



MANAGEMENT PROCEDURES FOR CAPE HAKE AND BALEEN WHALE RESOURCES

by

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Part I

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ABSTRACT

This thesis considers the selection of a "best" management procedure to provide a basis for TAC recommendations for the South African west coast Cape hake resource from a large number of alternatives. It also outlines the development of a management procedure which could potentially be used to regulate catch limits for baleen whale resources. All of the management procedures investigated are of a feedback (closed loop) nature. The ability of a procedure to achieve the objectives specified by the management authority is assessed using Monte-Carlo simulation methods. The two resources considered differ in that Cape hake are relatively productive and short-lived, whereas baleen whales are characterized by slow dynamics and extreme longevity.

A model of the west coast Cape hake resource and its associated fishery involving the effects of age- and length-structure, cannibalism, spatial effects and stochasticity is constructed and parameterized using available data. The operating models used to generate data to test candidate procedures are all sub-models of this general model. Most of these sub-models ignore length-structure, cannibalism and spatial effects, but many still take account of the two species (*Merluccius capensis* and *M. paradoxus*) nature of the resource. There are currently two defensible assessments of the resource. The production model assessment [based on the observation error version of Butterworth and Andrew's (1984) model (Schaefer form, with the biomass at the start of exploitation set equal to carrying capacity, i.e. $B_{1917}=K$)] indicates that the resource is increasing following overexploitation, but that it is still a little below the biomass at which Maximum Sustainable Yield is achieved. In contrast, the standard application of the Virtual Population Analysis (VPA) assessment method suggests that the resource is both much more depleted and less productive than does the production model. In order to take account of this uncertainty regarding current status, two sets of values for the operating model stock-recruitment relationship parameters are chosen (one reflecting the production model and the other the VPA assessment).

One hundred and nine management procedures (based on production model, partially age-structured model, *ad hoc* tuned VPA, and Integrated Analysis estimators) are applied to the "base case" operating models which ignore the two-species nature of the resource. Many of the procedures perform dismally. Procedures based on estimating time-trends in catchability-at-age within a VPA, on total least squares estimators and on Integrated Analysis estimators all fail, probably as a consequence of the attempt to estimate too many model parameters. The failure of the latter two classes of procedure may, however, also be a consequence of frequent convergence to a local rather than the global maximum of the likelihood function. Procedures which attempt to estimate the biomass (relative to the pre-exploitation level) at which the surplus production function attains its maximum, those based on partially age-structured models, and those which utilize survey but not CPUE data also perform poorly.

On the basis of these results, the number of procedures considered is reduced to 24. These remaining procedures are then tested using operating models which take the two-species nature of the resource into account, and are also applied to a large number of variants of these two-species "base case" operating models. As the results for these two-species trials are not markedly different from those for the corresponding single-species cases, it can be concluded that treating the west coast hake resource as one stock for management purposes is unlikely to lead to poor performance. The procedures are generally robust to the uncertainties considered. Robustness trials which do result in notable degradation in performance are ones involving: substantially increased recruitment variability, and time-trends in the catchability coefficient or in the age-at-50%-selectivity. Introducing length-structure or spatial effects into the operating model does not compromise performance, but the introduction of cannibalism leads to substantial changes. However, cannibalism results in the resource being naturally more resilient, so that conservation-related objectives are easier to attain if cannibalism is substantial.

Of the procedures considered, none of those based on age-structure data are able to perform satisfactorily, particularly because they exhibit unacceptably large inter-annual catch variability, and are often associated with a high probability of reducing the resource to an undesirably low level. This raises the question of the necessity for future collection of these data. In contrast, certain of the production model based procedures are able to perform satisfactorily, and show an ability to achieve reasonable resource recovery even if the substantial overexploitation indicated by the VPA assessment of the resource reflects the actual situation. These results are specific to the fishery considered, and are undoubtedly sensitive to the life-history parameters, current status and exploitation pattern of the resource. Future research should consider the extent to which they are general by examining performance over a wide range of these factors.

It is clear that the estimator component of a "best" procedure must be based on the observation error version of a production model. Of the harvesting strategies investigated, the RY strategy is eliminated as a viable alternative by its inability to effect sufficient resource recovery when the VPA assessment reflects the actual situation. $MAC_{0,n}$ strategies are preferable to $f_{0,n}$ strategies as they achieve similar total catches, but do this with less inter-annual catch variability and often with slightly less risk of unintended depletion. The Fox form of the surplus production function performs best of those investigated. Thus, the "best" procedure of those considered is the Butterworth-Andrew (Fox form; $B_{1917}=K$) observation error estimator together with either the $MAC_{0,1}$ or the $MAC_{0,2}$ harvesting strategy. As the choice between these strategies rests on relative preferences for higher catches and for lower risks of unintended depletion, it cannot be made on purely scientific grounds and is a matter for the management authority.

The procedure developed to manage baleen whale populations in cases in which stock identity is known, is based on a simple age-aggregated population dynamics model. The estimator component of the procedure assumes observation error dominance and utilizes only sightings survey estimates of absolute abundance (which become available once every five years). It incorporates a prior distribution on the MSY rate (MSYR) parameter to reduce inter-assessment fluctuations in the estimates of this parameter, and hence in catch limits. The catch control law includes a probing component which gives greater catches if the resource is assessed to be near its pre-exploitation size, to increase data contrast and hence improve parameter estimation. Catch limit variation restriction rules and rules to handle situations in which the period between surveys is longer than expected are also incorporated.

This single-stock procedure performs remarkably well, showing robustness to most of the uncertainties considered. It has a low risk of unintentionally further reducing stocks which are already substantially depleted, and leads to reasonable resource utilization over a wide range of values for MSYR and the initial depletion level. The procedure can be easily tuned to achieve different trade-offs between the major objectives of high catches, a low risk of extinction and small inter-annual catch limit variability. The scenarios in which the performance of the procedure are slightly disappointing are when the population size is 0.6K when management commences, and when the absolute abundance estimates are biased. Further, the procedure grossly underutilizes resources with MSYR=7%.

"Jacket" procedures, built around the single-stock procedure, are developed to handle the problems associated with uncertain stock identity, which are of interest in the "pelagic" and "coastal" whaling trials developed by the IWC's Scientific Committee. The key difficulty in these situations is to avoid unintentional depletion of stocks with low productivity. This problem is overcome by managing by small subareas instead of by large management areas, and by judicious choice of values for control parameters. However, risk levels can only be kept low at the expense of gross underutilization of stocks with high productivity.

Issues which should receive further attention are whether or not the range assumed for MSYR (in terms of the recruited component of the population) for the trials (1 - 7%) is appropriate, and whether or not generic or case-specific management procedures are to be preferred for multi-stock situations.

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CHAPTER 1 -INTRODUCTION

1.1 The objectives of this study

Ideally, the management of renewable marine resources should be based on applications of management procedures (though, in reality, it is never possible to remove the political/ economic considerations from decision making). The management procedure is a complete management "package" which incorporates specifications related to how and when data should be collected, how these data are to be processed to obtain estimates of current resource size and potential productivity levels, and finally, how to obtain management recommendations from such estimates. The process of applying a management procedure in an actual situation is illustrated in Figure 1.1.

The selection of a management procedure for a particular resource involves consideration of the extent to which various alternative management procedures are able to satisfy the (often conflicting) objectives of the different use groups (in their broadest terms, these objectives relate to the needs of the entire country/region concerned). Thus sound selection of a management procedure necessarily involves considerable interaction between scientists, decision makers, the industry and interest groups, and this interaction should preferably take place in an iterative manner.

This thesis considers the development of management procedures for two resources. The first, Cape hake off the west coast of South Africa, are relatively short-lived but quite productive animals, whereas the

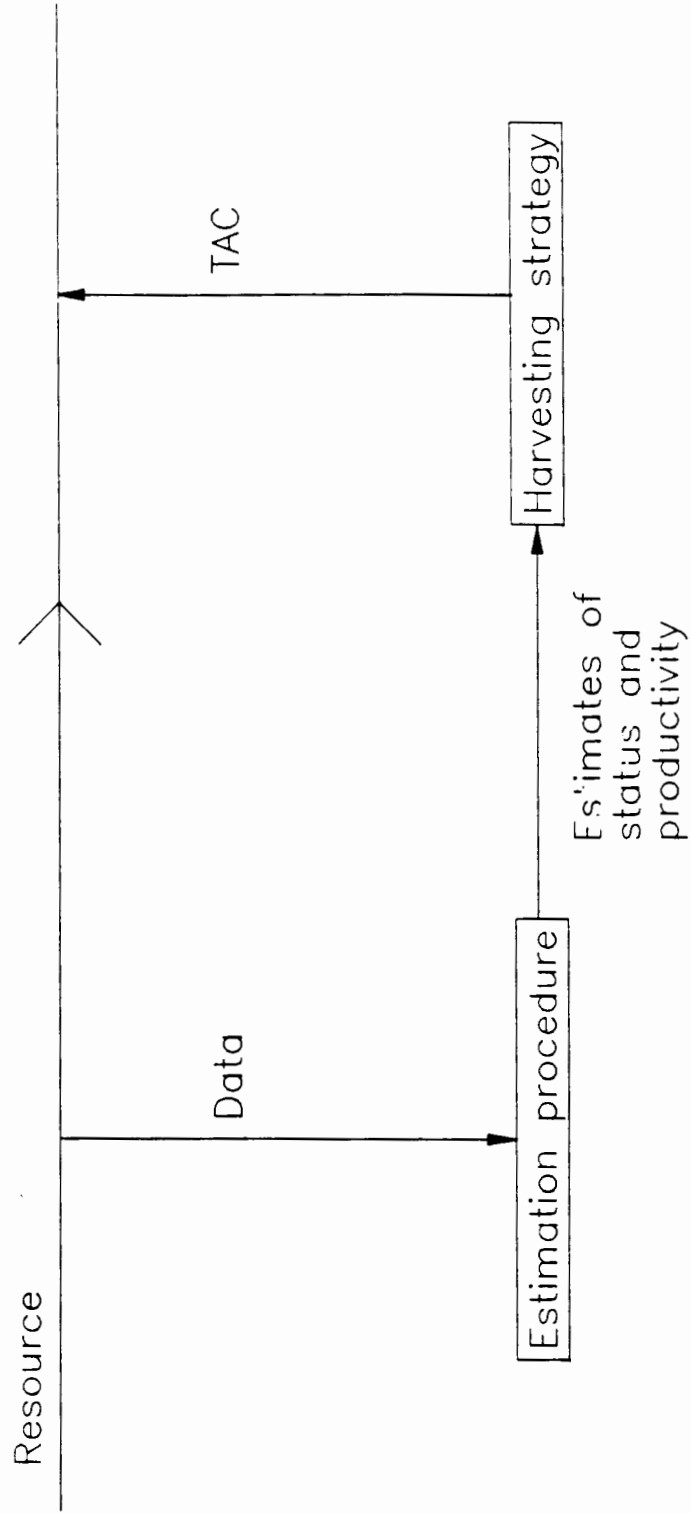


Figure 1.1 : The process of applying a management procedure. Only one "loop" of the process is shown. This "loop" is repeated each management interval (typically each year for fisheries).

second, baleen whales, are characterized by slow dynamics and extreme longevity.

The choice of management procedures made here might be useful as starting points for the development of management procedures for other local resources - for example, the procedure developed for baleen whales may provide a basis for one for kingklip, which is also a long-lived species (Japp 1990).

The thesis has been divided into two parts. The first part (Chapters 1 to 15) deals with the determination of the performances (in terms of trade-offs between the various management objectives) of a number of alternative management procedures for the Cape hake resource considered. The second (Chapters 16 to 19) considers the construction and testing of a management procedure for baleen whale resources.

The remainder of this Chapter addresses five topics. Section 1.2 defines some of the terminology used throughout the thesis. Sections 1.3 and 1.4 deal with what the managers of a renewable resource are likely to consider as their objectives in utilizing that resource (in somewhat general terms), and what they may deem desirable features of a management procedure. Section 1.5 describes how to select a management procedure once a set of objectives has been specified quantitatively. Section 1.6 provides a more detailed introduction to the section of the thesis dealing with hake, and Section 1.7 does likewise for baleen whales. To aid the reader, short summaries have been given at the start of the following Chapters.

1.2 Terminology

Certain terms used frequently throughout this thesis are often used loosely or with different definitions in the fisheries literature. The meanings of these terms as used in this thesis are detailed below. A full list of the symbols used in the thesis is given in the glossary.

1.2.1 Overexploitation

Although various interpretations of the term "overexploited" exist (see further discussion in Section 1.3), in this thesis it denotes biological overexploitation, i.e. it indicates that the current biomass is below that which corresponds to the production of maximum sustainable yield. This last level is referred to as the MSY level, or MSYL. The term "depletion" refers to stock size as a fraction of its average pristine (pre-exploitation) level (often expressed as a percentage). Note that "depletion" therefore reflects the extent to which (not by which) a stock has been reduced.

1.2.2 Estimation

In order to reduce confusion resulting from the paucity of suitable synonyms for the word "estimate", the term "model-estimation procedure" is used to indicate the combination of a model and the method employed for estimating its parameters. [The "method" (or "estimator") referred to here embodies an assumption about the error structure of the model, and the specification of a minimization criterion to be used in the estimation process.] The acronym "VPA" refers to the age-structured estimation technique Virtual Population Analysis (see Chapter 7). For

ease of presentation, estimates are not denoted by " $\hat{}$ " in some of the formulae because it is obvious from the context that they are such. "Accurate" and "precise" refer respectively to low bias and small variance.

1.2.3 Other definitions

The word "significant" is used only to indicate results which are statistically significant at the 5% level. A management procedure is said to dominate another if it is able to achieve all of the objectives equally or more satisfactorily than the other.

1.3 The objectives of management

Any management process involves defining a policy, collecting information, making decisions and evaluating the results of those decisions (ACMRR 1979). In marine resource management, decisions relating the system inputs, i.e. the resources available (biomass, vessels, etc.) to the outputs, i.e. the desired objectives (maximum profits and protein, etc.) should be based on scientific advice, the quality of which depends on the scientists' ability to assess the state of a resource both accurately and precisely.

The objectives of the different user groups involved in the management of marine resources are often at variance. Factors which must be considered before making decisions include:

- a) catch - this should be as large as possible over an unlimited period (Babayán and Kizner 1988),

- b) economic rent (profit) - the industry requires profits for continued operation, and these are likely to be greatest if catches remain stable (to avoid the additional costs associated with fluctuations in production levels) and catch rates remain high,
- c) depletion - the probability of depleting the resource below some critical level at which its production would be substantially impaired should be small,
- d) multispecies considerations - undesirable changes to the marine ecosystem in which the targeted species exists should be minimized (for example, a reduction in a prey species may result in a reduction in its a predator population - this may be undesirable for a fishery based on the predator population), and
- e) social and political goals - these, although important, are often very difficult to quantify and are therefore usually excluded from explicit consideration by scientists when making management recommendations.

In most cases, the dominant aim of the managers is to avoid "overexploitation". Obviously this term is related in some way to the extent to which a resource is depleted, but it is usually not defined. Suggested interpretations include (Walters 1986):

- (i) biological overexploitation - the resource is depleted below the biomass level which provides the maximum sustainable yield (MSY),
- (ii) recruitment overexploitation - the spawning stock is reduced to a level at which the average recruitment is markedly diminished,

- (iii) economic overexploitation - either the net economic rent (profit) from harvesting is nil (or negative) or, more strictly, fishing effort is above the level which provides maximum economic rent, and
- (iv) overcapacity - there is more capital/labour available in the fishery than is necessary to harvest the allowable catch, or to realize the maximum economic rent.

Objectives a) to e) cannot be satisfied simultaneously and compromises must be reached. For example, stable catches can only be achieved by accepting a lower average catch or a higher probability of depleting the resource below the critical level. All of the management procedures considered in this study implicitly select a particular trade-off between the three major objectives considered (namely: maximizing average catch, achieving stable catches, and preventing an unintended degree of depletion). Although some of these trade-offs are clearly undesirable in most circumstances (for example, those for a management procedure which provides high average catches in the short term is associated with a very high probability of resource collapse), it is often very difficult to select between different trade-offs (or even to suggest appropriate approaches to addressing this selection problem).

1.4 Potentially desirable properties of management procedures

In order to achieve the management objectives (which should be defined quantitatively) as best as is possible, a management procedure should have certain properties. Some of these are discussed below.

1.4.1 Ability to make accurate and precise predictions of management-related quantities.

The trade-off between accuracy and precision is of considerable importance when designing a management procedure. Biases in the estimates of parameters and variables can arise if the management procedure does not model the underlying processes adequately. [This is referred to as "error due to approximation" by Linhart and Zucchini (1986).] Lack of precision [or "error due to approximation" - Linhart and Zucchini (op. cit.)] arises from attempting to estimate the model parameters from a data set of finite size. Hence, even if a model (with its associated error structure) is a very accurate representation of the underlying fishery (low "error due to approximation"), predictions, and therefore possibly management performance, may be impaired if the available data prevent precise estimation of the model parameters (high "error due to estimation"). The "total error" is a combination of the bias and variance. The estimation process should aim at minimizing this total error. In some cases, simple models are able to make better predictions than more complex models in this sense, because gains of lesser bias for the complex models are more than offset by greater variance, in contributions to the total error.

Although the accuracy and precision of management related quantities *per se* are of interest to managers, a management procedure can be designed so that its performance is not severely affected if the estimates of some of the management-related quantities are somewhat biased or imprecise. Nevertheless, if the estimates of quantities on

which the management procedure is critically dependent are very poorly determined, then that procedure is unlikely to perform well.

1.4.2 The models underlying the management procedure should represent the basic causal factors

Many people, even some scientists, take the validity of this statement to be self-evident. But is it actually correct? Modelling even the most simple causal process may require considerable complexity. As noted above, the ability to predict well (a prerequisite for sound resource management) depends on the trade-off between bias and variance. None of the management procedures considered in this thesis are based on model-estimation procedures which attempt to model the basic causal biological processes in detail, because the data sets available are too small to justify the estimation of many (any) more than approximately three parameters.

1.4.3 The management procedure should be self correcting

All the management procedures considered in this study are of a feedback (closed-loop) nature. This means that they update their assessment of the status and productivity of the resource as more data become available, using either or both of future catch (and sometimes effort) data and future information independent of the fishery (e.g. research surveys). This means that if successive sets of parameters estimated from finite sized data sets result in TACs (for example) which are substantially larger or smaller than those which should be taken, the resultant change (beyond that intended) in the biomass will be reflected in the data collected, so that later assessments will

provide more accurate and precise estimates of the management-related quantities. This may allow the management procedure to start to correct its mistakes, so that subsequent TACs are closer to the values required to meet objectives.

De la Mare (1990) states that management procedures which are able to learn by means of their feedback nature are more accurate, provide more stable catch limits, are better able to attain management objectives and are less sensitive to error than management procedures which ignore future data. It is, in fact, difficult to imagine a scenario in which managers would not prefer to update assessments as more data become available. (Assuming, of course, "impartial" managers - fishing company executives may be tempted to ignore the biomass estimate from a recent research survey if this turned out to be less than had been expected!)

1.5 Determination of whether a management procedure can satisfy the management objectives

The determination of the performance of a management procedure cannot be achieved by applying it to a real resource for a number of reasons, some of them 'in principle', and others related to practical considerations. Certain of these reasons are listed below.

- a) The time period required. Management objectives often pertain to a lengthy period of time (even decades). Hence, if an experiment is actually to be performed, it would take a considerable amount of time before the results were available. Furthermore, it is unlikely that the management procedure would be applied exactly as intended throughout the period

- considered - this would complicate assessment of the procedure's performance.
- b) It would probably be impossible, even after applying a management procedure over a long period, to determine exactly what the status of the resource was relative to what had been the objective in that respect. It would thus not then be possible to assess accurately whether or not that objective had been achieved.
 - c) The consequences of the application of what proves to be a poor management procedure to a real resource may be highly undesirable. For example, if a management procedure results in excessive depletion of a resource, it may take several years (or even decades) to correct this, with concomitant major industrial and social dislocation.

Because of these difficulties, determination of the appropriateness of management procedures is carried out by means of Monte-Carlo simulation [for example, Hilborn (1979); Fournier and Warburton (1989)].

Simulation testing involves constructing a number of mathematical/statistical models ["operating models" (Linhart and Zucchini 1986)] of the fishery. These should reflect various alternative (yet plausible) scenarios for the fishery. Linhart and Zucchini (op. cit.) define an operating model to be "the nearest representation of the true situation which it is possible to construct by means of a probability model".

A large number of simulations (reasonable computer time restrictions dictated a value of 100 in most instances considered in this thesis),

each allowing the management procedure to manage the simulated resource(s) (represented by the operating model) for a number of years, is performed (Figure 1.2). The results of these simulations are summarized by means of a small set of performance statistics (also referred to as the summary statistics). These statistics are used to determine the trade-offs between the objectives specified by the managers.

A number of alternative operating models are considered instead of only one, because for a real fishery, the true situation is usually only very poorly understood. In this thesis, the performance of management procedures for a "base case" operating model and variants of this model (usually each involving only a single change to the "base case" model) are investigated. These results are used to determine the sensitivity of the performance of management procedures to the assumptions of and parameter values for "base case" operating model. It is desirable that a management procedure be insensitive (robust) to such changes, because it is never certain which (if any) of the operating models considered best reflect the "true" situation in the fishery.

Simulation testing removes the problems associated with experimentation on real resources because:

- a) poor procedures only exterminate "resources" on a computer,
- b) the state of the resource is known exactly (to the analyst, though not to the procedure under test) and at all times, and
- c) each simulation only takes a small amount of computer time (usually a few minutes) to carry out.

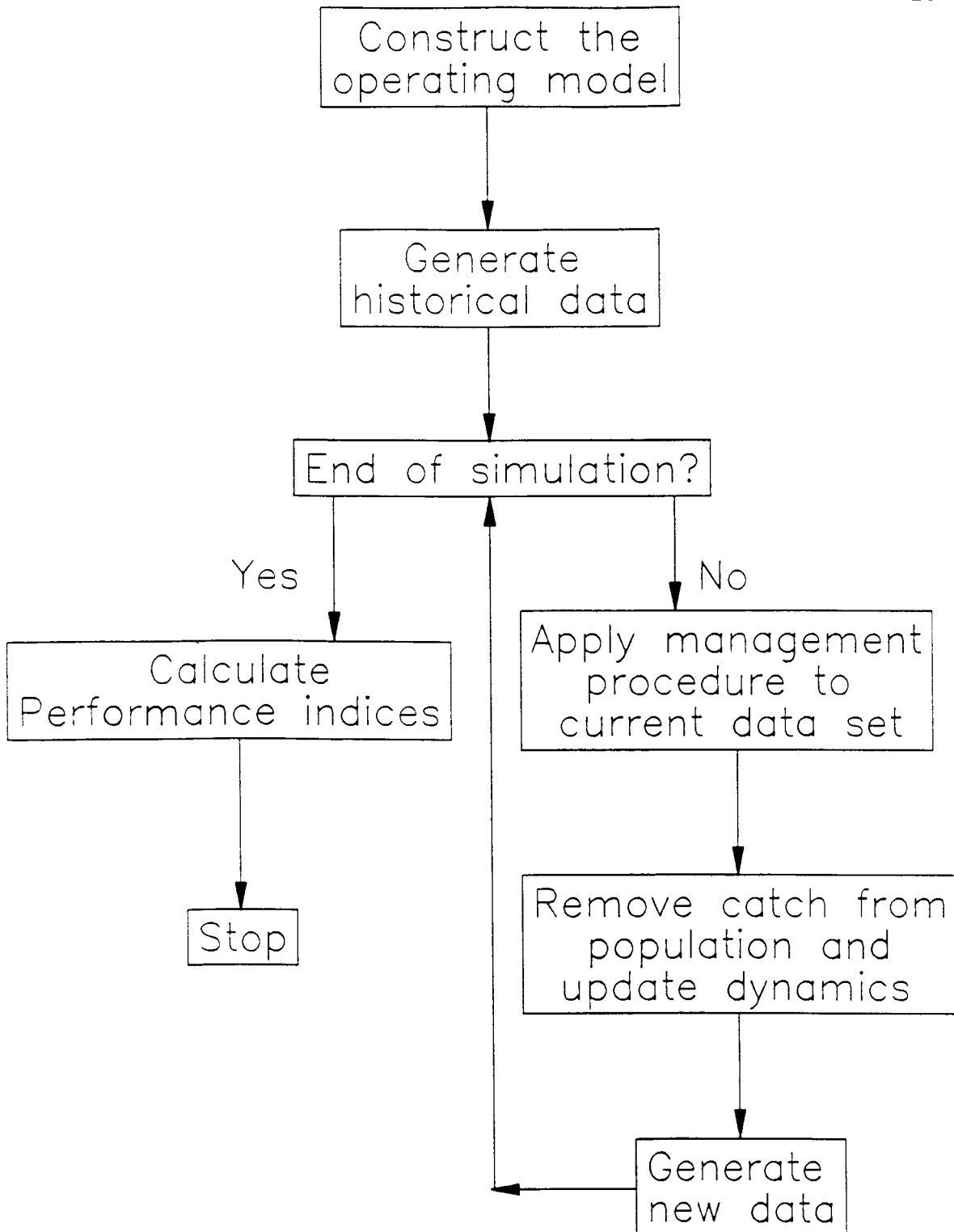


Figure 1.2 : The approach used to determine the performance of a management procedure.

Simulation testing does, however, have some disadvantages. The major problems are selecting appropriate operating models which reflect plausible alternative scenarios for the fishery, and choosing appropriate performance measures. For example, if the scenarios considered are too extreme, a management procedure might be selected which is unnecessarily conservative and will lead to wastage of potential yield.

1.6 Overview of the Cape hake section of the thesis

The Cape hake fishery off the South African coast is of considerable social and economic value to South Africa. Cape hakes (which actually comprise two species) form the bulk of the catch of the South African demersal industry, and the wholesale value of the catch of these species in 1989 exceeded 300 million Rand. This is almost 30% of the wholesale value of all marine species harvested off the South African coast. As the fisheries for hake off the South African west and south coasts are assessed separately, and the catches off the west coast are often more than double those taken off the south coast, this study has concentrated exclusively on the fishery off the South African west coast.

Scientific TAC recommendations for the South African hake fishery are presently developed from the results of a production-model based model-estimation procedure which uses catch, CPUE and survey biomass data (cognizance is also taken of the results of an *ad hoc* tuned VPA). Both of these assessment methods have been criticized in the past for reasons which are detailed in Chapters 6, 7 and 10. This thesis

examines the trade-offs achieved by a large number of candidate management procedures (including that currently used), and attempts to select the management procedure(s) which perform(s) most satisfactorily for this fishery. A large number of alternative operating models are considered in this process. Another of the objectives of this study is to identify which operating models result in the most marked deteriorations in performance. Such identification is necessary so that future management procedures for this resource can be built on attempts to overcome these problems.

Few quantitative analyses have been carried out to date to select management procedures for this hake resource. However, considerable research has been directed at determining the relative performances of alternative estimators (i.e. assessment methods) for the Cape hake resources off southern Africa. For example, Butterworth and Andrew (1987) compared the performances of three estimators by calculating the variances of their predictions, and by investigating the sensitivity of their TAC estimates to alternative minimization criteria. Punt (1988) compared the performances of a large number of alternative estimators for the Cape hake stock off northern Namibia using simulation methods. An investigation into the performance of management procedures has been carried out by Hilborn (1979). Ludwig and Walters (1985), Ludwig *et al.* (1988) Pope and Shepherd (1985), ICES (1987, 1988), and Fournier and Warburton (1989) *inter alia* have investigated the performance of different estimators. Unfortunately, as the latter studies have not been constrained by the specifics of the Cape hake resource, it is

unlikely that their results can provide anything other than some qualitative guidance for this fishery.

Chapter 2 deals with a number of aspects related to hake biology while Chapter 3 provides a historic background to the fishery for Cape hake off southern Africa. Chapter 4 details the data that are available for assessment purposes. Chapters 5 to 8 provide detailed descriptions of the model-estimation procedures considered in this study and Chapter 9 details the harvesting strategies investigated. Chapter 10 provides a number of alternative assessments of the resource, while Chapter 11 details the operating models considered. Chapters 12 to 14 provide a discussion of the results of applying the management procedures considered to the operating models, to assess performance over a 20-year period (1990-2009). Finally, Chapter 15 lists the conclusions drawn from the study.

1.7 Overview of the baleen whale section of the thesis

The current status of baleen whale stocks worldwide range from nearly extinct [the East-Greenland-Spitsbergen stock of Bowhead whales (Christensen *et al.* 1990)] to virtually unexploited [the "stock" of minke whales in Area V in the Antarctic (IWC 1990a)]. Baleen whales are characterized by extremely low productivity (ratios of MSY to the population size at which MSY is achieved of 1% or less have been suggested by some scientists). This means that any management procedure for baleen whale resources needs to be robust to uncertainties related to initial size (not that it is likely that harvesting in the near

future will be planned for any of the stocks currently at very low levels) and to productivity.

The management procedure which is developed in Chapters 16 to 19 is that co-developed by the author as part of the "Comprehensive Assessment" carried out to determine the effects of the moratorium on commercial whaling which came into effect in 1986. The features of and parameter values for this procedure have been selected so that satisfactory performance is achieved for virtually all of the scenarios considered.

Chapter 16 provides a more detailed background to the problem. The development of a "revised" whale management procedure centred initially on the construction of a procedure for situations in which stock identity is certain. A description and discussion of the algorithms developed for this purpose and the resultant performance statistics are given in Chapter 17. The details of the development of procedures for handling situations in which stock identity is not certain are provided in Chapter 18. A summary discussion of the results, as well as some general conclusions are given in Chapter 19.

CHAPTER 2 - THE BIOLOGY OF CAPE HAKE OFF THE SOUTH AFRICAN WEST COAST

Three species of hake (*Merluccius capensis*, *M. paradoxus* and *M. polli*) occur in the southeast Atlantic. *M. polli* is only found off northern Namibia and Angola and is thus not considered in this thesis. The other two species are collectively called the Cape hakes. *M. paradoxus* is found in deeper water than *M. capensis*, but because there is a size graduation with depth, larger *M. capensis* cohabit with smaller *M. paradoxus*. A logistic model is fitted to the fraction of mature fish by length. By converting the lengths at different levels of maturity into ages, males are found to mature earlier than females, although there is little difference in lengths at different levels of sexual maturity for the two species. Growth is found to be nearly linear for females, but the growth curve for males shows more curvature. Females grow slightly faster than males. Estimates of the annual consumption by and daily ration for the two species are estimated from stomach content data. Cape hake are opportunistic feeders, juveniles feeding intensively on crustaceans with the diet becoming increasingly piscivorous with age. The estimates of annual consumption by hake are found to be sensitive to the treatment of everted stomachs, and to vary considerably depending upon which survey is used to provide the estimates of numbers-at-length. The estimates are also often imprecisely determined because of the effects of small sample sizes and the opportunistic nature of feeding. Both species are cannibalistic, and *M. capensis* feeds on *M. paradoxus* (but not vice versa because of distributional effects). Point estimates of hake consumption by hake are substantial, and suggest that model-estimation procedures which assume a rate of natural mortality which is independent of age and year may be inappropriate. One of the aims of this study is therefore to determine the extent of performance degradation which occurs if conventional single-species management procedures are applied to a fishery in which quantitatively substantial multispecies interactions occur.

2.1. Introduction.

Despite the considerable commercial importance of the Cape hake fishery off the South African west coast (ICSEAF Division 1.6), relatively little research of a biological nature has been carried out on the hakes in this area. Many of the biological studies that have been directed at hakes off southern Africa have been conducted off Namibia (ICSEAF Divisions 1.3+1.4 and 1.5) and off the South African south coast (ICSEAF Divisions 2.1+2.2). However, the results of these studies are not necessarily applicable to the hake resource off the west coast

because there appear to be biological differences between the hake stocks in different areas. For example, Bohl *et al.* (1971) note differences in ages-at-maturity, Chlapowski (1975) notes differences in horizontal distribution and Cushing (1968) notes differences in vertical distribution.

This chapter provides an overview of the biology of the Cape hake resource off the South African west coast, and presents the results of the analysis of some of the data collected during recent direct surveys of this resource. A description of the methodology used during these surveys, as well as details of the data collected during the cruises are given in Appendix 2.A

2.2. Genealogy

Three hake species [*Merluccius capensis* Castelnau (shallow water hake), *Merluccius paradoxus* Franca (deep water hake) and *Merluccius polli* Cadenat (Benguela hake)] are caught in the hake fishery in the southeast Atlantic. *M. polli* is only caught off northern Namibia and Angola, and so will not be considered further in this study. The other two species are collectively called Cape hake. All three species are related to the cod-like fishes of the Order Anacanthini and have behaviour patterns similar to those of the European hake (*M. merluccius*). *M. paradoxus* and *M. capensis* were originally regarded as sub-species of *M. merluccius* [e.g. Franca (1954, 1962); Botha (1970)], but the two are now regarded as separate species.

Although very similar, differences in gill structures, otolith morphology, pectoral fin length, number of vertebrae, relative eye

size, colour of the anal fin and body shape for *M. capensis* and *M. paradoxus* have been noted by Van Eck (1969) and Botha (1970, 1971, 1980). The number of vertebrae has been shown by Van Eck (op. cit.) to be significantly different [49.9 (s.e. 0.7) and 55.2 (s.e. 0.69) for *M. capensis* and *M. paradoxus* respectively]. Franca (1954) also noted that the number of vertebrae differed between the two species. Differences in the gill tubercles have also been noted [Bentz (1976), Botha (op. cit.)], *M. paradoxus* having a dark pigment at the top of its tubercles which is not present in *M. capensis*, the latter normally having uniformly white tubercles. The tubercles of *M. paradoxus* also appear to be longer than those of *M. capensis*. The taxonomic status, distribution and biology of the genus *Merluccius* are reviewed by Botha (op. cit.) and by Inada (1981).

2.3. Distribution

Cape hake are caught in shelf waters off the southern African coast from Bahia de Farto in Angola (12°S) (Botha 1980) to as far east as northern Natal (see Figure 2.1). *M. capensis* lives closer inshore, apparently preferring the area where the shelf is widest and the slope steepest. On the other hand, *M. paradoxus* appears to prefer the area where the shelf is narrower and less sloped, but because there is a size graduation by depth - larger fish living deeper than small ones (Botha op. cit.) - large *M. capensis* cohabit with, and feed extensively on, small *M. paradoxus* (Payne et al. 1987). Kawahara and Nagai (1980) suggest that there may be limited horizontal migrations during the year off Namibia at least. Table 2.1 provides a rough classification of the

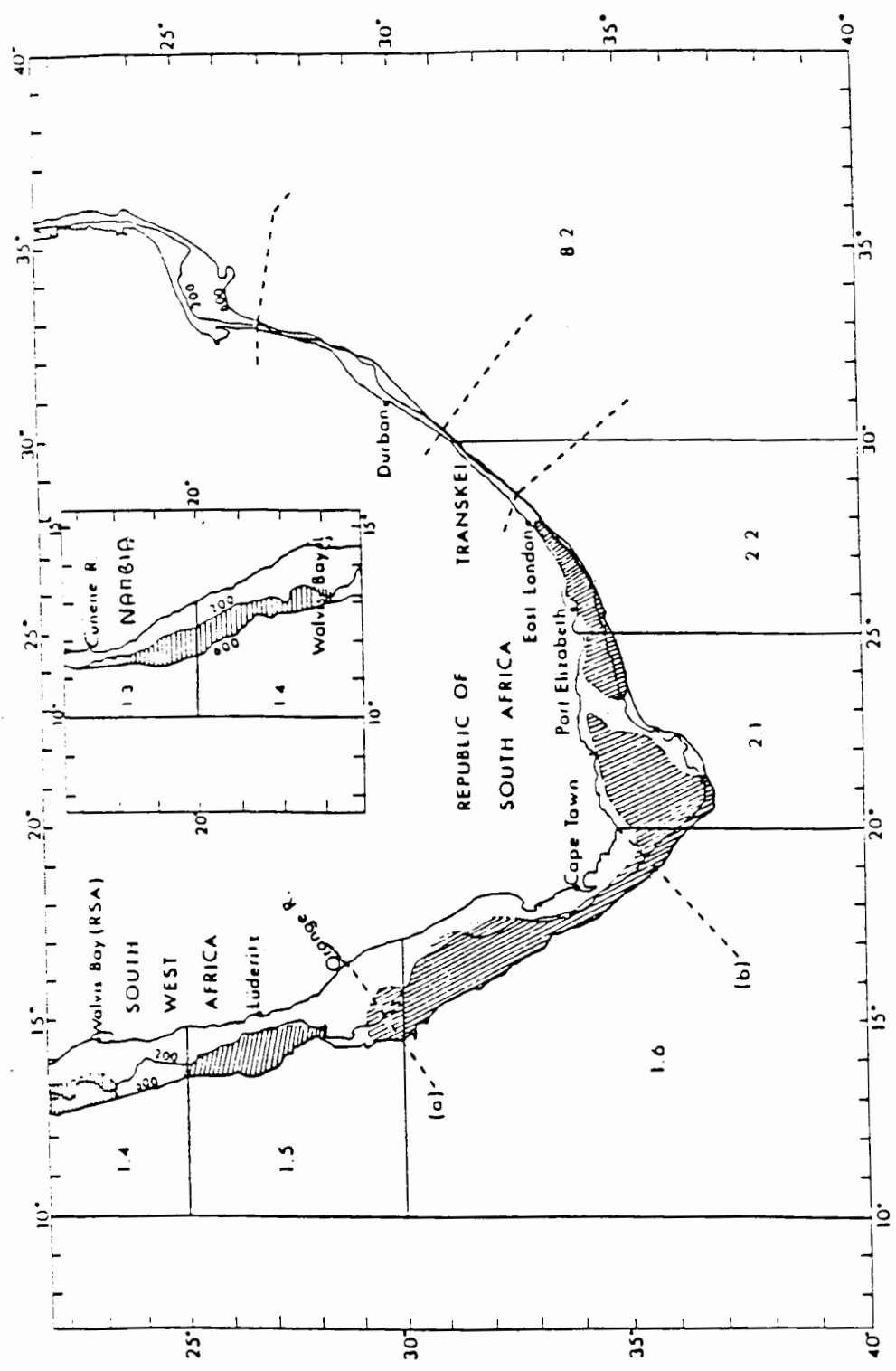


Figure 2.1: The major Cape hake trawling grounds and the ICSEAF Division boundaries off southern Africa (based on unpublished SFRI and ICSEAF statistics). The boundaries marked (a) and (b) refer respectively to the boundaries of Division 1.6 and of the adjoining Divisions which have been specified for domestic (South African) purposes [after Andrew and Butterworth (1987a)].

ICSEAF Divisions (see Figure 2.1) according to the dominant hake species caught in each.

The major proportion of the hake resource off the South African west coast is currently found at depths of between 200 and 400m (Payne *et al.* 1988), although a large number of small *M. capensis* live inshore of this. Botha (1980) notes that *M. capensis* of length less than 40cm are generally found in water up to 220m deep, with the larger individuals at depths of up to 440m. The maximum density of this species occurs at 150m (Botha *op. cit.*). *M. paradoxus* is found south of 22°S at depths of between 150 and 920m along the continental shelf (Botha *op. cit.*). The maximum density of this species occurs at 330m (Botha *op. cit.*). Between the depths of 150-440m (Botha *op. cit.*) and 200-400m (Inada 1981) juveniles of *M. paradoxus* coexist with the larger individuals of *M. capensis* [Botha (*op. cit.*), Inada (*op. cit.*)]. This phenomenon is uncommon among the adults of the two species (Badenhorst 1984), although it is possible that it may occur in some areas because of the narrowness of the continental shelf.

Botha (1980) notes that for the study area at least, fishing mortality should be lower on juvenile *M. capensis* and older *M. paradoxus* than on adult *M. capensis* and younger *M. paradoxus*, because the former pair occur in areas which are seldom subject to heavy exploitation.

2.4 Growth

2.4.1 Length-at-age

Previous attempts at fitting growth equations to lake length-at-age data [e.g. Botha (1971), Draganik (1976) and Leslie (1986)] have been performed using fitting procedures, such as Ford-Walford plots (Ricker 1958), now considered to be suboptimal [see, for example, Vaughan and Kanciruk (1982) and Sundberg (1984)]. The approaches used previously were undesirable because, in general:

- a) they considered the mean lengths-at-age without explicitly considering the distribution of data points about these means,
- b) they treated the mean length-at-age for each age as having equal weight when fitting the growth equation, even though there may have been substantially more data points for some ages than for others,
- c) they had implicit and often unrealistic error structures, and
- d) standard goodness-of-fit tests [e.g. the runs test - Draper and Smith (1966)] could not be applied to their results.

All four of these difficulties can be overcome if the selected growth equation is fitted to the raw length-at-age data set using non-linear least squares. For this approach, the age-length sample must be a random sample from the population under consideration. This assumption is often not justified in a real situation, and it is necessary to correct for sampling biases in these cases. The process of fitting a growth equation thus involves minimizing the quantity:

$$SS = \sum_i [f\{\ell_i\} - f\{\hat{\ell}_i\}]^2 \quad (2.1)$$

where ℓ_i is the observed length of the i 'th fish,
 $\hat{\ell}_i$ is the model-predicted length of the i 'th fish, and
 f is some transformation function.

The transformation function f is selected so that the statistical properties of randomness and homoscedascity of residuals are achieved [Draper and Smith (1966), Butterworth *et al.* (1989)]. Minimization of function (2.1) circumvents the four problems listed because it treats each data point separately. This means that ages with a large number of data points are accorded high weight and the distributions of lengths about the means are taken account of directly. A further positive feature of minimizing function (2.1) instead of making use of an *ad hoc* method is that standard approaches to variance estimation (e.g. the jack-knife and bootstrap [Efron 1982, 1985]) can be applied.

Hake length-at-age has traditionally been modelled by the von Bertalanffy growth equation:

$$\ell_a = \ell_\infty (1 - e^{-\kappa(a - t_0)}) \quad (2.2)$$

where ℓ_a is the mean length of a fish aged a years,
 ℓ_∞ is the asymptotic length,
 κ is a growth rate parameter, and
 t_0 is the 'age' at zero length.

However, more general growth equations, such as the Schnute growth equation (Schnute 1981), or more parsimonious parameterizations such as

a linear growth equation, could be considered instead. The form of the Schnute growth equation depends on the values of its shape parameters a^S and b^S . For $a^S > 0$ and $b^S > 0$, the form of this growth equation is:

$$l_a = \left[l_1 + (l_2 - l_1) \frac{1 - e^{-a^S(a - a_1)}}{1 - e^{-a^S(a_2 - a_1)}} \right]^{1/b^S} \quad (2.3)$$

where l_a is the mean length of a fish aged a years,
 a^S and b^S are the Schnute shape parameters,
 l_1 is the estimated length of a fish aged a_1 years, and
 l_2 is the estimated length of a fish aged a_2 years.

Both the von Bertalanffy and linear growth equations are special cases of the Schnute growth equation. The von Bertalanffy growth equation corresponds to $a^S > 0$ and $b^S = 1$ and the Linear growth equation corresponds to $b^S = 1$ and the limit as $a^S \rightarrow 0$. Schnute (1981) shows how to calculate the values of the von Bertalanffy parameters, l_∞ and t_0 , from values for the Schnute growth equation parameters, a^S , b^S , l_1 and l_2 (the von Bertalanffy parameter κ equals the Schnute parameter a^S). Standard statistical techniques, such as the likelihood ratio test (Draper and Smith 1966), can be used to compare fits of these three growth equations to an age-length data set, because they are nested.

The age-length data obtained from the research cruises cannot be used in conjunction with the Equation (2.1) approach to obtain growth equation parameter estimates. This is because the age is determined of approximately the same number of animals per 1cm length-class. Use of such data without correcting for the sampling procedure would result in

positively biased estimates of mean length-at-age, because larger (older) fish would be over-represented in the age-length samples.

The estimates of the growth equation parameter values are obtained instead by maximizing the likelihood function (L):

$$L = K' \prod_{a=0}^{12} \prod_{\ell=1}^{110} [p(\hat{a}|\ell)]^{A_{a,\ell}} \quad (2.4)$$

$$p(\hat{a}|\ell) = \frac{p(\hat{\ell}|a) p(a)}{p(\ell)} \quad (2.5)$$

where $A_{a,\ell}$ is the number of fish in length-class ℓ whose age is determined to be a years old in the actual data set,

K' is a combinatorial factor which is not a function of the model parameters,

$\hat{\ell}_a$ is the model-estimate of the mean length of a fish aged a years,

$p(\ell)$ is the probability that a randomly selected fish is in length-class ℓ ,

$p(\hat{\ell}|a)$ is an estimate of the probability that a fish aged a years is in length-class ℓ :

$$p(\hat{\ell}|a) = \int \frac{1}{\sqrt{2\pi} \sigma} e^{-\frac{(\ell - \hat{\ell}_a)^2}{(2\sigma^2)}} d\ell \quad (2.6)$$

where the integral is taken over length-class ℓ ,

$p(a)$ is the probability that a randomly selected fish is aged a years,

$p(\hat{a}|\ell)$ is an estimate of the probability that a fish in length-

class l is aged a years, and
 σ is the standard deviation of length-at-age (assumed to be independent of age).

The form of Equation (2.4) results from the assumptions that the growth rate has not changed over the period during which the samples were collected, that the estimates of the length composition are not subject to any error (except the bias resulting from differential selectivity of length-classes by the trawl gear) and that fish in each length-class are selected multinomially. In addition to the growth equation parameters (e.g. κ , t_0 and l_∞ for the von Bertalanffy growth equation), it is necessary to estimate the standard deviation of the length-at-ages (σ) in the non-linear maximization process. Equation (2.4) is such that length-classes for which large samples of age-at-length are available are given greater weight than length-classes for which only a few fish have been sampled. The approach of Equation (2.4) has all the advantages associated with that of Equation (2.1), as each data point is treated separately and because Equation (2.4) is derived on the basis of a specific (and defensible) choice of error structure. Although Equation (2.4) could have been extended to allow for the variability in the estimates of length composition data, this has not been considered here because the length composition estimates are obtained from much larger samples than the age-at-length distributions.

The age of a fish was determined by examining the number of concentric bands on the otolith. Otoliths have been selected for age determination for hake because they are large, easily accessible (Botha 1970) and relatively easy to read. Botha (1971) reports that ages could not be

assigned to only 7% of the otoliths in his collection. Of these, 3% could not be read at all and the remaining 4% could be read, but yielded conflicting ages (Botha 1971). An opaque zone is deposited on the otolith edge from December to January and a hyaline zone from August to November (Botha 1971). From this evidence, Botha concluded that the number of bands on the otolith could be used to determine the age of hake.

Of the otoliths sampled during 1989, the age estimates provided by the first two age-readers disagreed in 20% of cases. In all of these cases, the age estimates only differed by one year. After a third reader investigated the otoliths in question, agreement was reached in all cases. The highest percentages of fish for which initial agreement of age estimate could not be reached came from age-classes 3 and 4.

Parameter estimates from fits of the Schnute, von Bertalanffy and linear growth equations to the age-length data for 1986 to 1989 for *M. capensis* and *M. paradoxus* separately and for both species combined were obtained by maximizing Equation (2.4). The estimates for the Schnute and von Bertalanffy fits are given in Table 2.2; the results for the linear model are omitted from this Table because of its poor fits to the data. Results are presented for males, females and both sexes combined. The bootstrap (see Section 10.2 for details) has been used to provide the estimates of precision given in Table 2.2. In order for the parameter estimates to be comparable, they have all been presented in the form of Schnute parameters. If l_{∞} and t_0 are defined for the estimated values of a^S and b^S (they need not be), they are given as well.

In a number of cases, the fit of the Schnute growth equation provides a statistically superior fit to the data (at the 5% level - Table 2.2). In most cases, however, the fit of the von Bertalanffy growth equation is not significantly poorer than that of the Schnute growth equation, and the differences in the estimates obtained from each growth equation of mean lengths-at-age for the age ranges sampled are usually quite small - compare the estimates of l_1 and l_2 for each species/sex combination given in Table 2.2.

Estimated von Bertalanffy growth equations for males and females of each species and of both species combined are provided in Figure 2.2. It seems that, for the age range considered, female growth is almost linear (particularly for *M. capensis*) whereas the growth equation for males exhibits some curvature over part of its range. Although the estimates of length-at-age over the age range sampled are consistently predicted by both growth equations, the estimates of l_∞ and t_0 differ quite markedly (Table 2.2). The parameter t_0 does not have to exist for the Schnute fits because it is possible for this growth equation to select parameter values which imply that length increases from a zero asymptote instead of passing through the age-axis. Attempts to use the (conditioned) parametric bootstrap to estimate the variance of the estimates of l_∞ and t_0 failed, because some of the bootstrap estimates of l_∞ were infinite and those of t_0 were negative infinity. This indicates that the precision of these quantities is very poor, which is not surprising because both l_∞ and t_0 are extrapolations from the data.

Botha (1980) concluded that females of both species grow faster than the males because the annual growth increment for females over the

Figure 2.2(a)

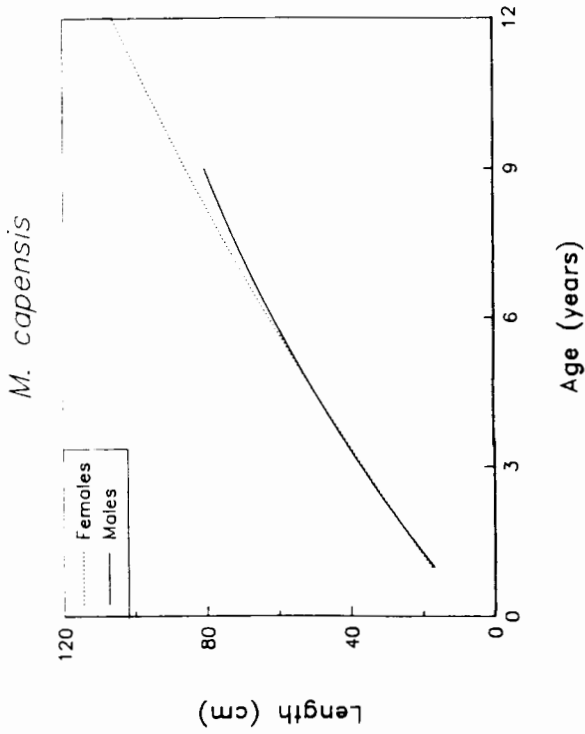


Figure 2.2(b)

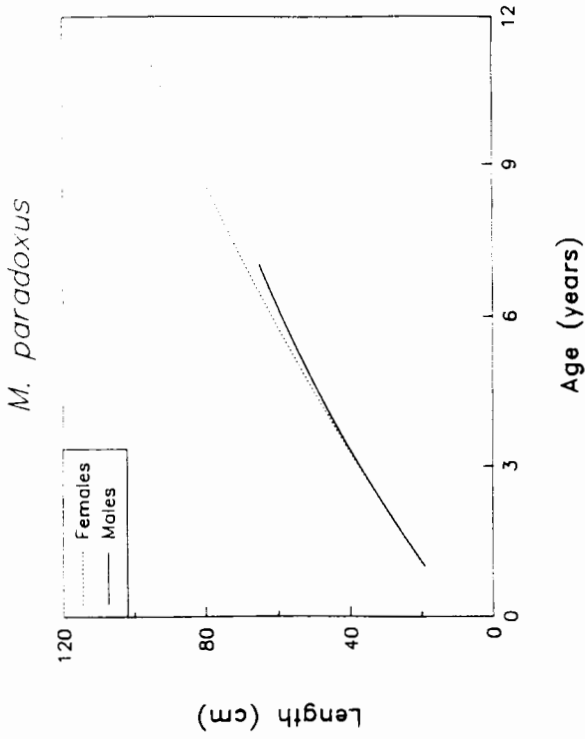


Figure 2.2(c)

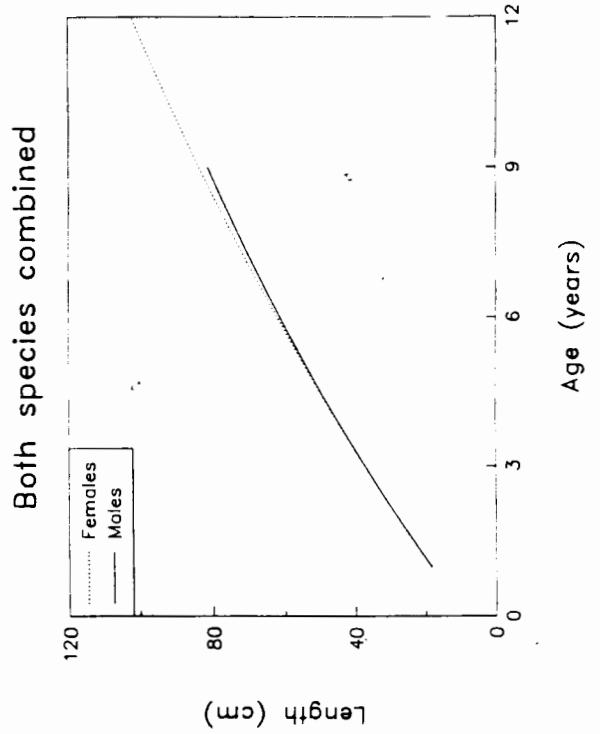


Figure 2.2 : Estimated mean lengths-at-age for males and females obtained from the fits of the von Bertalanffy growth equation: (a) *M. capensis*, (b) *M. paradoxus*, and (c) both species combined.

range of ages for which the data for the sexes overlap is larger than that for males. His estimates of the parameters of the von Bertalanffy growth equation suggest this quite strongly [see Figures 6 and 7 of Botha (op. cit.)]. Although the results of Table 2.2 exhibit this feature, the differences between length-at-age for males and females are not significant at the 5% level. Therefore, although the results support Botha's (op. cit.) hypothesis that females grow faster than males, they indicate that the magnitude of the effect is less than suggested by Botha (op. cit.). The phenomenon that females grow faster than males has been shown for a number of hake species. For example, Hickling (1933) demonstrated that female European hake *M. merluccius* grow faster than males from the age of three years and Dark (1975) found that female Pacific hake *M. productus* grow faster than the males.

2.4.2 Mass-at-length

Mass as a function of length is well represented by the relationship:

$$w = a_w(\ell)^{b_w} \quad (2.8)$$

where w is the mass of a fish of length ℓ .

The values of the parameters a_w and b_w , obtained by fitting Equation (2.8) to the length-mass data collected during the research cruises from 1986 to 1989 by means of linear regression, are given in Table 2.3. Estimates of the standard errors of $\ln a_w$ and b_w , as well as the values of the correlation coefficient (r) are given in Table 2.3. Linear regression of log-transformed lengths and masses was selected because the variance of mass-at-length appears to increase with length

and because it is more straightforward to obtain the standard errors of $\ln a_w$ and b_w if model (2.8) is linearized. The fits to the data are exceptionally good and are all highly statistically significant. Except for the largest fish, the curves pass virtually exactly through the mean masses-at-length.

Although the fits of Equation (2.8) to data for males and females separately are generally significantly different (Table 2.3), these differences do not appear to be substantial. Therefore, it would seem defensible to use a single mass-length relationship for both sexes for modelling purposes.

2.5 Maturation

Some spawning takes place virtually all year round (Botha 1980) and it is not impossible that a female may spawn more than once a year (Payne 1989). The spawning season for hake on South Africa's west coast is bimodal, with a major peak in November/December and a less intensive period in February/March dominated by *M. paradoxus* (Botha op. cit.). Although adults of the two species do co-exist, recent work [Becker *et al.* (1988), Grant *et al.* (1988)] shows that the two species of hake are genetically distinct and that the possibility of hybridization is remote.

Although each fish sampled was classified according to a maturity schedule (Payne *et al.* 1987), this study has only considered whether or not a fish is mature (defined here as the gonads showing evidence for egg/sperm production) or immature. Any attempts to model the data by maturity stage would have yielded anomalous results owing to the lack

of actively spawning fish in the samples. Movement to mid water to spawn may explain why few spawning females are found in commercial catches (Botha 1973).

Estimates of total length at various levels of sexual maturity are given in Table 2.4. These estimates have been obtained by fitting a logistic model to the maturation data collected during research cruises between 1986 and 1989.

$$\hat{m}_l = [1 + \exp(-(\bar{l} - l_m)/\delta_m)]^{-1} \quad (2.9)$$

where \hat{m}_l is the model-estimate of the fraction of fish in length-class l which are mature,

\bar{l} is the midpoint of length-class l ,

l_m is the length-at-50%-maturity, and

δ_m is a parameter which determines the width of the maturation ogive.

The model was fitted by minimizing the function:

$$SS = \sum_l n_l (m_l - \hat{m}_l)^2 / \{\hat{m}_l (1 - \hat{m}_l)\} \quad (2.10)$$

where n_l is the number of fish in length-class l for which maturity has been determined (each length-class has been taken to be 1cm wide), and

m_l is the observed fraction of fish in length-class l which are mature.

The term $n_l / \{\hat{m}_l (1 - \hat{m}_l)\}$ is the inverse of the variance of \hat{m}_l under the

assumption that m_{ℓ} is binomially distributed (not an unreasonable assumption, given that a fish is either mature or immature). Minimization criterion (2.10) thus assigns greater weight to length-classes for which more fish have been analyzed for maturation state and for which the variability of the estimates of the maturation fraction should be precisely determined. It was decided to minimize absolute deviations rather than differences of say logit- or log-transformed fractions for two reasons. First, most of the fits did not show evidence that the selection of minimization criteria (2.10) had led to lack of randomness. Second, selection of either the logit- or log-transformation would have necessitated *ad hoc* specifications to deal with zero (and, for the logit-transformation, unity) fractions, because these would have resulted in attempts to divide by zero and take the logarithm of zero. In order to minimize Equation (2.10), an iterative re-weighting approach was used. In general, convergence of this procedure occurred after only a few iterations (sometimes as few as two).

