of numbers as well as biomass. The dynamics are based on the Deriso-Schnute equations and are described in more detail in Walters *et al.* 2000b. More recently, the latest version of EwE includes a facility to model fully age-structured population dynamics with multiple life history stanzas and recommends the use of this approach in favour of the adult/juvenile splitting implemented earlier (Christensen and Walters 2004). Note that although ECOSIM focuses on feeding interactions, it also includes a facility in the form of a (seasonal or longer term) forcing function routine to represent the mediation of physical or other environmental parameters influencing these trophic interactions (Christensen *et al.* 2000).

In many respects, EwE achieves a good balance in model structure between simplicity and the level of complexity that often accompanies other ecosystem model representations. Although users have tended to include a large number of components in their EwE models, it can also be used in more of a Minimum Realistic Model (MRM) sense (V. Christensen, pers. commn, see chapter 12).

### 11.3 A CLOSER LOOK AT EwE

In the sections below, some weaknesses in model structure and potential pitfalls in the application of EwE as a tool to address management issues in situations such as those summarised in Fig. 11.1 are reviewed. Points to be borne in mind in interpreting output from the EwE software packages are also discussed.

#### 11.3.1 “Equilibrium” / “steady-state”<sup>3</sup> assumption

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<sup>3</sup> Note footnote 2 regarding the use of the terms “equilibrium” and “steady-state”.



The “steady-state” requirement of ECOPATH models implies that the model outputs apply only to the period for which the inputs are deemed valid (Christensen and Pauly 1992). Specification of the period for which these models are expected to have predictive power is therefore critical. For example, the choice of a decade for a slowly-changing ecosystem such as a coral reef (e.g. Arias-Gonzalez *et al.* 1997) seems appropriate; however the choice of a decade as a suitable time scale for models of upwelling systems (e.g. Jarre-Teichmann 1998, Shannon and Jarre-Teichmann 1999) seems open to question because of the fundamental role of variability in mediating important interactions on smaller spatial and temporal scales. There is the added danger that the construction of such models over different (arbitrary) time intervals, e.g. for the 1980’s to compare with the 1990’s, may lead to the conclusion that a sudden regime shift has occurred, as distinct from the possibility that the observed changes are simply a reflection of a steady trend. However, given short-lived small pelagic species have highly variable annual recruitment, averages over ten year periods of random variability can differ appreciably even though there is in fact no underlying change in the system.

It is possible to include trends (e.g. in the biomass of groups) in the ECOPATH analysis that is used to initiate projections, but the critical concern remains: the parameter values themselves are constrained by “steady-state” assumptions. Hollowed *et al.* (2000) point out that such models fail in adequately determining the impacts of fishing on marine ecosystems precisely because the multi-species interactions of most interest are precisely those that could cause marked departures from the current situation (and hence likely also from values of key associated parameters).

As an example, Christensen (1998) concludes that ECOSIM shows promise for setting up management options for multi-species fisheries through, for example, simulations to predict the exploited state of the ecosystem based upon knowledge of the pre-exploitation state. Although ECOSIM users are generally cautioned against studying the impact of drastic changes in fishing pressure, Christensen (1998) does explore the degree to which the heavily fished state of the Gulf of Thailand in 1980 could be reproduced based on a model of the pre-exploitation state of the ecosystem in 1963. As many predicted biomass changes were in reasonable agreement with the changes observed, he suggests that his results indicate that the imposed constraints which keep parameters values at their pre-exploitation “equilibrium” values does not seriously compromise ECOSIM’s predictive ability. However, an alternative conclusion could be that it provides an example of how readily it is possible to obtain a

reasonable fit to a set of data without correctly specifying the underlying causal mechanisms. Indeed, whereas the estimated changes in the biomass of some groups were in reasonably good agreement with changes estimated from trawl surveys, at the other extreme ECOSIM's predicted 1980 / 1963 biomass change of crabs and lobsters (admittedly only a minor part of the system) was >1000-fold compared with only a three-fold change observed. Nonlinearities in species interactions and in the response of individual species to fishing pressure are likely to play an increasingly important role the greater the departure from initial assumed "equilibrium" conditions. Managers must always take care when inferences drawn from models have involved substantial extrapolation, as in this case.

The use by some authors of the same  $P/B$  ratio per species group for two ECOPATH models of the same ecosystem for different time periods is a weakness. Even some of the most erudite ECOSIM applications can be criticised on this point. For example, Trites *et al.* (1999) assume the same constant  $P/B$  ratio ( $0.5 \text{ yr}^{-1}$ ) for adult pollock *Theragra chalcogramma* when trying to explain the large changes between the 1950s (prior to large-scale commercial fisheries) and the 1980s (characterised by an increase of groundfish such as pollock) in the eastern Bering Sea ecosystem, despite a 500% increase in the adult pollock biomass over this period.

Potentially one of the most useful applications of ECOSIM - to assist in resolving the debate concerning the extent to which major changes in ecosystem structure between two periods can be explained by the relative roles of fishing, predation and the environment (see e.g. Bundy 2001, Trites *et al.* 1999) – is similarly handicapped by the "equilibrium" assumption. A valuable suggestion put forward by Bundy (2001) is to assemble a second ECOPATH model for the later period, to be used in conjunction with direct biological parameter estimates for that period to assess the extent of change in the values assumed for the model of the first period.

### 11.3.2 Parameterisation

In traditional ecosystem models, modellers often confess to some level of *ad hoc* adjustment or "tuning" in an effort to find a feasible solution. An attraction of the ECOPATH approach is that for each species/species group, one of the quantities  $B$ ,  $P/B$ ,  $Q/B$  or  $EE$  is obtained by solving the linear equations and hence, given the other values needed, a unique mass-balance

solution is always obtained. However, the robustness of the model output to variability and uncertainty in these input values needs to be better explored.

The Bayes-like approach incorporated in the ECORANGER routine allows explicit consideration of uncertainty in input values (Pauly *et al.* 2000) and is an important step forward for the EwE approach. Analyses based on ECOPATH models that do not fully explore the consequences of uncertainty in the inputs are potentially misleading, especially when current knowledge of all the components in an ecosystem is such that, for example, actual biomass estimates for some species may vary by as much as a factor of ten (in contrast to the ECORANGER default of 10%). Moreover, the multiplicative effects of uncertainty in all input values (for diet proportions as well as abundance and  $P/B$  ratios) also need to be considered. Although ECOPATH analyses do assist in understanding an ecosystem, it is not justifiable to present them as a possible basis for multi-species management approaches without an accompanying appraisal of the associated levels of uncertainty. Unfortunately to date few ECOPATH applications appear to have attempted to represent and consider uncertainty about parameter values adequately.

A further issue with the use of the ECORANGER approach is that Monte Carlo integration over a strongly peaked function is computationally inefficient. This difficulty arises because the very large number of constraints imposed by the ECOPATH relations, such as that all EE parameters in the mass-balance equations must be less than 1, means that the feasible region is a very small proportion of the total parameter space. ECORANGER does incorporate a Sampling-Importance-Resampling (SIR) scheme (McAllister *et al.* 1994) to improve computational efficiency. This approach depends on a user-defined input function which measures the extent to which alternative solutions best satisfy some criterion (Christensen *et al.* 2000), and which presumably serves as the Importance function for the SIR process. The difficulty is that unless most of the mass of this Importance function falls within the feasible region (i.e. is consistent with the ECOPATH related constraints, which anyway would be difficult to know before conducting the analysis), little increase in computational efficiency would be expected. This situation might be improved by the use of the MCMC (Monte Carlo Markov Chain – see, e.g. Gilks *et al.* 1996) approach to such integration, which could readily incorporate the ECOPATH constraints directly in the function effectively integrated (see e.g. Somhlaba *et al.* 2004).

The possible double use of information is also a concern here because of the data intensive requirements of ecosystem approaches. Caution must be exercised to ensure the independence of parameter values input and the data used for fitting purposes. A further potential abuse of these models lies in the practice of parameter value “swapping”, whereby the choice of unknown parameter values is justified on the basis of values obtained by other researchers – often for different temporal and spatial scales. Careful thought needs to be given to the source of values for parameters. A common fault in EwE applications lies in the use of diet composition data from a different time period without adjusting to account for the differences in the relative abundances of prey species between the two time periods. This is because even if no prey-switching occurs, the use of non-representative dietary data will compromise both the ECOPATH mass balance results and projections using ECOSIM.

Improvements in this regard may be achievable through the use of techniques to smooth spatial and temporal variability in food composition and predator abundance, such as the geostatistical approach of kriging (Bulgakova *et al.* 2001). A good example of the importance of separating prey size preference from prey availability is given in Floeter and Temming (2003) (who consider North Sea cod).

One of the most important steps necessary to enhance the utility of the EwE approach in management contexts is an extension towards more soundly based statistical estimation methods. The application by Savenkoff *et al.* (2001) of inverse models to minimise the imbalances between inputs and outputs in an ECOPATH model constitutes a move in the right direction. Rather than the *ad hoc* adjustments generally made to achieve mass balance, Savenkoff *et al.* (2001) propose obtaining an optimal (balanced) solution by estimating trophic flows using a specific least squares criterion. The criterion used attempts to minimise the sum of the flows through the foodweb, with the mass balance thus being closed by residuals (inputs – outputs) rather than ecotrophic efficiencies as in ECOPATH. A particular advantage of that approach is its ability to refine estimates of diet composition in a manner that weights the evidence from different sources in a statistically defensible manner (Savenkoff *et al.* 2001).

Large ecosystem models based on poor data obviously have no place in the management arena. However, a viable solution is not necessarily to be found in the call for more holistic studies and future data collections (see e.g. Toral-Granda *et al.* 1999). A timely suggestion by

Mace (2001) concerns evaluating what is attainable in an ecosystem management context relative to the costs and benefits associated with obtaining the necessary information (see also next chapter).

### 11.3.3 Differences in energetic content of prey

There is no energetic content parameter included in the rightmost term of Equation (1), so alternative prey types are treated as energetically equivalent in EwE.

Contrary to this assumption, Winship and Trites (2003) recently drew attention to the need to consider not only the quantity, but also the distribution and quality of food available to predators. They estimate that in southeast Alaska, seasonal changes in the energy density of the diet of Steller sea lions *Eumetopias jubatus* resulted in animals requiring approximately 45-60% more food per day in early spring than in late summer. Regional differences in the energy density of the diet similarly accounted for substantial (up to 24% based on summer diets) differences in food requirements among the South-East and Western populations of Alaska Steller sea lions (Winship and Trites 2003). The same considerations presumably apply in the case of some fish species.

### 11.3.4 Prey-switching and the stability of prey suitabilities for predators<sup>4</sup>

Prey-switching occurs when the suitability of a prey for a predator exhibits a sudden change as a function of prey abundance, such as when a predator switches to another prey species when its original preferred species drops below a certain level of abundance. Mechanisms responsible for switching include predators developing a search image for a certain prey type, an increase in capture success with increasing abundance and the occupation of different sub-habitats by different prey types (Murdoch *et al.* 1975). The extent of prey-switching is still open to debate, with some authors finding no evidence of prey-switching (e.g. Bell *et al.* 1999 – bluefish *Pomatomus saltatrix*) and others arguing that prey-switching has occurred (see e.g. Murdoch *et al.* 1995 – fishes, Estes *et al.* 1998 - marine mammals, Mayfield (1998) – rock lobsters).

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<sup>4</sup> “Suitability” is used here in its technical sense. This reflects how desirable prey of one type are to a predator relative to other prey types: specifically, it expresses the relative frequency of selection of a prey type if all prey types were equally abundant. In contrast, the term “preference” is used here in a purely colloquial sense.

As noted by Walters *et al.* (1997), a potential problem with the use of ECOSIM to model situations far from the initial trophic “equilibrium” is that it cannot replicate the phenomenon of prey-switching by predators. This can be problematic in modelling highly dynamic ecosystems such as upwelling systems, where species have adapted to their variable environment by evolving extremely flexible feeding behaviour, with both feeding behaviour and rate of food consumption determined by the size and density of available prey (e.g. James and Findlay 1989). Walters *et al.* (1997) furnish further examples and discussion of situations in which the ECOSIM equations are likely to fail. The impacts of various multi-species harvesting strategies on community structure and fishery yields will most likely be incorrectly predicted (in a qualitative sense) for situations far from the initial “equilibrium”, *inter alia* because it is under such scenarios that prey-switching is more likely to manifest itself.

Both EwE and MSVPA models assume constant suitabilities (also referenced in the literature as the constant predator preference assumption). Some support for using near constant suitabilities derives from the extensive analyses of North Sea field collection data for MSVPA. Rice *et al.* (1991) deduced that suitabilities were quite stable over time when the effects of changing abundance and age structure are taken into account, such that multi-species models parameterised with good field data have the potential to be useful in management. Subsequently, however, Rindorf *et al.* (1998) concluded that cod and whiting *Micromesistius poutassou* exhibit positive switching, i.e. they keep eating a given food item to a greater extent than suggested by the constant suitability model when the food item becomes relatively scarce. Their findings highlight the need for caution in assuming constant suitabilities (as in both EwE and MSVPA models), particularly when extrapolating to situations far from the initial model state.

Adlerstein and Welleman (2000) note a potential bias in predation-mortality estimates because predator stomach contents can vary appreciably within a 24-hr period. They stress the need for time stratification in stomach sampling to improve the quality of the data used to implement multi-species models.

## 11.4 FORAGING ARENA CONSIDERATIONS

### 11.4.1 Direction of flows: bottom-up versus top-down control

ECOSIM users are able to set vulnerability parameters to control the extent to which the model moves towards “top-down” and away from “bottom-up” control. The two-state (vulnerable/invulnerable) representation of prey biomass (see equations 2-5) is a first approximation to modelling the complex spatial and temporal overlaps between predators and prey (Christensen and Walters 2000), and represents a substantial improvement on predation rate formulations employed by most other multi-species models. The major disadvantage relates to the difficulties of choosing appropriate values for prey vulnerability parameters because of the obvious problems in trying to quantify these processes. This is one of the biggest potential limitations to modelling ecosystem interactions with any confidence because, as emphasised by Shannon *et al.* (2000), ECOSIM predictions can be highly sensitive to the choice of vulnerability settings.

Utility in a management context therefore warrants careful explorations of model sensitivity to a range of energy flow assumptions, as has been done, for example, by Bundy (2001) in her study of Newfoundland-Labrador ecosystems. Another example of the way in which uncertainties as regards vulnerability settings have not necessarily impeded attempts to provide guidance for fisheries managers is provided by Arreguín-Sánchez (2000). By conducting ECOSIM simulations over a range of vulnerability and fishing mortality scenarios, he was able to investigate the importance of the interdependence between two important fishery resources, the octopus *Octopus maya* and red grouper *Epinephelus morio* (which preys on the former), in relation to the impact of fishing on octopus biomass. Daskalov (2002) used an EwE approach to assess the relative roles of overfishing (top-down forcing) and eutrophication (bottom-up forcing) in mediating changes in the Black Sea ecosystem through a trophic cascade mechanism. These examples show that ECOSIM studies can play an important role in investigating the impacts of changes in fishing mortality in the context of the relative roles of changes in top-down and bottom-up forcing.

### 11.4.2 Functional response<sup>5</sup> formulation and the problem of using default values

Given the difficulties associated with an appropriate choice of vulnerability values, there is a strong temptation for users to revert to using the default values and/or to assume the same value for all species. The default vulnerability value implies that the current abundance corresponds to the “half-saturation” point on the consumption curve as shown in Fig. 11.3. Specifying such a default for all predators seems questionable because differences in their prior exploitation histories mean that the current abundances of different predators would correspond to different proportions of their pre-exploitation levels, and likely also to different points on their consumption curves. The use of the same vulnerability value for all species is therefore problematic because, *inter alia*, it assumes that the species are all currently at the same point of the curve shown in Fig. 11.3, irrespective of their prior exploitation histories. Moreover, *a priori* assumptions of vulnerability settings can strongly influence model outcomes.

Consider, for example, the fundamental ECOSIM relationship between total consumption  $Q_{ij}$  of prey pool  $i$  by marine mammal predator  $j$ , and the abundance of predators  $B_j$  as shown in Fig. 11.3. Marine mammals have generally been heavily harvested in the past and therefore have been reduced to levels which are low compared with their pre-exploitation levels. As is evident from Fig. 11.3, if a scenario is run in which numbers of marine mammals (as predators) which were low reach numbers that are “high” (towards the right hand side of the Fig. 11.3 plot) and these numbers are subsequently halved and maintained at that level (say e.g. through culling), the total consumption  $Q_{ij}$  hardly drops using ECOSIM’s default vulnerability values. This is because the per capita consumption ( $Q_{ij}/B_j$ ) nearly doubles. Thus ECOSIM cannot yield pure-replacement results when predicting the extent to which consumption by another “predator” (a fishing fleet, say, that acts identically in terms of prey selection) can increase sustainably in response to a marine mammal cull. Expressed another way, this is to argue that default parameter value selections for the model effectively hard-wire it to such an extent that they effectively swamp other signals pertinent to predicting the effects of a marine mammal population reduction. Confidence in the ECOSIM predictions would in this case hinge on the extent to which the ECOSIM interaction representations (with their associated vulnerability parameter settings) are considered realistic and reliable. The

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<sup>5</sup> See Appendix 12.1 for a summary of different functional response types.



argument above serves also as a partial explanation of the simulation results of Aydin and Friday (2001), who implemented a simple three-compartment model (zooplankton – fish – marine mammals) in ECOSIM to investigate the effects of removing a top predator from the ecosystem. They found that reducing the abundance of marine mammals in the system by harvesting them (given default vulnerability settings) hardly changed the maximum sustainable yield (*MSY*) for commercial fish.

Unlike in the Lotka-Volterra representation of a predator-prey interaction, the foraging arena model in ECOSIM does not treat predators and prey symmetrically. If one raises or lowers the number of PREDATORS (while keeping prey numbers fixed), in view of the fact that ECOSIM has a “predator interference” functional response (equation (11.5)), the individual predators then eat less or more respectively because this formulation assumes implicitly that the predators are competing with one another for limited prey. Thus, as shown above, the immediate effect of this (if predator numbers are towards the right on the curve in Fig. 11.3) is that the total consumption of prey stays about the same. On the other hand, consumption (both total and *per capita*) is a linear function of prey abundance in ECOSIM (see implication (3) in Appendix 11.1) so that if PREY abundance is halved (e.g. through fishing), total consumption of prey by the predators is decreased proportionately. Thus, for example, if fishing reduces the abundance of a commercial fish species by half, this will result in a halving of the total consumption of that species by marine mammals and hence the ultimate effects of this fishing on marine mammals may be appreciable.

ECOSIM incorporates a “Seek evolutionary stable strategy” routine that can be used to either set *all* vulnerability parameters to the same “evolutionary stable” value or permits estimation of vulnerability values (that make sense in evolutionary terms) for each functional group (Christensen *et al.* 2000). The EwE manual (Christensen *et al.* 2000) notes that the routine that sets a single vulnerability value for all groups usually produces scaled values in the range 0.3 – 0.5, and that this corresponds to the range that produces reasonable<sup>6</sup> ecosystem behaviour when performing simulations. Although several ECOSIM studies include analyses making alternative assumptions, such as that vulnerabilities are proportional to trophic level (e.g. Bundy 2002, Mackinson 2002), the majority of published studies set a blanket value for

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<sup>6</sup> “Reasonable” ecosystem behaviour is described in the EwE manual as “species unlikely to drop out due to predation or competition, predator-prey cycles rare or absent, recruitment relationships with realistic compensatory responses”.

all groups. A useful insight is to be found in the work of Fulton and Smith (2002) who conclude that their model of the Port Philip Bay ecosystem often performs best (in the sense of providing better fits to historical time series data) when prey vulnerabilities for higher trophic levels or heavily depleted groups are high, whereas values for lower trophic levels are in the range 0.4 – 0.5.

The assertion by Walters and Kitchell (2001) (based on unpublished data) to motivate the foraging arena model that “predators with full stomachs are not a common field observation” (and hence are usually hungry and seeking food) is one that remains open to debate. Digestion time constraints likely put a cap on the consumption rates of both marine mammals (e.g. Rosen and Trites 2000) and fishes (e.g. Punt and Butterworth 1995, Jeschke *et al.* 2002, Rindorf 2002, Gill 2003). Moreover, in a study of the effect of stomach fullness on food intake of whiting (in the wild) in the North Sea (Rindorf 2002), the presence of food in the stomach had an appreciable dampening effect on search activity. Although whiting intensify their search for food as their stomach empties, they do not continue searching until the stomach is completely filled (Rindorf 2002).

The foraging arena model is fundamentally different to approaches such as MSVPA and associated predictive models such as MSFOR, which assume that a predator is always able to consume its desired daily ration of food. In contrast, in the ECOSIM formulation, a predator competes with others of the same species for a limited proportion of the prey population which are “vulnerable” to be consumed by that predator species alone. As a consequence, MSVPA and ECOSIM approaches may be categorized as “efficient predator models” and “hungry predator models” respectively (see Chapter 12).

The choice between such feeding interactions in a model can lead to large differences in the model’s predictions. For example, both Mackinson *et al.* (2003) and Koen-Alonso and Yodzis (in press) explore the consequences of alternative feeding interactions and demonstrate greatly different predicted model outcomes depending on the type of functional response formulation being implemented. Koen-Alonso and Yodzis (in press) fitted a trophodynamic model, constructed using a bioenergetic-allometric approach, with five different functional response formulations. The models that provided the best fits to the data shared Type III (which is predator independent) functional responses whereas the ECOSIM

functional response<sup>7</sup> performed the worst, in terms of its Akaike Information Criterion score corrected for sample size ( $AIC_c$ ) (Burnham and Anderson 2002). The ECOSIM functional response could be viewed as a Type 0 (i.e. linear and without saturation – see also discussion in Appendix 11.1) single species functional response (because the predation rate on a prey species is unaffected by changes in the abundance of alternative prey) (Koen-Alonso and Yodzis in press). They underscore the need to scrutinize the nonlinear mathematical structures of multispecies models (such as the functional response) with the same level of rigour as accorded to parameter estimates by traditional fishery modellers. Mackinson *et al.* (2003) have made a start in this regard by evaluating the effects of particular combinations of ECOSIM settings that can be used to produce alternative “emergent” forms of functional responses, specifically Type I and II behaviours. This is discussed further in the following sections.

#### **11.4.3 Dealing with an increasing food supply: a critical look at the food allocation and predation hypotheses**

The maturation of an ECOSIM model into a predictive modelling tool warrants *inter alia* a more thorough analysis of some of its ecological assumptions. This point is illustrated by focusing on attempts by Walters *et al.* (2000b) to rectify one of the more serious conflicts between field observations and model predictions in their first version of ECOSIM II: the model-predicted increase in average adult body weight of an apex predator (for example, tuna) with an increase in fishing rate. Considering the reduced abundance of most apex predators (and many tuna stocks in particular) over the past few decades at least, it is highly implausible that they are currently food-limited and hence that a further increase in predator/tuna fishing rates would lead to an increase in feeding rates and somatic growth. This unlikely prediction from the first version of ECOSIM II arises because reproductive and somatic growth rates are formulated as proportional to *per capita* food consumption. This formulation is presumably also responsible for “interesting” results such as those of Arreguín-Sánchez (2000) who found a predicted increase in the frequency of oscillations in abundance with increasing trophic level.

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<sup>7</sup> The “ECOSIM functional response” as referenced here corresponds to the formulation as given in Equation (11.5) of this chapter. This is the form used by the majority of ECOSIM users so that it has come to be taken as synonymous with ECOSIM. However, as demonstrated by Mackinson *et al.* (2003), by choosing certain parameter values in the EwE model package (e.g. feeding time adjustment rate) in more complex versions of the

Walters *et al.* (2000b) introduce two options to counter strong compensatory increases in somatic growth rate predicted by the first version of the ECOSIM II model. With increasing food supply, animals may either:

- i. allocate surplus to reproduction rather than growth (“food allocation hypothesis”); or
- ii. spend less time foraging so as to decrease time at risk to predation (so that the density dependent response to population reduction would be in the form of a drop in natural mortality) (“predation hypothesis”).

***i) The food allocation hypothesis and its implications for stock-recruitment relationships***

Users of later versions of ECOSIM II specify a “life history weighting factor”  $W_g$  that essentially determines the proportion of net intake (the  $\sum_j Q_{ji}$  term in Equation (11.8)) allocated to somatic growth compared to reproduction as follows (Walters *et al.* 2000b):

$$P_g^{\text{realized}} = W_g P_g^{\text{constant growth}} + (1 - W_g) P_o \quad (11.9)$$

where  $P_o$  is the proportion at the initial equilibrium ( $t=0$ ), and  $P_g^{\text{constant growth}}$  is the proportion which will maintain somatic growth at the same level as initially (i.e. the amount allocated to growth remains the same, but the proportion changes because of changes in the net intake rate). As the weighting factor  $W_g$  changes from 0 to 1, model behaviour shifts from one of these extremes to the other. Recruitment levels at each time step in the model are then determined by a combination of the current status of the adult stock relative to that at time  $t = 0$  ( $N_{A,t}/N_{A,0}$ ), and the ratio of food consumption per unit biomass allocated to reproduction  $(1 - P_g^{\text{realized}})Q_{A,t}$  to baseline ( $t=0$ ) food allocated  $(1 - P_o)c_{A,0}$ . The recruitment function  $R$  is thus scaled relative to baseline recruitment  $R_o^*$  as follows (Walters *et al.* 2000b):

$$R(B_{A,t}, N_{A,t}, Q_{A,t}) = R_o^* (N_{A,t}/N_{A,0}) \left[ \frac{(1 - P_g^{\text{realized}})Q_{A,t}}{(1 - P_o)c_{A,0}} \right]^r \quad (11.10)$$

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foraging arena model (see, e.g. equation A.17 in the Appendix), one can get a different “emergent” functional response type.

where  $Q_{A,t}$  is the adult food consumption (per unit biomass) at time  $t$  (that varies with feeding opportunities as given by equation (11.5) above – see equation A.17 in Appendix for a revised form of equation (11.5)); and  
 $c_{A,0}$  is the baseline adult food consumption (per unit biomass) at time  $t=0$ <sup>8</sup>.

The formulation above includes an option for generating non-linear effects of the food environment on *per capita* recruitment rates (by setting power parameter  $r \ll 1$ ). For simplicity, the discussion below assumes the default value of  $r = 1$ .

The assertion that when food availability increases, adults may increase the proportion of net intake allocated to reproduction at the expense of the proportion (though not the amount) allocated to growth is unlikely to hold unless adults are currently food limited. Given the general trend of fishing down food webs (Pauly *et al.* 1998), it seems unlikely that many commercial fish stocks are currently food limited. Most studies citing density-dependent changes in growth rates in marine systems (e.g. Anthony and Fogarty 1985, Ross and Nelson 1992, Bigler *et al.* 1996) are consistent with the notion that density-dependent effects on somatic growth rate should be most pronounced when stock density is high, and reduced or absent under low stock densities (Ross and Almeida 1986). In contrast, equation (10) above (with default value  $r = 1$ ) depicts the *per capita* rate of reproduction as linearly proportional to *per capita* food consumption for all levels of stock density. The scaling of equation (10) as relative to conditions at the ECOPATH starting “equilibrium” ( $t=0$ ) rather than relative to a measure of “true” base food consumption (from metabolic considerations for example) and/or pre-exploitation population levels merits further consideration.

Equation (10), with  $r = 1$ , can be rewritten in a simpler form, familiar to stock assessment scientists, that shows the relationship between recruits per spawner ( $R/S_N$  where  $S_{N,t} = N_{A,t}$  in the EwE notation) and *per capita* food consumption:

$$(R/S_N)_t = c_1 \left[ c_2 \frac{Q_{A,t}}{c_{A,0}} \right]^1 \quad (11.11)$$

where  $c_1$  is a constant ( $R_0^*/N_{A,0}$ ) that describes the “baseline” number of age 0 recruits per adult at time  $t = 0$  (the ECOPATH starting “equilibrium”); and  $c_2$  is another constant  $\left[\frac{1 - P_g^{realized}}{1 - P_0}\right]$  that effectively scales the linear relationship between  $R/S$  and food consumption.

ECOSIM converts from numbers (of adults) to adult biomass  $B_{A,t}$  using the relation:

$$B_{A,t} = \alpha_{A,t}(C_{A,t})N_{A,t} \quad (11.12)$$

where  $\alpha_{A,t}(C_{A,t})$  is the Ford-Brody growth model intercept, assumed to depend on (total) adult food consumption  $C_{A,t}$  at time  $t$  (Christensen *et al.* 2000).

Rearranging equation (5) to represent *per capita* food consumption and substituting equations (5) (with default setting  $v_{ij} = v'_{ij}$ ) and (12) into equation (11) yields (after some algebra):

$$(R/S)_t = \frac{\alpha_{A,t}(C_{A,t}) * c_1 c_2 / 2c_{A,0} a_{ij} B_{i,t}}{1 + a_{ij} / 2v_{ij} S_t} \quad (11.13)$$

where  $S_t$  is the adult (spawning stock) biomass (of a predator) at time  $t$ .

Note that the arguments hereunder hold also given the updated form of the consumption equation (see Appendix Equation A.17).

For a simple (static) case such that  $w_A = \alpha_{A,t}(C_{A,t}) = \text{constant}$  and with the prey biomass  $B_{i,t}$  held constant, the above formulation is identical to a traditional Beverton-Holt stock-recruit relationship:

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<sup>8</sup> Note that this thesis has retained the conventional EwE notation, but here, as in many other published descriptions of EwE, this notation is somewhat confusing: thus, for example,  $Q$  represents *total* food consumption in equation (11.5), but here represents consumption *per unit biomass*. Note also that  $c_{A,0} = Q_{A,0}$ .

$$(R/S)_t = \frac{\alpha}{1 + \beta S_t} \quad (11.14)$$

where  $\alpha = w_A * c_1 c_2 / 2c_{A,0} a_{ij} B_{i,t}$  (11.15)

and  $\beta = a_{ij} / 2v_{ij}$  (11.16)

Note that the Beverton-Holt model shows compensation i.e. as spawner abundance  $S_t$  decreases, the stock “compensates” by increasing its *per capita* birth rate. In a practical implementation, equations (11.13) and hence (11.14) hold for the ECOPATH starting “equilibrium” with recruitment  $R_0^*$  and spawning biomass  $S_0$ . This provides one relation to determine the  $\alpha$  and  $\beta$  parameters of equation (11.14), viz.:

$$R_0^* / S_0 = \frac{\alpha}{1 + \beta \cdot S_0} \quad (11.17)$$

The further relation required to solve for  $\alpha$  and  $\beta$  is provided by equations (11.15) and (11.16). Input data determine some of the quantities involved, but the analyst has some flexibility of choice for the parameters  $v$ ,  $r$  and  $W_g$ . Before adopting standard defaults for these quantities, it would seem to be important to check that the resultant value for  $\alpha$ , the recruits per unit spawner biomass in the limit of low abundance, is consistent with values typical of the species concerned as determined by the analyses of Myers *et al.* (1999) of empirical recruitment / spawner biomass time series.

Stock assessment scientists are typically more familiar with the steepness parameter ( $h$ ) than  $\alpha$  as characterising the shape of a stock recruitment function. Steepness is defined as the proportion of pristine recruitment ( $R_k$ ) obtained when spawner biomass is reduced to 20% of its pristine level ( $K^{sp}$ ). Thus from equation (14):

$$\left. \begin{aligned} R_k / K^{sp} &= \alpha / (1 + \beta \cdot K^{sp}) \\ hR_k / (0.2K^{sp}) &= \alpha / (1 + \beta \cdot 0.2K^{sp}) \end{aligned} \right\} \quad (11.18)$$

so that values for  $\alpha$  and  $\beta$  imply values for  $K^{sp}$  and  $h$ . Essentially therefore the ECOSIM analyst’s choices for  $v$ ,  $r$  and  $W_g$  amount to independent specification of stock-recruitment

steepness for the species concerned, rather than linking this to empirical data (Myers *et al.* 1999 also provide estimates of  $h$  for the time series they analyse).

It is particularly important to appreciate the implications of the default choice  $v_{ij}^* = 0.3$  for stock-recruit relationships. From equations (11.7) and (11.16) it follows that:

$$\beta = \frac{a_{ij}}{2v_{ij}} = \frac{1}{B_{A,0}} = \frac{1}{S_0} \quad (11.19)$$

Now the parameter  $\beta$  reflects the rate at which recruits per unit spawner biomass decline as abundance increases – from equations (11.14) and (11.19):

$$\begin{aligned} (R/S)_t / (R/S)_{S=0} &= \frac{1}{(1 + \beta \cdot S_t)} \\ &= \frac{1}{1 + S_t/S_0} \end{aligned} \quad (11.20)$$

Thus the default choice for the vulnerability parameter corresponds to the initial spawning biomass being the sole determinant of this aspect of the slope (curvature) of the stock-recruitment function. However the value of  $S_0$  will be primarily determined by the prior exploitation history of the species, rather than (only) biological characteristics. This points to the unsatisfactory nature of adopting the default value for vulnerability for all species in ECOSIM applications. More generally and importantly, it serves to stress that vulnerability parameter values should not be set to the same constant for all species, nor be based on general biological aspects only, but need to take account also of species-specific exploitation histories given the manner in which ECOSIM has been configured to depend upon the initial ECOPATH “equilibrium”.

To better understand why EwE’s “emergent” stock-recruitment function results in stock-recruitment patterns intermediate between the traditional Beverton-Holt and Ricker relationships readers are referred to Fig. 11.4. As is evident from equation (11.15),  $\alpha$  is linearly proportional to prey biomass in ECOSIM, so that an increase in prey biomass results in an upward shift of the curve (Fig. 11.4a). Now the feeding interactions are such that prey abundance will tend to increase as predator numbers decrease. Hence as predator spawner abundance decreases in Fig. 11.4a, the change in prey sees a move from point A to B to C to



D, and hence a tendency towards an overcompensatory curve in which (unlike Beverton-Holt) recruitment initially increases as spawner abundance declines.

However, a problem with EwE is that consumption rates (in total or per predator) are essentially a linearly increasing function of prey abundance (see A.17-A.19 in the Appendix). In contrast to Fig. 11.4a, Fig. 11.4b shows a set of equilibria for which  $\alpha$  is an asymptotic function of prey abundance (as would be expected given physiological upper bounds to *per capita* predator feeding rates). In this case the resultant recruitment curve bears a closer resemblance to a Beverton-Holt than to a Ricker curve. Importantly, enhancement in recruitment as abundance declines is not as appreciable in this case as the EwE formulation in Fig. 11.4a suggests.

### *ii) The predation hypothesis*

Some concerns regarding the inclusion and formulation of the predation hypothesis (the notion that animals spend less time foraging so as to decrease time at risk to predation) are discussed in Walters *et al.* (2000b). The inclusion of a routine that uses an optimisation criterion for setting foraging time is an encouraging development (Christensen and Walters 2000). Moreover, the inclusion of the predation hypothesis routine in ECOSIM has played an important role in highlighting the risks of compensatory decreases in juvenile fish survival and recruitment (Walters and Kitchell 2001).

One criticism of the predation hypothesis is that under changing conditions, for marine mammals in particular, adult natural mortality is conventionally considered one of the population parameters least liable to change. For long-lived mammals, density dependence is thought to first affect the rate of immature survival, then the age of sexual maturity and the birth rate, and only finally adult survival rate (Boyd *et al.* 1995, Gaillard *et al.* 1998, Wade 2002). In pinnipeds, for example, pups may die as a result of being abandoned or separated for long periods from lactating females that are forced to compensate for reduced resources by increasing the duration of foraging trips (Boyd 1999). The primary compensatory mechanism is thus not necessarily an increase in foraging time at lower food densities leading to increased predation risk, which brings into question whether or not EwE's predation hypothesis provides a reasonable proxy for the true underlying processes. For some marine mammals this hypothesis may be adequate given, for example, the effect of sharks in models

of fishery impacts on monk seals *Monachus schauinslandi* (Stevens *et al.* 2000) in the Hawaiian Islands, and that Heithaus and Dill (2002) demonstrated that the distributions of foraging bottlenose dolphins (*Tursiops aduncus*) in Shark Bay, Western Australia, reflect a trade-off between predation risk and food availability. However, it seems unlikely that this applies in the case of marine mammals such as the widely-ranging large whales.

#### 11.4.4 Life history considerations

To model a life history response appropriately, it is important to assess not only *per capita* food availability, but also the relative rates of fishing mortality applied to the juvenile compared to the adult stock pool. Life history theory predicts that an increase in mortality during one age class (*j*) selects for an increase in reproductive effort before and a decrease after the afflicted age class.

Life-history theory predictions have been most strongly tested in elegant field and laboratory experiments with guppies (Reznick and Endler 1982; Reznick and Bryga 1987; Reznick 1989; Reznick *et al.* 1990). When guppies were introduced into streams with a pike cichlid *Crenicichla alta* and other predators that preyed mostly on adult guppies (similar to human fisheries that prey on adult age classes), the guppies matured earlier (i.e. at smaller size), showed greater reproductive efforts/output, and produced more and smaller offspring, than did guppies introduced to streams with a predator that preyed mostly on juveniles. Reznick (1983) notes that fish that faced high juvenile predation and low adult predation, invested in growth at the expense of reproduction - but selection was probably acting on growth rate (faster growth means that fish escape predation earlier), not reproductive rate.

Unlike most single-species models used to provide management advice, EwE simulations are often conducted over periods of several decades and may thus encompass several generations of shorter-lived species. Given a variety of different scenarios in which juvenile or adult predation pressure is high, it may be inadequate to model all life history responses in the same way and some of the life history assumptions upon which the model is based may need to be revisited.

Complex trophic interactions and responses need to be included in and tested in ecosystem frameworks, but it is important to bear in mind that unless they are strongly

supported and validated, they may well erroneously predict both the magnitude and direction of a population's response to a signal. Aydin and Friday (2001) and Aydin (2004) provide further examples of the need for caution in interpreting ECOSIM model outputs in the management arena before more thorough analyses regarding underlying metabolic/life history parameters have been conducted, because of the sensitivity of model outputs to the values of such parameters. In this sense, models such as EwE are quite different from the simpler single-species models often used to provide management advice, as the latter rely on an empirical representation of density dependence, with the associated parameters estimated by fitting to data on abundance levels rather than measured separately.

#### **11.4.5 Cautions in applying EwE to marine mammals and seabirds**

Models such as EwE are customarily constructed using a generic template tailored to fish species which generally produce thousands of eggs and can show dramatic annual differences in productivity. Marine mammals and seabirds (and also a few shark species) have very different life histories from most fish and have a restricted scope to increase reproductive outputs given, for example, that they may produce an average of one young or less per year. Life-history parameters such as fecundity and age-at-maturity differ dramatically between fishes and marine mammals. Thus even though some fish may have the same expected growth rate as a marine mammal or seabird, differences in basic life history parameters mean that the range of distribution about the average is limited for this latter group but can be much more extensive for fish.

Fowler (1981) provides evidence that in species such as large mammals that have low reproductive rates and long life-spans, most density dependent changes in vital rates occur at high population levels (close to carrying capacity) so the population levels providing the maximum sustainable yield tend to occur above the 50% of carrying capacity suggested by the Schaefer model. The reverse is true for species with life history strategies typical of most fishes. In long-lived species such as marine mammals, one expects an appreciable change in natural mortality only when *per capita* prey availability falls below a low threshold value (see earlier discussion under predation hypothesis section). Notwithstanding that others (e.g. de la Mare 1994) have argued against some of the conclusions drawn by Fowler (1981), their posited counter-argument of hyper-compensation remains essentially speculative in nature (Butterworth and Punt 2003a).

The fact that life-history parameters for fishes and marine mammals differ substantially with respect to density-dependent responses is supported further by the recent work of Oli and Dobson (2003) who show that the ratio of fecundity to age-at-maturity (life-history parameters that differ dramatically between fishes and marine mammals) is a good determinant of population growth rates.

#### 11.4.6 Micro-scale to macro-scale representation problems

ECOSIM's foraging arena model structure arises from biology and physics operating at the micro-scale (Walters and Kitchell 2001). Commendable though such insights and attempts to incorporate them in population models might be, there may nevertheless be problems in the associated necessary extrapolation from this micro-scale to the macro-scale levels pertinent to populations.

As an example, Lindstrøm and Haug (2001) found that prey preferences determined at local levels by experiment may not be representative of inputs required for interaction models at the population level because of spatio-temporal variations in predator-prey overlap over the region of interest (Lindstrøm and Haug 2001). Even at the local level, estimates may be biased. Harbitz and Lindstrøm (2001) demonstrate the use of a stochastic spatial analysis framework to derive relationships between expected proportions of prey biomass in the sea and in the diet of minke whales *Balaenoptera acutorostrata*. They illustrate, for example, that while minke whales in the Barents Sea appear to actively select capelin *Mallotus villosus* in preference to other species present, this is no longer the case once the preference of minke whales for foraging in the upper water layers is also taken into account. Interpreting results at the correct spatial scale, as in these examples, is the same kind of problem as ensuring that mechanisms in a model that are based on processes (and parameter) values operating at the micro-scale adequately represent the processes at larger scales.

From a mathematical viewpoint, extrapolations from the micro-scale to the macro-scale require integrating the form of a functional response (the foraging arena model in this case) over the area concerned, with parameter values (such as those in Appendix equation A.17) changing over that area. When the form concerned is linear in the quantities integrated out, the functional form (i.e. shape) is preserved. However, because this is not the case in this

instance, there is no guarantee that the shape of the function at the macro-level, and its predicted density-dependence, will fall within the range of emergent behaviours of ECOSIM's generalised foraging arena functional relationship. ECOSIM users should therefore be aware that in some instances model results based on careful representation of micro-scale processes may nonetheless fail to adequately capture broader scale trends or may even translate into spurious functional response shapes when considered at the macro-scale level. For the same reasons, independent estimates of parameters at the micro-scale will not necessarily remain invariant if the same functional form is assumed to govern macro-scale behaviour.

Modifying ECOSIM's functional response formulation on the basis of behaviour at the micro-scale to represent macro-scale processes may be particularly problematic in instances where this is used as a surrogate for representing other processes. For example, as discussed earlier, consideration needs to be given to the extent to which behaviours such as seal pup deaths from abandonment by cows (under situations of reduced food resources) can reliably be represented by a model assuming that cows suffer extra mortality from predators because they are searching longer for food.

## **11.5 EwE IN A BROADER CONTEXT**

### **11.5.1 Meta-analyses**

Meta-analyses of a range of published estimates from ECOPATH analyses can have considerable import, such as the computations by Pauly and Christensen (1995) of the primary production required to sustain global fisheries. These studies are undeniably interesting and have the potential to shed light on hitherto difficult-to-answer questions, but it is important to recognise that they are built on estimates derived from models reflecting a wide range in the quality of the input data. The comparability of the "steady-state" assumptions in the various models is also questionable when extrapolated to much larger scales. Confidence in the results of meta-analyses would be greatly strengthened if restricted to estimates where robustness to the underlying model assumptions and uncertainties in input data has been demonstrated.

### 11.5.2 Comparisons with single species models

Three processes are primarily responsible for governing the dynamics of populations: I. competition, II. predation, III. environmental variation (Hollowed *et al.* 2000).

- I. Single-species models applied as management tools typically ignore interspecific competition, but generally implicitly include the effects of intraspecific competition, for instance, by including a logistic-type term in the equations to reflect density dependence. Most commercially harvested species are well below pre-exploitation levels, and management is likely neither to try to rapidly recover nor to rapidly further deplete such resources, so change would be slow. Despite well intended but lofty stated general goals, such as that by the World Summit on Sustainable Development 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*" (WSSD 2002), the socio-economic reality in most cases of resources below their *MSY* level is that the large short term catch reductions needed to achieve anything other than a relatively slow rate of recovery would not be politically acceptable. This suggests that at a first approximation one can reasonably assume multi-species impacts not to change. Equally, however, it is not advisable to ignore interspecific competition in situations managed for rapid large changes.
- II. Predation interactions are implicitly included in the mortality terms of most single species models. This approach obviously fails in instances in which predation pressure varies dramatically over time, as may be the case for mid-trophic level species in particular. Ecosystem models have proved useful for exploring the relative importance of fishery and predation mortality effects (Link 1999).
- III. Whereas single species models generally ignore any effect of environmental variation on mortality, the effect of such variations on recruitment processes is often incorporated in a statistical (rather than a causal) framework, e.g. by including stochastic fluctuations about a stock-recruitment curve in contrast to the causal basis of the differential-equation-based EwE.