Estimates of the precision of the model predictions, computed by means of the (conditioned) parametric bootstrap variance-estimation procedure, are also provided in Table 2.4.

Fits of model (2.9) were performed for:

- a) each cruise for each species,
- b) all summer cruises combined and all winter cruises combined for each species,
- c) all cruises combined for each species, and

d) all data combined.

As the sample sizes for individual cruises are relatively small (~200 fish per species per sex per cruise), there is considerable variability in parameter estimates when model (2.9) is fitted to data for individual cruises. Hence, the discussion will focus on the results for the pooled data.

Results are presented for fits to the data for males and females separately and for males and females combined in Table 2.4. The fits of model (2.9) to the maturation data for males, females, and males and females combined are given in Figure 2.3 for both species combined. Whereas some of the fits may fail a runs test (Draper and Smith 1966), most of the fits appear reasonably satisfactory, given the binary nature of the data.

The surprisingly high precision of the estimates of length at various levels of sexual maturity evident in Table 2.4 (C.V.s < 5% in most cases) may be a consequence of negative bias in the variance-estimation procedure used. Although tests of the ability of the (conditioned) parametric bootstrap to estimate variances for quantities estimated by logistic models have yet to be performed, Punt (1988) has shown that this variance-estimation procedure provides negatively biased estimates of standard error for a number of dynamic production model-estimation procedures.

Ignoring the possible bias in the estimates of precision, the estimated C.V.s conform to what would have been expected, viz. the estimates obtained from small data sets have larger C.V.s than those obtained

Figure 2.3(a)

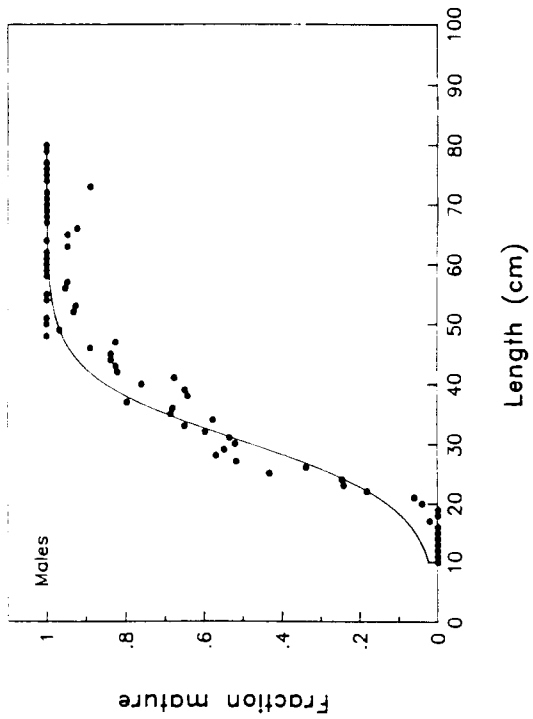


Figure 2.3(b)

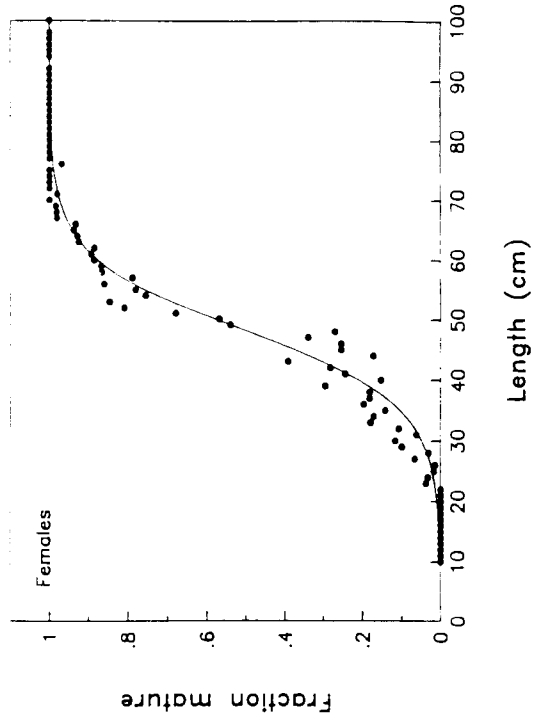


Figure 2.3(c)

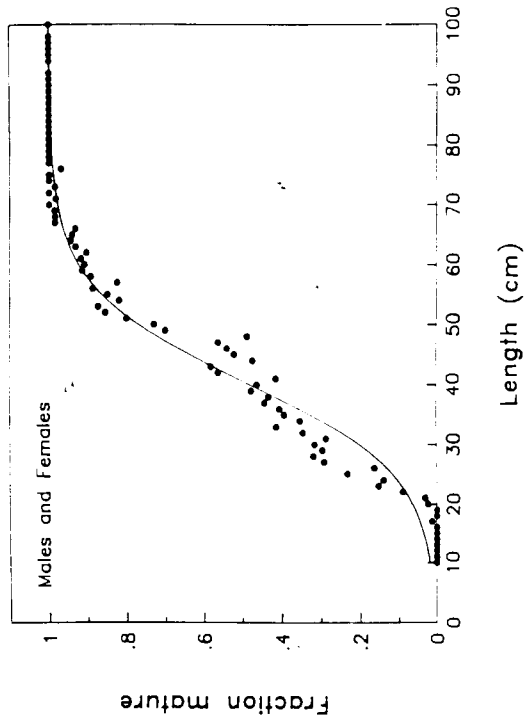


Figure 2.3 : Fits of model (2.9) to the maturation data for both species combined: (a) males, (b) females, and (c) both sexes combined.

from larger data sets. The C.V.s for males and females exhibit different trends with increasing maturity. For males, the C.V.s increase while for females, there is an intermediate length at which the lowest C.V. is achieved. The C.V.s for the case in which the data for males and females are pooled are higher than would have been expected, given the larger sample sizes. However, this is probably the result of pooling data which suggest different trends. The estimates of the parameter δ_m are generally less precisely determined than the estimates of total lengths at various levels of sexual maturity. This is not altogether surprising. The estimates of the length-at-50%-maturity can be bounded directly from the data, whereas the estimate of δ_m cannot.

There appears to be a substantial difference between the winter and summer estimates of the length-at-50%-maturity for female *M. capensis* [Table 2.4(b) rows 'Cap Win' and 'Cap Sum' (42cm and 50cm respectively)], although the estimates of δ_m are quite similar (0.139 and 0.115 respectively). There are two possible explanations for this phenomenon. First, the fit of model (2.9) to the data for female *M. capensis* for the summer cruises does not appear to be particularly good (even though the C.V.s for this fit are reasonably small); therefore the results of this fit may not be reliable. However, the more plausible explanation for the difference is that, for some cruises, fish were classified as active (i.e. gonads contained developing eggs) or inactive (no egg development). No attempt was made to distinguish between mature fish with inactive gonads (i.e. post-spawning fish) and immature fish. During summer, when spawning is most intensive, there will be a relatively high fraction of fish which have recently spawned.

Hence it would be expected that a number of these post-spawning fish may be misclassified as immature. This effect is likely to be most marked for *M. capensis*, which does not spawn intensively in February/March (Botha 1986).

Table 2.4(a) suggests that 25% of males are mature by length 25.1cm and 75% by a length of 37.7cm. These results pertain to the fits for both species combined - the results for each species separately are not markedly different. Attempts to infer at which length 5% or 95% of males are mature would be inappropriate, because such estimates are sometimes very imprecisely determined. Further, the fitted curves do not fit the data particularly well at the 5% and 95% levels [see, for example, Figure 2.3(a)]. Inferences from the estimates of lengths at these levels could be somewhat misleading. The estimates of the lengths at 25%- and 75%-maturity for males are considerably lower than the corresponding values reported by Botha (1986) [see Table 2.5]. However, the estimates of male 50% maturity provided by Botha (1986) [37cm] and by this study [31.4cm] are both substantially lower than that of Jones (1974a) [45cm]. The 34cm estimate of male length-at-50%-maturity obtained by Van Eck (1969) lies between the estimate obtained in this study and that obtained by Botha (op. cit.).

The situation for females is slightly different. There is less than a 3cm difference between the estimates of the lengths-at-25%-maturity and length-at-75%-maturity for both species combined, obtained during this study and those documented by Botha (1986). In this case, the length-at-50%-maturity obtained by Jones (1974a) [42cm], is less than the 46cm obtained by Botha (op. cit.) and the 47.5cm of this study. The

corresponding estimate provided by Van Eck (1969) is at 60cm, substantially higher.

There do not appear to be substantial differences between the lengths-at-sexual-maturity for *M. capensis* and *M. paradoxus* (except at the 5 and 95% levels) [compare the rows "Cap all" and "Par all" in Tables 2.4(a) and 2.4(b)] nor between lengths-at-age (see Figure 2.2). The former result is consistent with the observation of Botha (1986).

The results suggest strongly that the lengths at various levels of sexual maturity differ between sexes [Figure 2.3]. As the growth equations for males and females do not differ too markedly, this means that males mature at an earlier age than females. The ages at various levels of sexual maturity are given in Table 2.6. These estimates were obtained by converting the estimates of lengths at the selected levels of maturity into ages, using the Schnute growth equations given in Table 2.2. The lower tails of the distributions reported in Table 2.6 are unrealistically low (particularly so for males). These extremely low ages are attributable to the method used to convert from length to age. Those 15cm fish estimated to be mature are likely to be above the average age for this length. From Table 2.6, it can be concluded that the age-at-50%-maturity for males is approximately 2-2.5 years and for females it is approximately 4.25 years.

Although the fits of model (2.9) to the male and female data sets separately result in markedly different parameter estimates [see Tables 2.4(a) and (b)], it is not possible to use standard statistical methods (such as the log-likelihood ratio test) to determine whether these

differences are statistically significant. This is because the models for males and females are not nested because of the presence in Equation (2.10) of the n_g factor. This factor results in the error structures of the two estimators being different. It is, however, possible to compare the estimates in pairwise fashion by assuming that each estimate is independent and normally distributed. These comparisons indicate that differences between lengths at various levels of sexual maturity are significant at the 5% level.

As the parameter estimates for males and females are so different, there does not seem much justification for pooling the data for males and females. However, pooling was carried out because most assessment techniques require a single age-at-50%-maturity as they do not consider each sex separately. Pooled results are given in Tables 2.4(c) and 2.6(c) and Figure 2.3(c).

2.5 Feeding ecology

Many studies related to the feeding of the Cape hakes have been carried out off South Africa and Namibia [e.g. Chlapowski (1977), Assorov and Kalinina (1979), Prenski (1980), Krzeptowski (1982), Konchina (1987), Andronov (1987) and Roel and Macpherson (1988) off Namibia; Rattray (1947), Davies (1949), Botha (1980) and Payne *et al.* (1987) off the South African west coast; and Hecht (1976), Smale (1984) and Payne (1986) off the South African south coast]. Therefore, in principle at least, there would seem to be a large stomach content data base available for the Cape hakes.

However, many of these studies utilized very small sample sizes [e.g. 45 fish - Konchina (op. cit.)] and were very localized [e.g. Botha (op. cit.), Davies (op. cit.) and Rattray (op. cit.)]. In any case, samples collected in areas other than off the South African west coast may be unrepresentative of the diet of Cape hake in that region, because the Cape hakes are opportunistic feeders. Therefore, the relative importance of the various food items in the diet is likely to differ from area to area because of differences in local abundance of prey species. Furthermore, diet composition and annual consumption of a particular species, changes over time as a result of differences in the relative abundance of prey species.

It is thus inappropriate (unless statistical analyses indicate otherwise) to pool stomach content data over long time periods or wide geographic areas. Note that there is a bias-variance trade-off involved here - increasing the stomach content sample sizes by pooling over long time periods or wide geographic areas might increase the precision with which quantities are estimated, but would also increase the bias of these estimates. It is important to select an appropriate stratification of the data to achieve a reasonable trade-off between bias and variance.

Only data from the July 1988, January 1989, July 1989 and January 1990 cruises have been considered here, because the data for the surveys prior to July 1988 were not recorded in a consistent manner. Preliminary analyses of the stomach content data collected during the cruises carried out between 1983 and 1986 are given in Payne *et al.*

(1987). In the following, the term "the predators" will be used to refer to the two hake species.

2.5.1 Consumption model

A plausible model is that the consumption of prey species s by predators in length-class ℓ , U_{ℓ}^s , is given by:

$$U_{\ell}^s = \frac{365 N_{\ell} M_{\ell}^s}{t_{\ell}^s N_{\ell}'} = 365 N_{\ell} D(\ell, s) \quad (2.11)$$

where $D(\ell, s)$ is the daily ration of prey species s for a predator in length-class ℓ ,

M_{ℓ}^s is the total ingested mass of prey species s in the stomachs analyzed of predators in length-class ℓ ,

N_{ℓ} is the average number of predators in length-class ℓ during the year

N_{ℓ}' is the number of stomachs analyzed, including those found empty (i.e. the number of stomachs corresponding to 1_{ℓ}^s), and

t_{ℓ}^s is the evacuation time of prey species s from the stomachs of predators in length-class ℓ in days.

An estimate of the total consumption of prey species s by hake is then given by the formula:

$$\sum_{\ell} U_{\ell}^s \quad (2.12)$$

Application of Equations (2.11) and (2.12) to obtain consumption estimates is not straightforward, however, because a number of the requisite quantities are not known at all accurately or precisely.

There is considerable controversy as to an appropriate model for evacuation time (Brownley 1989) since evacuation time is affected by meal size (Brownley *op. cit.*), temperature and predator length (Jones 1974b). No estimates of t_{\emptyset}^s for Cape hake in South African waters have been published [Payne *et al.* (1987) identify the estimation of evacuation times as a research priority]. The values for the evacuation time used here are those used by Payne *et al.* (*op. cit.*), i.e.: fish - 3 days, cephalopods - 1.5 days, and crustaceans - 1 day. The values for crustaceans and fish were obtained by Prenske (1980) using data collected in Namibia, while the value for cephalopods is an educated guess based on the assumption that it should take less time to digest a cephalopod than to digest a fish but more time than to digest a crustacean. None of these values can be regarded as being particularly accurate or precise, but in the absence of any other data, they have been used.

The evacuation times used differ between the three groups of prey species, although it has been assumed that the evacuation time for all the animals within each of these groups is the same and is independent of length, temperature and meal size. This must lead to some further bias in estimates of annual consumption and daily ration. Nevertheless, although the assumption that the evacuation time is constant is crude and inaccurate, it is a standard assumption made when estimating daily ration [e.g. Durbin *et al.* (1983); Dwyer *et al.* (1987)].

It is possible, in principle, to convert the stomach content data from observed mass (i.e. the mass of the stomach contents) to ingested mass (i.e. the mass of the stomach contents when they were originally consumed). This is achieved by using prey length-mass relationships and/or relationships between the size of "hard parts" (e.g. otoliths for teleost fishes) and mass. Unfortunately, total lengths and/or sizes of "hard parts" are not available for all prey items, and the requisite relationships are not available for most prey species. Therefore, this approach cannot be used at the current time. It was assumed that, on average, the stomach contents reflect material half-way through the digestion cycle and that digestion is a linear process. Thus, the total ingested mass is estimated by multiplying the observed mass of stomach contents by two. The assumptions that the stomach contents reflect the state of the stomach half-way through the digestion cycle and that the prey is half digested at this time are somewhat questionable, because Jones (1974b) found evacuation to be a non-linear function of time. It is not possible to use Jones' (op. cit.) evacuation model directly because of the lack of requisite data on prey lengths. However, the assumption of a linear evacuation process seems the best possible under the current circumstances.

The number of stomachs analyzed would seem to be an easily determined quantity. Unfortunately, this is not the case because of the presence of everted stomachs (the proportion of which increases markedly with the size of the fish, particularly for *M. paradoxus* (see Table 2.7 and Payne et al. 1987)). Reasons postulated to explain why Cape hake evert their stomachs include depth, gut fullness, gonad size,

depressurization and shock (Payne *et al.* op. cit.). While there appears little certainty as to which of these factors is dominant, it is nevertheless clear that larger fish evert their stomachs far more frequently than smaller fish.

Payne *et al.* (1987), in their preliminary attempt to quantify hake predation, assumed that the ratio of empty to non-empty stomachs was the same in everted and non-everted cases. While this would seem reasonable as a first approximation, it does however appear that the probability that everted stomachs contain food is larger than the probability that they are empty (as noted by Payne *et al.* op. cit.). The approach of Payne *et al.* (op. cit.) is thus likely to lead to negatively biased estimates of annual consumption and daily ration. Because of these concerns, the sensitivity of the consumption and daily ration estimates to three assumptions regarding stomach eversion has been investigated here:

- i) the ratio of empty to non-empty stomachs is the same in everted and non-everted cases,
- ii) all everted stomachs are empty, and
- iii) no everted stomachs are empty (all having, on average, the same contents as non-empty stomachs).

Note that, although assumption (iii) results in the largest estimates of daily ration and annual consumption, making this assumption does not necessarily provide positively biased estimates of these quantities. This is because it may well be that only fish with extremely full guts evert their stomachs during the hauling process. Assuming that everted

stomachs have the same contents as non-empty stomachs would thus result in negatively biased estimates of daily ration and annual consumption, because the non-empty stomachs in the data base range from virtually empty to gorged.

Since survey biomass estimates are known to be negatively biased [for example, because of net avoidance (DeAlteris *et al.* 1989)] the consumption estimates will also be negatively biased. Because the survey bias is size-specific, reliable comparisons between the annual consumption of different length-classes cannot be made.

2.5.2 Food items and diet consumption

Table 2.8 lists the prey species encountered in the stomachs considered in this study, as well as those prey species recorded by Payne *et al.* (1987). For comparative purposes, the prey species have been grouped into the 13 categories used by Payne *et al.* (*op. cit.*) [see Table 2.8]. It is noticeable that several of the prey species which were recorded during the cruises analyzed here, were not recorded by Payne *et al.* (*op. cit.*), and *vice versa*.

The difference in diet composition between two studies of the same area at different times could be the result of a number of factors. It could be a consequence of changes in the abundance of the various prey species or it could be the result of changes in predator/prey distributional patterns. Finally, the differences could be attributable to the effects of inadequate sample sizes. While it is not currently possible to distinguish between these three hypotheses, the actual

reason is probably a combination of all three factors. This is because, in most cases, the differences in diet composition relate to species which make up only a very small proportion of the diet.

Unlike Payne *et al.* (*op. cit.*), for most of the analyses of this study, the data were pooled across geographic areas. Although this undoubtedly increases the bias of the consumption and daily ration estimates (as prey densities differ across the area covered by the surveys), it reduces the variance of these estimates by increasing the sample sizes. Table 2.9 provides a breakdown of the diet (by mass) for each hake species by length-class and prey species group. These data are illustrated for *M. capensis* and *M. paradoxus* separately by means of histograms in Figure 2.4. A breakdown of the diet (by mass) for each hake species by age-class and prey species group is given in Table 2.10. From Tables 2.8, 2.9 and 2.10, it is clear that the diet of *M. capensis* encompasses a broader spread of species groups than that of *M. paradoxus*. Pelagic fish (particularly anchovy *Engraulis capensis* and round herring *Etrumeus whiteheadi*) form a much larger component of the diet of the inshore *M. capensis* than that of the offshore *M. paradoxus*, whereas lightfish *Maurolicus muelleri*, myctophids (predominantly *Lampanyctodes hectoris*) and cephalopods make up a larger portion of the diet of *M. paradoxus*. Miscellaneous demersal fish form a larger proportion of the diet of *M. capensis* than that of *M. paradoxus*. These findings apply to both the present analysis and that of Payne *et al.* (1987).

For juvenile *M. capensis* (<30cm), a substantial component of the diet (by mass) appears to be anchovy [Figure 2.4(a)]. This was not noted by

Figure 2.4(a)

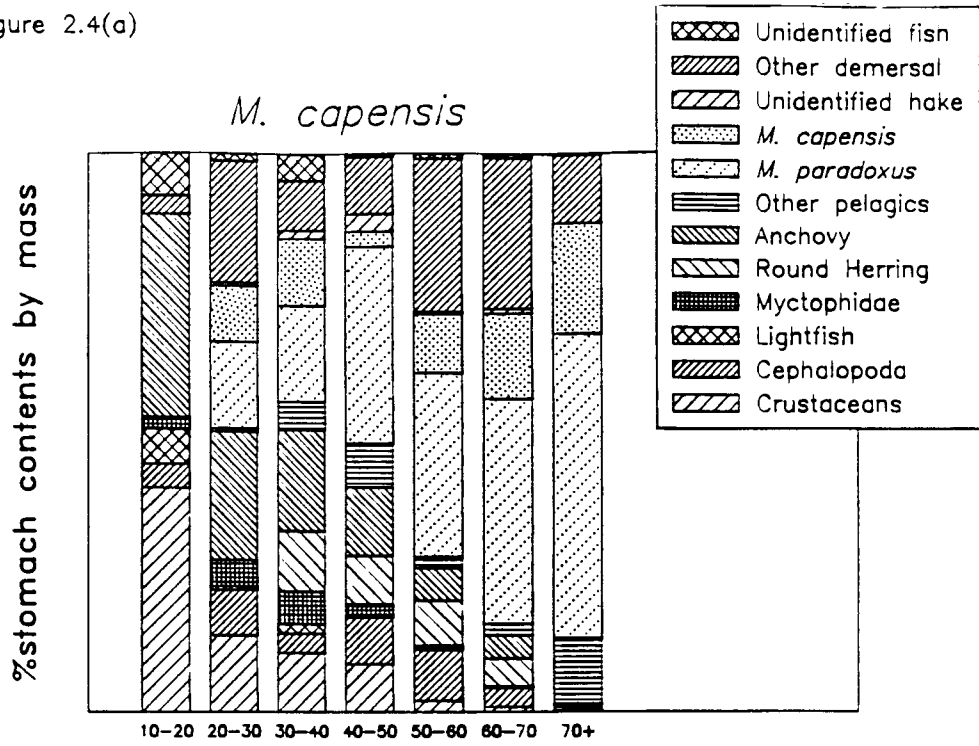


Figure 2.4(b)

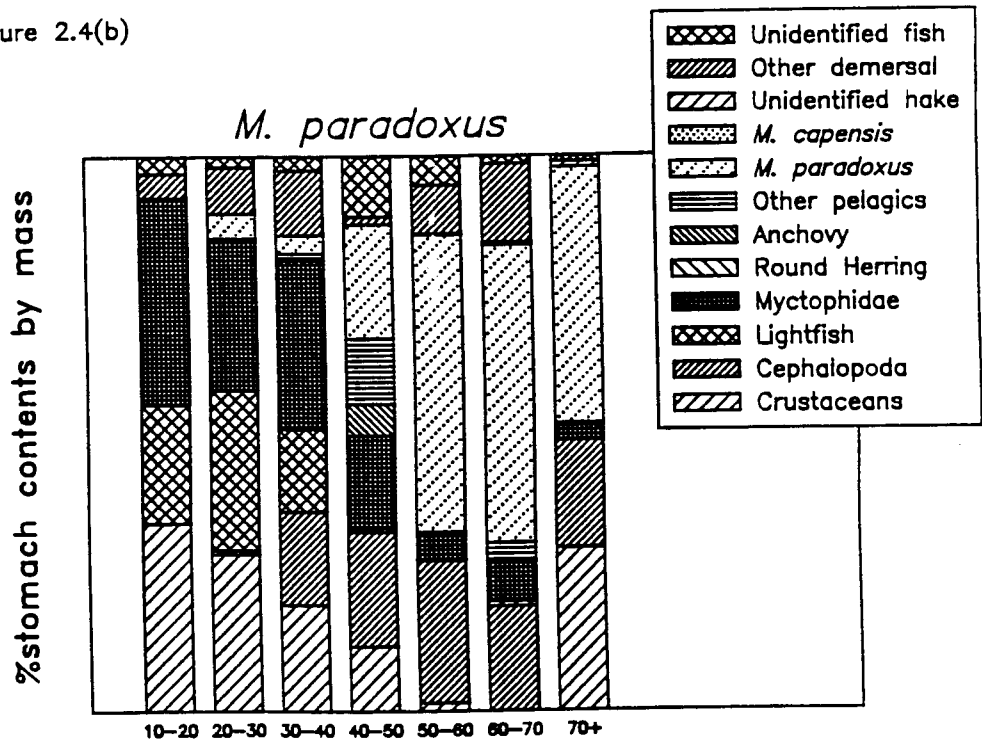


Figure 2.4 : The diet composition by mass for (a) *M. capensis* and (b) *M. paradoxus*. A list of the species making up the various prey species groups can be found in Table 2.8.

Payne *et al.* (1987). However, the biomass of anchovy off South Africa's west coast was much bigger in 1987 and 1988 than in earlier years (see the estimates of biomass at the start of 1986 and 1987 given in Cochrane *et al.* (1991)). If the Cape hakes do not actively search for anchovies, this serves to illustrate the opportunistic feeding nature of these species. The high percentage of crustaceans (virtually all *Funchalia woodwardi*) in the diet of large (>70cm) *M. paradoxus* is probably a sampling artifact (only 11 fish of this species larger than 70cm with "full" stomachs were encountered).

In order to determine the effects of the geographic distributions of prey and predators, the breakdown of the stomach contents by mass, prey species group and predator length-class was performed for each of three sub-divisions of the survey area (north of 32°S, 32°S-34°S, and south of 34°S). The results were difficult to interpret because of the small number of predators per length-class for some of the sub-divisions. This problem was particularly severe for the larger length-classes. The results for this stratification have not been tabulated in detail, but notable features were that anchovy were only found in the stomachs of *M. capensis* sampled south of 32°S. Above this latitude, the diet of *M. capensis* consists of larger percentages of round herring and other demersal species (notably gobies). Moving from north to south, the percentage of crustaceans in the diet of *M. paradoxus* dropped, with crustaceans being replaced mainly by lightfish and Myctophidae. The diet of individuals of *M. capensis* smaller than 30cm sampled between 32°S and 34°S and individuals of *M. capensis* smaller than 60cm sampled south of 34°S were dominated by anchovy. There were also large

differences in the percentages of other prey species for some of the length-classes, but given the small sample sizes, it was not possible to state with any certainty that these effects were merely chance events. These results suggest that the violation of the assumption that predators and prey are distributed homogeneously across the survey area is not likely to introduce substantial bias except, perhaps, as regards the pelagic species.

The percentage of fish in the diet of both species shows an upward trend with age [Table 2.10], although crustaceans continue to contribute to the diet even for fish of age 7 and above. Botha (1980) reported that after age 4, fish are the staple food item and Bentz (1976) attempted to relate this to gill raker morphology. Botha's statement would generally appear to be borne out for *M. paradoxus* [Table 2.10(b)], although there is a high percentage of cephalopods for age 5. However, it does not appear to be appropriate for *M. capensis*, because fish make up over 50% of the diet from the age of 2. This latter observation may be a result of the high availability of anchovy during 1987 and 1988 referenced above and not a general phenomenon. However, in the geographic sub-division in which anchovy were not found in the stomachs of predators (north of 32°S), considerable quantities of demersal fish and hake were found in the stomachs of *M. capensis* indicating that this may well be a general conclusion. Cephalopods make a notable contribution to the diet of all age-classes except 1- and 2-year-old *M. paradoxus* (Table 2.10).

Hake in the diet at a predator length of approximately 20-30cm (ages 2 and 3), and smaller hakes form the most common item (by mass) in the

diet of larger (>50cm) hake. Although Botha (1980) reports that 90% of the food of mature hake consists of small hake, this does not appear to be substantiated by the results of this study nor by those of Payne *et al.* (1987). It should, however, be noted that Botha's (1980) study was localized, and thus only reflected the diet of hake in a relatively small area west of Cape Town.

M. capensis consumes *M. paradoxus* and is also cannibalistic. *M. paradoxus* does not feed on *M. capensis* (Tables 2.9 and 2.10). This is not surprising because, where the distributions of the two species do overlap, the *M. capensis* are generally larger than the *M. paradoxus*.

2.5.3 Daily ration

The estimates of daily ration by predator length-class and prey species group are given in Tables 2.11 and 2.12. Bootstrap C.V.s are also given, and the results are provided for each of the three assumptions regarding the contents of everted stomachs. *M. capensis* is estimated to have a higher daily ration than *M. paradoxus* (compare the estimates of daily ration for each length-class for each assumption regarding the contents of everted stomachs). This is also evident in qualitative terms from Table 2.7 in which, for all length-classes, *M. paradoxus* has the higher proportion of empty stomachs. This difference is most marked for the 40-50cm length-class for assumption (i) for which the difference is a factor of five. This might be attributable to the fact that fish contribute more to the diet of *M. capensis* in the 40-50cm and 50-60cm length-classes than to the diet of *M. paradoxus* in these length-classes (Table 2.9). This is because it is likely that small

prey items (such as crustaceans) are regurgitated more easily than large prey items (such as fish) (B. Rose, pers. comm), so that any regurgitation will effect *M. paradoxus* more than *M. capensis*. Note that it is not always possible to identify fish that have regurgitated some of their stomach contents, so that these fish have not been classified as such.

Although the C.V.s of the daily ration estimates are relatively small (C.V.s < 30% for most length-classes, and often much smaller), the C.V.s of the daily ration by prey species are far higher. In fact these C.V.s, in most cases, are so large as to render the estimates of daily ration by prey species of little value. Perhaps, not surprisingly, the estimates of daily ration (and their components) for larger fish are the most sensitive (in absolute terms) to the assumptions regarding treatment of everted stomachs.

C.V.s can give very misleading impressions of the reliability of quantities if the distributions of these quantities are markedly skew. In order to determine whether or not this effect is playing a role here, the 95% confidence limits for the daily ration estimates by prey species were obtained using the percentile method (Efron (1982) - see Section 10.21). Although there is some evidence for distributional skewness, it is generally the case that a normal approximation to these distributions would be reasonable. Note that, because the estimates of daily ration are relatively precisely determined, while the estimates of daily ration by prey species group are not, there must be substantial covariance between the estimates of daily ration by prey species group.

Table 2.13 provides estimates of daily ration by length-class as a percentage of the average mass of fish in the length-class for each assumption regarding treatment of everted stomachs. The percentages reported in Table 2.13 are not inconsistent with similar estimates reported for other hake species. The range for *M. bilinearis* reported by Durbin *et al.* (1983) is 0.6 -> 4.6%. Livingston (1983) reported a value of 2.5% for *M. productus* and Francis (1982) obtained a range of 0.4 -> 1.1% for the same species from the results of a bioenergetics study.

The daily ration estimates provided in Tables 2.11 and 2.12 were computed using the stomach content data pooled across geographic areas and seasons. However, the pooling will have introduced some biases. For this reason, estimates of daily ration by species and season have been computed using the data from the winter and summer cruises separately, and estimates of daily ration by species and geographic sub-division have also been computed. The estimates and their C.V.s are contrasted with the results given in Tables 2.11 and 2.12, in Table 2.14. There are no marked differences between the estimates obtained for the two seasons except for the 70+cm length-class. However, this is probably due to the small sample size for this length-class. For *M. capensis*, the daily ration does not vary considerably between geographic sub-divisions. The larger variance of estimates for the 60-70cm and 70+cm length-classes is attributable more to small sample sizes than to differences in daily ration. It is extremely difficult to conclude much from the estimates of daily ration by geographic sub-division for *M. paradoxus* because of the enormous C.V.s. There are almost no cases in

which analyzing the data by geographic sub-division or by season results in lower C.V.s. It thus appears that the sample size effects and natural variability are sufficiently large to offset any advantages of reduced bias gained by stratifying the data by season or by sub-division.

2.5.4 Annual consumption

The results of the application of Equations (2.11) and (2.12) to estimate annual consumption by hake are given in Table 2.15. Results are presented for the case in which the data are pooled across all strata, as well as for the case in which the data are analyzed by geographic sub-division. The results for assumption (i) (and the "pooled" analysis) are illustrated in Figure 2.5. The C.V.s of the estimates of annual consumption are also given in Table 2.15. The estimates of annual consumption differ markedly depending on whether the numbers-at-length are estimated from the results of the August 1988 or the July 1989 cruises. These differences are most noticeable for prey species which are found mainly in the diet of 20-40cm *M. capensis* (i.e. crustaceans, cephalopods, round herring, and anchovy) because the July 1989 cruise provided substantially larger estimates of the number of small *M. capensis* than the August 1988 cruise (see Table 2.15). Not surprisingly, the results are also sensitive to the approach used to handle everted stomachs; this effect is most marked for the prey of large hake. However, the magnitude of this effect is insubstantial compared to the effects of selecting different cruises to obtain estimates of numbers-at-length.

Figure 2.5(a)

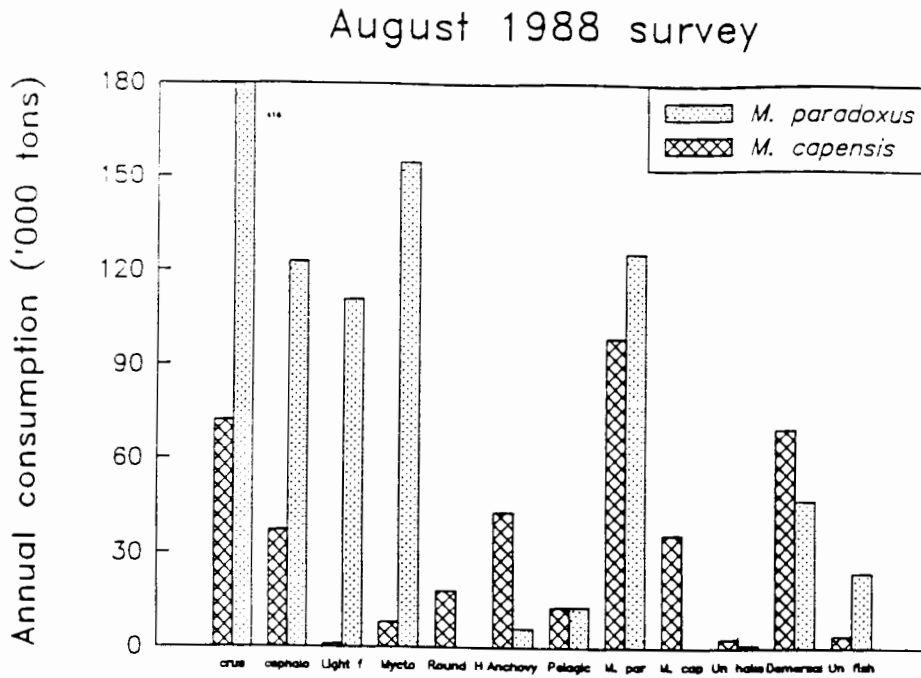


Figure 2.5(b)

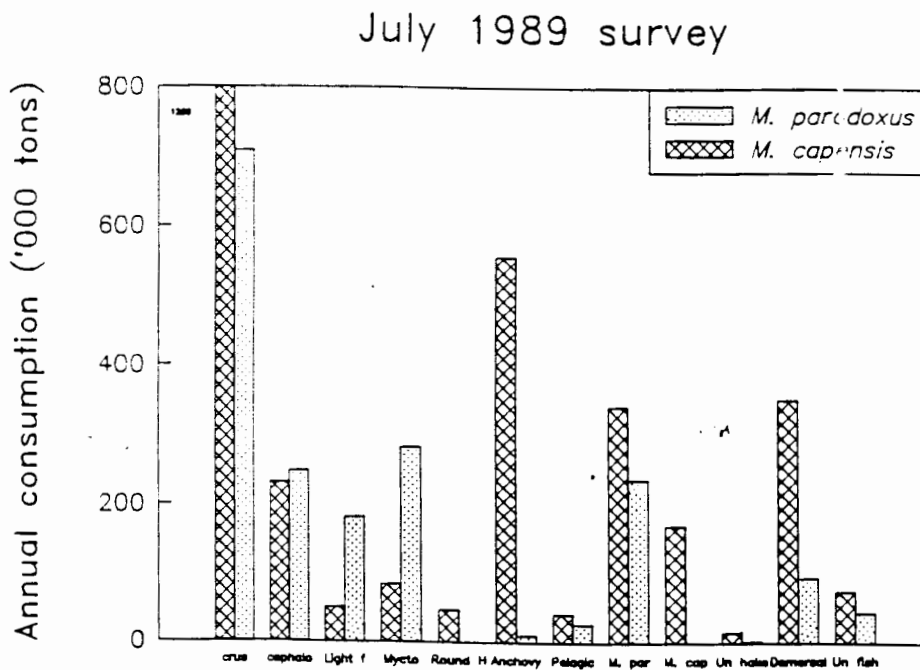


Figure 2.5 : Estimates of annual consumption of various prey species groups by *M. capensis* and *M. paradoxus* obtained using the estimates of the numbers at length of hake from a) the August 1988 cruise and b) the July 1989 cruise. The estimates are given for assumption (i) re everted stomachs. A list of the species making up the various prey species groups can be found in Table 2.8.

The C.V.s obtained by analyzing pooled and geographically disaggregated data are not substantially different. Nevertheless, the pooled results are generally more precise [see the estimates of the C.V.s of the estimated total consumption (all prey species combined) in Table 2.15]. This suggests that pooling across geographic areas is justified in this case. However, there are some marked differences in the point estimates of consumption provided by the pooled and disaggregated analyses. The most substantial difference occurs for predation of anchovy by *M. capensis*: when the data are disaggregated by geographic sub-division, the consumption estimates are markedly lower. This difference is attributable to the observation already noted that no anchovy were found in stomachs of *M. capensis* sampled to the north of 32°S.

The estimated consumption of all species is lower for the analysis of geographically disaggregated data than for aggregated data. This observation would seem to be at variance with the conclusion based on Table 2.14 that daily ration is not sensitive to whether the data are analyzed by geographic sub-division or not. However, the formula applied to obtain the consumption estimates includes the number of fish per length-class [see Equation (2.11)]. The number of fish per length-class differs when different stratifications of the survey area are used, because the densities of hake across the survey area are not uniform. Applying a multiplicative factor correcting for this effect to the results for pooled analyses yields results relatively close to those obtained using geographically disaggregated analysis. This confirms that most of the differences in consumption estimates are a consequence of the differences in the hake biomass estimates.

Nevertheless, not surprisingly (for the reasons discussed above), there is still a marked difference in the estimates of anchovy consumption by *M. capensis*.

The results in Table 2.15 and Figure 2.5 should be interpreted with considerable caution. For example, Table 2.15 suggests that the annual consumption of hake by hake lies between 80 and 92% of the biomass used to calculate the estimates of annual consumption. This is the range of the values of the estimates of total hake consumption by hake divided by the combined biomass used in the calculation inferred from Table 2.15. This might seem to imply that the production to biomass ratio for hake is bounded below by a number which must lie in this range. However, the effects of small fish not being fully available to the fishing gear cause the estimates of biomass from the surveys to be negatively biased, even though the estimates of consumption for hake need not be. Note that this bias is size-specific. Therefore, the real production to biomass ratio may be substantially lower than 80 - 92%.

The fact that small hake (<30cm) are undersampled by the surveys means that the estimates of the consumption of those species which are preyed on by small hake are negatively biased. It therefore follows that the estimates of consumption of crustaceans (mainly *Euphausia lucens*) and amphipods (mainly *Themisto gaudichaudi*) are negatively biased. Such a conclusion was also reached by Payne *et al.* (1987). The C.V.s reported in Table 2.15 indicate that, in some cases, the precision of the consumption estimates are very poor. For example, the C.V. of the annual consumption of *M. capensis* by *M. capensis* using survey data for

July 1989 is 58% or (considerably) more for all three assumptions regarding everted stomachs.

One cause of the high estimates of consumption obtained from the July 1989 numbers-at-length may be the use of numbers-at-length and stomach content data for slightly different periods. Table 2.16 reports "pooled" estimates of annual consumption and their C.V.s obtained using the stomach data collected during the July 1989 survey only. The estimates of annual consumption differ quite markedly from those in all the subtables of Table 2.15. However, not surprisingly, the C.V.s of the annual consumption estimates increase markedly because the number of stomachs analyzed is smaller. This serves once again to illustrate the bias-variance trade-off involved here. The estimates in Table 2.16 should have lower bias than those in Table 2.15, but it is likely in this case that the reduction in bias is more than offset by the increase in variance.

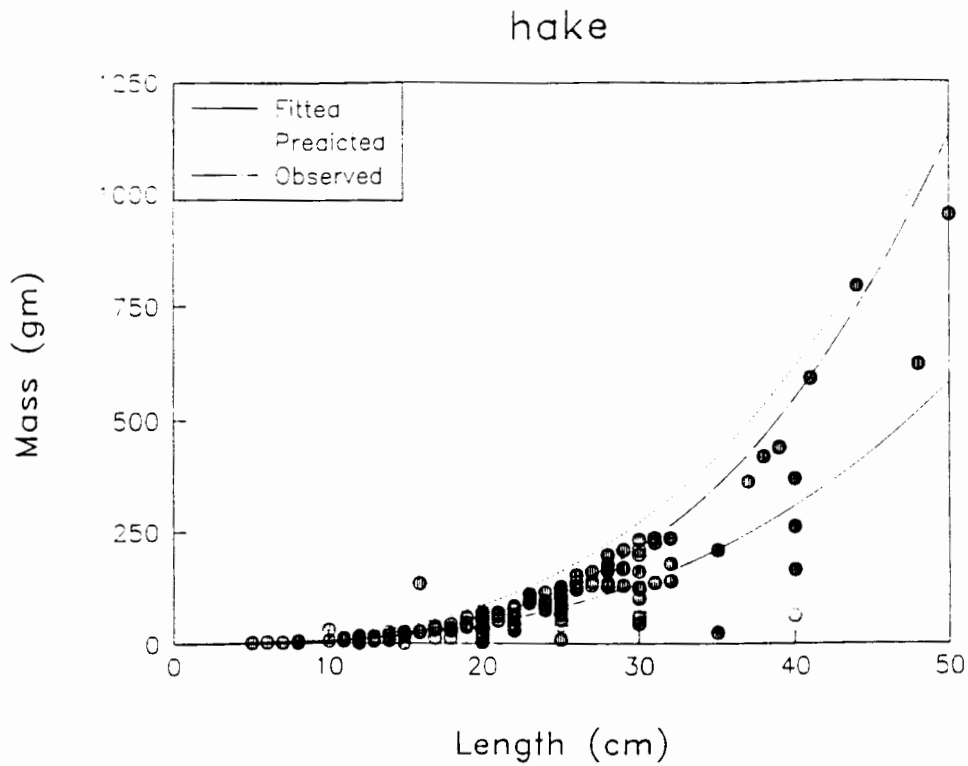
2.5.5 Further considerations

The estimates of daily ration and annual consumption of prey species are very imprecise. Even so, the C.V.s reported are negatively biased, because the variance of the values of the evacuation times and of the estimates of the numbers-at-length have not been taken into account. In addition to questions of precision, the possibility of bias in the estimates of daily ration and annual consumption must also be considered. Four aspects which may have given rise to bias have already been mentioned, namely the estimation of the evacuation times, the treatment of everted stomachs, pooling across geographic areas and

seasons, and the survey biases. A further possible source of bias is the violation of the assumptions that the stomach contents reflect material halfway through the digestion cycle, at which time the food is half digested. In order to determine the extent to which this assumption is defensible, a power curve was fitted (by means of linear regression after logarithm transformation) to masses-at-length for hake and anchovy prey items. The resultant fitted curve, a curve for undigested fish, the fitted curve multiplied by two and the raw data are illustrated for these two species in Figures 2.6(a) and (b). The sources for the curves for undigested fish are Table 2.3 for hake and B.A. Payne (SFRI, pers. comm) for anchovy. If the assumptions regarding material in the stomach are valid, multiplying the fitted curve by two (to give a "predicted" curve) should give a result closely approximating the "observed" curve for undigested fish.

While the "predicted" and "observed" curves for hake are quite similar [see Figure 2.6(a)], the corresponding curves for anchovy are not [Figure 2.6(b)]. It appears that most of the anchovy found in the stomachs considered in this study were virtually undigested [the "observed" curve for undigested fish passes through the raw data - Figure 2.6(b)]. This may, of course, merely be a chance event, but, if it is not, it could lead to some further bias in estimates of daily ration/annual consumption of anchovy by hake. However, it may be that virtually all the anchovy sampled were undigested because it can be difficult to differentiate pelagic prey items to the species level once they become fairly digested.

Figure 2.6(a):



60

Figure 2.6(b):

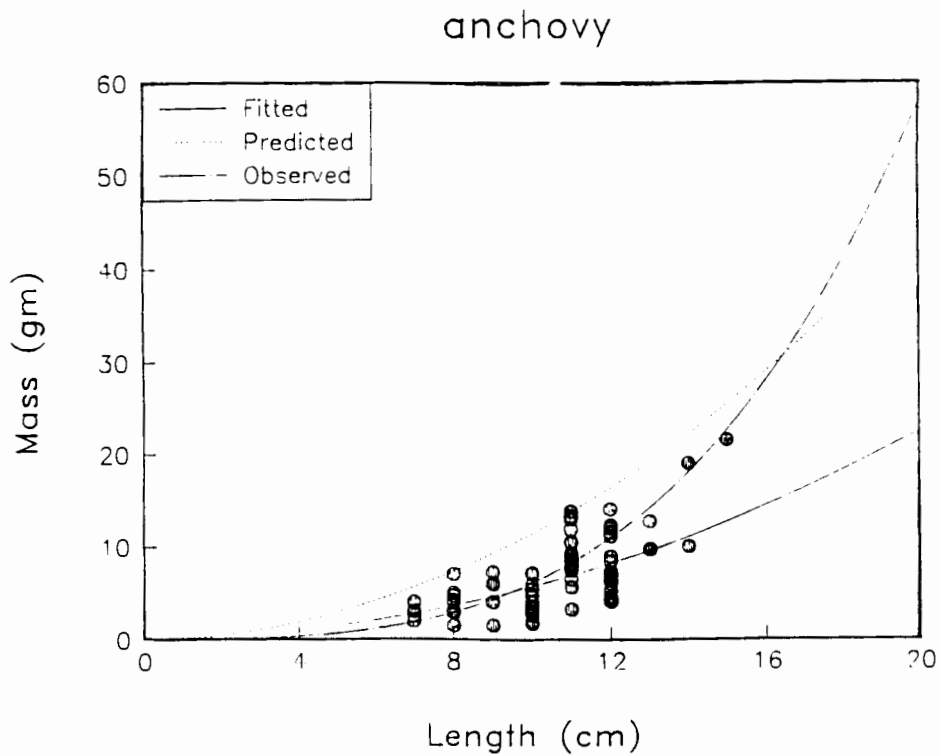


Figure 2.6 : Prey masses versus length. The curves reflect a fit to the stomach content data ("fitted"), a fit to data on undigested fish ("observed"), as well as an estimate of the latter (in terms of assumptions regarding digestion rates) obtained by multiplying the former by two ("predicted"). Plots are given in (a) for hake and (b) for anchovy.

No attempt has been made to perform similar analyses for the other prey species because of the lack of data on masses-at-length for those species. However, the analyses which have been performed suggest that the appropriateness of the assumption that the material in the stomach is (on average) half digested is by no means certain. Further work should be carried out to investigate the consequences of this effect and how, if possible, it can be avoided. While the estimation of more reliable evacuation rates would remove this problem, it is unlikely that this will be possible for a number of years.

2.5.6 Implications for assessments of hake

The estimates of annual consumption of some of the prey species are substantial compared to their biomasses, so that any marked changes in hake abundance are likely to lead to some changes in the rates of natural mortality for other species. In particular, the effect of hake predation on hake appears to be substantial because the consumption estimates and the survey biomass estimates of abundance are of similar magnitude. This appears to confirm the suggestions of Botha (1980) and Payne *et al.* (1987) that hake are very important predators of their own kind. It suggests perhaps, that a multispecies model-estimation procedure should be developed to model the South African west coast marine fisheries (or at least that "cannibalism" needs to be taken into account in assessments of the two hake species combined).

Further, from the estimates presented, it appears highly likely that the "traditional" age-independent estimate of natural mortality of 0.3yr^{-1} used for VPA assessments (e.g. Punt and Butterworth 1989a) is

two low. This finding, therefore, suggests that assessments based on VPA with a fixed age-independent value for M are inappropriate for the management of Cape hakes, and that a procedure which is able to take account of the dynamic nature of natural mortality should be preferred. However, the sample sizes for the stomach content data are small, so that when the resultant variances are combined with the natural variability associated with feeding, the variances of the estimates of daily ration by prey species occasionally become enormous. This poses some serious difficulties regarding the direction in which the development of model-estimation procedures should proceed because the assumptions underlying conventional single-species approaches are of very dubious validity, while the development of multispecies approaches will be hampered by the poor precision with which key parameter values can be estimated.

It is clear that it is necessary to determine the extent of performance degradation which occurs if conventional (constant M) single-species VPA is applied to a fishery in which quantitatively substantial multispecies interactions occur (as such applications are performed at present and would seem likely to continue to be performed for at least the short term for the hake fisheries). This is because it does not necessarily follow that the use of an inaccurate age-independent estimate of natural mortality will lead to serious performance degradation.

2.6 Natural mortality

Natural predators of Cape hake include sharks (Payne *et al.* 1987), snoek (Nepgen 1979), kingklip (Payne 1988), seals and dolphins (Bergh 1986). However, because of their dominance off the South African west coast, hakes are probably their own major predators.

Estimates (in yr^{-1}) of the instantaneous rate of natural mortality, M (generally assumed to be age and time-invariant), range from low values of 0.2-0.25 [Draganik (1976), Leonart *et al.* (1985a), Newman *et al.* (1976), Prenski (1978)] and 0.3 [Davies *et al.* (1980), Leslie (1985), Prenski (1980)] to high estimates larger than 0.4 [Andrew (1986), Assorov and Shcherbitch (1979)]. Andrew (*op. cit.*), using Pauly's (1980) formula and some of the estimates of length and growth parameters presented in the literature, reports that estimates of M may be far larger than previously suspected. Botha (*op. cit.*) estimated separate values of M for males and females using the method of Rihkter and Efanov (1977), i.e.:

$$M = \frac{1.525}{x^{0.72}} - 0.155 \quad (2.4)$$

where x is the age-at-50%-maturity.

However, it must be noted that both the Rihkter and Efanov (1977) and the Pauly (1980) methods of estimating M are highly imprecise [Gulland (1988), Butterworth *et al.* (1989)], so that the estimates reported above should be interpreted with considerable caution. Furthermore, although most of the assessments of the hake resources in the ICSEAF Convention Area have assumed a time-and age-invariant rate of natural

mortality (e.g. Punt and Butterworth 1989a), the effect of hake-on-hake predation (see Section 2.5) is such that natural mortality must depend on the age-structure of the hake population. This is investigated further in Chapter 10.

APPENDIX A : DESCRIPTION OF THE RESEARCH CRUISES

Research cruises have been carried out biannually (usually January and June/July) since July 1983 by the Sea Fisheries Research Institute from a 2452 GRT stern trawler, the RS AFRICANA. The standard gear employed has been a 180ft German trawl net with 1500kg polyvalent doors. The codend is constructed of 75mm mesh lined with 27.5mm pilchard netting to ensure that escapement through the meshes is minimal. The methodology used to select the stations is described in Payne *et al.* (1984). Each trawl was designed to last 30 minutes, but, because of the irregular topography of the sea bend, some trawls were shorter (Payne *et al.* 1987).

The primary purpose of these surveys was to provide a fishery-independent index of abundance which could be used for assessment purposes. Such a relative abundance index has two major advantages over CPUE data. First, because the cruises were carried out by scientists who can distinguish between the two species of hake, biomass indices could be provided for each species separately instead of for both combined. Secondly, the biomass indices should not be markedly affected by systematic changes in catchability which may be the case for CPUE data.

The total catch from each trawl is sorted by species and weighed. Whenever possible, all hake caught are measured. Otherwise, stratified random subsamples of the hake catch are taken to estimate the length composition and the number of hake in the trawl. A small subsample of the catch from each trawl (usually 10 fish) is selected for biological analysis. These fish are selected so that at least three fish per 1cm

length-class are analyzed during each half of the cruise. In the early phase of each half of the cruise, the selection of fish tends to be random, but towards the end of each half, fish in certain length-classes are selected to ensure reasonable sampling coverage. The purpose of this sampling scheme is to obtain a representative sample of the component of the population which is vulnerable to the gear (Payne *et al.* 1987). Clearly, because the distribution of fish is not uniform across the survey area, this sampling scheme will lead to some biases. Random selection throughout might have reduced such bias, but would have led to some lengths of hake (particularly the large fish) being poorly sampled, if at all.

From each fish selected for biological analysis, the age, length and mass are determined, the maturation state assessed visually and the stomach contents analyzed. The stomach content analysis involves first classifying the stomach by its degree of fullness. Payne *et al.* (1987) describe the six degrees of classification which are used. Many hake stomachs are everted [stomachs extruded into the mouth during the hauling process - Payne *et al.* (op. cit.)] and cannot be analyzed. The non-everted stomachs are removed whole and their contents examined. The length and mass of each prey item is noted, and the prey item is classified to the lowest possible taxon. Any items which were deemed to have been swallowed in the net are discounted.

Table 2.1 : Divisions of the ICSEAF Convention area as categorized by dominant hake species caught in each. The limit given in column 3 is the approximate depth at which dominance changes (Kawahara and Nagai 1980).

Division	Dominant shallow water species	Limit (m)	Dominant deep water species
1.3	<i>M. capensis</i>	400	<i>M. polli</i>
1.4	<i>M. capensis</i>	450	<i>M. paradoxus</i>
1.5	<i>M. capensis</i>	450	<i>M. paradoxus</i>
1.6	<i>M. capensis</i>	300 - 350	<i>M. paradoxus</i>
Subarea 2	<i>M. capensis</i>	300 - 350	<i>M. paradoxus</i>

Table 2.2 : Estimates of Schnute growth equation parameters and their standard errors (in parentheses) obtained by fitting the von Bertalanffy and Schnute growth equations by maximizing Equation (2.4). $-\ln L$ refers to the value of the negative of the log of the likelihood function and "#PNTS" refers to the number of points used in the estimation. Estimates of standard error are only provided for the parameters a^S , b^S , l_1 and l_2 if the values of these parameters were not fixed when performing the estimation. Lengths (l) are in cm and ages (a) are in years. Row ' χ^2 ' contains the results of the application of the likelihood ratio test (Draper and Smith 1966) to determine whether or not the von Bertalanffy growth equation provides a significantly poorer fit than the Schnute growth equation. Results marked with an asterisk indicate fits which are significantly poorer than the corresponding Schnute fit at the 5% level.

(Table 2.2 Continued)

(a) *M. capensis*

PARAM- ETER	Males		Females		Males+Females	
	$a_1 = 1$	$a_2 = 9$	$a_1 = 1$	$a_2 = 12$	$a_1 = 1$	$a_2 = 12$
von Bertalanffy fit						
a^S	0.086	(0.007)	0.041	(0.003)	0.039	(0.003)
b^S	1		1		1	
$l_1(a=a_1)$	16.939	(0.231)	17.630	(0.215)	17.657	(0.153)
$l_2(a=a_2)$	80.451	(0.550)	105.924	(0.565)	105.888	(0.542)
σ	3.273		3.383		3.405	
l_∞	144.6		260.9		270.6	
t_0	-0.449		-0.707		-0.730	
$-\ln L$	462.362		802.659		1284.533	
Schnute fit						
a^S	0.117	(0.016)	0.038	(0.005)	0.011	(0.001)
b^S	0.846	(0.075)	1.032	(0.028)	1.228	(0.012)
$l_1(a=a_1)$	17.182	(0.252)	17.552	(0.181)	17.182	(0.153)
$l_2(a=a_2)$	80.078	(0.530)	105.562	(0.531)	105.221	(0.344)
σ	3.263		3.308		3.356	
l_∞	126.2		268.9		568.4	
t_0	-0.749		-0.623		-0.246	
$-\ln L$	460.737		801.043		1273.300	
#PNTS	1043		1619		2662	
χ^2	3.250		3.231		22.466*	

(Table 2.2 Continued)

(b) *M. paradoxus*

PARAM- ETER	Males		Females		Males+Females	
	$a_1 = 1$	$a_2 = 7$	$a_1 = 1$	$a_2 = 11$	$a_1 = 1$	$a_2 = 11$
von Bertalanffy fit						
a^S	0.093	(0.016)	0.051	(0.003)	0.049	(0.003)
b^S	1		1		1	
$k_1(a=a_1)$	19.058	(0.208)	19.315	(0.193)	19.194	(0.110)
$k_2(a=a_2)$	65.029	(0.929)	94.649	(0.401)	94.802	(0.407)
σ	2.914		3.130		3.089	
l_∞	126.6		207.9		219.4	
t_0	-0.755		-0.912		-0.914	
$-\ln L$	329.344		925.801		1265.695	
Schnute fit						
a^S	0.623	(0.038)	0.051	(0.002)	0.049	(0.002)
b^S	-1.279	(0.171)	1.000	(0.022)	1.000	(0.015)
$k_1(a=a_1)$	19.681	(0.195)	19.315	(0.206)	19.194	(0.126)
$k_2(a=a_2)$	60.739	(0.729)	94.649	(0.372)	94.802	(0.324)
σ	2.938		3.130		3.089	
l_∞	64.8		207.9		214.4	
t_0	-		-0.912		-0.914	
$-\ln L$	313.014		925.801		1265.695	
#PNTS	881		1746		2627	
χ^2	32.659*		0.000		0.000	

(Table 2.2 Continued)

(c) Both species combined

PARAM- ETER	Males		Females		Males+Females	
	$a_1 = 1$	$a_2 = 9$	$a_1 = 1$	$a_2 = 12$	$a_1 = 1$	$a_2 = 12$
von Bertalanffy fit						
a^S	0.065	(0.005)	0.050	(0.003)	0.046	(0.002)
b^S	1		1		1	
$k_1(a=a_1)$	18.332	(0.135)	18.465	(0.145)	18.542	(0.097)
$k_2(a=a_2)$	81.194	(0.507)	102.081	(0.508)	102.620	(0.362)
σ	3.106		3.277		3.221	
l_∞	173.4		216.1		230.3	
t_0	-0.719		-0.786		-0.825	
$-\ln L$	815.637		1781.779		2597.587	
Schnute fit						
a^S	0.118	(0.004)	0.055	(0.001)	0.048	(0.002)
b^S	0.723	(0.019)	0.962	(0.012)	0.999	(0.009)
$k_1(a=a_1)$	18.638	(0.163)	18.680	(0.135)	18.512	(0.087)
$k_2(a=a_2)$	80.368	(0.409)	101.697	(0.312)	102.346	(0.283)
σ	3.148		3.264		3.220	
l_∞	130.0		205.5		223.0	
t_0	-1.388		-0.907		-0.810	
$-\ln L$	807.279		1781.252		2596.983	
#PNTS	1924		3365		5289	
χ^2	16.715*		1.054		1.207	

Table 2.3 : Estimates of the mass-length relationship parameters obtained by fitting Equation (2.8) to data obtained from the research cruises. Column 'F' is obtained from the application of the log-likelihood ratio test (Draper and Smith 1966) to determine whether the fits for males and females are significantly different. SS is the residual sum of squares.

(a) Males

SPECIES	$\ln(a_w)$	b_w	SS	r	#PNTs
Cap Jan	-5.323 (0.056)	3.119 (0.015)	11.491	0.993	547
Cap Jul	-5.204 (0.042)	3.091 (0.011)	9.525	0.996	650
Par Jan	-5.111 (0.056)	3.043 (0.017)	11.309	0.992	535
Par Jul	-4.929 (0.038)	3.004 (0.011)	5.430	0.996	554
Cap All	-5.262 (0.034)	3.105 (0.009)	21.131	0.995	1197
Par All	-5.013 (0.034)	3.022 (0.010)	17.496	0.994	1089
All Data	-5.205 (0.023)	3.085 (0.007)	40.059	0.995	2286

(b) Females

SPECIES	$\ln(a_w)$	b_w	SS	r	#PNTs
Cap Jan	-5.303 (0.051)	3.111 (0.013)	25.403	0.990	1112
Cap Jul	-5.304 (0.043)	3.123 (0.011)	20.581	0.994	980
Par Jan	-5.175 (0.032)	3.067 (0.008)	18.722	0.996	1100
Par Jul	-5.022 (0.029)	3.031 (0.008)	15.704	0.997	1106
Cap All	-5.300 (0.033)	3.116 (0.009)	47.075	0.992	2092
Par All	-5.094 (0.022)	3.048 (0.006)	34.740	0.996	2206
All Data	-5.199 (0.019)	3.083 (0.005)	85.661	0.994	4298

(c) Males and Females combined

SPECIES	$\ln(a_w)$	b_w	SS	r	#PNTs	F
Cap Jan	-5.291 (0.036)	3.109 (0.010)	36.940	0.992	1659	1.038
Cap Jul	-5.275 (0.030)	3.114 (0.008)	30.326	0.995	1630	5.946
Par Jan	-5.183 (0.025)	3.068 (0.007)	30.180	0.996	1635	4.061
Par Jul	-4.996 (0.021)	3.024 (0.006)	21.178	0.997	1660	1.718
Cap All	-5.289 (0.024)	3.113 (0.006)	68.227	0.994	3289	0.497
Par All	-5.091 (0.017)	3.046 (0.005)	52.380	0.996	3295	4.555
All Data	-5.203 (0.014)	3.084 (0.004)	125.728	0.995	6584	0.208

Table 2.4 : Estimates of total length (in cm) at selected levels of sexual maturity, obtained by fitting model (2.9) to the maturation data from the research cruises during the period 1986-1989. The estimates of C.V. (expressed as a percentage) given in the second row of each block were computed using the (conditioned) parametric bootstrap procedure. "Cap" refers to *M. capensis*, "Par" refers to *M. paradoxus*, "Sum" refers to the summer cruises and "Win" refers to the winter cruises. SS is the residual sum of squares (see Equation 2.10) and n is the number of data points.

(a) Males

CRUISE	δ_m	5%	25%	50%	75%	95%	n	SS/n
Cap Sum	0.183 3.84	14.05 2.35	24.15 2.90	30.15 3.05	36.16 3.15	46.25 3.29	547	0.200
Cap Win	0.187 6.16	13.91 4.89	23.75 5.94	29.61 6.25	35.47 6.48	45.32 6.73	650	0.933
Par Sum	0.171 13.60	15.92 15.39	26.71 15.35	33.13 15.88	39.55 16.36	50.33 16.99	535	1.034
Par Win	0.161 4.95	15.30 3.73	26.76 4.41	33.59 4.61	40.41 4.75	51.88 4.90	553	0.272
Cap All	0.188 2.59	13.82 1.66	23.62 2.12	29.46 2.21	35.30 2.29	45.10 2.37	1197	0.442
Par All	0.162 2.52	15.20 1.71	26.57 1.84	33.33 1.95	40.09 2.02	51.45 2.12	1088	0.468
All Data	0.175 1.60	14.50 1.03	25.07 1.28	31.36 1.34	37.66 1.38	48.23 1.43	2285	0.151

(Table 2.4 Continued)

(b) Females

CRUISE	δ_m	5%	25%	50%	75%	95%	n	SS/n
Cap Sum	0.115 5.38	24.46 6.99	40.49 2.54	50.04 1.64	59.58 1.49	75.61 1.96	986	0.102
Cap Win	0.139 5.23	20.92 5.40	34.24 2.39	42.17 2.02	50.10 2.10	63.42 2.48	816	0.075
Par Sum	0.233 12.33	37.07 3.43	45.00 2.11	49.72 2.33	54.44 2.87	62.36 3.82	1013	0.230
Par Win	0.378 13.40	39.01 2.85	43.90 1.55	46.80 1.22	49.71 1.29	54.60 1.90	891	0.071
Cap All	0.118 3.98	21.82 5.64	37.50 1.97	46.83 1.22	56.16 1.07	71.85 1.41	1802	0.049
Par All	0.268 9.58	37.63 2.31	44.52 1.15	48.63 1.46	52.73 2.05	59.63 3.00	1904	0.104
All Data	0.159 3.56	28.93 2.63	40.56 1.11	47.47 0.80	54.39 0.85	66.02 1.15	3706	0.026

(Table 2.4 Continued)

(c) Males and females combined

CRUISE	δ_m	5%	25%	50%	75%	95%	n	SS/n
Cap Sum	0.122 2.30	18.52 1.67	33.60 1.96	42.57 2.02	51.54 2.08	66.62 2.12	1533	0.168
Cap Win	0.148 1.73	16.21 1.11	28.72 1.32	36.16 1.41	43.60 1.44	56.10 1.52	1466	0.079
Par Sum	0.126 3.77	19.23 4.42	33.87 1.74	42.58 1.43	51.29 1.52	65.93 1.80	1548	0.068
Par Win	0.145 3.62	19.76 4.76	32.49 2.03	40.07 1.55	47.65 1.45	60.39 1.64	1444	0.094
Cap All	0.129 1.61	17.86 1.12	32.19 1.30	40.73 1.37	49.26 1.40	63.59 1.43	2999	0.083
Par All	0.132 3.10	19.25 3.53	33.21 1.39	41.51 1.16	49.82 1.22	63.78 1.51	2992	0.047
All Data	0.129 1.32	17.85 0.95	32.18 1.09	40.71 1.13	49.24 1.16	63.57 1.20	5991	0.037

Table 2.5 : Estimates of total length (in cm) at selected levels of sexual maturity obtained by Botha (1986).

% Maturity	<i>M. capensis</i>		<i>M. paradoxus</i>		Both species combined		
	Males	Females	Males	Females	Males	Females	Both
0.5	22	28	18	18	-	-	-
25	32	41	30	43	30	43	35
50	36	48	38	47	37	46	42
75	48	53	42	51	43	51	48
100	58	67	58	62	-	-	-
Sample Size	730	1109	1633	2655			

Table 2.6 : Estimates of ages at selected levels of sexual maturity, obtained by converting the estimates of lengths at those maturity levels (Table 2.4) to ages using the Schnute growth equation parameter values given in Table 2.2.

Data Set	5%	25%	50%	75%	95%
Males					
<i>M. capensis</i>	0.68	1.62	2.20	2.81	3.89
<i>M. paradoxus</i>	0.35	1.84	2.58	3.32	4.77
Both species	0.55	1.68	2.36	3.06	4.29
Females					
<i>M. capensis</i>	1.43	3.07	4.11	5.20	7.16
<i>M. paradoxus</i>	3.00	3.81	4.31	4.82	5.72
Both species	2.08	3.38	4.18	5.03	6.52
Males and Females					
<i>M. capensis</i>	1.06	2.47	3.40	4.38	6.15
<i>M. paradoxus</i>	1.01	2.52	3.48	4.48	6.29
Both species	0.93	2.44	3.40	4.40	6.19

Table 2.7 : Classification of the stomachs considered in this study into empty, "full" (containing food), and everted, by length-class.

Length-class (cm)	Total number of stomachs sampled	Frequency (%)		
		Empty	Full	Everted
a) <i>M. capensis</i>				
10 - 20	70	10.0	75.7	14.3
20 - 30	173	22.0	54.3	23.7
30 - 40	218	20.6	49.5	29.8
40 - 50	210	18.1	50.5	31.4
50 - 60	202	16.3	37.6	46.0
60 - 70	157	14.0	40.8	45.2
70+	110	17.3	43.6	39.1
b) <i>M. paradoxus</i>				
10 - 20	182	15.4	70.9	13.7
20 - 30	230	23.0	53.5	23.5
30 - 40	241	27.4	37.3	35.3
40 - 50	191	28.8	24.1	47.1
50 - 60	182	18.7	22.0	59.3
60 - 70	168	14.9	16.7	68.5
70+	84	17.9	13.1	69.0

Table 2.8 : Food items found in the stomachs of Cape hake off the South African west coast [a : from Payne et al. (1987); b : *M. capensis* - this study; c : *M. paradoxus* - this study]. Column 'GROUP' indicates the group to which the species has been assigned. Entries marked with asterisks indicate species observed by Payne et al. (op. cit.) but not included in their analysis.

TAXON	SPECIES	GROUP	a	b	c
ARTHROPODA					
CRUSTACEA					
	Crustacean remains	1	*	X	X
Euphausiidae	<i>Euphausia lucens</i>	1	X	X	X
	<i>Euphausia hanseni</i>	1		X	X
	<i>Nyctiphanes capensis</i>	1	X		
	Unidentified Euphausiidae	1	*	X	X
Amphipoda	<i>Themisto gaudichaudi</i>	1	X	X	X
	Unidentified Amphipoda	1	*	X	X
Mysidacea		1	X	X	X
Stomatopoda	<i>Pterygosquilla armata capensis</i>	1	X	X	
Decapoda	<i>Pasiphaea</i> spp.	1		X	X
	<i>Funchalia woodwardi</i>	1	X	X	X
	<i>Parapagurus dimorphus</i>	1		X	
	Penaeidae	1	X	X	X
	Megalopa larvae	1	X		
	Unidentified Decapoda	1	X		
MOLLUSCA					
CEPHALOPODA					
	<i>Loligo vulgaris reynaudii</i>	2	X		
	<i>Todaropsis eblanae</i>	3	X	X	X
	<i>Todarodes angolenis</i>	3	X	X	X
	<i>Lolliguncula mercatoris</i>	3	X	X	X
	<i>Austrorossia mastigop</i>	3		X	
	<i>Lycoteuthis diadema</i>	3			X
	<i>Octopus</i> spp.	3	X		
	<i>Sepia</i> spp.	3	X	X	
	Unidentified Cephalopoda	3	X	X	X

(Table 2.8 Continued)

TAXON	SPECIES	GROUP	a	b	c
CHORDATA					
PISCES					
Sternoptychidae	<i>Maurolicus muelleri</i> (lightfish)	4	X	X	X
Myctophidae	<i>Lampanyctodes hectoris</i>	5	X	X	X
	<i>Diaphus</i> spp.	5	X		
	<i>Symbolophorus boops</i>	5			X
	Other Myctophidae	5	X		
Round herring	<i>Etrumeus whiteheadi</i>	6	X	X	
Anchovy	<i>Engraulis capensis</i>	7	X	X	X
Other pelagics	<i>Sardinops ocellatus</i>	8	X	X	
	<i>Scomber japonicus</i>	8	X	X	
	<i>Scomberesox saurus</i>	8	X	X	
	<i>Photichthys argenteus</i>	8	X		X
	<i>Thyrsites atun</i>	8	X	X	
Merluccidae	<i>Merluccius paradoxus</i>	9	X	X	X
	<i>Merluccius capensis</i>	10	X	X	
	Unidentified hake	11	X	X	X
Other demersals	<i>Chelidonichthys capensis</i>	12		X	
	<i>Chlorophthalmus agassizi</i>	12		X	
	<i>Coelorinchus fasciatus</i>	12	X	X	X
	<i>Cynoglossus zanzibarensis</i>	12	X	X	X
	<i>Emmelichthys nitidus</i>	12	X	X	
	<i>Epigonus</i> spp.	12	X		
	Gempylidae	12			
	<i>Genypterus capensis</i>	12	X		
	<i>Gnathophis</i> spp.	12		X	
	<i>Helicolenus dactylopterus</i>	12	X	X	
	<i>Hoplostethus</i> spp.	12			
	<i>Lepidopus caudatus</i>	12	X	X	
	<i>Nezumia</i> spp.	12			X
	Ophichthidae	12	X		
	<i>Paracallionymus costatus</i>	12	X	X	X
	Paralepididae	12	X	X	X
	<i>Stomias boa</i>	12			X
	<i>Sufflogobius bibarbatus</i>	12	X	X	X
	<i>Trachurus trachurus capensis</i>	12	X	X	
	<i>Tripterochysis gilchristi</i>	12			X
	<i>Zeus capensis</i>	12	X		
	Unidentified fish	13	X	X	X

Table 2.9 : Breakdown of the stomach contents of the two Cape hake species by mass, prey species group and predator length-class. Results are expressed as percentages. The data for fish larger than 70cm have been pooled. "UNID" refers to "unidentified".

(a) *M. capensis*

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	40.2	13.4	10.9	8.9	2.0	0.6	0.3
CEPHALOPODA	4.3	8.3	3.6	8.8	9.5	3.7	0.0
LIGHTFISH	6.2	0.6	1.7	0.1	0.0	0.0	0.0
MYCTOPHIDAE	2.1	4.7	6.1	2.4	0.9	0.3	0.0
ROUND HERRING	0.0	0.0	11.0	9.0	7.9	3.4	0.7
ANCHOVY	36.3	23.0	18.4	12.9	6.1	4.9	0.6
OTHER PELAGIC	0.0	0.6	5.2	8.1	2.2	2.4	13.2
<i>M. PARADOXUS</i>	0.0	15.4	15.0	34.8	34.3	34.3	50.6
<i>M. CAPENSIS</i>	0.0	9.9	12.4	3.0	11.0	17.5	21.1
UNID HAKE	0.0	0.6	1.5	2.3	0.4	0.9	0.0
OTHER DEMERSAL	3.4	22.1	9.3	9.2	25.2	31.4	13.3
UNID FISH	7.5	1.4	4.9	0.6	0.6	0.7	0.1

(b) *M. paradoxus*

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	34.0	29.3	19.2	11.7	1.7	0.0	29.2
CEPHALOPODA	0.0	0.7	17.1	20.6	23.9	18.9	19.5
LIGHTFISH	21.5	29.9	15.1	0.7	0.0	0.9	0.0
MYCTOPHIDAE	37.2	27.4	29.8	16.9	5.7	7.1	3.1
ROUND HERRING	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ANCHOVY	0.0	0.0	0.7	5.7	0.0	0.0	0.0
OTHER PELAGIC	0.0	0.1	1.3	11.7	0.0	3.5	0.0
<i>M. PARADOXUS</i>	0.0	4.5	3.3	20.5	58.8	53.6	46.1
<i>M. CAPENSIS</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0
UNID HAKE	0.0	0.0	0.0	0.0	0.2	0.5	1.0
OTHER DEMERSAL	4.3	6.1	10.9	1.4	4.4	14.1	0.0
UNID FISH	3.0	1.9	2.6	10.8	5.3	1.4	1.0

Table 2.10 : Breakdown of the stomach contents of the two Cape hake species by mass, prey species group and predator age. Results are expressed as percentages. The data for fish older than 7 years have been pooled. "UNID" refers to "unidentified".

(a) *M. capensis*

PREY SPECIES GROUP	AGE (yr)						
	1	2	3	4	5	6	7+
CRUSTACEA	83.6	24.5	13.2	8.5	7.0	2.2	0.8
CEPHALOPODA	11.4	3.8	8.6	7.6	5.3	10.9	4.2
LIGHTFISH	0.2	3.9	2.3	0.2	0.1	0.0	0.0
MYCTOPHIDAE	0.0	5.8	4.7	3.1	2.7	0.8	0.3
ROUND HERRING	0.0	0.0	3.0	10.8	7.9	9.7	4.2
ANCHOVY	0.7	25.6	12.0	17.9	15.7	5.7	5.6
OTHER PELAGIC	0.0	0.0	5.7	7.8	4.2	2.4	2.4
<i>M. PARADOXUS</i>	0.0	2.6	13.6	32.3	33.8	33.5	35.8
<i>M. CAPENSIS</i>	0.0	0.0	20.6	1.3	7.0	10.0	18.3
UNID HAKE	0.0	0.0	0.3	0.9	3.1	0.8	0.9
OTHER DEMERSAL	0.0	30.4	13.4	7.6	12.4	23.2	26.8
UNID FISH	4.0	3.5	2.6	1.9	0.9	0.8	0.7

(b) *M. paradoxus*

PREY SPECIES GROUP	AGE (yr)						
	1	2	3	4	5	6	7+
CRUSTACEA	30.9	37.2	22.6	13.7	5.6	1.2	1.5
CEPHALOPODA	0.0	1.3	10.0	13.5	37.6	17.8	18.3
LIGHTFISH	24.8	25.3	24.5	2.5	0.3	0.4	0.9
MYCTOPHIDAE	37.3	31.8	28.0	22.9	15.4	3.7	6.3
ROUND HERRING	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ANCHOVY	0.0	0.0	0.2	5.7	0.6	0.0	0.0
OTHER PELAGIC	0.0	0.2	0.0	12.5	0.7	0.4	3.1
<i>M. PARADOXUS</i>	0.0	0.0	3.2	18.8	26.3	57.3	55.6
<i>M. CAPENSIS</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0
UNID HAKE	0.0	0.0	0.0	0.0	0.5	0.3	0.5
OTHER DEMERSAL	0.0	3.0	8.6	7.9	1.6	14.2	12.1
UNID FISH	7.1	1.2	2.9	2.5	11.4	4.7	1.8

Table 2.11 : Estimates and estimated C.V.s (expressed as percentages) of the daily ration in grammes for *M. capensis* by prey species group. The designations (i)-(iii) refer to the three assumptions concerning the treatment of everted stomachs.

Assumption (i)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10-20	20-30	30-40	40-50	50-60	60-70	70+
CRUSTACEA	0.57	0.69	1.55	3.35	1.93	1.00	0.98
CEPHALAPODS	0.03	0.21	0.25	1.66	4.48	2.94	0.03
LIGHTFISH	0.03	0.01	0.08	0.02	.0	0	0
MYCTOPHIDAE	0.01	0.08	0.29	0.30	0.28	0.14	0
ROUND HERRING	0	0	0.52	1.13	2.57	1.78	0.77
ANCHOVY	0.19	0.40	0.87	1.61	1.97	2.57	0.66
OTHER PELAGIC	0	0.01	0.25	1.01	0.71	1.28	13.98
<i>M. PARADOXUS</i>	0	0.27	0.71	4.36	11.14	18.13	53.42
<i>M. CAPENSIS</i>	0	0.17	0.59	0.37	3.58	9.23	22.24
UNID HAKE	0	0.01	0.07	0.29	0.13	0.47	0.04
OTHER DEMESAL	0.02	0.38	0.44	1.15	8.16	16.60	14.04
UNID FISH	0.04	0.02	0.23	0.07	0.18	0.39	0.15
DAILY RATION	0.9	2.3	5.9	15.3	35.2	54.5	106.3

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10-20	20-30	30-40	40-50	50-60	60-70	70+
CRUSTACEA	15	18	21	21	36	42	37
CEPHALAPODS	72	77	55	46	47	45	99
LIGHTFISH	65	90	55	60	-	-	-
MYCTOPHIDAE	102	56	40	66	39	80	-
ROUND HERRING	-	-	49	52	32	99	100
ANCHOVY	45	26	35	32	39	63	88
OTHER PELAGIC	-	98	57	59	64	64	61
<i>M. PARADOXUS</i>	-	47	44	27	25	30	25
<i>M. CAPENSIS</i>	-	121	34	82	44	55	53
UNID HAKE	-	87	95	53	76	52	94
OTHER DEMESAL	99	40	27	30	34	33	35
UNID FISH	99	60	75	87	74	64	83
DAILY RATION	12	15	10	12	11	17	21

* Ratio of empty to non-empty stomachs is the same in everted and non-everted cases.

(Table 2.11 Continued)

Assumption (ii)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	0.49	0.53	1.09	2.30	1.04	0.55	0.59
CEPHALAPODS	0.03	0.16	0.18	1.14	2.42	1.61	0.02
LIGHTFISH	0.03	0.01	0.06	0.01	0	0	0
MYCTOPHIDAE	0.01	0.06	0.20	0.20	0.15	0.07	0
ROUND HERRING	0	0	0.37	0.77	1.39	0.98	0.47
ANCHOVY	0.16	0.30	0.61	1.11	1.07	1.41	0.40
OTHER PELAGIC	0	0.01	0.17	0.69	0.38	0.70	8.51
<i>M. PARADOXUS</i>	0	0.20	0.50	2.99	6.01	9.93	32.54
<i>M. CAPENSIS</i>	0	0.13	0.41	0.25	1.93	5.06	13.55
UNID HAKE	0	0.01	0.05	0.20	0.07	0.26	0.02
OTHER DEMESAL	0.02	0.29	0.31	0.79	4.41	9.09	8.55
UNID FISH	0.03	0.02	0.16	0.05	0.10	0.22	0.09
DAILY RATION	0.8	1.7	4.1	10.5	19.0	30.0	64.7

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	16	18	21	20	36	42	39
CEPHALAPODS	62	78	53	46	45	45	88
LIGHTFISH	56	69	52	82	-	-	-
MYCTOPHIDAE	89	58	41	67	39	88	-
ROUND HERRING	-	-	48	53	32	101	102
ANCHOVY	46	27	34	33	37	61	89
OTHER PELAGIC	-	75	58	60	66	63	63
<i>M. PARADOXUS</i>	-	49	43	28	25	31	27
<i>M. CAPENSIS</i>	-	118	34	85	45	53	54
UNID HAKE	-	66	93	53	75	52	117
OTHER DEMESAL	85	40	27	31	34	33	36
UNID FISH	114	46	73	85	74	63	84
DAILY RATION	13	15	9	13	13	17	22

* All everted stomachs are empty

(Table 2.11 Continued)

Assumption (iii)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	0.58	0.76	1.74	3.73	2.32	1.16	1.13
CEPHALAPODS	0.03	0.23	0.28	1.85	5.37	3.40	0.03
LIGHTFISH	0.03	0.01	0.09	0.02	0	0	0
MYCTOPHIDAE	0.01	0.09	0.33	0.33	0.33	0.16	0
ROUND HERRING	0	0	0.59	1.26	3.09	2.06	0.88
ANCHOVY	0.19	0.43	0.98	1.79	2.37	2.96	0.76
OTHER PELAGIC	0	0.01	0.28	1.13	0.85	1.47	16.14
<i>M. PARADOXUS</i>	0	0.29	0.80	4.86	13.37	20.95	61.69
<i>M. CAPENSIS</i>	0	0.19	0.66	0.41	4.29	10.67	25.68
UNID HAKE	0	0.01	0.08	0.32	0.15	0.54	0.04
OTHER DEMESAL	0.02	0.42	0.50	1.28	9.80	19.18	16.21
UNID FISH	0.04	0.03	0.26	0.08	0.22	0.46	0.17
DAILY RATION	0.9	2.5	6.6	17.1	42.2	63.0	121.9

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	15	18	20	20	35	41	37
CEPHALAPODS	73	78	54	45	47	44	115
LIGHTFISH	67	98	55	67	-	-	-
MYCTOPHIDAE	104	54	40	67	40	81	-
ROUND HERRING	-	-	48	52	31	98	100
ANCHOVY	45	26	34	32	40	64	89
OTHER PELAGIC	-	108	57	58	64	65	61
<i>M. PARADOXUS</i>	-	48	43	27	25	29	24
<i>M. CAPENSIS</i>	-	121	34	82	44	54	53
UNID HAKE	-	96	93	54	80	52	108
OTHER DEMESAL	101	40	26	30	34	33	34
UNID FISH	102	43	74	85	71	63	84
DAILY RATION	11	14	9	11	11	17	18

* No everted stomachs are empty

Table 2.12 : Estimates and estimated C.V.s (expressed as percentages) of the daily ration in grammes for *M. paradoxus* by prey species group. The designations (i)-(iii) refer to the three assumptions concerning the treatment everted stomachs.

Assumption (i)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10-20	20-30	30-40	40-50	50-60	60-70	70+
CRUSTACEA	0.44	0.90	1.17	0.82	0.47	0.01	35.69
CEPHALAPODS	0	0.01	0.52	0.73	3.39	6.48	11.96
LIGHTFISH	0.09	0.30	0.31	0.02	0	0.20	0
MYCTOPHIDAE	0.16	0.28	0.61	0.40	0.54	1.63	1.28
ROUND HERRING	0	0	0	0	0	0	0
ANCHOVY	0	0	0.01	0.13	0	0	0
OTHER PELAGIC	0	0	0.03	0.27	0	0.79	0
<i>M. PARADOXUS</i>	0	0.05	0.07	0.48	5.56	12.24	18.82
<i>M. CAPENSIS</i>	0	0	0	0	0	0	0
UNID HAKE	0	0	0	0	0.02	0.12	0.42
OTHER DEMESAL	0.02	0.06	0.22	0.03	0.42	3.22	0
UNID FISH	0.01	0.02	0.05	0.25	0.50	0.33	0.42
DAILY RATION	0.7	1.6	3.0	3.1	10.9	25.0	68.6

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10-20	20-30	30-40	40-50	50-60	60-70	70+
CRUSTACEA	18	19	18	29	37	72	78
CEPHALAPODS	-	79	58	51	57	51	95
LIGHTFISH	22	22	34	53	-	106	-
MYCTOPHIDAE	23	25	27	28	47	66	79
ROUND HERRING	-	-	-	-	-	-	-
ANCHOVY	-	-	133	100	-	-	-
OTHER PELAGIC	-	-	79	68	-	66	-
<i>M. PARADOXUS</i>	-	57	72	86	40	36	67
<i>M. CAPENSIS</i>	-	-	-	-	-	-	-
UNID HAKE	-	-	-	-	97	104	101
OTHER DEMESAL	96	60	59	81	61	47	-
UNID FISH	67	37	42	62	35	51	53
DAILY RATION	12	12	15	23	23	22	48

* Ratio of empty to non-empty stomachs is the same in everted and non-everted cases.

(Table 2.12 Continued)

Assumption (ii)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	0.38	0.69	0.76	0.43	0.19	0	11.05
CEPHALAPODS	0	0.01	0.34	0.38	1.38	2.04	3.70
LIGHTFISH	0.08	0.23	0.20	0.01	0	0.06	0
MYCTOPHIDAE	0.14	0.21	0.39	0.21	0.22	0.51	0.40
ROUND HERRING	0	0	0	0	0	0	0
ANCHOVY	0	0	0.01	0.07	0	0	0
OTHER PELAGIC	0	0	0.02	0.14	0	0.25	0
<i>M. PARADOXUS</i>	0	0.04	0.04	0.25	2.26	3.86	5.83
<i>M. CAPENSIS</i>	0	0	0	0	0	0	0
UNID HAKE	0	0	0	0	0.01	0.04	0.13
OTHER DEMESAL	0.02	0.05	0.14	0.02	0.17	1.02	0
UNID FISH	0.01	0.01	0.03	0.13	0.20	0.10	0.13
DAILY RATION	0.6	1.2	1.9	1.7	4.4	7.9	21.2

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	18	19	19	31	39	-	80
CEPHALAPODS	-	60	57	51	57	51	98
LIGHTFISH	21	22	36	59	-	105	-
MYCTOPHIDAE	23	26	27	30	46	65	81
ROUND HERRING	-	-	-	-	-	-	-
ANCHOVY	-	-	86	98	-	-	-
OTHER PELAGIC	-	-	76	68	-	65	-
<i>M. PARADOXUS</i>	-	54	80	88	41	36	64
<i>M. CAPENSIS</i>	-	-	-	-	-	-	-
UNID HAKE	-	-	-	-	79	93	102
OTHER DEMESAL	84	55	61	65	61	49	-
UNID FISH	59	57	45	64	38	52	57
DAILY RATION	13	12	15	24	24	23	50

* All everted stomachs are empty

(Table 2.12 Continued)

Assumption (iii)* - Estimates

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	0.45	0.99	1.48	1.28	0.71	0.01	69.30
CEPHALAPODS	0	0.01	0.66	1.13	5.11	10.43	23.22
LIGHTFISH	0.10	0.34	0.39	0.03	0	0.32	0
MYCTOPHIDAE	0.16	0.31	0.77	0.62	0.81	2.62	2.49
ROUND HERRING	0	0	0	0	0	0	0
ANCHOVY	0	0	0.02	0.21	0	0	0
OTHER PELAGIC	0	0	0.03	0.43	0	1.28	0
<i>M. PARADOXUS</i>	0	0.05	0.09	0.75	8.36	19.72	36.54
<i>M. CAPENSIS</i>	0	0	0	0	0	0	0
UNID HAKE	0	0	0	0	0.03	0.19	0.81
OTHER DEMESAL	0.02	0.07	0.28	0.05	0.62	5.19	0
UNID FISH	0.01	0.02	0.07	0.40	0.75	0.53	0.82
DAILY RATION	0.7	1.8	3.8	4.9	16.4	40.3	133.2

C.V.s

PREY SPECIES GROUP	LENGTH-CLASS (cm)						
	10- 20	20- 30	30- 40	40- 50	50- 60	60- 70	70+
CRUSTACEA	18	19	17	28	36	12*	77
CEPHALAPODS	-	88	58	51	56	50	89
LIGHTFISH	20	21	34	54	-	105	-
MYCTOPHIDAE	23	25	27	26	46	63	80
ROUND HERRING	-	-	-	-	-	-	-
ANCHOVY	-	-	83	97	-	-	-
OTHER PELAGIC	-	-	100	64	-	65	-
<i>M. PARADOXUS</i>	-	63	71	83	38	35	61
<i>M. CAPENSIS</i>	-	-	-	-	-	-	-
UNID HAKE	-	-	-	-	99	104	109
OTHER DEMESAL	98	56	58	75	64	45	-
UNID FISH	69	41	38	60	36	48	50
DAILY RATION	12	12	14	20	19	19	39

* No everted stomachs are empty

Table 2.13 : Estimates of daily ration as a percentage of body mass for three assumptions concerning the treatment of everted stomachs.

Length-class (cm)	Average Mass (gm)	Daily ration (%)		
		(i)	(ii)	(iii)
<i>M. capensis</i>				
10 - 20	29	3.1	2.7	3.1
20 - 30	109	2.0	1.6	2.3
30 - 40	292	2.0	1.4	2.3
40 - 50	709	2.2	1.5	2.4
50 - 60	1294	2.7	1.5	3.3
60 - 70	2110	2.5	1.4	3.0
70+	3632	2.9	1.8	3.4
<i>M. paradoxus</i>				
10 - 20	39	1.8	1.6	1.8
20 - 30	123	1.3	1.1	1.4
30 - 40	267	1.1	0.7	1.4
40 - 50	644	0.5	0.2	0.8
50 - 60	1153	0.9	0.4	1.3
60 - 70	1957	1.3	0.2	2.1
70+	3351	2.0	0.6	4.0

Table 2.14 : Annual consumption estimates in grammes per day for the two hake species. Results for "pooled", seasonally disaggregated, and geographically disaggregated analyses are presented. '-' indicates a lack of data. All results were obtained by assuming that the ratio of empty to non-empty stomachs is the same in everted and non-everted cases [assumption i)]. C.V. estimates (expressed as percentages) are given in parentheses.

a) *M. capensis*

Length-class	Combined	Summer	Winter
10 - 20	0.9 (12)	0.8 (28)	0.9 (18)
20 - 30	2.3 (15)	1.8 (28)	2.4 (19)
30 - 40	5.9 (10)	5.3 (19)	6.2 (15)
40 - 50	15.3 (12)	18.5 (20)	13.3 (15)
50 - 60	35.2 (11)	41.7 (20)	31.3 (18)
60 - 70	54.5 (17)	61.9 (33)	50.2 (23)
70+	106.3 (21)	81.4 (35)	119.3 (24)
Length-class	North of 32 ⁰ S	32 ⁰ S - 34 ⁰ S	South of 34 ⁰ S
10 - 20	0.8 (13)	1.6 (30)	- (-)
20 - 30	2.4 (20)	1.8 (24)	2.6 (35)
30 - 40	5.9 (14)	5.5 (22)	5.9 (19)
40 - 50	16.7 (15)	15.2 (35)	13.1 (16)
50 - 60	34.5 (19)	39.8 (28)	33.7 (19)
60 - 70	63.5 (25)	80.0 (33)	29.0 (24)
70+	58.5 (85)	52.8 (59)	125.7 (20)

(Table 2.14 Continued)

b) *M. paradoxus*

Length-class	Combined	Summer	Winter
10 - 20	0.7 (12)	0.4 (18)	0.8 (13)
20 - 30	1.6 (12)	2.1 (28)	1.4 (14)
30 - 40	3.0 (15)	4.0 (21)	2.4 (20)
40 - 50	3.1 (23)	3.8 (27)	2.7 (30)
50 - 60	10.9 (23)	15.1 (31)	7.7 (38)
60 - 70	25.0 (22)	21.1 (40)	27.2 (30)
70+	68.6 (48)	2.2 (98)	84.4 (45)

Length-class	North of 32°S	32°S - 34°S	South of 34°S
10 - 20	0.8 (12)	0.5 (17)	0.7 (25)
20 - 30	2.0 (14)	1.3 (35)	0.8 (27)
30 - 40	2.0 (25)	4.7 (28)	4.3 (30)
40 - 50	2.9 (27)	4.7 (48)	1.8 (61)
50 - 60	10.9 (29)	20.4 (52)	0.7 (54)
60 - 70	32.6 (33)	23.0 (54)	16.7 (36)
70+	1.8 (118)	32.7 (107)	112.4 (47)

Table 2.15 : Estimates of the annual consumption of food by hake in '000 tons and their percentage C.V.s for each prey species group. The designations (i)-(iii) refer to the three assumptions concerning the treatment of everted stomachs.

(a) *M. capensis* - hake numbers at length from August 1988 cruise
 "Pooled" analysis
 [Biomass - *M. capensis* - 49.3 ± 10.2 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	72	52	80	11	11	11
CEPHALAPODS	37	24	42	27	29	27
LIGHTFISH	1	1	1	51	39	56
MYCTOPHIDAE	8	6	9	32	32	31
ROUND HERRING	18	11	21	23	24	22
ANCHOVY	43	30	48	17	17	17
OTHER PELAGIC	13	8	14	33	35	34
<i>M. PARADOXUS</i>	99	61	113	14	15	14
<i>M. CAPENSIS</i>	36	23	42	35	37	33
UNID HAKE	3	2	4	36	37	30
OTHER DEMESAL	70	43	80	17	17	17
TOTAL	404	264	458	5.9	6.3	5.6

Geographically disaggregated analysis
 [Biomass - *M. capensis* - 41.1 ± 7.7 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	54	38	61	12	12	11
CEPHALAPODS	36	22	42	30	32	30
LIGHTFISH	1	1	1	38	28	42
MYCTOPHIDAE	8	5	9	31	34	32
ROUND HERRING	17	10	20	28	33	27
ANCHOVY	24	17	26	14	14	14
OTHER PELAGIC	13	8	15	31	33	32
<i>M. PARADOXUS</i>	101	64	115	14	15	14
<i>M. CAPENSIS</i>	30	19	36	30	31	29
UNID HAKE	3	2	4	47	46	41
OTHER DEMESAL	53	32	61	19	18	21
UNID FISH	3	3	3	37	29	41
TOTAL	343	221	393	6.5	7.1	5.9

(Table 2.15 Continued)

(b) *M. capensis* - hake numbers at length from July 1989 cruise
 "Pooled" analysis
 [Biomass - *M. capensis* - 358.7 ± 81.3 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	1266	1022	1337	11	11	10
CEPHALAPODS	231	174	252	47	49	48
LIGHTFISH	49	41	51	50	51	49
MYCTOPHIDAE	83	63	90	41	42	40
ROUND HERRING	46	31	53	30	31	29
ANCHOVY	556	440	594	22	23	21
OTHER PELAGIC	41	28	47	35	35	33
<i>M. PARADOXUS</i>	342	239	382	26	28	25
<i>M. CAPENSIS</i>	170	122	190	81	84	81
UNID HAKE	15	11	16	46	46	47
OTHER DEMESAL	352	258	388	29	30	29
UNID FISH	74	60	78	66	70	64
TOTAL	3225	2489	3478	7.6	8.1	7.3

Geographically disaggregated analysis

[Biomass - *M. capensis* - 235.4 ± 37.5 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	804	631	855	11	11	11
CEPHALAPODS	178	127	198	51	50	51
LIGHTFISH	30	25	32	48	48	46
MYCTOPHIDAE	68	48	76	36	36	36
ROUND HERRING	39	27	44	30	32	30
ANCHOVY	206	180	215	16	17	15
OTHER PELAGIC	35	25	40	32	33	32
<i>M. PARADOXUS</i>	270	183	305	21	22	21
<i>M. CAPENSIS</i>	133	91	151	59	60	58
UNID HAKE	12	9	13	75	68	81
OTHER DEMESAL	201	138	223	24	25	24
UNID FISH	45	37	46	62	63	62
TOTAL	2021	1521	2198	7.9	8.1	7.7

(Table 2.15 Continued)

(c) *M. paradoxus* - hake numbers at length from August 1988 cruise
 "Pooled" analysis
 [Biomass - *M. paradoxus* - 282.8 ± 79.6 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	414	289	504	13	12	14
CEPHALAPODS	123	61	181	25	28	24
LIGHTFISH	111	83	126	17	17	16
MYCTOPHIDAE	155	107	188	15	16	15
ROUND HERRING	0	0	0	-	-	-
ANCHOVY	6	3	8	73	78	85
OTHER PELAGIC	13	7	20	49	48	48
<i>M. PARADOXUS</i>	126	54	194	23	25	23
<i>M. CAPENSIS</i>	0	0	0	-	-	-
UNID HAKE	1	0	1	48	-	87
OTHER DEMESAL	47	29	61	27	29	26
UNID FISH	24	14	33	25	25	27
TOTAL	1020	647	1316	6.5	6.8	6.6

Geographically disaggregated analysis

[Biomass - *M. paradoxus* - 228.9 ± 54.7 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	320	232	366	17	16	16
CEPHALAPODS	129	62	207	40	43	41
LIGHTFISH	59	44	69	17	17	17
MYCTOPHIDAE	206	138	249	25	25	25
ROUND HERRING	0	0	0	-	-	-
ANCHOVY	1	0	1	61	-	113
OTHER PELAGIC	4	1	5	42	93	47
<i>M. PARADOXUS</i>	147	58	252	30	31	34
<i>M. CAPENSIS</i>	0	0	0	-	-	-
UNID HAKE	1	0	3	76	-	48
OTHER DEMESAL	34	18	45	26	31	25
UNID FISH	13	7	17	19	19	20
TOTAL	914	560	1214	9.6	10.5	9.6

(Table 2.15 Continued)

(d) *M. paradoxus* - hake numbers at length from July 1989 cruise
 "Pooled analysis"
 [Biomass - *M. paradoxus* - 473.1 ± 124.0 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	709	492	870	11	11	13
CEPHALAPODS	248	126	361	26	30	25
LIGHTFISH	181	133	208	17	17	17
MYCTOPHIDAE	283	193	347	15	15	15
ROUND HERRING	0	0	0	-	-	-
ANCHOVY	10	6	15	78	71	80
OTHER PELAGIC	26	13	39	46	48	45
<i>M. PARADOXUS</i>	237	99	367	22	23	22
<i>M. CAPENSIS</i>	0	0	0	-	-	-
UNID HAKE	2	1	3	49	30	57
OTHER DEMESAL	94	56	124	29	30	28
UNID FISH	43	25	60	25	24	26
TOTAL	1833	1144	2394	6.4	6.6	6.4

Geographically disaggregated analysis

[Biomass - *M. paradoxus* - 388.2 ± 42.5 ('000 tons)]

PREY SPECIES	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEA	715	519	847	12	12	12
CEPHALAPODS	218	125	309	39	49	35
LIGHTFISH	154	115	181	18	19	18
MYCTOPHIDAE	244	163	303	15	15	15
ROUND HERRING	0	0	0	-	-	-
ANCHOVY	24	15	46	85	78	84
OTHER PELAGIC	30	17	40	41	46	41
<i>M. PARADOXUS</i>	146	61	240	29	28	30
<i>M. CAPENSIS</i>	0	0	0	-	-	-
UNID HAKE	0	0	0	-	-	-
OTHER DEMESAL	64	45	78	35	35	35
UNID FISH	37	24	51	27	28	25
TOTAL	1632	1084	2095	7.7	8.7	7.2

Table 2.16 : Estimates of annual consumption of food by hake in '000 tons and the associated percentage C.V.s for each prey species group obtained using data for the July 1989 cruise only. The data are pooled across geographic subdivisions. The designation (i) to (iii) refer to the three assumptions concerning the treatment of everted stomachs.

(a) *M. capensis*

PREY SPECIES GROUP	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEANS	871	715	919	16	16	16
CEPHALOPODS	536	465	563	75	77	74
LIGHTFISH	68	55	71	47	47	48
MYCTOPHIDS	21	17	23	67	65	65
ROUND HERRING	34	27	37	53	52	54
ANCHOVY	314	263	340	52	55	50
OTHER PELAGIC	41	29	49	32	34	31
<i>M. PARADOXUS</i>	808	683	868	46	48	45
<i>M. CAPENSIS</i>	499	435	527	86	89	85
UNID HAKE	11	9	12	91	100	92
OTHER DEMERSAL	766	669	808	48	49	47
UNID FISH	105	83	111	83	82	82
TOTAL	4074	3450	4328	17.2	18.6	16.3

(b) *M. paradoxus*

PREY SPECIES GROUP	ESTIMATE ASSUMPTION			C.V. ASSUMPTION		
	(i)	(ii)	(iii)	(i)	(ii)	(iii)
CRUSTACEANS	192	152	246	21	20	23
CEPHALOPODS	195	89	425	53	51	50
LIGHTFISH	314	267	351	30	30	30
MYCTOPHIDS	387	315	455	32	33	30
ROUND HERRING	0	0	0	0	0	0
ANCHOVY	41	28	73	71	64	84
OTHER PELAGIC	2	2	2	100	100	150
<i>M. PARADOXUS</i>	142	64	247	51	50	52
<i>M. CAPENSIS</i>	0	0	0	0	0	0
UNID HAKE	0	0	0	0	0	0
OTHER DEMERSAL	39	33	44	69	70	68
UNID FISH	20	12	34	35	33	38
TOTAL	1332	962	1877	13.6	14.0	12.9

CHAPTER 3 - HISTORICAL BACKGROUND TO THE HAKE FISHERY OFF THE SOUTH AFRICAN WEST COAST

The fishery for Cape hake off South Africa commenced at the turn of the century. Catches increased steadily and all four stocks off South Africa and Namibia had become biologically overexploited by the mid-1970s, with the South African fishery almost uneconomical at that time. The International Commission for the South East Atlantic Fisheries (ICSEAF) was established in 1972 to investigate and control the fisheries off southern Africa. Since 1978, the hake TACs for the South African west coast were set with the aim of rebuilding the stock, generally by applying an $f_{0.1}$ or $f_{0.2}$ harvesting strategy. Since then, catch rates have been increasing, indicating resource recovery, although this could be a consequence of improvements in vessel efficiency. From 1990, management of this resource has been based on $f_{0.2}$ strategy TACs provided by the Butterworth-Andrew (B₁₉₁₇=K; Schaefer form) observation error estimator. In deciding upon this strategy to set TACs for this resource, cognizance has also been taken of the results of an *ad hoc* tuned VPA assessment.

3.1. Development of the fishery

The fishery for Cape hake in the southeast Atlantic (of which the hake fishery off the South African west coast is a major component) is one of the largest in the world. For example, in 1983 the catch of Cape hake was the 20th largest of any fish species worldwide (Csirke 1988). The total catches taken from the southeast Atlantic are listed Table 3.1, and those from 1955 illustrated in Figure 3.1. The importance of the hake fishery in South African waters is such that it has constituted well over 70% by landed mass of all demersal fish species caught by the South African fishery in recent years [R.W. Leslie, SFRI, pers. comm].

The bottom-trawl fishery off South Africa commenced at the turn of the century, although it was directed at sole rather than at hake at that time (Payne 1988). In the early years of the fishery, the bulk of the hake catch taken must have been *M. capensis* rather than *M. paradoxus*,

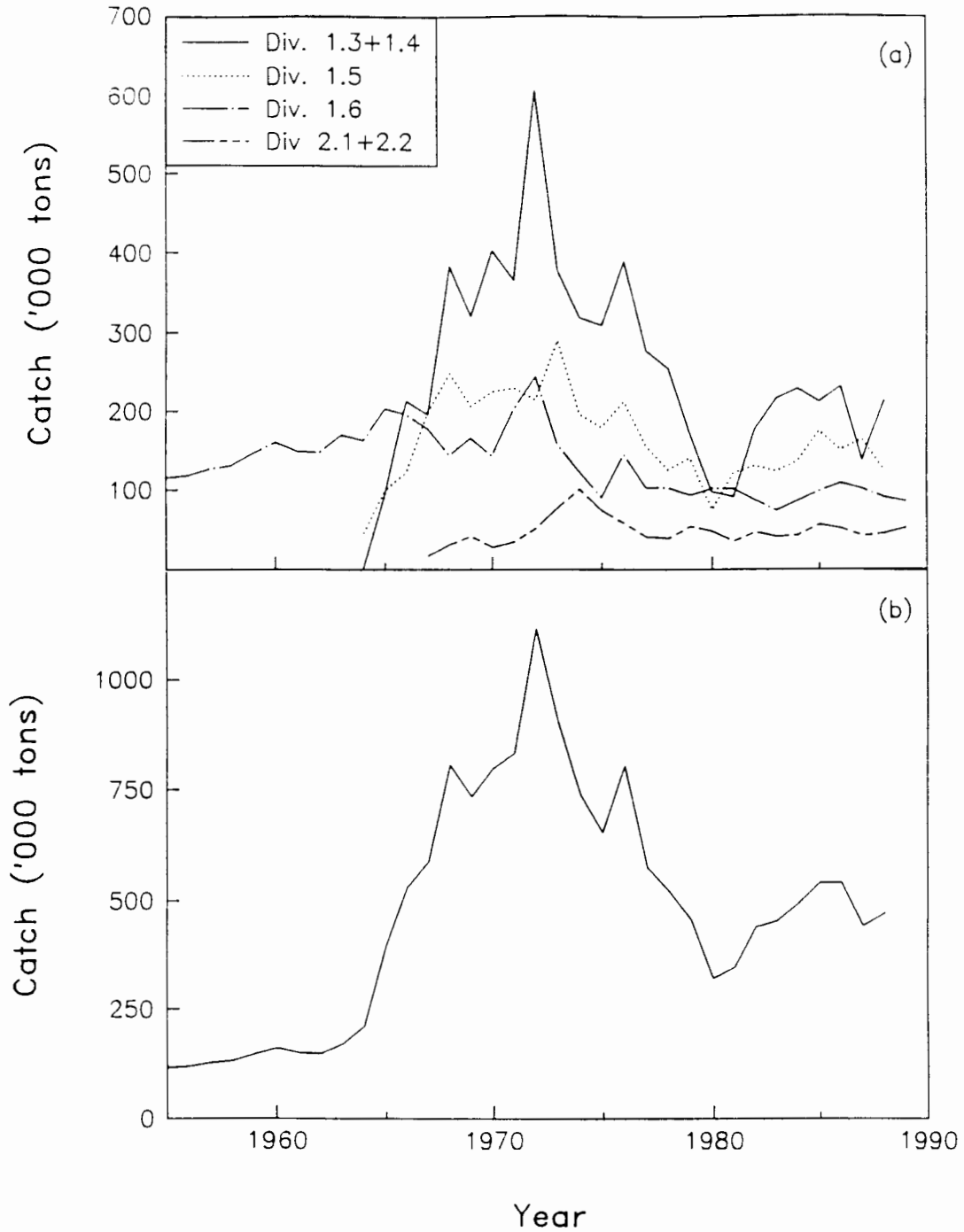


Figure 3.1: Annual catches from the hake resources in the ICSEAF Convention Area since 1955 (a) by Division, and (b) total.

because the fishing vessels only fished close to the Cape Peninsula (Jones 1974a). The hake fishery grew gradually [only by 1917 did the catch exceed 1000t - Chalmers (1976)], catches exceeded those of kabeljou (*Johnius holoepidotus*) only in the early 1920s [Table 3 of Chalmers (op. cit.)]. Scott (1950) reports that, before 1950, trawlers operated only up to 65km off shore, mainly in the region between Cape Point (34°50'S) and Saldana Bay (33°S) (Figure 2.1). Annual hake catches increased steadily between the two world wars and, by 1945, had already reached 30,000t. After 1945, the fishery expanded even more rapidly, and a catch exceeding 100,000t was achieved for the first time in 1954.

In 1962, foreign vessels joined the fishery, and the fishing area expanded from the Cape Peninsula and Agulhas Bank grounds to cover virtually the entire continental shelf from the northern boundary of Namibia to East London (see Figure 2.1). At first, the fishery concentrated on areas of high abundance, extending its range only later as stock densities in these areas decreased. The catches of Cape hake off southern Africa have been dominated by those of three countries, namely South Africa, Spain and the USSR. Prior to the declaration of a 200Nm Exclusive Fishing Zone (EFZ) by South Africa in 1977, up to 50% of the hake catch off the South African west coast was being taken by foreign fleets.

The increasing fishing effort made a substantial impact on stock densities, and CPUE indices in all the ICSEAF Divisions dropped markedly between the years 1965 and 1972. Off the South African west coast, for example, the CPUE had declined by over 50%, indicating (in a

Schaefer model context) that the stock was biologically overexploited (Figure 3.2 and Table 3.2). By this time, catch rates had reached levels that rendered the South African fishery almost uneconomic (Payne 1988).

3.2 Management

As a result of the rapid increase in fishing effort, the South African Division of Sea Fisheries (now the Sea Fisheries Research Institute - SFRI) initiated a research programme in 1966 which concentrated on the collection of catch-effort statistics for hake. In 1972, ICSEAF (the International Commission for the South East Atlantic Fisheries) was established to investigate and control exploitation of the fish resources in the southeast Atlantic. This large area was divided into eight subareas (some of which are shown in Figure 2.1), of which Subareas 1 and 2 correspond to the regions which provide the bulk of the Cape hake catch. For management purposes, each Subarea was further divided into Divisions. The hake populations in four sets of these Divisions 1.3+1.4, 1.5, 1.6 (the study area for this thesis) and 2.1+2.2 were assumed to be separate, and in consequence they have been managed independently. The boundary between Divisions 1.6 and 2.1+2.2 was placed on the basis of a hiatus in catch rates (A.I.L. Payne, SFRI, pers. commn). [Recent work by Payne *et al.* (1988) suggests, however, that the stocks in Divisions 1.5 and 1.6 may be related, because some of the recruitment of *M. capensis* into Division 1.6 appears to originate from Division 1.5.]

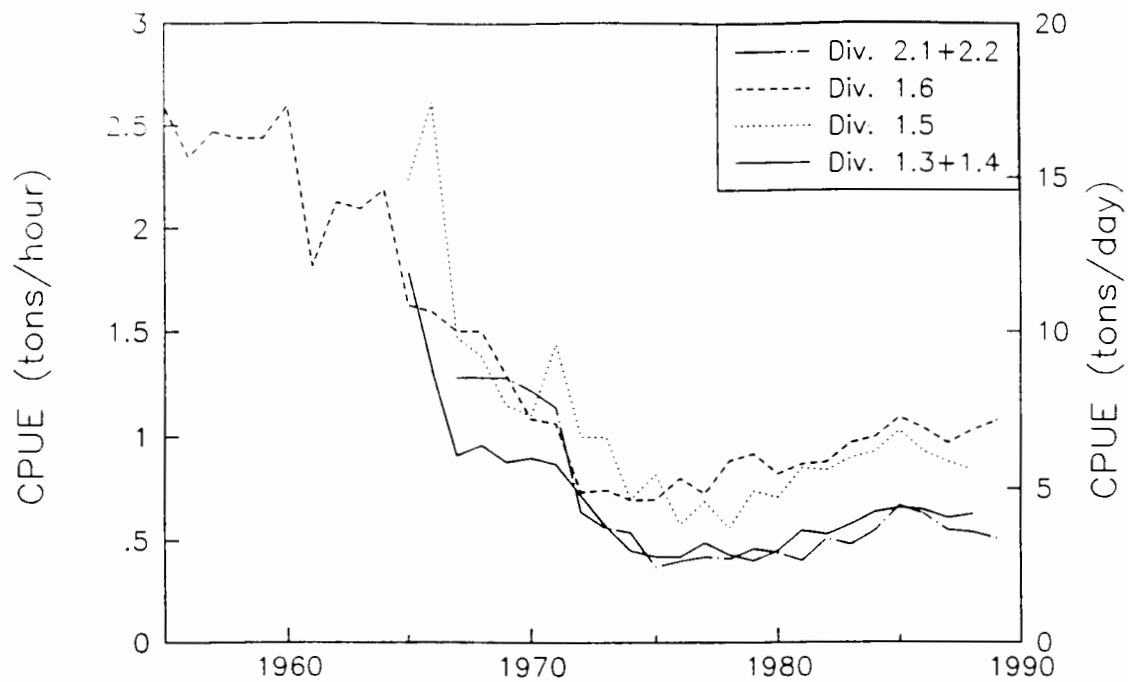


Figure 3.2: Catch rate data since 1955 for the hake resources in the ICSEAF Convention Area.

After South Africa declared a 200Nm Exclusive Fishing Zone in November 1977, TACs for Division 1.6 and Subarea 2 have been set by the South African authorities, taking account of recommendations made by ICSEAF. The goal of management in Division 1.6 has been to increase the stock biomass and catch rate.

The Cape hake resources in the southeast Atlantic have been managed on a species-aggregated basis for three main reasons:

- a) in most Divisions, one of the two hake species dominates the catch (e.g. in the study area, approximately 90% of the catch (by number) is made up of *M. paradoxus* (Botha 1980)),
- b) commercial catches are not separated into species, and
- c) there is little reliable quantitative knowledge of the biological interaction between the two hake species.

Botha (1980) states that "contrary to the often held view that these species should be separated for stock assessment purposes, they may be conveniently combined for this purpose. The resultant negligible loss of accuracy would most likely be more than compensated for by saving on elaborate research methods aimed at separating the two species in commercial catches". The investigation of the validity of this statement constitutes a major objective of this study.

The first attempt at managing the hake fishery off the South African coast was the decision to increase of the minimum mesh size from 2.1/2 inches to 4 inches in 1938 (Scott 1950). At that time, over 50% of the catch was discarded. In its initial attempt to reduce fishing effort and to provide some protection for juveniles, ICSEAF recommended a

minimum stretched-mesh size. For hake this was initially 102mm, but after further research had indicated that this should be increased, a recommendation for a minimum stretched-mesh size of 110mm was adopted in July 1977. However, Davies *et al.* (1980) concluded from yield-per-recruit analyses that this restriction probably had little or no effect in terms of altering the age composition of the catch. A system of international inspection of the hake fishery was adopted by ICSEAF in June 1976, and TACs for hake were set from 1978 onwards.

From 1977 to 1983, the TACs recommended by ICSEAF were based on the use of Gulland's (1961) effort-averaging procedure in conjunction with Fox's (1970) formulation of the surplus production function (Table 3.3). It is possible that the Fox form of the surplus production function was chosen over other alternatives because it provided more accurate fits to the data available at the time (Andrew 1986). In order to allow the stocks to recover without restricting the industry excessively, a policy was implemented which aimed at maintaining catches below annual sustainable yields, generally by use of the $f_{0.1}$ harvesting strategy (see Chapter 9). However, after it was argued (ICSEAF 1983) that the Gulland procedure was producing unsatisfactory fits to the data, and that allowance should be made for dynamic effects in the estimation of TACs, a procedure of recommending TACs based on the average results of three "dynamic" production models was subsequently employed. As a result (presumably) of the reduced catches from 1977, catch rates began to improve over the whole management area (Figure 3.2). However, this increase may be due in part to undetected

increases in vessel efficiency, as only crude analyses of power factors have ever been carried out.