The above suggests that in terms of describing the dynamics of a commercially harvested species, it is unlikely that approaches such as ECOSIM could afford much of an improvement on single species management tools. The main potential would be to refine estimates (particularly on a temporal scale) of predation mortality. However, because of the greater uncertainty attached to input data required to estimate this source of mortality (as well as

uncertainty about the associated functional forms) in ecosystem models, this will not necessarily lead to an improvement. Christensen and Walters (2000) noted that at that time no substantial improvements in fits to data on population trends had been obtained with the use of ECOSIM (i.e. by explicitly accounting for variation due to trophic interactions).

There are nevertheless a number of instances in which ecosystem approaches have greater utility than single species approaches:

- (1) to improve understanding of ecosystem functioning;
- (2) to derive indices of ecosystem health, e.g. EwE's routine for computing the system particle size distribution (PSD) (Christensen and Walters 2000);
- (3) to try to quantify the impacts of harvesting on other species in the ecosystem, including those of little commercial value;
- (4) to assess the effects of fishing on ecosystem stability (a key point being that ecosystem models introduce the possibility of multiple stable states; this raises the importance of the stability issue, which is trivially straightforward for conventional single species models) and other ecosystem properties;
- (5) to increase understanding of trophic cascade mechanisms (e.g. Daskalov 2002);
- (6) for interpreting the effects of ecosystem-scale changes in productivity resulting from regime shifts (Christensen and Walters 2000);
- (7) for highlighting complex nonlinear responses and interactions. An excellent example in this regard concerns the recommendations for fisheries management proposed by Walters and Kitchell (2001) on the basis of ECOSIM simulations that demonstrate the dangers of compensatory recruitment effects attributable to trophic interactions.

In earlier sections, criticism has been levelled at uncritical use of default values for certain parameters in ECOSIM applications and their general lack of evaluations of sensitivity to such choices. Could not the same be said regarding practice in single-species assessment and management for parameters such as natural mortality  $M$  and stock-recruit steepness  $h$ ? Some important differences apply:

- 1) general (though admittedly not universal) practice in single species analyses emphasises the key importance of exploring sensitivity to input assumptions; such tests will usually be conducted prior to management advice being given, except perhaps in circumstances when experience with earlier analyses has shown such sensitivity to be low;

- 2) there is already a large body of literature (e.g. Myers *et al.* 1999) available upon which to base choices of such input parameter values (or Bayesian priors) by analogy with existing estimates for other stocks of the same species or for similar species; in contrast this is hardly the case at present (nor likely to shortly become so) for key ECOSIM parameters such as vulnerability  $v$ .

These current deficiencies in ECOSIM applications will presumably be addressed once the results start contributing directly to management advice.

### 11.5.3 EwE as an operating model in an OMP/MSE context

One challenge for EwE users is to motivate the utility of this approach to provide the “operating models” of the underlying resource dynamics used to evaluate prospective management approaches to setting harvest limits in an Operational Management Procedure (OMP) (Butterworth and Punt 1999), or analogously Management Strategy Evaluation (MSE) (Smith *et al.* 1999) framework (Cochrane 1998). Although some encouraging attempts have been made explicitly to include multi-species models in MSE analyses (see Schweder *et al.* 2000), practical applications involving fairly complex models such as ECOSIM are often limited by the paucity of data and insufficient system understanding to permit discrimination between radically different harvest policy options (such as whether or not predator culls would prove beneficial to harvesters of their prey) (Butterworth and Punt 2001). In principle, ECOSIM models based on good data could be used as operating models, although it remains to be seen whether or not the associated levels of uncertainty can be adequately constrained to yield scientifically defensible and practically useful conclusions (Cochrane 1998, 2002, Sainsbury *et al.* 2000). Whereas there is clearly an immediate role for ecosystem models as the operating models used to test OMPs, the development of tactical ecosystem models as the basis for computing harvest limits within the OMPs themselves still seems some time off (see Chapter 12).

Increasingly complex management objectives (implicit in the frameworks indicated by Fig. 11.1 B-D) generally require increasingly complex operating models. Apart from the associated dangers of model mis-specification and inadequate parameter estimability, a major difficulty arises with the interpretation of simulation outputs. This is because of the need to consider the trade-offs between performance statistics (such as between average catch levels and inter-annual catch variability) not only for each resource separately, but now also



between resources (Butterworth and Punt 2003b). It should therefore be understood that only relatively simple EwE models based on good data have *immediate* potential utility for this purpose.

Nonetheless, coupling the EwE and OMP approaches should be encouraged because the latter can complement the former through the OMP approach's focus on the identification and modelling of uncertainties as well as in balancing different resource dynamics representations and associated trophic dependencies and interactions (Sainsbury *et al.* 2000). Even so, data limitations will likely mean that only a small subset of EwE models are likely to reach the stage of being considered viable operating models to assist in the management of target species. Following the lead from the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), attention should be focused on the need to account for key levels of uncertainty (Constable *et al.* 2000), preferably within a strategic and practical framework for developing an ecosystem approach to management.

## **11.6 POTENTIAL APPLICATION OF EwE AS A MANAGEMENT TOOL IN SOUTHERN AFRICA AND THE SOUTHERN OCEAN**

The potential of the ECOPATH with ECOSIM approach is considered below in relation to its ability to contribute to answering/addressing five important multi-species questions in these two ecosystems.

### ***I. Hake and fur seals in the southern Benguela upwelling system***

There is much interest and controversy world-wide surrounding the topic of competition between marine mammals and fishers for food resources (see chapter 13), with the various multi-species management objectives put forth spanning the full range of options as indicated by Fig. 11.1. The importance of this topic is illustrated by the fact that the hake fisheries off South Africa and Namibia make up some 50% or more of the landed values of the catches by these countries (Butterworth and Rademeyer in press a). The commercially valuable hake consists of two species, a shallow-water (*Merluccius capensis*) and a deep-water species (*M. paradoxus*), with the larger of the shallow-water species eating the smaller individuals of the deep-water species. The fact that, off the South African west coast, the fur seal population

(*Arctocephalus pusillus pusillus*) is estimated to consume about as much hake as is landed by fishers, begs the question as to whether the hake fishery would benefit in response to a seal cull. Computations by Punt and Butterworth (1995) suggested that a reduction in the number of seals would result in less hake overall because the associated increase in the numbers of shallow-water hake (the preferred prey of seals) would in turn result in increased predation on small deep-water hake, leading to less hake overall.

Can the use of the EwE approach shed further light on this issue (see Fig 11.1 D)? On the positive side:

- i. This approach would allow the inclusion of some essential elements of the analysis, such as split juvenile-adult pools for the hake and the incorporation of cannibalism.
- ii. The likely appreciable sensitivity of model results to the prey vulnerabilities ( $v$ ) assumed would require some careful consideration but is not necessarily a major impediment. In fact it could be a useful proxy for representing some of the spatial structure associated with the dynamics.
- iii. The complexities of the predation/cannibalism interactions pertaining to juvenile hake in particular are difficult to represent in a model, though in the most recent version of ECOSIM (v. 5.0) it is possible to create a set of biomass groups representing life history stages or stanzas for species that have complex trophic ontogeny. The latest version of EwE recommends use of such full age structure population dynamics in place of juvenile/adult age splitting. Mortality rates and diet composition are assumed to be similar for individuals within each stanza. For ECOPATH mass balance calculations, the total mortality rate  $Z$  entered for each stanza-group is used to replace the ECOPATH  $P/B$  for that group.

On the negative side, examples of some of the major problems foreseen are as follows.

- i. ECOSIM with standard default parameter settings cannot yield pure-replacement results in the sense of depicting the effects of a “predator” (a fishing fleet, say, that acts identically in terms of prey selection) being able to increase its consumption sustainably in response to a marine mammal cull (see earlier discussion and also chapter 13). As an example, under the ECOSIM default assumption  $v = 0.3$ , if seals (as predators) become present in high numbers and these numbers are subsequently halved (e.g. through culling), the total consumption  $Q_{ij}$  hardly drops (because  $Q_{ij} / B_j$  nearly doubles) – see Fig. 11.3. Thus rather than providing insight into the effects of a seal

cull on the system, the hard-wiring of the ECOSIM interaction representation means that in this case one could conclude *a priori* that a seal cull will not lead to much increase in sustainable fishery yields. Cooke (2002) reinforces these concerns with the ECOSIM defaults by demonstrating through the use of a simple model that whether or not a reduction in whale numbers results in higher fishery yields than would otherwise, other things being equal, be obtained, depends critically on the assumed vulnerability of fish prey to the whales. It is only under scenarios which assume a relatively high vulnerability of fish to whales that fishery yields are predicted to be sensitive to the abundance of whales.

- ii. EwE assumes that direct interference between predator species is inherently different from within-species interference (see chapter 13). EwE ignores the first of these interactions so that it cannot straightforwardly depict instances where the foraging arena vulnerability pools are used simultaneously by multiple predators<sup>9</sup>. In this specific example, however, there are likely to be appreciable overlaps amongst the vulnerability pools available to seals, to the hake fishery, and to predators which target hake.
- iii. Testing for the effect on the ecosystem of a large seal cull will move the system appreciably away from the initial “equilibrium” situation assumed, and hence the use of some of the ECOPATH-based parameter estimates may no longer be justified.
- iv. The correct choice of representative diet composition data for the various aggregated taxonomic groups is perhaps the major challenge to be met by any multi-species model. Apart from obtaining the information on seal diet composition required for input to an EwE model such as this with sufficient accuracy, there is the additional problem of obtaining this information for all the interlinked species groups included in the model. For example, anchovy is an important prey source not only for hake, but for several other predators too, in addition to its role as an important mesozooplankton predator. In fairness, the criticism that multi-species outputs will be uncertain until there is an improved understanding of how predators respond to changes in *per capita* food availability (Butterworth and Punt 2003b) applies to all multi-species approaches. In response, it could be argued that this is particularly problematic in ecosystem models with their many components, in contrast to, say, “minimal realistic models” as considered in the analysis of this issue by Punt and Butterworth (1995). On the other

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<sup>9</sup> See also discussion under point 1 in Appendix.

hand, the incorporation of greater ecosystem detail enables at least a first attempt at investigating the effect on juvenile *M. paradoxus* of changes in food supply in addition to the changes in predation and cannibalism rates incorporated in the minimum realistic model of Punt and Butterworth (1995). Fulton *et al.* (2003) provide some guidelines, such as advocating the use of functional groups (chosen on the basis of similar body size and shared predators and prey), to achieve an appropriate degree of trophic aggregation.

The explicit incorporation of cannibalism can have a dramatic effect on model predictions, so that it is important to thoroughly and systematically explore both the stabilising and destabilising effects that cannibalism can have on both externally and internally generated population fluctuations. Magnússon (1999) notes that it is model structure that largely influences which of these effects cannibalism will have on a population. Christensen *et al.* (2000) warn EwE users to avoid situations where the fraction of the food for a box taken from that same box (i.e. cannibalism) exceeds 0.10, and advise splitting such boxes into two components instead, such as adults and juveniles.

## ***II. Pelagic fish dynamics in the southern Benguela upwelling system***

Planktivorous predators such as anchovy *Engraulis capensis* and sardine *Sardinops sagax* support a purse seine fishery of considerable economic importance to the southern Benguela upwelling region (Hutchings *et al.* 1998). Some of the important questions pertaining to this fishery that have multi-species connotations include developing an improved understanding (and predictive ability) both for the substantial recruitment variability and for the changes in the relative abundances of anchovy and sardine. Moreover, an important research area that has been identified (BENEFIT/NRF 2002) concerns the development of methods to assess the level of escapement of pelagic fish necessary to maintain populations of threatened and scarce seabirds in the region. Can the EwE approach assist in addressing these questions as they pertain to management (see Fig 11.1 C)?

In this specific context, an EwE model would be inadequate primarily because of the following.

- i. Given the large level of variability about any underlying relationship evident in spawning stock-recruitment plots, any differential equation-based approach

is unlikely to fully capture the highly variable dynamics of species such as anchovy and sardine unless further stochastic components can be added.

- ii. Mesoscale spatial and temporal variability, not included in the EwE approach, is generally considered the scale of variability most appropriate to upwelling systems.
- iii. The EwE recruitment functions are inadequate for representing the large interannual variability in pelagic fish recruitment levels, especially as these are largely attributable to mesoscale spatial and temporal processes. Plagányi *et al.* (2000) demonstrate that measures of absolute prey abundance provide useful correlates of anchovy somatic growth rate only if they are considered within a match-mismatch hypothesis framework (*sensu* Cushing 1990).
- iv. The hypothesis of insufficient food availability in this system affecting the survival rates of immature African penguins *Spheniscus demersus* and Cape Gannets *Morus capensis* (Crawford 2001) is not easily represented in the EwE framework. As alluded to throughout this review, the results of EwE need to be interpreted with caution when relevant to “apex” predators such as marine mammals and seabirds. In the case of seabirds, for example, there is unlikely to be any appreciable response of population variables to increasing food availability above a certain threshold value. ECOSIM presents two options for dealing with scenarios of increasing food supply (see the earlier section: A Critical Look at the Food Allocation and Predation Hypotheses). However, the first option of allocating the surplus to reproduction, with the amount allocated dependent in part on the difference between the current and starting (“equilibrium”) stock status, is problematic in the case of seabirds (and marine mammals). In contrast to fish, these animals can generally produce no more than 1-2 offspring per year so that there is little scope to increase production in this manner. Instead, the effects of increasing food supply for seabirds (and marine mammals) could be modelled using ECOSIM’s second option, that of spending less time foraging so as to decrease time at risk to predation. Again, however, this assumption becomes problematic when applied to an apex predator with few natural predators of its own.
- v. Populations of some seabirds in the region are currently declining (and hence are not stationary), so that it would be difficult to model them within the constraints of the EwE approach. Although a negative biomass accumulation

term could be used in ECOPATH, the effects of the initial imbalance between production and mortality transfers into the initial rates in ECOSIM, and as such is equivalent to pre-setting a condition for decrease (A. Bundy, pers. commn).

Food-web models such as EwE may have limited predictive capabilities in highly dynamic upwelling systems in the light of the predictions of Benedetti-Cecchi (2000) that, in systems characterised by high residual variabilities (and reflected, for example, by appreciable variations about a deterministic stock-recruitment relationship which results from environmental fluctuations), even strong trophic interactions may be insufficient to increase the spatial and temporal variability in the abundance of a species. Food-web and interaction-web models need to recognise the limits of explaining population dynamics in terms of the variance of trophic interactions in situations where the residual variability of a resource is high. The conclusion of Jacobson *et al.* (2001) that environmental effects contribute more variance to the annual surplus production rates in anchovies and sardines than density-dependent effects, suggests that trophic models are unlikely to contribute substantially to the understanding of pelagic fish dynamics.

### ***III. Abalone and rock lobster off the southern Cape coast, South Africa***

The first section of this thesis outlined some of the grave concerns regarding the commercially-valuable South African abalone *Haliotis midae* resource, especially because of considerable takes by illegal fishing activities and an observed recruitment failure that has been attributed to multi-species effects. Diving surveys indicate a dramatic and sustained (since about 1996) decrease in young abalone (5-40mm) in two of the four main fishing zones (Tarr *et al.* 1996). This has been ascribed to an increase in juvenile abalone mortality as a result of a decrease in the abundance of Cape urchins (*Parechinus angulosus*), considered important as providers of shelter from predation for young abalone (Mayfield and Branch 2000). The urchin decrease is in turn attributed to a local increase (an effect of localised immigration, not a trend in the resource size overall) in the abundance of one of their main predators, the rock lobster *Jasus lalandii* (also a commercially-valuable species). Although there is still debate as to the exact causal mechanisms responsible for the dramatic changes in this system, the complexity of these interactions emphasises the need for an ecosystem approach for the management of these stocks (Mayfield and Branch 2000).

Could the EwE approach assist in answering questions such as what effect removing rock lobsters might have on system dynamics, as well as improving understanding of the relative impacts on abalone population dynamics of fishing, poaching (illegal harvests that include substantial numbers of animals below the legal minimum size limit) and this multi-species recruitment effect? This thesis shows that it is difficult to interpret the abalone dynamics without at least some spatial considerations being incorporated in the analysis. There are a number of concerns (in addition to some of those outlined for the two cases above) to be borne in mind if intending to apply an EwE model to this ecosystem.

- i. Abalone, rock lobster and urchin populations in this region are all currently in flux.
- ii. Precise biomass estimates of all of the major components in this system (e.g. kelp, abalone, urchins, rock lobsters) are at best coarse, if available at all.
- iii. Rock lobsters are highly selective feeders and are known to exhibit prey-switching (Mayfield 1998).
- iv. Different sectors of the abalone fishery exhibit different selectivity patterns with age. It may nevertheless be possible to capture something of this effect in an EwE framework, e.g. by scaling the total mortality rate for juvenile abalone upwards to account for the fact that harvests of sub-legal sized animals have a bigger impact on the population dynamics (see chapter 7).
- v. Model outputs would need to be presented over a range of vulnerability ( $v$ ) settings because of uncertainty as to the relative roles of top-down compared to bottom-up processes such as subsidisation of urchins by drift kelp (Day and Branch 2002).

There is nevertheless a need for ecosystem models and their contribution towards improving understanding of complex multi-species interactions such as the abalone-urchin-rock lobster case. The points above are made merely to query whether the specific EwE approach has any utility in the short- to medium-term to contribute to management decisions pertaining to both the abalone and rock lobster resources in the South African region.

One advantage of the EwE approach is that it includes a facility that could be used to simulate (approximately at least) the effect on abalone recruitment of decreasing urchin numbers. EwE incorporates trophic mediation functions, which can be used to modify the vulnerabilities of selected prey types to their predators (Christensen *et al.* 2000). It may thus

be possible to set a threshold urchin density below which the survival of juvenile abalone is limited (Mayfield and Branch 2000), by appropriately sketching a function in EwE in which the vulnerability of abalone recruits is increased in response to a decline in urchin numbers.

The abalone resource is currently managed using a single species age-structured production model (ASPM) that incorporates recent reductions in abalone recruitment in an *ad hoc* manner only (see chapters 6-10). From a management point of view, estimates of quantities such as the maximum sustainable yield (*MSY*) and current replacement yield (*RY*) are of critical concern. The statistical methods employed to fit single species models (such as that applied to the abalone resource) to data on abundance trends ensure that even in cases where model estimates of absolute resource abundance are perhaps incorrect, the productivity of the resource is likely to be reasonably estimated because of the trade-offs in fitting productivity-related parameters such as *K* (pristine biomass), *M* (natural mortality) and *h* (the stock-recruitment function “steepness” parameter). This relates to the fact that robust estimation of Replacement Yield may often be possible, even if other aspects of the model are suspect.

Attempts to obtain an absolute estimate of abalone biomass to refine the management of this resource have failed for a number of reasons (see chapter 2). Given the uncertainty in the biomass of abalone as well as that of coupled species such as kelp, the trial and error method of varying ecotrophic efficiencies to satisfy the mass balance relationships in ECOPATH models is a potential weakness because it does not encourage logically consistent relationships between the values of the productivity parameters. Resource productivity may thus all too easily be under- or overestimated. The latter is of particular concern.

#### ***IV. Post-exploitation recovery of whale stocks off Antarctica***

The substantial reduction through historic overexploitation of large baleen whale populations is argued to have resulted in some 150 million tons of “surplus” krill (*Euphausia superba*) becoming available annually to other predators in the Antarctic ecosystem (Laws 1977). Subsequent probable population increases of several krill-eating marine mammals, such as Southern Hemisphere minke whales, crabeater seals (*Lobodon carcinophagus*) and Antarctic fur seals (*Arctocephalus gazella*) (e.g. Shaughnessy *et al.* 2000), have been interpreted by some as evidence of a temporary large increase in krill abundance. This raises important questions for management such as whether the recovery of species such as the blue whale (*B.*



*musculus*) is being retarded by insufficient food availability due to interspecific competition from species which have increased in abundance. Could the EwE approach assist in answering questions such as whether there are any potential benefits to blue whale stocks if minke whale stocks were to be reduced, or what the effects of a large krill fishery might be (see Figs. 11.1 B and C)? Some points to consider include the following.

- i. Given the dramatic changes over the past two centuries in both Antarctic seal and whale populations, an initial (i.e. *circa* 2000) equilibrium assumption (without any *BA* terms, which would be difficult to specify for many species) seems overly constraining.
- ii. The schooling behaviour of krill is thought to have implications for all aspects of its biology (Hamner and Hamner 2000), and seasonal and spatial match/mismatches that characterise the feeding relationships between whales and their prey means that the choice of an appropriate spatial and temporal framework is critical. ECOSPACE is potentially useful in this regard.
- iii. A major problem with the EwE approach applied to this problem lies in its formulation of the response of vital population parameters to a change in food availability. ECOSIM is capable of simulating increased individual somatic growth rates of minke whales in response to increased food availability, but is unable to represent likely related changes in the average age at transition<sup>10</sup> of minke whales (Thomson *et al.* 1999) and crabeater seals (Bengtson and Laws 1984).
- iv. Another major problem concerns the likely appreciable sensitivity of model conclusions to the choice of vulnerability ( $v$ ) settings for the various krill predators. In this instance vulnerabilities would act in part as proxies for the relative competitive abilities of the krill predators. Thus, because krill is by far the largest component of the diet of the majority of the large vertebrate predators in this ecosystem, changes in krill abundance will likely affect all the top predators, but model predictions will likely be a function of the (*a priori*) input vulnerabilities for the different species. Sensitivity analyses exploring alternative vulnerability settings would therefore be imperative for this particular example.

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<sup>10</sup> The age at transition has been shown to be related to the age of both physical and sexual maturity. It is estimated from whale earplugs, where a change in the pattern of growth layers (the transition phase) occurs at this age (Lockyer 1972, Cooke *et al.* 1997).

### ***V. Antarctic krill and predator requirements off Antarctica***

The Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR) was a watershed in embracing an ecosystem-based approach to management through its acknowledgement (in Article II of the Convention) of the importance of maintaining the ecological relationships between harvested, dependent and related populations of marine resources (e.g. Butterworth 1986). Considering the numbers of Southern Ocean predators dependent on krill as a food source, concerns have been expressed as to the possible impacts on these predators of a rapidly expanding krill fishery (see Fig. 11.1 A). For example, using a combined standardized index (CSI) to represent variability of the marine ecosystem at South Georgia, Boyd and Murray (2001) demonstrated a Holling type II functional response between penguins or seals and their food supply. Modelling procedures such as that applied by Thomson *et al.* (2000) are required to estimate the level of krill fishing intensity that would reduce krill availability, and hence the population of a predator to a particular level. Can EwE contribute to refining management decisions related to the choice of appropriate krill harvest levels under this CCAMLR mandate?

A focus on the South Georgia ecosystem highlights the following issues.

- i. Krill densities exhibit large interannual fluctuations in areas such as around South Georgia and the Elephant Island – Antarctic Peninsula region. For example, Siegel (2000) documents inter-annual fluctuations by a multiplicative factor of about 20 in the biomass in the Antarctic Peninsula region. Moreover, Siegel notes that local or regional population processes such as mortality and recruitment are unable to explain the extent of these fluctuations. They are thought to be caused instead by processes such as advection, the extent of sea-ice cover (see Siegel and Loeb 1995, Murphy *et al.* 1998) and oceanographic variability (Trathan *et al.* 2003). Additional factors such as sea ice age are also thought to exert an important effect on krill population dynamics (Brierley and Watkins 2000). Unless these factors are taken into account, EwE may be unlikely to be able to model krill dynamics satisfactorily, even if appropriate time and space scales (Steele and Henderson 1994) are selected for model development.
- ii. Assuming that the variability in krill biomass could be adequately captured in an ECOSIM model (via coupling with a physical model for example), it then

becomes encouraging to note that there are fairly good correlations between krill availability and the proportions of krill in the diet of some predators (Fig. 11.5), as is implied by the ECOSIM equations. However, of equal importance is that, unless krill biomass is exceptionally low, other predators appear to selectively target krill irrespective of krill biomass (Fig. 11.5).

- iii. An important aim of an ecosystem model constructed for the South Georgia region would be to simulate changes in Antarctic fur seal and seabird (e.g. macaroni penguin (*Eudyptes chrysolophus*), gentoo penguin (*Pygoscelis papua*) and black-browed albatross (*Diomedea melanophris*)) populations in response to changes in krill abundance and hence availability (e.g. due to increased harvest levels). In that seals and seabirds are not fish (obviously!), a potential problem arises in relation to the mechanisms used in ECOSIM to represent compensatory changes in natural mortality rate. If food availability falls below a certain threshold level, seal and seabird populations are likely to exhibit increased juvenile natural mortality largely due to reduced body condition (see e.g. Boyd *et al.* 1995) rather than as a consequence of the ECOSIM options of variable predation risk or changes in adult energy allocation to reproduction. Boyd (1999) shows that female Antarctic fur seals increase foraging trip durations in response to reduced availability of prey. But the primary effect of these increased foraging trips is not to increase the mortality of the adult females (due to predators) but to increase juvenile mortality due to poor body condition. However, there is no apparent appreciable response of population variables to increasing food availability above a certain threshold level (e.g. Fig. 11.6). This is currently difficult to represent satisfactorily in ECOSIM, even through the use of proxies.
- iv. An ecosystem framework is potentially useful to increase understanding of some of the associated complex dynamics in the Antarctic system, such as the competition between salps and krill for phytoplankton (Loeb *et al.* 1997). A second example concerns the suggestion (Agnew *et al.* 1998) that the apparent substantial variations in natural mortality rates of the mackerel icefish (*Champsocephalus gunnari*) at South Georgia are due to variable predation by Antarctic fur seals, in turn a function of krill availability. When krill abundance increases, the seals eat substantially less fish and change back to eating krill. EwE can potentially assist in unravelling some of the complexities

of these interactions, but would be unable to mimic, for example, the change in feeding behaviour of these fur seals.

- v. ECOSIM's vulnerability settings and/or ECOSPACE may have some utility in capturing the probable disproportionate effect of krill harvesting on land-breeding predators with restricted foraging ranges compared to pelagic-habitat predators such as cetaceans (Constable *et al.* 2000).

## 11.7 DISCUSSION

It seems obvious that it will not always be effective or best to base fisheries management policies solely on single-species considerations. However, while the need for multi-species fisheries management has been recognised for a number of years, progress in this field has been impeded by the difficulties in building adequate models which can provide reliable scientific advice. This is due to the complexity of the dynamics of, and interactions between, the various components of an ecosystem and the current poor understanding of these. The ECOPATH with ECOSIM (EwE) modelling approach is currently the most popular and widely applied used to evaluate the ecosystem effects of fishing and to explore the consequences of different management policy options. One needs to ask, however, whether it is up to the task.

### Advantages of EwE

Some of the main advantages of the EwE approach in this context include:

- the ECOPATH framework is useful to evaluate the compatibility of the various data inputs (Christensen and Walters 2000);
- the use of a common framework (generic model) is useful for making comparisons between systems studied by different researchers;
- the ECOPATH constraints act as a rigorous analytical framework (in contrast to an *ad hoc* type model);
- the level of detail included in the model structure and equations is a good attempt at including only those processes deemed most important;
- an improved understanding of ecosystem structure and functioning is achievable with EwE;

- given good input data, EwE has utility to provide a first-order perturbation analysis (Aydin and Friday 2001).

### **Shortcomings of EwE model structure**

Some of the main shortcomings of EwE are summarised in Table 1. The following have been identified as aspects of the actual EwE model structure that may merit further attention or are potentially problematic:

- There are some (apparently not far-reaching) mathematical inconsistencies underlying the ECOSIM interaction term (see Appendix 11.1).
- Predator consumption rates (in total or per predator) are essentially a linearly increasing function of prey abundance, despite the addition of a handling time term (see Appendix 11.1).
- The form of the ECOSIM interaction term involves the questionably realistic assumption that prey vulnerable to one predator are distinct from those vulnerable to another; yet in contrast, the complete prey population is vulnerable to a fishery (see Appendix 11.1 and also comments in Walters *et al.* 2000b regarding the additivity of mortality components).
- The notion that predators are usually hungry and seeking food underlies the foraging arena formulation but is open to debate and highlights the need to explore the consequences of alternative feeding interactions. By choosing appropriate parameter combinations, EwE can generate “emergent” functional response Types I and II, but not III.
- Model formulations based on behaviour at the micro-scale may fail adequately to represent macro-scale processes, particularly in instances where these are used as surrogates for representing other processes.
- ECORANGER is likely computationally inefficient and could be improved.
- The absence of an energetic content parameter (see Equation (11.1)) is problematic in scenarios in which there are substantial differences in the energy density of prey and hence in the food requirements of a predator as prey abundances change.
- The life history assumptions in EwE are flawed in some respects. Sadovy (2001) highlights the need for more work to evaluate compensatory, density-dependent

responses to fishing and notes that such responses are highly variable and may be inadequate to allow full recovery.

In common with most other multi-species models, the constant suitability assumption (of EwE) may be violated under some circumstances, particularly when extrapolating to situations far from the initial model state.

### **Shortcomings of EwE – user applications**

There are also some shortcomings of EwE applications that are attributable to user misuse (or insufficient use!) rather than to the actual model structure. Greater focus on these issues may assist in trying to move the EwE suite closer to the sphere of practical fishery management.

- The development and implementation of statistical procedures for estimating model parameters is one of the major challenges facing ECOSIM (Aydin and Friday 2001). Model outputs need to be interpreted in a statistical framework that summarises the confidence that can be attached to model results/predictions.
- Model properties and behaviour are currently insufficiently explored (in particular for simple scenarios for which such analyses are tractable).
- As noted by Larska and Wootton (1998), quantification of interaction strength (vulnerability settings in this context) permits isolation of a few more likely possibilities from a vast range. Although far from perfect, the dynamic regression approach proposed by Larska and Wootton (1998) may be a potentially useful method in some systems to narrow the uncertainty attached to vulnerability parameter estimates. The approach involves using multiple regression techniques to evaluate the relationship between empirically observed rates of change and species abundances (and hence works best for situations far from “equilibrium”).
- Uncritical use of default parameter settings or setting of vulnerability values to the same constant for all species is unsatisfactory because *inter alia* it assumes the same prior exploitation history for all species and may result in overcompensatory stock-recruitment relationships.
- Model applications that include marine mammals, seabirds and some species of shark need to account as far as possible for the different life histories of these species.

As emphasised by Walters and Kitchell (2000), there is a need to make ecosystem predictions, albeit skeptically, because the only “proof” of a proposed model is to check

whether its predictions stand the test of time. Assumptions such as ECOSIM's additivity in mortality components (see Walters *et al.* 2000b) could similarly buckle or stand with confidence only if given a chance to be tested.

### **Suggestions for users regarding potential pitfalls**

The user-friendly EwE interface introduces positive management possibilities such as the potential to be highly community-based in the sense of fishing community groupings being afforded the opportunity to explore management options using the model (Okey and Pauly 1999). However, because of the potential for misuse of a “black-box type” model such as EwE, users are cautioned against some of the following potential pitfalls in model applications.

- The number of aggregated taxonomic groups included in the model should be a function not only of the need to include important groups, but also of the quality of the available data. The age-old model tenet of “start simple” is recommended. Yodzis (1998) provides a useful example of a way to simplify foodweb models through the removal of “weak links” that do not appreciably affect model predictions.
- Data should be input to the model in a form that as best as possible represents the chosen time and space scales of the model under construction. A common flaw (not necessarily avoidable!) is the use of temporally non-representative diet composition data. This is particularly difficult to achieve if diets are highly diversified and in the case of aggregated taxonomic groups.
- The sensitivity of model results to vulnerability parameter settings is a critical factor to consider in establishing whether model predictions are robust. Given arguments presented here as to the unsatisfactory nature of adopting the default value for vulnerability for all species in ECOSIM applications, the following are suggested as some alternatives:
  - (1) Group-specific values should be sought rather than setting vulnerability to be the same for all species.
  - (2) The model could be run to “equilibrium” under zero catch, followed by iteration on the vulnerability settings to get the inputs to give desired values at the pre-exploitation “equilibrium” stage – so that these are purely biologically determined/related and uninfluenced by catch histories.

- (3) Where time series data are available, Christensen *et al.* (2000) suggest searching for vulnerability estimates that give better “fits” of ECOSIM to such series.
  - (4) Vulnerability values for marine mammals and other top predators should be set at or close to one unless there is evidence to suggest interference competition for food.
- Given the model complexity and the associated uncertainty in the data inputs, a single representation of a system’s structure and dynamics is near meaningless. To be useful in the management context, models and their predictions should rather be presented in the form of a range of likely scenarios.

It is important to bear in mind EwE’s limitations if it is to be used as the basis for management recommendations or decisions (e.g. Pitcher and Cochrane 2002). The details of foodweb structure can make a large difference in predicting the ecosystem effects of fishing (Yodzis 1998). Therefore, to be of use in managing a system, lack of knowledge needs to be acknowledged by calculating probability distributions for a response (Yodzis 1998). One way forward is to account for ecosystem considerations in fisheries management decisions through using statistically sound EwE model implementations as a basis for testing Operational Management Procedure (OMP) candidates (Butterworth and Punt 2003a).

### **EwE’s potential in a local context**

The potential of the EwE approach to assist in addressing five important multi-species management questions pertaining to the southern African and Antarctic marine ecosystems has been assessed above. Although EwE has some potential, especially if used in conjunction with ECOSPACE, problems are foreseen in all cases. The major limitations identified (summarised in Table 11.2) are as follows.

- The ECOPATH-based initial values are considered overly constraining for all five cases. The highly dynamic nature of some systems makes the choice of representative average values disputable. Moreover, although biomass values need not be at “equilibrium” for the reference year (or time period) because of an option to include a rate of biomass “accumulation” (or depletion) (Christensen and Walters 2000), the pre-setting of a condition for increase or decrease (A. Bundy, pers. commn) confounds the identification of the true underlying mechanism. The fact that extrapolations to situations far from “equilibrium” are considered tenuous in EwE (and indeed in any models) is problematic



in an ecosystem management context because it is generally extreme, and not average, ecosystem effects that are likely to be regarded as ecologically important variations to which managers must respond in setting regulations (e.g. Constable *et al.* 2000).

- The density-dependent EwE model formulations appear problematic in most instances, particularly when applied to marine mammals and birds.
- Complex trophic ontogenies are not always easily accommodated in the EwE framework because, for example, even if the diet composition of a juvenile group is well documented, there are problems in explicitly modelling prey species about which less is known. Size-dependent cannibalism interactions that span a number of age-classes are also problematic to include in the EwE framework (though recent development of EwE software may help to address this problem).
- Long-term average predictions, whilst improving understanding, are not always appropriate in management contexts such as in instances where management decisions need to be sensitive to environmental processes operating at the mesoscale level.
- The utility of EwE depends in part on the relative roles of biotic and abiotic factors in mediating ecosystem change. EwE likely has limited predictive capability in systems characterised by high residual variability.

The most serious limitations for any multi-species or ecosystem modelling approach undoubtedly lie in the availability of good representative data and the need for an improved understanding of the complexities of predator-prey interactions. The art of constructing a multi-species model for the purpose of contributing to fisheries management advice entails achieving an appropriate balance between the level of model complexity and the quality and quantity of available data. Marine science has undoubtedly progressed beyond the stage where only single-species assessments are deemed appropriate on the basis of data limitations. However, an immediate quantum leap into a melee of multi-species models is not justified either. An uncalibrated EwE model including hundreds of parameters with unknown values may have utility as a framework for a data gathering and cataloguing exercise, but it clearly should not be relied upon to underpin management advice. At the other extreme, prudent EwE applications that address uncertainty and are based on good data have the potential to make an important contribution to fisheries management advice in some situations.

### APPENDIX 11.1: Technical considerations in applying the “foraging arena” concept to develop the form of the interaction terms in ECOSIM

As discussed in the main text, for prey  $i$  and predator  $j$ , Walters and Kitchell (2001) model the dynamics of the vulnerable ( $V_{ij}$ ) and non-vulnerable ( $N_i - V_{ij}$ ) components of the prey abundance (by number) as:

$$\frac{d(N_i - V_{ij})}{dt} = -v_{ij}(N_i - V_{ij}) + v'_{ij} V_{ij} \quad (\text{A.1})$$

$$\frac{dV_{ij}}{dt} = +v_{ij}(N_i - V_{ij}) - v'_{ij} V_{ij} - a_{ij} V_{ij} N_j \quad (\text{A.2})$$

where the consumption rate  $Q_{ij}$  of prey  $i$  by predator  $j$  is  $a_{ij} V_{ij} N_j$ , and  $N_j$  represents the number of predators.

Summing these two equations yields:

$$\begin{aligned} \frac{dN_i}{dt} &= -a_{ij} V_{ij} N_j \\ &= -Z_{ij} N_i \end{aligned} \quad (\text{A.3})$$

where  $Z_{ij}$  is the instantaneous mortality rate effected by predator  $j$  on prey  $i$ .

$$\text{Thus } a_{ij} N_j = (N_i / V_{ij}) \cdot Z_{ij} \quad (\text{A.4})$$

Substituting into  $V_{ij} = v_{ij} N_i / (v_{ij} + v'_{ij} + a_{ij} N_j)$  (equation 4 in main text) yields:

$$\frac{V_{ij}}{N_i} = v_{ij} / \left[ v_{ij} + v'_{ij} + Z_{ij} \frac{N_i}{V_{ij}} \right]$$

$$\text{or } \frac{V_{ij}}{N_i}(v_{ij} + v'_{ij}) + Z_{ij} = v_{ij}$$

$$\text{i.e. } \frac{V_{ij}}{N_i} = \frac{v_{ij} - Z_{ij}}{v_{ij} + v'_{ij}} \quad (\text{A.5})$$

The standard ECOSIM default:

$$v_{ij} = v'_{ij} \quad (\text{A.6})$$

$$\text{then gives: } \frac{V_{ij}}{N_i} = \frac{1}{2} \left( 1 - \frac{Z_{ij}}{v_{ij}} \right) \quad (\text{A.7})$$

Note that slow dynamics for the  $N_i$  compared to the  $V_{ij}$  implies:

$$Z_{ij} \ll v_{ij}$$

$$\text{so that } \frac{V_{ij}}{N_i} \approx \frac{1}{2} \quad (\text{A.8})$$

## IMPLICATIONS

- (1) The default assumption of equation (A.6) implies that about half the prey population numbers  $N_i$  are vulnerable to predator  $j$ , effectively irrespective of the value of  $v_{ij}$ . This hardly seems consistent with the idea of only a “small” proportion of the prey being vulnerable at any time. This suggests that one should default to a choice of  $v'_{ij}$  rather greater than  $v_{ij}$  to obtain a  $V_{ij}/N_i$  ratio well below 0.5.

A further difficulty of large  $V_{ij}/N_i$  ratios is that the standard ECOSIM interaction term (equation 5 in main text) tacitly assumes that the prey vulnerable to predator  $j$  ( $V_{ij}$ ) are distinct from those vulnerable to predators  $k$  ( $V_{ik}$ ),  $l$  ( $V_{il}$ ), etc. One

cannot consistently maintain a large  $V_{ij}/N_i$  for all  $j$  when necessarily

$$V_{ij} + V_{ik} + V_{il} + \dots < N_i.$$

(2) Prey dynamics are governed by:

$$\frac{dN_i}{dt} = -Z_i N_i = -\left(\sum_j Z_{ij} + F_i + M_i^0\right) N_i \quad (\text{A.9})$$

where  $F_i$  is the fishing mortality and  $M_i^0$  some residual natural mortality other than that caused by the predators  $j$ .

$$\text{Thus} \quad Z_{ij} < Z_i \quad (\text{A.10})$$

and further the assumption of slow dynamics for the prey overall ( $N_i$ ) compared with its vulnerable component ( $V_{ij}$ ) requires:

$$Z_i \ll v_{ij} \quad (\text{A.11})$$

so that:

$$Z_{ij} < Z_i \ll v_{ij} \quad (\text{A.12})$$

Now the vulnerability  $v_{ij}^*$  as input to ECOSIM is a rescaling of the vulnerability parameter  $v_{ij}$  above such that the recommended ECOSIM input default  $v_{ij}^* = 0.3$  corresponds to (K. Aydin, pers. commn):

$$R_{ij} = \frac{2v_{ij}}{a_{ij}N_j} = 1 \quad (\text{A.13})$$

$$\text{or} \quad a_{ij}N_j = 2v_{ij} \quad (\text{A.14})$$

$$\text{However, by equation (A.3): } Z_{ij} = a_{ij}N_j \quad (\text{A.15})$$

$$\text{so} \quad Z_{ij} = 2v_{ij} \quad (\text{A.16})$$

Inequality (A.12) and equation (A.16) clearly contradict each other. This then seems to indicate that the derivation of the ECOSIM interaction term (equation (5) in main text) is invalid for the standard defaults ( $v'_{ij} = v_{ij}$ ;  $v_{ij}^* = 0.3$ ) recommended for ECOSIM. The derivation would hold only in the limit of  $v_{ij} \gg Z_{ij}$ , i.e.  $v_{ij}^*$  close to 1, corresponding to “top-down” control.

This suggests that ECOSIM should be integrating equations (A.1) and (A.2) directly, rather than using equation (4) of the main text. However Walters (pers. commn) advises that computations show that equation (4) provides a reasonable surrogate for the behaviour of equations (A.1) and (A.2) even when  $v_{ij} \sim Z_{ij}$ , i.e.  $v_{ij}^*$  is (possibly well) below rather than close to 1.

- (3). Christensen and Walters (2000) advise that the consumption equation has been amended in subsequent versions of ECOSIM to the form:

$$Q_{ij} = \frac{a_{ij}v_{ij}B_iP_jT_iT_jS_{ij}M_{ij} / D_j}{v_{ij} + v_{ij}T_iM_{ij} + a_{ij}M_{ij}P_jS_{ij}T_j / D_j} \quad (\text{A.17})$$

where  $T_i$  is the prey ( $i$ ) relative feeding time;  
 $T_j$  is the predator ( $j$ ) relative feeding time;  
 $S_{ij}$  are the user-defined seasonal or long-term forcing effects;  
 $M_{ij}$  represents mediation forcing effects; and  
 $D_j$  accounts for handling time limitations on consumption rate by predator  $j$  as follows:

$$D_j = \frac{h_jT_j}{1 + \sum_k a_{kj}B_kT_kM_{kj}} \quad (\text{A.18})$$

Now if some prey population becomes very large (some  $B_k \rightarrow \infty$ ), equation (A.18) implies that  $D_j$  becomes small. The final term in the denominator of equation (A.17) will then dominate that denominator, with consequent cancellations yielding:

$$Q_{ij} \rightarrow v_{ij} B_i T_i \quad (\text{A.19})$$

Thus handling time limitations appear not to restrict consumption rates (in total or per predator): these remain a linearly increasing function of prey abundance  $B_i$ . (Text under “Foraging time and predation risk” in Walters *et al.* (2000b) appears not to resolve this: seemingly this suggests an inverse relationship between (time-dependent)  $v_{ij}$  and  $T_i$ , but such that  $v_{ij} T_i$  would remain constant.)

Table 11.1. Summary of key weaknesses of ECOPATH with ECOSIM (versions II-IV) in the context of its utility as a predictive multi-species tool for fishery management purposes.