ICSEAF was dissolved in 1990 (following the independence of Namibia), and since then management advice for the hake resources off South Africa has been provided by South African scientists only. The TACs recommended for these resources have been based on the $f_{0.2}$ harvesting strategy. The estimator used to provide the estimates needed to apply this strategy has been the Butterworth-Andrew observation error estimator (the Schaefer form, with $B_{1917}=K$ for the west coast stock) (Butterworth and Andrew 1984) (see Chapter 6). The $f_{0.2}$ harvesting strategy was selected over alternatives for the following reasons (SFRI 1989).

- a) Concerns about possible undetected increases in vessel efficiency.
- b) The fact that the rate of recovery over the period 1983 - 1988 estimated by the model exceeded that inferred directly from the CPUE and survey biomass data available at that time.
- c) The results of Punt (1988), who had concluded that the estimator concerned provided positively biased $f_{0.1}$ harvesting strategy TACs for the hake stock off northern Namibia.
- d) The fact that the resource was estimated to be biologically overexploited (i.e. still below its MSY level).
- e) Alternative TAC estimates provided by an *ad hoc* tuned VPA assessment based on the Laurec-Shepherd tuning algorithm (see

Chapter 7), which were substantially lower than those provided by the production model-estimation procedure.

Table 3.1: Total catches of Cape hake in the ICSEAF Convention Area
 [source: R.W. Leslie, SFRI, pers. commn]. Units '000 tons.

YEAR	DIVISIONS 1.3+1.4	DIVISION 1.5	DIVISION 1.6	DIVISIONS 2.1+2.2	TOTAL
1917			1.000		1.000
1918			1.100		1.100
1919			1.900		1.900
1920			0.000		0.000
1921			1.300		1.300
1922			1.000		1.000
1923			2.500		2.500
1924			1.500		1.500
1925			1.900		1.900
1926			1.400		1.400
1927			0.800		0.800
1928			2.600		2.600
1929			3.800		3.800
1930			4.400		4.400
1931			2.800		2.800
1932			14.300		14.300
1933			11.100		11.100
1934			13.800		13.800
1935			15.000		15.000
1936			17.700		17.700
1937			20.200		20.200
1938			21.100		21.100
1939			20.000		20.000
1940			28.600		28.600
1941			30.600		30.600
1942			34.500		34.500
1943			37.900		37.900
1944			34.100		34.100
1945			29.200		29.200
1946			40.400		40.400
1947			41.400		41.400
1948			58.800		58.800
1949			57.400		57.400
1950			72.000		72.000
1951			89.500		89.500
1952			88.800		88.800
1953			93.500		93.500
1954			105.400		105.400

(Table 3.1 Continued)

YEAR	DIVISIONS 1.3+1.4	DIVISION 1.5	DIVISION 1.6	DIVISIONS 2.1+2.2	TOTAL
1955			115.400		115.400
1956			118.200		118.200
1957			126.400		126.400
1958			130.700		130.700
1959			146.000		146.000
1960			159.900		159.900
1961			148.700		148.700
1962			147.600		147.600
1963			169.500		169.500
1964	1.815	46.037	162.300		210.152
1965	93.510	99.690	203.000		396.200
1966	212.444	122.183	195.000		529.627
1967	195.032	199.413	176.700	17.340	588.485
1968	382.712	247.680	143.600	31.370	805.362
1969	320.430	206.227	165.100	41.700	733.457
1970	402.467	224.731	142.500	27.800	797.498
1971	365.557	229.658	202.000	34.500	831.715
1972	606.084	214.026	243.933	51.388	1115.431
1973	377.642	290.323	157.782	77.356	903.103
1974	318.836	195.722	123.000	100.909	738.467
1975	309.374	178.834	90.270	73.835	652.313
1976	389.020	212.025	144.547	57.670	803.262
1977	276.901	154.582	102.378	40.472	574.333
1978	254.251	125.139	102.142	38.889	520.421
1979	170.006	140.169	92.904	53.831	456.910
1980	97.181	74.667	101.678	47.571	321.097
1981	90.523	121.011	101.058	35.138	347.730
1982	176.532	130.546	86.750	46.826	440.654
1983	216.181	123.409	74.061	41.174	454.825
1984	228.672	136.321	86.407	43.196	494.596
1985	212.177	174.007	98.536	56.223	540.943
1986	231.179	150.010	108.370	51.167	540.726
1987	136.942	163.307	100.640	41.826	442.715
1988	212.000	124.000	90.131	44.969	471.100
1989	-	-	84.896	51.772	-
1990	-	-	78.724	58.279	-

- Entries marked '-' denote data which are currently unavailable.

Table 3.2: Catch rate (CPUE) data for Cape hake in the ICSEAF Convention Area [source: R.W. Leslie, SFRI, pers. commn].

YEAR	DIVISIONS 1.3+1.4 ESP OTB-7 t/h	DIVISION 1.5 ESP OTB-7 t/h	DIVISION 1.6 ZAF t/d	DIVISIONS 2.1+2.2 ZAF t/h
1955			17.31	
1956			15.64	
1957			16.47	
1958			16.26	
1959			16.26	
1960			17.31	
1961			12.09	
1962			14.18	
1963			13.97	
1964			14.60	
1965	1.78	2.24	10.84	
1966	1.31	2.62	10.63	
1967	0.91	1.47	10.01	
1968	0.96	1.38	10.01	
1969	0.88	1.15	8.62	1.28
1970	0.90	1.10	7.23	1.22
1971	0.87	1.44	7.09	1.14
1972	0.72	1.00	4.90	0.64
1973	0.57	1.00	4.97	0.56
1974	0.45	0.70	4.65	0.54
1975	0.42	0.82	4.66	0.37
1976	0.42	0.58	5.35	0.40
1977	0.49	0.69	4.84	0.42
1978	0.43	0.56	5.90	0.41
1979	0.40	0.74	6.13	0.46
1980	0.45	0.71	5.48	0.44
1981	0.55	0.85	5.81	0.40
1982	0.53	0.84	5.87	0.51
1983	0.58	0.90	6.49	0.48
1984	0.64	0.93	6.67	0.55
1985	0.66	1.03	7.29	0.67
1986	0.65	0.93	6.93	0.63
1987	0.61	0.88	6.46	0.55
1988	0.63	0.84	6.88	0.54
1989	-	-	7.18	0.51
1989	-	-	7.29	0.60

- Entries marked '-' denote data which are currently unavailable.

CHAPTER 4 - DATA AVAILABLE TO MODEL-ESTIMATION PROCEDURES

There are five types of data available for the assessment of the hake resource off the South African west coast. The catch-by-mass data are available from 1917, but may be biased because of the discarding of small fish, and imprecise because of the use of logbook data to estimate pre-1954 catches and to disaggregate catches made over both the west and south coast grounds. CPUE data are available from 1955. However, the adequacy of the power factors used in the calculation of these data is currently a cause for concern. Reasonably reliable catch-at-age data are available from 1978, but these data may be biased because of inadequate sampling, discarding and errors in age-reading. Biomass indices, as well as estimates of the age- and size-composition of the stock, are available from research cruises which have been carried out since June 1983. These surveys also provide biological data. Because of the effects of fish passing through and over the net, (*inter alia*), these biomass indices cannot be regarded as measures of absolute abundance.

The data which are available to assess the hake resource off the South African west coast can be divided into five groups:

- a) catch-by-mass data,
- b) CPUE/effort data,
- c) size/age-structure data,
- d) survey biomass data, and
- e) biological data.

4.1 Catch-by-mass data

As noted in Chapter 3, the catch-by-mass data collected have never distinguished between the two hake species. The reported catch-by-mass for year y , C_y , is an estimate of the total mass of fish caught during that year. The nominal (or live) mass, rather than landed mass (which corresponds to headed, filleted and/or gutted fish), has been reported. To calculate the nominal mass from the landed mass, correction factors which depend on the landed size, and whether fish have been headed,

filleted or gutted, have been applied. A general correction factor of 1.46 (Chalmers 1976) was applied when the breakdown of the landed catch by size was not known. This correction factor is an average of the correction factors for a number of different methods of heading, filleting and gutting. The total catch-by-mass estimates are thus likely to be subject to some error due to changes in methods of heading and gutting.

After 1954, the catches-by-mass have been estimated by taking all fish landed into account, whereas the catches-by-mass for the years prior to 1954 were estimated from data reported in logbooks. As this data source gives only the skippers estimates of the mass of each haul (the fish are only weighed once the vessel lands its catch), these estimates of the catch-by-mass are likely to be less precise than the estimates for years after 1954.

A far more serious problem with the catch-by-mass data is the practice of discarding, which has been practised because until relatively recently, there has not been a market for small fish. The catches up to 1972 were increased by 39% to take into account the bias caused by discarding during that period (ICSEAF 1978). However, it is highly unlikely that this correction factor is appropriate for all those years. For example, in years of good recruitment to the fishery, the correction factor will be too small, and in years of bad recruitment it will be too large. Furthermore, it is almost certain that the rate of discarding decreased steadily over time, so that assuming a time-independent correction factor will lead to bias in the estimates of catch-by-mass.

Catches taken during trips off the South African coast as a whole need to be divided into those taken off the west coast and those off the south coast. This problem arises because TACs are set for the whole of South Africa's waters, and some vessels fish off both coasts during one trip. In order to determine the component of the landed catch which was caught off the west coast, the estimates of the mass of hake in each haul made by the vessel skippers and entered in the logbooks are used. As these are only estimates, the catches-by-mass are subject to further imprecision.

In any case, the reported catch always tends to underestimate the total mortality due to fishing, because the mass of fish which die as a result of passing into the net and subsequently slipping out through it, is not known.

When performing hake assessments, it is always (implicitly) assumed that the combined effect of these four factors is sufficiently small to justify ignoring any bias resulting from them. The estimates of the catches-by-mass for the west coast are given in Tables 4.1 (1955-1989) and 4.2 (1917-1954). Table 4.3 provides a breakdown of these catches-by-mass by national fleet. It is evident from this Table that the bulk of hake catches off the South African west coast in recent years has been taken by the local (South African) fleet.

4.2. CPUE/effort data

The catch-per-unit-effort (CPUE) series, which is available from 1955 (Table 4.1), is shorter than the corresponding catch series which commences in 1917. This is not particularly surprising because, in

general, the collection of effort statistics usually starts only after a fishery has developed sufficiently to warrant the research needed for scientifically based management.

The fishing effort of year y , E_y , is defined as:

$$E_y = C_y / (C/E)_y \quad (4.1)$$

where $(C/E)_y$ is the CPUE for year y , and

C_y is the total catch-by-mass taken during year y .

The CPUE for year y is calculated by dividing the catch-by-mass taken by all vessels whose fishing was directed at hake by the sum of the standard effort of those vessels. The standard effort of a particular vessel is defined as the product of its fishing power and the "time" it spent fishing (actual effort). The fishing power of a vessel is its efficiency relative to a standard (hypothetical) fishing vessel and is calculated by fitting a log-linear model to the CPUE for different vessel classes (Kimura 1981). The effort unit of trawler days (the number of days on which a vessel fishes during the year) was selected because only these data are available from 1955. In order for trawler days to provide an unbiased relative estimate of the total number of hours trawled, the steaming time to the fishing grounds should be minimal. Further, the assumption needs to be made that, over the period for which CPUE data are desired, the same fraction of the day has been spent towing the trawl nets. These two assumptions do not seem particularly unreasonable, because the nets are usually dropped after only a few hours steaming from harbour and because three trawls per day have been made each day throughout most of the history of the fishery.

The effort calculated will also be biased if power factors are incorrectly calculated or not updated sufficiently frequently. Whether the existing estimates of power factors are adequate is currently an area of concern (Butterworth 1988). Punt and Butterworth (1989b) have shown that the effect of only a small undetected increase in efficiency has a substantial impact on the estimates of current resource status obtained from the Butterworth-Andrew (Schaefer form; $B_1=K$) observation error estimator.

4.3 Size/Age-structure data

Extensive collection of catch length composition data has been carried out off the South African west coast (>20000 fish per year in recent years). If sampling was a multinomial process, this would imply that summary statistics for these data would be very precisely determined. However, there is an aspect of the sampling methodology applied which is likely to lead to some bias in the estimates of the proportion of the catch falling into each length-class. Sampling is only carried out after return to port, so that the length composition data reflect only the post-discard catches. Length frequency data collected at sea, which samples the pre-discard catches (and could as a result provide some information on the proportion of the catch which has been discarded) are few. When several such data sets have been collected, they were usually obtained on the same trip and hence reflect the length frequencies from individual hauls by the same vessel. As a single vessel is unlikely to be representative of the behaviour of the whole fleet, these data are of little use. Commercial length frequencies for the period 1981-1989 are given in Table 4.4.

The collection of otoliths for ageing purposes (Botha 1971) and catch length frequency data permits the breakdown of the total catch-by-mass into catch-at-age estimates. Appendix 4.A gives the algorithm currently used to do this, as well as some suggestions on how this algorithm could be improved. Catch-at-age data obtained using this "improved" catch-at-age estimator are given in Table 4.5. Although, in principle, these data should be more useful than the age-aggregated total catch-by-mass data, they do suffer from a number of problems.

- a) Allowance for discards. As noted above, the catches-by-mass for the South African west coast up to 1972 have been increased by 39% to allow for the effects of discarding. It appears that the associated correction to catch-at-age data was effected by similarly increasing all the catches-at-age by 39%. As discarding affects primarily only the small fish, this means that, prior to 1972, recorded estimates of catches-at-age for the older fish are probably positively biased, and those of the catches-at-age for younger fish are negatively biased. Note, however, that catch-at-age data for the years prior to 1978 are not used for assessment purposes in this thesis because the age-length key for 1978 was used in calculation of catch-at-age data for all years prior to 1978 (R.W. Leslie, SFRI, pers. commn), and the practice of applying an age-length key from one year to length frequency data from another can severely distort assessment results. From 1978, separate age-length keys are available for each year.

- b) Small sample sizes. If the samples used to estimate length frequencies, mass-length relationships and age-length keys are not sufficiently large, considerable error in the catch-at-age estimates (both bias and variance) can be expected (Punt 1989a). The bias arises from the lack of data for some length-classes. This results in the number of fish allocated to those length-classes being zero. Furthermore, even with relatively large sample sizes, catch-at-age data can still be rather imprecise [particularly for the oldest and youngest ages - Punt (1989b)]. Again, this problem is not likely to be extremely serious for the resource considered in this thesis because the catch-at-age data for years prior to 1978 (when small sampling may have been a problem) are not used for assessment purposes here.
- c) Unrepresentative sampling. Much of the basic data used to construct the historical catch-at-age data may have come from samples which were all collected at the same time, from the same location or from the same vessel. This can severely bias the catch-at-age estimates if the times/areas/locations are not representative of the fishery as a whole.
- d) Ageing error. Although hake are not particularly difficult fish to age (R.W. Leslie, SFRI, pers. commn), incorrect age assignments must nevertheless occur to some extent. This will lead to errors when model-estimation procedures which assume that the catch-at-age estimates are exact (such as *ad hoc* tuned VPA) are applied.

4.4 Survey biomass data

The data obtained from research survey cruises are particularly useful for assessment purposes. This is because the survey effort is too small to perturb the fishery, the geographic and seasonal coverage are known and can be controlled, and vessel characteristics can be kept constant. This means that abundance indices obtained from these cruises are unlikely to be affected by technological changes in the fishing operation (i.e. are less subject to consequential biases as an index of abundance than is CPUE). A number of different data sets are collected from the survey cruises off the west coast of South Africa.

- a) Estimates of absolute abundance. Because the proportion of fish which lie in the path of the net but nevertheless avoid it is unknown, swept area surveys can only provide relative indices of abundance (fish can avoid the net by swimming over the top of or through it.) The indices produced are thus likely to be negatively biased because of net avoidance and not surveying the component of the population higher in the water column than the depth range swept by net. The calculation of the survey biomass indices and their standard deviations is described in Appendix 4.B. The biomass indices for the period 1983 - 1990 are given in Table 4.6. These relative abundance indices are very useful when performing assessments, because they provide an alternative index of abundance which can be used if CPUE proves unsuitable (possibly because of the effects of undetected increases in efficiency). The algorithm applied to obtain the biomass indices for the whole South African west

coast can also be used to provide these indices by depth stratum (see Table 4.7).

- b) Estimate of numbers-at-age. The surveys provide estimates of the age-composition of the stock. Although these estimates are unlikely to be absolute indices for the reasons mentioned above, they can be used as relative indices of the abundance of each age-class. The algorithm used to estimate the numbers-at-age from a survey is described in Appendix 4.B. The estimates of numbers-at-age per depth stratum per cruise for the years 1986 to 1989 are given in Tables 4.8 and 4.9.
- c) Length frequency data. The length frequency data collected during the cruises are of considerable value because the two hake species are distinguished in the collection process. Furthermore, these data are collected at each station (the length-frequencies thus sample virtually the entire stock area), and are "sea-based" (i.e. the length-frequencies reflect pre-discard catches).
- d) Biological information. The data used to determine distribution, feeding ecology, reproduction, etc. (see Chapter 2) are all obtained from the samples collected during surveys.

4.5 Biological parameters

Estimates of biological parameters such as growth and natural mortality rates are usually available from species studies (see Chapter 2). These parameters are used in some of the model-estimation procedures, e.g. to fix M for the Shepherd surplus production function (see Chapter 6), or to estimate growth rates for integrated analyses (see Chapter 8).

Table 4.1 : Total catch and CPUE data, and total effort estimates obtained therefrom, for the Cape hake stock off the west coast (Source R. Leslie, A. Badenhorst, SFRI, pers. commn).

Year	Total catch (tons)	CPUE (ZAF tons/ std day)	Total effort (ZAF standard days)
1955	115 400	17.31	6 667
1956	118 200	15.64	7 558
1957	126 400	16.47	7 675
1958	130 700	16.26	8 038
1959	146 000	16.26	8 979
1960	159 900	17.31	9 237
1961	148 700	12.09	12 299
1962	147 600	14.18	10 409
1963	169 500	13.97	12 133
1964	162 300	14.60	11 116
1965	203 000	10.84	18 727
1966	195 000	10.63	18 344
1967	176 700	10.01	17 652
1968	143 600	10.01	14 346
1969	165 100	8.62	19 153
1970	142 500	7.23	19 710
1971	202 000	7.09	28 491
1972	243 933	4.90	49 782
1973	157 782	4.97	31 747
1974	123 000	4.65	26 452
1975	89 617	4.66	19 231
1976	143 894	5.35	26 896
1977	102 328	4.84	21 142
1978	101 140	5.90	17 142
1979	92 704	6.13	15 123
1980	101 538	5.48	18 529
1981	100 678	5.81	17 328
1982	85 970	5.87	14 646
1983	73 677	6.49	11 352
1984	88 410	6.67	13 255
1985	99 590	7.29	13 661
1986	109 091	6.93	15 742
1987	104 010	6.46	16 101
1988	90 131	6.88	13 100
1989	84 896	7.18	11 824
1990	78 724	7.29	10 799

Table 4.2 : Total hake catches off the South African west coast from 1917 to 1954. Calculation of the catches was based on historical records of landed catches published by Chalmers (1976). Correction for discarding of small hake has been made by increasing the catches reported by Chalmers (op. cit.) by 39%.

YEAR	CATCH	YEAR	CATCH	YEAR	CATCH
1917	1.0	1930	4.4	1943	37.9
1918	1.1	1931	2.8	1944	34.1
1919	1.9	1932	14.3	1945	29.2
1920	-	1933	11.1	1946	40.4
1921	1.3	1934	13.8	1947	41.4
1922	1.0	1935	15.0	1948	58.8
1923	2.5	1936	17.7	1949	57.4
1924	1.5	1937	20.2	1950	72.0
1925	1.9	1938	21.1	1951	89.5
1926	1.4	1939	20.0	1952	88.8
1927	0.8	1940	28.6	1953	93.5
1928	2.6	1941	30.6	1954	105.4
1929	3.8	1942	34.5		

Table 4.3 : Catches-by-mass for the Cape hake resource off the South African west coast by national fleet. Data for 1989 and 1990 are unavailable.

YEAR	ZAF	ESP	SUN	JPN	POL	ISR	BGR	OTHERS	TOTAL
1972	52219	20015	127182	28881		2691	3247		234235
1973	25147	120083	94115	17229	21735	6300	2920	1533	289062
1974	61018	6784	36847	8059	89		5832	936	119565
1975	54991	2625	14365	12180	204		1139		85504
1976	70524	2934	22555	38565	918	3600		4055	143151
1977	64372			23892	10	1100			89374
1978	100233			587		321			101141
1979	89513			950					90463
1980	93388	3319		597					97304
1981	95139	1768		32					96939
1982	81478	1051		2					82531
1983	70344	1632		1		1700			73677
1984	79374	1703				1686			82763
1985	68355	1796		3		1804			71958
1986	104938	1877		25		2251			109091
1987	99333	1913		2		2762			104010
1988	88812	1318							90131

ZAF - South Africa

SUN - USSR

POL - Poland

BGR - Bulgaria

ESP - Spain

JPN - Japan

ISR - Israel

Table 4.4 : Length frequencies for the Cape hake resource off the South African west coast. Note that a length-class of 35cm includes hake whose lengths were measured to lie between 34.5 and 35.5cm. Units of mass are grammes.

LENGTH- CLASS (cm)	YEAR				
	1981	1982	1983	1984	1985
19	52563	30606	2791	0	0
21	211361	218707	42661	21212	779
23	475997	668049	157311	66040	4785
25	968724	1048977	476704	408211	18992
27	1364030	1343491	787626	811685	65104
29	1292008	1459081	926307	913766	190161
31	973483	1140251	1119332	1002852	502338
33	698245	811688	1133760	799727	637705
35	528046	514415	980501	660981	579696
37	383568	402740	772273	587044	520047
39	249303	244738	577364	443034	420592
41	189519	158530	391610	258436	310117
43	157487	130054	303137	209951	229187
45	103385	107202	248375	180504	158925
47	87062	105303	211086	164590	126488
49	74604	84070	173692	134709	108966
51	59889	68255	131404	102312	79839
53	47795	50903	104076	73959	72380
55	40739	37194	81665	66208	61211
57	34590	30493	60040	59843	58539
59	28243	24811	47169	51547	46807
61	20845	20816	39583	41494	37651
63	15395	16680	29773	34907	31474
65	13314	13154	25203	28760	27922
67	9965	9398	19480	23077	20869
69	8345	7410	15472	16915	20525
71	6594	5626	12818	12729	16353
73	4406	4189	8683	8956	12000
75	3063	3087	7390	6521	7444
77	2296	2184	6151	3956	4642
79	1515	1825	4557	2755	3434
81	1404	1506	2919	2257	1994
83	888	691	2292	1281	1152
85	706	557	1554	776	914
87	528	497	763	612	390
89	282	208	674	402	267
91	146	160	335	218	603
93	102	118	261	146	165
95	74	110	138	43	42
97	40	14	74	19	70
99	10	9	57	5	34

(Table 4.4 Continued)

LENGTH- CLASS	YEAR				
	1981	1982	1983	1984	1985
101	3	3	59	0	3
103	0	0	8	25	3
105	0	0	0	0	0
107	0	0	0	0	0
109	0	0	0	0	31
#SAMPLES	52	40	68	64	47
#MEASURED	277424	23007	42970	35232	25833
EST #CAUGHT	8110562	8767800	8907128	7202465	4380640
MEAN LEN	31.43	31.16	35.45	35.25	39.13
MEAN MASS	259	302	396	419	562
SAMPLE MASS	102576	24758	51195	43255	32499
CATCH MASS	2101224	2645712	3529210	3015332	2463794

(Table 4.4 Continued)

LENGTH- CLASS	YEAR			
	1986	1987	1988	1989
19	0	0	0	1380
21	0	3126	2462	28199
23	11373	39819	89793	168753
25	51449	278548	352459	527289
27	183831	748308	1044042	1102048
29	315716	1244285	1379609	1519450
31	591645	1467446	1534834	1545521
33	861423	1437435	1264372	1165687
35	929054	1122860	995357	846569
37	845253	758530	725107	626866
39	704206	613217	486486	427283
41	474346	478481	372528	320667
43	315191	412674	277338	247689
45	211207	372359	193887	187250
47	162218	300963	168124	167797
49	121876	270865	144367	126135
51	94769	201174	117380	95044
53	69210	168576	108568	74926
55	56526	122253	89440	67765
57	50399	101316	86529	63453
59	40003	82235	73162	56807
61	36188	58872	72873	53318
63	35747	43912	54337	48361
65	32409	35756	40854	38369
67	29766	29441	32387	27401
69	22806	23761	23145	20381
71	22540	18511	15496	14136
73	16535	14865	9955	8378
75	13001	11242	7443	5378
77	7821	7090	4228	4249
79	6379	6866	3259	2537
81	3688	4621	2779	1457
83	2504	3518	1602	840
85	1317	1833	817	498
87	1143	1702	786	202
89	598	1090	454	151
91	350	523	253	69
93	186	282	169	68
95	151	191	112	108
97	75	97	45	0
99	39	18	46	40

(Table 4.4 Continued)

LENGTH- CLASS	YEAR			
	1986	1987	1988	1989
101	0	5	32	0
103	51	37	0	18
105	0	19	0	0
107	0	0	0	0
109	0	0	0	0
#SAMPLES	60	70	60	55
#MEASURED	34441	40247	34218	31864
EST #CAUGHT	6322989	9999999	9776916	9592537
MEAN LEN	38.35	36.64	35.16	34.20
MEAN MASS	512	442	410	414
SAMPLE MASS	41575	48574	39779	37766
CATCH MASS	3236618	4634437	4007705	3968984

Table 4.5 : Aggregated catch-at-age ($C_{y,a}$) data and mid-year masses-at-age for the Cape hake fishery off the South African west coast. Units are millions and grammes respectively.

YEAR	AGE (a)								
	1	2	3	4	5	6	7	8	9+
1978	26.88	266.44	56.38	14.40	5.89	1.81	0.35	0.00	0.00
1979	29.45	141.22	55.71	16.44	11.99	3.39	0.86	0.06	0.00
1980	12.97	109.75	67.13	26.02	11.06	3.94	1.25	0.37	0.12
1981	74.53	148.46	51.87	24.36	8.65	3.75	1.23	0.45	0.11
1982	121.39	161.02	38.11	13.71	7.72	3.08	0.90	0.21	0.08
1983	22.27	91.71	47.14	15.70	8.04	2.94	1.10	0.25	0.07
1984	19.00	96.55	54.61	21.77	9.12	3.45	1.13	0.43	0.07
1985	11.37	109.07	63.44	20.63	11.00	5.17	1.76	0.30	0.05
1986	5.10	73.24	77.93	25.86	13.01	5.74	2.06	0.54	0.18
1987	6.00	103.54	71.23	26.08	11.93	2.83	1.03	0.44	0.14
1988	9.65	142.70	56.50	12.85	9.14	4.84	1.06	0.28	0.06
1989	0.84	43.46	67.96	21.99	10.05	4.44	1.48	0.23	0.07
$w_{a+1/2}$	61	190	452	827	1341	2005	2785	3694	4735

Table 4.6 : Hake biomass survey estimates (in tons) for the South African west coast.

CRUISE	BIOMASS ESTIMATE	STANDARD ERROR
JULY 1983	200 988	28 520
JANUARY 1984	140 430	21 147
JULY 1984	250 040	34 850
JANUARY 1985	292 745	43 080
JULY 1985	446 452	59 674
JANUARY 1986	313 744	43 379
JULY 1986	292 147	30 402
JANUARY 1987	365 755	55 334
JUNE 1987	282 952	45 820
FEBRUARY 1988	225 539	29 432
AUGUST 1988	279 201	64 740
JULY 1989	792 831	141 863
JANUARY 1990	738 094	156 612
JULY 1990	343 737	44 476

Table 4.7 : Biomass estimates and their standard errors obtained from the random stratified sampling cruises carried out by the SFRI, disaggregated by hake species and depth zones. Units are tons.

(a) *M. capensis*

Time of survey	Depth zone (m)					TOTAL
	0-100	100-200	200-300	300-400	400-500	
Jun/Jul 1983	3925 (1721)	10500 (2050)	11129 (2729)	10595 (3646)	250 (159)	36387 (5277)
Jan 1984	12976 (5318)	15623 (2787)	12716 (2247)	3033 (2274)	13 (13)	44361 (6802)
Jul 1984	61050 (16098)	28675 (7564)	9144 (2140)	3334 (1944)	58 (58)	102260 (18020)
Jan 1985	61658 (20529)	44607 (9270)	15535 (2677)	2664 (1106)	158 (158)	124622 (22711)
Jul 1985	48239 (22290)	69829 (10086)	48289 (9741)	15160 (7851)	0	181517 (27480)
Jan/Feb 1986	17936 (7807)	55275 (13618)	39605 (17510)	4907 (2478)	0	117723 (23646)
Jul 1986	5759 (4370)	43955 (8703)	50469 (14260)	19336 (6615)	92 (92)	119611 (18492)
Jan 1987	7689 (5036)	42116 (7325)	24749 (5014)	1138 (462)	0	75692 (10216)
Jun 1987	11991 (6725)	33153 (6252)	31047 (4109)	11216 (4925)	0	87406 (11201)
Feb 1988	13496 (6345)	27402 (4735)	17007 (5872)	8831 (4333)	0	66736 (10767)
Aug 1988	12433 (6536)	11526 (2711)	16110 (5507)	6010 (3293)	0	46079 (9552)
Jul 1989	165659 (60104)	101820 (24735)	42003 (14144)	14395 (10266)	0	323878 (67304)
Jan 1990	225787 (126588)	173255 (45146)	50659 (14531)	5916 (4399)	244 (177)	455861 (135253)
Jul 1990	27680 (7731)	43941 (8269)	64053 (16959)	17012 (7911)	225 (117)	152912 (21871)

(Table 4.7 Continued)

(b) *M. paradoxus*

Time of survey	Depth zone (m)					TOTAL
	0-100	100-200	200-300	300-400	400-500	
Jun/Jul 1983	0 ()	2652 (1300)	16575 (7133)	58958 (15268)	32312 (10591)	110497 (19947)
Jan 1984	0 ()	6289 (2294)	29612 (7083)	34638 (15154)	28142 (11588)	76039 (16934)
Jul 1984	0 ()	623 (270)	11748 (2790)	38616 (11812)	28142 (11588)	79130 (16783)
Jan 1985	0 ()	8094 (3779)	47340 (14556)	82471 (32430)	30219 (7888)	168125 (36607)
Jul 1985	0 ()	2108 (609)	51386 (14186)	149120 (47812)	62321 (17837)	264935 (52970)
Jan/Feb 1986	0 ()	27741 (21557)	98236 (23452)	54476 (16768)	15568 (5177)	196021 (36368)
Jul 1986	0 ()	3312 (1139)	47691 (10979)	80304 (18606)	41229 (10689)	172536 (24131)
Jan 1987	0 ()	45267 (15290)	157002 (47562)	56923 (13225)	30872 (16932)	290063 (54383)
Jun 1987	0 ()	9974 (2335)	52945 (22825)	82344 (25931)	50283 (27843)	195546 (44430)
Feb 1988	0 ()	24623 (5680)	50732 (12102)	60212 (18638)	23236 (14974)	158803 (27392)
Aug 1988	0 ()	6246 (1500)	28537 (7691)	113380 (34229)	84958 (53543)	233122 (64031)
Jul 1989	0 ()	9810 (2346)	75499 (19556)	311304 (119290)	72339 (31261)	468953 (124881)
Jan 1990	0 ()	5241 (2279)	193735 (76434)	60148 (17573)	23109 (8825)	282233 (78956)
Jul 1990	0 ()	8596 (2669)	44644 (9861)	62452 (25373)	75134 (27416)	190825 (38727)

Table 4.8 : Estimates of the number of *M. capensis* by age and depth zone obtained from the results of the direct surveys

(a) January 1986

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	2530733	10326397	0	0	0
1	52004800	26526480	292792	0	0
2	88015160	154739712	45855996	0	0
3	805936	29112890	27805662	0	0
4	0	2746983	8551164	228949	0
5	0	2007813	4880380	548844	0
6	0	1333703	1578616	992239	0
7	0	473350	357125	443827	0
8	0	95244	206318	68480	0
9	0	0	13887	7155	0
10	0	0	0	0	0
11	0	0	0	0	0
12	0	0	0	0	0

(b) July 1986

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	1151587	0	0	0
1	35621996	41294768	122611	0	0
2	22547324	76352952	3483416	0	0
3	380120	61117140	47508440	1619221	0
4	15302	4023533	13900370	2781206	0
5	17852	1242652	4264319	3435306	1891
6	0	801866	2543515	3036977	39720
7	0	477308	931958	1036381	7566
8	0	191169	501441	299640	0
9	0	0	162012	105480	0
10	0	0	0	0	0
11	0	0	16532	0	0
12	0	0	0	0	0

(Table 4.8 Continued)

(c) January 1987

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	402581	254420	0	0	0
1	10830811	5002270	1887	0	0
2	31415542	48413224	581741	0	0
3	4886015	36793444	6077501	0	0
4	40620	14746806	14672733	101581	0
5	0	1597356	3636984	127958	0
6	0	600684	1408249	193302	0
7	0	298609	337044	139516	0
8	0	20086	117170	17171	0
9	0	0	28931	0	0
10	0	0	0	0	0
11	0	0	0	0	0
12	0	0	0	0	0

(d) June 1987

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	0	0	0	0
1	74286400	9899929	6613	0	0
2	56907544	20415454	489344	0	0
3	1904365	32979244	11408296	436825	0
4	13814	9225065	9212951	2150499	0
5	0	4246372	7152042	3622846	0
6	0	803928	1698293	949823	0
7	0	366414	850640	475993	0
8	0	59127	301294	139581	0
9	0	33743	186397	50343	0
10	0	0	38574	10407	0
11	0	0	16532	0	0
12	0	0	16532	0	0

(Table 4.8 Continued)

(e) February 1988

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	29407338	1479346	0	0	0
1	94974328	11611338	0	0	0
2	54959176	37011276	623125	0	0
3	1284706	23500250	7041489	0	0
4	22201	3730639	4863290	150484	0
5	39962	2027830	3301778	1800790	0
6	0	773058	1519514	2086656	0
7	0	247417	222356	513025	0
8	0	78565	69801	85272	0
9	0	3691	0	0	0
10	0	0	38574	0	0
11	0	0	0	0	0
12	0	0	0	0	0

(f) August 1988

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	3504820	1465655	0	0	0
1	48714776	13497434	44711	0	0
2	65958300	30543346	1195105	0	0
3	634425	5532345	5184640	419674	0
4	51613	1069478	6026554	1962344	0
5	15787	825988	3519739	1580569	0
6	0	613823	1242055	650376	0
7	0	367115	348184	239948	0
8	0	75609	75220	28619	0
9	0	0	17358	0	0
10	0	11632	17358	0	0
11	0	0	0	0	0
12	0	0	0	0	0

(Table 4.8 Continued)

(g) July 1989

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	64623200	80983392	604653	0	0
1	1917139328	877716992	40250880	0	0
2	1022898368	294400288	123391296	0	0
3	56211308	84173440	19788270	2099123	0
4	2544243	21380742	15156026	9896877	0
5	19340	2493151	4029203	2085275	0
6	0	1011751	1623391	547644	0
7	0	387701	776344	403614	0
8	0	93087	221045	123923	0
9	0	17274	44843	11447	0
10	0	0	0	0	0
11	0	0	0	0	0
12	0	0	0	0	0

Table 4.9 : Estimates of the number of *M. paradoxus* by age and depth zone obtained from the results of the direct surveys

(a) January 1986

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	8092302	1717059	0	0
1	0	83504352	238536592	24822532	1144132
2	0	48633088	264621008	105630888	6045614
3	0	32811280	55973204	56328336	15277104
4	0	110396	2107253	2449078	2777300
5	0	10823	1084889	2119053	1467749
6	0	0	350607	1396956	584828
7	0	0	106928	1450013	505793
8	0	0	0	260668	215734
9	0	0	69434	32196	65862
10	0	0	0	17171	28687
11	0	0	0	0	0
12	0	0	0	0	0

(b) July 1986

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	5526904	244304	28619	19124
1	0	69846040	158897072	39357844	1241034
2	0	2954818	158369568	112383864	23670176
3	0	220042	17614226	45704108	23013158
4	0	0	1698985	20016825	6937892
5	0	0	842142	8621379	4528057
6	0	0	501983	341365	3807354
7	0	0	107040	1032016	2049038
8	0	0	121234	212534	362804
9	0	0	60617	69503	19561
10	0	0	0	14309	16392
11	0	0	0	0	0
12	0	0	0	0	0

(Table 4.9 Continued)

(c) January 1987

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	3853798	865031	0	0
1	0	347934080	222956528	13578792	1553034
2	0	149117248	586578624	96489000	17301268
3	0	3925943	72614192	47876528	27865312
4	0	99459	16145846	12183742	9683571
5	0	0	2949209	3949314	2685482
6	0	0	1095023	1100085	944989
7	0	0	143339	467032	588460
8	0	0	1607	498144	1329864
9	0	0	0	12521	92890
10	0	0	0	0	13660
11	0	0	0	0	13660
12	0	0	0	0	0

(d) June 1987

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	15727093	2433192	131977	0
1	0	200082272	153048448	25890582	603409
2	0	14692348	215872800	133055320	24900284
3	0	190781	16954840	39135596	20288676
4	0	11144	1438883	14585478	10351262
5	0	3715	321545	14362284	9855123
6	0	0	0	4217849	2986500
7	0	0	0	1741775	2145961
8	0	0	0	751024	1957970
9	0	0	0	234153	591944
10	0	0	0	15610	138879
11	0	0	0	15610	0
12	0	0	0	0	0

(Table 4.9 Continued)

(e) February 1988

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	14502722	1016781	0	0
1	0	367972544	164964240	8066653	8196
2	0	50130616	123607192	87476184	4210289
3	0	963279	30194628	14788871	7890548
4	0	17851	6489719	11027114	1401828
5	0	14764	2777681	11852966	2012652
6	0	0	612174	5012729	4202531
7	0	0	239775	2000122	1785185
8	0	0	88838	1013813	529163
9	0	0	12858	399493	187575
10	0	0	0	0	0
11	0	0	0	0	24588
12	0	0	0	0	0

(f) August 1988

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	19566794	7371083	0	15368
1	0	123967728	160482848	21649742	506039
2	0	6581990	136181200	346706688	70824200
3	0	182958	2634026	60139284	49280896
4	0	0	109201	12214768	30787900
5	0	17448	69434	8960977	9694803
6	0	0	40503	3660478	2737655
7	0	0	9644	1807131	1094760
8	0	0	1929	701352	726951
9	0	0	0	36795	145994
10	0	0	0	18398	3842
11	0	0	0	0	0
12	0	0	0	0	0

(Table 4.9 Continued)

(g) July 1989

AGE	Depth Zone (m)				
	0-100	100-200	200-300	300-400	400-500
0	0	33202096	13565897	70996	0
1	0	266622448	240341552	23638154	334403
2	0	10231512	280106080	342707968	74703200
3	0	2667970	101611992	373902912	162119088
4	0	61264	3436131	146547472	15330026
5	0	0	1993989	36501600	1598101
6	0	0	1513738	15426136	761795
7	0	0	475254	7634164	283214
8	0	0	0	658230	0
9	0	0	0	0	0
10	0	0	0	0	0
11	0	0	0	0	0
12	0	0	0	0	0

**APPENDIX 4.A: THE ALGORITHM CURRENTLY USED TO CALCULATE CATCH-AT-AGE
DATA FROM CATCH-BY-MASS DATA**

In order to convert catch-by-mass data (C_y - the total catch-by-mass in year y) into catch-at-age data ($C_{y,a}$ - the number of fish of age a caught in year y) using this algorithm, the following data are required in addition to the catch-by-mass data:

- a) the average mass of a fish in length-class l ,
- b) the proportion of the catch (in numbers) which fall into length-class l during year y ($f_{l,y}$), and
- c) the number of fish in length-class l which were aged to be a years during the surveys carried out in year y ($n_{l,a,y}$).

The algorithm that has been applied by the SFRI is as follows.

- 1) Equation (2.8) is fitted to the length-mass data using linear regression (after logarithm transformation), thus providing estimates of the length-mass relationship parameters a_w and b_w .
- 2) A Von Bertalanffy growth equation is fitted to the age-length data pooled over time by minimizing the function:

$$SS = \sum_{a=1}^m \left[l_{\infty} (1 - e^{-\kappa(a-t_0)}) - \left\{ \frac{\sum_{l=1}^{n_l} l n_{l,a}}{\sum_{l=1}^{n_l} n_{l,a}} \right\} \right]^2 \quad (4.A.1)$$

where l_{∞} , κ and t_0 are the Von Bertalanffy growth curve parameters,

$$n_{l,a} = \sum_y n_{l,a,y}$$

n_l is the number of length-classes, and

m is the oldest age considered.

- 3) The proportion of fish aged a years in year y is calculated using the formula:

$$P_{a,y} = \frac{\sum_{\ell} n_{\ell,a,y} f_{\ell,y}}{\sum_{\ell} n_{\ell,a,y}} \quad (4.A.2)$$

where $n_{\ell,a,y} = n_{\ell,a,y} / \sum_a n_{\ell,a,y}$

- 4) The mean mass of fish in the catch for year y , w_y , is then determined using the equation:

$$\bar{w}_y = \frac{\sum_{a=1}^m P_{a,y} a_w [\ell_{\infty} (1 - e^{-\kappa(a - t_0)})]^{b_w}}{\sum_{a=1}^m P_{a,y}} \quad (4.A.3)$$

where ℓ_{∞} , κ and t_0 are the Von Bertalanffy growth curve parameters,

and

a_w and b_w are the length-mass regression parameters.

- 5) The total-catch-in-numbers for year y (C_y^N) is then obtained using the formula:

$$C_y^N = C_y / \bar{w}_y \quad (4.A.4)$$

- 6) The total-catch-in-numbers is broken down into ages using the formula:

$$C_{y,a} = C_y^N P_{a,y} \quad (4.A.5)$$

There are some problems with this approach. In particular, the method of fitting a growth curve to the pooled age-length key in order to estimate the mean catch mass is inappropriate. This is because the data upon which the age-length keys are based are collected by means of random stratified sampling. The larger fish are thus overrepresented in the age-length samples, so that use of the formula:

$$\frac{\sum_{\ell=1}^{n_{\ell}} \ell n_{\ell,a}}{\sum_{\ell=1}^{n_{\ell}} n_{\ell,a}} \quad (4.A.6)$$

to estimate the mean length-at-age would lead to positively biased estimates.

A further problem occurs in the estimating mass-at-age. Essentially, equation (4.A.3) implies that the mean mass of a fish of age a is given by:

$$a_w \bar{\ell}_a^{b_w} \quad (4.A.7)$$

However, this formula provides biased estimates of mean mass-at-age, because it does not take the distribution of length about the mean length-at-age into account. This is because the expected value of a non-linear function of a random variable is not the same function of the expected value of the random variable.

Both of the problems outlined above can be solved by estimating mass-at-age using the formula:

$$\sum_{\ell=1}^{n_{\ell}} a_w \bar{\ell}^{b_w} n_{\ell,a} f_{\ell} \quad (4.A.8)$$

where $\bar{\ell}$ is the mid-point of length-class ℓ .

APPENDIX 4.B: ESTIMATION OF THE SURVEY BIOMASS INDICES AND NUMBERS AT AGE

The biomass indices and their standard errors are computed as follows:

a) The fish density at each station is computed using the formula:

$$d_{i,j} = X_{i,j}/A_{i,j} \quad (4.B.1)$$

where $d_{i,j}$ is the areal density at the j 'th station in the i 'th stratum,

$X_{i,j}$ is the catch in mass at the j 'th station in the i 'th stratum, and

$A_{i,j}$ is the area swept at the j 'th station in the i 'th stratum.

b) The biomass in each stratum is computed by

$$B_i^S = \bar{A}_i \cdot \bar{d}_i \quad (4.B.2)$$

where B_i^S is the biomass in the i 'th stratum,

\bar{A}_i is the area of stratum i ,

\bar{d}_i is the mean density in the i 'th stratum $(\sum_j d_{i,j} / n_i)$, and

n_i is the number of hauls in stratum i .

c) The total biomass is then computed using the formula:

$$B^S = \sum_i B_i^S \quad (4.B.3)$$

d) The standard error of the total biomass is computed using the formula:

$$\sigma^S = \sqrt{\sum_i A_i^2 \cdot sd_i^2 / n_i} \quad (4.B.4)$$

where sd_i is the standard deviation of the $d_{i,j}$'s, and

σ^S is the standard error of B^S .

In order to estimate the number of fish aged a years in the population, the above algorithm is repeated, except that $X_{i,j}$ is replaced by $Y_{i,j,a}$, where $Y_{i,j,a}$ is the estimate of the number of a -year-olds at the j 'th station in the i 'th stratum:

$$Y_{i,j,a} = \sum_{\ell=1}^{n_{\ell}} n_{\ell,a,y} f_{\ell,i,j} X_{i,j} / \bar{w}_{i,j} \quad (4.B.5)$$

where $n_{\ell,a,y}$ is the age-length key for the survey,

$f_{\ell,i,j}$ is the proportion of fish in length-class ℓ at the j 'th station in the i 'th stratum, and

$\bar{w}_{i,j}$ is the mean mass of fish caught at the j 'th station in the i 'th stratum.

In order to estimate the number of fish in length-class ℓ in the population, the above algorithm is repeated, except that $X_{i,j}$ is replaced by $Z_{i,j,\ell}$ where $Z_{i,j,\ell}$ is the estimate of the number of fish in length-class ℓ at the j 'th station in the i 'th stratum:

$$Z_{i,j,\ell} = f_{\ell,i,j} X_{i,j} / \bar{w}_{i,j} \quad (4.B.6)$$

In order to apply these algorithms, it is necessary to have more than one haul per stratum (otherwise the standard deviation of the densities cannot be estimated). However, in many cases there is only one haul in the originally defined strata [Payne *et al.* (1984)], so that each depth zone has been taken to be a stratum in the calculations reported in Tables 4.6, 4.7 and 4.8. This re-stratification results in the estimates having lower precision than would have been expected had it been possible to use the original stratification, which was based on

prior knowledge of density variations in the area, and an allocation of survey effort designed to achieve a minimum variance biomass estimate.

CHAPTER 5 - MANAGEMENT PROCEDURES

A management procedure usually consists of two parts: the estimator and the catch control law. The estimator involves fitting a model to data to provide estimates of management-related quantities. Three types of estimators are considered in this thesis; these are based on surplus-production, partially age-structured and fully age-structured models. Surplus-production estimators assume that it is possible to mimic the dynamics of a resource by modelling the exploitable biomass only, and that the effects of natural mortality, tissue growth and recruitment can be represented by a single function. Surplus-production estimators can be criticized as over-simplistic. Partially and fully age-structured estimators consider the processes leading to changes in population size explicitly. Partially age-structured estimators differ from their fully age-structured counterparts in that the range of functional forms used to model the biological processes is restricted, so that it is possible to collapse the age-structured dynamics equation into a single age-aggregated equation. Problems related to fitting models to data include: time-dependence of parameter values, substantial emigration and immigration effects, other spatial effects, and non-linearity in CPUE-abundance relationships.

5.1 Introduction

A management procedure consists of a set of rules (perhaps augmented by equations) which, when provided with a pre-specified set of data, produces a management measure for recommendation. Most of the management procedures considered in this study can be divided into two parts: the estimator (or the "model-estimation procedure") and the catch control law (or the "harvesting strategy").

The estimator uses the data to obtain estimates of the various management related quantities which are required by the catch control law in order to produce the management measure. There are a large number of "management measures" which can be used to control a fishery (e.g. total allowable catch, maximum effort level, closed season, closed area, minimum mesh size, minimum landed size, bag limit, etc.). However, in this study, the only management measure which has been

considered is the setting of a total allowable catch (TAC) for the entire management area (i.e. the South African west coast). This selection has been made because, traditionally, management of the Cape hake resource off the South African west coast has been primarily by means of this method.

The remainder of this Chapter is devoted to discussing the estimator at a generic level. The specifics of the estimators and the catch control laws considered in this study are detailed in Chapters 6 to 9.

5.2 The estimator

All of the estimators considered in this study involve "fitting" a model to some data. The term "fitting" is placed in quotes because some of the fully age-structured estimators to be considered use what are essentially *ad hoc* fitting methods (see Section 7.4), rather than methods with a rigorous statistical basis.

All the models considered in this study make a common set of assumptions regarding the resource being modelled. It is assumed to be closed (or a unit stock), i.e. rates of immigration and emigration are considered to be negligible compared with the rates of resource growth and natural mortality. The stock size is assumed to increase, both as a result of recruitment into the population of young fish which have become large enough to be selected by the fishing gear, as well as of tissue growth in individual fish. Natural and fishing mortalities are effects which are assumed to reduce the population size.

The magnitude of these two effects depends, *inter alia*, on factors such as the age-, sex- and size-structure of the population, environmental variation, spatial density patterns across the fishing grounds, and interaction with other species (through food availability, predation etc.). It is possible to model such effects; for example, Butterworth *et al.* (1990) consider the effects of, *inter alia*, changes in environmental carrying capacity, when attempting to reconcile trends in absolute abundance of the Eastern North Pacific gray whale with historic catch data. However, paucity of data forces estimators to assume that most (if not all) of these factors are either insubstantial or essentially random.

The estimators considered in this study can be divided into three categories: surplus production, partially age-structured and fully age-structured. After a discussion of the CPUE-abundance relationship, each of these estimators will be discussed at a generic level.

5.2.1 Relationships between CPUE and abundance

An assumption which is frequently made in fisheries assessment is that catch-per-unit-effort is related to abundance, i.e.:

$$(C/E)_Y = f(\tilde{B}_Y, E_Y, C_Y) \quad (5.1)$$

where \tilde{B}_Y is some measure of the abundance of the resource.

Of the many functional forms which have been suggested for $f()$ [see, for example, Cooke (1985); Cooke and Beddington (1984); Walters (1986)], that which has most commonly been assumed is linear proportionality, i.e.:

$$(C/E)_y = q\tilde{B}_y \quad (5.2)$$

This relationship arises from the assumption that the population and/or the fishing effort are randomly distributed over the fishing grounds, so that a single additional unit of effort or of biomass produces a proportional increase in the catch. The parameter q in this Equation is called the catchability coefficient, and measures the effectiveness of the fishing gear in conjunction with the availability and vulnerability of the population.

Thus, although \tilde{B}_y is seldom (if ever) known, relation (5.2) indicates that $(C/E)_y$ may be used as an index of abundance. In reality, Equation (5.2) (or the "observation model") has random error superimposed on it, which is termed "observation error". The stochastic equivalent of Equation (5.2) is assumed to be:

$$(C/E)_y = q\tilde{B}_y e^{v_y} + z_y \quad (5.3)$$

The two noise terms in Equation (5.3) reflect two common assumptions of how noise can occur in the relationship between CPUE and abundance. Details of scenarios in which these two noise terms can arise are given in Section 6.3. (Note that, although errors in the measurement of catch and effort have not been considered in the above analysis, these may occur and could readily be incorporated into an extension of Equation (5.3).)

5.2.2 Surplus production model-estimation procedures

The use of surplus production (lumped or stock production) model-estimation procedures is based on the assumption that it is possible to mimic the dynamics of a resource by modelling only one state variable. The variable which is usually modelled is exploitable biomass. These model-estimation procedures are sometimes used in fisheries management instead of methods that require catch-at-age information (such as VPA) which might be unreliable (or unavailable). Surplus production models have minimal data requirements (usually only annually-aggregated catch and effort information is used).

A surplus production model assumes that the effects of reproduction (recruitment), tissue growth and natural mortality can be subsumed into a single function. This function (the surplus production function) is usually assumed to be a function of the current (exploitable) biomass alone. It is thus implicitly assumed that the age-structure of the population has a negligible effect on the production rate (i.e. the size of the biomass rather than its age-structure determines the rate of production). For this to be true, the age composition of the stock (i.e. the fraction of the total biomass falling into each age-class) must remain reasonably constant throughout the history of the fishery, because fecundity, growth and mortality are age-specific.

Any proposed deterministic surplus production function should be consistent with certain basic biological constraints:

- (i) it should pass through the origin (i.e. no adults, no production),

- (ii) the function should be zero at the average pristine level; the existence of such a level follows from the fact that there must be natural limits to the growth of the resource (in the absence of exploitation), and
- (iii) the function is positive for at least some biomass range below the average pristine level.

These assumptions imply that, in the absence of fishing, a stock once disturbed will tend to return to its average pristine level, which is called the environmental carrying capacity. The functional forms chosen for use as surplus production functions have maxima at biomasses which are below this level. As no state variables other than biomass are used in the surplus production function, it is assumed that the population is regulated by density dependent effects (e.g. cannibalism, food availability, etc.) which are implicitly incorporated into the function itself.

Surplus production functions which become negative below a certain biomass are said to exhibit critical depensation. If the population falls below this level, it will eventually be driven to extinction (even in the absence of fishing).

The choice of an appropriate surplus production function for a stock is usually determined by how well this function fits the data, and whether the data exhibit sufficient contrast to allow all the function's parameters to be estimated with reasonable precision. However, in most cases, paucity of data precludes selection between alternative surplus production functions [e.g. Butterworth and Andrew (1987b)]. Punt

(1988), on the other hand, selects between alternative surplus production functions on the basis of their robustness to the underlying true dynamics of the resource.

Punt (1988, 1989c, 1989d) has investigated the performance of a number of catch-effort based model-estimation procedures using simulation methods. A large proportion of those considered performed very poorly [e.g. procedures based on Gullands (1961) effort-averaging approach and the Leonart *et al.* (1985b) model]. Others were found to perform almost identically [e.g. the Butterworth-Andrew (1984) and Schnute (1977) approaches]. In order to reduce the amount of computation required, only one surplus production model has been considered later in this thesis (Chapter 6). However, a large number of model-estimation procedures can be constructed from this model.

5.2.3 Fully age-structured model-estimation procedures

Surplus-production models are mathematically and conceptually simple. They usually have minimal data requirements and are relatively easy to use. However, most biologists are more comfortable with analyses that explicitly include the biological realism of age-structure (Hilborn and Walters 1987). Furthermore, much of the data which are collected by management (e.g. growth, mortality rates, and, particularly, catch-at-age data) can be used more easily by an age-structured model-estimation procedure than by a surplus-production model-estimation procedure. Furthermore, intuitively, it would seem that a model-estimation procedure based on an age-structured model should be able to provide

more reliable estimates of management quantities than surplus-production model-estimation procedures.

The three main differences between surplus-production and age-structured model-estimation procedures are that the latter:

- a) usually consider numbers rather than biomasses,
- b) take explicit account of the age-structured nature of a fish population, and
- c) generally model the processes leading to changes in population numbers explicitly (viz. natural mortality, fishing mortality, tissue growth and recruitment).

5.2.4 Partially age-structured model-estimation procedures

Surplus-production models (such as the Butterworth-Andrew model) can be criticized on the grounds of over-simplification. For example, Deriso (1980) criticizes surplus production models as they do not take seasonal breeding into account and they often imply that the progeny age instantaneously to adulthood. In addition, the parameters of a surplus-production model rarely have any specific biological interpretation. It is therefore seldom possible to validate (or invalidate) parameter values by comparison with observable phenomena (Deriso *op. cit.*).

The delay difference models of Deriso (1980) and Schnute (1985) incorporate age-structure effects (subject to certain assumptions) and treat each component of production and mortality separately. Each parameter of these models has a specific biological interpretation.

These models are thus not subject to the criticisms of oversimplification levelled at surplus production models, and also do not require a large amount of data for estimation purposes [e.g. the model-estimation procedures proposed by Deriso (op. cit.) and Schnute (op. cit.) require only a time series of catch and effort data, given independent availability of values of certain biological parameters]. Furthermore, in order to reduce the number of estimable parameters when auxiliary information is available, each parameter for which such data is available can be set to the value indicated by those data.

5.3 Fitting models to data

In order to fit a model to data, an error model must be assumed. The selection of an error model is important for obtaining estimates of management quantities which have as small a variance as is possible (Butterworth and Andrew 1987a). The values of the parameters which minimize the statistical "goodness-of-fit" (or minimization) criterion are then determined. The selection of an error model involves deciding (perhaps semi-arbitrarily) which noise terms in the dynamics equation are dominant and which can be safely ignored as being insubstantial. To enable sensible estimates of parameters to be obtained from a fit of a model to historical data, the statistical criteria of repeatability and stationarity must be assumed.

Walters (1986) defines a strictly repeatable process as a "process from which the same mean response is obtained before and after any disturbance of the stock, irrespective of the size of the disturbance". For example, most surplus-production model-estimation procedures assume

that recruitment is dependent on (current) stock size alone and not on the age-structure, so that the expected recruitment at a given stock size always does not change with time. No stock condition can be strictly repeatable because large disturbances lead to irreversible loss of genetic and spatial structures (Walters *op. cit.*). However, in a "well-managed" stock, large disturbances are unlikely to occur, and so processes need only be weakly repeatable (i.e. the same mean response will be obtained after any disturbance that is likely to occur in practice). If a process is non-repeatable, certain basic sub-processes are missing from the model, and biased estimates may be obtained if historical data are used for estimation purposes (Walters *op. cit.*).

A stationary process is one for which the probability distribution of a response from the process is stable over time. This is a stronger condition than repeatability, which only requires that the mean value remains stable.

5.4 Problems associated with the models considered

The models considered here make several implicit assumptions which are unlikely to hold in practice.