Assumption / Problem	Interpreting output in the management context	Possible solutions/developments
“Equilibrium” / “steady-state” as starting point for ECOSIM	Extrapolations tenuous when moved far from “equilibrium” (e.g. because of a change in fishing pressure); inappropriate constraints on parameter values	Full fitting to data sets a partial solution
Parameter estimation in a rigorous statistical setting not routine; precision of parameter estimates seldom reported satisfactorily	Garbage in –garbage out basic tenet of modelling	ECORANGER (Pauly <i>et al.</i> 2000); Inverse methods (Savenkoff <i>et al.</i> 2001)
Aggregated functional groups	Errors in choice of representative diet composition (because of complex spatial and temporal patterns and ontogenetic changes in feeding) may be important	Split pools only a partial solution. Model output must take into account uncertainty in diet compositions. Test sensitivity of models to varying levels of aggregation.
Vulnerabilities of biomass pools	Model results need to be interpreted taking account of their sensitivity to vulnerability settings (which determine the strength and direction of trophic flows)	If the degree of top-down vs bottom-up control is unknown, model sensitivity and robustness to this setting <i>must</i> be explored
Prey-switching	Where important, leads to under- or over-estimation of predicted responses to an environmental pulse	Introduction of stochastic component to dynamics; not important in all contexts
Life history assumptions	Errors in formulations may generate e.g. overly strong compensatory responses (see text)	Thorough testing and validation of assumptions, or limit to inclusion of responses based on accepted state-of-the-art theory
Density dependent effects scaled relative to ECOPATH biomass estimates (numbers at time $t = 0$ ) rather than pre-exploitation levels	Questionable degree of linearity assumed in the response of an aggregated taxonomic group to a change in food availability (see text)	Test the necessity to refine this aspect of the model
Expanding model complexity through addition of e.g. behavioural plasticity effects such as the predation risk avoidance hypothesis	Effect on model predictions may be disproportionately large relative to the importance of the behavioural response included	Careful <i>a priori</i> consideration of whether additional model complexity is justified for the case under consideration
Bioenergetic formulations	Sensitivity to these assumptions can result in overestimates of maximum sustainable yields (MSY) as demonstrated by Aydin and Friday (2001)	Suggestion by Aydin and Friday (2001) to merge bioenergetic models with population mass-balance models
Mesoscale gap	The effects on population dynamics of mesoscale spatial and temporal variations cannot safely be ignored in some fishery management contexts	Proxies could be used such as the seasonal and long term “shaper” interface in ECOSIM II (Walters <i>et al.</i> 2000b) but this is still inadequate in some contexts

Constant physical forcing	Physical forcing important in some contexts. Could result in incorrectly attributing ecosystem changes to predation/fishing effects. On the other hand, including physical forcing in the model is problematic because of constraints on “equilibrium”-based parameters	Inclusion of physical forcing possible with e.g. ECOSIM II “shaper” interface or by linking to e.g. climate models, but a differential-equation-based model will struggle to adequately capture the response to strong physical forcing. Biogeochemical models have been successful under these conditions, but this is partly a reflection of the focus of these models on the dynamics of low to middle trophic groups as compared with EwE, which concentrates more on the higher trophic groups (Fulton 2001, Fulton and Smith 2004).
Linear relationship between recruitment and consumption	Sensitivity to this function may result in overly optimistic recruitment predictions in some circumstances (see text for details). Caution is necessary in interpreting results for systems where other factors are presumed more important in determining recruitment levels	Explore the effect of introducing random fluctuations about the stock-recruit relationship
Constant / knife-edge selectivity function	Problematic in modelling fisheries with a history of large/frequent changes in fishing selectivity or where multi-fleet effects are considered important	Robustness to this formulation should be explored for “nonsplit pools” in particular. Fishing mortality rates $F_i$ may need to be adjusted in an <i>ad hoc</i> manner to account for the disproportionately large effect on an aggregated taxonomic group’s dynamics of selectively removing large older individuals compared to juveniles
Unexplored properties (e.g. regions of dynamic stability)	Maturation of this approach into a predictive multi-species tool is currently impeded by the lack of sufficient explorations (preferably using simple model constructions) of model properties and behaviour. Aydin and Friday (2001) wisely suggest that ecosystem models “must be built, examined and assembled piece by piece”	The properties of these models should be further explored using simple examples. The reconceptualisation (to reduce structural uncertainty) of the ECOPATH approach as an iterative process (Pauly <i>et al.</i> 2000) is a promising development
Easily accessible user-friendly software (black-box approach)	In the absence of alternatives and faced with strong pressures (and indeed the need for) multi-species advice, there is much potential for the misuse and misinterpretation of model applications. Pre-set parameter values are almost certainly not appropriate in all ecosystems. Users and managers need to bear in mind that all ECOSIM models are not created equally! The confidence attached to the predictions from one of these models is in part a product of the quality of the input data and the level of statistical testing	Users should ensure they have a good understanding of what the approach can and cannot achieve (Pauly <i>et al.</i> 2000). Model applications destined for input into management decisions but <i>sans</i> an assessment (statistical or otherwise) of the robustness of the results should be viewed with suspicion. The enabling of a “pedigree” model rating based on the quality of the input data is a step forward (Pauly <i>et al.</i> 2000)



Table 11.2. Summary of some of the main problems/uncertainties in using the ECOPATH with ECOSIM (versions II – IV) approach to address five multi-species management quandaries outlined in the text.

Ecosystem	Southern Benguela upwelling system	Southern Benguela upwelling system	Southern Cape kelp ecosystem	Southern ocean Antarctic ecosystem	Southern ocean e.g. South Georgia ecosystem
Multi-species management question	Hake-seal-fishery interaction	Pelagic fish dynamics	Abalone-urchin-rock lobster interaction	Whale-seal-fishery interactions	Krill predator requirements
<i>Data requirements/relative quality:</i>					
Biomass of major groups of interest (t.km <sup>-2</sup> )	Hake – uncertain; seals OK; non-commercial spp. – poor/absent	Annual average questionably representative for pelagic fish; seabirds OK; other spp. - poor	Abalone, lobsters – uncertain; urchin and other spp. – poor/absent	Whales and seals OK; krill problematic (see e.g. Hamner and Hamner 2000); other spp. – poor	Seals, penguins and krill OK e.g. for South Georgia area. other spp. –poor
P/B (year <sup>-1</sup> ) and Q/B (year <sup>-1</sup> )	Representative values problematic	Representative values problematic	Representative values problematic	Representative values problematic	Representative values problematic
Diet composition	Particularly sensitive to assumptions re seal diet	Representative data difficult to obtain	Abalone OK; sensitive to assumptions re lobster diet	Whales and seals OK as feed almost exclusively on krill in southernmost regions	Data better than most
Flow control (relates to EwE vulnerability parameter $\nu$ )	Uncertain	Some information	Some information	Conclusions will be highly sensitive to these assumptions	Some evidence for bottom-up forcing
Mass balance assumption (i.e. $BA = 0$ )	Problematic	Problematic	Problematic	Problematic	Problematic
Spatial considerations critical?	Proxy may be OK	Proxy may be OK	Preferable to include spatial considerations	Proxy may be OK	Preferable to include spatial considerations
Relative importance of mesoscale processes	Not central concern	Important role	Not central concern	Not central concern	Cannot ignore in this context
Problems associated with density-dependent model formulations?	Yes	Not major	Yes	Yes	Yes
Complex trophic ontogenies problematic?	Yes	Yes	Yes	No	No

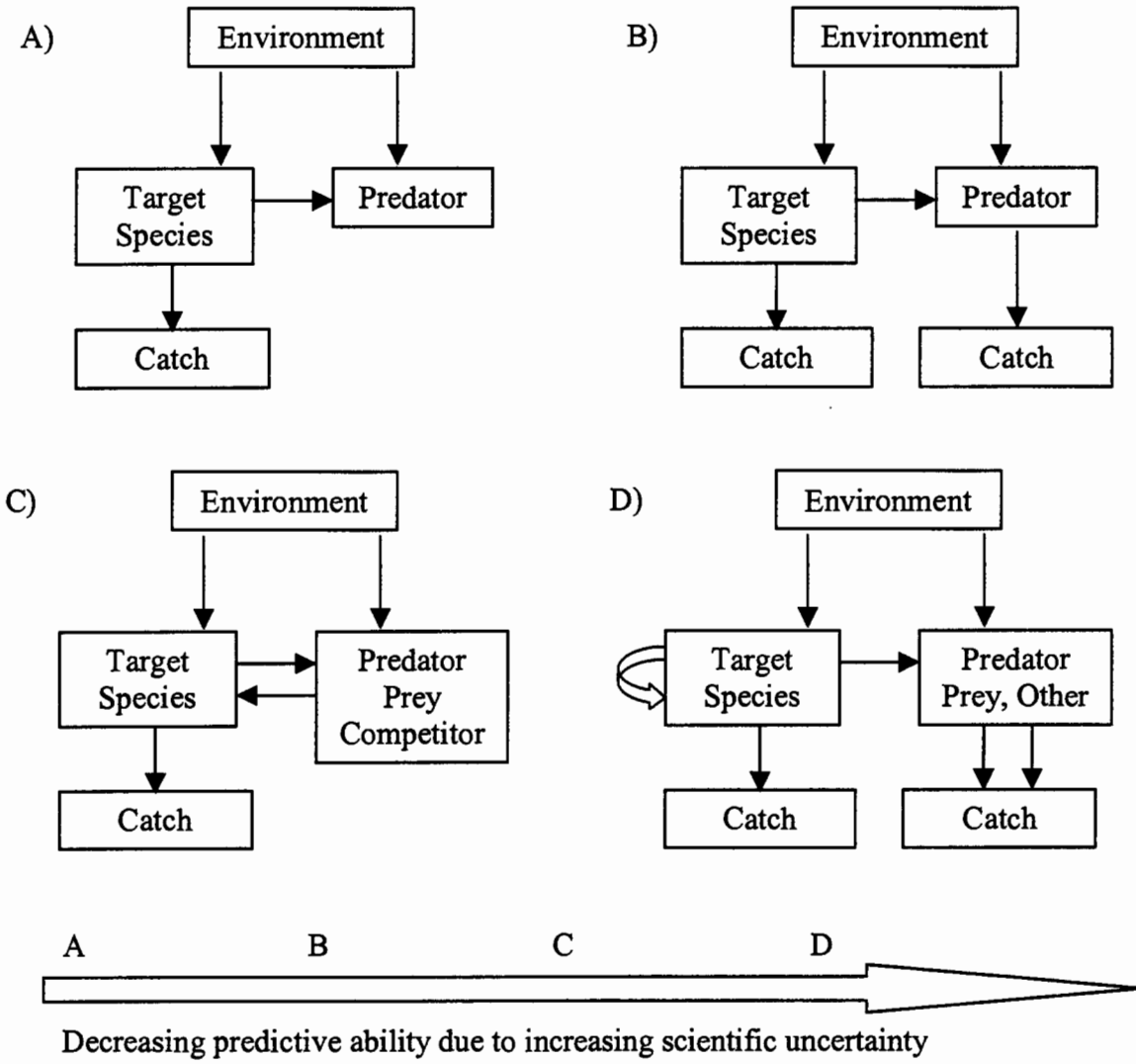


Fig. 11.1. Schematic of an increasing hierarchy of multi-species model complexity to account for biological (as distinct from technical) interactions that pertain to commercially important species. Basic multi-species models (A) may be used to explore how to harvest a target population appropriately, whilst simultaneously accounting for the needs of a predator dependent on that population as prey. If both predator and prey are subject to exploitation (B), it is necessary to simultaneously model both predator and prey populations as functions of physical variability, catch levels and the strength and nature of the functional relationship between the two populations. If an intermediate trophic level species is targeted (in a “wasp-waist” system in particular), it may be necessary to account for the functional relationships between the targeted species and its key predators, competitors and prey items (C). In this case appropriate catch levels are likely to be affected by variability in both upper and lower trophic levels. The most complex multi-species models (D) strive to suggest modifications in the catch level of a species based on the direct and indirect predation and competition effects associated with the simultaneous removal of other food web components. In addition, it may be necessary to consider negative feedback loops such as cannibalism (D). The major disadvantage of increasing model complexity to take better account of biological realism is the associated increase in scientific uncertainty, as a result both of lack of knowledge of functional relationships and of imprecision in estimates of the associated parameter values.

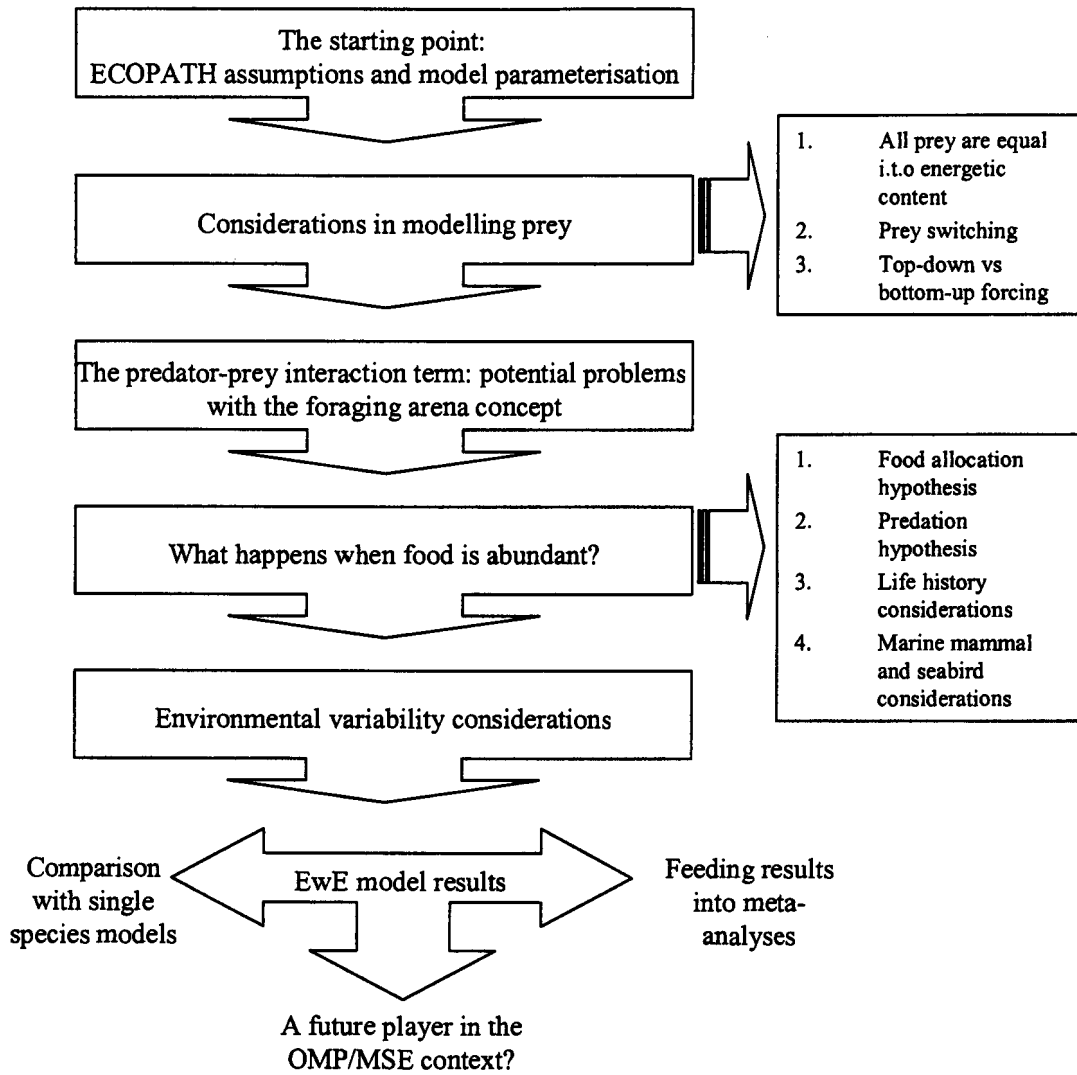


Fig. 11.2. Summary of the structure of this article’s walk through EwE showing selected detours for more detailed discussion.

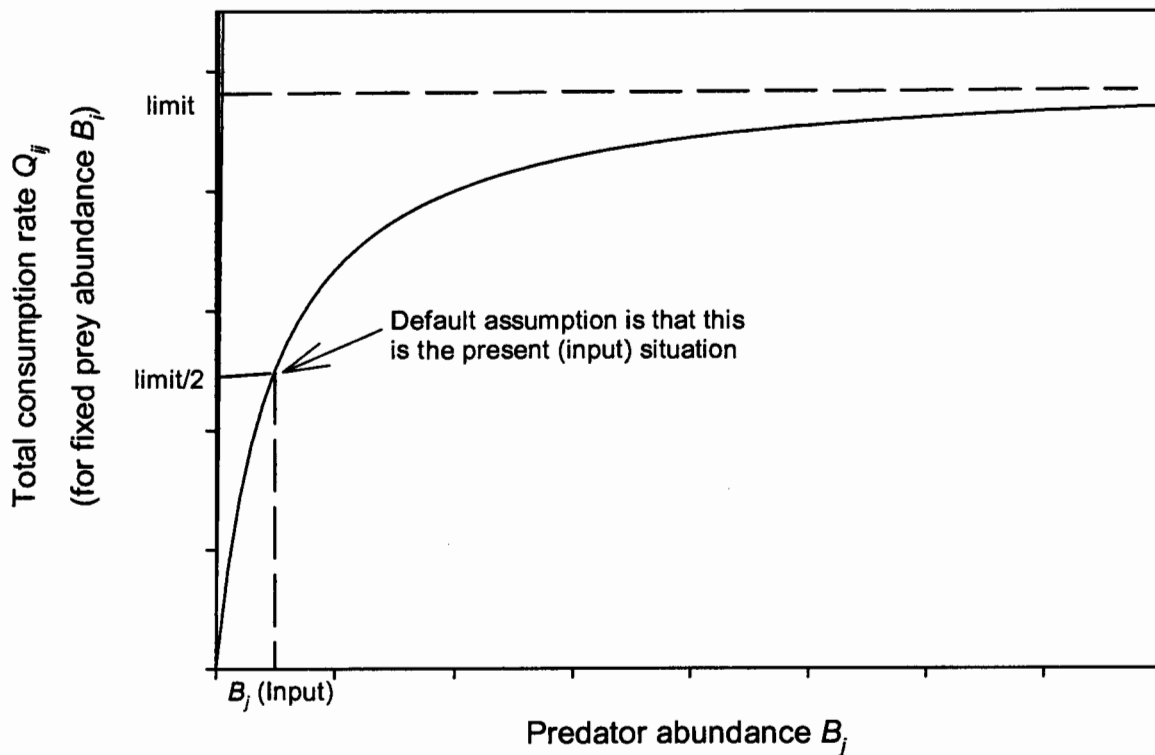


Fig. 11.3. A depiction of the ECOSIM “foraging arena” representation and the implications of default vulnerability settings for model computations of the total consumption  $Q_{ij}$  of species  $i$  by an abundance  $B_j$  of predators of species  $j$ . The curve shown corresponds to the default setting of the vulnerability parameter  $v^* = 0.3$  (see text for details).

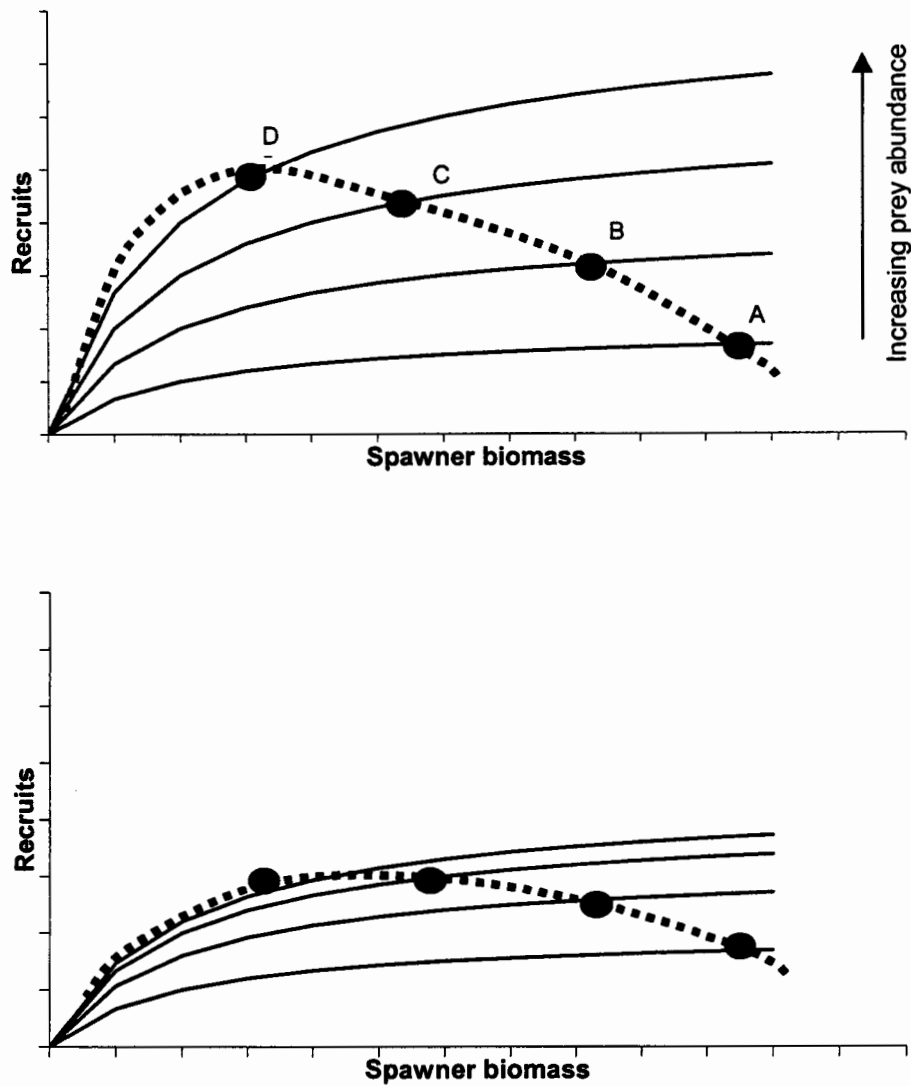


Fig. 11.4. An illustration of the emergent predator spawner stock-recruitment relationship in EwE is given in a). The solid curves reflect the asymptotically flat Beverton-Holt relationships that apply for fixed prey biomass. However, as predator consumption (and hence recruitment under further assumptions) increases linearly with prey abundance in EwE, and because for equilibrium situations fewer predators will correspond to greater prey abundance, the emergent relationship (the dashed curve) shows non-monotonic Ricker-like behaviour with recruitment first increasing as spawner abundance declines. The four Beverton-Holt curves in b) reflect the same prey abundance levels as in a), but here *per capita* consumption is asymptotic rather than linear in prey abundance. In these circumstances the emergent stock-recruitment curve exhibits less compensation as spawner biomass decreases than for a).

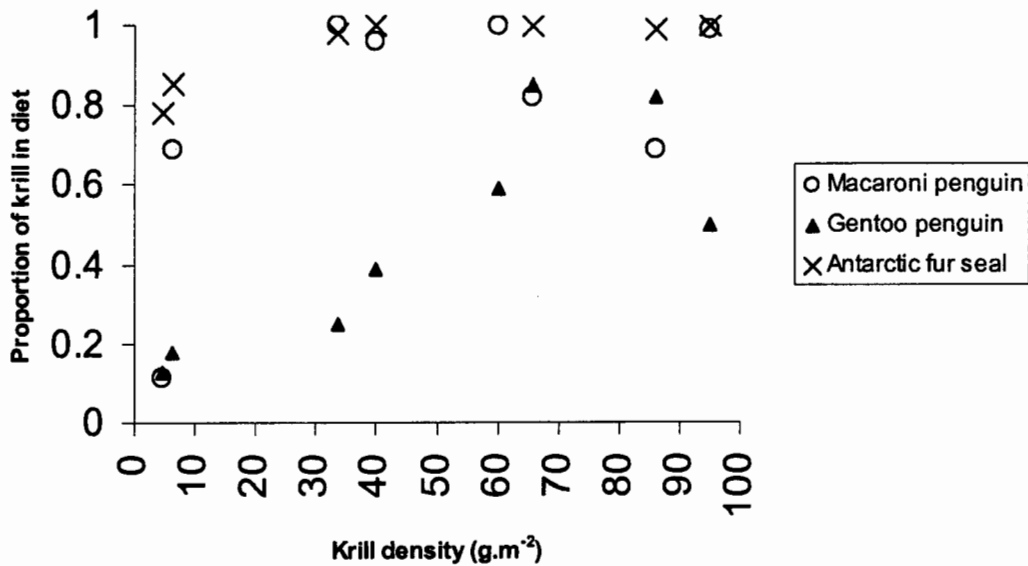


Fig. 11.5. The proportion of krill in the diets of two seabird krill predators and the proportion of Antarctic fur seal scats containing krill (data from Boyd and Murray 2001) as a function of krill *Euphausia superba* density estimates for South Georgia (data from Brierley *et al.* 1999). The data illustrate typical functional responses to changes in prey availability as well as the need for caution in selecting representative diet composition data as inputs to models such as EwE.

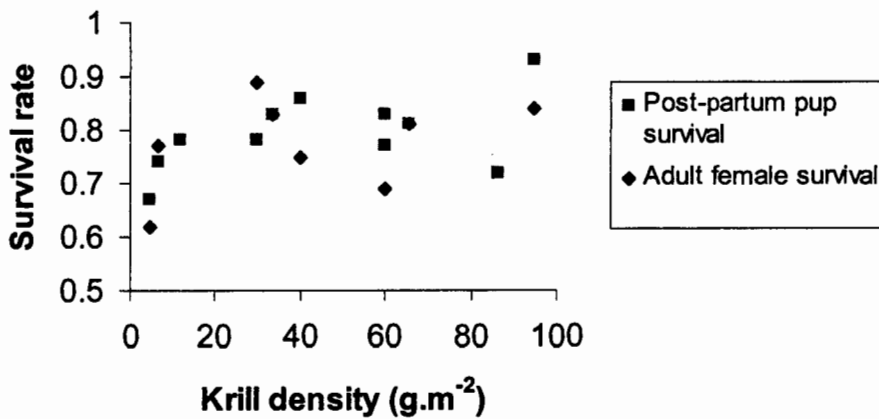


Fig. 11.6. Plot illustrating the non-linear relationship between adult and juvenile Antarctic fur seal survival rates at Bird Island, South Georgia (data from Boyd and Murray 2001) and the availability of krill (data from Brierley *et al.* 1999), the dominant constituent of the diet of lactating adult females. Whereas pup survival rates are considered responses to concurrent estimates of food availability, note that the adult rates may have been influenced by prior events (Boyd and Murray 2001).



## Chapter 12

### **Perspectives on multi-species/ecosystem modelling approaches in the context of their possible application in the management of South African fisheries**

#### SUMMARY

An overview is given of some of the potential approaches to modelling multi-species/ecosystem effects in the context of their possible application in the management of South African fisheries. In particular, various approaches are compared in terms of their potential ability to model both intra- and inter-species interactions between the Cape hakes *Merluccius capensis* and *M. paradoxus* as well as extending this to consider interactions with other components of the ecosystem, most notably Cape fur seals *Arctocephalus pusillus pusillus*. A summary description of the Punt and Butterworth (1995) “minimally realistic model” of the hake-seal system is provided. Suggestions are made for revisiting this work to account for, *inter alia*, changes in and extensions to the data used as inputs as well as a now improved understanding of the distribution and dynamics of the species involved. Some of the research topics identified as being particularly important in this regard include a) analysing hake stomach content data available since the earlier analyses and b) giving consideration as to what are the appropriate functional response formulations to be considered in models of hake-hake and hake-seal interactions. The need to carefully consider the relative merits and costs of the various approaches is stressed. Discussions of the relative merits of the various multi-species approaches deals also with issues of data availability and quality, as well as suggestions for additional data required to extend current analyses substantially.

More generally, five questions are posed related to the potential application of multi-species/ecosystem modelling to advise the management of South African fisheries. Some responses are suggested and relate to the development of a framework for multi-species/ecosystem modelling to contribute to South African fisheries management. Questions posed are: should such models be used for testing or making decisions; do they appreciably reduce uncertainties associated with single species models; are whole ecosystem or minimum realistic models more appropriate; what computer software is best suited to implement such approaches; and what are the overall cost implications? Caution is expressed that general scientific acceptance of the predictive reliability for such models (as required for their use for management) is unlikely in the short term, and will probably require considerable data collection and complex analysis at a not insubstantial cost. The discussions presented are intended to complement a locally-held workshop (December 2002) on the topic “An ecosystem approach to fisheries management in the southern Benguela: introducing the concept and looking at our options”.



## 12.1 INTRODUCTION

This chapter presents an overview of some of the potential approaches to modelling multi-species/ecosystem effects in the context of their possible application in the management of South African fisheries. The previous chapter focused on a detailed review of ECOPATH with ECOSIM (EwE), because it is currently the most widely employed approach to assess the ecosystem effects of fishing. The aim of this chapter is to take a more in-depth look at other multi-species approaches such as multi-species virtual population analysis MSVPA/MSFOR (Pope 1991, Sparre 1991, Magnússon 1995, Vinther 2001) and GADGET (Globally applicable Area-Disaggregated General Ecosystem Toolbox) (see e.g. webpage <http://www.hafro.is/dst2/report2/> ; coordinator G. Stefánsson). In view of the fact that this is such a large topic, it is beyond the scope of this thesis to dissect each of these approaches to the same extent as for EwE in the preceding chapter. The focus of this chapter is once again on the potential of the various approaches to provide information that can usefully be incorporated into *practical* fishery management advice, in the sense of providing at least qualitative and ideally defensible quantitative guidance on the modifications in annual catch levels deemed necessary because of the predicted effects that fishing on a target species will have on other components of the ecosystem.

In the preceding chapter, the potential of EwE to contribute to five important multi-species management questions in the marine environments off southern Africa and Antarctica was assessed. For reasons of brevity, this chapter deals only with the first of these, namely that of the biological interactions among the Cape fur seal and the two Cape hake populations on the South African west coast (Fig. 12.1). The hake example has been chosen because of its importance in a local context - the hake fisheries off South Africa and Namibia contribute some 50% or more of the landed values of the catches by these countries (Butterworth and Rademeyer in press). Moreover, consideration of the indirect interactions between marine mammals and fisheries (see Chapter 13 for a review) is a research field in need of urgent attention, and also an appropriate starting point for developing and testing multi-species models because of the lesser number of foodweb linkages for apex predators (Butterworth and Punt 2003).

Somewhat more time is spent discussing the model of Punt and Butterworth (1995) because it represents a considerable amount of work that has already been conducted in this field in

southern Africa. Their approach involved the construction of a “minimum realistic model” (MRM) of the hake-seal system (Fig. 12.1). This work now needs to be revisited *inter alia* because of changes in and extensions to the data used as inputs as well as a substantially improved understanding of the distribution and dynamics of the species involved. A summary description of the Punt and Butterworth (1995) model is given in this chapter, including some of the major assumptions and inputs. The discussion in each case commences with a brief description of a particular multi-species approach and then some specifics regarding the application of that approach to addressing questions pertaining to the hake-seal system.

The discussion is divided into sections as follows:

- 1) Extension of current assessment models
- 2) Considering predators as an additional fishing fleet
- 3) Punt and Butterworth MRM approach
- 4) ECOPATH/ECOSIM approaches
- 5) MSVPA – lessons to be learnt?
- 6) GADGET as a potential tool

Noting that data constraints ultimately limit any progress in this field, some discussion of data availability, quality and additional requirements is presented. The focus here relates to what could be achieved practically without being completely compromised by issues of uncertainty (a central focus in the single-species stock assessment exercises).

The second part of this chapter takes a broader view that looks more generally at points to be borne in mind in assessing the relative pros and cons of these various approaches perhaps being employed to incorporate ecosystem considerations in the management of South African fisheries. The discussions presented are designed to complement a locally-held workshop (December 2002) on the topic “An ecosystem approach to fisheries management in the southern Benguela: introducing the concept and looking at our options”. The workshop had two objectives, stated as:

- (I) To introduce the concept of ecosystem-based fisheries management to South African fisheries scientists and to present modelling tools to achieve this, in particular the ECOPATH/ECOSIM (Polovina 1984, Christensen and Pauly 1992, Walters *et al.* 1997, 1999) approach; and

- (II) To propose a framework of practical ways in which the incorporation of ecosystem considerations (potentially using information from ECOPATH/ECOSIM and other types of multi-species modelling approaches) into current Operational Management Procedures (OMPs) and other management strategies for South Africa's marine resources could be attempted.

By way of explanation, an OMP is the combination of a prescribed set of data to be collected and the analysis procedure to be applied to these data, to provide a scientific recommendation for a management measure, such as a Total Allowable Catch (TAC), for a resource (Butterworth *et al.* 1997, Butterworth and Punt 1999, Cooke 1999). A key aspect of the OMP approach is that the analysis procedure has been tested across a wide range of scenarios for the underlying dynamics of the resource using computer simulation. This is to ensure that the likely performance of the OMP in terms of attributes such as (high) expected catch and (low) risk of unintended depletion is reasonably robust to the primary uncertainties about such dynamics. This approach is used at present to manage South Africa's three most valuable fisheries: for hake, for pilchard and anchovy, and for west coast rock lobster (Geromont *et al.* 1999).

The preceding chapter and first part of this chapter relate to Workshop objective (I) above. The second part of this chapter seeks to provide an introduction to objective (II) above. It begins by clarifying the different roles played by models in the OMP approach, and raises questions about the costs of the data collection (in particular) needed to apply a multi-species modelling approach in South African fisheries management. This leads to the identification of five questions relating to the framework envisaged in objective (II) above to incorporate ecosystem considerations in the management of South African fisheries.

## **12.2 SUMMARY OF MODELLING APPROACHES AND THEIR POTENTIAL WITH RESPECT TO THE HAKE-SEAL-FISHERY EXAMPLE**

The variety of approaches that have been used for multi-species/ecosystem modelling may be conveniently separated into two groups:

### **1. Whole ecosystem models**

Such approaches attempt to take all trophic levels in the ecosystem into account, from primary producers to top predators. Quite sweeping simplifications and assumptions may

need to be made in this process. Examples are the ECOPATH with ECOSIM (EwE) framework (Chapter 11), which is usually applied in this manner, the models developed by Fulton (2001) and the multi-species individual-based model OSMOSE (Object-oriented Simulator of Marine ecOSystem Exploitation) (Shin *et al.* 2004).

## **2. Minimum Realistic Models**

The Punt and Butterworth (1995) approach was founded in the recommendations of a workshop held in Cape Town in 1991 to develop a basis to evaluate fur seal-fishery interactions off the west coast of South Africa (Butterworth and Harwood 1991). This led to the coining of the term Minimum Realistic Model (MRM) to describe the concept of restricting a model to those species most likely to have important interactions with the species of interest.

### ***12.2.1 Background to the hake-seal example and specific objectives***

The January 2004 BENEFIT-NRF-BCLME (hereafter referred to as BENEFIT) International Stock Assessment Workshop (BENEFIT 2004) aimed to critically review past assessments and management procedure evaluations applied as tools to assist in setting annual catch limits for managing the Namibian and South African hake fisheries and to make recommendations for future research.

A number of possible goals for multi-species modelling / ecosystem studies (centred on hake) were developed by workshop participants. Broadly these can be consolidated into three major goals (BENEFIT 2004):

1. A recent Marine Stewardship Council (MSC) review of the South African hake trawl fishery identified the following condition as one of those for continuing certification: *“Further research (perhaps through expanding the existing modelling approaches) should be undertaken to improve the understanding of ecosystem impacts of the fishery. This should be directed towards the assessment of the capacity of the ecosystem (in terms of productivity and diversity) to recover from fishery-induced impacts. Liaison between ecosystem and stock assessment modelling should be investigated.”* The first goal concerns the Marine Stewardship Council’s (MSC) condition that gaps in the understanding of ecosystem relationships should be addressed by appropriate research.

2. The second goal concerns providing ballpark figures for the implications of the “hake-hake-seal” subset of interactions and more generally to provide a better basis for the evaluation of future OMP’s.
3. The third goal concerns moving towards an Ecosystem Approach to Fisheries (EAF). This broad goal was seen as incorporating several sub-aims such as describing and possibly explaining regime shifts as well as achieving a better understanding of by-catch and damage to non-target species.

The following discussions are restricted to the second goal above – specifically that of considering possibilities for including multi-species effects in assessments, particularly hake cannibalism and inter-species predation, and/or modelling these interactions within a separate framework. As noted in the previous chapter, the commercially valuable hake consists of two species, a shallow-water (*Merluccius capensis*) and a deep-water species (*M. paradoxus*), with the larger of the shallow-water species eating the smaller individuals of the deep-water species. Off the South African west coast, the fur seal population (*Arctocephalus pusillus pusillus*) is estimated to consume about as much hake as is landed by fishers (Butterworth *et al.* 1995), begging the question of whether the hake fishery would benefit in response to a seal cull.

### ***12.2.2 Extension of current assessment models***

As a first step, further work is required to improve understanding of stock structure and distribution. There are three pertinent questions in this regard: 1. Are there separate west and south coast stocks of shallow-water hake (*M. capensis*) off South Africa? 2. Are there separate west and south coast stocks of deep-water hake (*M. paradoxus*) off South Africa? and 3. Are the deep-water hake (*M. paradoxus*) caught off Namibia and South Africa part of the same stock?

The answers to these questions have fundamental bearing on the way in which these populations should be modelled and managed. Thus whereas previous assessments of South African hake have been based on the assumption of separate west and south coast populations (Rademeyer 2003), a recent review (BENEFIT 2004) suggests that it is more likely that there is only a single stock of shallow-water hake and a single deep-water hake stock off South Africa.

Moreover, there are some indications that the deep-water hake off Namibia and South Africa may comprise the same stock, in contrast to the assumption made in present management approaches that they are separate. Thus, for example, it may be necessary to construct a combined SA/Namibia spatial and species-disaggregated model (Butterworth and Rademeyer 2004). The management implications of a shared single stock involve a particularly tricky negotiating exercise with few clear international precedents or guidelines! Several initiatives are currently underway to encourage close collaboration between hake scientists and population geneticists to address stock structure issues, especially those related to trans-national-boundary questions.

In constructing multi-species/ecosystem models for this system, it is therefore important to bear in mind the importance of issues related to hake stock structure. The 2004 BENEFIT Workshop suggested that if Goal 2 (see preceding section) is to be addressed, a first step should be to adapt existing models to provide estimates of the predation mortality on hake that is generated by the two hake species. The next step would involve extending such models to include seal predation on the hake species. Finally, predation by other fish on hake or the effect on hake mortality of including alternative prey could be considered. It was noted that such studies would be essentially hake-centric and aim to provide a better basis to evaluate hake OMPs (BENEFIT 2004).

### ***12.2.3 Considering predators as an additional fishing fleet***

Preliminary explorations have been conducted by including the various predators simply as alternative “fishing fleets”, rather than estimating their effects as part of a “natural mortality” term (which complicates the computations). The approach described is similar to that of Livingston and Methot (2000) and Hollowed *et al.* (2000b) who explicitly modelled predation mortality in a catch-at-age stock assessment model applied to the Gulf of Alaska walleye pollock. They modelled the effect of three predators: arrowtooth flounder (*Atheresthes stomias*), Pacific halibut (*Hippoglossus stenolepis*) and Steller sea lion (*Eumetopias jubatus*) by defining predation mortality as a type of fishery. Two important features of this approach were the use of a flexible functional response form (see Appendix 12.1 for a summary of functional response types) capable of simulating varying levels of predator satiation, and the use of statistical methods to fit the model to the data. Tjelmeland

and Lindstrøm (in press) provide a further example of the incorporation of predators into standard fish stock assessment models. They incorporated predation by northeast Atlantic minke whales in the SeaStar herring stock assessment model and estimated the parameters of the consumption formula by directly including the consumption term in the total log-likelihood function.

The “predators as a fishing fleet” approach was applied in this instance (see also Chapter 9 for an additional application) to a simple representation that incorporated the two Cape hake species as two separate species with *M. capensis* preying on *M. paradoxus* and both of the hake species acting as a predator on juveniles of their own species to emulate the cannibalism known to occur. Seals were included as a separate “fishing fleet” that preyed on *M. capensis*. Each predator was ascribed a selectivity function (based on stomach content data). The two hake species were modelled simultaneously using a simple age-structured production model (ASPM) approach and by fitting to provisional disaggregated GLM standardised CPUE data obtained at that time from Jean Glazer, MCM. Von Bertalanffy growth parameters for the two hake species (and the case in which they are treated as one species) are from Punt and Leslie (1991). There were no age composition data included in the model and the “older” catch data were split assuming that the two species comprised fixed proportions of the catches (Geromont *et al.* 1995). The basic ASPM specifications and parameter values are as described in Geromont and Butterworth (1997).

As in a typical ASPM (see Appendix, Chapter 6), the predator-specific catch by mass in year  $y$  is given by:

$$C_y^{pred} = \sum_{a=1}^m w_a N_{y,a} S_a^{pred} F_y^{pred} \quad (12.1)$$

and similarly, the proportion of the resource harvested each year ( $F_y^{pred}$ ) by predator  $pred$  is therefore given by:

$$F_y^{pred} = C_y^{pred} / \hat{B}_y^{pred} \quad (12.2)$$

with 
$$C_{y,a}^{pred} = S_a^{pred} F_y^{pred} N_{y,a} \quad (12.3)$$

The major challenge in constructing such a model obviously lies in the choice of a suitable interaction term. The simplest way to estimate the predator-specific catch by mass in year  $y$  is to use a Lotka-Volterra-type interaction of the form:

$$C_y^{pred} = a^{pred} B_y^{pred} B_y^{prey} \quad (12.4)$$

where  $a^{pred}$  is an “availability” constant (i.e. the interaction constant). However, this is a particularly strong interaction form and some experimentation was conducted with alternative forms such as:

$$C_y^{pred} = a^{pred} B_y^{pred} B_y^{prey} / (1 + b^{pred} B_y^{prey}) \quad (12.5)$$

which allow for predator satiation. More complicated functional response formulations (such as the various Holling functional response formulations or ECOSIM’s foraging arena formulation) can readily be incorporated in a simple model of this form.

Biomasses of the two hake species were simultaneously estimated in the model fitting process, and initial attempts were made to fit the extra parameters, namely the interaction constants corresponding to each interaction (e.g. estimate  $a^{pred}$  describing predation by *M. capensis* on *M. paradoxus*). Initial investigations suggested that the data were not sufficient to support estimation of (all of) these additional parameters. However, given appropriate data, it may be possible to input estimates of the predator-specific catch by mass in year  $y$  directly e.g. seal predation on *M. capensis* could be fixed in a base-case.

Seals were not modelled explicitly – instead an index of relative abundance was used under the assumption that temporal changes in predation pressure have the biggest effect on multi-species dynamics, i.e. the dramatic increase in seals during the last century means that natural mortality  $M$  cannot be considered as time independent in the case of *M. capensis* (Fig. 12.2). Annual numbers of seals from Butterworth *et al.* (1998) were used as inputs to the model and are shown graphically in Fig. 12.2. Preliminary results indicated that whether or not multi-



species interactions were included in the model (i.e. non-zero interaction constants) had a marked effect on the resultant population biomass trajectories. A single illustrative example for *M. capensis* is shown in Fig. 12.3. Note that this analysis used data available at the time (2000) and hence needs to be updated. However, this analysis is not presented in detail in this thesis but rather is intended as an illustration of an approach to incorporate predation interactions in a stock assessment model.

The development of a simple “Fishing Fleet” type model as described in this preliminary exercise was considered by the BENEFIT Workshop to be a good starting point to address goal 2 outlined above, particularly because it could be based upon current single-species models (possibly length-based). The approach could be improved by building on length-structured models given that most feeding interactions are strongly sized-based. By building these models in a stepwise fashion, they could be extended to achieve greater realism or moulded to provide greater insight into predation-mediated changes in hake recruitment (BENEFIT 2004). The aims of these models would be a clearer understanding of the population dynamics of the two hake species and as a basis for the operating models used to test OMPs.

#### ***12.2.4 Minimum Realistic Model (MRM) approach***

##### **12.2.4.1 Background**

The first so-called MRM (Punt and Butterworth 1995) was developed in response to a need to quantify the potential effect of seals on hake, simultaneously the most valuable fishery in South Africa and Namibia. The MULTSPEC model of the major fish and marine mammal species in the Barents Sea (Bogstad *et al.* 1997) is also of the MRM type, as is multi-species virtual population analysis (MSVPA) (Sparre 1991, Magnusson 1995) and its derivatives which project into the future (e.g. Vinther 2001). MSVPA has been widely applied to species in the North Sea in particular. It is an extension of VPA, which assesses historic population size by summing catches made from different cohorts (year-classes), while making allowance for losses to natural mortality  $M$ , usually under the assumption that  $M$  is a constant. The extension to MSVPA accounts for estimated levels of consumption by major predators in computing these losses to natural mortality, and has led to the important insight that  $M$  is typically much higher during the earliest years of a fish’s life (Pope 1991, Magnússon 1995).

The specification of an MRM raises the important question: what is the optimal level of complexity for multi-species models? Reducing the number of species considered, or aggregating similar species into groups, reduces the number of inter-species links which need to be modelled, but consequently also reduces the number of weak links included in the model. Yodzis (1998) used a food web model of the Benguela ecosystem to show that the exclusion of feeding links representing less than 10% of consumption both by and of any species had minimal effect on model predictions, but that above this threshold for linkage strength the model predictions start to become unreliable.

#### 12.2.4.2 Revisiting the “original” MRM

The Punt and Butterworth (1995) model was restricted to the two species comprising the hake resource, seals, a grouped category of large predatory fish, and the hake fishery (Fig. 12.1). Together these were estimated to account for more than 90% of all mortality of hake. The level of detail taken into account for each component depends on that considered necessary to capture the key aspects of its dynamics. Thus fully age-structured models were used for the two hake species (to capture cannibalism and interspecies predation effects), but the “other” predatory fish components were simply lumped into either a small or large fish category.

One advantage of the Punt and Butterworth (1995) model is that a realistic population dynamics model (Butterworth *et al.* 1995) was used to simulate the seal population, in contrast to the more usual practice of trying to adapt models originally constructed to simulate fish dynamics. A summary of the major features and assumptions of this approach is given in Plagányi and Butterworth (BEN/JAN04/HB/5a<sup>1</sup>). A few features of relevance in the current context are listed below:

- The model is discrete (with half-year time-steps).
- The dynamics of the two hake species are modelled separately using a (modified) age-structured production model. The two species are treated as one in a sensitivity test.
- The model includes both cannibalism and interspecific predation.

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<sup>1</sup> See reference list in Appendix A1 at end of thesis

- Equations (12.6) and (12.7) below include noise terms which were ignored for all the deterministic calculations and handled in a rather *ad hoc* way for the stochastic runs. This aspect could be improved, for example, through the use of Bayesian methods (A.E. Punt, School of Aquatic and Fishery Sciences, University of Washington, pers. commn).
- Natural mortality for hake has four sources:
  1. **Predation/cannibalism by hake:** this is affected by three factors: the number of predators, the number of prey and the “desirability” of different species/age-classes to a particular predator. The daily hake ration of a predator of species  $j$  (either seals, *M. capensis* or *M. paradoxus*) is assumed to be given by a Holling Type II feeding function relationship, as recommended by Butterworth and Harwood (1991), on the grounds of simplicity and availability of sufficient data to allow parameter estimation. The daily hake ration of a predator of species  $j$  and age  $a$  during the first half of the year  $y$  is thus given by:

$$R_{y,a}^j = \tilde{R}_a^j \left( 1 - \exp(-\kappa_a^j V_{y,a}^j e^{\eta_{y,a}^j - \sigma_\eta^2/2}) \right) \quad (12.6)$$

$\eta_{y,a}^j$  from  $N(0; \sigma_\eta^2)$

- where  $R_{y,a}^j$  is the mass of hake consumed each day by predators of species  $j$  and age  $a$  during year  $y$ ;
- $\tilde{R}_a^j$  is the total daily ration for a predator of species  $j$  and age  $a$  (assumed to be independent of prey abundance);
- $\kappa_a^j$  determines the extent of saturation in the feeding function relationship, and
- $V_{y,a}^j$  is the total biomass of hake which is available for consumption by predators of species  $j$  and age  $a$  during the first half of year  $y$ ; and
- $\sigma_\eta$  reflects the extent of the annual variation in the diet.

2. **Predation by seals** – the same form as above.

3. **Predation by “other predatory fish”** (e.g. snoek *Thyrsites atun*, kingklip *Genypterus capensis* and sharks): assumed that the number of hake of species  $i$  and age  $a$  which are eaten by these fish is related to the abundance of such hake by a Holling Type II feeding relationship. The number of hake of species  $i$  and age  $a$  which are eaten is given by:

$$D_{y,a}^{i,predfish} = u_a^i B_y^{opf} \left[ 1 - \exp\left(-v_a^i w_{a+\frac{1}{4}}^i N_{y,a}^i \times e^{\eta_{y,a}^{i,predfish} - \sigma_\eta^2 / 2}\right) \right] \quad (12.7)$$

$\eta_{y,a}^{i,predfish}$  from  $N(0; \sigma_\eta^2)$

where  $u_a^i$  is the maximum number of hake of species  $i$  and age  $a$  which could plausibly be eaten (pre-exploitation level);

$B_y^{opf}$  is the biomass of “other predatory fish”, as a fraction of the pristine level;

$v_a^i$  determines the extent of saturation in the feeding function relationship; and

$\sigma_\eta$  reflects the extent of the annual variation in the diet.

Note that  $u_a^i$  and  $v_a^i$  were pre-specified inputs (sensitivity to their values was examined).

4. **Basal rate ( $M_b$ )** – mortality attributed to “other causes” not included in the model.

This was somewhat arbitrarily set to  $0.1 \text{ yr}^{-1}$ .

Of the many factors considered in the sensitivity tests by Punt and Butterworth (1995), notable changes to the base-case trial were obtained only by increasing the extent of predation by seals on *M. paradoxus*. There thus exists a need to examine more recent data to check the validity of the assumption in the original model that seals feed mainly in shallow waters, and hence that their hake consumption is presumably nearly all *M. capensis*. A second aspect of the Butterworth *et al.* (1995) seal model which may need to be revised concerns the absence of any feedback between a paucity of hake and a population-dynamic response in (for example) weight-at-age, survival and reproduction of seals, i.e. it was assumed that there was always sufficient “other” food for such predators.

The hake model used a Holling Type II feeding function relationship and assumed a fixed daily ration for predators. The way in which the daily ration of a predator is comprised of different hake species and age-classes depends in part on the “desirability” ( $\gamma_{a',a}^{j,i}$  - see eqn. App.II.12 in Punt and Butterworth 1995) which predators of species  $j$  and age  $a'$  exhibit for hake of  $i$  and age  $a$ , as estimated from available feeding data. It is assumed that the observed species-/age-composition of the diet is the same as the pristine species-/age-composition.

Punt and Leslie (1995) computed estimates of diet composition and daily ration for the Cape hakes using information on stomach contents collected during demersal trawl surveys by the SFRI (Sea Fisheries Research Institute – now MCM) between 1988 and 1994. Estimates of evacuation rates for Cape hake stomach content analyses were obtained using a model of the stomach evacuation process and data for juvenile Cape hake and other gadoids. Of interest is that their estimates of evacuation time were notably larger than those used in earlier analyses, suggesting that the time to evacuate 90% of a prey item ranges from 2 to 10 days depending on the meal size and the size of the predator. A key feature of this study was the conclusion that hake inter-meal frequency decreased rapidly with hake size, so that the largest hake were feeding only about once every 10 days. Without this low feeding rate, the model produced a perpetual-fishing-machine - large hake would be so effective at eating small ones, that the harder one fished and removed larger hake, the more smaller hake escaped such predation and became available to make for even larger sustainable fishery catches (D.S. Butterworth, UCT, pers commn)!!!

The fact that digestion time constraints likely put a cap on the consumption rates of hakes is important in discussing the appropriate form of the functional response because, for example, it runs counter to one of the assumptions underlying ECOSIM's functional response formulation, namely that “predators with full stomachs are not a common field observation” (Walters and Kitchell 2001 - based on unpublished data; compare to Table I of Punt and Leslie 1995 which suggests the reverse) (see also Chapter 11 of this thesis).

A summary of the total number of hake stomachs (disaggregated by species and length-class and for years 1988-1994) considered in the Punt and Leslie (1995) study is given in Table 12.1. It is clear that the number of hake stomachs sampled was not exhaustive, contained

biases (as admitted by the authors) and constituted a reasonable sample size only when all data were pooled. Nonetheless, given that there are equivalent stomach content data (disaggregated by species and length-class but currently in raw form) for all years since 1994 (R. Leslie, MCM, pers. commn), it may be possible to derive comparable estimates of daily ration and diet composition for the period 1995-2003. Given that the sampling intensity per survey (6 fish per species per 1-cm length class) has remained the same (R. Leslie, MCM, pers. commn), the number of hake stomach samples for this later period should be similar to that used by Punt and Leslie in their original analyses. This would be useful both to check the extent to which these estimates have changed and/or to use as an updated estimate for, say, 1999 compared to the 1991 estimate.

A potentially bigger problem than straightforwardly updating the “desirability” parameters concerns the fact that these are assumed to be independent of density. This could be addressed to some extent by a more intensive stomach sampling exercise, for example by using techniques to smooth spatial and temporal variability in food composition and predator abundance, such as the geostatistical approach of kriging (Bulgakova *et al.* 2001). A further example of methods used to separate prey size preference from prey availability is given in Floeter and Temming (2003) (who consider North Sea cod).

In summary:

- It would be useful to analyse hake stomach content data for the period 1995-2003 using an analogous method to that applied by Punt and Leslie (1995) to data available for the period 1988-1994.
- If possible, a more intensive stomach sampling exercise should be conducted (over one year say) to improve understanding of differences due to, for example, spatial, seasonal and sex effects.

#### **12.2.4.3 Management Procedure Considerations**

A noteworthy feature incorporated in the Punt and Butterworth (1995) approach involved taking explicit account of uncertainty and management issues through the use of a simulation framework that incorporated the feedback control rules actually in place for setting TACs for the hake fishery. The purpose of this approach was to check whether, even if a seal reduction

did increase hake sustainable yields, the management system applied to compute TACs was such as to be able to take advantage of this. In similar context, Cooke (2002) stresses the importance of considering management constraints and issues of uncertainty as integral components of attempts to assess the effects of changing cetacean abundance on fishery yields. There is currently a paucity of studies addressing this matter in a multi-species context, and the approach of Punt and Butterworth (1995) provides a useful framework for further work in this field.

### ***12.2.5 ECOPATH with ECOSIM***

Considerable work in implementing ECOPATH has already been carried out for the southern African region (e.g. Shannon *et al.* 2000, Shannon *et al.* 2003). Care needs to be taken to accurately reflect current estimates of imprecision in the inputs to EwE analyses. Preliminary results from the application of the MCMC approach to explore whether the constraints provided by the ECOPATH mass-balance equation improve existing estimates of resource abundance and productivity (BEN/JAN04/SAH/5a and BEN/JAN04/NH/5a – listed in Appendix A1) suggest that the Benguela system is not particularly tightly constrained by mass conservation and maximum conversion efficiency considerations, and hence that more work needs to be done to validate and improve model inputs to be able to reduce the uncertainty associated with model outputs. For example, some of the input data used by Shannon *et al.* (2003) are based on a 1989 workshop but seem questionable (e.g.  $P/B$  for seals =  $0.95 \text{ yr}^{-1}$  for a relatively long-lived animal for which natural mortality  $M$  is generally thought to be in the vicinity of  $0.1 \text{ yr}^{-1}$ ). Some of these estimates have recently been revised (L. Shannon, MCM, pers. commn).

The previous chapter provides a detailed summary of some of the potential advantages and disadvantages of applying an EwE model in the current context.

### ***12.2.6 MSVPA approach***

Multi-species Virtual Population Analysis (MSVPA) is a technique that uses commercial fisheries catch-at-age and fish stomach-content data to estimate both the past fishing mortalities and the predation mortalities on some of the chief fish species of interest (see e.g. Sparre 1991, Magnússon 1995). Unlike VPA (Virtual Population Analysis) which assumes

that the natural mortality rate remains the same over time and usually also age, here natural mortality is split into two components: predation due to predators explicitly included in the model ( $M_2$ ) which depends on time and age because of variations in predator abundance, and residual mortality ( $M_1$ ) due to all additional factors which are customarily taken to be constant. Based on the estimates of  $M_2$  that result, forward-looking simulations (MSFOR) are then used to determine the average long-term consequences of changing patterns of fishing.

One disadvantage of this approach is that it requires substantial data pertaining to the predation ecology of the predators included in the model, to the extent that tens of thousands of stomachs were sampled in the North Sea in 1981 and 1991, the “Years of the Stomach”, under the auspices of the International Council for the Exploration of the Sea (ICES). MSVPA applications have thus far focused primarily on the North Sea, with the considerable data requirements generally impeding the application of this approach to other areas, although similar approaches have been applied to the Bering, Baltic and Barents Sea as well as to the Gulf of Maine. Although it is generally considered difficult to sample marine mammal diets, the 1991 ICES stomach sampling scheme included grey seals because scat analyses were used to estimate food suitability constants (“suitability” is an important input to MSVPA, and specifies the relative preference that a predator has for different prey species, if all were present in equal abundances); although far from perfect, this is a pragmatic approach that could be emulated.

A second potential problem with MSVPAs in general is that they concentrate on the impacts of predators on prey but ignore any potential effects that changing prey populations may have on the predators themselves (because of its constant ration assumption – see below). Nonetheless, the approach has some utility in quantifying the relative losses in prey biomass attributable to other predatory fish, marine mammals and commercial fisheries. Moreover, the MSVPA studies have made a start (e.g. Rice *et al.* 1991, Rindorf *et al.* 1998) in trying to determine the extent to which the consumption of a given prey is a simple linear function of its relative abundance in an ecosystem (the constant suitability assumption).

Although there are insufficient data in the southern Benguela region to permit the application of a full MSVPA approach such as that applied in the North Sea (with “attendant hefty price tag” – Hilborn and Walters (1992)), there is the possibility of applying a slightly simpler or even hybrid version. The data intensive requirements of MSVPA could be reduced



(obviously at the expense of increasing model uncertainty) by restricting the focus to a smaller subset of the ecosystem (hake-seals-fishery in this context) and by making various assumptions regarding the length of the time period over which data such as age-length keys and stomach samples are assumed to be adequately representative.

One possibility in the current context would be to use a modified version of the hybrid-type approach used by Mohn and Bowen (1996) to model the impact of Grey seal (*Halichoerus grypus*) predation on Atlantic cod (*Gadus morhua*) on the eastern Scotian Shelf. Their approach involved first running a standard VPA using commercial landings and research survey data, and then adding the consumption of cod by grey seals to the commercial landings and repeating the VPA which was retuned to take grey seal predation into account. They incorporated two alternative models of food consumption by seals (a constant ration predation model in which the fraction of cod in the diet was assumed constant, and a proportional ration model in which the fraction of cod in the diet was assumed proportional to cod abundance), with the two predation models yielding substantially different estimates of the amount of cod consumed by grey seals.

The different functional form of interactions in the foraging arena (per-capita consumption by a predator decreases with the overall abundance of that predator) compared to the constant ration model (per-capita consumption is set equal to the predator's required daily ration) for predator feeding has important implications for model behaviour and predictions. It tends (desirably) to damp the large amplitude oscillations in population size that are frequently predicted by multi-species models (see, for example, Mori and Butterworth 2004). However, this has additional consequences as detailed below.

Butterworth and Plagányi (2004) contrast the assumptions of the MSVPA (and its associated derivatives that provide projections) and ECOSIM approaches, which they categorize as “efficient predator models” and “hungry predator models” respectively. MSVPA assumes that a predator is always able to consume its desired daily ration of food. If  $N_j$  is the number of predators of species  $j$ , and the number of their prey species  $i$  ( $N_i$ ) is kept fixed, then Fig. 12.4a shows the implication of the MSVPA assumption for how the total consumption rate  $Q_{ij}$  of prey  $i$  by predator  $j$  grows as the number of predators increases: linear proportionality.

On the other hand, as explained in the previous section, ECOSIM is based upon the foraging arena model (Walters *et al.* 1997) which leads to the form of relationship between total consumption rate  $Q_{ij}$  and the number of predators  $N_j$  as shown in Fig. 12.4b. An MSVPA model is almost certainly not adequate on its own as a tool for investigating hake-seal-fishery interactions, but together with an ECOSIM model, these two approaches can at least make a first attempt at bounding the likely impact on a fishery of a reduction in seal numbers in that, based on the assumed forms of interaction, the former approach is likely to overestimate the effect and the latter to underestimate it (at least when using default or low vulnerability settings) (see Chapter 11). The data hungry nature of MSVPA does not necessarily preclude the use of MSFOR to predict forwards, provided the model is initialised using sensible assumptions based on at least some data (IWC 2004).

### ***12.2.7 GADGET as a potential tool***

In Chapter 13 it is argued that GADGET shows great promise as a tool for modelling marine mammal–fishery interactions and has been recommended for such (NAMMCO 2003).

One of the conclusions arising from a workshop held by the Scientific Committee of an international marine mammal commission during 2002 (NAMMCO 2003) was that while the output from a model such as GADGET was not expected to be able to predict all aspects of future states of the ecosystem, the model was seen to have potential utility for management through testing scenarios where abundances of target species are manipulated. In addition, the workshop recommended the development of a generic (or “template”) North Atlantic model, based on GADGET, and including major fish and marine mammal species. The main use of such a model was seen to be to identify the inputs which had the greatest effect on model predictions, and hence to guide research priorities in different regions each subject to different deficiencies in data.

## **12.3 DATA ANALYSIS**

The summary below is intended to relate predominantly to a) hake-hake and b) hake-seal biological (feeding) interactions rather than a more general summary of data inputs and

modelling research topics for all the inter-related species in the southern Benguela region. Below follows a summary of some recommendations for possible research topics:

#### 1) Hake feeding studies

- Importance of analysing hake stomach content data for the period 1995-2003 using an analogous method to that applied by Punt and Leslie (1995) to data available for the period 1988-1994.
- Obtain updates of “desirability” estimates.
- Investigate feasibility of a more intensive stomach sampling program (over a limited time period), in particular to try to quantify prey preference as a function of prey density/availability.

#### 2) Other demersals feeding studies

- Although some work has been done in this area (e.g. Pillar and Barange 1998), there is a current paucity of studies describing aspects of the feeding behaviour of demersal species (other than hake) in the Benguela upwelling system.

#### 3) Seal feeding studies

- Identify biases in the data due to seals scavenging from trawlers.
- Investigate whether hake consumption by seals is nearly all *M. capensis*.
- Update estimates of the fraction of hake in the diet of seals.

### 12.4 ANALYSIS AND MODELLING ISSUES – SYNTHESIS OF KEY POINTS

One of the key issues in attempting to model hake-hake and hake-seal interactions is the appropriate form of the functional response formulations to be considered in the models. Mackinson *et al.* (in press) note that most multi-species models utilize a hyperbolic (Type II) functional relationship. However, they caution that such an assumption could overestimate the effect that generalist predators, such as whales, exert on depleted prey stocks. As a consequence, they suggest that a sigmoidal (Type III) functional response might be more appropriate at lower prey densities (although difficult to implement because additional parameters need to be estimated). There is a need for careful consideration of the most appropriate choice of model form to represent feeding behaviour: at opposite extremes,

formulations such as that used by ECOSIM depict per-capita consumption by a predator as decreasing with increasing overall abundance of that predator, whereas constant ration formulations (such as that used in MSVPA approaches and the Punt and Butterworth MRM approach) set per-capita consumption as equal to the predator's required daily ration.

Walters *et al.* (2000) advance two arguments to support the foraging arena over the constant ration model, but there appears to be little observational evidence to distinguish the two models, and IWC (2004a) describes the biological underpinnings of the foraging arena model as “controversial and uncertain”. Clearly therefore, evaluations to provide advice on the impact of, say, a predator cull on fisheries for prey species cannot be based on the foraging arena model alone as a representation of species interactions; rather the robustness of results across a range of plausible functional forms needs to be considered.

A further difficult issue in multi-species modelling is that of estimating prey suitability. Experimental estimates of suitability often refer only to the microscale, but multi-species models require parameter values that reflect effective responses at the macroscale level (Lindstrøm and Haug 2001). Reliable integration of microscale estimates of suitabilities over the spatio-temporal distributions for both predators and prey to provide macroscale parameter values, is likely a realistic objective for the longer term only; in the shorter term, regression approaches will probably be needed to attempt to relate macroscale changes in diet to variations in prey abundance. Studies comparing the performance or predictions of models representing processes at different scales and / or with different levels of spatial aggregation can also be informative (Fulton *et al.* 2003).

As highlighted by an international review panel at the 2004 BENEFIT Stock Assessment Workshop (BENEFIT 2004), the choice of which multi-species models to use needs to be linked to scientific goals and / or management objectives. For objectives related to broad-scale questions regarding the structure of the ecosystem ECOPATH / ECOSIM models might be used; other models may be more appropriate for more specific questions. Unlike EwE, individually tailored approaches such as MRMs have more flexibility in modelling the dynamics of marine mammals, but usually ignore any potential effects that changing prey populations may have on the predators themselves. Fulton and Smith (2004) strongly recommend that ideally a suite of different “minimum-realistic” marine ecosystem models should be constructed and their results compared. However, given limited person-power and

pressure to produce results, it is important to engage in discussions regarding which are the preferred modelling approach/es to be pursued in each context. Thus, for example, as a first attempt to address hake-multi-species interactions, the 2004 BENEFIT Workshop recommended that existing models should be adapted to provide estimates of the predation mortality on hake that is generated by the two hake species. Similarly, CCAMLR has tended to consider simpler predator-prey type models for the Southern Ocean (e.g. Thomson *et al.* 2000).

Nevertheless, EwE clearly has an important role to play, given that none of the other models discussed in this chapter are suitable for exploring broader ecosystem questions. It may be instructive to investigate possibilities of closer links between ECOPATH data inputs and single-species stock assessment models. In considering ECOPATH's potential to contribute to single-species models, there is a need to pursue the question of whether the constraints provided by the ECOPATH mass-balance equation appreciably reduce uncertainties associated with single species models. On the other hand, outputs from single-species stock assessment models may have some utility for improving biomass and productivity estimates (and their associated variance estimates) used as inputs to ECOPATH and hence ECOSIM. Recent additions to the EwE software (Christensen and Walters 2004) mean that it may be possible to experiment with including more life history stages of hake in an updated ECOSIM model.

Regarding MSVPA, consideration could be given to implementing a simplified less data-intensive version of MSVPA that is focused on hake-seal-fishery interactions. For example, limited stomach sampling data could be assumed representative for several years when computing predator-prey suitabilities. The most pragmatic implementation method may be to modify an existing ASPM-based model to include some of the main MSVPA assumptions.

The development of GADGET is an ongoing process and as yet no work has been done in attempting to apply it to the example discussed here.

The agreed conclusion of the IWC's Scientific Committee following discussion of the report of its Workshop (IWC 2004a) provides some useful insights and reads:

“for no system at present are we in the position, in terms of data availability and model development, to provide quantitative management advice on the

impact of cetaceans on fisheries, or of fisheries on cetaceans. However, this does not rule out the possibility of providing qualitative advice if a number of different approaches yield qualitatively similar results.”

## 12.5 ROLES FOR MODELS IN OMPs: TESTING vs DECISION

Models play two very different roles in the process of developing and implementing OMPs.

“*Decision Models*” essentially integrate information available from monitoring a resource (e.g. *CPUE*, survey indices of abundance) together with a control rule to provide a scientific recommendation for management such as a TAC. Typically these are simple population models, fitted to data which indicate trends in abundance, that seek to provide “feedback correction” (e.g. if future data indicate a downward trend in abundance, the TAC is reduced to attempt to arrest and reverse this trend). Some *Decision Models* are even simpler than this: for example, the annual TAC for the South African directed pilchard fishery is set as a fixed proportion of the hydroacoustic estimate of abundance forthcoming from an annual spawning biomass survey, where computer simulation testing was used to advise the choice of the specific proportion set (De Oliveira *et al.* 1998, Geromont *et al.* 1999).

*Decision Models* are not intended to necessarily provide an accurate representation of the possible underlying resource dynamics. Rather the basis for their choice is that computer simulation tests show them to be likely to provide robust achievement of the objectives sought by the management authority.

In contrast “*Testing Models*” (which are often termed “*Operating Models*”) do seek to accurately reflect alternative possibilities for the true underlying dynamics of the resource or resources under consideration. They may seek a high degree of realism, and hence may be quite complex (e.g. IWC 2003). The role of these *Testing Models* is to provide the basis for computer simulations to project resource trends into the future to test how well alternative candidate *Decision Models* achieve the objectives sought by the management authority.

What potential role do multi-species/ecosystem models have as *Decision* and as *Testing Models*? It will likely be many years before such models might be used as *Decision* models for OMPs, primarily because of the uncertainty surrounding appropriate choices for the

numerous parameter values and the functional forms to describe species interactions, as reiterated throughout Section II of this thesis. Rather it seems that *Testing Models* are the more appropriate level at which multi-species/ecosystem models might first be used under the OMP approach. To date, such implementation in OMP evaluation exercises generally has been implicit only. For example, in the computer simulation testing of its Revised Management Procedure, the Scientific Committee of the International Whaling Commission (IWC) did not develop testing models with multiple interacting species; instead it allowed for time-dependence in the intrinsic growth rate and carrying capacity parameters of the testing model for the population under harvest, to mimic the typical impacts on that population of changing levels of other predator and prey species (IWC 1989).

## 12.6 COSTS

A word of caution seems appropriate before decisions might be taken to advocate greatly enhanced multi-species/ecosystem modelling efforts to contribute to South African fisheries management. Single-species models rely primarily on time series of catches and abundance indices for the population concerned. Multi-species models require in addition not only dietary information for that population, but also abundance and dietary data for its major predators and prey. This can add substantially to research and monitoring costs, and this needs to be justified by likely resultant improvements to management advice.

A recent review of stock assessment needs for fisheries in the USA (NMFS 2001) provides some figures which assist in placing this aspect in perspective. By way of background, in 1999 a US National Marine Fisheries Service Report to Congress on the Status of Fisheries (NMFS 1999) listed 904 marine stocks, but information sufficient to assess status is available for only some 40% of these. Only 119 of these US stocks are routinely assessed at the state-of-the-art single species level, although this group does include most species of high value, volume or profile.

NMFS (2001) identify further research vessel surveys and observer programmes as the two most important needs to enhance scientific advice for fisheries management in the USA. For a balanced overall approach, they group the areas requiring further resources into three categories, with the following increases required to the present US fisheries research budget:

Improve stock assessments using existing data:	11%
Elevate stock assessments to new national standards of excellence:	32%
Next generation assessments:	25%

Such an overall 68% budget increase corresponds to some US \$90 million per annum<sup>2</sup>. It is only the “Next generation” assessments which are envisaged to “explicitly incorporate ecosystem considerations such as multi-species interactions and environmental effects, fisheries oceanography, and spatial and seasonal analyses”. The budget estimates above assume that this would be attempted only for “core species” – of the order of 20-40 stocks for the USA as a whole.

Coarsely comparing South African to US fisheries, one would be dealing with approaching an order of magnitude fewer stocks, and costs are lower. But even taking these factors into account, the figures above suggest that full incorporation of ecosystem considerations into the assessment and management of, say, the half dozen or so most important South African fisheries could entail additional annual research costs of the order of a few tens of millions of Rands.

Clearly therefore, research priorities in this area need to be carefully and realistically chosen, and weighed against other research needs, such as the further research vessel surveys and observer programmes stressed by NMFS (2001), for improved management of South African fisheries. Nevertheless, these priorities also need to be assessed in the light of various policy obligations, by South Africa and other nations, to broaden the current focus on target stocks to include consideration of the ecosystem effects of fishing (Gislason *et al.* 2000).

## 12.7 A FRAMEWORK FOR SOUTH AFRICA

As outlined in the Introduction section of this chapter, Objective (II) of the locally-held “ecosystem approach” workshop was to develop a framework to incorporate multi-species/ecosystem modelling research in South African fisheries management (particularly in an OMP context).

<sup>2</sup> These figures relate to staffing and minor associated infrastructure needs only. If other costs such as vessel time and major infrastructure are added, the US \$ 90 million sum quoted virtually doubles (P. Mace, U.S. National Marine Fisheries Service, pers. comm).



Arising from the issues discussed above, five questions that need to be addressed in this development are offered, together with brief associated commentary.

- 1) *In an OMP context, is the immediate role for multi-species/ecosystem models as “Testing” or “Decision” Models?*

As elaborated above, an appropriate initial role seems to be as “Testing” (Operating) Models.

- 2) *Do mass-balance constraints appreciably reduce uncertainty about current single-species management model estimates of abundance and productivity?*

While predictive multi-species population models may have limited management impact in the short-term, if only because of considerations of lack of data, model complexity and uncertainty, and research costs, there are some initiatives that are being pursued with the information that is to hand at present. The mass-balance relationships of the ECOPATH approach (Christensen and Pauly 1992) provide some information beyond that conventionally incorporated in single-species assessments, and do so essentially independent of concerns about how best to model the functional forms of species interactions. However, as explained under Section 12.2.5, preliminary computations (BEN/JAN04/SAH/5a and BEN/JAN04/NH/5a – listed in Appendix A1) suggest that for the Benguela system, the precision of single-species assessment estimates is unlikely to be improved through taking account of mass-balance constraints.

- 3) *What immediate relative emphasis should be placed on “Whole Ecosystem” vs “Minimum Realistic Model” analyses?*

The view expressed in this thesis is that until “Whole Ecosystem” approaches have been shown to demonstrate adequate robustness in their predictions to uncertainties in input data and alternative plausible choices for the functional forms of interactions between species, they should have lower priority than the development of Minimum Realistic Models, given an aim of providing inputs to local fisheries management advice. At the very least, MRMs would seem the obvious first step to take in the process of moving from single-species models to the extremely ambitious and demanding aim of a reliable predictive model for all major ecosystem components.

Given the dominance of hake amongst local demersal species, and the high value of the South African hake fishery, an immediate candidate for such an MRM study is hake cannibalism and inter-species predation, taking account of considerable further data which have become available since the earlier analyses of Punt and Butterworth (1995).

4) *What are the most appropriate analysis platforms for such exercises?*

The selection here seems to lie between ECOPATH with ECOSIM, GADGET and *ab initio* coding. EWE, which could also be tailored to implement MRMs, is readily available and has been widely applied. GADGET has some promising features (see Chapter 13), but is not as yet fully developed. Hence, the appropriate choice would likely depend on the specific question being addressed, but particular weight must be given to the importance of being readily able to investigate the sensitivity of results to alternative functional forms for interactions between species. The ideal (if not always practical) scenario is one in which a plurality of approaches is applied, so that there is an opportunity for the comparative and confirmatory use of multiple MRMs (Fulton *et al.* 2003).

5) *What are the cost implications for data collection and analysis?*

The cost estimates for multi-species assessments provided by the US study by NMFS (2001) discussed above are sobering. Clearly costs depend on the extent of the activities proposed. However, given that population projections will likely prove highly sensitive to the functional forms chosen for species interactions, it is difficult to imagine how alternative models might be discriminated without recourse to regular dietary data collection exercises. This is so as to be able to check how diet changes in response to changes in the relative abundances of prey species. In principle, spatial variation in a single sampling exercise could serve in place of repeated temporal sampling, but it seems very doubtful that the necessary degree of spatial resolution and precision would be achievable locally. Such activities will not be of low cost. What seems important is that careful cost estimates are made for data collection and analysis exercises as part of the planning of research programmes to address management questions which have a multi-species/ecosystem context, so that these can be contrasted with the likely resultant improvements in management.

## **12.8 CONCLUDING REMARKS**

Multi-species/ecosystem modelling which aims to advise fisheries management decisions is an interesting and challenging activity that needs to be pursued. However, it is necessary to caution against unrealistic expectations of substantial progress towards this aim in the short and even medium term. Considerable data collection and complex analysis at a substantial cost and over a period of more than a few years will likely be necessary to achieve such reliability. Research priorities in this area need to be carefully and realistically chosen, and weighed against other research needs. Chapter 14 summarises recommendations of some immediate future research priorities that could contribute to an EAF both in southern Africa and the Southern Ocean.

Table 12.1. Summary of number of hake stomachs sampled across the whole study area over the period 1988-1994 (from Punt and Leslie 1995). These stomach content data were used (following further analyses) as inputs in the Punt and Butterworth (1995) model with the key values of the fraction of the diet consisting of Cape hakes shown in the lower part of the Table (from Punt and Leslie 1995).

a) Number of samples - all data pooled		
Length-class (cm)	<i>M. capensis</i>	<i>M. paradoxus</i>
1-20	214	299
20-30	425	509
30-40	524	454
40-50	468	396
50-60	485	382
60-70	437	353
70+	388	263
TOTAL	2941	2656

b) Fraction of diet (%) consisting of hake		
Predator age	<i>M. capensis</i>	<i>M. paradoxus</i>
1	8	93
2	9	54
3	13	60
4	13	36
5	46	71
6	19	30
7	22	31
8+	20	21

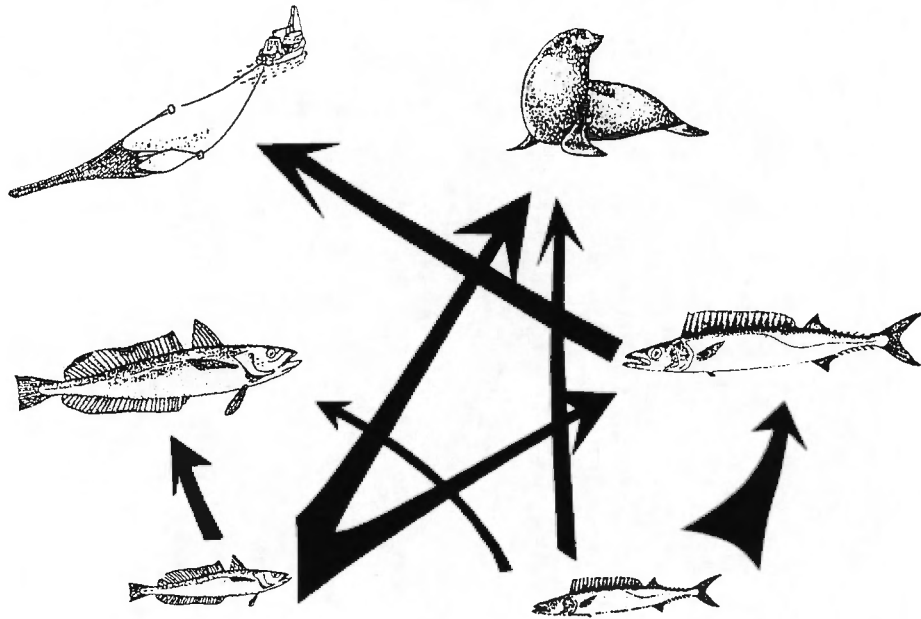


Fig. 12.1. Schematic summary of a minimal realistic model of Cape fur seals and Cape hake interactions off South Africa (from Punt, 1994).

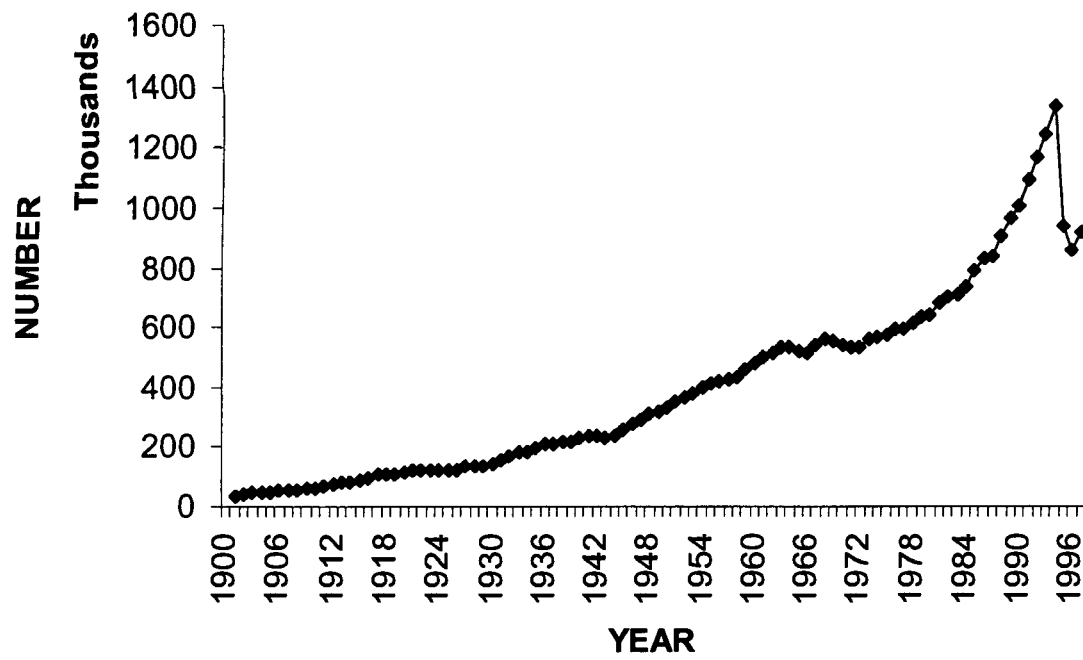


Fig. 12.2. Total numbers of Namibian seals estimated over the period 1900 to 1997 (from Butterworth *et al.* 1998).

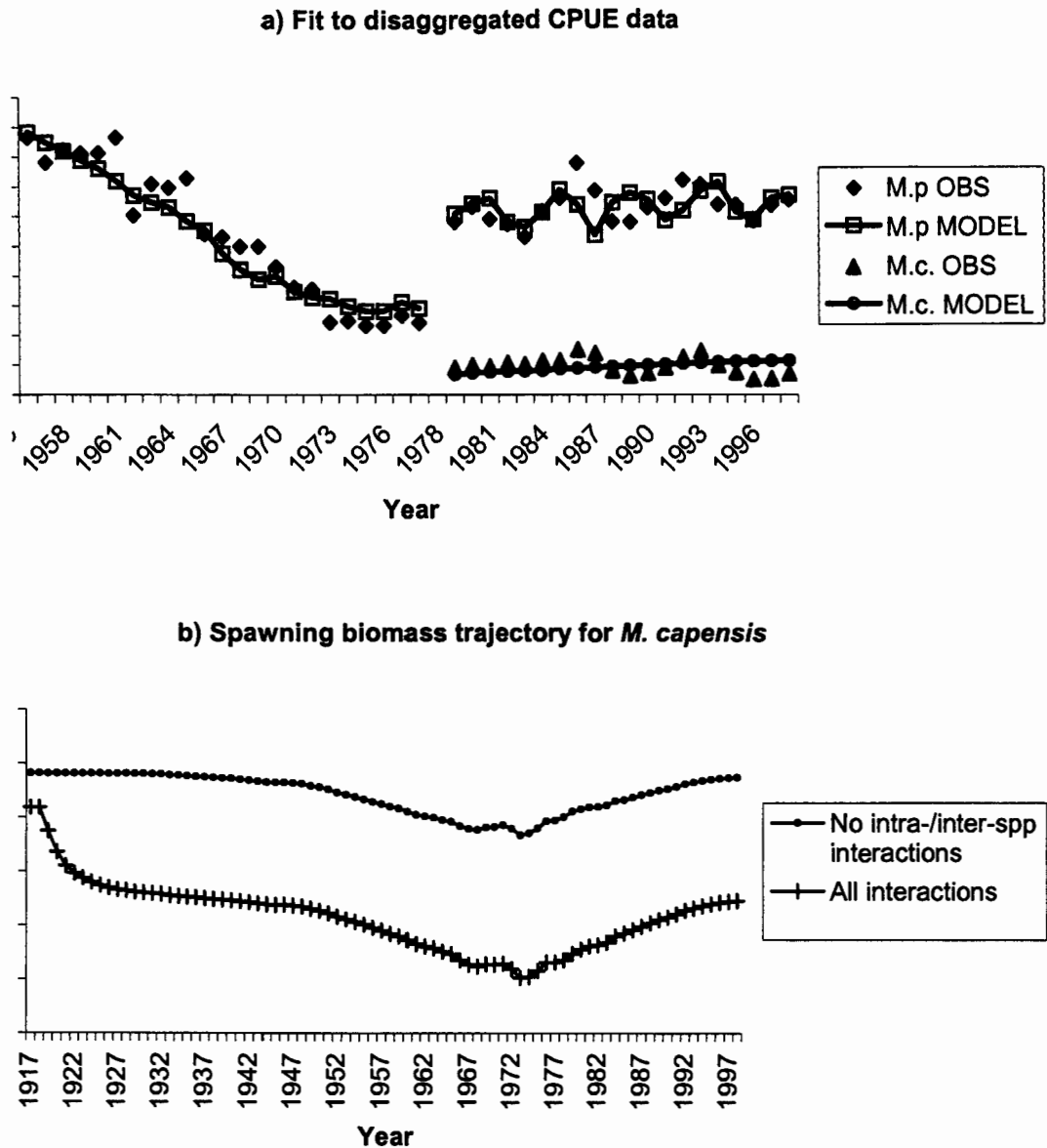


Fig. 12.3. Selected preliminary result from a multi-species hake-seal model treating predation/cannibalism interactions in the same way as an alternative fishing fleet. Note *M.c.* = *M. capensis* and *M.p.* = *M. paradoxus*. Provisional disaggregated and GLM-standardised CPUE data were obtained at that time from Jean Glazer, MCM and are similar to more recent updated values as presented in Glazer and Butterworth (2004). The top panel shows the fit of the model when including predation and cannibalism effects (with some arbitrary choices for interaction constants), while the lower panel (b) gives an example of a marked change in the predicted spawning biomass trajectory (for *M. capensis* in this case) when intra- and inter-species interaction effects are included in the model.