- 1) The values of the parameters of the model are time-invariant. It is probable that some of the parameter values will change systematically over the history of the fishery. Such variations can be expected as the fine structure of the population is altered by fishing, the more accessible subpopulations, for example, tending to be most heavily depleted. The values of some of the model

parameters, such as the average pristine population (K), may vary systematically over time as a result of long term environmental changes.

2) The catchability coefficient, q , remains constant over the entire history of exploitation. Possible reasons why this is unlikely to be true in reality include:

- (i) abnormally high catch rates in the early stages of exploitation caused by first fishing down the small areas of highest density [a possible explanation for the high initial CPUE values recorded in all ICSEAF Divisions - see Table 3.1],
- (ii) changes in mesh size which affect age-specific selectivity patterns,
- (iii) an initial "learning" process as areas of higher density are discovered, which tends to reduce the effect described in (i),
- (iv) inadequate allowance for changing vessel characteristics (i.e. inadequate power factors), and
- (v) undetected increases/decreases in efficiency. Butterworth and Andrew (1987b) illustrate that (for their model) an undetected increase in fishing efficiency partly due, perhaps, to changes in catchability, may result in a positive bias of up to 40% of the estimated $f_{0.1}$ harvesting strategy TAC value. They also show that even the estimates of quantities which are independent of current biomass (such as MSY) tend to vary linearly as a function of an (undetected) increase in catching power. Attempts by these authors (Andrew and Butterworth 1988) to quantify suspected changes in q for the ICSEAF hake stocks

as a result of mesh size changes, have been unsuccessful, although they conclude that this may be possible with a larger data set.

- 3) The stocks are not substantially affected by emigration and immigration. However, Payne *et al.* (1988) note that the hake stocks off southern Namibia and the South African west coast are closely related. The assumption that emigration and immigration effects are negligible is thus open to question. In most management situations, in addition to the problems associated with emigration and immigration, the substocks within the area assumed to be occupied by a stock may exhibit considerable fine structure (Walters 1986).
- 4) Fishing takes place at random over the area occupied by the stock. However, fishing is a decidedly non-random operation with respect to spatial distribution. Accessible high density areas will always tend to be subjected to greater fishing intensities. The assumption that effort is randomly distributed over the fishing grounds may therefore often be invalid and fish movement insufficiently rapid to compensate for this.
- 5) A linear relationship exists between CPUE and stock density. This assumption has been questioned by Cooke (1985a), who lists a number of scenarios in which CPUE is more likely to be related to abundance by the power relationship:

$$(C/E)_y = (\tilde{q}B_y)^p \quad \text{where } p < 1 \quad (5.4)$$

Some of these scenarios are:

- (i) effort calculations include those occasions on which q is zero (e.g. during bad weather),
- (ii) effort calculations include the time spent steaming to the fishing grounds and handling the catch [Cooke (*op. cit.*) shows that even if this time is subtracted from the estimated effort, nonlinear relationships of the form of Equation (5.4) still result because of fluctuations in catchability q],
- (iii) searching is non-random,
- (iv) stock density is not constant across the fishing grounds, and
- (v) scouting vessels are used to locate areas of highest concentration.

Another reason why CPUE may not be related to abundance is the difficulty in defining directed effort in a multispecies fishery.

In addition to the potential problems associated with the models that are listed above, serious statistical difficulties may arise when fitting a model. These include the following.

- (i) Certain of the critical model parameters may be confounded, i.e. it is possible to estimate only the value of a function of these parameters combined, rather than the values of the individual parameters, because there is insufficient contrast in the data on stock size and fishing effort (Hilborn 1979).
- (ii) Measurement error (e.g. sampling error in survey estimates) may disguise a *bona fide* relationship between variables, so that they appear to be independent of each other [the "errors in variables" problem noted by Walters and Ludwig (1981)]. Even if

this is not the case, measurement error may nevertheless bias parameter estimates substantially.

- (iii) The precision of the estimates may not improve over time, as most management strategies aim to stabilize the biomass at some "optimum" level, thus having the effect of dampening out data contrast.
- (iv) The error model selected is likely to be wrong. In practice, it is unlikely that there is only one dominant source of error. Rather, there will be a number of them, so that no estimator will model the error structure correctly.

CHAPTER 6 - THE SURPLUS PRODUCTION MODEL-ESTIMATION PROCEDURES

All the surplus-production model-estimation procedures considered in this thesis are based on the Butterworth-Andrew (1984) discrete dynamics time series model. This model assumes that the inter-annual change in biomass is made up of only three effects: surplus-production, catch, and process error. It also assumes that the annual CPUE is proportional to the mid-year biomass (subject to some observation error). Seven surplus-production functions are considered: Schaefer (1954), Fox (1970), Pella-Tomlinson (1969) and four Shepherd (1982) forms. The "base case" applications of these estimators utilize both CPUE and survey biomass data. Because of the poor performance of estimators based on the assumption of (pure) process error, only estimators based on (pure) observation error and "total least squares" [Ludwig and Walters (1988)] estimators are considered. The fitting procedures applied assume that the error in the relationship between CPUE and abundance is due to catchability fluctuations, and that the survey data are relative indices of abundance which are normally distributed about their expected values.

6.1 The generalized Butterworth-Andrew model

The model which will form the basis for all the surplus production model-estimation procedures considered in this study is a generalization of the Butterworth-Andrew (1984) discrete dynamic time series model.

The population dynamics are assumed to be governed by the Equation:

$$B_{y+1} = [B_y + g(B_y)]e^{u_y} - C_y + s_y \quad (6.1)$$

where B_y is the (exploitable) biomass at the start of year y ,

$g()$ is the surplus production function, comprising the combined effects of all natural (i.e. excluding the fishery) processes which increase or decrease the population, and

u_y, s_y are components of noise (possibly related to environmental and species interaction effects) which are not explicitly modelled in Equation (6.1); these are sometimes termed

"process errors". The process error components in Equation (6.1) are a convenient representation of the different mechanisms which lead to errors in the assumption of a deterministic stock dynamics equation.

The change in biomass from one year to the next is thus made up of only three effects:

- (i) the surplus production (which is assumed to depend only on the current biomass, i.e. the population is assumed to react instantaneously to changes in biomass),
- (ii) C_y , the catch taken during the year, and
- (iii) u_y and s_y , which represent unmodelled effects (process error contributions) during year y .

It is possible to generalize model (6.1) in many ways. For example, Walter (1973) illustrates how time delays can be incorporated into a dynamic model similar to model (6.1). Time delays can be caused by, for example, the time required for juveniles to grow large enough to be selected by the fishery. However, such extensions have not been considered here, because, in actuality, there is seldom enough data to justify the estimation of the requisite additional parameters. For example, Andrew and Butterworth (1988) attempted to generalize model (6.1) to allow for changes in mesh size for the hake stocks in the ICSEAF Convention area which they considered, but found that the data were inadequate to permit precise estimation of extra parameters needed.

The CPUE-abundance relationship considered is a variant of Equation (5.3) which assumes that annual CPUE is proportional to mid-year biomass, i.e.:

$$(C/E)_y = q \left(\frac{B_y + B_{y+1}}{2} \right) e^{v_y} + z_y \quad (6.2)$$

where $(C/E)_y$ is the CPUE for year y ,

q is the catchability coefficient, and

v_y, z_y are observation error terms.

6.2 The surplus production functions considered

The seven surplus production functions which have been considered here are:

i) Schaefer (1954,1957):	$g(B) = rB(1 - B/K)$
ii) Fox (1970):	$g(B) = rB[1 - \ln(B)/\ln(K)]$
iii) Pella-Tomlinson (1969):	$g(B) = rB[1 - (B/K)^h]$
iv) Shepherd (1982):	$g(B) = \alpha B/[1 + (B/\beta)^\gamma] - M^*B$
v) Shepherd (1987):	$g(B) = \alpha B/(1 + B/\beta) - M^*B$
vi) Shepherd (1982)-like:	$g(B) = \alpha B/(1 + B/\beta)^\gamma - M^*B$
vii) Ricker (1954)-like	$g(B) = \alpha B e^{-\beta B} - M^*B$

where B is the current biomass,

r is the intrinsic growth rate parameter (although for some of the cases $g(B)/B$ does not tend to r as B tends to 0),

K is the equilibrium pristine biomass (carrying capacity),

h is the Pella-Tomlinson shape parameter,

α, β, γ are parameters of functions related to stock-recruitment

and tissue growth effects, and
 M'' is related to natural mortality.

The last four of these surplus production functions are derived by dividing surplus production, $g(B)$, into two components. The first reflects recruitment and tissue growth, while the second relates to natural mortality (M'' is generally independently estimated or guessed from auxiliary information, and so is not estimated in the non-linear search - M'' has been taken to be 0.3 here). Shepherd (1982) argues for the use of such surplus-production functions in place of the more traditional forms (such as the Pella-Tomlinson form). These traditional forms quickly become large and negative when $B > K$; because tissue growth and recruitment are necessarily positive effects, these forms imply that natural mortality must become density-dependent when $B > K$. As exploitable biomass generally refers to adult fish, and density dependent natural mortality is usually thought to occur primarily for very young fish (for example, as a result of cannibalism), the traditional forms seem unrealistic.

The choice of an appropriate surplus production function for a stock is often determined by how well the various forms fit the data, and whether the data exhibit sufficient contrast to allow all the parameters of the best fitting form to be estimated with reasonable precision. However, paucity of data usually precludes selection among alternative surplus production functions [for example, Butterworth and Andrew (1987b)]. Punt (1988), on the other hand, argued that selection of an appropriate surplus production function should be based on the relative robustness properties of the various forms. He used Monte

Carlo simulation techniques to select between alternative surplus production functions for the hake stock in ICSEAF Divisions 1.3+1.4 (off northern Namibia), and concluded that the Schaefer form provided more robust estimation than the Fox form, even when the underlying ("true") surplus production function closely resembled the latter form.

6.3 Estimating the parameters

The parameters of the Butterworth-Andrew model are estimated by minimizing the (negative of the) log of the likelihood function. After removal of constants, this becomes the minimization of the following function of the observed and predicted CPUE and survey biomass estimates:

$$-\ln L = \tilde{n} \ln \hat{\sigma} + \tilde{n}/2 + SS_S \quad (6.3)$$

where \tilde{n} is the number of years for which CPUE data are available,
 $\hat{\sigma}$ is an estimate of the residual variance, and
 SS_S is the contribution (if any) of the survey biomass data.

6.3.1 The contribution of the CPUE data to the log-likelihood

The residual variance, σ , is estimated by SS/n where:

$$SS = \sum_Y [f(C/E)_Y - f(\hat{C}/\hat{E})_Y]^2 \quad (6.4)$$

where $(C/E)_Y$ is the observed CPUE in year y ,
 $(\hat{C}/\hat{E})_Y$ is the model-predicted CPUE for year y ,
 Σ' is summation over those years (y) for which CPUE data are available, and
 f is a transformation.

The choice of the error model (transformation function f) depends on where the dominant noise term is assumed to occur in the dynamic Equations (5.3) and (6.1). The two extreme types of error model are "process error (only)" and "observation error (only)". For the purposes of this study, it is assumed that estimates of catches are reasonably accurate, and that the CPUE reflects all the variability in the catch-effort data. Note that, in order that minimization of Equation (6.3) provides minimum variance estimates, it is necessary that the transformed residuals are random, normal and homoscedastic.

- The process error model

This type of error model assumes that all noise is caused by natural processes (e.g. temperature anomalies, increase in the abundance of predators, etc.). The error terms (u_y , s_y) in Equation (6.1) are assumed to dominate those in the observation model, which expresses the relationship between CPUE and biomass (see Equation 6.2). The observation model is thus assumed to be exact, i.e. it contains no error. Here, the choice of the transformation function f will depend on *a priori* ideas of the process error variance as a function of biomass, because paucity of data usually precludes the use of statistical methods which have the power to distinguish between alternative transformations under tests for heteroscedasticity.

- The observation error model

The dominant noise is assumed to be in Equation (6.2) (i.e. in the observation model), which provides an index of the population size,

while the population dynamics [Equation (6.1)] are assumed to be deterministic. For this type of error model, an estimate of the biomass level at the commencement of exploitation (B_{1917}) is needed in the model fitting process in order to start the biomass series. If such a model is selected, but the dominant noise is in Equation (6.1), serial correlation can be expected in the residuals and this may bias the parameter and variance estimates obtained by minimizing Equation (6.4). Two of the possible minimization criteria (transformation functions f) which have been proposed for the observation error model are described below; the choice between them depends on which of the noise terms in Equation (6.2) is dominant. These noise contributions may arise from:

- (i) Sampling error (z_y). The precision of CPUE as a relative index of abundance depends on the size of the catch taken. Kirkwood (1981) suggests that fishing may be approximated by a Poisson-like process (i.e. with variance proportional to the mean) and advocates the use of \sqrt{C} minimization, as this then provides asymptotic homoscedasticity of the residuals. The contribution of the CPUE data is then:

$$\begin{aligned} SS &= \sum_Y (\sqrt{C_Y} - \widehat{\sqrt{C_Y}})^2 \\ &= \sum_Y E_Y \{ \sqrt{(C/E)_Y} - \widehat{\sqrt{(C/E)_Y}} \}^2 \end{aligned} \quad (6.5)$$

CPUE data for whale populations have been analyzed by de la Mare (1986b), who found after linearly detrending the data, that the C.V.s of the residuals, did not decrease as the size of the catch increases, as would be expected if sampling error dominated. In such situations, the square-root transformation does not provide

homoscedastic residuals, as large catches receive too large a weighting, and estimates obtained by least squares tend to be less precise as a result. De la Mare (*op. cit.*) concluded that large catches do not improve the precision of the associated CPUE estimates as indices of abundance (although they do provide greater contrast in the data set and may thereby facilitate improved estimation of the other model parameters).

- (ii) Catchability fluctuations (v_y). Changing environmental factors, seasonal migration and behavioural/distributional changes tend to produce inter-annual catchability fluctuations. As one would expect catchability to be influenced by a large number of these factors, each of which may well be independent and have a multiplicative effect, the central limit theorem implies that the sum of the logarithms of the magnitudes of these factors approaches a normal distribution, and thus that taking logarithms is the most appropriate transformation to use:

$$SS = \sum_y [\ln(C/E)_y - \ln(\hat{C}/\hat{E})_y]^2 \quad (6.6)$$

Gulland (1956) noted that this transformation had the effect of stabilizing the variance of catch-effort data for the North Sea demersal trawl fishery. Furthermore, empirical distributions of CPUE are often highly positively skewed (Gulland *op. cit.*; Moyle and Lound 1960; Kimura 1981; Bannerot and Austin 1983), as would be expected if CPUE is log-normally distributed.

In the light of de la Mare's (*op. cit.*) results in particular, Equation (6.6) will be used in preference to equation (6.5) for the analysis that follows.

One of the above transformation functions f has always been selected, and an explicit choice of either process or observation error dominance made, when assessments of the hake resource off the west coast have been performed [e.g. Butterworth and Andrew (1984); Punt (1989e); Punt (1990a)]. However, there is no *a priori* reason why there should not be comparably substantial error in both the stock dynamics and the index of abundance (i.e. both process and observation error with neither dominating). There are, however, serious statistical difficulties associated with attempts to incorporate noise occurring simultaneously in both the catching process and in the stock dynamics, into a simple minimization criterion (Reed 1986).

Ludwig *et al.* (1988) describe an approximate method of incorporating both observation and process error into an estimation procedure. Following the nomenclature of these authors, this will be called the total least squares (TLS) method. The method involves minimizing the weighted sum of the squares of the estimated observation errors $\{\hat{v}_Y\}$ and the estimated process errors $\{\hat{u}_Y\}$:

$$SS = \frac{1}{1 - \lambda} \sum_Y \hat{u}_Y^2 + \frac{1}{\lambda} \sum_Y \hat{v}_Y^2 \quad (6.7)$$

Ludwig *et al.* (*op. cit.*) report that SS depends only weakly on the choice of λ [the ratio of σ_u^2 to $(\sigma_u^2 + \sigma_v^2)$] and, following their

example, λ is taken to be 0.5 here. (Note that λ has to be pre-specified; it cannot be estimated satisfactorily from the data.)

The $\{\hat{v}_y\}$ are treated as estimable parameters along with the other parameters of the model. Given a series of estimated observation errors $\{\hat{v}_y\}$ and the estimates of the values of the model parameters, estimates of the process errors can be obtained as follows:

- a) the estimated effort in year y , \hat{e}_y , is estimated by $E_y e^{-\hat{v}_y}$
- b) the biomass in year $y+1$ is estimated from the relationship between catch, effort and biomass by replacing effort by its estimated value, \hat{e}_y , i.e.

$$B_{y+1} = 2(C/E)_y e^{\hat{v}_y} / q - B_y \quad (6.8)$$

- c) u_y can then be calculated from Equation (6.1).

6.3.2 The contribution of survey biomass data

The survey biomass estimates are assumed to be relative indices of abundance and normally distributed about their expected values (Andrew *et al.* 1989). The contribution of the survey biomass data to the log-likelihood is thus:

$$SS_S = \sum_j \left[2(\sigma_j^S)^{-2} \right] [B_j^S - \Omega B_j]^2 \quad (6.9)$$

where Σ is summation over all available survey biomass estimates,

Ω is a constant of proportionality (the relative bias of the survey biomass estimates),

B_j^S is the j 'th absolute abundance estimate,

σ_j^S is an estimate of the standard error of B_j^S , and
 B_j is the resource biomass corresponding to survey estimate B_j^S
 (either B_k if the survey took place at the start of year k , or
 $(B_k + B_{k+1})/2$ if it was carried out in the middle of year k).

This formalism probably gives too much weight to the survey data because σ_j^S is an estimate of the sampling variability associated with the survey biomass estimate only. In reality, factors such as incomplete coverage of the area inhabited by the stock (so that differing proportions of the stock are in the survey area each year), as well as changes in the net opening, probably mean that the variance associated with B_j^S is somewhat greater than its sampling component.

6.4 Surplus-production model-estimation procedures considered

Punt (1988) shows that pure process error estimators (based on the Butterworth-Andrew as well as other models) perform very poorly unless further assumptions can be made regarding the relationship between CPUE at the start of exploitation and the average pristine size. For this to be possible, it is necessary to have CPUE data for the period during which substantial exploitation began. Such data are not available for the hake resource off the South African west coast (see Chapter 4), and so process error estimators have not been considered here. Further details of the performance of process error estimators can be found in Punt (1988).

In order to apply both observation error and TLS estimators, it is necessary to have an estimate of the biomass at the start of the CPUE

series (B_{1955}). Traditionally, two means of obtaining this estimate have been considered (Butterworth and Andrew 1984):

$B_{1955} = \phi K$ [where K is the (average) biomass of the pristine (pre-exploitation) resource], and

B_{1955} is estimated along with the other model parameters in the non-linear search.

The assumption that $B_{1955}=K$ (i.e. $\phi=1$) is not entirely realistic for the resource under consideration because substantial catches off the South African west coast occurred before 1955 [see Figure 3.1]. However, it is realistic to assume that $B_{1917}=K$ and project this biomass forward deterministically to 1955; ϕ is then estimated by B_{1955}/B_{1917} . Although this method can be used for observation error estimators, it cannot be applied with TLS estimators because they require effort data for each year to estimate the biomass series (see Equation 6.7).

Table 6.1 summarizes the fifteen surplus-production model-estimation procedures considered in this study. The algorithm used to select these procedures was to choose two "base case" procedures and to construct variants of these by altering a single feature of either "base case" procedure. The "base case" procedures incorporate the Schaefer and Fox forms of the surplus-production function, assume that $B_{1917}=K$, and use survey biomass data in the estimation procedure. These "base case" procedures are the surplus-production model-estimation procedures which are currently used to assess the resource (see Chapter 10).

6.5 Problems specifically associated with surplus production model-estimation procedures

- 1) The surplus production function is stationary. Walters (1986) defines a relationship to be stationary if the error distribution about the relationship is not serially correlated. Stationarity of the surplus production function is unlikely in practice because, for example, although it is assumed that surplus production is a function of exploitable biomass alone, in reality it is likely to be a function of the size-, sex- and age-structure of the population. In addition, interaction with the ecosystem can result in the "noise" about the surplus-production relationship being serially correlated.
- 2) The surplus production function does not take lags into account. None of the surplus production functions considered here incorporate a time-lag. In addition, Shepherd (1982) argues that some surplus production functions have unrealistic associated implications (specifically density dependent adult natural mortality).
- 3) The surplus production function is an unrealistic oversimplification. It is possible to obtain only very crude estimates of the historical and current status of the resource.

Table 6.1 : The 15 surplus-production model-estimation procedures considered in this study. The algorithm used to construct the acronyms is given in Appendix 11.A. Column P contains the estimable parameters.

Estimator	Method of estimating B_{1955}	Surplus production function	Survey /CPUE data	Acronym	P+
*Observation	Set to ϕK	Schaefer	Both	PROK-SC	r K
*Observation	Set to ϕK	Fox	Both	PROK-FO	r K
Observation	Set to ϕK	Pella-Tomlinson	Both	PROK-PT	r K h
Observation	Set to ϕK	Shepherd (1987)	Both	PROK-S1	α β
Observation	Set to ϕK	Shepherd (1982)	Both	PROK-S2	α β γ
Observation	Set to ϕK	Shepherd (1982)-L	Both	PROK-S3	α β γ
Observation	Set to ϕK	Ricker (1954)-L	Both	PROK-RI	α β
Observation	Estimate	Schaefer	Both	PROE-SC	r K
Observation	Estimate	Fox	Both	PROE-FO	B_{1917} r K
TLS	Estimate	Schaefer	Both	PR E-SC	B_{1917} r K
TLS	Estimate	Fox	Both	PRTE-FO	v_y r K
Observation	Set to ϕK	Schaefer	CPUE	PCOK-SC	v_y r K
Observation	Set to ϕK	Fox	CPUE	PCOK-FO	r K
Observation	Set to ϕK	Schaefer	Survey	PSOK-SC	r K
Observation	Set to ϕK	Fox	Survey	PSOK-FO	r K

* Base case procedure

+ All procedures which used CPUE data estimate the parameter q and all procedures which use survey data estimate the parameter Ω .

CHAPTER 7 - THE FULLY AGE-STRUCTURED MODEL-ESTIMATION PROCEDURES

Fully age-structured model-estimation procedures take explicit account of growth, mortality and recruitment. *Ad hoc* tuned VPA involves selecting fishing mortality matrices which are consistent with auxiliary information. The tuning process removes the problem associated with traditional VPA that estimates are essentially determined by the initial guesses for the terminal fishing mortalities. Six different approaches to tuning the most-recent-year terminal fishing mortalities are detailed, namely the Armstrong, Modified-Armstrong, Gamma, Modified Gamma, Laurec-Shepherd and Hybrid methods. *Ad hoc* tuned VPA can be criticized for a number of reasons related to statistical appropriateness. Integrated Analysis approaches overcome these problems at the expense of increased complexity and computer running time. The Integrated Analysis estimator considered here utilizes standard statistical methods to fit an age-structured population dynamics model to the available data. The approach is flexible as it can take almost any data (catch-at-age, catch-by-mass, effort and survey biomass estimates are considered) and additional assumptions (a relationship between spawner-biomass and recruitment is considered) into account when estimating the model parameter values.

7.1 Fundamentals

To highlight the basic principles behind age-structured model-estimation techniques, the simple case of a single cohort (a group of fish born at the same time, which is taken here, for convenience, to be the start of the year - this corresponds roughly to the hake spawning season - see Chapter 2) is considered. The fundamental equation governing the dynamics of a cohort is:

$$N_{a+1} = N_a e^{-Z_a} \quad (a = 1, \dots, m-1) \quad (7.1)$$

where N_a is the number of fish aged a years at the start of the year,

$Z_a = F_a + M_a$ is the instantaneous rate of mortality during the year from all causes (total mortality) on fish aged a years,

F_a is the instantaneous rate of fishing mortality on fish aged a years,

M_a is the instantaneous rate of natural mortality on fish aged a years, and

m is the oldest age considered.

The number of fish in a cohort thus declines due to the combined influence of natural effects [primarily predation - Gulland (1988)] and fishing. If Z is independent of age and year, the decline is exponential.

It can be shown [see, for example, Andrew (1986)], that the number of fish aged a years removed from a cohort by fishing is governed by the "Catch equation" (Baranov 1918):

$$C_a = \frac{F_a N_a}{F_a + M_a} (1 - e^{-Z_a}) \quad (7.2)$$

where C_a is the catch in numbers of fish aged a years .

Equations (7.1) and (7.2) can be generalized to deal with multiple cohorts. Equation (7.1) becomes:

$$N_{y+1,a+1} = N_{y,a} e^{-Z_{y,a}} \quad (y = y_1, \dots, n-1; a = 1, \dots, m-1) \quad (7.3)$$

where y_1 is the first year considered (for ease of presentation, y_1 will be taken to be 1 in the remainder of the discussion on age-structured model-estimation procedures), and

n is the most-recent-year.

Equation (7.2) becomes:

$$C_{y,a} = \frac{F_{y,a} N_{y,a}}{F_{y,a} + M_{y,a}} (1 - e^{-Z_{y,a}}) \quad (7.4a)$$

or [by substituting Equation (7.3) into Equation (7.4a)]:

$$C_{y,a} = \frac{F_{y,a} N_{y+1,a+1}}{F_{y,a} + M_{y,a}} (e^{Z_{y,a}} - 1) \quad (7.4b)$$

The primary purpose of age-structured model-estimation procedures is to provide estimates of the matrices $\{N_{y,a}:y=1,\dots,n;a=1,\dots,m\}$, $\{F_{y,a}:y=1,\dots,n;a=1,\dots,m\}$ and $\{M_{y,a}:y=1,\dots,n;a=1,\dots,m\}$. Estimation of these matrices using only Equations (7.3) and (7.4) and the observed catches-at-age $\{C_{y,a}:y=1,\dots,n;a=1,\dots,m\}$ is impossible because the problem is underdetermined [there are fewer equations $(2nm-n-m+1)$ than unknowns $(3nm)$]. Additional assumptions and/or data are required if the F, N and M matrices are to be estimated. Note that in order to perform these calculations, it is necessary that there are no zeros in the $C_{y,a}$ matrix for $y = n$ and for $a = m$. If there are zeros in the $C_{y,a}$ matrix for $a=m$, the $C_{y,a}$ matrix must be truncated to avoid them.

7.2. Traditional VPA

Traditional VPA (Virtual Population Analysis) [Gulland (1965), Murphy (1965)] is a simple technique for estimating the N and F matrices. The method is based on guessing the terminal fishing mortalities for each cohort in the catch-at-age matrix [i.e. $F_{y,m} : y = 1,\dots,n; F_{n,a}:a=1,\dots,m-1$] as well as independently specifying the matrix $\{M_{y,a}:y=1,\dots,n;a=1,\dots,m\}$ (usually setting $M_{y,a}=M$). Using the $n+m-1$ terminal fishing mortalities, Equations (7.4) and (7.5) can be

used successively to compute the remaining elements of the F and N matrices. The process of computing the F 's and N 's in this manner (see Figure 7.1) is known as back-calculation.

Note that traditional VPA, along with many of the other age-structured model-estimation procedures considered here, involves solving Equation (7.4b) for $F_{y,a}$ given values for $M_{y,a}$, $C_{y,a}$ and $N_{y+1,a+1}$. A number of approximate solutions for this Equation exist [e.g. Pope (1972); MacCall (1986)], but it has been solved numerically (using a bisection technique) throughout this study.

Traditional VPA is not considered further here, because the results obtained using this method are essentially arbitrary as they are functions of the terminal fishing mortalities selected [Pope and Shepherd (1982), Butterworth *et al.* (1990)]. However, VPA back-calculation does form an integral part of the *ad hoc* tured VPA methods.

7.3. The separability assumption

Fishing mortality is often assumed to be related to fishing effort in some way (certainly, at least, no fishing effort implies zero fishing mortality). However, because of the effects of fish and fishing vessel distribution patterns, and of gear selectivity (i.e. the effects of partial recruitment to the fishery), one unit of fishing effort applied at a particular place and time will not necessarily inflict the same fishing mortality on all age-classes (or even the population as a whole). To take age-specificity into account in a simple manner, the most common assumption made when applying age-

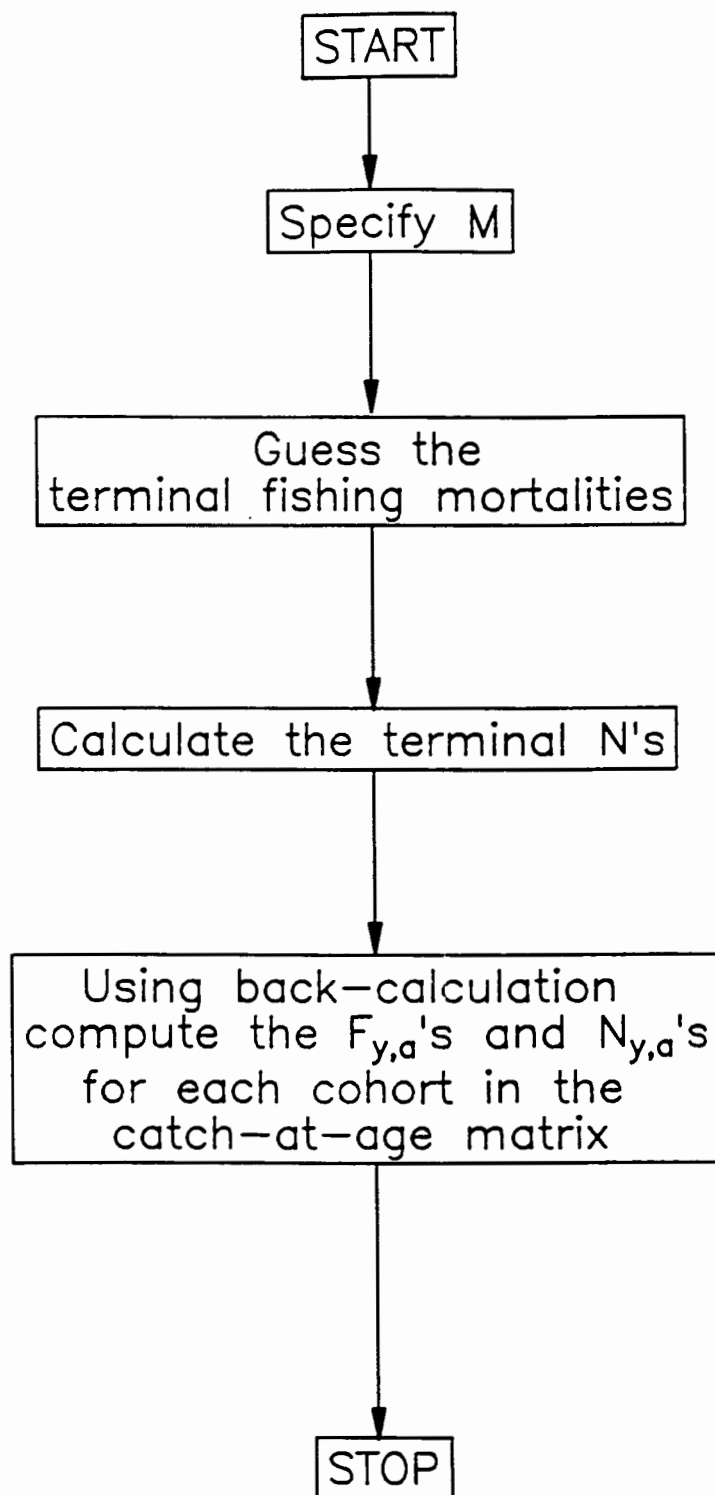


Figure 7.1 : The traditional VPA approach

structured techniques is that fishing mortality-at-age ($F_{Y,a}$) can be separated into an age-component which is common to all years (age-specific selectivity - S_a) and a year-component which is common to all ages within a year (fully-selected fishing mortality - F_Y). This assumption is justifiable if the fish and fishing vessel distribution patterns do not vary substantially from one year to the next. Two scenarios in which the separability assumption would not be justified (and which might be pertinent to the hake fishery) are:

- a) if the mesh size changes, and
- b) if the distribution of the fishing fleet across the fishing grounds changes as a result of changes in market demand for different sizes of fish.

Fully-selected fishing mortality is the quantity which is usually related to fishing effort. Mathematically, the relationship between fishing mortality-at-age and fishing effort can be described by the Equation:

$$F_{Y,a} = S_a F_Y = S_a f(E_Y) \quad (7.5)$$

where $F_{Y,a}$ is the instantaneous rate of fishing mortality on fish aged a years during year y ,

S_a is the selectivity of the fishery on fish aged a years ($0 \leq S_a \leq 1$),

F_Y is the fishing mortality on fully selected fish (i.e. fish with $S_a=1$) in year y ,

$f()$ is the relationship between F_Y and E_Y - usually assumed to be linear proportionality, i.e. $F_Y = q E_Y$, and

q is the catchability coefficient.

7.4 *Ad hoc* tuned VPA

A number of authors [e.g. Pope and Shepherd (1982); Butterworth *et al.* (1990)] have shown that the results of traditional VPA are essentially arbitrary because the terminal fishing mortalities (which are guessed rather than objectively chosen) completely determine the estimates of the F and N matrices. *Ad hoc* tuned VPA attempts to remove the arbitrariness of the traditional VPA solution by applying a "tuning algorithm", which re-estimates the terminal fishing mortalities iteratively until a solution which is consistent with auxiliary information is found. This iterative process removes the influence of the initial guesses of the terminal fishing mortalities on the final estimates. Figure 7.2 presents the *ad hoc* tuned VPA approach in the form of a flowchart.

Tuning algorithms utilize additional assumptions regarding the fishing mortalities in order to improve estimates of the terminal fishing mortalities. In general, the algorithms used to tune the most-recent-year terminal fishing mortalities $\{F_{n,a}:a=1,\dots,m-1\}$ utilize additional information such as effort data or survey abundance indices to estimate catchabilities-at-age for each year. Some plausible model is fitted to these catchabilities, and is used to predict the catchability-at-age vector for the most-recent-year, and thence the corresponding "tuned" values for the most-recent-year terminal fishing mortalities. Tuning of the oldest-age terminal fishing mortalities $\{F_{y,m}:y=1,\dots,n\}$ is usually achieved by assuming

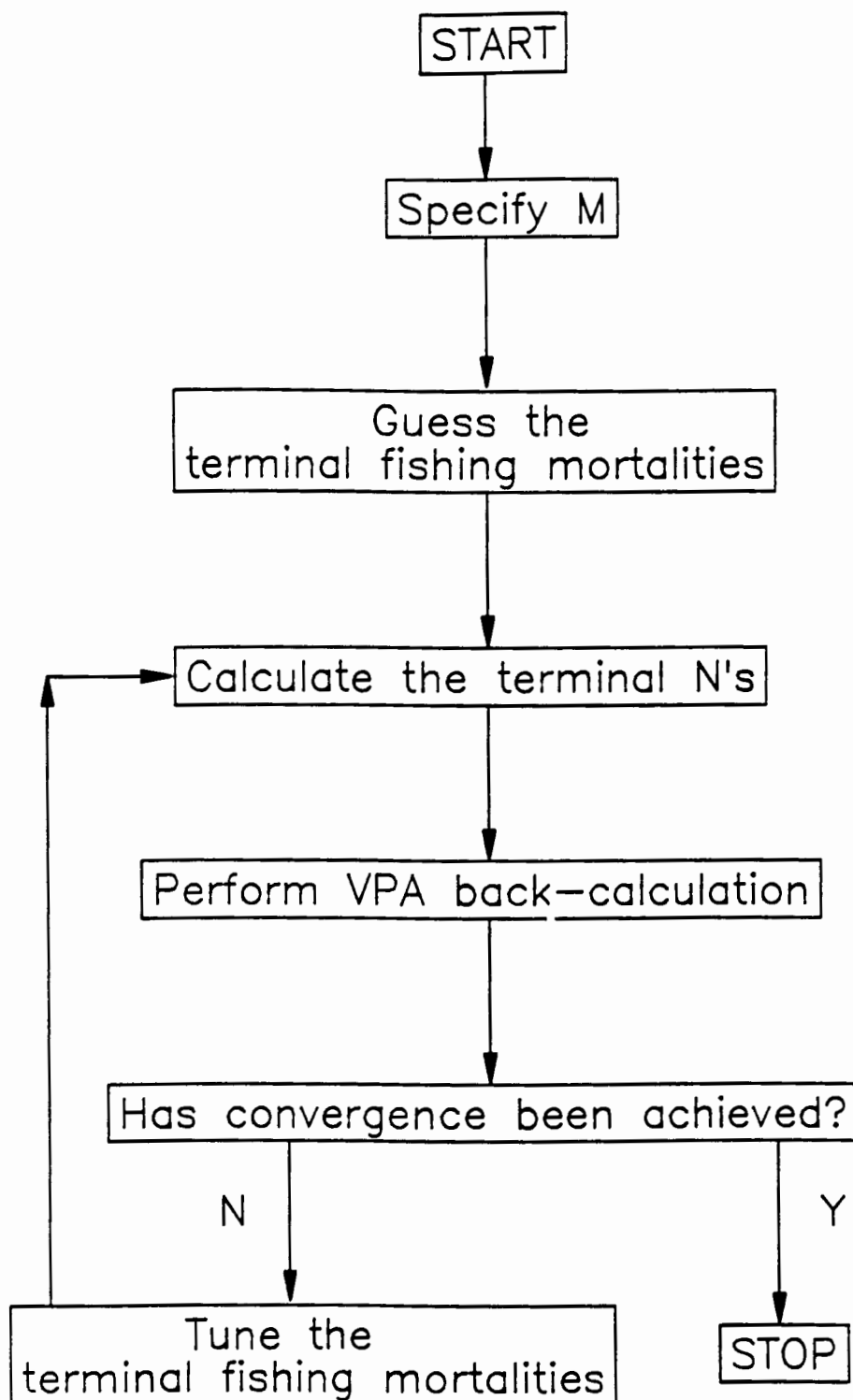


Figure 7.2 : The ad hoc tuned VPA approach

that the age-specific selectivity function (S_a) follows some plausible model.

A large number of algorithms for tuning the oldest-age and the most-recent-year terminal fishing mortalities exist [see the summaries presented in Pope and Shepherd (1985) and ICES (1988)]. Pope and Shepherd (op. cit.) investigated the performances of a number of algorithms for tuning the most-recent-year terminal fishing mortalities using Monte-Carlo simulation. They estimated the biases of a number of management-related quantities for each tuning algorithm which they considered, for a number of scenarios. They concluded that, in general, the tuning algorithms which they considered performed satisfactorily when catchability-at-age was constant, but could give biased estimates if catchability-at-age varied systematically.

Table 7.1 lists the eight *ad hoc* tuned VPA model-estimation procedures selected for investigation in this study, while Appendices 7.A and 7.B document the tuning algorithms used by these procedures. Selection of the eight procedures was based primarily on (a) their performances in the simulations carried out by Pope and Shepherd (1985) (suggesting the inclusion of the Laurec-Shepherd and Hybrid algorithms), and (b) their use by ICES (the Gamma algorithm). Selection of the other algorithms was based on achieving a "balanced design" (i.e. if an algorithm which fits a model to log-catchability was selected, an algorithm which fits the same model to the untransformed catchabilities was also included).

7.5 Integrated analysis approaches

Although it appears to be a vast improvement over traditional VPA, *ad hoc* tuned VPA can be criticized for a number of reasons:

- a) it does not seek to maximize any statistically-based objective function (unlike the surplus-production model-estimation procedures) and so the estimates obtained may not be minimum variance estimates (Butterworth 1988),
- b) it is very difficult to test the validity of the models underlying the tuning algorithms (Lewy 1988),
- c) estimation of the precision of the management quantities estimated using *ad hoc* tuned VPA can be difficult (Lewy 1988),
- d) it is not easy to utilize more than one data type in a tuning algorithm, and
- e) the tuning algorithms make the assumption that catch-at-age and effort data are measured without error, when it is well known that these data can be subject to considerable error.

Integrated analysis approaches are designed to overcome many of the problems of *ad hoc* tuned VPA though at the expense of increased complexity and computer running time. Most integrated analysis approaches are based on two principles: flexibility and the explicit consideration of the natural stochascity of the input data. The model underlying an integrated analysis approach should be flexible so that a number of alternative hypotheses regarding the factors which control the dynamics of the stock can be incorporated into it easily. In addition, it should be able to take virtually any data into account as auxiliary information. For example, the technique of

Fournier and Archibald (1982) can utilize the information on the following:

- the structure of the relationship between fishing effort and fishing mortality,
- the precision of the annual estimates of total catches (by number),
- the precision of the aging techniques used, and
- the structure of the relationship between spawner biomass and recruitment.

Other information which can be incorporated into an integrated analysis approach are:

- the effects of the environment on recruitment,
- mesh size changes, and
- survey biomass indices.

Because integrated analyses are based on firm statistical foundations, it is possible to use statistical techniques (such as investigations of the residuals for serial correlation and normality) to determine whether a model fit is adequate (Gudmunsson 1986). Another method of determining whether a model fit is adequate is by determining whether any generalization of the model is able provide a statistically superior fit to the data (Gudmunsson op. cit.).

The approach which has been investigated here is a variant of the "adaptive framework" (ICES 1988). Parameter values are selected which

minimize a sum of the squared differences between the observations and the model predictions (i.e. the objective function):

$$\sum_j \lambda_j \sum_y (O_{j,y} - P_{j,y})^2 \quad (7.6)$$

where λ_j is the weight assigned to data type j (as these weights cannot, in general, be estimated satisfactorily from the data, they are almost always pre-specified),

$O_{j,y}$ is the observed value for data type j in year y , and

$P_{j,y}$ is the value for data type j in year y predicted by the model.

These values of the parameters are then used to calculate the F , N and M matrices.

Thus, virtually any information (data or structural relationships) can be incorporated in this estimation process, because addition of new information involves merely adding an extra term to the objective function which is to be minimized. The actual number of estimable parameters depends on the specific assumptions made.

A major problem when applying an integrated analysis approach is how to obtain suitable initial approximations for the parameter values for the iterative minimization process. This is because an inappropriate choice of initial values could lead to convergence of the non-linear minimization routine to a local minimum of the function (7.6). This problem has been handled here by using the results of an application of an *ad hoc* tuned VPA based on the Laurec-Shepherd and log-normal tuning algorithms as the initial parameter

estimates. However, since this does not guarantee convergence to the global minimum, the results reported for applications of the integrated analysis approach may be biased to some extent.

7.5.1 The model

The description of the model underlying the integrated analysis approach investigated here will be compartmentalized as follows. The basic model which uses only catch-at-age data to estimate the model parameters will be described first. Following this, the extensions which are required in order to incorporate other information into the objective function will be described. Note that, although it is possible in principle to estimate all the model parameters using only catch-at-age data, results obtained on this basis are unreliable [Pope and Shepherd (1982), Cooke (1985b), Bergh (1986)].

7.5.1.1 The basic model

The integrated analysis approach is based on the two fundamental age-structure Equations:

$$\hat{N}_{y+1,a+1} = \hat{N}_{y,a} e^{-(\hat{M}_{y,a} + \hat{F}_{y,a})} \quad (7.7)$$

and

$$\hat{C}_{y,a} = \frac{\hat{N}_{y,a} \hat{F}_{y,a}}{\hat{M}_{y,a} + \hat{F}_{y,a}} (1 - e^{-(\hat{M}_{y,a} + \hat{F}_{y,a})}) \quad (7.8)$$

where $\hat{N}_{y,a}$ is the model estimate of the number of fish aged a years at the start of year y ,

$\hat{F}_{y,a}$ is the model estimate of the instantaneous rate of fishing mortality on fish aged a years during year y ,
 $\hat{M}_{y,a}$ is the model estimate of the instantaneous rate of natural mortality on fish aged a years during year y , and
 $\hat{C}_{y,a}$ is the model estimate of the catch of fish aged a years during year y .

In order to obtain an estimate of the matrix $\{C_{y,a}: y = 1, \dots, n; a = 1, \dots, m\}$, estimates of the numbers-at-age at the start of the $n+m+1$ cohorts represented in the catch-at-age matrix $\{\hat{N}_{y,1}: y=1, \dots, n; \hat{N}_{1,a}: a=2, \dots, m\}$ are required. The remainder of the numbers-at-age matrix is then obtained by applying Equation (7.7) repeatedly. [Note that estimates of the matrices $\{F_{y,a}: y=1, \dots, n; a=1, \dots, m\}$ and $\{M_{y,a}: y=1, \dots, n; a=1, \dots, m\}$ are required to do this.] Once an estimate of the numbers-at-age matrix has been obtained, Equation (7.8) can be used to obtain an estimate of the catch-at-age matrix.

The total number of parameters required (the $n+m-1$ initial numbers-at-age, the nm $M_{y,a}$'s and the nm $F_{y,a}$'s) vastly exceeds the nm $C_{y,a}$ observations. The problem is thus underdetermined, so that some simplifications and/or additional assumptions are required if the method is to be of any use. The assumption that natural mortality is time- and age-invariant (i.e. adding nm equations of the form $M_{y,a} = M$ to Equations 7.7 and 7.8) reduces the number of estimable parameters considerably. It is very difficult to estimate M using only catch-at-age data (even given additional data), so that in most applications of integrated analysis approaches this parameter is fixed using auxiliary information [Fournier and Archibald (1982), Pope and

Shepherd (1982), Bergh (1986)]. Although this assumption reduces the number of estimable parameters by nm , the problem remains underdetermined.

A overdetermined problem can be obtained if the separability assumption is invoked (i.e. adding nm equations of the form $F_{y,a} = S_a \cdot F_y$). This reduces the nm fishing mortality-at-age parameters to $n+m-1$ F_y and S_a parameters. Note that one of the S_a must be fixed, otherwise the product $S_a \cdot F_y$ is not uniquely determined (i.e. if the sets $\{S_a\}$ and $\{F_y\}$ minimize the objective function, then any sets of the form $\{\alpha S_a\}$ and $\{F_y/\alpha\}$ will also minimize it). It is now, in principle, possible to estimate the remaining $2n+2m-2$ model parameters using the nm observations as long as $2m+2n-2 > nm$.

In the implementation of the "adaptive framework" considered, the term in the objective function related to the catch-at-age data has been taken to have the form:

$$SS_1 = \lambda_1 \sum_{y=1}^n \sum_{a=1}^m [\ln(C_{y,a}/\hat{C}_{y,a})]^2 \quad (7.9)$$

where λ_1 is a weighting factor.

It is possible to argue in favour of many other ways of incorporating the contribution of the catch-at-age data in the objective function [see, for example, Fournier and Archibald (1982) and Deriso *et al* (1985)]. The particular form of (7.9) has been selected because of numerical simplicity. The form gives equal weight to equal relative differences, and is appropriate if the catch-at-age data are log-

normally distributed about their expected values and if the C.V.s of the catch-at-age data are independent of age and time.

7.5.1.2 Relationships between fishing mortality and fishing effort

If effort data are available and there is no strong reason to argue against a linear relationship between fishing effort and fully-selected fishing mortality, then a term of the following form can be added to the objective function.

$$SS_2 = \lambda_2 \sum_{y=1}^n (\ln \hat{F}_y - \ln \hat{q} - \ln E_y)^2 \quad (7.10)$$

where λ_2 is a weighting factor,

\hat{F}_y is the model-estimate of the fully-selected fishing mortality in year y , and

\hat{q} is the model-estimate of the catchability coefficient.

In this implementation, the parameter q is not included in the non-linear search, but is rather estimated using the method of Cooke (1983) [a closed form solution being provided by partial differentiation of (7.10) with respect to q and setting the result equal to zero].

A very small value of λ_2 means that little weight is to be placed on the fishing mortality versus fishing effort relationship when fitting the model to the data. If, on the other hand, $\lambda_2 = \infty$ and the separability assumption has been made, Equation (7.8) becomes:

$$\hat{C}_{y,a} = \frac{\hat{N}_{y,a} \hat{S}_a \hat{q} \hat{E}_y}{\hat{M}_{y,a} + \hat{S}_a \cdot \hat{q} \cdot \hat{E}_y} (1 - e^{-(\hat{M}_{y,a} + \hat{S}_a \cdot \hat{q} \cdot \hat{E}_y)}) \quad (7.11)$$

This amounts to assuming no error in measuring effort and no fluctuations in catchability-at-age. Setting λ_2 to ∞ has the advantage of reducing the number of estimable parameters by $n-1$ (the n F_y parameters are replaced by one q parameter), but has the disadvantage that the effort data are treated as being exact.

It is possible to allow for a more general relationship between fishing mortality and fishing effort. For example, the term:

$$SS_2 = \lambda_2 \sum_{y=1}^n (\ln \hat{F}_y - \ln q - \ln y - \ln E_y)^2 \quad (7.12)$$

allows for the possibility of a linear trend in log-catchability with time.

7.5.1.3 Relationships between spawner biomass and recruitment

If it is considered appropriate to assume some relationship between recruitment and spawner biomass, a term of the following form is added to the objective function.

$$SS_3 = \lambda_3 \sum_{y=1}^n (\ln \hat{N}_{y,1} - \ln \hat{N}_{y,1}^R)^2 \quad (7.13)$$

where λ_3 is the weighting factor, and

$\hat{N}_{y,1}^R$ is the estimate of the recruitment in year y obtained from the spawner biomass-recruitment relationship.

A number of spawner biomass-recruitment relationships have been proposed in the scientific literature. Of these, the two which have been considered here are as follows.

a) The Beverton-Holt (1957) form:

$$\hat{N}_{y,1}^R = \alpha SB_y / (\beta + SB_y) \quad (7.14)$$

where SB_y is the spawner biomass at the start of year y , and α, β are spawner biomass-recruitment relationship parameters.

b) The Ricker (1954) form:

$$\hat{N}_{y,1}^R = \alpha SB_y e^{-\beta SB_y} \quad (7.15)$$

where SB_y is the spawner biomass at the start of year y , and α, β are spawner biomass-recruitment relationship parameters.

If λ_3 is set to ∞ , recruitment becomes deterministically related to spawner biomass. This is essentially equivalent to making an "observation error" assumption (i.e. no error in the population dynamics equation), and reduces the number of estimable parameters by $n-2$ (the n $N_{y,1}$ parameters being replaced by the two spawner biomass recruitment relationship parameters).

It is possible, within the framework of Equation (7.13), to incorporate the effects of the environment on recruitment. This is achieved by adding a term incorporating the relevant environmental variable into Equation (7.13). Such an approach to modelling the effect of the environment on recruitment is objective because it is possible to determine whether or not incorporation of the

environmental variable provides a statistically significantly superior fit to the data.

7.5.1.4 Incorporation of survey biomass data

A number of the types of data collected from survey biomass cruises can be incorporated into the objective function and hence could possibly be used to improve parameter estimates. In this study, only survey estimates of biomass are considered. These data are incorporated into the objective function by adding the following term:

$$SS_4 = \sum (B_j^S - \hat{e} B_j)^2 / [2(\sigma_j^S)^2] \quad (7.16)$$

where B_j^S is the j 'th survey biomass estimate - assumed here to be a relative index of exploitable biomass and normally distributed about its expected value,

σ_j^S is the standard error of the j 'th survey biomass estimate,
 $\hat{e} B_j$ is the model estimate of the exploitable biomass at the time that the j 'th biomass survey was carried out, which is either:

$$\hat{e} B_j = \sum_{a=1}^m S_a w_a \hat{N}_{Y,a} \quad (7.17a)$$

if the j 'th survey was carried out at the start of year k ,
 or

$$\hat{e} B_j = \sum_{a=1}^m S_a w_{a+1/2} \hat{N}_{Y,a} \cdot e^{-(\hat{M}_{Y,a} + \hat{F}_{Y,a})/2} \quad (7.17b)$$

if the j 'th survey was carried out in the middle of year k [Note that these forms are simple approximations to exploitable biomass - a more appropriate but algebraically more complex approach is outlined in Chapter 11],

w_a is the mass of a fish aged a years (assumed temporally invariant), and

Ω is a constant of proportionality related to the bias of the survey biomass estimates, which is estimated from the data in the application considered here.

7.5.1.5 Incorporation of total catch-by-mass data

The estimates of annual catch-by-mass can be incorporated in the estimation of the model parameters by adding the following term to the objective function:

$$SS_5 = \lambda_4 \sum_{y=1}^n (C_y - \sum_{a=1}^m C_{y,a} w_{a+1/2})^2 \quad (7.18)$$

Setting λ_4 to ∞ , it is possible to reduce the number of estimable parameters by n . As it has been argued earlier that the estimates of catch-by-mass are by far the most accurate and precise data which are available for assessment purposes for the hake fisheries (see Section 4.1), all the estimators considered here assume that $\lambda_4 = \infty$ (i.e. that the actual catches-by-mass are exactly the same as those predicted by the model).

7.5.2. Simplifications

It is possible to further reduce the number of parameters that need to be estimated when fitting the model to the data, over and above the reductions that follow from setting the weighting factors to infinity, as discussed above.

7.5.2.1 Use of parsimonious parameterizations

It is often possible to reduce the number of estimable parameters considerably by means of adding extra relationships which make the values of some of the parameters deterministically related to the values of other parameters. One approach to achieving parsimony is to specify a functional relationship between S_a and a . For example, instead of treating all the $m-1$ S_a parameters as separately estimable in the non-linear search, S_a can be assumed to be knife-edged:

$$S_a = \begin{cases} 0 & \text{if } a < a_r \\ 1 & \text{otherwise} \end{cases} \quad (7.19)$$

where a_r is the age-recruitment to the fishery.

An assumption which is probably more appropriate for hake is that age-specific selectivity follows a logistic curve:

$$S_a = 1 / (1 + e^{-(a-a_r)/\delta_r}) \quad (7.20)$$

where a_r is now the age-at-50%-recruitment to the fishery, and

δ_r is a parameter which determines the width of the selectivity ogive.

This parameterization reduces the number of estimable parameters by $m-3$ (i.e. the $m-1$ S_a parameters are replaced by the parameters δ_r and a_r).

7.5.2.2 Specification of initial conditions

If it is known that the first year of the catch series corresponds to the year in which exploitation commenced, it may be reasonable to incorporate an assumption that the resource was at its pristine equilibrium biomass at that time with an age-structure corresponding to deterministic equilibrium, i.e.:

$$\hat{K} = \sum_{a=1}^m w_a \hat{N}_{1,a} \quad (7.21)$$

and

$$\hat{N}_{1,a+1} = \hat{N}_{1,a} e^{-\hat{M}_{1,a}} \quad (7.22)$$

These assumptions replace the m $N_{1,a}$ parameters by the parameter \hat{K} .

7.5.3. Further generalizations

The model and estimators developed above can be generalized considerably. For example, if a change in mesh size has occurred, the model can be extended by permitting S_a to be a function of time (i.e. $S_a \rightarrow S_{y,a}$). Other extensions of this sort include permitting mass-at-age to be a function of time as well as age.

7.6 Integrated analysis approaches considered here

It is not possible to investigate the performances of even a small subset of all the possible versions of integrated analysis approach described above because of the enormous computer time required to test these model-estimation procedures. In order to select which integrated analysis approaches to investigate, a "base case" version has been selected, and then five variants of this "base case" procedure constructed. The "base case" integrated analysis approach involves considering ages in the range 1 to 7, and all years from 1978 to year n . Furthermore, M is taken to be 0.3yr^{-1} , λ_4 is fixed at ∞ , survey biomass data are used to fit the model, the Ricker stock-recruitment relationship is fitted to the recruitment-spawner biomass data, and the age-at-maturity is assumed to be 4 years. The values assigned to the weight parameters (λ_1 , λ_2 , and λ_3) are obtained from the "base case" operating model. $\lambda_1=1/(0.4)^2$; $\lambda_2=1/(0.16)^2$ and $\lambda_3=1/(0.3)^2$. These weights correspond to the inverse of the catch-at-age, catchability, and recruitment variability in the age-structured operating model (see Chapter 11).

The variants considered [see Table 7.2] involve reducing the number of estimable parameter by assuming that selectivity follows a logistic curve, by changing the value of the rate of natural mortality, setting λ_3 to ∞ , and by replacing the Ricker form with the Beverton-Holt form.

7.7 Problems associated specifically with age-structured model-estimation procedures

There are a number of problems which are specific to age-structured model-estimation procedures. Some of these are likely to be pertinent to their application to data for this Cape hake resource.

- 1) The catch-at-age data may be subject to considerable error. Unlike catch-by-mass data which are obtained using census-like methods (see Section 4.1), catch-at-age data are determined using sampling methods (see Section 4.3), so that considerable sampling errors may arise.
- 2) Even given the assumption that natural mortality is time and age-invariant, all the age-structured techniques still require an estimate of the instantaneous rate of natural mortality (M). This is a very difficult parameter to estimate from the data, and the alternative of prespecification of what might in fact be an incorrect value may lead to an distorted perception of the status of the stock.
- 3) Most of the age-structured techniques provide highly imprecise estimates of the current status of the resource [see the estimates of current biomass obtained for the hake resource off northern Namibia by Butterworth *et al* (1990)]. Unfortunately, it is the current status of the resource which is usually of greatest importance when setting TACs.

Table 7.1 : *Ad hoc* tuned VPA model-estimation procedures considered in this study. Details of the tuning algorithms are given in Appendices 7.A and 7.B which also contain definitions of the terms used in this Table.