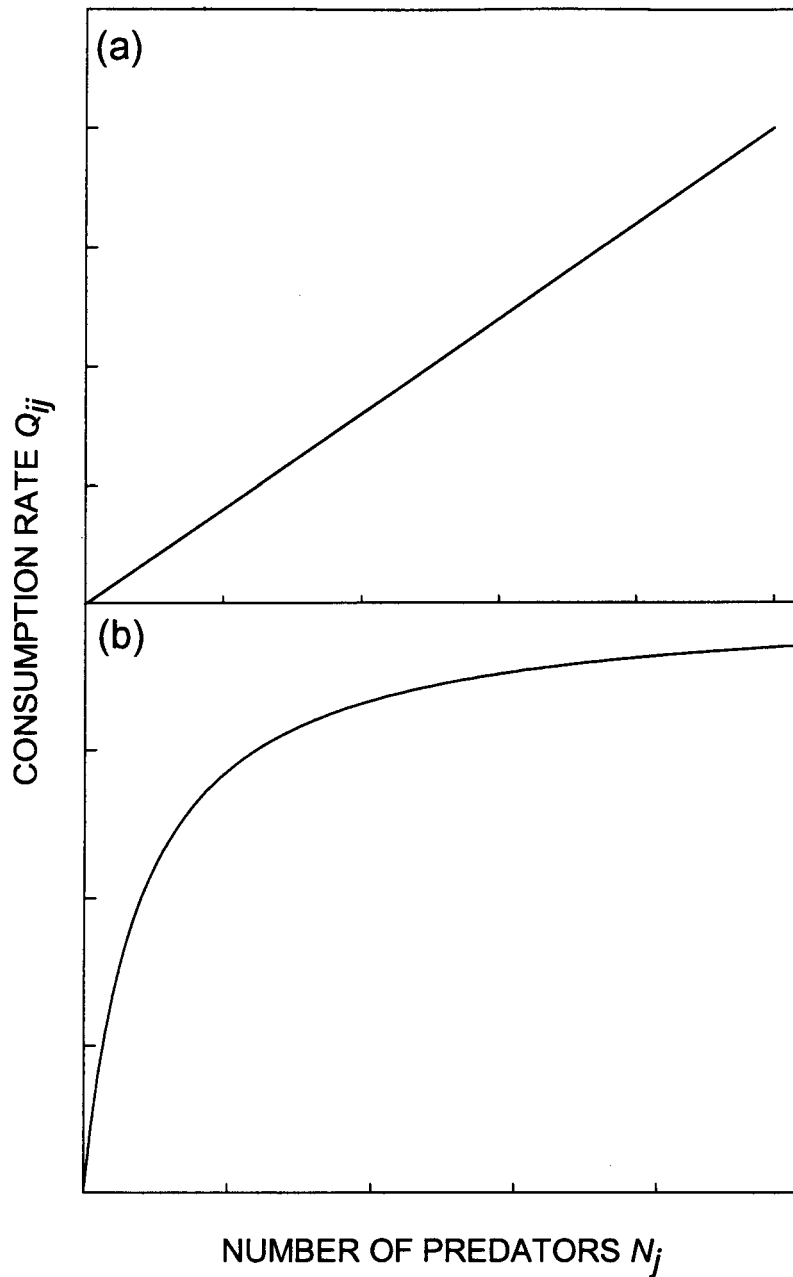


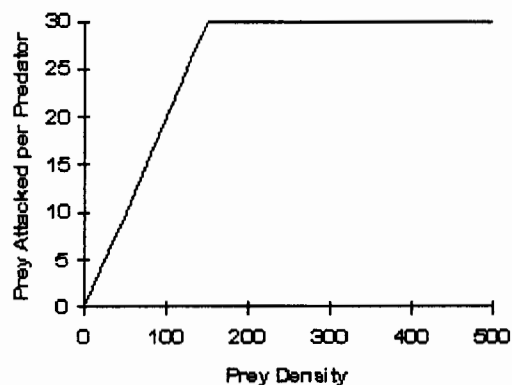
Fig. 12.4. Schematic showing how the total consumption rate  $Q_{ij}$  of prey species  $i$  by predator species  $j$  grows as the number of predators increase for the two contrasted cases: a) MSVPA (and its derivatives providing projections) showing a linear proportionality relationship, and b) ECOSIM's foraging arena-based model in which the total rate saturates at a constant level for high numbers of predators.



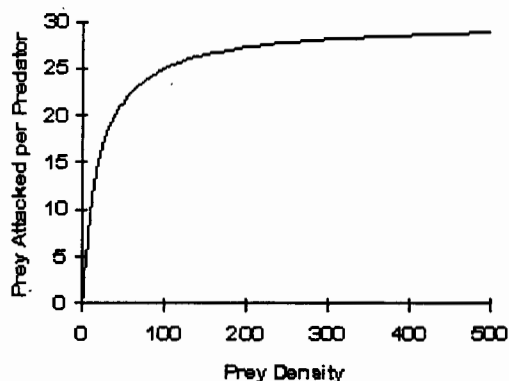
Appendix 12.1 Background to functional response types.

Holling (1959) identified three basic types of functional responses:

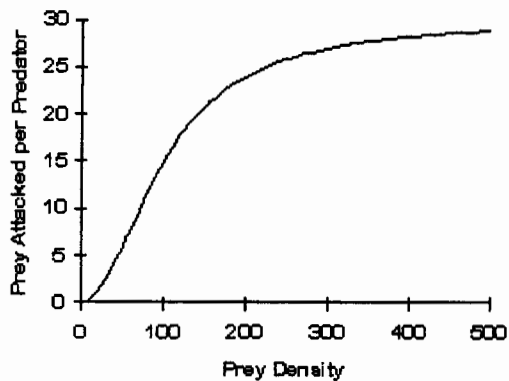
A *Type I (linear)* response is one in which the predation rate increases linearly with prey density but then plateaus when the consumer is satiated.



In a *Type II* functional response, the attack rate increases at a decreasing rate with prey density until it becomes constant at satiation.



A *Type III* or sigmoid functional response is one in which the predation rate accelerates at first and then decelerates towards satiation.



## Chapter 13

### Indirect fishery interactions: assessing the feeding-related interactions between marine mammals and fisheries

#### SUMMARY

Indirect fishery interactions between marine mammals and fisheries are defined as ecological, in contrast to technical, interactions and include:

- 1) Direct competition for commercial species;
- 2) Indirect competition for commercial species;
- 3) Habitat degradation due to trawling and other fishing practices;
- 4) Marine mammals as hosts for parasites to commercial fish species;
- 5) Indirect competition for primary production;
- 6) Changes in shark/killer whale predation rates affecting marine mammals;
- 7) Fishery-induced shifts in the size structure of ecosystems;
- 8) Alterations in foraging strategies in response to fishing effects; and
- 9) Effects of noise from fishing operations on marine mammal foraging activities and effectiveness.

Perceived competitive effects are a primary concern but the evidence for their impact is inconclusive because of the difficulties of substantiating, incontrovertibly, claims that predation by marine mammals is adversely affecting a fishery or *vice versa*. Given the predictions of a global shortfall between supply and demand over the next few decades coupled with the realisation that food consumption by marine mammals worldwide may be as much as 3-5 times the current commercial fisheries catches, there is an urgent need for scientific evaluations to estimate the nature and extent of these conflicts.

Some 17 marine mammal species (Common Minke whale, Antarctic Minke whale, Bryde's whale, Humpback whale, Harbor porpoise, Short-beaked common dolphin, Common Bottlenose dolphin, Killer whale, Gray seal, Harbor seal, Harp seal, Hawaiian monk seal, Steller sea lion, Antarctic fur seal, Cape fur seal, Sea otter, Dugong) are identified as having high priority in terms of data acquisition and focused modelling studies. The overriding importance is stressed of prioritising further investigations to collect data (experimental and/or field-based, and on foraging in particular) to assist in resolving issues related to the relationships between the rates of predation and the types and densities of available prey, so as to determine the appropriate forms of the functional responses. Data in the form of time series and / or spatial contrast data are needed rather than more conventional snapshot-type estimates.

Additional important considerations for further modelling studies include:

- 1) The need to present results that reflect uncertainty in data and model structure;
- 2) The need to consider operational (i.e. management) issues;

- 3) Examination of the extent to which underlying model assumptions predetermine or have implications for the results obtained;
- 4) The need for more realistic modelling of marine mammal populations;
- 5) The need to focus on specific areas/systems where there is the greatest chance of success (either because of the simpler structure of the ecosystem or because there are large signals in the data);
- 6) The need for systematic investigations of the numbers of links that have to be included in a non-trivial ecosystem model for reliable predictive ability; and
- 7) The choice of an appropriate spatial and temporal framework for data collection and modelling investigations.

It is proposed that GADGET (Globally applicable Area-Disaggregated General Ecosystem Toolbox) and MRMs (Minimum Realistic Models) are the most appropriate tools to assess competition interactions between marine mammals and fisheries.

Regarding habitat degradation, research needs to focus on:

- 1) investigations of the ecological implications of the short- and long-term effects of trawling; and
- 2) the extent to which acoustic harassment devices (as used by fisheries) may displace marine mammals from preferred habitats.

Trawler exclusion zones (particularly in nearshore waters) and no-take zones within Marine Protected Areas (MPAs) are potentially valuable tools for protecting the prey base for marine mammals, their breeding habitats and as a refuge from noise and other disturbances caused by fishing operations.

### 13.1 INTRODUCTION

Indirect fishery interactions encompass ecological interactions between marine mammals and fisheries. This is in contrast to directed marine mammal takes and operational (also termed technical) interactions in which marine mammals damage or become entangled in fishing gear with negative consequences for both the fishery and the animals. Biological and operational conflicts are sometimes difficult to separate because, for example, animals may damage fishing gear in the process of removing fish from the gear. There are many more published studies pertaining to operational interactions than to so-called biological or ecological interactions because the former are more obvious and easier to quantify (see Northridge 1991 for a comprehensive review of both forms of interaction). These can be distinguished further between so-called direct and indirect competition and other ecological aspects. "Direct competition" is defined as exploitative competition (in ecologists' terminology), being that which involves reduction (by consumption or utilisation) of a limited resource, but with no direct interactions between the competing species (Clapham and Brownell 1996), as when a marine mammal eats a fish that could otherwise have been caught by a fisherman. "Indirect competition" is defined similarly (as a sub-definition of

exploitative competition), except that in this instance the competitors may target different resources but these are linked because of a foodweb effect (e.g. when a marine mammal consumes a fish that is an important prey species of a commercially desirable fish species). Interference competition in which species actively disrupt the activities of others (Clapham and Brownell 1996) is not actually referred to as such in this chapter, but would include, for example, disruption to the feeding activities of marine mammals due to disturbances such as noise from fishing operations. A final set of indirect interactions (here referred to simply as “additional ecological aspects”) again involves no direct interactions between marine mammals and fisheries but considers the effects on one party of, for example, habitat destruction by the other party.

In almost all cases the available evidence for such forms of competition between marine mammals and fisheries is currently inconclusive. However, perceived conflicts between the two in pursuit of common sources of food have come increasingly to the fore in recent years. Escalating pressures on shared resources are expected in the future because of both increasing marine mammal populations and the needs of an increasing human population. Reductions in direct takes in response to recognition that many populations of marine mammals were heavily over-exploited in the 19<sup>th</sup> and earlier part of the 20<sup>th</sup> century, as well as a widespread change in people’s perceptions of whether marine mammals should still be regarded as renewable resources available for harvest, have meant that some marine mammal populations are currently increasing, sometimes by as much as 5-12 percent per annum (e.g. Shaughnessy *et al.* 2000, Bowen *et al.* 2004). In contrast, long-term (20-30 year) declines in the populations of several pinnipeds have been attributed to a combination of environmental factors and the depletion of prey species by commercial fisheries (Wade 2002), although the relative importance of these factors remains unknown.

The Food and Agriculture Organization of the United Nations (FAO) has estimated that more than one billion people worldwide currently rely on fish and shellfish for more than thirty percent of their animal protein (FAO 2002). The predicted increase in the global annual per capita consumption of fish is of concern given that marine capture fisheries are unlikely to appreciably exceed the present level of global landings (FAO 2002). The magnitude of the expected global shortfall between supply and demand over

the next few decades will depend in part on the growth in aquaculture (FAO 2002). In short, if the world's human population continues to increase faster than the total food fish supply from aquaculture, pressure will mount to increase fisheries production.

It is therefore imperative that scientists focus their efforts to increase understanding of the nature and implications of indirect interactions. Concerns about the consequences for fisheries of an increasing marine mammal population have already been expressed in southern Africa, for example, where, as described in Chapter 11, in 1990 Cape fur seals (*Arctocephalus pusillus pusillus*) were estimated to consume some two million tonnes of food a year. Considering that this amount was about the same as the annual human catch of fish in the region, and that the fur seal population was anticipated to increase further, the reasons for concern and potential for conflict are obvious.

Another example is in the Pacific Ocean, where marine mammals are estimated to consume about 150 million tonnes of food per annum, which is some three times the current annual fish harvest by humans (Trites *et al.* 1997). Tamura (2003) similarly deduced that, on a worldwide scale, fish consumption by *cetaceans* is approximately equivalent to commercial fisheries catches. He claims that competition between cetaceans and commercial fisheries is particularly severe in the North Pacific and North Atlantic Oceans. The Japanese Whale Research Programs in this region (JARPN and JARPN II) aim to address some of these issues, with their research emphasis having consequently shifted to studies of the feeding ecology of cetaceans to establish both prey consumption and preferences (Tamura 2003).

In Newfoundland, following the collapse of the cod fisheries in the late 1980s and early 1990s, the possible impact of an increasing harp seal (*Pagophilus groenlandicus*) population became a focal topic (Stenson *et al.* 1997, Stenson *et al.* 2002). Given the difficult nature of this problem and consequent lack of clear scientific guidelines, lobbying and political pandering have tended to dominate, as exemplified by the confident declarations of the then Canadian fisheries minister Brian Tobin, who in 1995 announced that the TAC for Northwest Atlantic harp seals was being increased from 186,000 to 250,000 seals on the basis that it was "...intellectually dishonest to claim that seals have not had a significant impact on recovering fish stocks...". He added that

“There is only one major player fishing that stock and his first name is Harp and his second name is Seal”, and concluded that the seals “must ...be harvested...in the context of building the groundfish stocks.”

In this chapter the direct competition for food and fishery resources between marine mammals and fisheries is first reviewed. A brief summary is presented of some specific examples that address the question of whether marine mammal populations have negatively impacted the potential yields from fisheries through competition. These examples generally concern *perceived* competitive interactions because in most cases the evidence is inconclusive. Then some examples pertinent to the reverse situation – whether fisheries negatively impact marine mammals - are summarised. As pressures mount to provide scientifically defensible answers to questions regarding marine mammal – fishery interactions, marine resource dynamics modellers are increasingly including marine mammals in their models or constructing new models in response. Given that models are appropriate tools for this task in that they effectively summarise the situation in terms of data availability and our understanding of how systems operate, the various approaches being employed at present are reviewed.

Moving one step further, into the realm where little is known at present, a brief overview is provided of additional factors to be borne in mind in assessing indirect interactions between fisheries and marine mammals. Specifically mention is made of potential instances where fishing has (or is likely to) indirectly impact marine mammals by damaging critical habitats upon which they depend, by altering the structure of ecosystems or by otherwise altering marine mammal population dynamics and/or population parameters. As knowledge of ecosystem structure and functioning continues to improve, concerns such as those listed here will increasingly be brought to the fore by scientists, conservationists and other interested and affected parties. Then returning to instances of potential negative impacts of increasing marine mammal populations, the serious economic consequences that can befall fisheries as a result of the spread of parasites from marine mammals to commercial fish species is briefly mentioned.

An initial attempt is made to rank the various indirect interactions in terms of their overall importance. Focusing then on the most important factors, criteria are identified

to assist in comparing these various indirect fisheries effects in terms of their potential impact. Finally, research needs are identified and prioritised.

### 13.2 FEEDING-RELATED INTERACTIONS

The following section first presents a brief summary of some specific examples that address the question of whether marine mammals negatively affect fisheries yields through competition. Secondly, some examples pertinent to the reverse – whether fisheries negatively affect marine mammals - are summarised.

A summary of the range of possible indirect interactions between marine mammals and fisheries is given in Fig. 13.1. The type of interaction may be either that of direct competition between fishermen and marine mammals for a target prey species, predator species or both (Fig. 13.1. A, B, C), or through an indirect foodweb effect whereby a fish species targeted by a fishery is in turn the predator or prey of another species that is an important component of the diet of a marine mammal (Fig. 13.1. D, E). The various interactions may be further complicated if cannibalism (of the target species in particular) occurs to a marked degree (Fig. 13.1. F). Additional interaction representations include scenarios in which a fishery on one species has a negative impact on the habitat of a prey species of a marine mammal (Fig. 13.1. G), and ones in which marine mammals indirectly reduce the value of the catch landed by a fishery due to the transmission of parasites (Fig. 13.1. H).

Table 13.1 gives a list of marine mammals and their potential interactions with fisheries. Ignoring arguments about taxonomy for present purposes, the list includes some 7 mysticete species, 47 odontocetes (including 20 beaked whale species not listed individually), 25 pinnipeds, two sirenians and two marine otters. Note that the information presented in the table is a first step only and should be interpreted broadly. It does not claim comprehensive capture of every record of small-scale marine mammal–fishery interactions. The following general features are evident from Table 13.1 and the associated literature:

- In almost all cases perceived interactions are difficult to establish conclusively.

- Reports in the literature are typically somewhat biased one way or the other depending on whether the concern arises from fishing interests (wanting to justify reducing marine mammals to improve fishery yields) or a conservationist group (wanting to ensure adequate food supplies remain for threatened or potentially threatened marine mammal species). However, whereas more than twenty of the species listed have at one time or another been killed (legitimately or otherwise) in the belief that reducing marine mammals will improve fishery yields, concerns that overfishing is negatively impacting local marine mammal populations have been documented for only a few species.
- Characteristics of marine mammal species presumed most at risk from indirect fishery interactions include the following:
  - Species that feed heavily on commercially fished species.
  - Species with restricted nearshore distributions or local populations of a species resident in areas overlapping commercial fishing grounds. Species without alternative feeding grounds may be particularly vulnerable if overlaps occur in their main feeding areas.
  - Species which are specialist feeders with narrow dietary ranges, e.g. those that target predominantly deepwater squid, may be at risk if deepwater squid fisheries were to expand in the future.
  - Very abundant species because of the large cumulative amounts of prey required to support their populations.
  - Species with low population numbers because of the danger that overfishing has already contributed to a population decline or may exacerbate future declines.
  - Highly social cetacean species because of the visual perception of the large amounts of prey consumed by such schools. Behavioural attributes such as vessel avoidance behaviour also presumably influence such perceptions with there being far fewer accusations of perceived competition from shy or elusive species. Large aggregations of pinnipeds similarly evoke concerns because of visual perceptions of their consumption requirements.



- Fisheries that allegedly suffer from consumption by marine mammals include the following:
  - Fisheries operating near pinniped breeding colonies.
  - Fisheries (both commercial and small-scale) operating in nearshore coastal areas (that overlap with restricted cetacean distributions such as that of the Indo-Pacific humpback dolphin).
  - Fisheries reliant on seasonal pulses in the abundance of a target species (e.g. pelagic schooling fish such as anchovy; migratory fish species such as tuna and yellowtail). Marine mammals generally congregate to take advantage of these pulses in abundance with the mammals' consequent increased visibility further fuelling suspicions that fewer marine mammals translate into more fish for the fishermen.
  - Fisheries reliant on species that are also important in the diet of marine mammals, particularly in situations where the mid-trophic layers are dominated by a few species only.
  - Fisheries on species where estimates of consumption of those species by marine mammals in the area of the fishery are substantial when compared with fishery takes.
  
- Most of the actual, perceived or predicted interactions between fisheries and marine mammals take the form of direct competition for a shared prey resource, even though the actual sizes of fish targeted may differ. However, there are several documented examples of indirect interactions whereby, for example, a fishery impacts a predator or prey species which is in turn the prey and predator respectively of another species fed upon by a marine mammal. With the exception of the dugong, West Indian manatee and walrus, there are very few documented cases (including speculations) that damage to a habitat by a fishery is indirectly responsible for reducing food supplies to a marine mammal.

### 13.2.1 *Instances of Potentially Detrimental Effects of Marine Mammals on Fisheries*

#### 13.2.1.1 **Pinnipeds (seals, sea lions and walruses)**

In the early 1990s, major collapses occurred in the Atlantic cod (*Gadus morhua*) fisheries on the East coast of Canada and several hypotheses have been posited to explain this. Harp seal populations off Newfoundland and Labrador have been increasing at an estimated rate of 5% per annum since the mid-1980s and are known to consume a substantial tonnage of juvenile cod (Stenson *et al.* 1997). Using a bioenergetics model, Stenson *et al.* (1997) estimated that in the Gulf of St Lawrence alone, harp seals annually consume some 445 000 tonnes of capelin, 20 000 tonnes of Arctic cod and 54 000 tonnes of Atlantic cod. There is some overlap between the sizes of capelin taken by the seals and the commercial capelin fishery, but the juvenile Atlantic cod age classes targeted by the seals are smaller than those recruited into the commercial fishery. Nonetheless, there is an obvious temptation to argue a causal relationship between the failure of the cod population to recover as rapidly as expected after its protection and the increase in harp seal abundance, especially considering that the socio-economic implications of the collapse of the cod fishery were substantial with some 40 000 fishermen rendered out of work. Although the results of at least one ecosystem modelling study (Trites *et al.* 1997) support the hypothesis that the recovery of these cod populations is being retarded to some extent by the increased biomass of harp seals, ecosystem models generally have poor predictive reliability, largely because of data limitations.

Demonstrating that either a fishery or a marine mammal will be adversely affected as a result of an increase in the removals of a limited resource by one of these two is not simple. Inferences based on assumptions of a linear relationship between predator and prey abundance are often incorrect because of the complex nonlinear interactions in an ecosystem. For example, off the west coast of South Africa, seals consume almost as much hake as is taken by the commercial fishery. However, the commercially valuable hake consists of two species, a shallow-water (*Merluccius capensis*) and a deep-water species (*M. paradoxus*), with the larger of the shallow-water species eating the smaller individuals of the deep-water species (see also Chapters 11 and 12). The results of multispecies models suggest that the net effect of a seal reduction would likely be less

hake overall because fewer seals, which are thought to predate primarily on shallow-water hake, would mean more shallow-water hake, and hence more predation on small deep-water hake (Punt and Butterworth 1995). This highlights the complexity of predation, food-fish and fishery interactions and hence the difficulties of demonstrating conclusively that marine mammals are in direct competition with humans for food fish, as may superficially appear to be the case.

#### 13.2.1.2 Whales

Numerous multispecies modelling studies have been employed to investigate the direct and indirect effects of minke whales (*Balaenoptera acutorostrata*) on the cod, herring (*Clupea harengus*) and capelin (*Mallotus villosus*) fisheries in the Greater Barents Sea (e.g. Schweder *et al.* 2000). Minke whales are abundant in this region and prey on all three species, prompting the question of whether or not fishermen could expect greater catches if the populations of these marine mammals were reduced. The indications of these studies are that there is competition between the whales and fishermen in this region, and that the fisheries are likely to respond roughly linearly to changes in whale abundance. They estimate that each minke whale reduces the potential annual sustainable catches of both cod and herring by some 5 tonnes (Schweder *et al.* 2000). Similarly, studies off Iceland suggest that the piscivorous minke, humpback (*Megaptera novaeangliae*) and fin whales (*B. physalus*) may be having a considerable impact on the region's cod stock (Stefansson *et al.* 1997). The cod fishery is of key importance to the Icelandic economy and the rebuilding of the cod population and catches are recognised as an important economic consideration. It is therefore not surprising that arguments have been put forward that there is a need to reduce whale populations to permit commercial fisheries to increase.

Of note is that the main research focus of the Japanese Whale Research Program in the North Pacific (JARPN II) has shifted from a study of the stock structure of minke whales to feeding ecology and ecosystem studies (Government of Japan 2002). Minke whales show seasonal and geographical changes in diet and it is postulated that there might be direct competition between minke whales in the Northwestern Pacific and the fishery for Pacific saury in particular (Tamura and Fujise 2002). As part of the JARPN II program, an ECOPATH with ECOSIM modelling approach (see Chapter 11) has

been applied to the western North Pacific, which suggests that both minke and sperm whales are “key species” in the region because their removals result in large changes to the biomass of prey species (Government of Japan 2002). Bryde’s whales have also been identified as an important species for consideration in the Japanese study given that they are abundant and reportedly play an important role in the Western North Pacific ecosystem. Work is also in progress to develop a MULTSPEC-type model (Bogstad *et al.* 1997) incorporating minke whales, Pacific saury, Japanese anchovy and krill (Government of Japan 2002).

Whereas marine mammals are thought to exert relatively minor influences on systems such as the North Sea and Baltic Sea, they have been identified as potentially serious competitors off, for example, the northeastern USA, a region that includes important fishery areas such as the Gulf of Maine and Georges Bank (Overholtz *et al.* 1991, Kenney *et al.* 1996). The latter region exemplifies the conflicts that can arise between fishery management plans tasked with rebuilding prey populations and prescriptions, by the USA Marine Mammal Protection Act in this case, to facilitate an increase in the abundance of marine mammal predators.

#### 13.2.1.3 Small cetaceans

In many areas of the world, coastal fishers consider dolphins as serious competitors and culls or control programs for fisheries have been carried out. In some instances a bounty for culling dolphins has been instigated (e.g. Kishiro and Kasuya 1993). In the Mediterranean, both short-beaked common dolphins (*Delphinus delphis*) and common bottlenose dolphins (*Tursiops truncatus*) were culled in the past because of *inter alia* their reputation as competitors with fisheries (Bearzi *et al.* 2003).

One of the largest culls on record occurred at Iki Island in Japan in response to declines in catches of yellowtail (*Seriola dorsalis*) (Kasuya 1985). Over the period 1976 to 1986, drive-fishing was responsible for the deaths of thousands of bottlenose dolphins and hundreds of Pacific white-sided dolphins (*Lagenorhynchus obliquidens*), false killer whales (*Pseudorca crassidens*) and Risso’s dolphins (*Grampus griseus*) (Kasuya 1985).

#### 13.2.1.4 Sea otters

Sea otters prey on a variety of marine invertebrates such as urchins and abalone. Off southern California, southern sea otters (*Enhydra lutris nereis*) have been labelled by some as responsible for the decline of the red abalone (*Haliotis rufescens*) fishery (Fanshawe *et al.* 2003), but there is little direct evidence to support this notion. The commercial abalone fishery off southern California was closed in late 1997, and factors such as commercial fishing, disease and changing environmental conditions are all thought to have contributed to the decline of these commercially valuable shellfish (Tegner *et al.* 2001). Although the southern sea otter population is not increasing, there are perceptions that the level of competition between otters and commercial abalone fishers has increased because of the recent southward movements of otters, which has enlarged the overlap between otter feeding grounds and abalone fishing areas (Fanshawe *et al.* 2003).

### 13.2.2 *Instances of Potentially Detrimental Effects of Fisheries on Marine Mammal Populations*

#### 13.2.2.1 Pinnipeds

The western Alaska population of Steller sea lions (*Eumetopias jubatus*) has shown a continuous decline since the 1970s (Loughlin *et al.* 1992) and was listed in 1997 as an endangered species under the U.S. Endangered Species Act. Several groups have argued that this decline is due in part to the large fishery harvests of walleye pollock (*Theragra chalcogramma*) and Atka mackerel (*Pleurogrammus monopterygius*) which are simultaneously key sources of food for sea lions and the basis for commercially valuable U.S. fisheries (Committee on the Alaska Groundfish Fishery and Steller Sea Lions 2003). This argument is supported by research demonstrating considerable overlap in the size of these species taken by the Alaskan western stock of sea lions and the commercial trawl fishery (Zeppelin *et al.* 2004). Measures to reduce the perceived competition between sea lions and fisheries for groundfish stocks include the establishment of “buffer” (no-trawl) areas to include important locations where the sea lions breed, feed and rest, as well as specifying a pollock harvest which is more evenly distributed over the remaining areas and spread throughout the year (Marchant 2001).

However, the results of modelling studies (Trites *et al.* 1999) indicate that the observed sea lion population decline cannot be explained solely through trophic interactions, and rather is more likely linked to shifts in environmental conditions which lead to changes in the favoured complex of species. Moreover, studies such as that by Trites *et al.* (1999) highlight the difficulties of predicting the direction and magnitude of a change in an ecosystem arising from a reduction in predation or fishing pressure. They posit that, paradoxically, Steller sea lion and northern fur seal (*Callorhinus ursinus*) populations might realize greater benefits if adult pollock and large flatfish were more heavily fished. This competitive release effect may occur because, for example, pollock are cannibalistic and hence decreased adult pollock abundance as a result of heavier fishing may result in increased numbers of juvenile pollock available to marine mammals.

An alternative hypothesis, the so-called “junk-food” hypothesis of Rosen and Trites (2000), links the decline of Steller sea lions (in the Gulf of Alaska and the Aleutian Islands) to the dominance of pollock (a gadoid fish) in their diet instead of higher-quality (in terms of energy content) fishes such as Pacific herring (*Clupea pallasii*) and sandlance (*Ammodytes hexapterus*). Winship and Trites (2003) recently estimated that in southeast Alaska, seasonal changes in the energy density of the diet of Steller sea lions resulted in animals requiring approximately 45-60% more food per day in early spring than in late summer. Regional differences in the energy density of the diet similarly accounted for substantial differences (up to 24% based on summer diets) in food requirements among the southeast and western Alaska Steller sea lion populations (Winship and Trites 2003). While the role of commercial fisheries (targeting pollock and herring) compared to the environment (e.g. through a shift in ocean climate in the late 1970s) in altering the forage fish abundance and community structure of this region remains unclear, the endangered-species listing of the Steller sea lion in the U.S. has drawn attention to the need for analyses of indirect interactions between fisheries and marine mammals to consider not only the quantity but also the distribution and quality of food available to marine mammal populations (Trites and Donnelly 2003).

A second example of a likely fisheries-generated lack of food concerns the mass migration of harp seals south along the coast of Norway in 1987 in response to the

collapse of the capelin stock in the mid-1980's (Nilssen *et al.* 1997, Livingston and Tjelmeland 2000). Heavy fishing played an important role in the collapse of the capelin stock (Gjøsæter 1998), as well as earlier declines in the cod and herring stocks, with the associated sequence of events further highlighting the complexities of cod, capelin, herring, and fishermen interactions in the Barents Sea. The effects of fishing should generally be easier to detect in relatively simple ecosystems such as the Barents Sea where most of the energy is channelled through a few species positioned at an intermediate level in the foodweb (Gislason 2003).

### 13.2.2.2 Whales

Competition effects are difficult to quantify, but it has been proposed by Whitehead and Carscadden (1985) that the collapse of the eastern Canadian capelin fishery in the 1970s had a negative effect on fin whales. They suggest that a shortage of capelin might have allowed humpback whales to outcompete fin whales because the latter rely principally on capelin as a prey source.

If competitive predation between a marine mammal and fishery occurs, this implies that the marine mammal population is limited by food availability and hence that it should be possible to demonstrate a response of some vital population parameters to a change in food availability. Recent probable population increases of several krill-eating marine mammals, such as minke whales, crabeater seals (*Lobodon carcinophagus*) and fur seals, have been attributed by some investigators to a likely large increase in the availability of krill (*Euphausia superba*) in Antarctic waters (Clapham and Brownell 1996). Following the substantial reduction through overexploitation of large whale populations during the early 20<sup>th</sup> century, some 150 million tonnes of “surplus” krill is argued to have become available annually to other predators (see also Chapter 11). This “krill surplus” theory enunciated by Laws (1977) has yet to be universally accepted, and supporting evidence has been sought (and argued to have been found) by addressing questions such as whether the mean age at maturity of minke whales (Thomson *et al.* 1999) and crabeater seals (Bengtson and Laws 1985) has dropped in more recent years as a response which might result from an increase in food availability, and whether minke whale assessments based upon catch-at-age data indicate population increases (Butterworth *et al.* 1999). Clapham and Brownell (1996)

stressed the following difficulties in conclusively linking purported changes in the abundance and reproduction of baleen whales to interspecific competition following large-scale commercial exploitation: insufficient data on levels of prey biomass and predator consumption, the validity of many of the purported changes, a wide range of alternative explanations and a lack of sufficient data to discriminate among these.

Modelling studies by Trites *et al.* (1999) suggest that marine mammal populations can be quickly reduced through reductions in prey abundance but show a generally slow recovery when abundant food becomes available.

### 13.2.2.3 Small cetaceans

Dolphin populations that have localised coastal distributions, such as the Indo-Pacific hump-backed dolphins (*Sousa chinensis*) off KwaZulu-Natal, South Africa, may be vulnerable to commercial fishery expansions because of increased competition with fishermen for limited food resources (Peddemors 1999). In both the Mediterranean and the Black Sea, prey depletion is considered of primary or secondary importance in causing habitat degradation and loss of at least four marine mammal species: common bottlenose dolphins, short-beaked common dolphins, striped dolphins (*Stenella coeruleoalba*) and harbor porpoises (*Phocoena phocoena*) (Notarbartolo di Sciara *et al.* 2002).

In the Mediterranean, declines in short-beaked common dolphins have been attributed to a number of human-induced threats, including both direct and incidental catches by the fishery, prey depletion, contamination by xenobiotics and global climate change (Bearzi *et al.* 2003). Of these, prey depletion is the most likely proximate cause to account for the observed trends (Politi and Bearzi in press, cited in Bearzi *et al.* 2003). Other marine mammals in the Mediterranean such as bottlenose dolphins also show signs of malnutrition presumably as a response to overfishing of their prey stocks and intensive trawling (Politi *et al.* 2000, cited in Bearzi *et al.* 2003). An additional complication in assessing the impacts of prey depletion concerns its role in compromising animal health such as is implicated in the large die-off of Mediterranean striped dolphins (due to an epizootic) in 1990-1992 (Aguilar 2000).



It is of concern that in response to the decline in sardines and anchovies that were previously preferentially targeted by fishermen in some areas of the Mediterranean, attention has now focused instead on other small pelagic fish (e.g. round sardinella, *Sardinella aurita* and garpike, *Belone belone*) that are simultaneously targeted by common dolphins and required by the growing aquaculture industry (Bearzi *et al.* 2003).

#### **13.2.2.4 Sea otters**

Dramatic declines in northern sea otter (*Enhydra lutris kenyoni*) populations have been indirectly linked to competition with fisheries. As outlined above, overfishing is argued to be one of the factors contributing to the decline of pinniped populations (harbor seals *Phoca vitulina* and Steller sea lions) in some of Alaska's Aleutian Islands (Loughlin and York 2000). Killer whales may feed preferentially on pinnipeds in this region, but as a result of the decline in pinnipeds, they may have switched to sea otters as prey. The work of Estes *et al.* (1998) argues that overfishing populations of fish prey responsible for providing high caloric and nutritive value to pinnipeds may impact not only directly on pinniped populations, but also indirectly on killer whale and sea otter populations.

#### **13.2.2.5 Sirenians (dugongs and manatees)**

These mammals feed mostly on vascular aquatic plants so that there is no direct competition with humans for a shared food resource, but rather indirect interactions due to the degradation and loss of their habitat, which is partly attributable to fishing activities such as trawling. For example, on the Great Barrier Reef intensive prawn trawling has been blamed for the decline in dugongs (*Dugong dugon*) following the elimination of seagrasses. Along the Andaman coast of Thailand there have been periodic problems with large trawlers encroaching into shallow coastal waters and damaging seagrass beds (Hines 2002).

#### **13.2.3 Food Web Interactions**

Trites *et al.* (1997) and Kaschner *et al.* (2001) estimated the degree of competition between fisheries and marine mammals for prey and primary production in the Pacific and North Atlantic Oceans respectively. In both instances, marine mammals are

estimated to consume about three times as much food as humans harvest, but most of this consumption is composed of species that are not of current commercial interest. Kaschner (2004) presents similar arguments based on a global analysis of catch and food consumption by marine mammals and fisheries. Trites *et al.* (1997) estimate that the greatest overlaps in the Pacific occur for pinnipeds (60%) and dolphins and porpoises (50%), though such estimates are likely subject to appreciable uncertainty. Similarly, Kaschner (2004) estimates that in the Northern Hemisphere resource overlaps are greatest for these groups, whereas in the Southern Hemisphere overlaps between baleen whales and large toothed whales are the most substantial.

Trites *et al.* (1997) argue that whilst direct competition between fisheries and marine mammals for prey appears limited, indirect competition for primary production may have an appreciable effect. Such so-called food web competition may occur if there is overlap between the trophic flows supporting the two groups (see Fig. 13.2). Evidence in support of food web competition between marine mammals and fisheries is provided by a negative correlation between estimates of the primary production required to support fisheries catches and to support the number of marine mammals estimated to be present in the different FAO Statistical Areas that span the Pacific Ocean (Trites *et al.* 1997).

#### **13.2.4 *Methods to Assess Interactions, and their Applicability***

While commercial fishing interests in many parts of the world perceive marine mammals as serious competitors for scarce resources, many other organisations argue that marine mammals are being used simply as scapegoats for failed fisheries management policies (Butterworth 1999). Scientific evidence is therefore increasingly being sought to settle these disputes, but there is increasing appreciation that the scientific methodologies required to address them are complex, time consuming and beset with difficulties. Rice (2000) notes that even though most modelling frameworks give competition a strong role in structuring marine communities, this is questionable considering that after decades of research in terrestrial ecosystems, the role of competition is still unresolved.

#### 13.2.4.1 Simple approaches

Initial approaches (e.g. Beverton 1985) to quantify the impact of consumption by marine mammals on fish catches took account of the fact that, particularly for pinnipeds, the sizes of fish eaten tend to be smaller than are taken by commercial fisheries. There is thus not an exact correspondence between one ton of a commercially desired fish species eaten by seals, say, and a ton caught by fishermen. This follows because although a fish eaten by a seal would have grown larger by the time it became vulnerable to fishing, it might also have died before reaching that size as a result of other sources of natural mortality.

Such computations, which essentially treated marine mammals as the equivalent of another fishing fleet, are now generally considered to be inadequate because of oversimplification, particularly as they disregard potentially important secondary (indirect) interactions. To perform more realistic computations while still accepting that both data and computing power limitations necessarily restrict the degree of complexity that is viable to incorporate in multispecies models, the following three complicating factors need to be addressed:

- a) How many of the large number of interacting species in any ecosystem need to be considered and what is the functional form of these interactions?
- b) Do age-structure effects need to be taken into account? This can become important when, for example, one species that predated on the small juveniles of a second, finds itself the prey of the larger adults of that same species.
- c) Is the assumption that species interactions occur homogeneously over space adequate or an oversimplification? Accurate representation of spatial overlaps in prey availability and marine mammal feeding distributions are complicated by the fact that for pinnipeds, for example, the distribution of breeding and resting sites does not necessarily reflect their feeding distributions (see e.g. Bradshaw *et al.* 2003).

More recent attempts to quantify marine mammal-fishery interactions have taken account of some or all of the above complicating factors, though studies have not yet reached the stage where all three are satisfactorily considered simultaneously. The

scientific methodologies required to address these aspects are complex, time consuming and beset with difficulties as is illustrated through selected examples below.

#### **13.2.4.2 More complex modelling approaches**

A brief critical review of the methods currently being used to provide a more reliable basis for the scientific evaluation of the competitive effects at issue is given below.

In discussing these different modelling approaches below, it is useful to classify models as either “Efficient predator” models or “Hungry predator” models (Butterworth and Plagányi, 2004). In the former set of models the predator is assumed to always get its daily ration (e.g. MSVPA, MULTSPEC), though the species composition of this ration may change with varying prey abundances over time. In contrast, in the latter set, predators are assumed to compete with others of the same (and possibly other) species for limited vulnerable proportions of prey (e.g. “foraging arena”-based models applied in approaches such as ECOSIM).

A second useful classification groups models as Minimally Realistic Models (MRM) on the one hand and “ecosystem” models on the other. As explained further in section 3.4.5 below (and in Chapter 12), a MRM seeks to include only those species considered likely to have important interactions with the species of primary interest. The MRM group includes MSVPA, MULTSPEC, BORMICON/GADGET, Seastar, Scenario Barents Sea and the original seal-hake MRM of Punt and Butterworth (1995). Shared characteristics of these models include the following (NAMMCO 2002):

- They are system specific.
- Only a small selected component of the ecosystem is modelled.
- Lower trophic levels and primary production are modelled as constant or varying stochastically.

In contrast, the ECOPATH/ECOSIM models, for example, are generic and capable of explicitly including most ecosystem components as well as incorporating lower trophic levels and primary production, though naturally they can also be applied in a simplified form closer to the MRM concept.

A general problem in reviewing the suite of models currently available to explore indirect interactions between marine mammals and fisheries is that many of these models have essentially been constructed using a generic template tailored to fish species which generally produce thousands of eggs and can show dramatic annual differences in productivity. Marine mammals have very different life histories to fish and do not possess much scope to increase reproductive outputs given, for example, that they produce an average of one young or less per year. There is evidence that in species such as large mammals that have low reproductive rates and long life spans, most density dependent changes in vital rates occur at high population levels (close to carrying capacity) (Fowler 1981), whereas the reverse is true for species with life history strategies typical of most fishes for which absolute recruitment levels generally drop appreciably below pristine levels only when spawning biomass has been reduced to well below such levels (Myers *et al.* 1995). In long-lived species such as marine mammals, one therefore expects a change in natural mortality only when per capita prey availability falls below a low threshold value. Notwithstanding that others (e.g. de la Mare 1994) have argued against some of the conclusions drawn by Fowler, the posited counter-argument of hyper-compensation remains essentially speculative in nature (Butterworth and Punt 2003). The fact that life-history characteristics of fish and marine mammals differ substantially with respect to density-dependent responses is supported further by the recent work of Oli and Dobson (2003) who show that the ratio of the magnitude and onset of reproduction (life-history variables that differ dramatically between fish and marine mammals) is the best determinant of population growth rates.

#### **13.2.4.3 ECOPATH with ECOSIM (EwE)**

Given that the ECOPATH (Polovina 1984, Christensen and Pauly 1992), ECOSIM (Walters *et al.* 1997) and ECOSPACE (Walters *et al.* 1999) suite are currently dominating attempts world-wide to provide information on how ecosystems are likely to respond to changes in fishery management practices, it is critical that the applicability of these approaches to answering questions in this context be carefully reviewed (Aydin and Friday 2001, Aydin 2004). A description of the ECOPATH with ECOSIM approach is given in Chapter 11. Following from this, attention is drawn in particular to the following limitations that need to be borne in mind when interpreting

any outputs of EwE pertaining to the question of the extent and nature of indirect interactions between marine mammals and fisheries.

#### **13.2.4.3.1 Bioenergetic considerations**

In the interests of simplicity, EwE does not make allowance for detailed bioenergetic considerations (Aydin and Friday 2001, Aydin 2004) and unfortunately some of this information is critical in the current context. For example, alternative prey types are treated as energetically equivalent in EwE. Detailed studies on Steller sea lions (see section 3.2.1) have pointed to the importance of considering not only the quantity but also the quality and distribution of different prey types.

#### **13.2.4.3.2 Life history considerations**

In ECOSIM, with increasing food supply, and a consequent increase in per capita predator consumption, animals may either (Walters *et al.* 2000):

- allocate surplus to reproduction rather than somatic growth (food allocation hypothesis); or
- spend less time foraging so as to decrease time at risk to predation (so that the density dependent response to population reduction would be in the form of a drop in natural mortality) (predation hypothesis).

For marine mammals in particular, there is a problem with both these limits in ECOSIM. Under changing conditions, natural mortality is conventionally considered one of the least labile population parameters for marine mammals. Furthermore, physiological limitations mean that above a certain threshold level there is generally no appreciable response of population variables to increasing food availability. One danger of inadvertently modelling marine mammal populations as overly compensatory (by treating them in the same way as fishes) is the possibility that this could lead to overly optimistic predictions of the likely recovery of populations following their reduction, or in response to increased food availability

#### ***13.2.4.3.3 Implications of the ECOSIM interaction representation***

Given current pressures to accurately quantify the predicted effects of marine mammal reductions on fisheries, or, conversely, the effects of increases/decreases in fishery catches on marine mammal populations, it is imperative that models employed for this purpose be closely scrutinised to understand the extent to which underlying model assumptions predetermine or have implications for the results obtained. This is particularly important in the case of “blackbox”-type models such as ECOSIM and hence this aspect is addressed briefly here. By virtue of it being packaged in a form that is readily digested by as many people as possible, the EwE modelling package is necessarily less transparent and the code less accessible than for other approaches.

The ECOSIM “foraging arena” concept (Walters and Kitchell 2001), as described in Chapter 11 (see Equation 11.5), is a novel functional response representation that is argued by these authors to be supported to some extent by studies of fish populations. However, complications to be borne in mind include the fact that EwE cannot straightforwardly depict instances where the foraging arena  $V$ 's (vulnerability pools) are used simultaneously by multiple predators. This may be important in instances in which a fish predator targets similar prey to those targeted by a marine mammal, or in which there are overlaps in the vulnerability pools available to marine mammals and to fisheries. EwE as presently configured implicitly assumes that direct interference between predator species (which it ignores) is inherently different from within-species interference (explicitly modelled by Equation (11.5)).

As explained in Chapter 11, ECOSIM cannot yield pure-replacement results when predicting the effects of a “predator” (a fishing fleet, say, that acts identically in terms of prey selection) in supplanting marine mammals. Expressed another way, this argument is that default parameter value selections for the model effectively hard-wire it to such an extent that they effectively swamp other signals pertinent to predicting the effects of a marine mammal reduction. Cooke (2002) similarly demonstrated through the use of a simple model that whether or not the reduction in cetaceans results in higher fishery yields than would otherwise, other things being

equal, be obtained, depends critically on the assumed vulnerability of the fish to the whales. It is only under scenarios assuming a high vulnerability of fish to whales that fishery yields are predicted to be sensitive to the abundance of whales. The results of Cooke (2002) highlight the importance of first exploring the likely relationship between whale abundance and whale consumption of a target fish species, because *a priori* assumptions in this regard strongly influence model outcomes in terms of whether or not they yield pure-replacement results.

#### 13.2.4.4 Multispecies Virtual Population Analysis (MSVPA)

Multispecies Virtual Population Analysis (MSVPA), described in the previous Chapter, is a technique that uses commercial fisheries catch-at-age and fish stomach-content data to estimate both the past fishing mortalities and the predation mortalities on some of the chief fish species of interest (see e.g. Sparre 1991, Magnússon 1995). MSVPA applications have thus far focused primarily on the North Sea, with the considerable data requirements generally impeding the application of this approach to other areas, although similar approaches have been applied in the Bering, Baltic and Barents Sea as well as the Gulf of Maine. Given the various difficulties associated with sampling marine mammal diets, it would be problematic to expand this approach to model indirect marine mammal-fishery interactions, although the 1991 ICES stomach sampling scheme did include gray seals. In the case of marine mammals, scat analyses were used to estimate food suitability constants (“suitability” is an important input to MSVPA, and specifies the relative preference that a predator has for different prey species, if all were present in equal abundances).

A second potential problem with the MSVPA approach is that it concentrates on the impacts of predators on prey but ignores any potential effects that changing prey populations may have on the predators themselves (because of its constant ration assumption – see next paragraph). Nonetheless, it has some utility in quantifying the relative losses in prey biomass attributable to other predatory fish, marine mammals and commercial fisheries. Interestingly, MSVPA results for the North Sea support the results of other studies suggesting that losses due to marine mammals are small relative



to losses attributable to other predatory fish (Bax 1998, Bogstad *et al.* 2000, Furness 2002 – North Sea). However, given that the consequences of overlaps in consumption likely depend heavily on the associated temporal and spatial scales, standard MSVPA (as with most other models) falls short in not explicitly modelling migration into and out of the area under analysis.

The MSVPA approach differs fundamentally from ECOSIM in its assumption that the intake by a predator is time invariant – a so-called “efficient predator” model in which predators always attain their daily ration in contrast to the “hungry predator” assumption of ECOSIM’s “foraging arena” model (Butterworth and Plagányi, 2004). Once again it is critical to recognise that this model assumption will lead to model predictions of a near-linear relationship between reductions in marine mammal biomass and increases in fishery yields. This stresses the need for improved understanding of feeding ecology and relationships before selections can be made between models which will have different implications for the key question under review here. The MSVPA studies have at least made a start (e.g. Rice *et al.* 1991, Rindorf *et al.* 1998) in trying to determine the extent to which the consumption of a given prey is a simple linear function of its relative abundance in an ecosystem (the constant suitability assumption). Put simply, such models have problems accounting for scenarios in which predators keep preferentially eating a given food item despite its becoming scarce, as has been demonstrated, for example, for fishes such as cod and whiting (Rindorf *et al.* 1998).

Marine mammals as a group presumably vary substantially in terms of the extent to which their preferences for different prey remain stable over time. At least a few species may prey switch (e.g. Estes *et al.* 1998). Harbitz and Lindstrøm (2001) demonstrate the use of a stochastic spatial analysis framework to derive relationships between expected proportions of prey biomass in the sea and in the diet of minke whales. They illustrate, for example, that while minke whales in the Barents Sea appear to actively select capelin in preference to other species present, this is no longer the case once the preference of minke whales for foraging in the upper water layers is also taken into account. Their methods, which highlight the importance of interpreting results at the correct spatial scale, could usefully be applied to other predator foraging studies.

An MSVPA model is almost certainly not adequate on its own as a tool for investigating indirect interactions involving marine mammals, but together with an ECOSIM model, these two approaches can at least make a first attempt at bounding the likely impact on a fishery of a marine mammal reduction in that, based on the assumed forms of interaction, the former approach is likely to overestimate the effect and the latter to underestimate it (at least when using default or low vulnerability settings). MSVPA is probably too data hungry to be widely applied, but that does not necessarily preclude the use of MSFOR to predict forwards, provided the model is initialised using sensible assumptions based on at least some data (IWC 2004).

#### **13.2.4.5 Minimum Realistic Models (MRM) – a specific example**

As described in the previous two chapters, Punt and Butterworth (1995) developed the first so-called Minimally Realistic Model (MRM) in response to recommendations to this effect made at a preceding international workshop (Butterworth and Harwood 1991). As detailed foodweb structure can make a large difference in predicting the ecosystem effects of fishing (Yodzis 1998), there is clearly a need for more work to systematically assess the effects of ignoring weak ecosystem links in models such as this. Moreover, Yodzis (1988) highlights the inherent indeterminacy of multispecies manipulations. Given the plethora of uncertainties, it is somewhat surprising that work to date has not concentrated more on calculating probability distributions for a response (Yodzis 1998).

A noteworthy feature incorporated in the Punt and Butterworth (1995) approach involved taking explicit account of uncertainty and management issues through the use of a simulation framework that used the feedback control rules factually in place or setting TACs for the hake fishery. The purpose of this approach was to check whether, even if a seal reduction did increase hake sustainable yields, the management system applied to compute TACs was such as to be able to take advantage of this. Cooke (2002) stresses the importance of considering management constraints and issues of uncertainty as integral components of attempts to assess the effects of changing

cetacean abundance on fishery yields. There is currently a paucity of studies addressing this aspect.

The models described below (section 3.4.6) can also be categorised as of the MRM type. The downside of such carefully tailored models is the considerable amount of work and expertise required, but this has to be weighed against the availability of suitable generalised modelling approaches that are sufficiently flexible to be adapted to local scenarios. As an example, models of marine mammal–fishery interactions can be expected to vary substantially between the Northern and Southern Hemispheres given the substantially greater importance of zooplankton (e.g. krill) in the diet of Southern Hemisphere marine mammals.

#### **13.2.4.6 MULTSPEC, BORMICON and GADGET**

These models (and others not mentioned in detail here such as Scenario Barents Sea (Schweder *et al.* 2000), Seastar (IWC 2004) and FLEXIBEST (IWC 2004) are all of Northern-Hemisphere origin and have variously incorporated predation by marine mammals. A common feature is that they are area-disaggregated which is a definite advantage given the migratory behavior of many marine mammals and the consequent importance of considering spatial-temporal overlaps between fisheries, marine mammals and shared prey species. In brief, MULTSPEC (see Bogstad *et al.* 1997) is a length-, age- and area-structured simulator for the Barents Sea that includes cod, capelin, herring, polar cod, harp seal and minke whales. Predation interactions are modeled only as one-way in the case of marine mammals, which do not react to changes in prey availability in the model. BORMICON (A BOREal Migration and CONsumption model) is another area-structured approach for the multispecies modelling of Arcto-boreal ecosystems (Stefansson and Palsson 1998).

Given that work is not currently continuing on MULTSPEC and that BORMICON is being incorporated as a special implementation of GADGET, the focus here falls instead on a brief review of GADGET (Globally applicable Area-Disaggregated General Ecosystem Toolbox) (see e.g. webpage <http://www.hafro.is/dst2/report2/> ; coordinator G. Stefansson). Current case studies include the Celtic Sea, Icelandic

waters and the North Sea and North Atlantic herring. Although marine mammals have not been included in any of the case studies to date, this project is still being developed and shows great promise for modelling these indirect interactions (and has been recommended for such – NAMMCO, 2002) because of some of the following features:

- Population trends can be split by species, size class, age group, area and time step.
- The model platform is flexible in permitting the easy addition/substitution of alternative model components of biological processes such as growth and predator-prey interactions (unlike EwE's restriction to the foraging arena model)
- Change from age- to length-based predation selectivity models is considered an advance given that length is a better predictor of prey vulnerability than age.
- Marked improvements in representing uncertainty are possible given the inclusion of a range of likelihood functions that can be maximized to obtain parameter estimates and their confidence intervals when fitting to data. The ability to conduct global maximization of the likelihood is a definite advantage as is the continuing work to derive improved statistical measures of uncertainty.
- GADGET incorporates a data warehouse that facilitates investigations using data at various levels of aggregation.
- Migration is implemented through movement matrices, so that even if the finer details are not known for marine mammals, these matrices can be used to capture broad seasonal patterns.

As with the other modelling approaches, a major impediment to applying this approach is the current lack of adequate data to describe feeding relationships, especially when considering situations far from the current level. At a more basic level, there is a general paucity of adequate data on cetacean diets (e.g. Steffansson *et al.* 1997).

#### **13.2.4.7 Bioenergetic models**

A separate suite of models include those based on allometric and bioenergetic reasoning. Yodzis (1998) used a 29-species foodweb model incorporating allometric reasoning to investigate the effects of a reduction of fur seals on fisheries in the Benguela ecosystem. However, the model structure was arguably too linear and lacked

age-, spatial- and seasonal structure. More recently, an improved bioenergetics model is being constructed to describe interaction between squid, anchovy, hake and sea lions off the Patagonian shelf (Koen-Alonso and Yodzis in press). These models reinforce the importance of correctly specifying the form of the functional response because of its critical influence on model behaviour. They address many of the problems outlined in the discussions above, but at the cost of requiring much more detailed data.

Although likely to be too data intensive for most investigations, the combined Geographic Information System (GIS) and energetics modelling approach of Bjørge *et al.* (2002) shows much promise. They used radio-tracking data to construct an energetics simulation model of a population of harbor seals in Norway. By integrating their results into a Geographic Information System (GIS) model, they were able to analyse the co-occurrence of fishing operations and seals. Interestingly, they concluded that harbor seal predation probably negatively impacted some fisheries but had a positive effect on shrimp catches due to the removal by seals of benthic-feeding fishes.

#### **13.2.4.8 CCAMLR predator-prey models**

The adoption of the Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR), and particularly Article II thereof (for a discussion of the implications see, e.g., Butterworth 1986), was a crucial step forward in acknowledging the importance of maintaining the ecological relationships between harvested, dependent and related populations of marine resources. Krill is the primary food source of a number of marine mammal species in the Antarctic, and concern has been expressed that the rapidly expanding krill fishery might negatively impact (retard) the recovery of previously overexploited populations such as the large baleen whales of the Southern Hemisphere.

Predator-prey modelling procedures have been developed through CCAMLR to assess the impact of Antarctic krill harvesting on krill predator populations and to explore means of incorporating the needs of these predators into the models that are used for recommending annual krill catch levels (see Chapter 11 for examples). Initial modelling procedures estimated the level of krill fishing intensity that would reduce

krill availability, and hence the population of a predator to a particular level (Butterworth and Thomson 1995, extended in Thomson *et al.* 2000).

A particular concern in CCAMLR has been the potential negative effects of concentration of krill fishing in the vicinity of land-based breeding colonies, which necessarily restrict the foraging range of parents. Mangel and Switzer (1998) developed a model at the level of the foraging trip for the effects of a fishery on krill (*Euphausia superba*) predators, using the Adelie penguin (*Pygoscelis adeliae*) as an example. Their approach of incorporating advection and diffusion processes in a spatial-temporal framework to model krill availability in relation to the location of breeding colonies could usefully be extended and applied to seal populations. Given the large interannual fluctuations observed in krill biomass, these models may also need to include the capacity to incorporate physical forcing of prey dynamics (Constable 2001). Moreover, the modelling approach of Constable (2001) recognises the need to consider potential competitive effects among a suite of predators.

In general, initiatives such as these pursued under CCAMLR recognise the need to balance the needs of predators with the socio-economic pressures underlying fishery harvests. As mentioned in Chapter 11, they represent a realistic step forward in resolving some of the management quandaries resulting from competition for limited marine resources through their focus on the need to account for key levels of uncertainty (Constable *et al.* 2000, Thomson *et al.* 2000), preferably within a strategic and practical framework for developing an ecosystem approach to management.

#### **13.2.4.9 Community indicators to evaluate fishery impacts**

Despite recent advances in this field (e.g. Rochet and Trenkel 2003), a robust well-defined theoretical framework for using ecosystem metrics to evaluate the effects of fishing on ecosystems is still in its infancy (Rice 2000). At most it seems likely that these approaches may provide an early warning that fishing has impacted an ecosystem to such an extent that it is probable that the marine mammal populations are under threat.

### 13.3 ADDITIONAL ECOLOGICAL ASPECTS

Section 3 above focused primarily on so-called feeding-related interactions (cf. Fig. 13.1 A-F), such as perceived direct competition between fisheries and marine mammals for target prey species. This section describes some examples of other more indirect ecological effects (cf. Fig. 13.1 G-H) such as the following:

- Habitat degradation caused by fishing gear and its possible negative impacts on cetaceans (section 4.1 below).
- Effects of noise from fishing operations on marine mammal foraging activities and effectiveness (section 4.1.3 below).
- The role of fishing in altering the structure of ecosystems (section 4.2 below).

Likely effects described include:

- Changes in size compositions of prey species targeted by both fishermen and marine mammals.
- Facilitation of marine mammal foraging because of the role of fishing gear in both concentrating prey and providing easy forage items in the form of discards.
- Fisheries-generated shifts in foraging tactics (for less direct reasons).
- The indirect effect of fisheries in changing predation mortalities (section 4.3).
- The role of marine mammals in acting as hosts for parasites of commercial fish species and the consequent economic costs to the associated fishery (section 4.4). This effect acts in the opposite direction to those mentioned above, i.e. in this case it is the marine mammals that have a negative indirect impact on a fishery.

#### 13.3.1 *Habitat Degradation through Fishing Activities*

##### 13.3.1.1 **Effects of trawling**

Bottom trawling is undoubtedly the greatest fishing-caused disturbance in the ocean, with bottom trawls reportedly annually scraping across an area of the ocean equivalent to half the entirety of the world's continental shelves. Trawling nets often have heavy rollers, chains and wooden "doors" attached to enable them to penetrate some

sediments to a depth of as much as 6 cm, and hence cause substantial non-catch mortalities and discard mortalities on benthic species in the trawls' path.

In a study of the long-term effects of trawling, McConnaughey *et al.* (2000) detected significant differences in the number and types of seafloor organisms between trawled and untrawled areas. However, further investigations of the ecological implications of the short- and long-term effects of trawling are necessary before any definitive conclusions can be drawn regarding effects propagated higher up the foodweb to the marine mammals. That such effects are quite possible is highlighted by a study modelling the effects on juvenile cod survival of creating a range of marine protected areas (MPAs) - of various sizes and configurations - where trawling and dredging are prohibited in portions of Stellwagen Bank National Marine Sanctuary off the coast of Massachusetts (Lindholm *et al.* 2001). They demonstrated dramatic increases in the survival rates of juvenile cod when the highly complex seafloor habitats in Stellwagen Bank were protected in a MPA. Their and other studies which link the survival of juvenile Atlantic cod with the type and quality of seafloor habitat where the juvenile cod settle, suggests that trawling may have important effects on fish populations upon which both fisheries and marine mammals depend.

Examples of marine mammals that are potentially highly sensitive to the deleterious effects of trawling on bottom sediments are the gray whale *Eschrichtius robustus*, which feeds primarily on species such as benthic amphipods, and the walrus *Odobenus rosmarus*, which feeds on molluscs and other invertebrates that live on the seafloor and in bottom sediments. Fisheries-generated habitat destruction may be particularly serious when it affects habitats that are expected to exhibit slow recovery times and/or are disproportionately important ecosystem components such as areas with bathymetric features that render them productivity "hot spots". One example concerns seamounts which are important feeding areas supporting a number of marine mammals. These habitats are in many cases being severely impacted by destructive deep-sea fishing practices (Koslow *et al.* 2000).



### 13.3.1.2 **Destructive fishing practices**

A number of other destructive fishing practices are on record such as the use of explosives or cyanide in coral areas. These practices seriously damage the habitat with consequent repercussions for both fish communities and the marine mammal populations that in turn rely on these habitats for their foraging. Given the mobile nature of cetaceans, they themselves have some capacity to mitigate against such habitat destruction provided this is not overly widespread.

### 13.3.1.3 **Noise pollution**

#### 13.3.1.3.1 *Noise from fishing fleets and fishing operations*

Although one of the least known impacts, noise pollution has been argued to be one of the major causes of concern for marine mammals because of the importance of sound in their adaptive fitness at all levels from the individual to the species (Roussel 2002). Noise pollution from fishing vessels is relatively minor compared to that from other sources. Nonetheless, noise from fishing vessels has been implicated in negatively affecting marine mammals in at least two ways:

- Noise from fishing operations may interfere with the sensitive echolocation systems of toothed cetaceans and thereby indirectly reduce their foraging efficiency. Perez *et al.* (in Roussel 2002) found a negative correlation between cetacean clicks and whistles and shipping noise (from sources such as commercial ships and fishing fleets), either because the cetaceans reduced their echolocation in response to shipping noise or, of equal concern, their sounds became masked by shipping noise.
- Noise from fishing operations may keep marine mammals away from or reduce the amount of time spent in preferred (presumably more productive) feeding areas.

#### 13.3.1.3.2 *Sonars and pingers associated with fisheries*

Echo-sounders are increasingly being used on board fishing vessels to assist in fish and depth detection, but their effect on marine mammals has not yet been documented

(Roussel 2002). The use of acoustic devices (pingers) to, for example, prevent net entanglement has received slightly more attention, but remains a controversial topic because of concerns related to their role in provoking cetaceans to reduce or stop echolocation (Tregenza in Roussel 2002). Moreover, acoustic harassment devices are implicated in displacing both target (e.g. harbor porpoises) and non-target species (e.g. killer whales) from areas where noise has deliberately been introduced into their environment (Morton and Symonds 2002, Roussel 2002).

### ***13.3.2 Role of Fishing in Altering the Structure of Ecosystems***

#### **13.3.2.1 Changes in prey size compositions**

One of the best known effects of fishing is change in the overall size distribution of fish. For example, fishing has caused the larger sizes and some larger species (e.g. halibut) in the North Sea to become uncommon (Pope 1991). This has two main implications for marine mammals. First, given that most marine mammals target somewhat smaller fish than commercial fishermen, a possible effect of an erosion of the larger sizes over time is to increase the competitive overlap between fishermen and marine mammals. On the other hand, the effect of an increased proportion of small fish may be positive in the case of marine mammals with a preference for smaller fish. Secondly (but less likely given the first point above), marine mammals that used to prey on larger fish species or individuals would be forced to shift to preying on smaller fish, with consequent (largely unquantified) negative effects on their bioenergetics.

#### **13.3.2.2 Facilitation of marine mammal foraging**

Some 16 cetacean species (Fertl and Leatherwood 1997) and several pinniped species (e.g. Shaughnessy and Payne 1979) have been documented as feeding in association with trawling. Trawlers can either act as a mobile patch, concentrating food to the benefit of marine mammals (Fertl and Leatherwood 1997), or they can broaden feeding resources by disturbing and/or bringing prey species higher in the water column where they are easier targets for marine mammals.

Bycatch and discards associated with trawling have to various extents played a role in altering benthic community structure (Alverson *et al.* 1994). Apart from habitat modifications resulting from the trawl gear itself, changes in bottom structure may result from factors such as reductions in oxygen saturation (affecting community structure) due to localised dumping of discards such as fish heads and increases in scavenger or decomposer species attracted to such areas (Alverson *et al.* 1994). Alverson *et al.* (1994) stress the fact that discards provide ready forage for several species (including dolphins and seals), and hence that population enhancements resulting from discards should be borne in mind when considering how to deal with discards. On the other hand, Fertl and Leatherwood (1997) caution that shifts in marine ecosystems as a result of trawling may initially seem beneficial to species such as dolphins but may ultimately prove detrimental, especially in cases where trawling ultimately results in severe declines in prey species. They note that whilst trawling often provides more easily captured food (e.g. the long-standing association between bottlenose dolphins and shrimp trawlers) and opens up new feeding niches for marine mammals, it probably destroys other niches. It also exposes marine mammals to greater risk of injury or death associated with fishing operations.

An example of an indirect effect of trawling is provided by the high-seas trawl fisheries off Patagonia which target hake (*Merluccius hubbsi*), a predator of anchovy, which in turn is the main prey item of dusky dolphins *Lagenorhynchus obscurus* (Crespo *et al.* 1997).

Trawling has undisputedly played a role in altering the structure of ecosystems, especially in heavily trawled areas such as the Bering Sea, Mediterranean and Gulf of Mexico, but the longer term effects on the ecology and population status of affected marine mammals is currently difficult to predict.

### 13.3.2.3 Fisheries-generated alterations in foraging strategy

Estes *et al.* (2003) highlighted the important but hitherto mostly ignored role of individual dietary variation in population and community ecology, using sea otters as an example. In response to intraspecific competition, dietary diversity in sea otters

decreases with decreasing population density (Estes *et al.* 1981 cited in Estes *et al.* 2003, Watt *et al.* 2000). That individual differences in dietary diversity may be promoted by strong intraspecific competition adds complexity to the indirect interactions between fisheries and marine mammals. This raises the possibility that reducing marine mammals to increase fishery yields may be much less effective than predicted if the consequent relaxation in intraspecific competition translates into reduced dietary variation among the surviving animals as they switch to feeding to a greater extent on a preferred prey species (which happens to be that of commercial interest). Note that this runs contrary to what the constant suitability assumption underlying MSVPA (see section 3.4.4 above) would predict.

It is noteworthy that several marine mammals display individual variation in diet (on a variety of spatial scales) and actively teach foraging skills to their offspring (Heithaus and Dill 2002). If overfishing substantially reduces the abundance of one prey species, and the principal effect of cultural transmission is one of inertia (because transmitted specialisations act as local optima), then the associated danger exists of apparently “suboptimal” specialisations being maintained over multiple generations (Estes *et al.* 2003). While still in their infancy, studies of interindividual and intraindividual variation in foraging tactics, and the circumstances in which they are employed, will improve the ability to predict the interactions between marine mammals and fisheries (Heithaus and Dill 2002). This could potentially further complicate the specification of functional responses in ecosystem models.

### **13.3.3 Role of Fishing in Altering Predation Mortality**

As already alluded to in section 3.2.4, fishing may play an indirect role in increasing the mortality rates of a marine mammal species as a result of predation by species such as killer whales. The interaction may be relatively complex as with the Steller sea lion – otter example, or more straightforward as when killer whales switch to targeting other marine mammals in instances where overfishing of their preferred fish stocks has occurred. Whether or not killer whales can in fact substantially negatively impact other marine mammal species is currently open to debate. A perhaps more worrisome negative effect of fishing resides in the possibility that as fisheries deplete prey stocks

upon which marine mammals depend, the latter become exposed to increased predation risk themselves by being forced to either increase their foraging time or to forage further afield in areas where they are themselves at higher predation risk (see also Section 11.4.3 in Chapter 11 of this thesis).

Turning the tables, shark fisheries may have a positive indirect effect on marine mammals because of the associated reduced predation on (immature in particular) marine mammals. Sharks represent an important predatory threat to most pinnipeds and some cetaceans (Weller 2002).

#### **13.3.4 Marine Mammals as Hosts for Parasites of Commercial Fish Species**

One of the best known examples of marine mammals indirectly influencing a fishery concerns so-called anisakid nematodes whose larvae use commercial fish and squid for transmission to marine mammals (Raga *et al.* 2002). Although the effects of the larvae scarcely go beyond rendering the fish unappealing to consumers, this naturally has serious economic (marketing) consequences for some major commercial species. For example, the parasite *Pseudoterranova decipiens* (“cod-worm”) infected cod in eastern Canada to such an extent that the losses it caused in 1982 alone were estimated to be \$20 (Canadian) million (Raga *et al.* 2002).

Elsewhere extensive investigations have been conducted concerning the dynamics of seal parasites, the second most common nematode being the “herring worm” *Anisakis simplex*, and their interactions with commercially important fish species (Podolska and Horbowy 2003). For example, Ólafsdóttir and Hauksson (1997) found geographical and seasonal differences in the anisakid infestations of Icelandic gray seals. Ultimately these studies may shed some light on mitigation methods to reduce infestation levels in commercial catches, by coupling to models incorporating seasonal and geographic dynamics of parasite infections. Management measures to ameliorate the adverse effects of this indirect interaction are considered the most sensible way forward given that it seems highly unlikely that any simple linear relationships exist between marine mammal populations (acting as hosts) and parasite loads of fish that feed, for example, on planktonic crustaceans that have in turn fed on free-living parasite larvae. Any

arguments put forward to reduce marine mammals on this basis would thus hardly have an unequivocal scientific basis.

### **13.4 ASSIGNING IMPORTANCE TO INDIRECT FISHERIES EFFECTS IN TERMS OF THEIR POTENTIAL IMPACT**

#### **13.4.1 *Summary and ranking of interaction types***

A number of different indirect fishery interactions have been described in this chapter and the potential importance of each depends to a large extent on the local context. Very generally an attempt has been made to rank the nine major interaction categories identified on a scale from 1 (very important) to 5 (least important) based on an overall impression of the following three factors:

- How widespread and likely the effect/s are.
- The economic repercussions (on a global scale) of the effect/s.
- The extent of likely ecological repercussions.

A summary of the different interactions and provisional rankings is given in Table 13.2. The direct competition for food and fishery resources between marine mammals and fisheries emerges as the major potential impact (to either or both). Again the relative importance of this factor can be expected to vary substantially from region to region.

#### **13.4.2 *Considerations in identifying important interactions***

Based on insights gained in reviewing this topic, it is proposed that the following represent important considerations in assigning importance to indirect fisheries effects in terms of their potential impact:

1. High degrees of overlap (known or assumed) in the type and/or size classes of prey targeted by marine mammals and a fishery. This needs to be assessed within a spatio-temporal framework to account, for example, for the disproportionately large impact of fisheries in instances where they overlap

spatially and/or temporally with increased seasonal or age-specific energetic demands of marine mammals.

2. Scenarios involving species with no easy access to alternative food sources – examples include species with restricted nearshore distributions or limited mobility.
3. Spatial overlaps between commercial fishing grounds and the foraging areas for pinniped breeding colonies (a special case of 2).
4. Scenarios involving species with high degrees of specialisation in prey species (stenophagous) – this applies particularly to species with physiological or other constraints that restrict the range of prey that can be eaten.
5. Trawling in nearshore waters and particularly in areas known to be inhabited by dugongs.
6. Scenarios involving marine mammals with very large and possibly localised population numbers because of the large total amounts of prey required by such populations.

The factors above recognise the importance of considering the impacts of effects acting in both directions. Examples of potentially important fishery effects based on these criteria would therefore include (but are not restricted to):

- Sea otter – killer whale – pinniped – fishery interactions off Alaska.
- Harp seal – fishery interactions in the North Atlantic.
- Cape fur seal – fishery interactions off the coast of southern Africa.
- Krill – predator interactions in the Antarctic.
- Minke whale – fishery interactions in the Barents Sea and North Pacific.
- Dugong – seagrass – trawling interactions.

#### **13.4.3 *Highlighting species of particular concern***

Based on the information presented in Table 13.1, a preliminary synopsis suggests that the following 17 species (not ordered in priority) should be given priority (in terms of data acquisition and focused modelling studies) when assessing the effects of indirect fishery interactions:

1. Antarctic minke whale (*Balaenoptera bonaerensis*)
2. Common minke whale (*Balaenoptera acutorostrata*)
3. Bryde's whale (*Balaenoptera edeni*)
4. Humpback whale (*Megaptera novaeangliae*)
5. Harbor porpoise (*Phocoena phocoena*)
6. Common bottlenose dolphin (*Tursiops truncatus*)
7. Short-beaked common dolphin (*Delphinus delphis*)
8. Killer whale (*Orcinus orca*)
9. Gray seal (*Halichoerus grypus*)
10. Harbor seal (*Phoca vitulina*)
11. Harp seal (*Pagophilus groenlandicus*)
12. Hawaiian monk seal (*Monachus schauinslandi*)
13. Steller sea lion (*Eumetopias jubatus*)
14. Cape fur seal (*Arctocephalus pusillus pusillus*)
15. Antarctic fur seal (*Arctocephalus gazelle*)
16. Sea otter (*Enhydra lutris*)
17. Dugong (*Dugong dugon*)

One additional consideration relates to the conservation status of a species. For example, sirenians face a multitude of threats which are potentially more serious than indirect fishery interactions (in the biological sense of trawling damaging their feeding grounds), but further consideration of these effects obviously needs to have high priority to avoid exacerbating population declines. Similar examples of indirect fishery effects that are important to consider despite being contributory, rather than major, threats to species are interactions involving blue whales (with krill) and humpback dolphins (competing with fishermen for estuary-associated fish – Peddemors 1999).

## **13.5 SUGGESTIONS FOR FUTURE INVESTIGATIONS**

### **13.5.1 *Feeding-related Interactions***

In analysing the diet composition of marine mammals to assess overlaps with fisheries, the following factors need to be borne in mind:



1. The possibilities for using non-lethal methods to determine diet composition and foraging strategies (e.g. Hooker *et al.* 2002).
2. The importance of distinguishing between prey size preference and prey availability. Floeter and Temming (2003) present a good example (albeit applied to North Sea cod) of methods for distinguishing the two. Furthermore, prey preferences determined at local levels by experiment may not be representative of inputs required for interaction models at the population level because of spatio-temporal variations in predator-prey overlap over the region of interest (Lindstrøm and Haug 2001) (discussed in more detail in Chapter 11).
3. The spatial-temporal overlap between marine mammals and their prey. This can be addressed, for example, by weighting prey ingestion estimates by local predator abundance (see e.g. Floeter and Temming 2003). It is particularly pertinent to take this into account because:
  - a) baleen whales, due to their migratory habits, are generally present on feeding grounds for only a portion of the year; and
  - b) some pinnipeds haul-out to breed so that there is a gradient in their use and need for prey as one moves away from their breeding grounds.
4. Age, sex and regional differences in the feeding behaviour of marine mammals (e.g. Flinn *et al.* 2002). Females in particular are likely to be more sensitive to indirect fishery impacts because of their increased energetic requirements and because, for example, the foraging range of female adult otariids is limited by their need to return regularly to the breeding colony to nurse their pup (Boyd 1998, Boyd *et al.* 1998). In some cases it may be important to account for intraspecies variation in diet (e.g. Estes *et al.* 2003).
5. The need to place investigations into an appropriate spatial and temporal framework (e.g. Bradshaw *et al.* 2002).
6. The logistical difficulties involved in working with large whales or with particularly cryptic or elusive animals such as the beaked whales.

### 13.5.2 *Ecological Aspects*

The complexity of marine ecosystems, coupled with strong environmental forcing effects and a range of human impacts, means that it is very difficult to isolate the ecosystem effects of fishing. Nonetheless, it is noteworthy that in most cases where major exploited fish stocks have collapsed, the most likely explanations involve a combination of intense fishing pressure and unfavourable environmental conditions (Gislason 2003). This highlights the importance of an integrated approach in considering the ecosystem effects of fishing.

A selection of points for consideration in improving current understanding of the ecological effects of fishing (on marine mammals in particular) are as follows:

1. Marine protected areas (MPAs) potentially play an important role in protecting foraging and breeding habitats and/or aggregations (Hyrenbach *et al.* 2000), while no-take zones improve the prey base for marine mammals and provide areas of reduced disturbance by fishing operations (Reeves 2000). They also play a potentially valuable role in serving as controls (given the absence of take) to study the broader impacts of fishing on ecological systems and in improving understanding of some of the processes (such as functional responses) (Hilborn *et al.* 2004).

Although static protected areas have only limited utility in protecting the habitat requirements of highly mobile marine mammals, they are the obvious choice in trying to protect important deep-sea habitats such as the Gully, a submarine canyon in the Nova Scotian shelf, that supports a diverse cetacean assemblage (Hooker *et al.* 1999). Research using geographical mapping systems should similarly be used to identify other biological “hot spots” (e.g. seamounts) of particular importance to marine mammals so that an attempt is made to protect at least some of these. For example, WWF (World Wide Fund for Nature) are currently compiling a directory of potential offshore MPAs (for the OSPAR maritime area) that includes areas such as:

- a) the Celtic shelf break that is of known importance to, *inter alia*, short-beaked common dolphins, harbor porpoises and Atlantic white-sided dolphins;
- b) the Fladen, an offshore bank situated in the Kattegat that is an important feeding ground for gray and harbor seals; and
- c) the Galicia Bank, a large seamount where cetaceans and other species tend to aggregate.

Holistic management schemes for entire ocean basins are, however, beset with problems, not least among them the degree of adherence to existing legislations (Hyrenbach *et al.* 2000).

2. Fishery management regimes need to include habitat management issues, but these are currently difficult to formulate given the lack of clearly defined objectives and criteria for habitat variables such as seabed condition (Kaiser *et al.* 2003). To make any appreciable progress in assessing the impacts of fishing on marine benthic habitats, it is important that efforts are focused on large-scale (compared to shorter, smaller-scale) press-and-relaxation experiments (see e.g. Sainsbury 1991, Sainsbury *et al.* 1997, Kaiser *et al.* 2003). The latter need to incorporate, for example, carefully designed scenarios including total and partial exclusions of towed bottom gears.

Given the current lack of information on noise pollution and the potential importance of this factor in disrupting the behaviour of cetaceans in particular, there is a general need for more research in this field and a specific need to identify the nature and extent of noise pollution from fisheries sources on a regional basis (see Hildebrand *in press*).

### **13.5.3 Data requirements**

A framework (UNEP 1999) for assessing the data requirements for scientific evaluation of proposals to reduce marine mammal populations to benefit fisheries is given in Table 13.3. However, these criteria cannot be met even in situations where relatively large data sets are available (Harwood and McLaren 2002). Data requirements for future investigations of fishery effects depend to a large degree on the local context being

considered, so that a comprehensive overview of this topic here would be impractical. Nonetheless, the following general points should be borne in mind:

1. Data requirements should be assessed in relation to modelling requirements, preferably with simulations conducted initially to highlight critical parameters.
2. Clarity concerning stock identity of cetaceans (IWC 2004) is important because a localised effect is likely to have a much greater impact if the animals involved constitute a small, localised stock rather than a component of a larger, rapidly mixing assemblage.
3. There is an overriding need to collect data (experimental and/or field-based, and on foraging in particular) to assist in resolving issues related to the relationships between the rates of predation and the types and densities of available prey (reiterated in Chapters 11 and 12). An example is provided by Lindstrøm *et al.* (2002), who recently showed that the dietary importance of herring to minke whales in the Barents Sea varies with herring abundance in a non-proportional manner. In particular, spatial contrast data or data in the form of time series are required, in contrast to the more conventional snapshot-type estimates. For example, the use of spatial management experiments has been recommended to assist in resolving uncertainties regarding Steller sea lion – groundfish interactions (Committee on the Alaska Groundfish Fishery and Steller Sea Lions 2003).
4. Methodological biases can affect results, as illustrated, for example, by Reid (2002), who cautions that the interpretation of somatic growth rate should not rely on *a priori* assumptions regarding the underlying growth pattern (see also NAMMCO 2002).
5. Data collection efforts should focus on areas where there is the greatest chance of success, either because of the structure of the ecosystem or because there are large signals in the data (e.g. the Antarctic).
6. Attention needs to be given to the consequences of data-related uncertainties and their effects on the bias and variance of quantities estimated from such data.

#### 13.5.4 Modelling studies

The North Atlantic Marine Mammal Commission (NAMMCO) has focused for a number of years on marine mammal-fisheries interactions. For example, workshops have been convened to investigate the role of minke whales, harp seals and hooded seals in the North Atlantic (NAMMCO 1998), the economic aspects of marine mammal-fisheries interactions (NAMMCO 2001), and the main uncertainties in extrapolating from feeding behaviour or stomach contents to annual consumption (NAMMCO 2002), and to model marine mammal-fisheries interactions in the North Atlantic (NAMMCO 2003). Given the conclusion of the first of these workshops, namely that marine mammals have substantial direct and/or indirect effects on commercial fisheries in the North Atlantic (NAMMCO 1998), attention was focused on studies related to competition and the economic aspects of marine mammal-fisheries interactions (e.g. NAMMCO 2001).

In light of uncertainties in calculations of consumption by marine mammals, concrete recommendations were sought with regard to estimating consumption in the North Atlantic (NAMMCO 2002). The next step was to review how available ecosystem models could be adapted to quantify marine mammal-fisheries interactions in the North Atlantic. The following requirements (NAMMCO 2003) are particularly relevant in identifying the desirable features of a multispecies modelling framework:

- 1) Flexibility of functions for prey selection.
- 2) Flexibility of age structuring (from fully age-structured to fully aggregated).
- 3) Accessible code and transparent operation (not “black-box”).
- 4) Able to be tailored to area and species of concern.
- 5) Includes interactions accounting for most of the natural mortality,  $M$ , for species of concern.
- 6) Spatial and temporal resolution able to be tailored for target species.
- 7) Uncertainty in data and model structure reflected in results.

The main groups of models discussed in section 3.4 have here been coarsely assessed in terms of these seven requirements, as well as the additional requirement that marine mammals be explicitly included, rather than treated as exogenous components (Table

13.4). This last requirement is included because the models presented differ substantially in terms of whether they represent:

- i. only the effects of marine mammals on a commercial prey species (e.g. MSVPA, BORMICON, and other models were originally constructed with the primary aim of assessing fish stocks);
- ii. only the effects of prey on marine mammal populations (e.g. CCAMLR models constructed with this aim in mind); or
- iii. effects operating in both directions (e.g. ECOSIM).

Although the finer details of the coarse assessment presented in Table 13.4 could be argued, the general impression is that GADGET and Minimally Realistic Models (MRM), such as the approach of Punt and Butterworth (1995), show the most promise as tools to assess indirect interactions between marine mammals and fisheries. Bioenergetic/allometric modelling approaches such as that of Koen-Alonso and Yodzis (in press) have a role to play too in attempting to characterise the finer details of these interactions. Given that the Antarctic marine ecosystem could be viewed as a case on its own, further development of the suite of CCAMLR predator-prey models (essentially also MRM-type models) is considered the most appropriate approach for this region. The importance of applying different modelling approaches to the same system is stressed (provided that appropriate resources, in terms of both person power and data, are available). This is particularly useful for qualitative cross-checking to determine whether different approaches give similar results, and therefore gauging how much confidence can be placed in their reliability. Furthermore, given the importance of comparing the outputs of different modelling approaches as well as the need to test model predictions both against simulations and against reality, there needs to be an internationally-coordinated effort to provide a structure within which model testing can take place (Boyd, pers. commn).

An appreciation for the need to understand the assumptions underlying each model emerged from both the NAMMCO meeting on modelling marine mammal–fisheries interactions (NAMMCO 2003) and the IWC meeting on cetacean–fishery competition (IWC 2004). Both meetings stressed the need for:

- Careful consideration as to whether or not underlying model assumptions are appropriate for the case under investigation.
- Tests of the sensitivity of predictions to alternative assumptions, particularly regarding interaction terms (e.g. Mackinson *et al.* 2003).
- Addressing uncertainty, in particular by focusing research on the discrimination of alternative assumptions that yield appreciably different predictions.

A further pragmatic recommendation from the IWC meeting was that modelling efforts should focus on specific areas/systems where there is the greatest chance of success. Table 13.5 summarises the key characteristics of such systems as identified at that workshop. One ideal ecosystem for such investigations is the Barents Sea, where there is evidence of relatively tight predator-prey coupling with only a few fish species (herring, cod, and capelin) playing key roles. Systems characterised by strong physical forcing (bottom-up control) are likely to show little or no response to the removal of predators because even strong trophic interactions may be insufficient to increase the spatial and temporal variability in the abundance of a species in systems characterised by high residual variabilities as a result of such physical forcing (Benedetti-Cecchi 2000). The Antarctic ecosystem has often been proposed as a suitable starting point for developing ecosystem models because it is a relatively simple ecosystem that has suffered large impacts from overfishing (e.g. Mori and Butterworth 2004). However, as with other high-latitude regions with short links to high trophic levels, it is subject to large physical variability that may need to be better understood before reliable conclusions can be drawn regarding trophic interactions.

In considering future investigations of marine mammal-fishery interactions, the following additional points are worth reiterating:

- The overriding importance of further investigations regarding the appropriate form for functional responses (the prey-predator interaction terms) and feeding selectivities/suitabilities.
- The need to consider operational (i.e. management) issues, such as how fisheries respond to variability in target stocks and whether perceived gains from a marine mammal reduction are actually realisable, given the various

constraints on fishing operations (including the manner in which total allowable catches are computed) (Cooke 2002).

- The need for more realistic modelling of marine mammal populations in the sense that they are not simply large fish, generally have more limited reproductive (pregnancy) rates, and may exhibit different functional responses.
- The importance of appropriately modelling fish-fish and fish-fishery dynamics before trying to draw conclusions regarding fish-fishery-marine mammal interactions.
- The need for further systematic investigations (presumably through simulation studies) of the numbers of links that have to be included in a non-trivial ecosystem model for reliable predictive ability.

### 13.5.5 *Management decisions*

Evaluating indirect fishery interactions in general requires detailed multi-species biological and fisheries models as well as appropriate datasets. This may become particularly challenging for multinational fisheries involving multiple marine mammal stocks. However, one view is that the debate over culling marine mammals (because of argued competition with fisheries) has less to do with science *per se* than values, attitudes and societal objectives (Lavigne 2003). Lavigne (2003) argues that there will never be an end to calls for culling marine mammals because of political reasons.

Given the demands arising from an increasing human population for more food from the marine environment, coupled to the continued overexploitation of a sizeable proportion of fisheries worldwide (FAO 2002), there is likely to be increased pressure on managers from both fishermen desiring greater yields and conservation groups concerned about the ecosystem effects of fishing and pressing for more cautious management. As has been demonstrated in Newfoundland, economic hardships can strongly influence local opinion as to the perceived impacts of marine mammals on fisheries. In contrast, organisations such as CCAMLR tend to focus on the negative effects of fishing on marine mammals, rather than *vice versa*. A primary difference between these two cases is that in the latter there is as yet no real pressure for greater



krill catches. The real clash on these issues is likely to surface only a decade or so hence, when the demand for fishery products seems likely to outstrip supply.

Because of the difficulties of providing definitive scientific advice on such questions, scientists often equivocate. It is virtually impossible to substantiate, incontrovertibly, claims that predation by marine mammals is adversely affecting a fishery or *vice versa*. In the absence of conclusive evidence, managers are increasingly called upon to apply the "Precautionary Principle" (Principle 15 of the UNCED Rio Declaration (Agenda 21) of 1992), which requires that "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (FAO 1995), but this has been argued both ways in this context: Either that a marine mammal cull should not take place in the absence of clear evidence that it will benefit fisheries, or alternatively that a marine mammal population should be held at its existing level in the absence of clear evidence that the additional consumption of fish accompanying its continued increase will not possibly damage fisheries (Butterworth *et al.* 1988, Butterworth 1999). Hilborn *et al.* (2001) argue that the Precautionary Approach should apply equally to the protection of fishing communities and the resources they depend upon.

### 13.6 SUMMARY AND CONCLUSIONS

Marine mammal species that potentially interact with fisheries at present (or may do so in the future) have been identified here (Table 13.1) and some 17 species that should be given priority in terms of data acquisition and modelling to assess indirect fishery interactions are highlighted. Although in most cases there is insufficient evidence at present to support claims related to indirect fishery interactions, allegations that they negatively affect fishery yields have been made for about 20% of the 63 species of marine mammals. Regarding the reverse concern, tentative links between overfishing and marine mammal population declines have been suggested for at least a dozen species. Two of the most striking arguments for fishing-generated food declines affecting marine mammals are the Steller sea lion and harp seal examples discussed in section 3.2.1. Until fairly recently, only claims of negative effects on fishery yields received much attention. However, the adoption of the CCAMLR Convention some

three decades ago, with its stress on the needs of dependent and related species in determining harvest levels (Butterworth 1986, Constable 2001), has proved a watershed, mandating management actions to ameliorate the potential effects of fisheries on, for example, marine mammals.

Nine different types of biological interactions between marine mammals and fisheries have been identified in this review, and a coarse attempt has been made to rank these on a global basis (Table 13.2). The principal factor of concern is the perceived competition between marine mammals and fisheries, which means that there is an urgent need for scientific evaluations to confirm the existence of such competition and to estimate its scale and extent in terms of its impact on either or both of the mammals and the fisheries.