TUNING ALGORITHMS		p	Age range	Acronym
most-recent-year	oldest-age			
Laurec-Shepherd	Log-normal	2	1 - 7	AD1
Laurec-Shepherd	Log-normal	3	1 - 7	AD2
Modified Gamma	Log-normal	2	1 - 7	AD3
Armstrong	Normal	2	1 - 7	AD4
Gamma	Log-normal	2	1 - 7	AD5
Hybrid	Log-normal	2	1 - 7	AD6
Modified Armstrong	Normal	2	1 - 7	AD7
Laurec-Shepherd	Log-normal	2	0 - 9	AD8

Table 7.2 : Integrated analysis approaches considered in this study. All procedures investigated fit the catch-by-mass data exactly (i.e. assume that $\lambda_4 = \infty$) and make use of catch-at-age and survey biomass data. Procedures IN2 - IN6 are variants of the "base case" procedure.

Details	S-R relation	Parameters estimated	Acronym
"Base case"	Ricker	$S_a, N_{y,1}$	IN1
Observation error assumption	Ricker	S_a	IN2
Observation error Logistic selectivity	Ricker	a_r, δ_r	IN3
Beverton-Holt S-R relation	Beverton Holt	$S_a, N_{y,1}$	IN4
Logistic selectivity	Ricker	$N_{y,1}, a_r, \delta_r$	IN5
$M = 0.4\text{yr}^{-1}$	Ricker	$S_a, N_{y,1}$	IN6

Notes: a) The details of the "base case" procedure are given in Section 7.6

- b) All six procedures also estimate the following parameters:
- i) α, β - the stock-recruitment relationship parameters
 - ii) Ω - the bias of the survey biomass estimates
 - iii) $N_{1,a}$ - the numbers-at-age at the start of 1978
 - iv) q - the catchability coefficient
- c) $N_{y,1}$ - denotes the vector of recruitments (1979-1989)
- d) S_a - denotes the $m-1$ selectivity parameters (ages 1 to $m-1$)
- e) a_r, δ_r - denotes the logistic selectivity function parameters

APPENDIX 7.A: ALGORITHMS FOR TUNING THE MOST-RECENT-YEAR TERMINAL
FISHING MORTALITIES

ICES (1985) recommends that whenever possible, fleet-disaggregated tuning algorithms should be applied. However, as the fishery for hake off the South African west coast has been conducted almost exclusively by South African vessels since 1978 [see Table 4.3] (the period used for tuning), only algorithms which involve tuning the overall ("fleet-aggregated") fishing mortality matrix have been considered. If fleet-disaggregated tuning were required, the tuning algorithms considered here could readily be extended to deal with this [see, for example, Pope and Shepherd (1985) and Butterworth *et al.* (1990)]. Although the implementations of these algorithms give equal weight to each of the data points (i.e. each of the years) used to tune the most-recent-year terminal fishing mortalities, it is, in principle, possible to give lower weight to less recent data, which may be less reliable (see ICES 1988).

The algorithms are referenced by the names given to them by ICES. Where an existing algorithm has been altered, the algorithm is referred to by its original name preceded by the word "Modified" (e.g. the "Modified-Armstrong" algorithm).

Note that when applying approaches which assume trends in catchability-at-age and then attempt to estimate these trends, it is necessary for stability reasons to have a reference age for which it is assumed that catchability-at-age does not exhibit such a trend. Age 4 has been selected as this reference age.

I] The Armstrong algorithm (Pope and Shepherd 1985)

This algorithm is based on the assumptions that catchability-at-age is time invariant, and that the error in the fishing mortality-at-age versus fishing effort relationships can be modelled as follows:

$$F_{y,a} / E_y = \alpha + e_y \quad (7.A.1)$$

[Note that here, and in the formulae that follow: $e_y \sim N(0; \sigma^2)$]

The tuned value of $F_{n,a}$ is obtained using the formula:

$$\hat{F}_{n,a} = \frac{E_n}{n-1978} \sum_{y=1978}^{n-1} (F_{y,a}/E_y) \quad (7.A.2)$$

II] The Modified-Armstrong algorithm

This variant of the Armstrong algorithm assumes that the catchabilities-at-age exhibit a linear trend with time. The fishing mortality-at-age versus fishing effort relationships are thus modelled by the equation:

$$F_{y,a} / E_y = \alpha + \beta y + e_y \quad (7.A.3)$$

The values of the parameters " α " and " β " are obtained by linear regression of $\{(F_{y,a}/E_y): y=1978, \dots, n-1\}$ on $\{y: y=1978, \dots, n-1\}$.

The tuned value of $F_{n,a}$ is then obtained using the formula:

$$\hat{F}_{n,a} = E_n (F_{n,a}/E_n) = E_n (\hat{\alpha} + \hat{\beta} n) \quad (7.A.4)$$

III] The Gamma algorithm (ICES 1981)

This algorithm is based on the assumption that CPUE-at-age is related to population size according to a power law. Relationships of the following form are assumed:

$$\ln(\bar{N}_{y,a}) = \alpha + \beta \ln(C_{y,a}/E_y) + e_y \quad (7.A.5)$$

where $\bar{N}_{y,a}$ is the mean number of fish of age a years during year y :

$$\bar{N}_{y,a} = \frac{N_{y,a}}{M_{y,a} + F_{y,a}} (1 - e^{-(M_{y,a} + F_{y,a})}) \quad (7.A.6)$$

The value of the parameters " α " and " β " are obtained by linear regression of $\{\ln \bar{N}_{y,a} : y=1978, \dots, n-1\}$ on $\{\ln(C_{y,a}/E_y) : y=1978, \dots, n-1\}$. An estimate of $\bar{N}_{n,a}$ is then obtained using the formula:

$$\bar{N}_{n,a} = e^{\alpha} (C_{n,a}/E_n)^{\beta} \quad (7.A.7)$$

and the tuned value of $F_{n,a}$ using the formula:

$$\hat{F}_{n,a} = C_{n,a} / \bar{N}_{n,a} \quad (7.A.8)$$

IV] The Modified-Gamma algorithm (Pope and Shepherd 1985)

This algorithm is also based on the assumption that CPUE-at-age is related to population size according to a power law, but with the parameter " β " equal to one (see Equation 7.A.5). This means that, unlike the case for the Gamma algorithm, catchability is not a function of population size. Relationships of the following form are assumed:

$$\ln(\bar{N}_{Y,a}) - \ln(C_{Y,a}/E_Y) = \alpha + e_Y \quad (7.A.9)$$

where $\bar{N}_{Y,a}$ is defined as in Equation (7.A.6).

An estimate of $\bar{N}_{n,a}$ is obtained using the formula:

$$\bar{N}_{n,a} = C_{n,a}/E_n \prod_{y=1978}^{n-1} (\bar{N}_{Y,a} E_Y/C_{Y,a})^{1/(n-1978)} \quad (7.A.10)$$

and the tuned value of $F_{n,a}$ using the formula:

$$\hat{F}_{n,a} = C_{n,a}/\bar{N}_{n,a} \quad (7.A.11)$$

VI The Laurec-Shepherd algorithm (Laurec and Shepherd 1983)

This algorithm is based on the assumptions that log-catchability-at-age is time invariant and that the error in the fishing mortality-at-age versus fishing effort relationships are of the form:

$$\ln(F_{Y,a} / E_Y) = \alpha + e_Y \quad (7.A.12)$$

The tuned value of $F_{n,a}$ is obtained using the formula:

$$\hat{F}_{n,a} = E_n \prod_{y=1978}^{n-1} (F_{Y,a}/E_Y)^{1/(n-1978)} \quad (7.A.13)$$

VII The Hybrid algorithm (Pope and Shepherd, 1985)

This variant of the Laurec-Shepherd algorithm assumes that log-catchability-at-age has been changing exponentially with time. The fishing effort versus fishing mortality-at-age relationships are modelled by the equation:

$$\ln(F_{Y,a} / E_Y) = \alpha + \beta Y + e_Y \quad (7.A.14)$$

The values of the parameters " α " and " β " are obtained by linear regression of $\{\ln(F_{y,a}/E_y): y=1978, \dots, n-1\}$ on $\{y: y=1978, \dots, n-1\}$. The tuned value of $F_{n,a}$ is then obtained using the formula:

$$\hat{F}_{n,a} = E_n (F_{n,a}/E_n) = E_n \exp(\hat{\alpha} + \hat{\beta}n) \quad (7.A.15)$$

**APPENDIX 7.B: ALGORITHMS FOR TUNING THE OLDEST-AGE TERMINAL FISHING
MORTALITIES**

Two very simple approaches to tuning the oldest-age terminal fishing mortalities have been considered. Both are based on the assumption that the age-specific selectivity function is flat (i.e. unchanging with age) for the oldest (p+1) ages. The difference between the two approaches is in the error structures assumed. The two algorithms are as follows.

I) The Normal algorithm

$$F_{Y,m} = \frac{1}{p} \sum_{a=m-p}^{m-1} F_{Y,a} \quad (7.B.1)$$

II) The Log-normal algorithm

$$F_{Y,m} = \frac{1}{p} \sum_{a=m-p}^{m-1} (F_{Y,a})^{1/p} \quad (7.B.2)$$

Algorithm (7.B.2) has been used with algorithms for tuning the most-recent-year terminal fishing mortalities which assume a relative (log-normal) error model (i.e., the Laurec-Shepherd, Hybrid, Gamma and Modified-Gamma algorithms), while algorithm (7.B.1) has been used with those algorithms which assume an absolute (normal) error model (i.e., the Armstrong and the Modified Armstrong algorithms).

CHAPTER 8 - PARTIALLY AGE-STRUCTURED MODEL-ESTIMATION PROCEDURES

Partially age-structured model-estimation procedures are based on models which, although fundamentally age-structured, collapse to a form which contains only age-aggregated quantities. This property follows from judicious choices for the functional forms used to model growth, mortality and recruitment. One such model is detailed and is incorporated into an observation error estimator.

8.1 The basic Deriso-Schnute model

For a number of marine species, growth after sexual maturity can be well represented by the Brody growth curve:

$$w_{a+1} = (1+\rho)w_a - \rho w_{a-1} \quad (8.1)$$

where ρ is the Brody growth coefficient (Ricker 1975).

Now, Deriso (1980) defines the exploitable biomass at the beginning of y . year B_y , as:

$$B_y = \sum_{a=a_r}^{\infty} w_a N_{y,a} \quad (8.2)$$

where $N_{y,a}$ is the number of fish aged a years at the start of year y ,
and

w_a is the average mass of an individual fish aged a years at the start of the year (assumed to be independent of year).

Equation (8.2) assumes that all fish aged a_r and older have recruited to the fishery, and that all those younger have not (the knife-edge selectivity assumption). [Deriso (1980) also describes a variation of his model in which a "pool" of young fish is used, rather than knife-

edge selectivity, but for simplicity this is not considered here.] The summation in Equation (8.2) is extended to age infinity because it can be assumed that very few adults attain the maximum physiologically possible age (Deriso op. cit.).

Now, if the same rate of annual survival in year y (τ_y) applies to all exploitable fish, the number of fish aged a_r and older in year $(y+1)$ can be related to the number in year y and the number of incoming recruits by:

$$\sum_{a=a_r}^{\infty} N_{y+1,a} = \sum_{a=a_r}^{\infty} \tau_y N_{y,a} + R_{y+1} \quad (8.3)$$

where R_{y+1} is the number of recruits during year y which are present in the exploitable biomass at the beginning of year $(y+1)$, but were not vulnerable to fishing during year y , and

$\sum_{a=a_r}^{\infty} \tau_y N_{y,a}$ represents the number of exploitable fish surviving from year y to year $(y+1)$.

Multiplying Equation (8.1) by $N_{y+1,a+1}$ and rearranging terms gives:

$$N_{y+1,a+1} w_{a+1} = (1+\rho) N_{y+1,a+1} w_a - \rho N_{y+1,a+1} w_{a-1} \quad (8.4)$$

Then, replacing $N_{y+1,a+1}$ by $(N_{y,a} \tau_y)$ in the first term and by $N_{y-1,a-1} \tau_{y-1} \tau_y$ in the second term on the RHS of Equation (8.4) gives:

$$N_{y+1,a+1} w_{a+1} = \begin{cases} (1+\rho) N_{y,a} \tau_y w_a - \rho N_{y-1,a-1} \tau_{y-1} \tau_y w_{a-1} & (a \geq a_r + 1) \\ (1+\rho) N_{y,a_r} \tau_y w_a - \rho N_{y-1,a_r-1} \tau_{y-1} e^{-M} w_{a_r-1} & (a = a_r) \end{cases} \quad (8.5)$$

where M is the instantaneous rate of natural mortality (assumed to be independent of age and year).

Summing the terms in Equation (8.5) from a_r to infinity yields:

$$\begin{aligned} \sum_{a=a_r}^{\infty} N_{Y+1,a+1}w_{a+1} &= (1+\rho) \sum_{a=a_r}^{\infty} N_{Y,a}\tau_Y w_a - \rho \sum_{a=a_r+1}^{\infty} N_{Y-1,a-1}\tau_Y\tau_{Y-1}w_{a-1} \\ &\quad - \rho N_{Y-1,a_r-1}\tau_Y e^{-M}w_{a_r-1} \end{aligned} \quad (8.6)$$

and substituting for the summations [from Equation (8.2)] gives:

$$B_{Y+1} - w_{a_r}R_{Y+1} = (1+\rho)B_Y\tau_Y - \rho\tau_Y\tau_{Y-1}B_{Y-1} - \rho\tau_Y R_Y w_{a_r-1} \quad (8.7)$$

Re-arrangement of Equation (8.7) then produces the basic Deriso-Schnute model:

$$B_{Y+1} = B_Y(1+\rho)\tau_Y - \rho\tau_Y\tau_{Y-1}B_{Y-1} - \rho\tau_Y R_Y w_{a_r-1} + R_{Y+1}w_{a_r} \quad (8.8)$$

8.2 Selection of functional forms

It now only remains to specify functional forms for τ_Y and R_Y .

The recruitment in numbers in year y , R_Y , depends on the biomass of mature adults a_r years before (as it is assumed that each individual takes a_r years to recruit into the fishery). The number of live births in year y , $N_{Y,0}$, can be modelled by a Beverton-Holt (1957) stock-recruitment relationship:

$$N_{Y,0} = \frac{\alpha B_Y}{(\beta + B_Y)} \quad (8.9a)$$

or a Ricker (1954) stock-recruitment relationship:

$$N_{y,0} = \alpha B_y e^{-\beta B_y} \quad (8.9b)$$

Note that these equations assume equality of the age-at-maturity and the age-at-recruitment to the fishery.

Converting either of these to the biomass recruiting to the fishery in year y requires that allowance be made for a_r years of natural mortality, and that the result be multiplied by w_{a_r} (the mass of an individual fish aged a_r years old at the start of the year). The recruiting biomass is thus:

$$w_{a_r} R_y = w_{a_r} N_{y,0} \exp(-a_r M) \quad (8.10)$$

The total survival rate, τ_y , is equal to $\exp(-(M+F_y))$, where F_y (the instantaneous rate of fishing mortality in year y - assumed to be independent of age) must satisfy the VPA catch equation:

$$\frac{C_y}{B_y} = \frac{F_y}{F_y + M} [1 - e^{-(F_y+M)}] \quad (8.11)$$

where C_y is the actual catch-by-mass in year y .

The equation above assumes that the rates of fishing and natural mortality are independent of age (after the age-at-recruitment a_r). This is why the LHS of the equation can be written in terms of biomass, rather than numbers as is conventional. The fishing mortality in year y is calculated by solving Equation (8.11) for F_y given values for M , C_y and B_y . (This is achieved using a bisection technique in the applications of the procedure investigated in this study.)

The parameters M , a_r and ρ can be estimated from auxiliary information. For applications to hake, M is taken to be 0.3yr^{-1} (as has been conventionally assumed when conducting hake assessments), a_r to be 3 yr and ρ is estimated from the mass-at-age data.

8.3 Partially age-structured model-estimation procedures considered

Deriso (1980) and Schnute (1985) constructed their estimators by making a process error assumption (i.e. the dominant error is in the resource dynamics equation). However, simulation studies have shown that such estimators perform particularly poorly [Ludwig and Walters (1985), Punt (1988)] and so only observation error estimators have been considered here. It is possible, in principle, to construct TLS estimators (see Ludwig *et al.* 1988) for the Deriso-Schnute model. However, considering the poor performance of model-estimation procedures based on this model in the studies of Ludwig and Walters (*op. cit.*) and Punt (*op. cit.*), the extra computation required would not seem likely to be warranted.

In order to construct an observation error estimator for this model, it is assumed that recruitment [Equation (8.9a) or Equation (8.9b)] and the stock dynamics [Equation (8.7)] are deterministic. Assuming that E_y (the observed effort in year y) is lognormally distributed about F_y/q , and that the survey biomass estimates are relative indices of abundance which are normally distributed about their expected values, leads to minimization of the following function to obtain the estimates of the model parameters:

$$-\ln L = \sum_Y \left[\ln(\sigma) + \frac{v_Y^2}{2\sigma^2} \right] + \sum_j \left[2(\sigma_j^S)^2 \right]^{-1} [B_j^S - \hat{\Omega} B_j]^2 \quad (8.12)$$

where \sum_Y is summation over all years (y) for which effort data are available,

\sum_j is summation over all available survey biomass estimates (j),

Ω is a constant of proportionality (the relative bias of the survey biomass estimates),

B_j^S is the j'th survey biomass estimate,

σ_j^S is an estimate of the standard error of B_j^S ,

$\hat{\Omega} B_j$ is the resource biomass corresponding to survey biomass estimate B_j^S (either B_k if the survey took place at the start of year k, or $(B_k + B_{k+1})/2$ if it was carried out in the middle of year k),

v_Y is the observation error in year y:

$$v_Y = \ln(F_Y) - \ln(qE_Y), \text{ and}$$

σ is an estimate of residual variance:

$$\sigma^2 = \sum_Y v_Y^2 / \sum_Y 1$$

As a result of the poor performance of partially age-structured model-estimation procedures in previous investigations [Ludwig and Walters (1985), Punt (1988)], only two of these model-estimation procedures are considered here. Both assume that the hake biomass in 1917 (and in all preceding years) was equal to the unexploited equilibrium biomass (K). The two estimators differ in the form of the

stock-recruitment relationship assumed (either Ricker or Beverton-Holt).

8.4 Problems specific to partially age-structured model-estimation procedures

The major problems which are associated with the application of partially age-structured model-estimation procedures to the hake resource are related to the simplifications necessary to derive the basic Deriso-Schnute model [Equation (8.8)]:

- 1) Growth is assumed to follow the Brody equation (Equation 8.1). Although the Brody growth equation may fit growth after maturity reasonably well (Ricker 1975), the fishing mortality on immature fish in many fisheries (including the hake fishery) is not insubstantial, so this growth equation may not provide an adequate fit to the data.
- 2) The age-at-recruitment and age-at-maturity are assumed to be the same.
- 3) Fishing mortality is age-independent. This assumption is not likely to be realistic for hake (see Chapter 10).

CHAPTER 9 - HARVESTING STRATEGIES

All of the harvesting strategies (catch control laws) considered in this thesis are based on achieving fundamentally biological objectives. Three harvesting strategies (the $f_{0.1}$, $MAC_{0.1}$ and RY strategies) are considered for the non-age-structured management procedures. The $f_{0.1}$ strategy sets TACs in an attempt to drive the resource to a target biomass level a little above the MSY level, by applying a constant fishing mortality. The $MAC_{0.1}$ strategy is identical to the $f_{0.1}$ strategy, except that the TAC cannot exceed the equilibrium catch for the target biomass level. The aim of this upper bound is to prevent possible subsequent (and industrially undesirable) reductions in TAC, to achieve lower inter-annual catch variability, and to avoid the possibility of setting inappropriately large TACs if the estimate of current relative stock size is positively biased. The RY harvesting strategy sets the TAC to the estimate of the current replacement yield in an attempt to keep the biomass at its current level. Two harvesting strategies (the $F_{0.1}$ and $F_{status-quo}$ strategies) are considered for age-structured management procedures. The $F_{0.1}$ strategy sets TACs using this (i.e. $F_{0.1}$) target fishing mortality level. The particular $F_{status-quo}$ strategy considered here sets TACs in an attempt to keep fishing mortality at its current level. Two short-cut methods ("SHOT" and "DROP"), which can be used to provide $F_{status-quo}$ strategy TACs in the absence of catch-at-age and effort data, are detailed.

9.1 Introduction

In order to achieve the objectives decided upon for a resource to be harvested (see Section 1.3), it is necessary for scientists to provide management advice and for the management agency to enforce the regulations which result from that advice. As noted in Section 1.3, however, no matter what management procedure is used, it is impossible to optimize performance on all possible management objectives simultaneously - some trade offs are inevitable.

Harvesting strategies are usually designed to regulate either the fishing mortality or the total catch. Although it is possible in principle to regulate the natural mortality (predation) on a species by culling its predators, there are many problems encountered when attempting to quantify predator-prey relationships [see, for example,

problems encountered with attempts to rationalize the culling of Cape fur seals (Butterworth *et al.* 1988)], so that this option is seldom considered when alternative harvesting strategies are compared.

Harvesting strategies can be divided into two types.

- a) "Biological" harvesting strategies. These strategies aim to stabilize the fishing mortality on or the biomass of a resource (in an average sense, given natural fluctuations) at some target level. This target level is usually selected to achieve an objective which is primarily biologically based. The traditional target level (e.g. Russel 1931) is that biomass (or fishing mortality) which provides the maximum sustainable yield (MSY) under average environmental conditions. Other target levels are often selected, whose rationale is frequently the avoidance of either growth or recruitment overfishing (see Section 1.3), e.g. fishing mortality levels $F_{0.1}$, or F_{low} , F_{med} , and F_{high} respectively.
- b) "Economic" harvesting strategies. Economic harvesting strategies recognize that fishing is primarily an economic activity and that management cannot ignore economic considerations if it is to be appropriate and effective. The primary economic variables are the selling price and harvesting cost per unit mass of fish landed. Economic considerations lead to different perceptions of how total allowable catches (TACs) should be set. For example, if harvesting costs are high, maximum profit may be achieved at a

fishing mortality which is much lower than that at which MSY is achieved.

Although it is widely recognized that economics plays an important role in determining the dynamics of a fishery, it is very difficult to base management recommendations on the effects of economic factors on a fishery. This is because the primary variables (cost and price) are unlikely to be constant (even after adjusting for inflation). They are more likely to fluctuate in response to market forces (which are in turn related to the availability of other food items) as well as to the current abundance of the species being managed. Allowance for the effects of capital costs, e.g. replacement of older vessels, make the problem even more complicated. For this reason, only "biological" harvesting strategies are considered in this study. "Economic" harvesting strategies are discussed further in other texts, e.g. Clark (1976, 1985).

A biological reference point is a biological y based fishing mortality level. As the following discussion will consider only situations in which the resource is in equilibrium, it is possible to refer to the biomass at which the resource would be stabilized under harvesting strategy R (B_R) as the "biological reference point". The equilibrium catch corresponding to B_R will be referred to as C_R , and the corresponding equilibrium effort as E_R . The total allowable catch in year (n+1) under the strategy which will eventually stabilize the biomass at B_R will be referred to as $TAC_R(n+1)$. The replacement yield for a resource is the annual catch which will maintain the population at the end of any given year at the same level as it was at the start

of that year (again in a deterministic dynamics context). In the absence of transient effects arising from non-equilibrium age-structure, the replacement yield is identical to the sustainable yield at that population level.

9.2 The objective of Maximum Sustainable Yield

A traditional aim of fishery management is to stabilize the resource at that biomass (B_{MSY}) which provides the maximum sustainable yield (MSY) available under average environmental conditions. However, stabilizing the resource at B_{MSY} is not necessarily the most appropriate objective in the presence of environmental fluctuations (which lead to poor recruitment on occasions), because in "bad" years the biomass will drop below B_{MSY} , resulting in classification of the stock as "biologically overexploited". This biomass reduction is particularly dangerous for a constant catch harvesting strategy (see Section 9.3.3) as maintaining a catch of MSY in these circumstances will continue to deplete the stock because MSY is not sustainable at population sizes lower than B_{MSY} .

9.3 Harvesting strategies for surplus-production and partially age-structured model-estimation procedures

The aim of the harvesting strategies described below is to drive the current biomass B_n [indexed by the current CPUE, $(C/E)_n$] to a biological reference point, B_R . Most of these strategies aim to achieve optimal performance (in a deterministic context) on only one of the many possible objectives for management.

9.3.1 The "constant effort" strategy

A common harvesting strategy is that of regulation by fixing fishing effort at a constant value. Here, we consider an implementation of this approach under which the managers do not regulate effort as such, but set TACs which are the catches which (it is estimated) will be taken under that level of effort. Convergence to B_R can be achieved by setting TACs according to the formula:

$$\begin{aligned} \text{TAC}_R(n+1) &= B_n g(B_R)/B_R & (9.1) \\ &= (C/E)_n E_R \end{aligned}$$

where $\text{TAC}_R(n+1)$ is the recommended TAC for year $n+1$, and

$g(B_R)$ is the equilibrium yield corresponding to B_R .

This strategy is optimal (in a deterministic context) in minimizing the variance of the annual fishing effort (at zero). Thus, if the fishing fleet can be reduced or increased to the size required to apply E_R units of effort, then this size need not change further over time.

For this strategy, B_R is a stable equilibrium point as long as $g(B)$ is not depensatory in the region $[B_n, B_R]$, i.e. $g''(B) < 0$ in this interval. Beddington and May (1977) show by simulation that even when considerable environmental noise is present, this harvesting strategy will still (on average) stabilize the population at B_R .

9.3.2 The "fixed escapement" strategy

An alternative to the constant effort strategy is the "fixed escapement" strategy [Clark (1976); Walters (1986)], in terms of which TACs are set according to the formula:

$$TAC_R(n+1) = \begin{cases} B_n + g(B_n) - B_R & \text{if } B_n + g(B_n) > B_R \\ 0 & \text{otherwise} \end{cases} \quad (9.2)$$

This strategy will effect a return to a biomass of B_R as quickly as possible, so that, if $B_R = B_{MSY}$, the total catch achieved by this harvesting strategy is the maximum possible (in a deterministic context). However, this strategy is unlikely to be acceptable to the fishing industry because of:

- a) the immediate large operating losses caused by the complete halt to fishing that will follow years of poor recruitment, and
- b) the very high inter-annual catch fluctuations that are almost certain to result.

9.3.3 The "constant catch" strategy

The "constant catch" strategy involves setting the TAC to C_R , where $B_R > B_{MSY}$. In a situation of purely compensatory density dependence, there will be a further (unstable) equilibrium biomass $B_R < B_{MSY}$ for which the sustainable yield is also C_R . Deterministically, if B_n is greater than B_R , the population stabilizes at B_R . However, if B_n is less than B_R , the population becomes extinct. Unless $B_R \gg B_{MSY}$ (and

hence $C_R \ll MSY$), the presence of environmental noise will ultimately lead to extinction of the stock (Beddington and May 1977). For this reason, the constant catch strategy is undesirable, even though it is optimal in terms of minimizing catch fluctuations.

9.3.4 The "Maximum allowable catch" strategy

A compromise between the constant catch and constant effort strategies is the MAC (Maximum Allowable Catch) strategy (Butterworth 1987), which sets TACs according to the formula:

$$TAC_{R(n+1)} = \begin{cases} \frac{q(B_R)}{B_R} B_n & \text{if } B_n < B_R \\ g(B_R) & \text{otherwise} \end{cases} \quad (9.3)$$

Like the constant effort strategy, this strategy stabilizes the population at B_R (deterministically), but if $B_n > B_R$ the TAC is held fixed at $g(B_R)$. This strategy is similar to the "New Management Policy" for whale management (Allen 1976). It tends to reduce inter-annual TAC fluctuations, as the MAC is independent of B_n if $B_n \geq B_R$, and so in this situation is altered only following a revised estimate of $g(B_R)$, i.e. when more data for estimation purposes become available. The precision with which the MAC can be determined may be greater than that for the TAC corresponding to the constant effort strategy, as B_n need not be estimated (although it must be known that $B_n > B_R$). As a result, this strategy should be more robust to model error. It also has the advantage that the TAC cannot be set larger than the estimate of $g(B_R)$, even if B_n is estimated to be near K . This reduces the possibility of unintended overexploitation if B_n is imprecisely or inaccurately determined.

Another objective of this harvesting strategy is to place an upper bound on the size of the TAC, because the fishing industry usually has much greater difficulties accepting a reduction in the TAC compared to an increase. For this reason, TACs in excess of MSY are undesirable. The MAC strategy seeks to avoid this situation.

9.3.5 The "maximum effort" strategy

The maximum effort strategy can be used if $B_n + g(B_n) > B_R$. Deterministically, this involves setting effort as high as possible until $B_n + g(B_n) = B_R$, and then harvesting the resource at $g(B_R)$. Although approaching the optimal solution in terms of maximal catch if $B_R = B_{MSY}$ (see Section 9.3.2), this strategy has serious limitations because an unregulated increase in effort encourages the development of overcapacity, and hence pressure from the industry to maintain catches in excess of sustainable yields once B_R is reached. In addition, estimates of $g(B_R)$ and B_n during the declining phase of a fishery are usually positively biased and imprecise due to insufficient contrast in the data (Hilborn 1979). This strategy corresponds to the fixed escapement strategy, except that it may not be possible to catch the TAC under that strategy [$B_n + g(B_n) - B_R$] given the maximum effort available.

9.3.6 The "replacement yield" (RY) strategy

Harvesting strategies such as those which are described above are optimal (deterministically) in achieving one particular management objective. However, this presupposes perfect information, i.e. that the relevant population dynamics parameters are known exactly. In reality,

the quantities required by these strategies [i.e. $g(B_R)$, B_R , E_R , B_n and C_R] have to be estimated from the available data, and are therefore often poorly determined, so that the "optimality" of these approaches is often of little consequence.

De la Mare and Cooke (1983) have shown that the current replacement yield, $g(B_n)$, can be estimated more reliably than many other management variables in certain circumstances. The replacement yield strategy sets the TAC for year $n+1$ to $g(B_{n+1})$ and thus "plays safe" in an attempt to avoid unintended depletion of the resource because of overestimation of $g(B_R)$. However, it essentially chooses the current population size as the target level and so does not attempt to drive the resource to any other level. This means that if the resource is larger than B_{MSY} , the average yield is lower (possibly much lower) than that which would be achieved by the other harvesting strategies. More seriously, if the resource has already been driven to biologically overexploited (and less productive) levels, the replacement yield strategy does not allow the resource to recover.

9.3.7 Harvesting strategies considered

Only three of the harvesting strategies described above (the "constant effort", maximum allowable catch and replacement yield strategies) will be considered further. The "constant effort" strategy has been selected as it is the harvesting strategy currently used as the basis for the management of the hake resource off the South African west coast, and because it seems to achieve a better balance between the various management objectives than do the two other "deterministically

optimal" harvesting strategies (the "constant catch" and "fixed escapement" strategies). The maximum allowable catch strategy has been included because it attempts to combine some of the best features of the "constant effort" and "constant catch" strategies, and because it may be more reliable than either. Finally, the replacement yield strategy has been chosen because, although it is not (deterministically) optimal in any sense, it may be more robust to problems of estimation than the other two strategies.

9.3.8 The $f_{0,n}$ harvesting strategy

Although three harvesting strategies have been selected for investigation, only one (the replacement yield strategy) is as yet fully specified. The other two strategies require the choice of an appropriate target biomass level, B_R . As noted above, a target level of B_{MSY} can be a dangerous selection given estimation imprecision and other uncertainties regarding the yield function and current biomass level, so that a more conservative choice of target level seems appropriate.

The $f_{0,n}$ strategies are not necessarily economically or biologically optimal, but do stabilize the biomass at a level greater than B_{MSY} , thus providing a hedge against the problems associated with the f_{MSY} strategy. In addition, steady state economic analysis of a fishery [discount rate $\delta = 0$ - see Clark (1976)] indicates that optimal effort ($E_{\delta=0}$) is smaller than E_{MSY} and corresponds to a biomass in excess of B_{MSY} . Given values for economic parameters (selling prices and harvesting costs), the value of n for which the $f_{0,n}$ strategy

corresponds to an effort level of $E_{\delta=0}$ could be calculated, but usually such data are not available. Accordingly, an $f_{0.n}$ strategy is adopted in the hope that this will fix effort at a level closer to $E_{\delta=0}$ than, for example, E_{MSY} .

Gulland (1968) outlines the concept of the marginal yield of a fishery. In Figure 9.1, the curve OACM is a general relation between equilibrium catch and effort. If the effort is increased from its current level at E_1 to a new level, E_2 , the equilibrium catch increases by an amount BC. The "efficiency" of this increase (the discrete "marginal yield") is BC/AB . The "marginal yield" at E_1 is defined as the limit of the discrete marginal yield between E_1 and $(E_1 + \Delta E)$ as ΔE tends to zero. This limit is dC/dE , evaluated at E_1 . The $f_{0.n}$ policy involves maintaining fishing effort at the level at which the marginal yield has dropped to a fraction $0.n$ of its level at the onset of fishing (i.e. at $E=0$). This definition can be written as follows:

$$\left. \frac{dC}{dE} \right|_{E = E_{0.n}} = 0.n \left. \frac{dC}{dE} \right|_{E = 0} \quad (9.4)$$

Appendix 9.A shows how, if CPUE is linearly related to biomass, the target biomass ($B_{0.n}$) corresponding to the $f_{0.n}$ strategy is found by solving the equation:

$$B_{0.n} \left. \frac{dg(B)}{dB} \right|_{B=B_{0.n}} = 0.nK \left[\left. \frac{dg(B)}{dB} \right|_{B=B_{0.n}} - \frac{g(B_{0.n})}{B_{0.n}} \right] \quad (9.5)$$

For the majority of the calculations performed in this study, B_R has been taken to be $B_{0.1}$. This choice was made even though the $f_{0.2}$

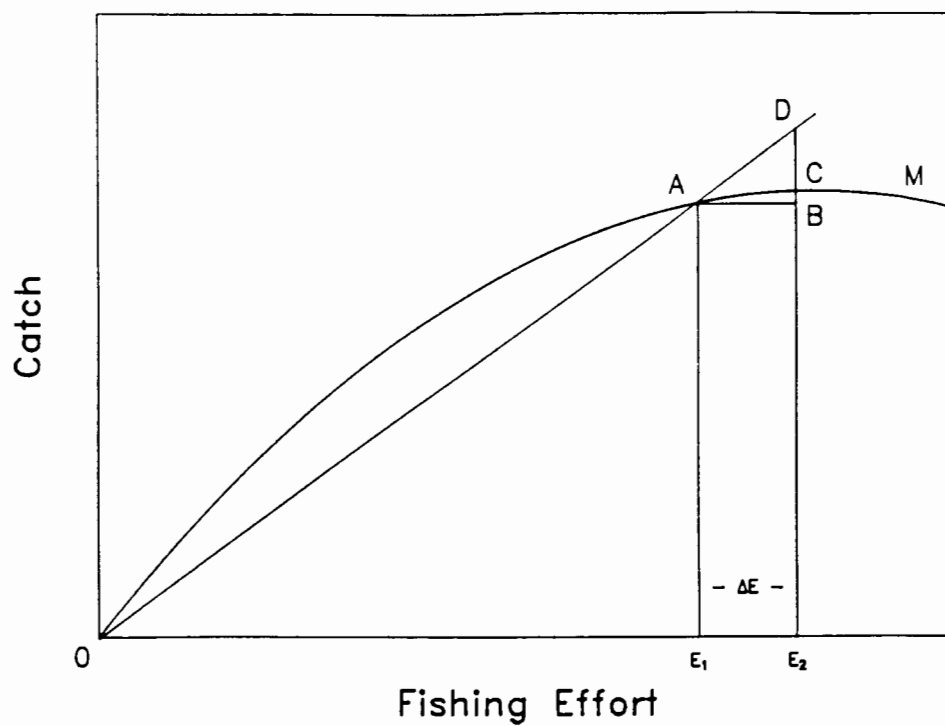


Figure 9.1 : A general equilibrium relationship between catch and effort. BC represents the increase in equilibrium catch as fishing effort is increased from E_1 to E_2 . The discrete marginal yield is the ratio BC/AB .

harvesting strategy is currently being used to provide TACs for the west coast hake resource (see Chapter 3). This is because $0.n=0.1$ is a more traditional choice, and because the $f_{0.2}$ harvesting strategy was first applied only after this study commenced (see Section 3.2). In principle, all calculations could have been performed for $0.n=0.1$ and $0.n=0.2$. However, this would have resulted in unreasonably long computation times. Instead, calculations have been performed with $0.n=0.1$ for the initial selections, and results for the choice $0.n=0.2$ taken into consideration only when drawing final conclusions regarding the "best" management procedure.

9.4 Harvesting strategies with age-structured model-estimation procedures

Most of the harvesting strategies described in Section 9.3 have analogues when the estimator is age-structured. The differences between harvesting strategies for age-structured model-estimation procedures and those for production model-estimation procedures is that the former are usually formulated in terms of fishing mortality rather than fishing effort, and that age-structure information is explicitly taken into consideration when setting TACs. If fishing effort is assumed to be linearly related to fishing mortality, then the first of these differences is not of consequence.

Only two harvesting strategies are considered here, viz. the $F_{0.1}$ (constant fishing mortality) and the $F_{\text{status-quo}}$ ("fixed" fishing mortality) harvesting strategies. These have been selected as they are the harvesting strategies which have been chosen most frequently when

applying age-structured model-estimation procedures to the hake resource [e.g. Butterworth *et al.* (1990), Punt and Butterworth (1989a)].

9.4.1 The $F_{0,n}$ harvesting strategy

The $F_{0,n}$ harvesting strategy sets the TAC for a particular year on the basis that the fishing mortality in that year will equal the target fishing mortality $F_{0,n}$. In the estimation of $F_{0,n}$ harvesting strategy TACs, the overall calculation includes the following four steps: specification of the future mean mass-at-age vector, specification of the age-specific selectivity pattern, estimation of $F_{0,n}$, and prediction of numbers-at-age (including recruitment) at the start of the year concerned. Each of these steps is described in turn.

9.4.1.1 Specification of the future mass-at-age vector

The masses-at-age used for TAC determination are computed according to the formula [as suggested by ICSEAF (1990)]:

$$w_{a+1/2} = [w_{a+1/2}(n) + w_{a+1/2}(n-1) + w_{a+1/2}(n-2)]/3 \quad (9.6)$$

where $w_{a+1/2}$ is the estimated mass of a fish aged $a+1/2$ years in years y ($y > n$), and

$w_{a+1/2}(k)$ is the observed mass of a fish aged $a+1/2$ years in year k ($k \leq n$).

9.4.1.2 Specification of the age-specific selectivity pattern

For those model-estimation procedures which do not explicitly estimate the age-specific selectivity pattern (e.g. *ad hoc* tuned VPA),

specification of the selectivities-at-age is achieved using the formula:

$$S_a = F_{a,n}/F_{m,n} \quad a = 1, \dots, m \quad (9.7)$$

where $F_{a,n}$ is the fishing mortality on fish aged a years in the final (n 'th) year.

9.4.1.3 Estimation of $F_{0,n}$

The yield which can be taken from a resource which is in equilibrium under a fully-selected fishing mortality of $F \text{ yr}^{-1}$, is given by the formula:

$$Y = \sum_{a=1}^m w_{a+1/2} S_a F N_a [1 - \exp\{-(M+S_a F)\}] / (M+S_a F) \quad (9.8)$$

$$\text{where } N_a = \begin{cases} N_{a-1} \exp\{-(M+S_{a-1}F)\} & \text{if } a > 1 \\ N_1 & \text{if } a = 1 \end{cases}$$

The value of N_1 (the equilibrium recruitment when fully-selected fishing mortality is equal to $F \text{ yr}^{-1}$) is set using one of three formulae, the selection of which depends on the form of the spawner biomass-recruitment relationship chosen:

$$\text{a) } \quad N_1 = \text{CONSTANT} \quad (9.9)$$

if there is assumed to be no dependence of recruitment on spawner biomass,

$$\text{b) } \quad N_1 = [\alpha \cdot SB(F) - \beta] / SB(F) \quad (9.10)$$

where $SB(F)$ is the effective equilibrium spawner biomass per recruit when fully-selected fishing mortality is $F \text{ yr}^{-1}$, i.e:

$$SB(F) = \sum_{a=1}^m f_a \exp\left[-\sum_{a'=1}^{a-1} (M + S_{a'} F)\right]$$

f_a is the fecundity index at age a (taken to be w_a if a is greater than the age-at-maturity)

if the Beverton-Holt spawner biomass-recruitment relationship is chosen, or

$$c) \quad N_1 = \ln\{\alpha SB(F)\} / \{\beta SB(F)\} \quad (9.11)$$

if the Ricker spawner biomass-recruitment relationship is chosen.

The value of $F_{0.n}$ is obtained numerically by solving the equation:

$$\left. \frac{dY}{dF} \right|_{F=F_{0.n}} = 0.n \left. \frac{dY}{dF} \right|_{F=0} \quad (9.12)$$

Traditionally, $F_{0.n}$ is defined in terms of yield-per-recruit (i.e Y/N_1) instead of yield (Beverton and Holt 1957). The extension to yield is necessary if the effects of the spawner biomass-recruitment relationship are to be taken into account in the estimation of $F_{0.n}$. The consequence of incorporating such relationships is usually to reduce the value of $F_{0.n}$. This is because increases in fishing mortality are associated with (eventual) reductions in equilibrium recruitment. Equilibrium recruitment declines monotonically with decreasing spawner biomass for the Beverton-Holt spawner biomass-recruitment relationship, but for the Ricker form there may be an initial increase before this eventual decline sets in.

9.4.1.4 Prediction of future numbers-at-age

An age-structured model-estimation procedure such as VPA will provide estimates of historic numbers-at-age. These include estimates of numbers-at-age at the start of the last year (n) for which catch-at-age data are available. If these estimates of numbers-at-age cover a sufficiently long time period, it may be possible to fit a spawner biomass versus recruitment curve to the historic estimates and to use this relationship to obtain predictions of future recruitment. However, spawner biomass versus recruitment relationships are usually very imprecisely determined, and so that estimating future recruitment to be the geometric mean of past recruitments may be a more reliable procedure:

$$\bar{R} = \left[\prod_{y=1}^{n-1} N_{y,1} \right]^{1/(n-1)} \quad (9.13)$$

The estimate of recruitment for the final year is omitted from the product, because it is usually very imprecisely determined by VPA.

Using an estimate of $N_{n+1,1}$ (either \bar{R} or a value predicted by a spawner biomass-recruitment relationship), and the formula:

$$N_{n+1,a+1} = N_{n,a} e^{-(M+F_{n,a})} \quad (9.14)$$

the numbers-at-age present at the start of year $n+1$ can be estimated. The $F_{0,n}$ strategy TAC for year k ($k > n$) is then provided by the equation:

$$TAC_{0.n}(k) = \sum_{a=1}^m w_{a+1/2} S_a F_{0.n} N_{k,a} \frac{1 - \exp\{- (M+S_a F_{0.n})\}}{M+S_a F_{0.n}} \quad (9.15)$$

where $w_{a+1/2}$ is the future mid-year mass-at-age vector,

$$N_{k,1} \quad (k \geq n+1) = \frac{\bar{R}}{\alpha SB_k / (\beta + SB_k)} \quad \begin{array}{l} \text{Recruitment independent} \\ \text{of spawner biomass} \end{array} \quad \text{Beverton-Holt form} \quad (9.16)$$

$$\alpha SB_k e^{-\beta SB_k} \quad \text{Ricker form}$$

SB_k is the spawner biomass in year k , and

$N_{k,a} \quad (k > n+1, a > 1)$ is obtained from the equation:

$$N_{k,a} = N_{k-1,a-1} e^{-(M+S_{a-1}F_{0.n})}$$

For those procedures which estimate $N_{k,1} \quad (k > n)$ using a spawner biomass-recruitment relationship, the parameters α and β need to be estimated. For some of the integrated analysis approaches, α and β are estimated directly in the non-linear minimization process. In this study, for those model-estimation procedures which do not estimate α and β directly, the spawner biomass-recruitment relationship has been transformed into a form linear in the parameters α and β . Linear regression has then been used to estimate α and β . This approach was chosen because it is computationally efficient. Other (less biased and more precise) approaches such as that of Walters and Ludwig (1981) could have been used, but this would have increased the computation time substantially.

Due to computer time restrictions, only the case $0.n = 0.1$ has been considered in this study.

9.4.2 The $F_{\text{status-quo}}$ harvesting strategy

In many applications of age-structured model-estimation procedures to the hake resources [e.g. Butterworth *et al.* (1989); Punt and Butterworth (1989a)], $F_{0,1}$ harvesting strategy TACs are found to be very imprecisely determined. Use of these TACs to manage a resource could therefore lead to undesirable consequences such as unintended depletion of the resource. The $F_{\text{status-quo}}$ harvesting strategy does not attempt to alter the fishing mortality in the future to some target value (e.g. $F_{0,n}$), but rather aims to keep it at its current level (the option considered further in this study) or some proportion of this level (Pope 1984). Such a harvesting strategy is not optimal (in a deterministic context) in terms of, say, maximizing total catch. However, it may be a more robust approach to management in the face of uncertainty, as the associated TACs are usually more precisely determined than the corresponding $F_{0,n}$ harvesting strategy TACs [Pope (1983), Pope and Gray (1983)].

The $F_{\text{status-quo}}$ harvesting strategy TAC for year k ($k > n$) is given by:

$$\text{TAC}_{\text{sq}}(k) = \sum_{a=1}^m w_{a+1/2} F_{n,a} N_{k,a} \frac{1 - \exp\{-(M+F_{n,a})\}}{M+F_{n,a}} \quad (9.17)$$

where $N_{k,a}$ ($k > n+1$) is obtained from the equation:

$$N_{k,a} = N_{k-1,a-1} e^{-(M+F_{n,a-1})}$$

$N_{k,1}$ ($k > n+1$) is obtained from Equation (9.16).

9.4.3 An alternative TAC estimation method

The variances of the recruitment estimates provided by VPA are usually very large for the last few years of the time series [see, for example, results in Punt and Butterworth (1989a) and Punt (1990a)], and estimation methods (such as those described above) which use these estimates are likely to provide relatively imprecise TAC estimates. If it can be assumed that there is no dependence of recruitment on spawner-biomass, an "alternative" method of estimating TACs is to replace the last k (k has been chosen to be 2 here) recruitment values (i.e. corresponding to the years $n-k+1$ to n) by an inverse variance weighted average of the VPA-estimated recruitment for that year and the estimated mean recruitment (J.G. Shepherd, pers. commn). The general technique is referred to as "shrinkage" towards the mean.

In order to remove the effects of trends in recruitment (perhaps related to trends in spawner biomass) affecting the ability of the mean recruitment to forecast future recruitment (see Equation 9.13), only the recruitments for years $n-k-7$ to $n-k$ are used in the calculation of the mean recruitment. The "alternative" estimate for year $n-i$ ($i \geq k-1$) is then:

$$N_{n-i,1}^a = \exp\{[\ln(N_{n-i,1})W_i^N + \ln(\bar{R})W_R] / (W_i^N + W_R)\} \quad (9.18)$$

where $W_i^N = [\text{var}\{\ln(N_{n-i,1})\}]^{-1}$

$$W_R = (\sigma_R^2)^{-1},$$

$$\sigma_R^2 = \frac{1}{7} \sum_{y=n-k-7}^{n-k} [\ln N_{y,1} - \ln \bar{R}]^2, \text{ and}$$

$$\bar{R} = \prod_{y=n-k-7}^{n-k} (N_{y,1})^{1/8}$$

Note that this estimator differs from that used for actual assessments, because a delta method (Seber 1982) is used to calculate $\text{var}\{\ln(N_{n-i,1})\}$. In the actual assessments, a bootstrap technique has been used [see Punt and Butterworth (1989a)]. The choice to use the delta method here was made to reduce computation time.

Because $\text{var}\{\ln(N_{n-i,1})\}$ values are usually large, this estimator results in estimates of recruitment for the last few years of the series which are very similar to the geometric mean recruitment. The complete numbers-at-age matrix is computed by forward-projecting the $N_{n-i,1}^a$ from Equation (9.18) using the historic catch-at-age data prior to calculating the TACs.

9.5 Short-cut methods

One approach to estimating $F_{\text{status-quo}}$ harvesting strategy TACs is based on the observation that the matrix of $\ln\{C_{y+1,a+1}/C_{y,a}\}$ can be rather well approximated by a linear sum of year and age effects, and hence analyzed as an ANOVA (Pope and Shepherd 1982). Pope (1984) shows that is not even necessary to use an age-structured model-estimation procedure in order to apply the $F_{\text{status-quo}}$ harvesting strategy. He demonstrates that "short-cut" methods based on catch-by-mass and recruitment indices may provide reasonably accurate estimates of $F_{\text{status-quo}}$ strategy TACs. Two of these "short-cut" methods are investigated here. The two methods differ in the way in which mass and mortality change with age (Pope op. cit.).

9.5.1 Shepherd's Hangover TAC (SHOT) method (Shepherd 1984)

There are a number of approaches to the calculation of "hangover" TACs [Shepherd (1984), ICES (1985)]. The one which has been chosen for investigation in this study is intermediate in its data requirements and sets the TAC for year $n+1$ using the formula:

$$TAC_{n+1} = hC_n + sR_{n+1} \quad (9.19)$$

where C_y is the catch-by-mass in year y ,

R_y is an index of the recruitment in year y , and

h and s are obtained by fitting the model:

$$C_y = hC_{y-1} + sR_y + \epsilon_y \quad \epsilon_y \sim N(0; \sigma^2) \quad (9.20)$$

Equation (9.19) can be derived from the Butterworth-Andrew model:

$$B_{y+1} = B_y + g(B_y) - C_y \quad (9.21)$$

Let F_y be the catch/biomass ratio in year y and assume that $F_y = F_{y-1} = \dots = F$. Multiplying Equation (9.21) by F gives:

$$C_{y+1} = FB_{y+1} = FB_y - FC_y + Fg(B_y) = C_y(1-F) + Fg(B_y) \quad (9.22)$$

Assuming that most of the surplus production is made up of the incoming recruits (i.e. assuming that the effects of natural mortality and tissue growth approximately cancel) implies that:

$$C_{y+1} = C_y(1-F) + FR_{y+1} \quad (9.23)$$

If R_{y+1} is a relative index of the recruitment in year $y+1$, then:

$$C_{y+1} = C_y(1-F) + F R_{y+1} \quad (9.24)$$

where \tilde{F} is the product of \tilde{F} and the constant of proportionality between R_y and \tilde{R}_y .

Equations (9.23) and (9.20) are then equivalent, with $L = 1 - \tilde{F}$ and $s = \tilde{F}$.

9.5.2 Deriso-Roff-Pope (DROP) method (Pope 1984)

The TAC for year $n+1$ is estimated using the formula:

$$TAC_{n+1} = hC_n + gC_{n-1} + sR_{n+1} \quad (9.25)$$

where C_y is the catch-by-mass in year y ,

R_y is an index of the recruitment in year y , and

h , g and s are obtained by fitting the model:

$$C_y = hC_{y-1} + gC_{y-2} + sR_y + \epsilon_y \quad (9.26)$$

Equation (9.25) can be derived similarly to Equation (9.19), except that the underlying dynamics equation is the Deriso-Schnute model [Equation (8.8)] with w_{a_r-1} set equal to zero (ICES 1985).

9.6 Probing strategies

Harvesting strategies can be divided into three classes. The harvesting strategies in the first class do not probe the resource at all. Examples of such harvesting strategies are the RY and $F_{\text{status-quo}}$ strategies. The remaining harvesting strategies investigated in this study fall into the second class. These strategies may probe the resource incidentally as they attempt to drive it to some target level,

but probing is not their main objective. Harvesting strategies in the first two classes are sometimes referred to as passive adaptation strategies, because they wait for nature to point out model error.

In contrast, strategies in the third class deliberately attempt to probe the resource. The rationale for use of such strategies is that occasional probing experiments should be considered as a means of detecting model error and possibly improving parameter estimation precision, thereby perhaps ultimately improving resource utilization. Such experiments involve making relatively large changes in fishing effort, one aim of which is to provide better estimates of the model parameters by increasing the contrast in the data. Experiments which involve only small disturbances are not likely to provide sufficient information to offset "wasted" yield. Nevertheless, experiments must be designed so that the probability of heavily depleting the stock is low.

Walters (1986) suggests several scenarios in which probing rather than passive management strategies are appropriate. These include situations in which:

- (i) the historical data (which may cover a wide range of stock sizes, and are often critical to parameter estimation) are becoming less reliable because, perhaps, of effects such as changes in the "true" parameter values with time,
- (ii) large measurement errors are known to exist in the historical data,
- (iii) there is insufficient contrast in the data to provide reasonably precise parameter estimates, which may lead to under- or overexploitation,

- (iv) the stock has high productivity, thus making the consequences of any reduction caused by probing less serious,
- (v) an accurate monitoring system is available for recording responses to probing, and
- (vi) the maintenance of good (stable) catches at the present time is not as important as avoiding low catches in the future (i.e. the discount rate is low).

Probing experiments may be used in conjunction with the strategies discussed above in order to speed improvements to estimates of model parameters, and thus to the estimates of the target biomass (or target fishing mortality). The information gained from well designed experiments should, in the long term, more than compensate for any loss of current yield. Note that, as the number of such experiments increases, the marginal rise in present value (the economically discounted total catch over time) will decrease until eventually, at some point which is dependent on the discount rate, the present value of the resource will fall. This is because the costs associated with such experiments (loss of catch, monitoring costs, etc.) will tend to reduce the present value of the resource. If the experiment is to be useful in maximizing present value, the resultant (discounted) increase in revenue must exceed these costs. In a fishery with a high discount rate (i.e. in which current catches make up a large proportion of the present value), any probing which involves a drop in fishing effort may reduce the present value of the resource substantially.

Although occasional probing experiments may increase the present value of a resource, the effects of such experiments may be particularly

disturbing to the fishing industry. It is seldom easy to motivate management to make large changes in fishing effort, even when a resource is assessed to be biologically overexploited. For obvious reasons it is difficult to motivate large changes in fishing effort using arguments involving possible higher future catches, particularly if these involve a short-term reduction in effort and hence in catch. On the other hand, occasional increases in fishing effort may be undesirable because, if the extra effort is to be applied by the local industry, this may encourage overcapacity, while if it is to be applied by foreign vessels, this is likely to meet with resistance from the local industry. Another problem which may arise with probing experiments is that, if average catches do not increase relatively soon after a probing experiment, the fishing industry may lose faith in the management agency.

While harvesting strategies which deliberately probe the resource may result in improved trade-offs in performance, no harvesting strategies of this nature are considered in this study.

APPENDIX 9.A. EQUILIBRIUM BIOMASS, CATCH AND EFFORT LEVELS
CORRESPONDING TO AN $f_{0.n}$ STRATEGY

The fishing effort corresponding to the $f_{0.n}$ strategy is that effort which satisfies the equation:

$$\left. \frac{dC}{dE} \right|_{E = E_{0.n}} = 0.n \left. \frac{dC}{dE} \right|_{E = 0} \quad (9.A.1)$$

where $C(E)$ is the equilibrium (sustainable) catch for effort E .

The f_{MSY} strategy corresponds to $0.n = 0$, and the $f_{0.1}$ strategy to $0.n = 0.1$. Assuming equilibrium conditions and that CPUE is linearly related to biomass (see Equation 5.2), the rate of change of equilibrium catch with biomass (dC/dB) may be related to the rate of change of equilibrium catch with effort:

$$\frac{dC}{dE} = \frac{dC}{dB} \frac{dB}{dE} \quad (9.A.2)$$

Substituting $(C/E)/q$ for B in Equation (9 A.2) gives:

$$\frac{dC}{dE} = \frac{1}{q} \frac{dC}{dB} \frac{d(C/E)}{dE} \quad (9.A.3)$$

Now

$$\frac{d(C/E)}{dE} = \left[\frac{dC}{dE} E - C \right] \frac{1}{E^2}$$

so that substitution of qB for C/E (see Equation 5.2) gives:

$$\frac{d(C/E)}{dE} = \left[\frac{dC}{dE} - qB \right] \frac{1}{E} \quad (9.A.4)$$

Substituting this result into Equation (9.A.3) and multiplying by q_E then gives:

$$q_E \frac{dC}{dE} = \frac{dC}{dB} \left[\frac{dC}{dE} - q_B \right] \quad (9.A.5)$$

Solving Equation (9.A.5) for dC/dE :

$$\frac{dC}{dE} = \frac{q_B \frac{dC}{dB}}{\frac{dC}{dB} - q_E} \quad (9.A.6)$$

and evaluating dC/dE at $E = 0$ (note that when $E = 0$, $B = K$) gives:

$$\begin{aligned} \left. \frac{dC}{dE} \right|_{E=0} &= \frac{q_K \frac{dC}{dB}}{\frac{dC}{dB} - q_0} \\ &= q_K \end{aligned} \quad (9.A.7)$$

Now, substituting these last two results into Equation (9.A.1) gives:

$$\frac{q_B \frac{dC}{dB}}{\frac{dC}{dB} - q_E} \Big|_{E = E_{0.n}} = 0.n q_K \quad (9.A.8)$$

Rearranging terms and noting that $B(E_{0.n}) = B_{0.n}$ gives:

$$B_{0.n} \frac{dC}{dB} \Big|_{B=B_{0.n}} = 0.nK \left[\frac{dC}{dB} \Big|_{B=B_{0.n}} - \frac{C}{B_{0.n}} \right] \quad (9.A.9)$$

Finally, replacing C by $g(B)$ gives:

$$B_{0.n} \frac{dg(B)}{dB} \Big|_{B=B_{0.n}} = 0.nK \left[\frac{dg(B)}{dB} \Big|_{B=B_{0.n}} - \frac{g(B_{0.n})}{B_{0.n}} \right] \quad (9.A.10)$$

which, given $g(B)$ and $0.n$, can be solved to provide the value of $B_{0.n}$.