Despite the persistent assumption by non-scientists that there is a mass for mass equivalence in the prey of marine mammals and the yields available to fisheries, such equivalence is called into question even under the simplest form of analysis. In fact, the complexity of ecosystems could well be such that the response to a marine mammal reduction could be highly dispersed through the food web, involving many other species. In some cases, competition effects are moderated because, for example, one of the putative competitors in fact reduces the abundance of a predatory fish species, in turn affecting the abundance of the target prey species. Although marine mammals are the most obvious scapegoats for fishermen because of their visibility, there is typically greater competitive overlap in the feeding “niches” of fishermen and fish that eat other fish. In spite of these considerations, there appears to be a dichotomy in the current thinking of policy makers, with most parties arguing either that fewer marine mammals would translate into greater fishery yields, or that such links are insubstantial and more consideration should be given to the impacts on marine mammals of prey reductions caused by fisheries.

As more and better information on marine mammal diets becomes available, limited understanding of the feeding strategies of marine mammals remains a key uncertainty. There is a need to quantify not only present diets and their spatial-temporal variation, but also the associated functional responses, i.e., how these diets will change as the

abundances of the various species change. It is important also to bear in mind that some highly specialized marine mammals (or, similarly, highly specialized fishermen) are particularly vulnerable to competition effects because they cannot simply change their diet in response to overfishing of an important food source. Rapidly expanding squid fisheries are one area in which future problems might arise. As yield from the world's major fisheries declines as a result of overexploitation and associated corrective management measures, attention is likely to shift to less-developed fisheries, such as those for cephalopods - the marine capture fisheries predicted by FAO to have the greatest growth in consumption (FAO 2002).

Given current pressures to quantify the predicted effects of marine mammal reductions on fisheries, or, conversely, the effects of increased/decreased fishery catches on marine mammal populations, it is imperative that the models employed be closely scrutinised to understand the extent to which their underlying assumptions predetermine or have implications for the results obtained. Having critically reviewed some of the main models that have or might have been applied in the current context, it is concluded that, in general, GADGET and Minimally Realistic Models (MRM), such as the approach of Punt and Butterworth (1995), show the most promise as tools to assess indirect effects between marine mammals and fisheries.

Table 13.1. List of marine mammals and their potential interactions with fisheries. Taxonomic and distribution information is based on Perrin *et al.* (2002) and Reeves *et al.* (2002). The form of possible competition is described as direct if the marine mammal species and fishery potentially compete for a common food source (see Figs 13.1 A, B, C, F) and as indirect if a fishery potentially affects a predator or prey species which is in turn the prey or predator, respectively, of another species fed upon by the marine mammal species (see Figs 13.1 D, E). Inferences presented in this table are preliminary and largely speculative due to the lack of information and difficulties of conclusively demonstrating linkages. Species thought to be appreciably affected (currently or historically) or particularly vulnerable to indirect fishery interactions are marked with an asterisk in the third column.

Taxonomic group	Common and scientific name	Form of possible competition / interaction	Geographic location	Notes
<b>Order Mysticeti</b>	<b>Baleen whales</b>			
Family Eschrichtiidae	Gray Whale <i>Eschrichtius robustus</i>	Indirect	North Pacific	Effect of trawling on bottom sediments may affect abundance of benthic amphipods, their principal prey
Family Balaenopteridae	Minke Whale (incl. Dwarf minke subsp.) <i>Balaenoptera acutorostrata</i>	* Direct / Indirect	Northern hemisphere Dwarf Minke whale subsp. – Southern hemisphere	Perceived competition as targets cod, herring, capelin and sandlance e.g. ostensible direct competition for cod with fisheries off Iceland and Norway; considered a “key species” competing for fishery resources in the western North Pacific
	Antarctic Minke Whale <i>Balaenoptera bonaerensis</i>	* Direct / Indirect	Circumpolar in Southern hemisphere	a) Possible direct competition with expanding krill fisheries b) Indirect interaction – krill surplus hypothesis (see text)
	Bryde’s Whale <i>Balaenoptera edeni</i>	* Direct / Indirect	Worldwide in tropical to temperate waters	Resident populations e.g. off the southern African coast rely on pelagic fish populations also targeted by commercial fishery; In the western North Pacific ostensibly competes directly with fisheries for Japanese anchovy and indirectly because both whales and skipjack tuna (an important commercial species) target anchovy.
	Fin Whale <i>Balaenoptera physalus</i>	* Direct	Cosmopolitan	Southern hemisphere – possible direct competition with fisheries for krill Northern hemisphere – possible direct competition for herring, capelin and sandlance Possible competition for cod off Iceland and Norway. Hypothesized indirect interaction due to collapse of Canadian capelin fishery (see text).

	Blue Whale <i>Balaenoptera musculus</i>	Direct / Indirect	Cosmopolitan	a) Expanding krill fisheries may impede recovery b) Indirect interaction – krill surplus hypothesis (see text)
	Humpback Whale <i>Megaptera novaeangliae</i>	* Direct / Indirect	Cosmopolitan	Southern hemisphere – possible direct competition with fisheries for krill Northern hemisphere – possible direct competition for herring, capelin and sandlance Possible competition for cod off Iceland and Norway. Hypothesized indirect interaction due to collapse of Canadian capelin fishery (see text).
<b>Order Odontoceti</b>	<b>Whales and dolphins</b>			
Family Physteridae	Sperm Whale <i>Physeter macrocephalus</i>	Direct	Cosmopolitan	Unlikely unless major expansion of deepwater squid fisheries; Ostensibly an important competitor in the western North Pacific; In the Southern Hemisphere competition very likely for deep-sea species such as orange roughy and Patagonian toothfish (eg. off Macquarie Island)
Family Kogiidae	Dwarf Sperm Whale <i>Kogia sima</i>	Direct	Worldwide in tropical and temperate latitudes	Possibility of direct competition for cephalopods, crustaceans and fish in nearshore waters
	Pygmy Sperm Whale <i>Kogia breviceps</i>	Direct	Worldwide in tropical and temperate latitudes	Possibility of direct competition for cephalopods, crustaceans and fish in nearshore waters
Family Ziphiidae	20 species – poorly known	Direct	Pelagic, open oceans	Unlikely unless major expansion of deepwater squid fisheries as many species solely dependent on squid
Family Monodontidae	Beluga <i>Delphinapterus leucas</i>	Direct	Arctic and subarctic waters	Unlikely – highly varied diet but has been deliberately killed in the past due to perceived competition with fisheries
	Narwhal <i>Monodon monoceros</i>	Direct	Arctic Ocean and adjacent waters	Varied diet including herring, cod, halibut, salmon, cephalopods, crustaceans reduces interactions; turbot fishery and prawn fishery could be important.
Family Phocoenidae	Dall's Porpoise <i>Phocoenoides dalli</i>	Direct	North Pacific Ocean	Unlikely, but diet includes several commercially fished species such as hake, herring, mackerel, capelin; Conflicts off Japan
	Harbor porpoise <i>Phocoena phocoena</i>	*Direct / Indirect	Coastal Northern Hemisphere	Diet includes commercially fished species such as herring, mackerel and anchovy; coastal distribution renders interactions likely
	Finless Porpoise <i>Neophocaena phocaenoides</i>	Direct / Indirect	Asian coastal waters	Diet includes small fish, prawns, squid, cuttlefish – may be vulnerable due to living near large population centres in Asia that have resulted in depletion of local fish stocks.
	Vaquita <i>Phocoena sinus</i>	Direct	Northern Gulf of California	Biosphere reserve and some other mitigation measures to try and prevent extinction of this endangered cetacean
	Burmeister's Porpoise <i>Phocoena spinipinnis</i>	Direct	Coastal South America	Unlikely, but possible overlaps with coastal fisheries for small schooling fishes

Family Dephinidae				
Tucuxi <i>Sotalia fluviatilis</i>	Direct	Eastern South America	Possible competition with nearshore and riverine fisheries	
Indo-Pacific Hump-backed Dolphin <i>Sousa chinensis</i>	Direct	Indian Ocean and western Pacific Ocean	Possible competition with nearshore fisheries as their restricted distribution often overlaps with heavily fished areas	
Atlantic Hump-backed Dolphin <i>Sousa teuszii</i>	Direct	Northwestern coast of Africa	Possible competition with nearshore fisheries as their restricted distribution often overlaps with heavily fished areas	
Atlantic White-sided Dolphin <i>Lagenorhynchus acutus</i>	Direct / Indirect	North Atlantic	<p>a) Direct competition unlikely although there is some overlap with commercially fished species</p> <p>b) Indirect: decline in herring in the Gulf of Maine (possibly due to overfishing) resulted in an increase in sandlances, thought to be a major prey item of these dolphins</p>	
Pacific White-sided Dolphin <i>Lagenorhynchus obliquidens</i>	Direct	North Pacific	Perceived competition with fisheries e.g. culling operations off Japan during the 1970s. Overlaps with fisheries in inshore waters for small schooling fishes	
Striped Dolphin <i>Stenella coeruleoalba</i>	Direct	Worldwide in tropical to warm temperate waters	Unlikely although possible threats from overfishing in the Mediterranean	
Spinner Dolphin <i>Stenella longirostris</i>	Direct	Pantropical	Unlikely although some effects on local populations possible	
Short-beaked Common Dolphin <i>Delphinus delphis</i>	* Direct	Tropical to warm temperate waters worldwide	In e.g. Black Sea expanding anchovy and sprat fisheries thought likely to negatively affect local dolphin populations; Conversely, perceived as a threat to the purse-seine and trawl fisheries in the Mediterranean	
Long-beaked Common Dolphin <i>Delphinus capensis</i>	Direct	Warm temperate and tropical coastal waters	Similar to above but less well known	
Common Bottlenose Dolphin <i>Tursiops truncatus</i>	* Direct	Cosmopolitan	Perceived competition with fisheries, especially in nearshore areas. Overfishing in the Mediterranean may be a threat to local populations.	
Indo-Pacific Bottlenose Dolphin <i>Tursiops aduncus</i>	Direct	Indian Ocean and western Pacific	Perceived competition with fisheries, especially in nearshore areas.	
Risso's Dolphin <i>Grampus griseus</i>	Direct	Tropical and temperate waters worldwide	Alleged competition with commercial fisheries (e.g. killed in Japan)	
Long-finned Pilot Whale <i>Globicephala melas</i>	* Direct	North Atlantic and Southern hemisphere	Diet includes several commercial species	
False Killer Whale <i>Pseudorca crassidens</i>	Direct	Worldwide in tropical and warm temperate	Perceived competition e.g. with yellowtail fishery off Japan	

Table 13.1

	Killer Whale <i>Orcinus orca</i>	* Direct / Indirect	Cosmopolitan	a) Direct – depends on locality but e.g. possible competition with herring fisheries in coastal Norway; salmon stock declines may negatively affect killer whale pods resident off western North America b) Indirect – possible altered diet due to overfishing (see text for details)
Subfamily Orcaellinae	Irrawaddy Dolphin <i>Orcaella brevirostris</i>	Direct	Coastal Indo-west Pacific	In coastal and fresh waters has been suggested reduced food supplies from overfishing may be a problem
Family Pontoporiidae (River dolphins – one species marine)	Franciscana <i>Pontoporia blainvillei</i>	Direct	Coastal marine waters from northern Argentina to southern Brazil	Possible direct competition for fish with nearshore fisheries in Brazil, Uruguay and Argentina
<b>Order Pinnipedia</b>	<b>Seals and sea lions</b>			
Superfamily Phocoidea				
Family Phocidae (“true” seals)	Hawaiian Monk Seal <i>Monachus schauinslandi</i>	Direct	Hawaii	Population decline at French Frigate Shoals possibly linked to commercial overfishing of lobsters and associated bycatch species that are important in the diet of these seals
	Mediterranean Monk Seal <i>Monachus monachus</i>	Direct	Mediterranean and Northwest African coast	Threatened species due to several other factors but can't exclude possibility of reduced food supply contributing to pup mortality. Deliberately killed as perceived to be competitor for fish and squid.
	Southern Elephant Seal <i>Mirounga leonina</i>	Direct	Southern hemisphere	Concern has been expressed that the ever-increasing volume of squid caught by commercial fishermen may be negatively impacting Peninsula Valdés population. Possible overlaps if deepwater fisheries expanded. Off e.g. Macquarie Island fisheries for Patagonian toothfish considered a threat to the elephant seals who depend heavily on this deep-sea species.
	Crabeater Seal <i>Lobodon carcinophaga</i>	Direct / Indirect	Southern Ocean	a) Direct: Possible direct competition for krill, its primary prey, if commercial krill harvests increase b) Indirect: krill surplus hypothesis
	Hooded Seal <i>Cystophora cristata</i>	* Direct	North Atlantic Ocean	Suspected direct competition for Greenland halibut and redfish in the Gulf of St Lawrence, although seals primarily target prerecruits to the commercial fishery.
	Gray Seal <i>Halichoerus grypus</i>	*Direct	North Pacific	On the eastern Scotian shelf seals implicated in causing the decline and preventing the early recovery of the cod stock, but model results suggest this is unlikely

	Harbor Seal <i>Phoca vitulina</i>	* Direct	Northern Hemisphere	Potential for competition, for example, in the Moray Firth, Scotland, seals showed evidence of reduced body condition when clupeids absent from inshore waters. Decline in western Gulf of Alaska population correlated with an increase in commercial trawl fisheries – for shrimp in particular. Killed both historically and currently (e.g. in Norway and Canada) due to perceived competition with commercial fishermen.
	Harp Seal <i>Pagophilus groenlandicus</i>	* Direct / Indirect	North Atlantic and White Sea	<p>a) Interactions likely as the harp seal is the most abundant Northern Hemisphere pinniped. Implicated as possibly retarding recovery of cod stocks off Newfoundland and Labrador. Direct competition considered a distinct possibility if fisheries for polar cod are expanded further in the Barents Sea, particularly given the link between harp seal condition and the collapse of the Barents Sea capelin stock in the 1980s. Considered a major predator on capelin in the Barents Sea ecosystem.</p> <p>b) Indirect interactions possible as Atlantic cod (a commercially sought after species) feed on polar cod, also eaten by the seals. Harp seals feed on juvenile Atlantic cod that are not recruited into the commercial fishery.</p>
	Ringed Seal <i>Pusa hispida</i>	Direct / Indirect	Arctic waters	<p>a) Direct competition a distinct possibility if fisheries for polar cod are expanded further in the Barents Sea</p> <p>b) Indirect interactions possible as Atlantic cod (a commercially sought after species) feed on polar cod, a preferred prey species of ringed seals</p>
	Ribbon Seal <i>Histiophoca fasciata</i>	Direct	Okhotsk, Bering and Chukchi Seas	Unknown – possible competition with pollock fishery
	Caspian Seal <i>Pusa caspica</i>	Direct	Caspian Sea	Possible decline exaggerated by loss of suitable food resources due to overfishing
Superfamily Otariidae				
subfamily Otariinae	Steller Sea Lion <i>Eumetopias jubatus</i>	*Direct / Indirect	North Pacific	<p>a) Direct: Population decline possibly linked to decline in food supply partly attributable to commercial fishing</p> <p>b) Indirect: possible competitive release effect re pollock (see text for details)</p>



Table 13.1

	California Sea Lion <i>Zalophus californianus</i>	Direct	California	Potential for competition with fisheries due to overlap of feeding and fishing habitats
	Galapagos Sea Lion <i>Zalophus wollebaeki</i>	Direct	Galapagos	Unlikely
	Southern Sea Lion <i>Otaria flavescens</i>	Direct	South America	Possible competition with expanding commercial fisheries for squid and nearshore fish species. Calls for reductions due to perceived competition e.g. Off Uruguay.
	Australian Sea Lion <i>Neophoca cinerea</i>	Direct	Australia	Unlikely?
	New Zealand Sea Lion <i>Phocartos hookeri</i>	Direct	New Zealand	Small population listed as threatened – uncertain whether competition for fish and cephalopod resources could be important
subfamily Arctocephalinae	South American Fur Seal <i>Arctocephalus australis</i>	Direct	South America	Direct competition likely as prey on several species also targeted by fishermen such as anchovy and lobsters; conflicts in Peru in particular
	New Zealand Fur Seal <i>Arctocephalus forsteri</i>	Direct	New Zealand and Australia	Some conflicts as diet includes some commercial finfish species such as hoki but apparently eat predominantly non-commercial species such as lanternfish, anchovies, octopus
	Antarctic Fur Seal <i>Arctocephalus gazella</i>	* Direct / Indirect	Antarctic	a) Direct: Commercial trawl fisheries pose a threat if allowed to expand near breeding colonies such as those at South Georgia and the Antarctic peninsula b) Indirect: krill surplus hypothesis
	Galápagos Fur Seal <i>Arctocephalus galapogoensis</i>	Direct	Galapagos	Conflicts possible but reduced due to the 1998 establishment of a no-fishing zone around the islands. Problems controlling no-fishing zone
	Juan Fernandez Fur Seal <i>Arctocephalus philippii</i>	Direct	Juan Fernandez Island region, Chile	Unlikely, although could be a problem if e.g. lanternfish and squid fisheries were to expand in the vicinity
	2 subspp. a) Cape Fur Seal <i>Arctocephalus pusillus pusillus</i> b) Australian Fur Seal <i>Arctocephalus p. doriferus</i>	*Direct	Southern Africa and Australia	South Africa – perceived competition with hake fishery in particular (see text) and possible competition with other fisheries as well. Australia – more than half the diet consists of commercially targeted fish and squid species hence concerns re conflicts with fisheries (Southern Squid Jig Fishery in particular) given that the seal population is rapidly recovering from past hunting
	Northern Fur Seal <i>Callorhinus ursinus</i>	Direct / Indirect	North Pacific ocean, Bering Sea, Sea of Okhotsk	Posited competition with Alaska Pollock fishery – interaction is indirect as seals prey on smaller pollock than those targeted by the fishery, but these smaller fish are in turn cannibalized by the larger fish (see text)

Family Odobenidae	Walrus <i>Odobenus rosmarus</i>	Direct / Indirect	Arctic	a) Direct : potential competition with shellfish fisheries b) Indirect: habitat destruction from bottom trawling
<b>Order Sirenia</b>				
Family Dugongidae	Dugong <i>Dugong dugon</i>	Indirect	Indo-Pacific	Trawling reported to damage seagrass beds upon which these sirenians depend
Family Trichechidae	West Indian manatee <i>Trichechus manatus</i>	Indirect	Warm coastal waters of western Atlantic Ocean	Trawling and boat propellers blamed for damaging seagrass beds
<b>Order Carnivora</b>				
<b>Otters</b>				
Family Mustelidae	Sea otter <i>Enhydra lutris</i>	* Direct / Indirect	North Pacific nearshore waters	a) Direct: perceived competition with abalone fishery off California b) Indirect: overfishing claimed responsible for pinniped decreases with the subsequent reduction in prey availability to killer whales resulting in their switching to preying on sea otters
	Marine otter <i>Lontra felina</i>	Direct	west coast of South America	Possible direct competition with fishermen for nearshore marine fish and shellfish

Table 13.2. Initial summary and “strawdog” ranking of types of biological interactions between marine mammals and fisheries. Rankings serve as a guide to the relative potential importance of each interaction (to either or both the fishery and the marine mammals) based on a global overview of the possible number and likelihood of occurrences, and the associated economic and ecological implications. The rankings are not based on any firm quantitative criteria but instead represent the views of the author based on a qualitative assessment.

RANKING	TYPE OF BIOLOGICAL INTERACTION
1 (most important)	➤ Direct competition for commercial fish species
2	➤ Indirect competition for commercial fish species
3	<ul style="list-style-type: none"> <li>➤ Habitat degradation due to trawling and other fishing practices</li> <li>➤ Marine mammals as hosts for parasites of commercial fish species</li> <li>➤ Indirect competition for primary production</li> <li>➤ Changes in shark/killer whale predation rates affecting marine mammals</li> </ul>
4	<ul style="list-style-type: none"> <li>➤ Fishery-induced shifts in the size structure of ecosystems</li> <li>➤ Alterations in foraging strategies in response to fishing effects</li> </ul>
5 (least important)	➤ Effects of noise from fishing operations on marine mammal foraging activities and effectiveness

Table 13.3. Data required to evaluate proposals to reduce marine mammal populations based on fishery impacts, from Table 2 of UNEP (1999).

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**(i) Marine mammal:**

- abundance, distribution and migration
- per capita food/energy consumption
- diet composition, including methods of sampling and estimation
- demographic parameters

**(ii) Target fish species:**

- abundance, distribution and migration
- demographic parameters (weight at age, age at spawning, etc., commercial catch per unit effort)
- details of assessment models and results

**(iii) Other predators and prey of the target species:**

- abundance, amounts consumed, details of stock assessment if any

**(iv) Other components of the ecosystem**

- 2-way matrix of "who eats whom" with estimated or guessed annual consumptions
  - estimated abundance by species
-

Table 13.4. Summary of modelling methods employed to investigate indirect interactions between marine mammals and fisheries. The different approaches are here evaluated in terms of the extent to which they “meet” (have met, or in principle could be applied to meet) the seven criteria identified in the NAMMCO (2003) Workshop Report as desirable features of a multispecies modelling approach (see text for further details), as well as the additional criterion of whether marine mammals are explicitly modelled or simply included as exogenous components of the model.

Model	1) Flexibility re prey selection ?	2) Flexibility re age structuring ?	3) Accessible code and transparent?##	4) Can be tailored to other areas and species of concern?	5) Accounts for most of <i>M</i> of species of concern?	6) Spatial and temporal resolution can be tailored?	7) Uncertainty adequately reflected ?	8) Marine mammal dynamics explicitly modelled?
ECOPATH with ECOSIM (EwE) #	No / some	Some	Not in general but is possible	Partly	Yes	Maybe some (ECOSPACE)	No	Yes
MSVPA	No	Yes	Yes	Limited	Yes	Yes	No	No
MULTSPEC	No	Yes	Yes	No	No / sometimes	Yes	No	No
BORMICON	No	Yes	Yes	No	Yes	Yes	Yes	No
GADGET §	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Minimum Realistic Model (MRM) such as Punt and Butterworth (1995)	Yes	Yes	Yes	Yes in general	Yes	Yes	Yes	Yes
Bioenergetic models (e.g. Yodzis 1998; Koen-Alonso and Yodzis in press.)	Yes	No	Yes	Data limited	Yes	Yes	Yes	Yes
CCAMLR predator-prey models (e.g. Thomson <i>et al.</i> 2000)	No / some	Yes	Yes	No	No	Yes	Yes	Yes

# Note comments w.r.t. EwE mostly refer to “the modelling package” that is widely used, not to modified versions such as considered in Mackinson *et al.* (2003).

§ Note comments w.r.t. GADGET refer more to the potential than demonstrated ability of this approach given that it is still under construction

## Comments here refer at times to the mathematical specification of the code, rather than necessarily to the code itself.

Table 13.5. Summary of system characteristics in relation to modelling feasibility, from Table 4 of the report of the IWC workshop on cetacean-fishery competition (IWC 2004).

<b>System Property</b>	<b>More Feasible to Model</b>	<b>Less Feasible to Model</b>
<b>Data Availability</b>	High or Reasonable	Low or Non-Existent
<b>Food Web Properties</b>		
No of species	Relatively Low	Moderate or High
No of species interactions (i.e. complexity of the food web)	Low	High
Species interaction strength	Strong	Weak or Diffuse
<b>Habitat Properties</b>		
System Openness	Relatively Closed	Wide Open
System Boundaries	Tight and Obvious	Loose or Merging
Depth	Shallow or Moderate Depth	Deep
Physics (e.g. Environmental Forcing)	Low – or else Obvious	High or else Unclear

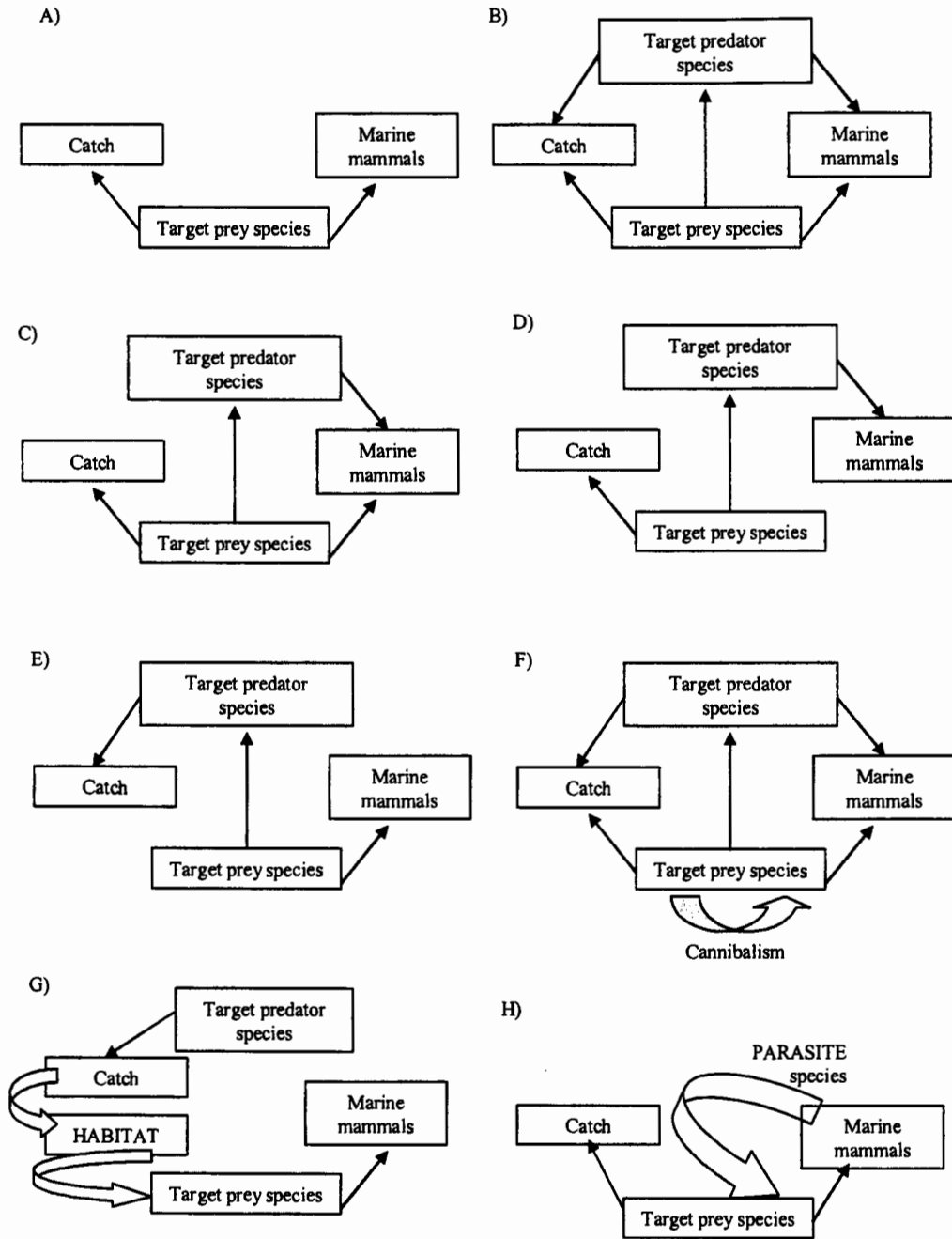


Fig. 13.1. Schematic summary of range of possible indirect interactions between marine mammals and fisheries. The type of interaction may be either that of direct competition between fishermen and marine mammals for a target prey species, predator species or both (A, B, C) or through an indirect foodweb effect whereby a fish species targeted by a fishery is in turn the predator or prey of another species comprising an important component of the diet of a marine mammal (D, E). The various interactions may be further complicated if cannibalism (of the target species in particular) occurs to a marked degree (F). Note that these simplified figures depict predation/fishery interactions only but these various interactions may be mediated to various degrees by environmental, spatial and age-structure effects. Additional interaction representations include scenarios in which a fishery has a negative impact on the habitat of a prey species (example shown in G) and scenarios in which marine mammals indirectly reduce the value of the catch landed by a fishery due to the transmission of parasites (example shown in H).

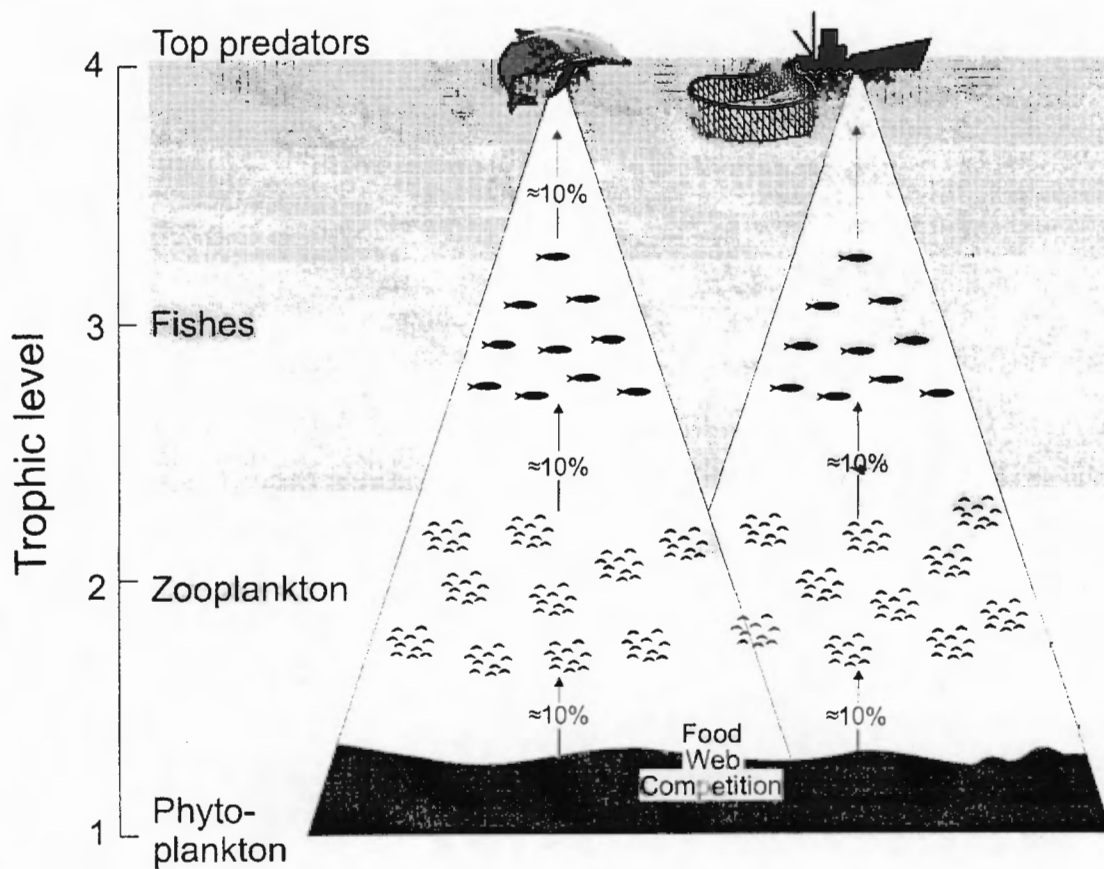


Fig. 13.2 Schematic example of indirect competition for food by marine mammals and fisheries. The representation shows how top predators, such as marine mammals, may be affected by fisheries because of limits on the primary productivity available to support the two groups. Thus even though the mammals' prey and species taken by fisheries may not overlap, so-called food web competition occurs at the base of the food pyramids. (reproduced from Trites *et al.* 1997).





## Chapter 14

### Lessons learnt and challenges ahead

This thesis deals with three topics:

- I) The construction of an age- and spatially-structured stock assessment model for the commercially valuable South Africa abalone *Haliotis midae* resource.
- II) A critical evaluation of ECOPATH with ECOSIM (EwE), which is currently the most widely-employed multi-species/ecosystem approach, as well as an examination of aspects of the potential application of other multi-species/ecosystem modelling approaches to advise the management of South African fisheries.
- III) A review of the indirect interactions between marine mammals and fisheries, as well as of the models used to quantify the nature and magnitude of the various interactions.

The study as a whole represents an attempt to walk the bridge from single- to multi-species approaches to fisheries management by developing a “traditional” single-species stock assessment model that is used for management purposes, assessing possibilities for extending the model to incorporate multi-species effects, and evaluating the potential of a range of multi-species approaches to contribute to the provision of practical management advice.

This chapter knits together some of the findings of this study and places them in a broader context. The discussion commences with a retrospective analysis of the abalone stock assessment and management process. This is followed by a consideration of the lessons to be learnt for multi-species modelling from a single-species stock assessment exercise and some conclusions are drawn regarding multi-species/ecosystem modelling initiatives in the South African context. Finally, recommendations are given for some immediate future multi-species research priorities in southern Africa and the Southern Ocean.

#### 14.1 THE ABALONE EXAMPLE IN A BROADER CONTEXT

A classic stock assessment approach (see Chapters 6 and 7) that attempts also to estimate the extent of the illegal take and the reduction in juvenile abalone survival due to an ecosystem

shift (in the form of a movement of rock lobster into a major part of the range of the abalone) has been used to set annual TACs for the South African abalone resource. However, as outlined in Chapters 2 to 9, this resource is subject to ever-increasing problems as a result of these two factors. This begs the question of whether a reasonably sound stock assessment exercise as applied in these circumstances has played any role other than that of accurately documenting the decline of the resource. Indeed, as highlighted by the review panel at the 2002 BENEFIT Stock Assessment Workshop (relevant extract given as Appendix 2.2), the current modelling methods are critical for highlighting the extent of the problem and evaluating the consequences of possible changes in harvest regime, but do not represent a management solution. The actual solution has been seen to reside in the development of alternative management approaches that better include local communities. In recognition of this, a new policy to define the process of allocating commercial abalone fishing rights was announced by MCM in October 2003 (see Chapter 2 for details).

At the time of writing this thesis, the management of the abalone resource is therefore in transition and the future remains uncertain. Given the mandate of the MCM Abalone Working Group (AWG) to provide advice on the sustainable utilisation of the abalone resource, the assessment model described in this study has contributed as much as has been possible, under challenging circumstances, over the past seven years. But the advent of a new management system requires co-evolution of the assessment method (see Section 14.2.5).

Although the focus of this thesis (and of the AWG) is on assessment rather than management, broader questions have nonetheless been raised such as the extent to which legal commercial fishers should be penalised because of the actions of illegal fishers, and whether classic stock assessment guidelines for recommending catch limits should still apply in non-classic situations. Despite being associated with a management regime that has failed to reconcile fisheries with conservation in this instance, in retrospect, there appear to have been clear economic and biological advantages to having continued these classic stock assessment exercises.

## **14.2 LESSONS FROM A RETROSPECTIVE ANALYSIS OF THE ASSESSMENT AND MANAGEMENT OF THE ABALONE RESOURCE**

### **14.2.1 Prior to the stock assessment model**

Prior to (and immediately following) the development of the first assessment model in 1997, a number of key issues were raised about the abalone resource, such as:

- 1) whether or not the CPUE data (increasing over the 1980s and early 1990s) were believable as indices of abundance – the modelling exercises showed that there was consistency given the catch history of the resource (see Chapter 6); and
- 2) the magnitude of the poaching – the model assisted in quantifying this, through the use of an entirely new index – the CPUPE – or confiscations per unit of policing effort (see Chapter 4).

In contrast to the results of a rigorous stock assessment model that effectively integrates all available information in an objective manner, there were exclamations from a variety of sectors along the lines that: “The resource is doomed – the fishery should be closed – abalone will be commercially extinct in 2-3 years”. The model told a somewhat different story and was criticised for not depicting the resource as sufficiently depleted.

In 1997, a first attempt was made to extend the qualitative decision rule based management approach (see Chapter 2) from consideration of short-term indicators only, to analyses that incorporated all past information and allowed medium-term projections of future trends.

### **14.2.2 Population modelling in retrospect - examples using Zones B and C**

Recent modelling analyses (see Chapter 7) indicate that the “poached” CP subarea of Zone C is currently the most depleted region, with the current spawning biomass estimated at *ca.* 24% of the pre-exploitation level. The inshore region is particularly depleted, with a current depletion estimate of only 12 %. Somewhat ironically, the abalone resource has not (yet!) been assessed to be as depleted to the same extent as its nemesis, the west coast rock lobster (also a commercially valuable species), for which current estimates of the fishable biomass are some 8% of the pristine level (Johnston and Butterworth in press). Whereas a few years previously poaching estimates were equal to or slightly less than the commercial TAC, the very recent explosion of poaching activities has resulted in a poaching estimate for Zones A-

D in 2003 which is more than seven times the legal 2003 commercial TAC for these zones (Chapter 7). These estimates are extremely high, but are quite plausible considering that they correspond to the assumption that, on average, as much as 36% of all poached abalone are confiscated. If the confiscation success rate is actually less than this, these estimates are negatively biased. The clear failure in this instance of fisheries management to achieve conservation begs the question of whether the population modelling pursued has played any role other than that of accurately documenting the demise of the resource?

The “initial” ASPM, developed in 1997, assumed knife-edge selectivities, did not fit to any catch-at-age data and estimated three parameters (the pre-exploitation (i.e. 1953) spawning biomass  $B_0^{sp}$ , the then present annual poaching take  $CP_{max}$  and the natural mortality rate  $M$ ). This model suggested that, at that time, the commercially exploitable biomass in Zone C was some 21% of the corresponding pre-exploitation level (Fig. 14.1), but this was criticised by some as being too high. Projecting the estimates from this original model forwards under “known” total catches and reduced recruitment levels since that time would suggest that the resource should currently be commercially extinct in that zone. Although the resource *is* rapidly headed towards that state, nonetheless, given the considerable takes from that zone, it is evident in retrospect that the Zone C resource could *not* have been more depleted than was indicated by the model in 1997, as some claimed at the time. For both Zones B and C, the updated 2003 assessment, which is based on a more refined model, suggests lower estimates of the pre-exploitation biomasses, but a higher value of natural mortality, and hence higher productivity (Fig. 14.1), than was the case in 1997.

For Zone B, which is not subject to the lobster effect and where heavy poaching started later, the “initial” and updated estimates of exploitable biomass depletion in 1997 are similar (46%), with a current depletion estimate of 26% (Fig. 14.1). However, because the “initial” assessment estimated slightly lower productivity, projections (following the recent very high catches) are more pessimistic under that scenario. The projections to 2010 shown in Fig. 14.1 were computed assuming that current estimated poaching levels are halved in future. However, the current assessment of the situation is that if no reduction in poaching occurs over the next 5 years, the resource will likely decrease to 15% of its pre-exploitation level by the end of that period. Under the “initial” assessment, the resource would be predicted to become extinct within the next few years under both poaching scenarios. Considering the

large catches over the past few years, it is clear (as for Zone C) that the resource in Zone B could not have been, as was claimed by some, much more depleted in 1997 than suggested by the model at that time.

### **14.2.3 Maintaining scientific credibility**

In an era where the Precautionary Principle is increasingly being called upon to guide scientific recommendations for management, these results point (contrastingly) to the need also for caution in “crying wolf”. There is clearly a need for a precautionary approach in making management decisions, but equally scientific credibility may become the real casualty if concerned scientists make unsubstantiated claims. Arguments based on best scientific evaluations, rather than upon qualitative impressions of the state of a resource, may better safeguard the interests of scientific credibility (and hence resource conservation) in the long run.

This abalone example provides a basis for focussing debate on this point. Clearly there has been more than enough reason for concern and precaution in managing this resource. However, the standard stock assessment exercises repeatedly indicated that the resource was not as depleted as some were suggesting. This meant that, based on model results, the TACs set for the industry could be slowly phased down, with greater catches being assigned to the less depleted zones, so as to maintain as much industrial stability as possible over a period when it was hoped to bring poaching under control. During the period 1997 – 2001, the assessment models therefore played an important role in trying to safeguard against further decreases in abundance in the zones most heavily hit by poaching, and to accommodate phase downs in legal allowable catches in these zones by allowing for small decreases in abundance in the less depleted zones (based on projections that took account of estimated future poaching catches). Had the efforts to curb the poaching been successful, this approach would have served the dual function of maintaining industrial stability and conserving the resource as best as possible under the circumstances. However, no-one could realistically have predicted the enormous extent to which poaching catches would continue to increase. Given that management failed to curtail poaching, it seems in retrospect that larger cuts or even complete closure of abalone fishing would simply have meant more abalone for the poachers.

The abalone resource is not the first resource for which catches have exceeded earlier estimates of resource abundance (e.g. VPA assessments in the early 1990s of the number of the largest bluefin (the plus-group) remaining in the East Atlantic and Mediterranean stock were exceeded by the following year's catch on more than one occasion (ICCAT 1994)). This abalone example serves rather to make the point that where there is cause for concern, it is sometimes better not to overstate the case. The original setting for the "tragedy of the commons" (Hardin 1968) was open-access fisheries in which there is little gain to a fisher in trying to conserve fish given that these would then simply be caught by someone else. This seemingly occurs also in situations where there are rumours or announcements of an imminent closure of a fishery, followed by a rush to take as much as possible before closure or before there is nothing left.

#### **14.2.4 Reducing catches synchronous with poaching increases**

Poaching first began in Zone C. Until that time commercial catches were steady, but as poaching continued, commercial catches were reduced in step to compensate (Fig. 7.16). The situation in Zone C was exacerbated by the ecosystem change effect. The situation in Zone B is similar except that poaching escalated rather later in this zone. However, by the time the poachers moved into this zone, the situation had spiralled out of control to the extent that they were removing some 4-8 times as many abalone (in terms of mass) as the commercial fishers (Chapter 7).

Once again, the commercial fishers were penalized because of the large poaching catches. Given this pessimistic appraisal, there was no obvious alternative other than to reduce commercial catches in step with increases in estimated poaching catches. Whereas management aimed to ensure at least some industrial stability, it also needed to try to prevent further resource declines so that the resource would be afforded a chance to recover in the event that poaching could be controlled. Clearly there were no conventional stock assessment linked reference point guidelines for setting TACs for a resource for scenarios in which, in many cases, replacement yields for commercial catch levels (after allowing for ecosystem change effects and anticipated poaching) were negative. For economic reasons, the needs of the industry had to be balanced with concerns that the resource should not simply be effectively handed over to the poachers.

### 14.2.5 The future

Without the assessment modelling discussed above, pre-2000 claims of the enormous levels of poaching would likely to have led to calls to stop all legal harvesting, and hence to close the industry, to “save the resource”. By estimating the level of poaching, the assessment models suggested that, provided poaching could be controlled to some extent, commercial catches would still need to be reduced, but not down to zero. Through such estimation and by keeping the industry going, there were economic advantages forthcoming from the assessment exercise. However, three years later, in 2003, the poaching situation had exacerbated to the extent that a radical change in management approach was necessary.

In the lobster-infected zones such as Zone C, the resource is predicted to decrease rapidly irrespective of whether future catches are high, medium or zero (Fig. 14.2). But there may still be some medium term future for the “normal” zones such as Zone B, depending on whether or not poaching can be controlled (Fig. 14.2).

To summarise:

- 1) The precautionary approach is increasingly being advocated as the basis for advice. Naïve application must be avoided and there must be a balance with the need to maintain scientific credibility through reliance on the best available science regarding a resource, rather than overstatements remembered later as “crying wolf”.
- 2) Given that the recent enormous escalation in poaching was hardly predictable, there appear to have been clear economic advantages to the industry in the classic stock assessment exercises having continued.
- 3) Given that in 2000, poaching was neither predicted to increase further, nor could any decrease through greater law enforcement efforts be guaranteed, there appears to have been no reasonable alternative to penalising legal commercial fishers because of the actions of illegal fishers.
- 4) Conventional resource management target guidelines played a minor role in categorising the situation, but were of no use in recommending commercial TACs.