CHAPTER 10 - ASSESSMENTS OF THE CAPE HAKE STOCK OFF THE SOUTH AFRICAN WEST COAST

Three assessment procedures are applied to data for the hake resource off the South African west coast. Two of these procedures (the Butterworth-Andrew ($B_1=K$; Schaefer form) observation error estimator and the 'M-independent-of-age' *ad hoc* tuned VPA) are routinely applied to provide management advice for this resource. The third procedure involves incorporating the effects of cannibalism into the VPA approach. The predation model included in this latter procedure is based on the assumption that each predator exerts a mortality rate proportional to its mass on its preferred size of prey. It incorporates factors which take account of the spatial distribution of prey and predators, and of the observed proportion of hake in the diet. The C.V.s of the management quantities provided by these assessment procedures are obtained by means of the (conditioned) parametric bootstrap variance estimation procedure.

The estimates of current stock size (both in absolute terms and expressed as a percentage of the average pristine level) and MSY provided by the standard ($M=0.3\text{yr}^{-1}$) 'M-independent-of-age' VPA are substantially different from the estimates of these quantities obtained from the production-model. This VPA assessment estimates that the resource is less productive, smaller and at a lower fraction of its average pristine level than does the production-model. The estimates of the quantities provided by the VPA are highly sensitive to the value assumed for M . By increasing M to 0.5yr^{-1} , it is possible to reconcile the production-model and VPA estimates of current depletion. However, irrespective of the value of M , there are still substantial differences between the estimates of current biomass in absolute terms. These differences can be removed by tuning the terminal fishing mortalities using survey data. However, the estimates provided by the latter procedure are very imprecisely determined.

The estimates of the mass of hake consumed by hake which are predicted by the 'M-cannibalistic' VPA range from virtually nothing to over 20 million tons per annum. The log-likelihood surface is insensitive to the value of the parameter which determines the extent of cannibalism, so that the data to which the model is fitted are unable to determine a value for this parameter. It is also not possible to select a value for this parameter by comparing the predictions of the 'M-cannibalistic' VPA with those obtained from analyses of the stomach content data (Section 2.5) because the latter are rather sensitive to assumptions and are often very imprecisely determined. However, the results of the 'M-cannibalistic' VPA are qualitatively similar to those of the 'M-independent-of-age' VPA in a number of respects.

10.1 Introduction

In order to select values for the parameters of the operating models to be utilized (see Chapter 11), it is necessary to have some idea of the historical trends and current status of the resource being modelled. It is also important to have some estimates of the precision of these quantities.

The two approaches which have been used recently to assess the hake resource off the South African west coast (Punt 1991) are *ad hoc* tuned VPA (based on the Laurec-Shepherd tuning algorithm) and a production model-estimation procedure [the Butterworth-Andrew ($B_{1917}=K$; Schaefer form) observation error estimator]. This Chapter describes the details of these two approaches. An extension of the *ad hoc* tuned VPA which makes allowance for cannibalism is then developed. These three approaches are applied to data for the resource under consideration.

However, before the approaches are described, the procedure used to estimate the precision of the quantities estimated will be detailed. This is the (conditioned) parametric bootstrap (Efron 1982, 1985). This approach was shown to perform the best of a number of variance-estimation procedures in estimating the standard errors and coefficients of variation of management quantities provided by a number of surplus production model-estimation procedures (Punt 1988). It has been implemented for VPA by Butterworth *et al.* (1990).

Note that there is substantial overlap between the *ad hoc* tuned VPA and production model approaches developed in Sections 10.3 and 10.5 respectively, and those described in Chapters 6 and 7. Some material is

repeated here to ease the presentation of the implementations of the (conditioned) parametric bootstrap method of variance-estimation, as this requires more details related to the error structures assumed by the model-estimation procedures to be specified than were given earlier.

10.2 Estimation of the precision of estimated quantities

The estimates of standard errors (S.E.s) and coefficients of variation (C.V.s) are obtained using the (conditioned) parametric bootstrap procedure (Efron 1982, 1985), and estimates of 95% confidence intervals by using the percentile method [Efron (1981), Buckland (1984)].

10.2.1 Estimation of standard deviations and coefficients of variation

The (conditioned) parametric bootstrap variance-estimation procedure will be illustrated by means of the example of estimating the standard error of the quantity $\{a + bx_1\}$, where a and b are obtained from a linear regression of $\{y_i: i=1, \dots, k\}$, against $\{x_i: i=1, \dots, k\}$. Note that, under the assumption that errors are homoscedastic, independent and normally distributed, it is of course not necessary to estimate the standard error of this quantity using the bootstrap, because its variance can be estimated analytically using normal distribution theory.

More generally, let \hat{Q} be the quantity for which an estimate of standard error is required. Now assume that \hat{Q} is calculated from the set of estimated model parameters $\hat{\beta}$, using the function ϕ , i.e.:

$$\hat{Q} = \phi(\hat{p}) \quad (10.1)$$

where for the linear regression example, $\hat{p} = \{\hat{\alpha}, \hat{\beta}\}$, and $\phi(\alpha, \beta)$ is $\{\alpha + \beta x_1\}$.

Now let θ be the estimator of \hat{p} , given a set of data \underline{x} , i.e.:

$$\hat{p} = \theta(\underline{x}) \quad (10.2)$$

In the example under consideration, \underline{x} is the set $\{x_i: i=1, \dots, k; y_i: i=1, \dots, k\}$ and the estimator θ is:

$$\theta = \begin{bmatrix} (\sum x_i y_i - \sum x_i \sum y_i / k) / (\sum x_i^2 - \sum x_i \sum x_i / k) \\ (\sum x_i^2 \sum y_i - \sum x_i y_i \sum x_i) / (k \sum x_i^2 - \sum x_i \sum x_i) \end{bmatrix}^T \quad (10.3)$$

Finally, let $\hat{D}(\underline{x})$ be an estimate of the distribution of the observed vector of data points \underline{x} . The various forms of the bootstrap (Efron 1981, 1982, 1985, 1987) differ as to how $\hat{D}(\underline{x})$ is constructed. The (conditioned) parametric bootstrap is "conditioned" in the sense that the values of independent variables (i.e. $\{x_i: i=1, \dots, k\}$ in the example considered) are assumed to be fixed. The procedure is "parametric" because, in order to construct $\hat{D}(\underline{x})$, it is assumed that the distributions of the dependent variables are those assumed when fitting the model. For the regression example considered here, these are:

$$x_i \sim N(x_i; 0)$$

$$y_i \sim N(\alpha + \beta x_i; \sigma^2)$$

The (conditioned) parametric bootstrap variance-estimation procedure estimates the standard error of \hat{Q} as follows: a large number (U_{\max} , where $U=1, \dots, U_{\max}$) of random bootstrap samples $\{X^U : U=1, \dots, U_{\max}\}$ are generated and the set $\{\hat{Q}^1, \hat{Q}^2, \dots, \hat{Q}^{U_{\max}}\}$ where $\hat{Q}^U = \phi|\theta(X^U)|$ is computed. For the regression example, the random bootstrap samples are generated as follows:

$$\begin{aligned} x_i^U &\sim \hat{D}(x_i) = x_i \\ y_i^U &\sim \hat{D}(y_i) = N(\hat{\alpha} + \hat{\beta}x_i; \hat{\sigma}^2) \end{aligned}$$

The variance of \hat{Q} is then estimated by:

$$\widehat{\text{VAR}}(\hat{Q}) = \frac{1}{U_{\max} - 1} \sum_{U=1}^{U_{\max}} [\hat{Q}^U - Q_{(\cdot)}]^2 \quad (10.4)$$

where $Q_{(\cdot)}$ is the mean of the \hat{Q}^U 's.

In order to implement the (conditioned) parametric bootstrap, it is necessary to specify the functions ϕ , θ and the distribution $\hat{D}(X)$. As ϕ is usually trivially defined (e.g. $MSY=rK/4$ for the Schaefer surplus production function) and θ is the estimator itself, the descriptions of the implementations of the (conditioned) parametric bootstrap for each approach will only detail the distribution $\hat{D}(X)$, i.e. how the random bootstrap samples are generated.

10.2.2 Confidence intervals

In order to estimate confidence intervals, the percentile method [Efron (1981), Buckland (1984)] has been used. The estimate of the confidence

interval is computed from the set of bootstrap estimates after sorting them into ascending order. If N Monte-Carlo trials are performed, the 95% confidence interval for \hat{Q} is then estimated by $[\hat{Q}_{0.025N}^U, \hat{Q}_{0.975N}^U]$, with $\hat{Q}_{0.025N}^U$ and $\hat{Q}_{0.975N}^U$ obtained by linear interpolation within the ordered set. Efron (1987) states that to use this method properly requires at least 1000 bootstrap simulations.

10.3 The "M-independent-of-age" *ad hoc* tuned VPA

The error structure corresponding to the tuning algorithms utilized is as follows:

$$F_{y,a} = q S_a E_y \exp(\epsilon_{y,a}) \quad \epsilon_{y,a} \text{ from } N(0; \sigma_a^2) \quad (10.5)$$

where $F_{y,a}$ is the fishing mortality on fish aged a years in year y ,

E_y is the effort in year y ,

S_a is the age-specific selectivity on age a , and

q is catchability ($q_a = qS_a$).

Thus, the actual fishing mortality is assumed to be log-normally distributed about its expected value and the residuals are assumed to be uncorrelated with respect to age and year. This particular error structure has been selected because it ensures that $F_{y,a}$ is positive for all values of $\epsilon_{y,a}$. This simplifies the generation of the bootstrap data sets considerably (see Section 10.3.3). The *ad hoc* tuned VPA procedure is detailed in Section 7.4. Here it is necessary only to describe the tuning algorithms used, which correspond to maximum likelihood estimation for the error structure assumed.

10.3.1 Tuning the oldest-age terminal fishing mortalities

The algorithm used to tune the oldest-age terminal fishing mortalities is based on the assumption that the age-specific selectivity function is flat (i.e. unchanging with age) for the oldest (p+1) ages:

$$\hat{F}_{Y,m} = \left[\prod_{a=m-p}^{m-1} F_{Y,a} \right]^{1/p} \quad Y = Y_1, \dots, n \quad (10.6)$$

where $\hat{F}_{Y,m}$ is the estimated oldest-age terminal fishing mortality in year y,

m is the oldest age considered,

n is the most-recent year, and

y_1 is the first year for which catch-at-age data are available.

10.3.2 Tuning the most-recent-year terminal fishing mortalities

The maximum likelihood estimator for the error structure assumed corresponds to the Laurec-Shepherd tuning algorithm (Pope and Shepherd 1985).

$$\hat{F}_{n,a} = \bar{q}_a E_n \quad a = 1, \dots, m-1 \quad (10.7)$$

$$\text{where } \bar{q}_a = \bar{\bar{\bar{q}}}_a = \left[\prod_{Y=Y_T}^{n-1} (F_{Y,a}/E_Y) \right]^{1/(n - Y_T)}$$

$\hat{F}_{n,a}$ is the estimated most-recent-year terminal fishing mortality, and

y_T is the first year for which effort data are available.

10.3.3 Variance estimation

Butterworth *et al.* (1990) suggest a method of applying the (conditioned) parametric bootstrap variance-estimation approach which involves generating a large number (U_{\max}) of alternative fishing mortality matrices. These fishing mortality matrices are conditioned on the assumption that the catch-at-age matrix, the effort vector and the natural mortality estimate are all exact. The alternative fishing mortality matrices are created by generating vectors of alternative most-recent-year and oldest-age terminal fishing mortalities, and then applying the standard VPA back-calculation process (once). Bootstrap estimates of the other management quantities are computed from each alternative fishing mortality matrix. The alternative most-recent-year terminal fishing mortality vector is generated by assuming that the residuals of the relation between $F_{n,a}$ and E_n are uncorrelated with respect to age, and are log-normally distributed (see Equation 10.5). The vector for the oldest age is handled similarly. These alternative sets of terminal fishing mortalities are thus generated as follows.

a) Most-recent-year terminal fishing mortalities

$$F_{n,a}^U = F_{n,a} \exp(\epsilon_{n,a}^U) \quad a = 1, \dots, m-1 \quad (10.8)$$

where $\epsilon_{n,a}^U$ is chosen at random from $N(0; (\hat{\sigma}_a^U)^2)$,

$$\begin{aligned} (\hat{\sigma}_a^U)^2 &= \frac{n - Y_T + 1}{n - Y_T} \hat{\sigma}_a^2, \\ \hat{\sigma}_a^2 &= \frac{1}{n - Y_T - 1} \sum_{Y=Y_T}^{n-1} |\ln(F_{Y,a}/E_Y) - \ln q_a|^2, \text{ and} \end{aligned} \quad (10.9)$$

$$\bar{q}_a = F_{n,a} / E_n.$$

b) Oldest-age terminal fishing mortalities

$$F_{y,m}^U = F_{y,m} \exp(\epsilon_{y,m}^U) \quad y = y_1, \dots, n \quad (10.10)$$

where $\epsilon_{y,m}^U$ is chosen at random from $N(0; (\hat{\sigma}^U)^2)$,

$$(\hat{\sigma}^U)^2 = \frac{1}{p} \hat{\sigma}^2 \quad \text{and}$$

$$\hat{\sigma}^2 = \frac{1}{(n-y_1)(p-1)} \sum_{y=y_1}^{n-1} \sum_{a=m-p}^{m-1} [\ln F_{y,a} - \ln F_{y,m}]^2 \quad (10.11)$$

Both of these approaches assume that the variance of the estimates of the terminal fishing mortalities is made up of two components: (a) the variance of the estimate of an average and (b) the variance of individual realizations about that average value.

10.3.4 Implementation of methodology

Tuning of the most-recent-year terminal fishing mortalities was achieved using effort data for the period 1978 to 1989 (see Table 4.1). In order to avoid zeros in the catch-at-age matrix (see Table 4.5), only ages in the range 1 to 7 were considered.

Tuning of the oldest-age terminal fishing mortalities was carried out with parameter $p=2$. Bootstrap variance calculations used $U_{\max}=1000$ resamples. In order to determine the sensitivity of the results to the value assumed for the age-independent instantaneous rate of natural mortality (M), calculations were carried out for $M=0.2; 0.3; 0.4$ and

0.5yr^{-1} . This range was selected as it encompasses the range of published estimates for this parameter [see, for example, Andrew (1986) Table A9.1.1]. Additional sensitivity tests were performed by tuning the most-recent-year terminal fishing mortalities using effort data for the period 1983 to 1989, and by tuning these terminal fishing mortalities using a "pseudo effort" series. The latter was constructed by dividing the annual catch (C_Y) by the survey biomass index obtained in the middle of the year (B_Y^S). An application using the survey biomass indices obtained at the start of each year was not possible, because the requisite biomass index for 1989 does not exist.

Two biomass series were computed for each application.

a) The 2^+ biomass at the start of year y (B_Y^{2+})

$$B_Y^{2+} = \sum_{a=2}^m w_a N_{Y,a} \quad (10.12)$$

where w_a is the mass of a fish aged a years, and

$N_{Y,a}$ is the estimate of the number of fish aged a years at the start of year y .

b) Exploitable biomass during year y (B_Y^e)

$$B_Y^e = \sum_{a=1}^m w_{a+1/2} S_a N_{Y,a} e^{-(F_{Y,a}+M)/2} \quad (10.13)$$

where selectivity $S_a = F_{n,a} / F_{n,7}$.

The estimate of F_{MSY} (and hence of MSY) is obtained by selecting the value of F which maximizes the equation:

$$SY(F) = \bar{R} \sum_{a=1}^m w_{a+1/2} F S_a \bar{N}_a (1 - e^{-(F S_a + M)}) / (F S_a + M) \quad (10.14)$$

where $SY(F)$ is the sustainable yield when fishing mortality is fixed at F .

\bar{R} is the geometric mean of $\{N_{Y,1} : Y = y_1, \dots, n-2\}$, and

$$\bar{N}_a = \exp\left[-\sum_{a'=1}^{a-1} (F S_{a'} + M)\right].$$

The estimates of the average (exploitable) pristine level (K^e) were obtained using Equation (10.13), except that $F_{Y,a}$ was taken to be zero, the resource was assumed to be in equilibrium (i.e. $N_a = N_{a-1} e^{-M}$), and the number of 1-year-olds was taken to be \bar{R} .

10.4 The "M-cannibalistic" *ad hoc* tuned VPA

The *ad hoc* tuned VPA formalism detailed in the previous section can be criticized because of its assumption that M is a constant, independent of both age and year. Specifically, the effects of hake predation/cannibalism are not taken into consideration, even though Sections 2.5 and 2.6 suggest that these effects may be substantial.

Although quantitative knowledge of biological interactions between fish species is usually poor (see, for example, Shepherd 1988), it is generally considered that the main effect of interspecific predation is on the smallest/youngest animals. The results of MSVPA [Multispecies VPA] (ICES 1986, 1989) indicate that the mortalities on the youngest/smallest fish may be substantially larger than those customarily assumed when single-species model-estimation procedures (such as the *ad*

hoc tuned VPA detailed in Section 10.2) are applied. This can, in turn, lead to different long-term predictions for the consequences of changing fishing intensity. This is because the resultant changes in predator abundance lead to changes in the natural mortality rates for prey species.

It is possible, in principle, to develop models of whole ecosystems. Such models would take account of all species in the ecosystem (from the smallest micro-organisms to the top predators), as well as spatial structure and hydrography. The models of Anderson and Ursin (1977), Laevastu *et al.* (1982) and Tjelmeland (1988) are of this type. Although these approaches are praiseworthy for their objectives, they are perhaps overly ambitious. This is because there are serious difficulties in selecting appropriate functional forms and estimating the many model parameters. As a result, in order to model the multispecies fishery in the North Sea, ICES (ICES 1986, 1989) have only considered the major commercial species in MSVPA calculations. The data base required to estimate the fishing and predation mortalities using MSVPA is substantial, and consists of information on food consumption as well as feeding rates. The amount of this type of information for the species found off the South African west coast, although not insubstantial, is almost certainly inadequate for application of MSVPA at this time. This is because there is almost no information on evacuation rates and the stomach content samples sizes are relatively small. Furthermore, the dynamics of some of the major commercial species harvested off the South African west coast (e.g. horse mackerel) are only poorly understood. Stomach content data exist only

for recent years (1983 onwards), which means that MSVPA cannot be used to estimate the historic biomass series anyway. A simpler approach to modelling predation is therefore necessary, and this is to model only the effects of hake predation/ cannibalism.

Two extreme approaches to modelling the natural mortality inflicted by a single predator have been suggested (e.g. Shepherd (1988); ICES (1986, 1989)). It is assumed either that each predator takes the same daily ration from the available prey, or that each predator exerts the same predation mortality per unit biomass on its desired prey species. Both approaches can be criticized. For example, the former approach requires that account be taken of all prey species which make up substantial proportions of the diet of the predator species - this is not possible in many cases (including the case under consideration) due to lack of sufficient data on the dynamics of the prey species. The second approach runs into difficulties if huge year-classes of prey occur. A predator then eats unrealistic amounts of prey, because this model makes no allowance for saturation. Although it is possible, in principle, to model predator saturation, the functional forms and parameter values required cannot often (if ever) be satisfactorily estimated from the available data.

The second approach described above has been used here (i.e. each predator is assumed to exert a mortality rate proportional to its mass on its preferred size of prey). The consequent mortalities are modified by two additional factors. The first takes into account the spatial distribution of the prey and the predators, while the other makes allowance for the proportion of hake in the diet. There is also a

component of natural mortality which is independent of predator biomass. This "basal" rate of natural mortality is the combined effect of all causes of natural mortality except predation by hake, and is for simplicity (and also the lack of data on predation of hake by other species) taken to be independent of age, year and species.

Thus, the natural mortality rate on a fish of species s in length-class ℓ during year y is given by:

$$M_{Y,\ell}^s = M_b + \sum_{s'} \sum_{\ell' > \ell} G_{\ell'}^{s'} \Gamma^{s,s'} SP(\ell, \ell') w_{\ell'}^{s'} N_{Y,\ell'}^{s'} \quad (10.15)$$

where $M_{Y,\ell}^s$ is the natural mortality on fish of species s in length-class ℓ during year y ,

M_b is the basal rate of natural mortality,

$\Gamma^{s,s'}$ is a species distribution factor which affects the rate of mortality which a predator of species s' inflicts on a prey of species s ; this factor is related to the spatial distribution of prey and predators - for example, the predation of *M. paradoxus* on *M. capensis* is zero because adult *M. paradoxus* are geographically separate from the juvenile *M. capensis*,

$SP(\ell, \ell')$ is the predator size preference function (assumed to be independent of predator species),

$G_{\ell'}^{s'}$ is the proportion of hake in the diet of predators of species s' in length-class ℓ' - this factor has to be introduced because, for example, hake smaller than 20cm are extremely numerous, but inflict virtually no

mortality on smaller hake because they prefer other prey species (see Section 2.5),

$w_{\ell}^{s'}$ is the mass of a fish of species s' in length-class ℓ' ,
and

$N_{y,\ell}^{s'}$ is the number of fish of species s' in length-class ℓ'
at the start of year y .

This model thus assumes that each predator has a preferential prey length and that the mortality due to predation increases if there are "more" predators. Specifically, it is assumed that predation mortality is proportional to predator biomass (rather than numbers). This is appropriate for hake, because larger hake have a larger daily ration (see Section 2.5). For the purposes of computational simplicity, the annual predation mortality rate is assumed to be a function of the biomass of predators at the start of the year rather than the average biomass during the year, which might be more accurate. While this assumption is not likely to result in substantial error as long as the predator biomass-at-age remains relatively constant during a year, it may have a more important effect on the results if the mortality rate on prey is high.

The size preference function $SP(\ell, \ell')$ is similar to that used by Anderson and Ursin (1977), and has the form:

$$SP(\ell, \ell') = \exp(-\{[\ln(\ell/\bar{\ell}') - \ln(R_f)]^2 / 2\sigma_w^2\}) \quad (10.16)$$

where R_f is the preferred prey/predator length ratio (assumed to be independent of predator species),

$\bar{\ell}$ is the mid-point of length-class ℓ , and

σ_w is the standard deviation of the distribution of the relative length of fish taken about the preferred length ratio.

This predation model can be criticized on two grounds. As discussed above, although the model allows for predator "starvation", it does not take saturation into account. Another questionable assumption is that M_b is independent of age and year (it would in reality be a function of both, as other predators will also have size-preference functions and fluctuate in abundance).

The function G_l^S has been modelled by a logistic curve:

$$G_l^S = [1 + \exp(-(\ell - \ell_{50}^P)/\delta^P)]^{-1} \quad (10.17)$$

where ℓ_{50}^P is the length at which 50% of the diet consists of hake, and

δ^P is a parameter which determines the width of the curve.

Although model (10.15) may be able to model hake predation/cannibalism reasonably realistically, it cannot be used within the framework of *ad hoc* tuned VPA. This is because VPA considers age rather than size. It is, however, possible to approximate Equation (10.15) by an equation involving quantities which could be estimated using VPA.

$$M_{y,a}^S = M_b^S + \sum_{s'} \sum_{a' > a} G_{s'}^{s'} \Gamma^{s,s'} SP(\ell_a^S, \ell_{a'}^{s'}) w_{a'}^{s'} N_{y,a'}^{s'} \quad (10.18)$$

where ℓ_a^S is the length of a fish of species s aged a years,
 w_a^S is the mass of a fish of species s aged a years,
 $N_{y,a}^S$ is the number of fish of species s aged a years at

the start of year y .

This approximation would be exact if each age-class corresponded to a unique length-class. As this is not the case for Cape hake, there will be some error when approximating Equation (10.15) by Equation (10.18).

Although the predation model (10.18) is now formulated in terms of numbers-at-age instead of numbers-at-length, it is still not possible to use it to assess the Cape hake resource off the South African west coast. This is because the catch and effort data do not distinguish between the two hake species, i.e. the data for the two species are pooled. This means that it is only possible to consider estimators which pool both species. Aggregating the two species in Equation (10.18) yields:

$$M_{y,a} = M_b + \sum_{a' > a} \frac{G}{l_{a'}} \Gamma SP(l_{a'}, l_a) w_{a'} N_{y,a'} \quad (10.19)$$

where $M_{y,a}$ is the natural mortality on fish aged a years during year y ,

Γ is natural mortality per unit predator biomass on the preferred size of fish,

G_{l_a} is the fraction of the diet of predators of age a which is composed of hake,

w_a is the begin-year mass of a predator aged a years, and

$N_{y,a}$ is the number of predators aged a years at the start of year y .

Once values for the parameters M_b , R_f , σ_w , Γ , l_{50}^P and δ^P are specified, it is possible to apply the *ad hoc* tuned VPA formalism as usual. The natural mortality on fish in the oldest age-class is M_b , because hake

in the this age-class are not preyed on by any other hake according to Equation (10.19). By setting $M=M_b$, it is thus possible to compute the number of fish in the oldest age-class. The number of fish in this age-class are then used to calculate the natural mortality on fish in the second oldest age-class (as the only fish which prey on these fish are those in the oldest age-class). This then allows the number of fish in the second oldest age-class to be computed. This back-calculation process is applied until the entire numbers-at-age matrix is computed. To reduce the number of parameters that need to be estimated by maximizing the likelihood function, values for the parameters L_{50}^P and δ^P were obtained by fitting the model of Equation (10.17) to data on the proportion by mass of hake in the diet. The resultant estimates are $L_{50}^P = 57.6\text{cm}$ (C.V. 2.7%) and $\delta^P = 19.6$ (C.V. 7.8%). The data and the fitted curve are illustrated in Figure 10.1. The fit of Equation (10.17) to these data is quite good, although a straight line would have fitted the data equally adequately.

The values of the four remaining parameters were obtained by maximizing the log-likelihood function:

$$\ln L = K'' \sum_a \sum_{a'} Z_{a,a'}' \ln \{ \hat{Z}_{a,a'} \} \quad (10.20)$$

where $Z_{a,a'}'$ is the observed number of hake in the stomachs of predators of age a' which are of age a years,

$\hat{Z}_{a,a'}$ is the model-predicted proportion of hake in the diet of predators of age a' which are of age a years (calculated from Equation 10.19), and

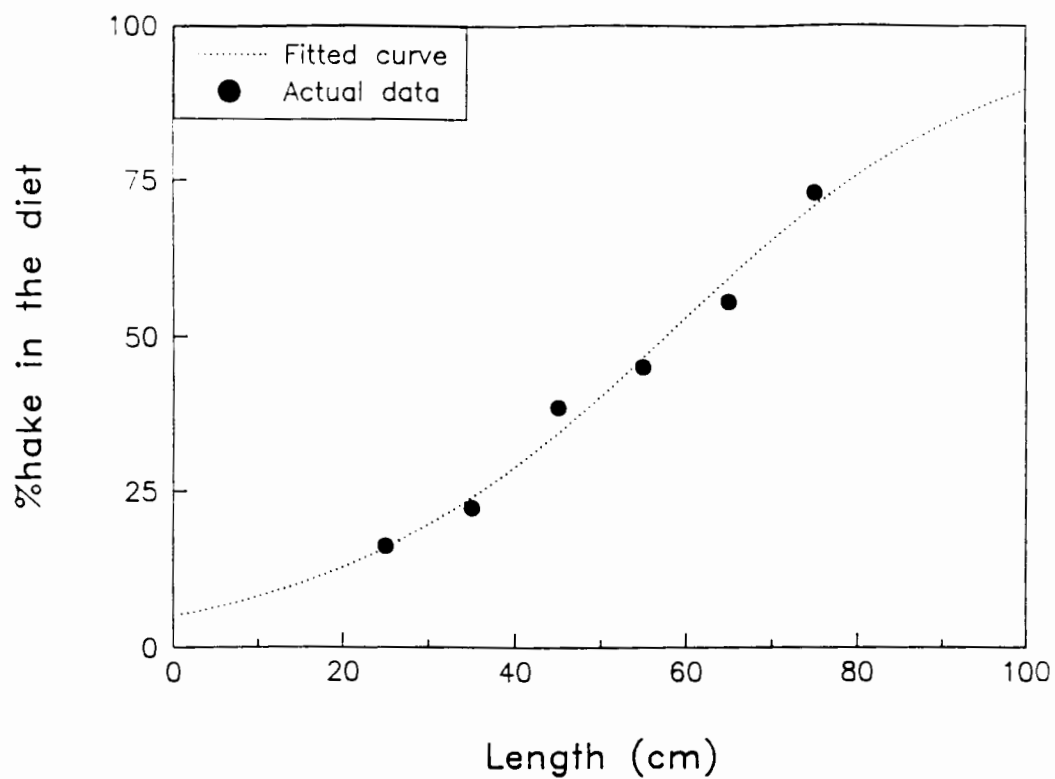


Figure 10.1 : The proportion by mass of hake in the diet of other hake (both species combined). The fit of a logistic curve to these data is indicated by the dotted line.

"K" is a combinatorial factor which is independent of the model parameters.

Transformation 10.20 was selected over alternatives (such the logit transformation), because it is appropriate if $\hat{Z}_{a,a'}$ is multinomially distributed. This assumption appears reasonable, because it does not seem unrealistic to imagine that predation involves selection of prey at random from the pool of available prey.

Perhaps not surprisingly, preliminary applications of this procedure suggested that the parameters Γ and M_b could not be determined satisfactorily from the data (see Table 10.8 and Section 10.8.2). For this reason, subsequent applications have been performed for various alternative fixed values of these parameters.

10.4.1 Variance estimation

In order to obtain bootstrap estimates of the variance of management quantities, a large number of fits of the model to alternative data sets were carried out. These alternative data sets were generated by conditioning on the catch-at-age data. The approach differs from that of Section 10.3.3 because error was added to the effort data to reflect the fluctuations in catchability. The bootstrap effort and stomach content data sets were generated as follows.

a) Effort data

$$E_y^U = E_y e^{v_y} \quad v_y \sim N(0; 0.16^2) \quad (10.21)$$

where E_y^U is the effort for year y in bootstrap data set U , and

E_y is the actual effort for year y .

The selection of 0.16 for σ_q is a compromise between the estimate of the residual variation (σ) of the fit of the Butterworth-Andrew observation error estimator to the CPUE data (~ 0.12) (Table 10.15), and the estimates of the residual variation in the fishing mortality versus fishing effort relationships for the *ad hoc* tuned VPA (~ 0.2) (Table 10.7).

b) Diet data

The diet composition data for each bootstrap data set is generated by assuming that each predator selects its prey multinomially. The $\hat{Z}_{a,a'}$ were therefore used to provide the multinomial probabilities used to generate the diet composition data for the bootstrap data sets.

10.4.2 Implementation of methodology

To retain comparability with the results of the "M-independent-of-age" *ad hoc* tuned VPA, tuning of the most-recent-year terminal fishing mortalities was achieved using effort data for the period 1978 to 1989. Ages in the range 1 to 7 were considered, and p was taken to be 2. Due to computer time restrictions, U_{\max} was set equal to 100. Fits have been performed for basal natural mortality (M_b) rates of 0.2 and 0.3yr^{-1} and for various values of Γ in the range 0 to $0.5\text{yr}^{-1}(\text{'000 tons})^{-1}$. The upper end of the latter range was selected, because setting Γ to a higher value resulted in completely unrealistic estimates of the mass of hake consumed by hake. In order to avoid undesirable oscillatory behaviour in a few of the applications, the maximum natural mortality considered was 2yr^{-1} .

10.5 The production model

The estimator which is used at present to provide TAC recommendations for the Cape hake resource off the South African west coast is the Butterworth-Andrew ($B_{1917}=K$; Schaefer form) observation error estimator (Punt 1990a, 1991).

10.5.1 The estimator

The fishery is modelled as follows:

$$B_{y+1} = B_y + g(B_y) - C_y \quad (10.22)$$

$$(C/E)_y = q \left(\frac{B_y + B_{y+1}}{2} \right) e^{v_y} \quad v_y \sim N(0, \sigma_v^2) \quad (10.23)$$

where B_y is the biomass at the start of year y and $B_{1917} = K$,

$g(B)$ is surplus production as a function of biomass,

$$g(B) = rB(1 - B/K) \text{ (Schaefer form),}$$

C_y is the actual catch during year y ,

$(C/E)_y$ is the actual CPUE for year y , and

σ_v^2 is the variance of the log of the observation error.

In addition to CPUE data, the estimator also uses biomass survey data to estimate the model parameters (Andrew *et. al.* 1989). The approach assumes that the biomass survey estimates are relative indices of abundance, and that they are normally distributed about their expected values. Estimates of the parameter values are then obtained by minimizing the (negative of the) appropriate log-likelihood function. By eliminating constants, the quantity minimized to obtain the parameter estimates can be shown to be:

$$-\ln L = \sum_Y' \left[\ln(\hat{\sigma}_V) + v_Y^2 / (2\hat{\sigma}_V^2) \right] + \sum_j'' [2(\sigma_j^S)^2]^{-1} [B_j^S - \hat{\Omega}' B_j]^2 \quad (10.24)$$

where Σ' is summation over all years (y) for which CPUE data are available,

Σ'' is summation over all available survey biomass estimates (j),

Ω is a constant of proportionality (the relative bias of the survey biomass estimates - assumed to be the constant across all years and seasons),

B_j^S is the j 'th absolute abundance estimate,

σ_j^S is an estimate of the standard error of B_j^S ,

$\hat{\Omega}' B_j^S$ is the resource biomass corresponding to survey estimate B_j^S (either B_k if the survey took place at the start of year k , or $(B_k + B_{k+1})/2$ if it was carried out in the middle of year k),

$v_Y = \ln(C/E)_Y - \ln(\hat{C}/\hat{E})_Y$, and

$\hat{\sigma}_V^2 = \Sigma' v_Y^2 / \Sigma' 1$.

In order to determine the sensitivity of the results of the surplus-production model-estimation procedure to its assumptions, a number of sensitivity tests were performed:

- a) $B_{1917/K}$ was estimated in the non-linear search instead of being fixed at unity,
- b) the Schaefer form of the surplus production function was replaced by the Fox form [$g(B) = rB(1 - \ln B / \ln K)$],
- c) the CPUE data were ignored when fitting the model, and
- d) the survey data were ignored when fitting the model.

10.5.2 Variance estimation

The 1000 bootstrap resamples, each of which contains a fixed catch series and random effort and survey biomass series, were generated using the predicted CPUE and survey biomass estimates obtained by fitting the Butterworth-Andrew model to the data. Error was then added to these predicted CPUE and survey biomass estimates according to the formulae:

$$(C/E)_Y^U = (\hat{C/E})_Y e^{v_Y} \quad v_Y \sim N(0; \sigma_v^2) \quad (10.25)$$

$$E_Y^U = C_Y / (C/E)_Y^U \quad (10.26)$$

$$B_j^{s,U} = \hat{\Omega} B_j + v_j \quad v_j \sim N(0; \sigma_j^2) \quad (10.27)$$

where $(\hat{C/E})_Y$ is the estimate of CPUE in year Y obtained by fitting the model-estimation procedure to the actual data,

$(C/E)_Y^U$ is the CPUE for year Y in bootstrap data set U ,

$\hat{\Omega} B_j$ is the model estimate corresponding to the j 'th survey biomass estimate, which is obtained by fitting the model-estimation procedure to the actual data, and

$B_j^{s,U}$ is the j 'th survey biomass estimate in bootstrap data set U .

Note that in order to use Equation (10.27), it is necessary to truncate the normal distribution $N(0; \sigma_j^2)$ so that there is a zero probability that $\hat{\Omega} B_j + v_j < 0$, but this is necessary only very infrequently in practice.

10.6 Data utilized in these assessments

The annual total hake catch-by-mass data appear in Table 4.1. This Table also gives the effort statistics for the South African fleet. Table 4.5 sets out the catch-at-age matrix and the mid-year masses-at-age. Table 4.6 lists the survey biomass estimates and their estimated standard errors.

Table 10.1 gives the number of hake in the stomachs of other hakes as a function of age. This matrix was obtained from the length distributions of hake in the these stomachs by applying the age-length key for 1989. [The year 1989 corresponds to the middle of the period during which the stomach content data were collected (August 1988 - January 1990).] Note that this use of the 1989 age-length key is likely to lead to some bias in the numbers estimated due to changes in year-class strength from one year to the next. The 1989 age-length key was used to split the distributions because it contains more data on the ages of larger fish than the August 1988 or January 1990 age-length keys.

10.7 Results

10.7.1 'M-independent of age' *ad hoc* tuned VPA

The fishing mortality-at-age ($F_{y,a}$) matrices obtained from the "base case" application of the 'M-independent-of-age' *ad hoc* tuned VPA model-estimation procedure and the five variants are given in Tables 10.2(a)-(f), while Tables 10.3(a)-(f) set out the numbers-at-age matrices obtained from these applications. The "base case" application has $M = 0.3\text{yr}^{-1}$ and is tuned using effort data from 1978 to 1989. Each of the

variants make a change to one of these two specifications. The 2^+ and exploitable biomass series and their C.V.s for each application are given in Table 10.4 and are illustrated in Figures 10.2 and 10.3. Table 10.5 provides estimates of annual recruitment ($N_{Y,1}$) and their C.V.s. These series are illustrated in Figure 10.4. Table 10.6 gives the measures of the variation in the fishing mortality-at-age versus fishing effort relationships (σ_a), while Table 10.7 sets out the estimates of age-specific selectivity (S_a).

10.7.2 'M-cannibalistic' *ad hoc* tuned VPA

Table 10.8 provides the results of applying a number of variants of the 'M-cannibalistic' *ad hoc* tuned VPA model-estimation procedure. These variants correspond to $M_b = 0.2\text{yr}^{-1}$ and 0.3yr^{-1} and Γ in the range 0 to $0.5\text{yr}^{-1}(\text{'000 tons})^{-1}$. Table 10.8 contains the value of $-\ln L$ and estimates of the parameters R_f and σ_w , the average exploitable and 2^+ biomasses over the period 1987 to 1989, the average mass of hake consumed by hake over this period, as well as the exploitable component of the average pristine level (K^e).

The fishing mortality-at-age ($F_{Y,a}$) and numbers-at-age ($N_{Y,a}$) matrices for six of these variants of the 'M-cannibalistic' *ad hoc* tuned VPA model-estimation procedure are given in Tables 10.9 and 10.10 respectively. These six variants were selected because they provided estimates of the ratio of the mass of hake consumed by hake to exploitable biomass, which encompass the corresponding range inferred from Section 2.5. Table 10.11 provides estimates of natural mortality by age and year ($M_{Y,a}$) for the six variants, while Table 10.12 provides

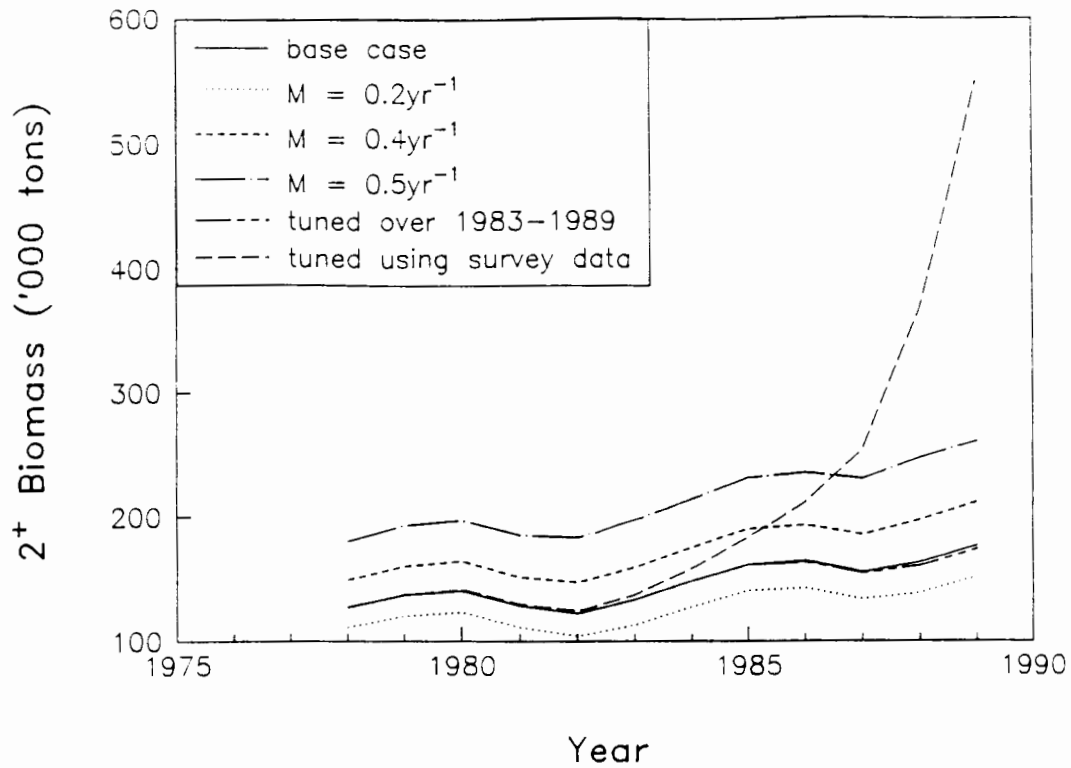


Figure 10.2 : The 2⁺ biomass time series for the six applications of the "M-independent-of-age" ad hoc tuned VPA.

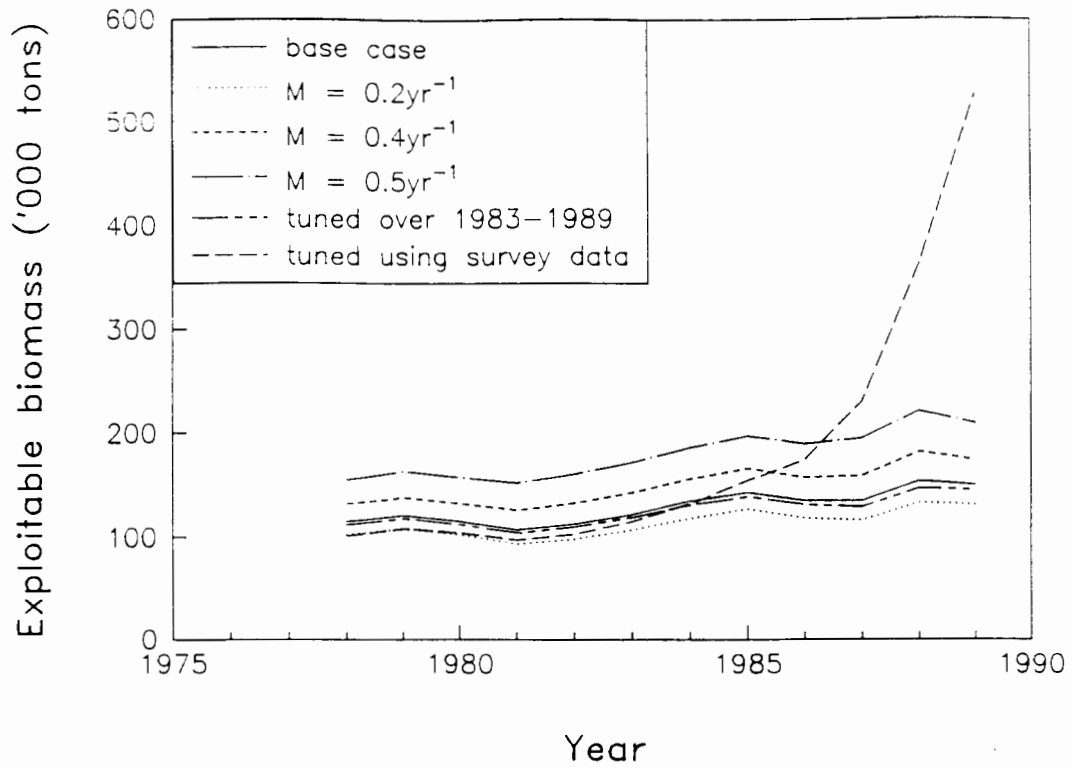


Figure 10.3 : The exploitable biomass time series for the six applications of the "M-independent-of-age" ad hoc tuned VPA.

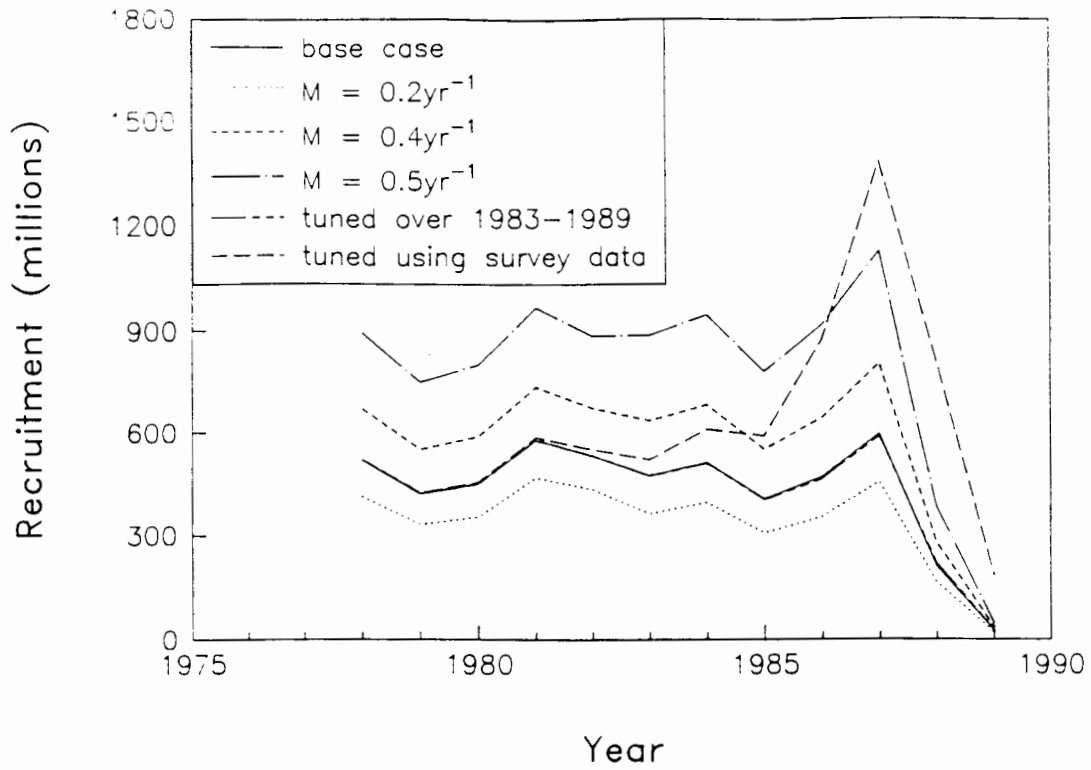


Figure 10.4 : The recruitment time series for the six applications of the "M-independent-of-age" ad hoc tuned VPA.

the residuals of the model fit to the stomach content data. Tables 10.13 and 10.14 give the estimates of the time series of exploitable biomass, 2+ biomass, recruitment, and hake consumed by hake with their estimated bootstrap C.V.s. The former three time series are illustrated for four of the six variants in Figures 10.5 to 10.7.

10.7.3 Production model

Tables 10.15(a)-(e) give the estimates of the model parameters and variables, their C.V.s and their 95% confidence intervals obtained from fits to the "base case" and the four alternative data set choices specified in Section 10.5.1. The estimated biomass series are illustrated in Figure 10.8. Figures 10.9(a)-(d) compare the estimated and actual CPUE time series. Time series of residuals are also shown in these Figures. These residuals are the differences between the logarithms of the observed and model-predicted CPUE values.

10.8 Discussion

10.8.1 'M-independent-of-age' *ad hoc* tuned VPA

The results of the "base case" application are considered first, in isolation. Differences evident for the other variants are discussed subsequently.

The biomass estimates obtained from the "base case" application of the 'M-independent-of-age' *ad hoc* tuned VPA model-estimation procedure are relatively well determined (C.V.s < 5% for all but the final two years of the series) [Table 10.4]. These series indicate that the resource has been increasing reasonably steadily, if slowly, over the last

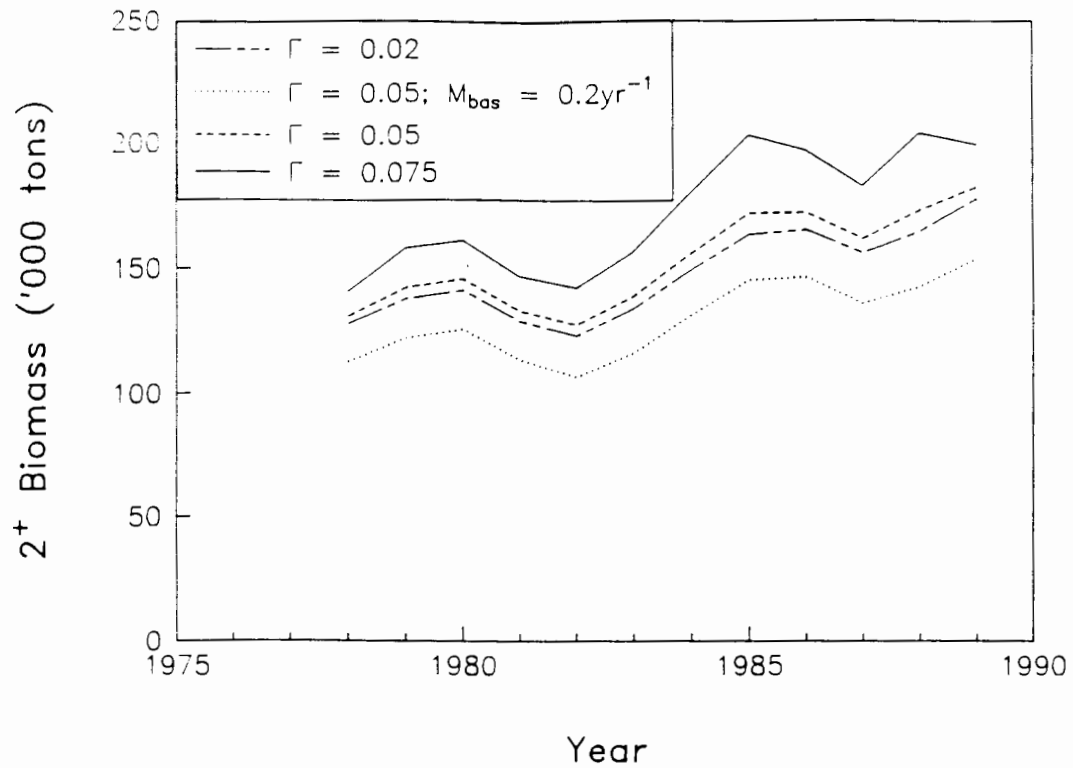


Figure 10.5 : The 2^+ biomass time series for four variants of the "M-cannibalistic" ad hoc tuned VPA. $M_D = 0.3yr^{-1}$ except where indicated.

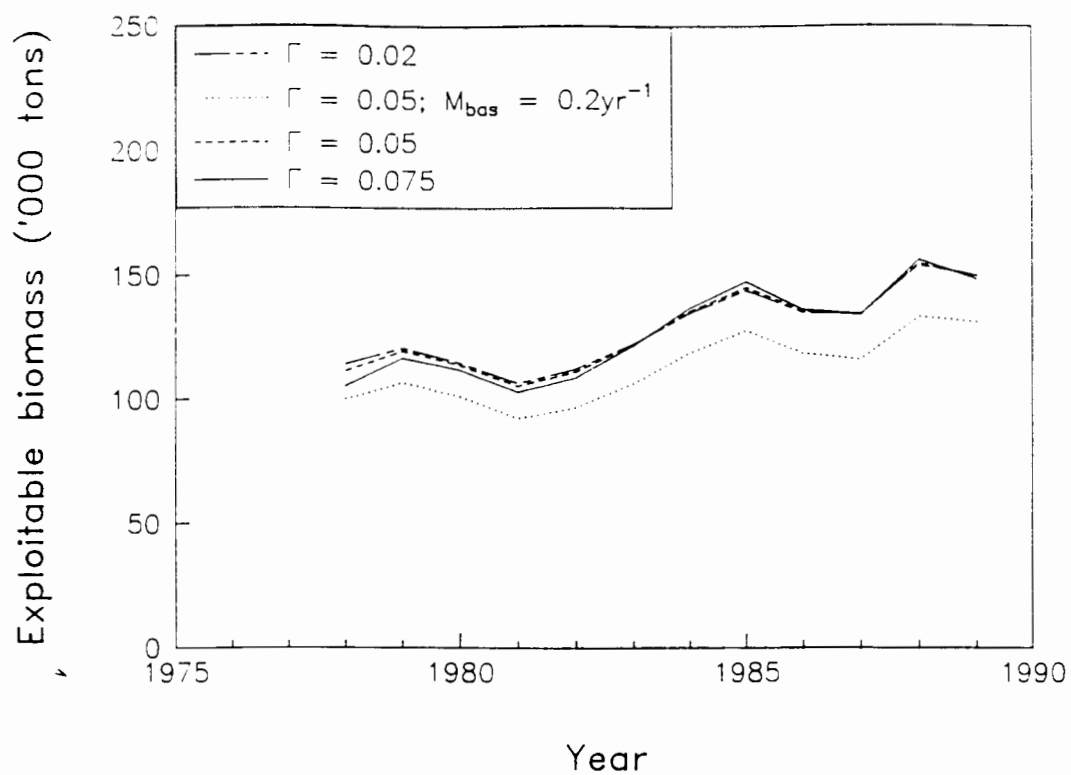


Figure 10.6 : The exploitable biomass time series for the same four variants of the "M-cannibalistic" ad hoc tuned VPA as in Figure 10.5.

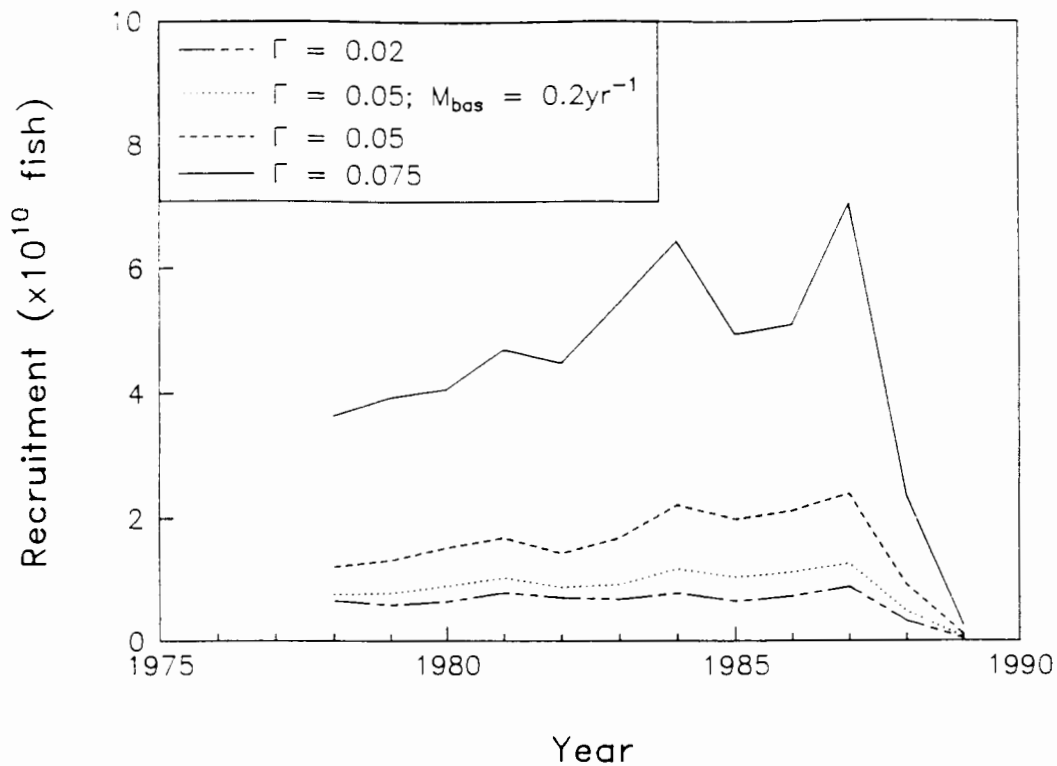


Figure 10.7 : The recruitment time series for the same four variants of the "M-cannibalistic" ad hoc tuned VPA as in Figure 10.5.

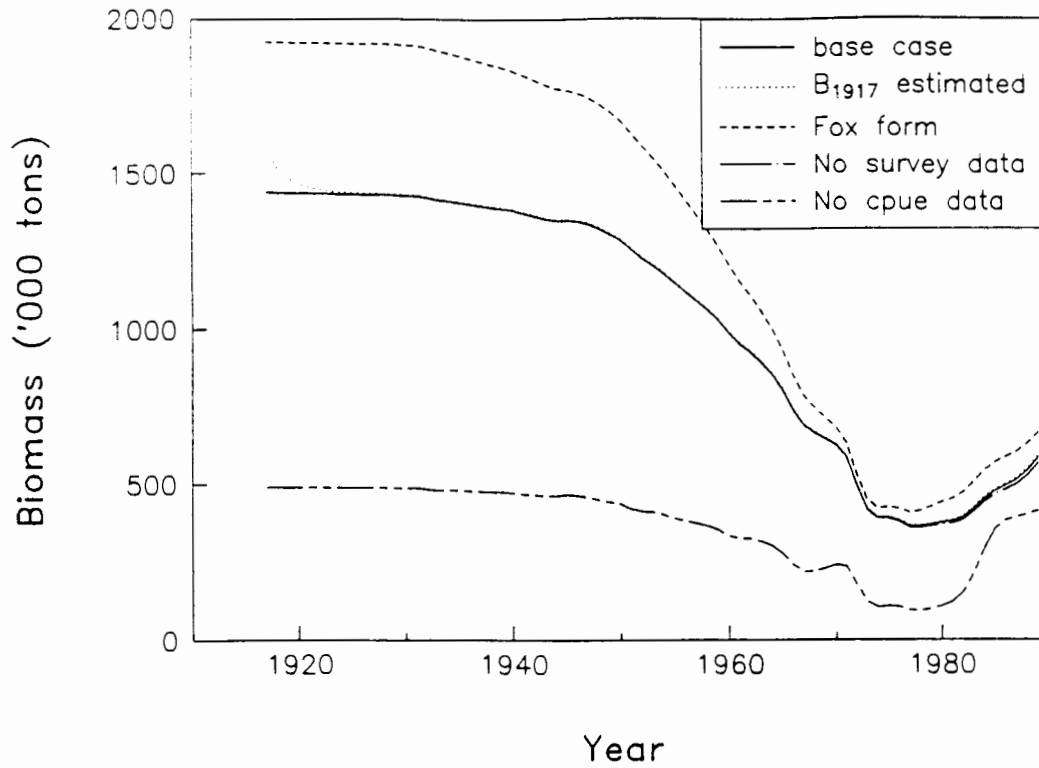


Figure 10.8 : Biomass time series for the "base case" and four alternative applications of the Butterworth-Andrew observation error estimator.