Given the failure of more traditional top-down management strategies for the South African abalone, a new policy to underpin the process of allocating commercial abalone fishing rights was announced by the government in October 2003 (see also Chapters 2 and 10). This policy



is akin to a Territorial User Rights in Fisheries (TURF) system (Stephenson and Lane 1995, Christy 1996, Caddy 1999, Castilla and Defeo 2001) and was inspired by the apparent success of this system when applied to the shellfish fishery of Chile (Parma *et al.* 2003). It introduces a radical shift in the way South Africa manages its abalone resource. The objectives are to instill a culture of “ownership” amongst right holders and members of the respective coastal communities, and to encourage co-management of the abalone resource (DEAT 2003). It remains to be seen whether or not this change will reduce illegal harvesting substantially.

The system implemented in South Africa differs in a number of respects from the large-scale TURF system which operates in Chile. The Chilean shellfish fishery targets several benthic species such as the “loco” *Concholepas concholepas* (superficially similar to abalone), sea urchins *Loxechinus albus* and key-hole limpets *Fissurella* spp., and includes more than 10 000 commercial divers based in about 250 fishing communities or “caletas” (Parma *et al.* 2003). Implementation of the TURF system in Chile started in 1996 so that the system has now had some time to mature. Thus, for example, new TURFs are only established following the approval of a two-year management plan and, once functioning, annual performance reports (including trends in estimated abundance) are required (Parma *et al.* 2003). One of the key features deemed important to ensure the success of the TURF approach involves empowering and educating fishers to enable equitable participation in management decisions (Parma *et al.* 2003).

In South Africa, preliminary attempts in this regard have met with some success (A. MacKenzie, MCM, pers. commn), but a number of problems remain. Ten-year rights have been issued to a handful of participants in each region or secondary zone (cf. Fig. 10.1), and the right to fish for abalone in a particular zone has (with a few exceptions) been awarded to fishers who reside close to that zone. But the successful rights holders do not necessarily constitute a “community” as such, so that the success of this TURF system will likely depend on the extent to which the local rights holders club together and emulate a “community” eager to enforce their own territorial rights.

However, even if monitoring / data gathering exercises are put in place to measure the performance of rights holders in the different zones, the interpretation of these data will be confounded because abalone are broadcast spawners. Parma *et al.* (2003) note that the

incentives for conserving a reproductive stock within a TURF may be compromised in the case of benthic shellfish because recruitment may originate in neighbouring areas. Lessons that could usefully be adopted from other co-management initiatives and experiments, such as the shellfish diving fishery of northern Argentine Patagonia (see Parma *et al.* 2003), include shifting the emphasis from an overall TAC as the primary management measure, to encouraging greater protection of sublegal animals and close monitoring of the fishing season.

#### **14.2.6 Linking the management of abalone and rock lobsters**

Given that the movement of rock lobsters into the area east of Cape Hanglip (EoH) has negatively impacted the abalone populations in the area, a number of proposals have been put forward to reduce the rock lobster population so as to allow the abalone population a chance to recover. The BENEFIT 2002 workshop agreed that there were no biological reasons to support continued closure of the EoH area to lobster fishing as had been the case, and recommended that the entire EoH catch should be allocated to as small an area (or areas) as possible so as to derive the maximum scientific benefit (in terms of the potential for approximating a large-scale experiment to examine the impacts of reduced numbers of lobsters EoH) (BENEFIT 2002). Moreover, the international review panel noted that consideration might be given to using these lobster allocations to alleviate some of the problems associated with abalone poaching by empowering persons who would otherwise poach to secure an income.

The new abalone policy described above intends to complement the policy that has been adopted for the management of the (commercially valuable) west coast rock lobster fishery east of Cape Hanglip. In an effort to reduce lobster numbers and to simultaneously reduce the pressures on the abalone resource, an allocation of some 100 tons of commercial west coast rock lobster was made on a limited basis to commercial applicants with effect from the last quarter of 2003. Long-term abalone rights holder applicants were allowed to hold abalone and rock lobster fishing permits simultaneously.

### **14.3 FROM SINGLE- TO MULTI-SPECIES MODELS AND FROM ABALONE TO WHALES**

The topics covered in this thesis span the full range from a small benthic shellfish species, the abalone (Section 1) to discussions centred on the largest of marine animals, the whales (Section 2; Chapter 13). Moreover, whereas the first section describes single-species modelling approaches, the focus of the second is on multi-species approaches. The common thread throughout has been the appraisal and application of quantitative approaches to practical fishery management. The study's focus on abalone was largely in response to a need for a quantitative stock assessment method to be applied to this commercially valuable resource. But a number of other important aspects of the study fit squarely with the subject matter of the second section of the thesis, and these are briefly summarised below.

#### **14.3.1 Lessons from and extensions to single-species stock assessment methodology**

In moving towards an ecosystem approach to fisheries management, it is advantageous to gain an understanding of the various single-species stock assessment methods currently in use to advise management. The view expressed in this thesis is that one of the most important steps necessary in this movement is the use of more soundly based statistical estimation methods, as are typically applied for single-species stock assessment models.

The Age-Structured Production Model (ASPM) approach applied here (see Chapter 6) has an advantage over Virtual Population Analysis (VPA) in that it does not require catch-at-age data for all the years considered, and can accommodate likely errors in such data, by making assumptions about the selectivity-at-age of the catch (Butterworth *et al.* 2003a). This is useful in a southern African context as data series are often not complete and may be of variable quality. This thesis presents two examples concerning ways in which the ASPM methodology could be extended to include multi-species interactions: predation of abalone by rock lobsters (Chapter 9) and hake-hake inter-species predation and cannibalism effects (Chapter 12). The ASPM methodology is well suited to this approach because the selectivity-at-age formulations can surrogate as prey suitability vectors, and it is straightforward to add terms to the likelihood function in order to fit to all available abundance indices and other data.

The abalone example called for broader consideration of multi-species and socio-economic factors. The latter were seen to lead to a management regime that proved ineffectual and as

argued earlier in this chapter, highlighted the inability of a stock assessment method (however rigorous) and associated recommendations to successfully achieve conservation when management fails. As stressed by Orensanz *et al.* (in press) and Parma *et al.* (2003), sustainability of a fishery is likely to succeed only when the right incentives are provided, such as in the form of secure long-term access rights.

Regarding multi-species interactions pertaining to abalone, preliminary explorations were made to investigate possible patterns of resource recovery and to caution regarding the trade-offs of embarking on a multi-species management approach (Chapter 8), a framework for a preliminary abalone – rock lobster – urchin multi-species model was constructed (Chapter 9), and the utility of applying ECOPATH with ECOSIM (EwE) was reviewed (Chapter 11).

### **14.3.2 Prudent use of the Precautionary Principle**

Given the difficulties of providing definitive scientific advice on stock status and ecosystem “quality” and interactions, managers are increasingly called upon to apply the “Precautionary Principle” (Principle 15 of the UNCED Rio Declaration (Agenda 21) of 1992), which requires that “where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation” (FAO 1995). However, as argued under section 14.2.3 using abalone as an example, naïve application must be avoided because unsubstantiated claims and overstatements can damage scientific credibility. Arguments based on best scientific evaluations, rather than upon qualitative impressions of the state of a resource, may better safeguard the interests of scientific credibility (and hence resource conservation) in the long run. For this same reason, Chapter 13 stresses the need to soundly and scientifically substantiate claims that predation by marine mammals is adversely affecting a fishery or *vice versa*.

## **14.4 MULTI-SPECIES MODELLING CONSIDERATIONS**

### **14.4.1 Lessons learnt**

As mentioned at various points throughout this thesis, attention worldwide is increasingly being concentrated on establishing frameworks for fisheries management that are ecosystem-orientated, notwithstanding that the operational aspects of this goal are fraught with difficulty (Hall and Mainprize 2004). This field is still very new and this thesis represents a small

contribution to bridging the gap between single-species and multi-species or ecosystem approaches to practical fishery management. This has been achieved by suggesting ways to extend the single-species ASPM methodology currently in use (in southern Africa in particular), and by critically reviewing widely applied approaches such as EwE with the aim of identifying ways to move the system forward to effectively contribute to practical fisheries management advice. The South African abalone resource has served as an interesting example because it has extended across this continuum, starting from the development of a single-species population dynamics model, to a realisation of the need to incorporate first some spatial structure followed by some multi-species considerations. In practice, the latter have been incorporated in an *ad hoc* manner only, because of the difficulties both in the conceptualisation and parameterisation of a more complicated multi-species model capable of explicitly representing the various interactions. Finally, the abalone example has served as an example of the need to consider political, social and economic factors when formulating management advice.

In Chapters 12 and 13 it is argued that consideration of the indirect interactions between marine mammals and fisheries (see Chapter 13) is an appropriate starting point for developing and testing multi-species models because of the lesser number of foodweb linkages for apex predators (Butterworth and Punt 2003). A focus on this issue led again to the conclusion that one of the key issues in moving the development of multi-species models forward is the appropriate form of the functional response formulations to be considered in the models. At opposite extremes, formulations such as that used by ECOSIM depict per-capita consumption by a predator as decreasing with the overall abundance of that predator, whereas constant ration formulations (such as that used in MSVPA approaches) set per-capita consumption as equal to the predator's required daily ration (see Chapter 12). It is strongly recommended that effort be focused on appropriate data collection and / or experiments to assist in shedding light as to the most appropriate choice of model form to represent feeding behaviour. However, extrapolations from the microscale to the macroscale require integrating the form of a functional response over the area concerned, and as cautioned in Chapter 11, independent estimates of parameters at the microscale will not necessarily remain appropriate if the same functional form is assumed to govern macroscale behaviour.

Apart from the ECOSIM functional response (see Chapter 11), which could be viewed as a Type 0 response (Koen-Alonso and Yodzis in review), most multi-species models utilize a

hyperbolic (Type II) functional relationship (Mackinson *et al.* in press). Although difficult to implement because additional parameters need to be estimated, a sigmoidal (Type III) functional response might be more appropriate when modelling generalist predators, such as whales (Mackinson *et al.* in press). This is because these predators are generalists and hence exert less of a strong effect on depleted prey stocks, as can be depicted using a sigmoidal relationship (see Appendix 12.1). Furthermore, a useful approach that could be pursued involves introducing Bayesian methods to take account of variability in and uncertainty about the feeding relationships.

As highlighted by an international review panel at the 2004 BENEFIT Stock Assessment Workshop (BENEFIT 2004) tasked with making recommendations for future research for Cape hakes, the choice of which multi-species models to use needs to be linked to scientific goals and / or management objectives. For objectives related to broad-scale questions regarding the structure of the ecosystem, ECOPATH / ECOSIM models might be used; other models may be more appropriate for more specific questions. Unlike EwE, individually tailored approaches such as MRMs have more flexibility in modelling specific population dynamics, but they often ignore any potential effects that changing prey populations may have on the predators themselves. Given limited person-power and pressure to produce results to assist managers, it is important to engage in discussions regarding which are the preferred modelling approach/es to be pursued in each context. Thus, for example, as a first attempt to address hake-multi-species interactions, the BENEFIT Workshop recommended that existing models should be adapted to provide estimates of the predation mortality on hake that is generated by the two hake species. Similarly, CCAMLR has tended in the past to consider simpler predator-prey type models for the Southern Ocean (e.g. Thomson *et al.* 2000).

While models such as EwE are needed for exploring broader ecosystem questions, it remains critical that users have an appreciation of the underlying assumptions and limitations of “black-box” approaches such as this. By taking into account some of the suggestions outlined in Chapter 11, it may be possible to move forward the agenda of having this approach contribute reliably to practical fisheries management advice. Ideally both simple and more complex multi-species models are needed, and their results should be compared to determine whether they indicate qualitatively similar results.

#### 14.4.2 Role of OMPs

One advantage in a local context is that South Africa has been at the forefront in implementing and promoting the use of an Operational Management Procedure (OMP) (Butterworth *et al.* 1997, de Oliveira *et al.* 1998, Butterworth and Punt 1999), or Management Strategy Evaluation (MSE) approach (Smith *et al.* 1999), to provide scientific recommendations for management measures such as TACs for its major fisheries. The OMP approach has the potential to complement multi-species approaches through its focus on the identification and modelling of uncertainties, as well as through balancing different resource dynamics representations and associated trophic dependencies and interactions (Sainsbury *et al.* 2000). While there is clearly an immediate role for ecosystem models as the “*Testing Models*” (also termed “*Operating Models*”) used to test OMPs, the development of tactical ecosystem models as the basis for computing harvest limits within the OMPs themselves still seems some time off (see Chapter 12). As explained in Chapter 12, Testing Models seek to accurately reflect alternative possibilities for the true underlying dynamics of the resource or resources under consideration. Multi-species or ecosystem models could play a role as Testing Models that provide the basis for computer simulations to project resource trends into the future, to test how well alternative candidate “*Decision Models*” achieve the objectives sought by the management authority. However, it remains to be seen whether or not the associated levels of uncertainty can be adequately constrained to yield scientifically defensible and practically useful conclusions (Cochrane 1998, 2002, Sainsbury *et al.* 2000).

OMPs undoubtedly provide a strategic and practical framework for developing an operational ecosystem approach to management. However, data limitations are likely to restrict the number of multi-species models that reach the stage of being considered viable operating models to assist in the management of target species. At the current level of development in southern Africa, multi-species models cannot provide quantitatively reliable predictions. However, if a variety of alternative plausible models yield qualitatively similar predictions, this could provide a basis for management response.

### 14.5 IMMEDIATE FUTURE RESEARCH PRIORITIES

Several factors have contributed to the current world-wide boom in developing multi-species and ecosystem models to advise fisheries management decisions, with interest in this topic evinced by a number of recent conferences on ecosystem considerations, including the ICES-

SCOR 1999 ecosystem effects of fishing symposium in Montpellier, France (ICES 2000), the 2002 expert consultation on ecosystem-based fisheries management held in Reykjavik, Iceland (FAO 2003b, see also Sinclair and Valdimarsson 2003), and, on the local front, the 2002 workshop on an ecosystem approach to fisheries management in the southern Benguela, held in Cape Town, South Africa (Shannon *et al.* 2004). A number of policy documents pertaining both to South Africa and other nations have attempted to set targets (notwithstanding that these are not necessarily always realistic targets e.g. WSSD 2002 – see below), establish universal definitions of terms such as an “ecosystem approach to fisheries” or EAF (Garcia *et al.* 2003) and formulate guidelines to operationalise EAF by suggesting ways of implementing it at a practical level (FAO 2003 a, b). These initiatives date roughly from the 1982 UN Convention on the Law of the Sea, to the influential 1995 FAO Code of Conduct for Responsible Fisheries and finally to the somewhat ambitious 2002 World Summit on Sustainable Development which “*encourage(d) the application by 2010 of the ecosystem approach..*” and set as a target 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*” (WSSD 2002). Unfortunately the socio-economic reality in most cases of resources below their *MSY* level is that the large short term catch reductions needed to achieve anything other than a relatively slow rate of recovery would not be politically acceptable. Locally, South Africa’s Marine Living Resources Act of 1998 recognises “*the need to protect the ecosystem as a whole, including species which are not targeted for exploitation*” (Anon. 1999).

As motivated in Section 2 of this thesis, it is necessary to caution against unrealistic expectations of substantial progress towards achieving an EAF in the short and even medium term. Locally it appears that considerable data collection and complex analysis over a period of more than a few years will likely be necessary to achieve such reliability, and the associated costs will not be insubstantial. Research priorities in this area need to be carefully and realistically chosen, and weighed against other research needs. It is important to acknowledge that an EAF is expensive to develop and implement, and may result in further costs to the fishing industry in that, in the short-term at least, it may result in less fish being made available to fishers. As stressed at the 2002 Cape Town workshop on an ecosystem approach to fisheries management in the southern Benguela, the long term benefits of an EAF need to be heavily emphasized and clearly explained (D.S. Butterworth, pers. commn), but efforts in this regard are impeded by the fact that there is a current paucity of examples of



successful case studies to show that an EAF is successful and beneficial (K.C. Cochrane, FAO, pers. commn).

In light of the above, summarised recommendations are given below of some immediate future research priorities that could contribute to an EAF both in southern Africa and the Southern Ocean. Given the focus of this thesis, suggestions (and their associated motivations) are listed below first for the abalone fishery and then more generally. The focus here is restricted to biological interactions, in contrast to operational or technical interactions, such as by-catch issues and problems of habitat modification. Comments are made acknowledging that considerably more groundwork needs to be done on general principles of ecosystem management such as deciding on optimal multi-species reference points and the choice of metrics for evaluating fishery impacts, but this aspect is not dealt with further here. The discussion hereunder also ignores that ecosystem objectives should include broader goals such as conservation of biodiversity, protection of endangered species and protection of critical habitat (K.C. Cochrane, FAO, pers. commn). In line with the theme of this thesis, the examples below are those considered important (from a biological and economic viewpoint) in contributing to practical fisheries management advice.

#### **14.5.1 Multi-species modelling recommendations pertaining to South African abalone**

##### *1. Modelling the relative economic gains and losses from the abalone and rock lobster resources in the East of Hangklip (EoH) area*

As described in Chapters 8, 9 and 11, detailed modelling of the abalone – rock lobster – urchin multi-species interactions is complex and not immediately feasible. However, given pressures to increase rock lobster quotas in the EoH region, an immediate priority relates to gaining an improved understanding of the trade-offs involved in harvesting rock lobster heavily in this region with the aim (in theory at least) of allowing some recovery of the abalone resource. The long time-scale (approx. 10-20 years) required for any appreciable recovery of the abalone resource in the “lobster-invaded” areas has been demonstrated at various points in this thesis. A practical starting point to address this issue would involve a relatively simple extension of the approach described in Chapter 8, which builds on the current abalone and rock lobster stock assessment models.

## 2. *Modelling the abalone – rock lobster – urchin multi-species interactions*

Given the paucity of available data and lack of full ecosystem understanding, it is debatable whether a detailed ecosystem approach to this problem will yield practically meaningful conclusions. The complexity of these interactions is also not easily accommodated within the relatively rigid structure of preset models such as EwE (see Chapter 11 for a summary of the potential advantages but also the problems of applying the EwE approach). A more pragmatic approach would be to extend the current abalone stock assessment model to include interactions with rock lobsters, urchins and possibly the substrate, as detailed in Chapter 9. This approach would be extremely flexible in permitting experimentation with a range of different interaction representations and scenarios. It has the added advantage that the consequences for management are immediately obvious within this framework, as results would be output in the same form as for current abalone assessments.

## 3. *Development of an OMP for abalone that takes account of multi-species interactions*

Ideally, an OMP needs to be developed for the abalone resource in the main fishery Zones A-D. As a first step, the current population model described in Chapter 6 could be used as the operating model for the underlying dynamics. Decision models would then need to be developed to take account of three critical factors:

- a) the recent trend in poaching in each secondary zone (or TURF);
- b) the recent trend in CPUE and survey indices in each secondary zone, as determined from finer spatial scale data than that input to the operating model; and
- c) an assessment of the impact of multi-species interactions.

The last of these could be based on any or several of the following:

- i) Data on abalone recruitment success from a dedicated recruitment survey or from a full population survey with coverage in at least one lobster-invaded and one “lobster-free” zone (as was the case for the 2002 MCM/Industry survey).
- ii) Information on the EoH proportion of the rock lobster TAC, in the event that it can be demonstrated that sufficient numbers of rock lobsters have been harvested to allow some recovery of the abalone resource. This relates to item 1. above – note also that this would become relevant only in a few years time given the time-scale needed for a noticeable recovery. Unfortunately it appears that the current EoH rock lobster allocations have *not* been constrained (for social reasons) to be taken from a sufficiently small area to be able to assist in starting to shed further light as

to both the likelihood and extent of a possible reversal of the “rock lobster effect”. On the other hand, given the possible continuing eastward encroachment of rock lobsters into “normal” Zones B and A (S. Brouwer, MCM, pers. commn), this aspect may need to be included in an OMP even if only to serve as a short-term mechanism for reducing TACs in response to an increasing encroachment of rock lobsters.

- iii) Information from models of abalone - rock lobster – urchin interactions. These could either be relatively simple models or more complicated whole ecosystem models (cf. item 2. above). Indications from these models of a short-term enhancement or reduction of the “rock lobster” effect could be fed into a decision model, provided such multi-species / ecosystem models are carefully parameterised and have demonstrated sufficient robustness of their conclusions to uncertainty in the data as well as to a range of plausible alternative hypotheses. In the case of abalone, the development of a tactical ecosystem model as the basis for computing harvest limits within an OMP itself would seem to be a very long way off.

Arguably the most important issue relates to the need for an overall strategic goal by the resource managers regarding the abalone and rock lobster resources in the East of Hangklip region (e.g. should effort be concentrated on removing rock lobsters from the EoH region – or even just from east of Danger Point – or should an optimal mix [if possible] of the two resources be attempted?).

Although the discussion above focuses on a modelling perspective, the best approach to this problem would likely depend on experimental studies and an adaptive management approach (e.g. Walters 1986, Hilborn and Walters 1992, Sainsbury *et al.* 2000). For example, an actively adaptive management strategy applied to the Australian multi-species fishery was successful in resolving key uncertainties about resource dynamics and sustainable resource use (Sainsbury *et al.* 1997). The approach involved identifying four different plausible hypotheses and adopting an experimental process involving the sequential closure of areas to trawl fishing. After a period of a few years, the experiment was successful in discriminating among the competing hypotheses (Sainsbury *et al.* 1997, 2000).

## 14.5.2 Multi-species modelling recommendations pertaining to southern Africa and the Southern Ocean

### 1. Review of underlying shortcomings and assumptions of available multi-species / ecosystem approaches

This aspect is seen as critical to advancing attempts to incorporate ecosystem considerations in practical fisheries management. Unfortunately endeavours in this regard appear to be lagging considerably behind the ever-growing number of documented applications of ecosystem models. Critical reviews of methods such as EwE, as presented in Chapter 11, assist in highlighting weaknesses and hence ultimately in strengthening applications of an ecosystem approach. Conventional single-species modelling approaches used to inform the management of commercially important stocks are typically subject to intense scrutiny. Ecosystem models are likely to be subject to a similar level of scrutiny when they reach the state of being used as the basis for management recommendations or decisions (with implications for economically valuable fisheries in particular). There is therefore a need for parallel processes of model development, application and scrutiny – otherwise the danger exists that considerable time and effort will have been wasted in developing an ecosystem model that is later rejected out of hand when it attempts to enter the management arena.

### 2. Models of inter-hake and hake-seal-fishery interactions

As discussed in Chapters 12 and 13, the Cape hake consists of two species, a shallow-water (*Merluccius capensis*) and a deep-water species (*M. paradoxus*), with the larger of the shallow-water species eating the smaller individuals of the deep-water species. Moreover, off the South African west coast, the fur seal population (*Arctocephalus pusillus*) consumes substantial quantities of hake. There are several reasons why further modelling of these interactions is seen as a priority, including:

- a) Hake is the most valuable of the local fisheries resources (Butterworth and Rademeyer in press b).
- b) There is an urgent need, both locally and world-wide, for methods and models to quantify the perceived competition between marine mammals and fisheries (Plagányi and Butterworth 2002, in press, see Chapter 13).
- c) Considerable work has already been completed (Punt and Butterworth 1995) in constructing a MRM of this system.

- d) At the very least, it is necessary to model the dynamics of the two hake species separately in order to more accurately assess the relative impacts (and/or optimal ratio) of trawling compared with long-lining, as these two methods of fishing impact the two species differentially.

Given prior work in this area, a logical first step would involve extending the current stock assessment models, for example, by including predation effects as a separate fishing fleet, as described in Chapter 12. “Tailor-made” modelling approaches require considerable expertise and personpower, but it may be possible to modify and update the Punt and Butterworth (1995) MRM following the suggestions outlined in Chapter 12. Finally, extensive work in implementing ECOPATH and ECOSIM models that include hake and fur seals has already been carried out for the southern African region (e.g. Shannon *et al.* 2000). As concluded in Chapter 11, these EwE models have yet to reach the stage where they can reliably contribute to answering management questions such as those outlined here. However, there is considerable scope for improving these approaches (see e.g. suggestions in Chapter 11) and they provide a useful over-arching structure for a preliminary overview of more general ecosystem aspects pertaining to hake.

### 3. *Systematic analyses of alternative functional response formulations to be considered in models*

Although progress in this field is primarily impeded by a lack of suitable data and experimental studies (noting that the focus here is on recommended modelling endeavours), simulation and modelling studies can nevertheless contribute. This issue is critical (as stressed in Chapters 11-13) and hence attention should be focused both on the need to carefully check model robustness to alternative interaction representation hypotheses, and on simulation exercises to systematically and thoroughly explore this issue.

### 4. *Pelagic fish escapement levels*

As explained in Chapter 11, a topical local question concerns the level of escapement of pelagic fish necessary to maintain populations of threatened and scarce seabirds in the region. This is seen as a priority both because of the important commercial value of the pelagic fishery (and hence concerns by the industry that catches should not be cut unnecessarily), as well as the conservation status of affected seabirds such as the

vulnerable African penguin *Spheniscus demersus* (Barnes 2000). Two of the most appropriate (and complimentary) ways to proceed are as follows:

- i) Rather than initially attempting a full ecosystem model, a first step might involve using a relatively simple seabird model to estimate seabird prey requirements. This information could then be fed into the existing OMP (de Oliveira *et al.* 1998, Cunningham and Butterworth 2004) in such a way that pelagic fish catches are reduced in the event that there is deemed to be inadequate prey to support populations of threatened or scarce seabirds. Obviously it would also be important to determine what are desirable population levels of these seabirds. Although prior models of seabird dynamics have been constructed in the region (e.g. Crawford *et al.* 1992, Shannon and Crawford 1999), these models cannot be straightforwardly adapted for this purpose because they failed to introduce density dependence into the seabird dynamics. As stressed by Butterworth and Plagányi (2003), it is important to appreciate with such models that assuming juvenile seabird survival rate depends only on pelagic fish abundance (and not also on the size of the seabird population) does not provide a viable model structure from which to draw inferences of the effect of pelagic fishing on seabirds such as penguins.
- ii) A spatial and temporal modelling framework is needed to assess the impact of closing to pelagic fishing areas in the vicinity of seabird (and seal) breeding areas to reduce any effect the fishery may be having on the foraging success of breeders. Such a model needs to take account of both diffusive mixing and removals by both fishery and predators. One suggestion would be to build on models such as that by Plagányi *et al.* (1999, 2000) used to explore the effect of different geometric distributions and degrees of synchrony in the abundance of anchovy and its zooplankton prey off the South African west coast. Plagányi *et al.* (2000) used a spatial framework to explicitly model shoals of anchovy recruits feeding on patches of zooplankton prey, and quantified the fish's performance through temporal and spatial integration of periods and patches of prey abundance and shortage. This work is accorded priority for the additional reason that it has the potential to inform similar initiatives by CCAMLR concerning krill fishing in the vicinity of krill-dependent predator colonies (CCAMLR 2004, see also Chapter 11, Section 11.6).

### 5. *Antarctic multi-species krill-whale models*

Relatively simple multi-species models such as that by Mori and Butterworth (in press) of Antarctic krill-baleen whale-fur seal interactions off Antarctica have the potential to contribute to multi-species dilemmas such as the extent of inter-specific competition for krill between blue whales (*Balaena musculus*) and minke whales (*Balaenoptera acutorostrata*) (see also Chapter 11, Section 11.6). The Antarctic is a pragmatic choice as an area for developing multi-species models because it is a relatively simple system in terms of the number of important species involved, and is simultaneously the region where the largest human induced perturbation of any marine ecosystem in the world has occurred (Mori and Butterworth in press). It therefore meets the requirement stressed in Chapter 13 that modelling efforts should focus on areas where there is the greatest chance of success, either because of the structure of the ecosystem or because there are large signals in the data. Moreover, approaches such as that by Mori and Butterworth (in press) provide a sound basis from which to formulate krill-centric MRMs to serve as operating models for testing candidate control rules for recommending krill TACs at a wide spatial scale level. Valuable lessons can therefore be learnt from the Antarctic experience and transferred to other regions.

### 6. *Models to advise regarding the expansion of “under-developed” fisheries (such as horse mackerel *Trachurus trachurus*) which are important prey for other commercially valuable fisheries*

Faced with growing socio-economic needs, there is likely to be increasing pressure to optimally exploit all available fishery resources. In a multi-species context, determination of optimal exploitation levels of a suite of species is far from trivial in the case where some of the species serve as prey for other species. This multi-species problem is considered a priority because it also ties into addressing a number of related questions such as that of the competition between marine mammals and fisheries – in this instance the same method could be used to compute optimal harvest levels of a commercial fish species that is also an important prey item for a protected marine mammal species. Any one of the various multi-species methods discussed in this thesis (e.g. EwE, GADGET, MRMs) could be used to shed light on this issue. The most appropriate choice depends on the availability of both suitable data and expertise.

In conclusion, some progress has already been made towards achieving an EAF in southern Africa and the Southern Ocean, but as yet it represents only a few pinnacles in a large ocean. As demonstrated in Section I of this thesis, the correct incentives and management structures need to be firmly in place if success is to be achieved. To reach this goal it is insufficient simply to perfect existing models. Stakeholder participation and dialogue need to be seen as integral components of multi-species fisheries management, and scientists need to avoid the temptation to use loosely constructed ecosystem models to justify a preferred point of view. There is considerable work ahead in ensuring that future generations are tasked with managing fisheries in a way that is both ecologically and economically sound.



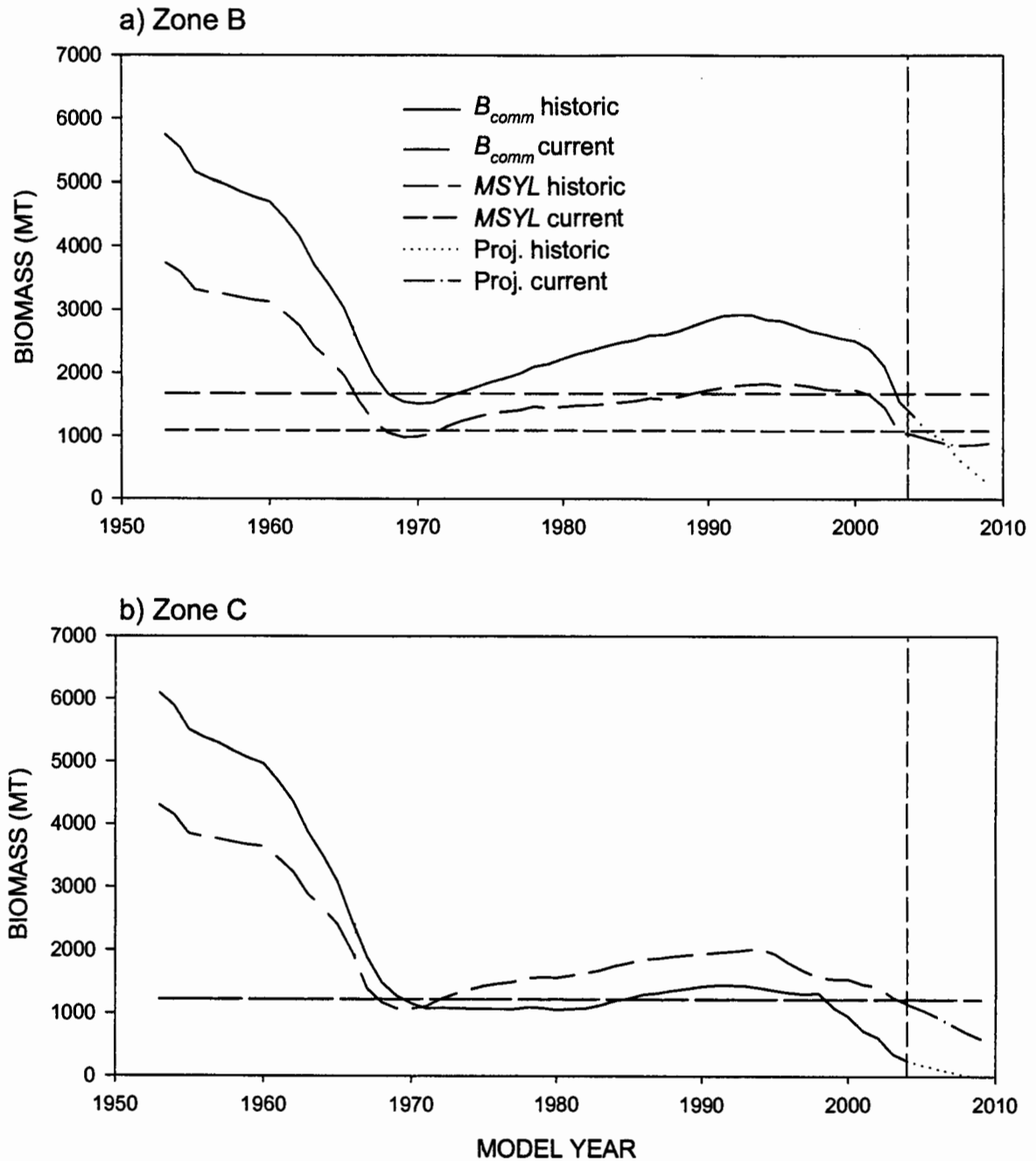


Fig. 14.1. Commercially exploitable biomass trajectories for a) Zone B and b) Zone C shown for current (2003) and initial (1997) assessments. Projections for the initial assessment are effected using catches subsequently taken, whereas future projections (after the vertical dashed separator) assume that future poaching is at half the current estimated level and that commercial catches remain fixed at the 2004 TAC. For Zone C, the same post-1990 juvenile mortality rates are assumed as in the current assessment.

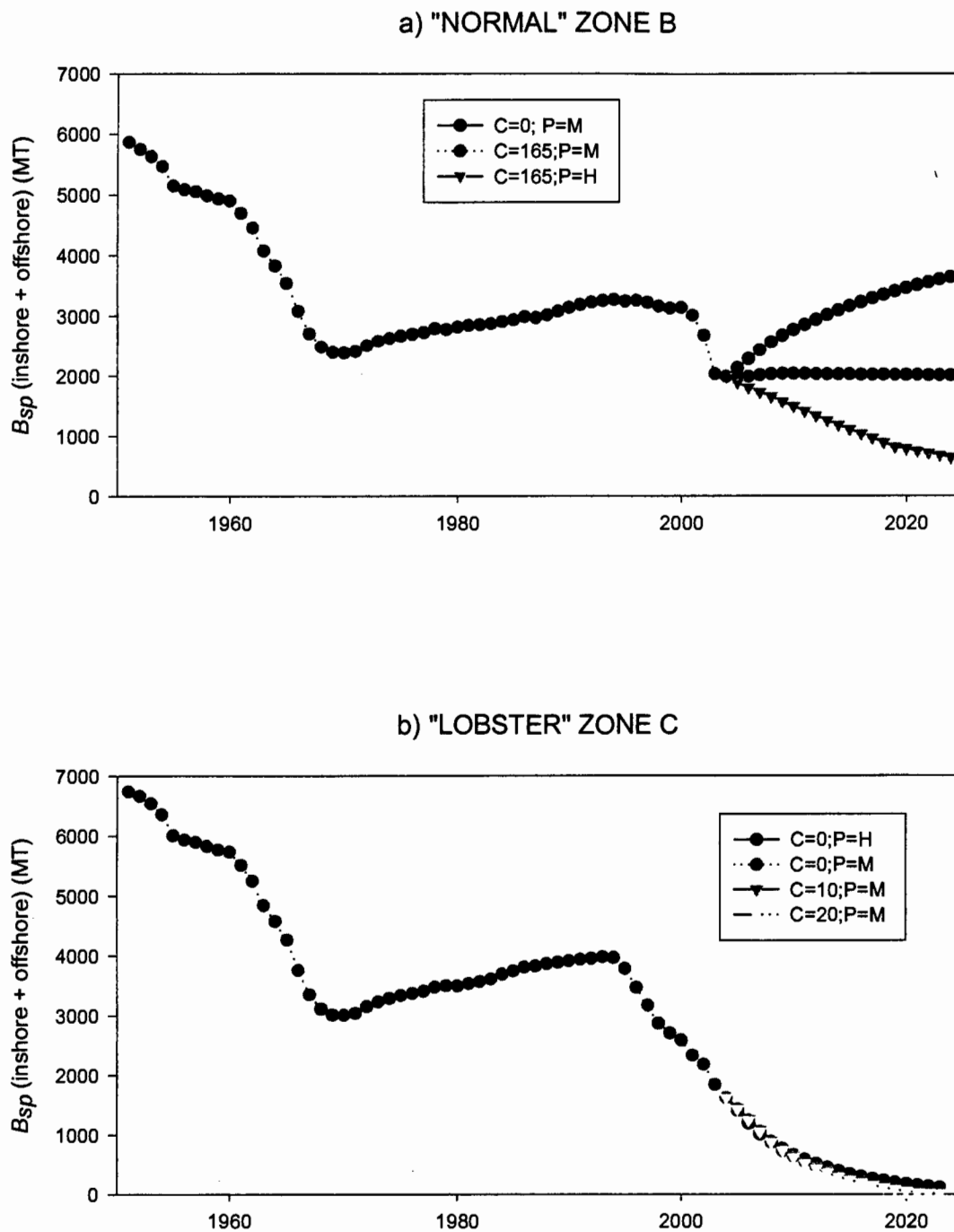


Fig. 14.2. Projected spawning biomass trajectories for a) Zone B and b) Zone C (CNP and CP combined) under alternative scenarios as indicated, where C represents the fixed future commercial catch (MT) , P=H implies future poaching is at the current (high) estimated level and P=M implies "Medium" future poaching at half the current estimated level.



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**APPENDIX A1: South African Abalone Working Group Documents authored / co-authored by the author of this thesis**

1. Plagányi, É.E. and D.S. Butterworth 2003. Summary of further projections for abalone in Zones C and D assuming a reduction in future poaching levels. Marine and Coastal Management task group document, 03/10/01: 5 pp.
2. Plagányi, É.E. and D.S. Butterworth 2003. Further projections for abalone in Zones C and D assuming a reduction in future poaching levels and some recovery of abalone in response to the removal of west coast rock lobsters from the East of Hangklip (EoH) region. Marine and Coastal Management task group document, 03/09/01: 9 pp.
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Table 12.1. Summary of number of hake stomachs sampled across the whole study area over the period 1988-1994 (from Punt and Leslie 1995). These stomach content data were used (following further analyses) as inputs in the Punt and Butterworth (1995) model with the key values of the fraction of the diet consisting of Cape hakes shown in the lower part of the Table (from Punt and Leslie 1995).

a) Number of samples - all data pooled		
Length-class (cm)	<i>M. capensis</i>	<i>M. paradoxus</i>
1-20	214	299
20-30	425	509
30-40	524	454
40-50	468	396
50-60	485	382
60-70	437	353
70+	388	263
TOTAL	2941	2656

b) Fraction of diet (%) consisting of hake		
Predator age	<i>M. capensis</i>	<i>M. paradoxus</i>
1	8	93
2	9	54
3	13	60
4	13	36
5	46	71
6	19	30
7	22	31
8+	20	21

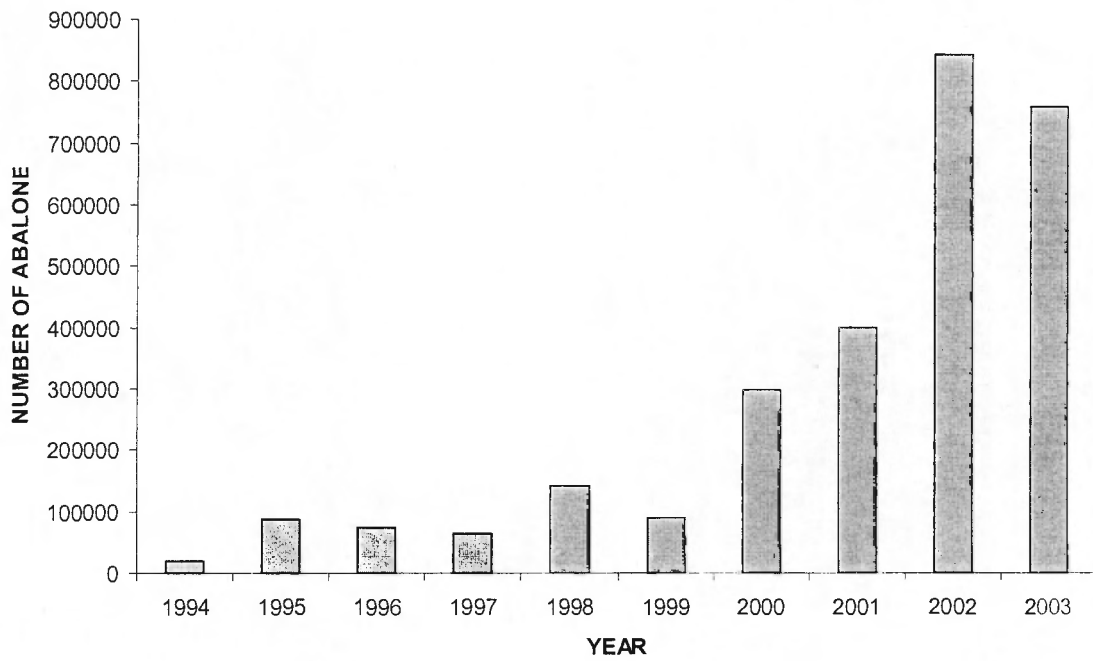


Fig. 2.3. The total number of poached abalone confiscated by law enforcement officers in South Africa from 1994 to present (data from A. MacKenzie, MCM, pers. commn).

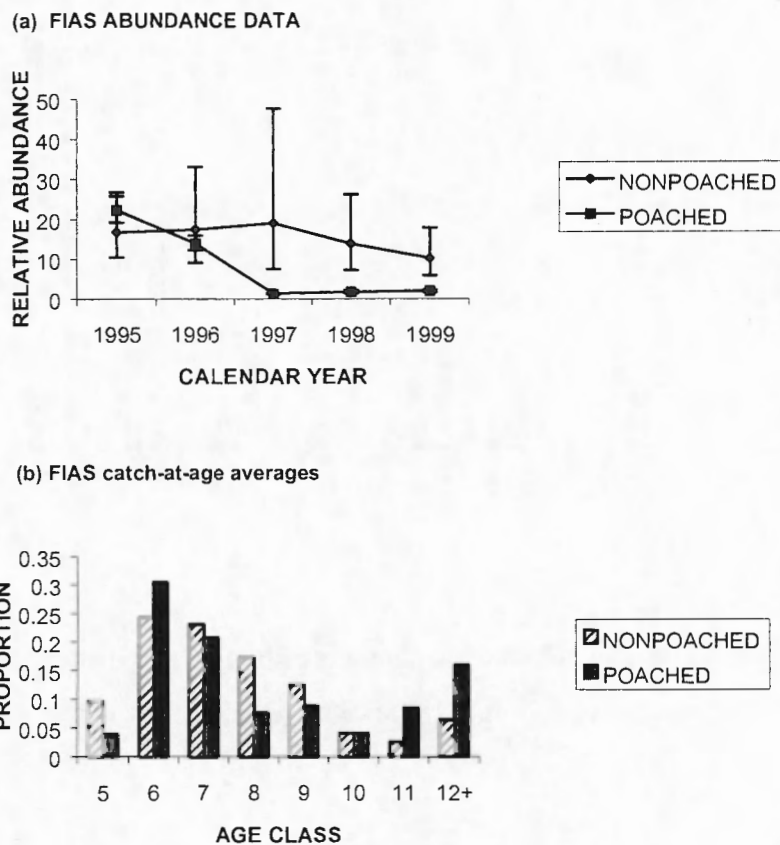


Fig. 2.4. Comparisons between FIAS data obtained for the “poached” (CP) and “nonpoached” (CNP) subareas of Zone C. (a) shows the trend in the relative abundance index (average number  $\bar{x}$  per 60 m<sup>2</sup>) in each subarea. The error bars indicate the 95% confidence intervals calculated as  $\bar{x} * e^{\pm 1.96 * CV}$ , i.e. assuming a lognormal error distribution. (b) compares the catch-at-age data averaged over the period 1995-1999. The catch-at-age data are derived from length distribution data (from A. McKenzie, MCM) that is cohort-sliced using a von Bertalanffy growth curve (Tarr 1995).