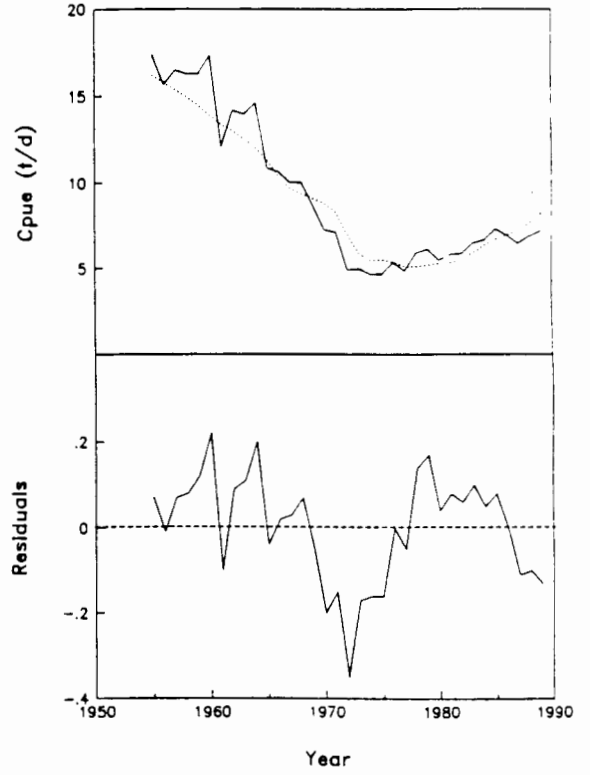
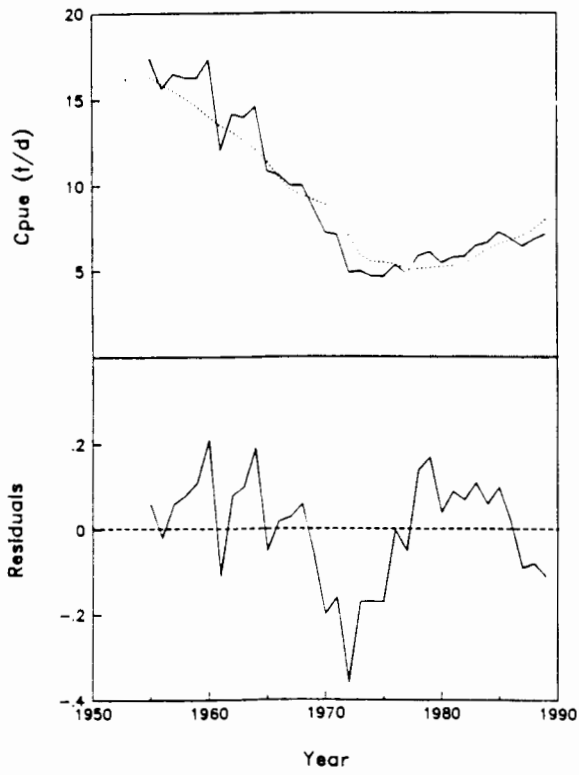
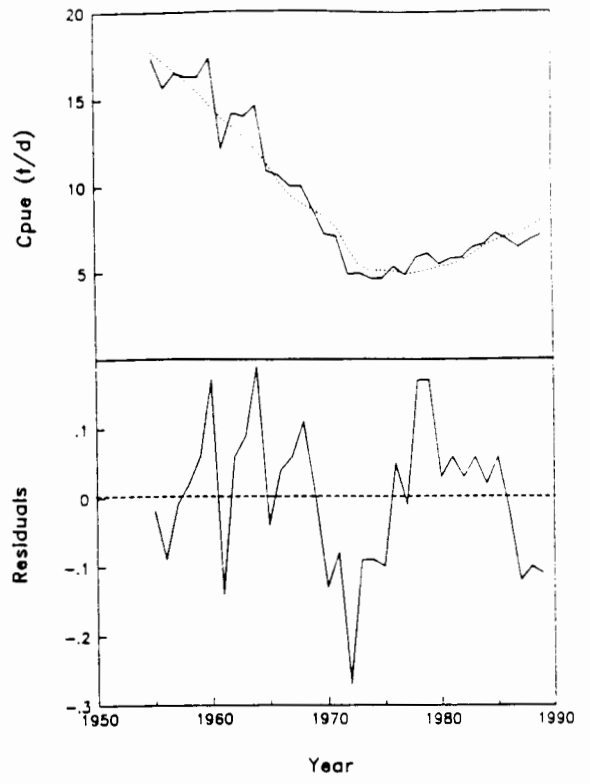
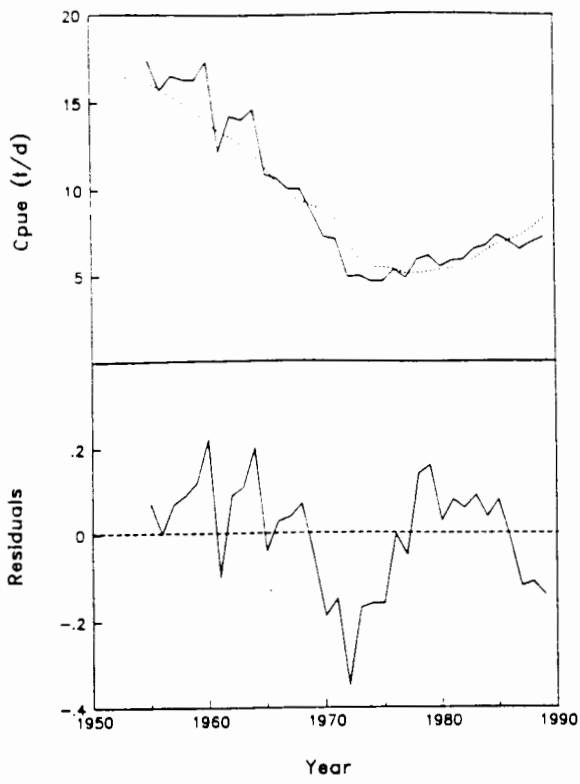


Figure 10.9 : CPUE residual time series for the four applications of the Butterworth-Andrew observation error estimator which make use of CPUE data.

twelve years (Figures 10.3 and 10.4). However, the exploitable biomass is estimated still to be a small fraction of its average pristine size ($B_{1989}^e / K^e = 19\%$). [Note, however, that the estimate of K^e is relatively poorly determined (C.V.=13.3%), and may be negatively biased. This is because, if a stock is depleted to relatively low levels (as appears to be the case for this hake resource), the average recruitment over the period 1978 to 1987 (which is used to estimate K^e - see Section 10.3.4) may be reduced due to the effects of a stock-recruitment relationship. On the other hand, the average recruitment may be positively biased because of the lack of large predators to cannibalize small fish, or because of a negatively biased value for M].

The number of older fish (>4 years) in the population is estimated to have increased by over 50% over the period 1978-1989 [Table 10.3], whereas the exploitable biomass has increased by less than 40% over this period [Table 10.4]. This is a result of the decreasing trend in recruitment [Table 10.5] in the last two years, coupled with the low age-at-50%-selectivity (<2 years - see Table 10.7). The estimates of recruitment for the last two years are very poorly determined (the C.V. of the estimate of recruitment for 1989 is larger than 100% for the "base case" application - see Table 10.5). Further, the estimates of σ_a reported in Table 10.6 suggest that the fishing mortality versus fishing effort relationship for age 1 has almost no predictive power ($\sigma_a=0.983$ for age 1). The σ_a 's for all the other ages are all appreciably smaller than this. Thus, the confidence which can be placed in the estimates of recruitment for the last two years is very poor.

This justifies the exclusion of these data points from the calculation of K^e and MSY.

The age-specific selectivity pattern obtained from the "base case" application is virtually flat from age 2 onwards [Table 10.7]. The selectivity on age 1 ($S_a=0.074$ - Table 10.7) is markedly smaller than that for age 2. This is probably a result of the size graduation by depth (the majority of 1-year-old *M. capensis* are found in the 0-100m depth zone in which very little trawling takes place) and the effects of discarding of small fish. As discarding is illegal, it is not possible to obtain even a qualitative assessment of this effect. The results of Table 10.7 suggest that the use of any integrated analysis approaches which make the assumption that selectivity-at-age can be modelled by a smooth curve over the complete age-range are likely to provide positively biased estimates of the selectivity on fish of age 1, because very few smooth curves could fit the data in Table 10.7 particularly well.

When M is larger than 0.3yr^{-1} , the biomass time series are larger than for the "base case" application and *vice versa* [Figures 10.2 and 10.3]. Another feature of the results for different values of M is that MSY varies considerably (from 99 000 tons when $M=0.4\text{yr}^{-1}$ to 120 000 tons for $M=0.2\text{yr}^{-1}$ - Table 10.4). The estimates of current depletion (B/K) are also sensitive to the value of M . These range from 13% [95% CI: 9 - 18%] for $M=0.2\text{yr}^{-1}$ to 36% [95% CI: 27 - 48%] for $M=0.5\text{yr}^{-1}$. This suggests that not too much confidence can be placed on the "base case" point estimate of current depletion (19%), as the value

of 0.3yr^{-1} for M is little more than a conventional guess which became "standard" primarily through its regular use by ICSEAF.

Tuning the VPA using the survey biomass indices instead of the effort data results in markedly different interpretations of the current status and productivity of the resource. This assessment suggests that the stock is currently above 50% of its average pre-exploitation level, that MSY is 140 000 tons and that the current exploitable biomass is 526 000 tons. These estimates are appreciably larger than the corresponding values obtained in the "base case" application [106 000 and 150 000 tons respectively]. The exceptionally large biomass for 1989 is a result of estimated huge recruitments between 1986 and 1988, coupled with a very low fishing mortality in recent years [Tables 10.3(f) and 10.2(f)]. One of the reasons for the low fishing mortality in 1989 is that the survey biomass index for 1989 was the largest ever recorded (see Table 4.6). This estimate is treated as exact when tuning the VPA. However, consideration of the C.V.s of the biomass and recruitment estimates [Tables 10.4 and 10.5] and of the σ_a 's (Table 10.6) shows that the estimates obtained by tuning using the survey data are particularly imprecise for the last 4-5 years, and these estimates therefore cannot be accorded much confidence. The effects of using effort data for 1983 - 1989 to tune the most-recent-year terminal fishing mortalities (i.e. ignoring effort data for 1978 to 1982) are hardly noticeable.

10.8.2 'M-cannibalistic' *ad hoc* tuned VPA

Not surprisingly, increasing the degree of cannibalism (Γ) from 0 to $0.5\text{yr}^{-1}(\text{'000 tons})^{-1}$ [Table 10.8] results in marked increases in the estimates of the annual average amount of hake consumed by hake over the period 1987 - 1989. These estimates range from unrealistically low (nothing) to unrealistically high (21 million tons per annum). While the 2^+ biomass estimates increase somewhat as Γ is increased, this effect is hardly noticeable for the exploitable biomass estimates [Tables 10.8 and 10.13, and Figures 10.5 and 10.6]. This is a consequence of the estimation of the selectivity pattern by the assessment procedure - as Γ is increased, the selectivity on smaller fish is reduced.

A notable feature of Table 10.8 is that the log-likelihood does not vary much across the wide range of Γ considered. This suggests that the data fitted cannot be used to distinguish between alternative values for Γ . While the value of σ_w is not sensitive to the value of Γ , there is a trend in the estimate of R_f with this parameter. The range of current depletion estimates can be inferred by dividing the estimate of the average exploitable biomass over the period 1987 to 1989 (B^e) by the estimates of the exploitable component of the average pristine level (K^e). These depletion estimates range from highly overexploited (13% - $\Gamma=0$; $M_b=0.2\text{yr}^{-1}$) to substantially above the average pristine level (637% - $\Gamma=0.5\text{yr}^{-1}(\text{'000 tons})^{-1}$; $M_b=0.3\text{yr}^{-1}$). This last result occurs because, if cannibalism is very intense, the removal of the larger hake by the fishery reduces the amount of cannibalism substantially, which leads to an increase in the overall biomass.

However, for the most part, the results for the 'M-cannibalistic' *ad hoc* tuned VPA model-estimation procedure are qualitatively similar to those for the 'M-independent-of-age' procedure. Both these procedures suggest that the exploitable biomass was at a low point in 1981, that it has been increasing since then, and that it peaked in 1988 (Tables 10.4 and 10.13). The two estimated trends in the increase in exploitable biomass since 1981 are also similar [3.9% per year for the $\{M_b=0.3\text{yr}^{-1}; \Gamma=0.05\text{yr}^{-1}(\text{'000 tons})^{-1}\}$ 'M-cannibalistic' application and 3.6% for the $(M=0.3\text{yr}^{-1})$ 'M-independent-of-age' application].

The most marked difference between the results of the 'M-cannibalistic' and 'M-independent-of-age' VPAs are the recruitment (1-year-class strength) estimates (compare Figures 10.4 and 10.7). The former estimates are markedly larger than the latter, although the trends in recruitment are relatively insensitive to the treatment of cannibalism. In addition, as for the 'M-independent-of-age' case, the magnitude of the recruitment estimates increases when M_b is increased (from 0.2 to 0.3yr^{-1}). For age 4 and older, the estimates of fishing-mortality-at-age and numbers-at-age are virtually identical to the 'M-independent-of-age' estimates (for a value of M_b equal to that used for M for the latter assessments).

The estimates of the mass of hake consumed by hake given in Table 10.14 are appreciably smaller than the estimates of the corresponding quantities given in Section 2.5, for all the values of Γ considered. However, the biomass estimates considered in the calculations reported in Section 2.5 are substantially larger than the exploitable biomasses estimated by the VPA. The estimates of the ratio of the mass of hake

consumed by hake to exploitable biomass ($PRED/B^e$) inferred from Table 10.8, and Tables 10.13 and 10.14, overlap the corresponding estimates inferred from the results provided in Section 2.5. In principle therefore, it should be possible to select a value for Γ on the basis of how well the estimates of $PRED/B^e$ inferred from the results of Section 2.5 are predicted by assessments for different values of M . This is not possible in this case, however, because the value of this ratio inferred from Section 2.5 depends on the treatment of everted stomachs, whether or not the stomach content data are analyzed by geographic strata, and which value is used for the evacuation time for fish consumed. In addition, the estimates of annual consumption reported in Section 2.5 are very imprecisely determined (even though the estimates of C.V.s shown are negatively biased). For these reasons, only those values of Γ which correspond to unrealistically high estimates of hake consumption by hake can be excluded.

The estimates of natural mortality by age and year (Table 10.11) show that cannibalism can be substantial on fish younger than three years, even for low values of Γ . Table 10.12 shows that, in general, the fits of the 'M-cannibalistic' model to the hake predation data are good.

The estimates of the preferred predator-prey length ratio ($R_f = 0.220 - 0.295$ for the six variants considered in detail - see Table 10.8) is higher than the value of a comparable parameter used by Shepherd (1988) in his preliminary investigations into predation by cod. Shepherd (op. cit.) considered a preferred predator-prey mass ratio of 0.001, in contrast to the range of 0.011 to 0.026 obtained here (calculated by cubing the extremes of the range of the estimates of the preferred

predator-prey length ratio). The values estimated may thus seem unrealistically high. However, it may be that the preferred predator-prey length ratio is driven by mainly spatial considerations rather than by desirability in this instance.

The long-term predictions of the 'M-cannibalistic' *ad hoc* tuned VPA are markedly different from those of the 'M-independent-of-age' version - K^e is often estimated to be below the current exploitable biomass; and for a few variants, MSY is estimated to be larger than that provided by the 'M-independent-of-age' *ad hoc* tuned VPA model-estimation procedure (compare Tables 10.4 and 10.13). However, the K^e estimates provided by the 'M-cannibalistic' *ad hoc* tuned VPA model-estimation procedure are poorly determined, while the MSY estimates correspond to extremely high fishing mortalities. At such levels of fishing mortality, the number of predators is much reduced, so that huge numbers of juveniles are available to be caught. These estimates of MSY may be unreliable because they correspond to very small numbers of fish older than the age-at-maturity. Had the calculations incorporated a stock-recruitment relationship, recruitment and hence sustainable yield at these levels of fishing intensity would undoubtedly have been much less. Further, for the calculations reported, the average size of fish caught at MSY would be very small. This is undesirable from an industrial marketing point of view.

The precision with which estimates are obtained by the 'M-cannibalistic' *ad hoc* tuned VPA is worse than for the 'M-independent-of-age' procedure (particularly for recruitment - see Table 10.14). Fixing the values of R_f and σ_w , and applying the 'M-cannibalistic' VPA

and the variance-estimation procedure described in Section 10.4.1, results in C.V.s for the recruitment estimates which are comparable with those obtained from the 'M-independent-of-age' procedure. This suggests that the imprecision of the recruitment estimates is a consequence of imprecision associated with the estimation of the predation model parameters R_f and σ_w from the stomach content data.

10.8.3. Production model

Examination of Figures 10.9(a) to (d) for patterns in the residuals of the production model fits to the CPUE data reveals that these fits are significantly (but not substantially) mis-specified. (Note the runs of negative residuals in the early 1970's, followed by positive residuals in the early 1980's.) This may not be too serious, however, because the effect of process error on observation error estimators is the introduction of positive correlation in the residuals [Walters (1986), Punt (1988)].

One advantage of the production model assessments is that estimates of historic (prior to 1978) biomass can be obtained. Although the resource is assessed to be increasing at present (Figure 10.8), the "base case" assessment indicates that it has not yet reached its MSY level (B_{MSY}) [Table 10.15]. The resource is, however, assessed to be far less depleted than suggested by the "base case" 'M-independent-of-age' *ad hoc* tuned VPA assessment [42% as opposed to 19%]. The differences between the VPA and production-model estimates of current depletion virtually disappear when M is taken to be 0.5yr^{-1} . The estimate of MSY obtained from the production model is larger than the MSY estimates for

the VPA assessments (whichever value is chosen for M), and also larger than those for the 'M-cannibalistic' VPA. The estimate of current biomass of 604 000 tons is more than four times larger the "base case" 'M-independent-of-age' *ad hoc* tuned VPA current exploitable biomass estimate and more than three times larger the $M_b=0.3\text{yr}^{-1}$ 'M-cannibalistic' *ad hoc* tuned VPA estimates. Agreement with the 'M-independent-of-age' assessments is improved if the value of M is increased, but even for $M=0.5\text{yr}^{-1}$ there is still a three fold discrepancy. Although better agreement between the VPA and production model current biomass estimates can be obtained by tuning the VPA using survey biomass data, the latter assessment is particularly imprecise, and the disagreements noted above remain evident for earlier years.

Estimating B_{1917}/K instead of fixing it at unity makes very little difference to the estimates of the model parameters. This is because, due to the long time span considered, any transient effects of B_{1917}/K ratios which are not unity will have died away by the time the catches became substantial (see Figure 10.8). The C.V.s of the estimates given in Table 10.15(b) are much smaller than those for the "base case" application because the log-likelihood surface becomes very flat when the parameter B_{1917}/K is estimated from the data, and the non-linear minimization procedure "converges" to values of parameters close to the initial choices for the minimization process.

If the Fox rather than the Schaefer form of the surplus production function is used in the estimator, the estimate of K increases. This is not surprising because the Fox form has a long tail at high biomass (nearer K) over which the surplus production is low, so that historic

catches comprise a greater proportion of reduction of standing stock and less of surplus production than for the Schaefer model. The Fox form estimates the stock to be virtually at B_{MSY} , but it should be remembered that B_{MSY}/K for this form is lower than for the Schaefer form for which the current depletion estimate is below MSY level.

Ignoring the survey biomass data when fitting the surplus production model makes little difference to the parameter estimates, although these estimates are marginally less precise as would be expected given fewer data. Ignoring the CPUE data when fitting the model results in estimates which are markedly different from the "base case" application. In particular, the stock is assessed to be well above B_{MSY} and to have a larger MSY (of 160 000 tons). These estimates are only slightly less precise than the corresponding "base case" estimates, so that this fit cannot be rejected on the grounds of imprecision. However, the large estimate for the parameter r (of 1.354) seems unrealistically high. In addition, this assessment is highly sensitive to the values of the last two data points, which suggests that its results are probably unreliable.

The applications of the production model also provide estimates of the bias of the survey biomass indices (which are reported as absolute values on the basis of "swept area" methodology - see Table 4.6 and the comments in Section 4.4). These range from 0.438 to 0.766 [Table 10.15]. The last of these estimates should be viewed with considerable caution as it is very imprecisely determined (C.V. >1000%). A more reasonable upper bound for this range is 0.522 (corresponding to the "base case" application).

10.8.4 General discussion

There thus appears to be a fundamental and substantial conflict between some of the results of the production-model and the age-structured assessment techniques. While it is possible to achieve similar estimates of current depletion by adjusting the value of M (or Γ), it is not possible to reconcile the estimates of biomass and MSY. Also of importance is that the precision with which the estimates of current biomass and MSY are provided by the 'M-independent-of-age' *ad hoc* tuned VPA are worse than those provided by the production-model.

One of the key questions which will be addressed in subsequent Chapters is whether either of the production model or the 'M-independent-of-age' *ad hoc* tuned VPA estimators can be rejected as being inappropriate for use in providing TAC recommendations for the Cape hake resource off the South African west coast.

Table 10.1 : Numbers of hake in the stomachs of other hake as a function of age, calculated from the observed length distributions of hake in these stomachs by applying the age-length key for 1989.

PREDATOR AGE	PREY AGE					
	1	2	3	4	5	6
1	-	-	-	-	-	-
2	7.062	-	-	-	-	-
3	14.909	0.551	-	-	-	-
4	12.842	3.759	-	-	-	-
5	17.517	10.727	0.322	-	-	-
6	19.059	11.089	1.764	0.016	0.002	-
7	7.635	10.694	1.791	0.620	0.085	-

Table 10.2 : Estimates of fishing mortality ($F_{y,a}$) matrices obtained from the 'M-independent-of-age' *ad hoc* tuned VPA. Units are yr^{-1} .

(a) "Base case" application

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.062	0.955	0.680	0.407	0.523	0.617	0.568
1979	0.084	0.591	0.607	0.486	0.819	0.758	0.788
1980	0.034	0.574	0.724	0.745	0.830	0.827	0.828
1981	0.161	0.743	0.681	0.735	0.687	0.893	0.783
1982	0.304	0.698	0.485	0.431	0.627	0.644	0.636
1983	0.056	0.448	0.512	0.427	0.554	0.595	0.574
1984	0.044	0.406	0.602	0.539	0.539	0.560	0.549
1985	0.033	0.422	0.583	0.547	0.666	0.784	0.723
1986	0.013	0.342	0.698	0.572	0.952	1.079	1.013
1987	0.012	0.424	0.756	0.611	0.653	0.636	0.644
1988	0.055	0.469	0.492	0.326	0.510	0.700	0.598
1989	0.040	0.418	0.485	0.407	0.519	0.572	0.545

(b) $M = 0.2\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.075	1.083	0.785	0.468	0.599	0.702	0.649
1979	0.103	0.679	0.697	0.556	0.923	0.855	0.888
1980	0.041	0.672	0.828	0.853	0.934	0.938	0.936
1981	0.193	0.872	0.802	0.847	0.793	1.019	0.899
1982	0.367	0.813	0.577	0.509	0.727	0.748	0.737
1983	0.070	0.525	0.597	0.500	0.642	0.688	0.665
1984	0.055	0.480	0.695	0.618	0.614	0.639	0.626
1985	0.042	0.494	0.679	0.623	0.748	0.879	0.811
1986	0.016	0.406	0.810	0.662	1.082	1.217	1.148
1987	0.015	0.508	0.892	0.714	0.750	0.737	0.744
1988	0.068	0.555	0.582	0.386	0.593	0.805	0.691
1989	0.050	0.489	0.566	0.472	0.594	0.653	0.623

(Table 10.2 Continued)

(c) $M = 0.4\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.050	0.825	0.577	0.348	0.449	0.533	0.489
1979	0.067	0.503	0.517	0.417	0.714	0.659	0.686
1980	0.027	0.478	0.619	0.635	0.722	0.715	0.718
1981	0.131	0.616	0.563	0.624	0.581	0.766	0.667
1982	0.245	0.586	0.398	0.357	0.529	0.543	0.536
1983	0.043	0.373	0.430	0.358	0.468	0.505	0.486
1984	0.034	0.334	0.510	0.462	0.465	0.482	0.473
1985	0.025	0.350	0.488	0.471	0.582	0.687	0.632
1986	0.010	0.281	0.587	0.483	0.819	0.937	0.876
1987	0.009	0.344	0.625	0.510	0.556	0.536	0.546
1988	0.043	0.386	0.406	0.270	0.429	0.596	0.506
1989	0.032	0.347	0.407	0.344	0.445	0.491	0.467

(d) $M = 0.5\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.039	0.692	0.476	0.289	0.375	0.449	0.410
1979	0.051	0.415	0.427	0.349	0.606	0.559	0.582
1980	0.021	0.383	0.514	0.525	0.612	0.600	0.606
1981	0.103	0.493	0.450	0.513	0.477	0.637	0.551
1982	0.190	0.477	0.317	0.287	0.434	0.445	0.439
1983	0.032	0.301	0.352	0.292	0.386	0.417	0.401
1984	0.026	0.265	0.420	0.387	0.391	0.404	0.398
1985	0.019	0.281	0.396	0.393	0.496	0.586	0.540
1986	0.007	0.221	0.477	0.396	0.683	0.791	0.735
1987	0.007	0.269	0.499	0.413	0.459	0.439	0.449
1988	0.033	0.306	0.325	0.216	0.351	0.493	0.416
1989	0.024	0.279	0.330	0.283	0.371	0.410	0.390

(Table 10.2 Continued)

(e) Tuned using effort data for 1983-1989

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.062	0.955	0.680	0.407	0.523	0.617	0.568
1979	0.084	0.591	0.607	0.486	0.819	0.758	0.788
1980	0.034	0.574	0.724	0.745	0.830	0.827	0.829
1981	0.161	0.743	0.681	0.735	0.687	0.893	0.783
1982	0.304	0.698	0.485	0.431	0.627	0.644	0.636
1983	0.056	0.448	0.513	0.427	0.554	0.596	0.575
1984	0.044	0.406	0.603	0.540	0.540	0.560	0.550
1985	0.033	0.424	0.585	0.549	0.667	0.786	0.724
1986	0.013	0.344	0.704	0.575	0.957	1.084	1.019
1987	0.012	0.431	0.763	0.621	0.659	0.644	0.651
1988	0.053	0.475	0.505	0.331	0.524	0.713	0.612
1989	0.040	0.398	0.495	0.424	0.531	0.600	0.564

(f) Tuned using survey biomass data

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.062	0.954	0.679	0.407	0.523	0.617	0.568
1979	0.084	0.589	0.606	0.485	0.818	0.756	0.786
1980	0.034	0.570	0.720	0.740	0.826	0.823	0.825
1981	0.158	0.732	0.671	0.727	0.679	0.884	0.775
1982	0.293	0.682	0.472	0.421	0.614	0.630	0.622
1983	0.051	0.424	0.491	0.409	0.532	0.572	0.552
1984	0.037	0.359	0.551	0.503	0.504	0.523	0.513
1985	0.023	0.339	0.481	0.470	0.589	0.691	0.638
1986	0.007	0.220	0.490	0.417	0.709	0.825	0.765
1987	0.005	0.205	0.387	0.337	0.389	0.365	0.377
1988	0.014	0.176	0.183	0.123	0.211	0.302	0.253
1989	0.005	0.089	0.132	0.112	0.149	0.168	0.158

Table 10.3 : Estimates of numbers-at-age ($N_{y,a}$) matrices obtained from the 'M-independent-of-age' *ad hoc* tuned VPA. Units are millions.

(a) "Base case" application

YEAR	AGE						
	1	2	3	4	5	6	7
1978	518	490	130	49	17	4	1
1979	421	361	140	49	24	7	2
1980	449	287	148	56	22	8	3
1981	578	322	120	53	20	7	3
1982	532	365	113	45	19	7	2
1983	474	291	134	52	22	7	3
1984	512	332	138	60	25	9	3
1985	405	363	164	56	26	11	4
1986	469	290	176	68	24	10	4
1987	596	343	153	65	28	7	2
1988	208	436	166	53	26	11	3
1989	25	146	202	75	28	12	4

(b) $M = 0.2\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	411	437	113	42	14	4	1
1979	332	313	121	42	22	6	2
1980	353	245	130	49	20	7	2
1981	467	278	102	46	17	6	2
1982	433	315	95	38	16	6	2
1983	363	246	114	44	19	6	2
1984	395	277	119	52	22	8	3
1985	306	306	140	49	23	10	3
1986	353	241	153	58	21	9	3
1987	454	284	131	56	25	6	2
1988	161	366	140	44	22	10	2
1989	19	123	172	64	25	10	3

(Table 10.3 Continued)

(c) $M = 0.4\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	669	560	153	59	19	5	1
1979	550	426	165	58	28	8	2
1980	588	345	173	66	25	9	3
1981	735	384	143	62	23	8	3
1982	673	432	139	55	22	9	3
1983	636	353	161	63	26	9	3
1984	683	408	163	70	29	11	4
1985	551	442	196	66	30	12	4
1986	643	360	209	81	27	11	4
1987	804	427	182	78	33	8	3
1988	277	534	203	65	31	13	3
1989	33	178	243	91	33	14	5

(d) $M = 0.5\text{yr}^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	890	659	186	72	24	6	1
1979	749	519	200	70	33	10	2
1980	800	432	208	79	30	11	3
1981	966	475	179	75	28	10	4
1982	884	529	176	69	27	11	3
1983	887	443	199	78	31	11	4
1984	945	521	199	85	35	13	4
1985	780	559	242	79	35	14	5
1986	916	464	256	99	32	13	5
1987	1126	552	226	96	40	10	4
1988	382	678	256	83	39	15	4
1989	45	224	303	112	41	16	6

(Table 10.3 Continued)

(e) Tuned using effort data for 1983-1989

YEAR	AGE						
	1	2	3	4	5	6	7
1978	518	490	130	49	17	4	1
1979	421	361	140	49	24	7	2
1980	449	287	148	56	22	8	3
1981	578	322	120	53	20	7	3
1982	532	364	113	45	19	7	2
1983	473	291	134	52	22	7	3
1984	510	331	138	60	25	9	3
1985	403	362	164	56	26	11	4
1986	463	289	175	68	24	10	4
1987	590	338	152	64	28	7	2
1988	216	432	163	52	26	11	3
1989	25	152	199	73	28	11	4

(f) Tuned using survey biomass data

YEAR	AGE						
	1	2	3	4	5	6	7
1978	519	490	130	49	17	4	1
1979	423	362	140	49	24	7	2
1980	454	288	149	57	22	8	3
1981	586	325	121	54	20	7	3
1982	550	371	116	46	19	8	2
1983	521	304	139	54	22	8	3
1984	610	367	147	63	26	10	3
1985	590	436	190	63	28	12	4
1986	874	428	230	87	29	12	4
1987	1382	643	254	105	42	11	4
1988	811	1019	388	128	55	21	5
1989	183	592	633	239	84	33	12

Table 10.4 : Biomass estimates in '000 tons with their corresponding estimated C.V.s (expressed as a percentage) in parenthesis obtained from the 'M-independent-of-age' *ad hoc* tuned VPA. K^e is the estimate of the exploitable component of the average pre-exploitation biomass.

YEAR	"BASE CASE"		M = 0.2yr ⁻¹	
	2+	EXPLOITABLE	2+	EXPLOITABLE
1978	127 (0.6)	115 (0.7)	111 (0.4)	102 (0.6)
1979	137 (0.7)	121 (0.8)	120 (0.5)	108 (0.7)
1980	140 (0.7)	115 (1.0)	123 (0.6)	102 (0.8)
1981	128 (0.9)	107 (1.1)	111 (0.7)	93 (0.9)
1982	122 (1.0)	113 (1.1)	104 (0.8)	98 (0.9)
1983	133 (1.0)	122 (1.1)	113 (0.8)	107 (1.0)
1984	148 (0.9)	135 (1.1)	127 (0.7)	118 (0.9)
1985	161 (0.8)	143 (1.1)	140 (0.7)	127 (0.9)
1986	164 (1.2)	135 (1.8)	142 (0.9)	118 (1.4)
1987	155 (2.4)	135 (3.6)	133 (1.8)	116 (3.0)
1988	163 (4.6)	154 (5.5)	138 (4.0)	133 (5.0)
1989	176 (6.5)	150 (8.2)	151 (6.0)	131 (7.9)
K^e	804 (13.3)		1017 (14.5)	
MSY	106 (13.3)		120 (14.5)	

YEAR	M = 0.4yr ⁻¹		M = 0.5yr ⁻¹	
	2+	EXPLOITABLE	2+	EXPLOITABLE
1978	149 (0.8)	132 (0.9)	180 (1.1)	155 (1.3)
1979	160 (0.9)	138 (1.1)	193 (1.2)	163 (1.4)
1980	164 (1.0)	132 (1.3)	197 (1.3)	157 (1.6)
1981	151 (1.1)	126 (1.4)	185 (1.5)	152 (1.8)
1982	147 (1.2)	133 (1.4)	183 (1.6)	161 (1.7)
1983	159 (1.2)	143 (1.3)	197 (1.5)	172 (1.7)
1984	175 (1.1)	156 (1.3)	214 (1.4)	186 (1.6)
1985	190 (1.1)	166 (1.4)	231 (1.4)	197 (1.9)
1986	193 (1.6)	157 (2.4)	235 (2.3)	189 (3.1)
1987	185 (3.0)	159 (4.3)	230 (3.9)	195 (5.2)
1988	197 (5.4)	182 (6.1)	247 (6.2)	221 (6.8)
1989	211 (7.0)	174 (8.5)	260 (7.5)	209 (8.8)
K^e	667 (12.6)		585 (12.4)	
MSY	99 (12.6)		101 (12.4)	

(Table 10.4 Continued)

YEAR	TUNED OVER 1983-1989		SURVEY DATA	
	2+	EXPLOITABLE	2+	EXPLOITABLE
1978	127 (0.6)	112 (0.7)	127 (0.6)	101 (0.8)
1979	137 (0.7)	118 (0.9)	137 (0.7)	108 (1.0)
1980	140 (0.7)	112 (1.0)	141 (0.8)	104 (1.2)
1981	128 (0.8)	104 (1.1)	129 (1.0)	97 (1.4)
1982	122 (0.9)	110 (1.1)	124 (1.1)	103 (1.4)
1983	133 (1.0)	119 (1.1)	137 (1.2)	115 (1.5)
1984	148 (0.9)	131 (1.1)	158 (1.2)	132 (1.5)
1985	161 (0.8)	139 (1.1)	183 (2.4)	154 (4.0)
1986	163 (1.1)	131 (1.7)	211 (7.6)	174 (10.4)
1987	154 (2.1)	129 (3.1)	253 (15.0)	230 (18.7)
1988	160 (4.1)	147 (5.1)	367 (24.5)	362 (26.1)
1989	173 (6.3)	145 (8.0)	549 (28.1)	526 (28.4)
K ^e	797 (13.3)		970 (37.9)	
MSY	107 (13.3)		140 (37.9)	

Table 10.5 : Recruitment ($N_{y,1}$) estimates in millions, with their estimated C.V.s (expressed as a percentage) in parenthesis obtained from the 'M-independent-of-age' *ad hoc* tuned VPA.

YEAR	"BASE CASE"	RECRUITMENT	
		M = 0.2yr ⁻¹	M = 0.4yr ⁻¹
1978	518 (0.4)	411 (0.2)	669 (0.7)
1979	421 (0.5)	332 (0.3)	550 (1.0)
1980	449 (0.4)	353 (0.2)	588 (0.7)
1981	578 (0.3)	467 (0.1)	735 (0.5)
1982	532 (0.3)	433 (0.2)	673 (0.6)
1983	474 (0.5)	363 (0.3)	636 (0.9)
1984	512 (1.5)	395 (1.0)	683 (2.4)
1985	405 (3.5)	306 (2.5)	551 (4.9)
1986	469 (7.2)	353 (5.7)	643 (9.1)
1987	596 (5.3)	454 (4.5)	804 (6.4)
1988	208 (22.8)	161 (20.6)	277 (25.2)
1989	25 (129.6)	19 (126.9)	33 (132.5)

YEAR	M = 0.5yr ⁻¹	RECRUITMENT	
		TUNED 83-89	TUNED SURVEY
1978	890 (1.2)	518 (0.4)	519 (0.5)
1979	749 (1.6)	421 (0.5)	423 (0.7)
1980	800 (1.3)	449 (0.4)	454 (0.6)
1981	966 (0.9)	578 (0.3)	586 (0.5)
1982	884 (1.1)	532 (0.3)	550 (0.8)
1983	887 (1.6)	473 (0.5)	521 (1.8)
1984	945 (3.6)	510 (1.4)	610 (13.0)
1985	780 (6.7)	403 (2.8)	590 (25.5)
1986	916 (11.3)	463 (6.3)	874 (47.1)
1987	1126 (7.6)	590 (6.2)	1382 (42.8)
1988	382 (28.1)	216 (27.2)	811 (48.9)
1989	45 (136.6)	25 (167.0)	183 (138.2)

Table 10.6 : Estimates of the standard deviation (σ_a) of the residuals in the $F_{y,a}$ versus E_y regressions obtained from the 'M-independent-of-age' *ad hoc* tuned VPA.

VARIANT	AGE					
	1	2	3	4	5	6
"Base case"	0.983	0.271	0.102	0.203	0.184	0.183
M = 0.2yr ⁻¹	0.969	0.259	0.099	0.197	0.179	0.178
M = 0.4yr ⁻¹	0.996	0.285	0.107	0.209	0.190	0.188
M = 0.5yr ⁻¹	1.014	0.302	0.114	0.215	0.196	0.194
Tuned 83-89	1.131	0.258	0.118	0.180	0.149	0.179
Tuned survey	0.985	0.440	0.448	0.570	0.480	0.460

Table 10.7 : Estimates of age-specific selectivity (S_a) obtained from the 'M-independent-of-age' *ad hoc* tuned VPA.

VARIANT	AGE						
	1	2	3	4	5	6	7
"Base case"	0.074	0.766	0.890	0.748	0.953	1.050	1.000
M = 0.2yr ⁻¹	0.080	0.786	0.908	0.757	0.954	1.049	1.000
M = 0.4yr ⁻¹	0.068	0.743	0.870	0.737	0.952	1.051	1.000
M = 0.5yr ⁻¹	0.062	0.717	0.847	0.726	0.951	1.052	1.000
Tuned 83-89	0.072	0.705	0.876	0.752	0.941	1.063	1.000
Tuned survey	0.034	0.560	0.837	0.709	0.942	1.062	1.000

Table 10.8 : Estimates of various parameters and management-related quantities obtained from a number of applications of the 'M-cannibalistic' *ad hoc* tuned VPA model-estimation procedure. Figures in parenthesis are percentage C.V.s. B^e , B^{2+} and PRED are the averages of the exploitable biomass, 2^+ biomass and hake consumed by hake over the period 1987 to 1989. The quantity $-\ln L$ is the value of the negative of the logarithm of the likelihood function (see Equation 10.20). K^e is the exploitable component of the average pristine biomass level.

Γ	M_D	R_w	σ_w	$-\ln L$	K^e '000 t	B^e '000 t	B^{2+} '000 t	PRED '000 t
0	0.2	-	-	-	1017	126.9	140.1	0
0	0.3	-	-	-	804	146.2	164.7	0
0.005	0.2	0.212	0.329	4.53	511	126.9	140.6	1.1
0.005	0.3	0.218	0.326	4.54	450	146.2	164.9	1.9
0.01	0.2	0.216	0.327	4.53	373	126.9	140.8	2.6
0.01	0.3	0.219	0.326	4.55	341	146.2	165.2	3.9
0.02	0.2	0.220 (84)	0.326 (45)	4.54	259	126.9	141.2	5.9
0.02	0.3	0.226 (78)	0.324 (45)	4.58	242	146.2	166.1	9.7
0.05	0.2	0.239 (81)	0.323 (45)	4.76	147	126.9	144.2	28.5
0.05	0.3	0.251 (79)	0.324 (46)	5.06	145	146.2	172.4	58.9
0.075	0.2	0.281 (67)	0.314 (49)	5.47	113	126.9	155.7	136.3
0.075	0.3	0.295 (60)	0.311 (51)	5.42	115	146.3	195.4	262.6
0.1	0.2	0.298	0.312	5.27	93.7	126.8	173.5	301.1
0.1	0.3	0.308	0.315	4.84	92.6	146.3	235.5	666.4
0.25	0.2	0.316	0.326	4.11	45.1	126.8	332.2	3308
0.25	0.3	0.316	0.327	4.11	43.6	147.3	448.6	5583
0.5	0.2	0.316	0.328	4.42	24.3	129.0	649.9	18572
0.5	0.3	0.315	0.331	4.53	23.2	147.7	844.0	21149

Table 10.9 : Estimates of fishing mortality ($F_{y,a}$) matrices obtained from the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure. Units are yr^{-1} .

(a) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.016	0.763	0.650	0.404	0.523	0.617	0.568
1979	0.017	0.411	0.568	0.480	0.818	0.757	0.787
1980	0.007	0.385	0.673	0.735	0.828	0.826	0.827
1981	0.037	0.516	0.633	0.726	0.685	0.892	0.782
1982	0.064	0.486	0.449	0.425	0.625	0.643	0.634
1983	0.009	0.286	0.470	0.420	0.552	0.594	0.573
1984	0.007	0.239	0.546	0.530	0.537	0.559	0.548
1985	0.005	0.238	0.522	0.536	0.663	0.783	0.721
1986	0.002	0.201	0.635	0.561	0.949	1.077	1.011
1987	0.002	0.265	0.698	0.602	0.651	0.635	0.643
1988	0.009	0.278	0.445	0.320	0.508	0.699	0.596
1989	0.008	0.270	0.447	0.401	0.517	0.571	0.543

(b) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.028	0.937	0.764	0.466	0.599	0.701	0.648
1979	0.028	0.532	0.669	0.557	0.922	0.855	0.888
1980	0.011	0.513	0.791	0.846	0.933	0.937	0.935
1981	0.060	0.686	0.766	0.840	0.792	1.019	0.898
1982	0.118	0.642	0.550	0.505	0.726	0.747	0.736
1983	0.017	0.391	0.566	0.495	0.641	0.688	0.664
1984	0.011	0.338	0.655	0.612	0.613	0.638	0.626
1985	0.008	0.335	0.633	0.616	0.746	0.878	0.810
1986	0.004	0.282	0.762	0.654	1.080	1.216	1.146
1987	0.003	0.374	0.850	0.708	0.749	0.736	0.742
1988	0.014	0.394	0.547	0.381	0.591	0.804	0.690
1989	0.012	0.366	0.537	0.467	0.592	0.652	0.621

(Table 10.9 Continued)

(c) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.036	0.892	0.672	0.407	0.523	0.617	0.568
1979	0.040	0.527	0.597	0.485	0.819	0.758	0.788
1980	0.016	0.505	0.710	0.742	0.829	0.827	0.828
1981	0.081	0.661	0.668	0.733	0.686	0.893	0.783
1982	0.156	0.622	0.476	0.430	0.627	0.644	0.635
1983	0.024	0.388	0.501	0.426	0.554	0.595	0.574
1984	0.017	0.343	0.587	0.537	0.539	0.559	0.549
1985	0.012	0.350	0.566	0.545	0.665	0.784	0.722
1986	0.005	0.288	0.681	0.569	0.951	1.078	1.013
1987	0.005	0.367	0.741	0.609	0.652	0.636	0.644
1988	0.021	0.397	0.479	0.324	0.509	0.700	0.597
1989	0.018	0.363	0.475	0.406	0.518	0.571	0.544

(d) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.051	1.040	0.780	0.468	0.599	0.702	0.648
1979	0.060	0.632	0.691	0.555	0.923	0.855	0.888
1980	0.024	0.619	0.819	0.852	0.934	0.938	0.936
1981	0.118	0.811	0.793	0.846	0.793	1.019	0.899
1982	0.231	0.758	0.571	0.508	0.727	0.747	0.737
1983	0.039	0.481	0.590	0.499	0.642	0.688	0.665
1984	0.027	0.432	0.685	0.617	0.614	0.639	0.626
1985	0.019	0.439	0.668	0.622	0.748	0.879	0.811
1986	0.008	0.363	0.798	0.660	1.082	1.216	1.147
1987	0.008	0.465	0.882	0.713	0.750	0.737	0.743
1988	0.034	0.502	0.573	0.385	0.592	0.805	0.690
1989	0.027	0.448	0.559	0.470	0.593	0.652	0.622

(Table 10.9 Continued)

(e) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.054	0.942	0.679	0.407	0.523	0.617	0.568
1979	0.069	0.577	0.606	0.486	0.819	0.758	0.788
1980	0.028	0.559	0.722	0.744	0.830	0.827	0.828
1981	0.134	0.725	0.679	0.735	0.687	0.893	0.783
1982	0.254	0.682	0.484	0.431	0.627	0.644	0.636
1983	0.045	0.434	0.510	0.427	0.554	0.595	0.574
1984	0.034	0.392	0.599	0.539	0.539	0.560	0.549
1985	0.025	0.405	0.580	0.547	0.666	0.784	0.723
1986	0.010	0.330	0.695	0.571	0.952	1.078	1.013
1987	0.009	0.412	0.754	0.611	0.652	0.636	0.644
1988	0.042	0.453	0.490	0.326	0.510	0.700	0.597
1989	0.032	0.405	0.483	0.407	0.519	0.571	0.544

(f) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.067	1.073	0.784	0.468	0.599	0.702	0.649
1979	0.088	0.668	0.696	0.556	0.923	0.855	0.888
1980	0.035	0.659	0.826	0.853	0.934	0.938	0.936
1981	0.167	0.857	0.800	0.847	0.793	1.019	0.899
1982	0.321	0.799	0.576	0.508	0.727	0.748	0.737
1983	0.059	0.514	0.596	0.499	0.642	0.688	0.665
1984	0.045	0.468	0.693	0.618	0.614	0.639	0.626
1985	0.033	0.480	0.677	0.623	0.748	0.879	0.811
1986	0.013	0.395	0.807	0.661	1.082	1.216	1.147
1987	0.012	0.498	0.890	0.714	0.750	0.737	0.743
1988	0.056	0.542	0.580	0.385	0.592	0.805	0.690
1989	0.042	0.479	0.564	0.471	0.593	0.652	0.622

Table 10.10 : Estimates of numbers-at-age ($N_{y,a}$) matrices obtained from the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure. Units are millions.

(a) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}('000 \text{ tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	3638	663	139	50	17	4	1
1979	3932	619	155	50	24	7	2
1980	4063	523	166	57	22	8	3
1981	4692	546	134	54	20	7	3
1982	4471	612	127	46	19	7	2
1983	5447	567	153	53	22	7	3
1984	6448	730	159	61	25	9	3
1985	4940	867	194	58	26	11	4
1986	5094	665	204	70	24	10	4
1987	7045	688	171	66	28	7	2
1988	2369	952	194	55	26	11	3
1989	253	318	237	78	29	12	4

(b) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}('000 \text{ tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	1790	532	117	42	14	4	1
1979	2285	451	129	43	22	6	2
1980	2685	369	139	50	20	7	2
1981	2690	395	110	47	17	6	2
1982	2164	443	102	38	16	6	2
1983	2919	384	124	44	19	6	2
1984	4192	473	130	52	22	8	3
1985	3214	561	156	50	23	10	3
1986	3345	431	168	59	21	9	3
1987	4568	451	141	56	25	6	2
1988	1577	616	154	45	22	10	2
1989	158	210	190	65	25	10	3

(Table 10.10 Continued)

(c) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	1214	539	132	49	17	4	1
1979	1320	431	143	49	24	7	2
1980	1522	350	153	57	22	8	3
1981	1686	384	123	53	20	7	3
1982	1432	433	117	45	19	7	2
1983	1687	364	139	52	22	7	3
1984	2215	431	143	60	25	9	3
1985	1983	488	171	56	26	11	4
1986	2121	385	184	68	24	10	4
1987	2395	430	157	65	28	7	2
1988	912	565	173	53	26	11	3
1989	103	191	211	76	28	12	4

(d) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	762	463	114	42	14	4	1
1979	781	350	123	42	22	6	2
1980	895	278	132	49	20	7	2
1981	1034	310	104	47	17	6	2
1982	877	350	97	38	16	6	2
1983	922	282	117	44	19	6	2
1984	1178	327	121	52	22	8	3
1985	1037	371	144	49	23	10	3
1986	1123	290	156	59	21	9	3
1987	1265	327	133	56	25	6	2
1988	479	430	143	44	22	10	2
1989	57	146	176	64	25	10	3

(Table 10.10 Continued)

(e) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}$ ('000 tons) $^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	654	499	130	49	17	4	1
1979	583	375	140	49	24	7	2
1980	641	299	149	56	22	8	3
1981	788	334	120	53	20	7	3
1982	704	378	114	45	19	7	2
1983	683	305	135	52	22	7	3
1984	782	351	139	60	25	9	3
1985	646	387	165	56	26	11	4
1986	731	309	178	68	24	10	4
1987	885	360	153	65	28	7	2
1988	320	461	167	53	26	11	3
1989	38	155	204	75	28	12	4

(f) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}$ ('000 tons) $^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	493	443	113	42	14	4	1
1979	430	321	121	42	22	6	2
1980	470	253	130	49	20	7	2
1981	596	285	103	46	17	6	2
1982	536	323	95	38	16	6	2
1983	483	254	115	44	19	6	2
1984	552	289	119	52	22	8	3
1985	448	321	141	49	23	10	3
1986	506	252	154	58	21	9	3
1987	619	294	132	56	25	6	2
1988	225	381	141	44	22	10	2
1989	27	128	173	64	25	10	3

Table 10.11 : Estimates of natural mortality-at-age ($M_{y,a}$) matrices obtained from the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure. Units are yr^{-1} .

(a) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	1.755	0.694	0.373	0.312	0.302	0.300	0.300
1979	2.000	0.908	0.421	0.321	0.303	0.300	0.300
1980	2.000	0.980	0.443	0.326	0.304	0.301	0.300
1981	2.000	0.939	0.437	0.325	0.304	0.301	0.300
1982	2.000	0.897	0.427	0.323	0.304	0.301	0.300
1983	2.000	0.986	0.449	0.328	0.305	0.301	0.300
1984	2.000	1.086	0.468	0.331	0.305	0.301	0.300
1985	2.000	1.207	0.501	0.338	0.307	0.301	0.300
1986	2.000	1.157	0.488	0.335	0.306	0.301	0.300
1987	2.000	1.001	0.442	0.325	0.304	0.301	0.300
1988	2.000	1.114	0.471	0.331	0.305	0.301	0.300
1989	2.000	1.286	0.514	0.340	0.307	0.301	0.300

(b) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.075\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	1.350	0.479	0.248	0.208	0.201	0.200	0.200
1979	1.794	0.644	0.281	0.213	0.202	0.200	0.200
1980	1.906	0.701	0.297	0.217	0.203	0.200	0.200
1981	1.745	0.667	0.292	0.216	0.203	0.200	0.200
1982	1.611	0.627	0.283	0.214	0.202	0.200	0.200
1983	1.804	0.691	0.298	0.217	0.203	0.200	0.200
1984	2.000	0.768	0.311	0.219	0.203	0.200	0.200
1985	2.000	0.872	0.336	0.224	0.204	0.201	0.200
1986	2.000	0.838	0.328	0.223	0.204	0.201	0.200
1987	2.000	0.700	0.293	0.216	0.202	0.200	0.200
1988	2.000	0.782	0.312	0.219	0.203	0.200	0.200
1989	2.000	0.911	0.342	0.225	0.204	0.201	0.200

(Table 10.11 Continued)

(c) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	1.001	0.432	0.319	0.303	0.300	0.300	0.300
1979	1.286	0.510	0.333	0.305	0.301	0.300	0.300
1980	1.362	0.540	0.339	0.306	0.301	0.300	0.300
1981	1.279	0.527	0.338	0.306	0.301	0.300	0.300
1982	1.215	0.512	0.335	0.305	0.301	0.300	0.300
1983	1.340	0.545	0.341	0.306	0.301	0.300	0.300
1984	1.496	0.579	0.346	0.307	0.301	0.300	0.300
1985	1.628	0.627	0.355	0.309	0.301	0.300	0.300
1986	1.590	0.607	0.352	0.308	0.301	0.300	0.300
1987	1.440	0.543	0.338	0.306	0.301	0.300	0.300
1988	1.545	0.587	0.346	0.307	0.301	0.300	0.300
1989	1.787	0.652	0.359	0.309	0.301	0.300	0.300

(d) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.728	0.287	0.211	0.201	0.200	0.200	0.200
1979	0.971	0.342	0.220	0.203	0.200	0.200	0.200
1980	1.037	0.364	0.224	0.203	0.200	0.200	0.200
1981	0.965	0.354	0.223	0.203	0.200	0.200	0.200
1982	0.901	0.340	0.220	0.203	0.200	0.200	0.200
1983	0.997	0.363	0.224	0.203	0.200	0.200	0.200
1984	1.128	0.387	0.227	0.204	0.201	0.200	0.200
1985	1.256	0.425	0.234	0.205	0.201	0.200	0.200
1986	1.225	0.413	0.232	0.204	0.201	0.200	0.200
1987	1.071	0.361	0.223	0.203	0.200	0.200	0.200
1988	1.154	0.391	0.227	0.204	0.201	0.200	0.200
1989	1.349	0.436	0.235	0.205	0.201	0.200	0.200

(Table 10.11 Continued)

(e) $M_b = 0.3\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.504	0.328	0.303	0.300	0.300	0.300	0.300
1979	0.597	0.346	0.306	0.301	0.300	0.300	0.300
1980	0.624	0.354	0.307	0.301	0.300	0.300	0.300
1981	0.601	0.351	0.307	0.301	0.300	0.300	0.300
1982	0.581	0.347	0.306	0.301	0.300	0.300	0.300
1983	0.620	0.355	0.307	0.301	0.300	0.300	0.300
1984	0.669	0.363	0.308	0.301	0.300	0.300	0.300
1985	0.715	0.374	0.310	0.301	0.300	0.300	0.300
1986	0.698	0.369	0.309	0.301	0.300	0.300	0.300
1987	0.643	0.354	0.307	0.301	0.300	0.300	0.300
1988	0.683	0.364	0.308	0.301	0.300	0.300	0.300
1989	0.759	0.379	0.310	0.301	0.300	0.300	0.300

(f) $M_b = 0.2\text{yr}^{-1}$; $\Gamma = 0.02\text{yr}^{-1}(\text{'000 tons})^{-1}$

YEAR	AGE						
	1	2	3	4	5	6	7
1978	0.361	0.221	0.202	0.200	0.200	0.200	0.200
1979	0.442	0.235	0.204	0.200	0.200	0.200	0.200
1980	0.465	0.241	0.205	0.201	0.200	0.200	0.200
1981	0.444	0.239	0.205	0.201	0.200	0.200	0.200
1982	0.424	0.235	0.204	0.201	0.200	0.200	0.200
1983	0.455	0.241	0.205	0.201	0.200	0.200	0.200
1984	0.496	0.247	0.206	0.201	0.200	0.200	0.200
1985	0.541	0.257	0.207	0.201	0.200	0.200	0.200
1986	0.529	0.254	0.207	0.201	0.200	0.200	0.200
1987	0.473	0.240	0.205	0.201	0.200	0.200	0.200
1988	0.505	0.248	0.206	0.201	0.200	0.200	0.200
1989	0.568	0.259	0.207	0.201	0.200	0.200	0.200

Table 10.13 : Biomass estimates in '000 tons with their corresponding C.V.s (expressed as a percentage) in parenthesis obtained from the applications of the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure. K^e is the estimate of the exploitable component of the average pre-exploitation size. Estimates of MSY are only provided when the estimate of F_{MSY} is less than $5yr^{-1}$.

YEAR	$\Gamma = 0.075$; $M_b = 0.3$		$\Gamma = 0.075$; $M_b = 0.2$	
	2+ BIOMASS	EXP BIOMASS	2+ BIOMASS	EXP BIOMASS
1978	141.0 (5.5)	105.3 (3.8)	118.2 (3.7)	95.8 (3.0)
1979	158.2 (7.9)	116.5 (1.9)	131.0 (5.2)	104.9 (1.3)
1980	161.1 (7.6)	111.8 (1.4)	134.1 (4.9)	99.9 (1.0)
1981	146.7 (7.4)	103.0 (1.8)	120.7 (4.8)	90.6 (1.4)
1982	142.2 (8.2)	108.8 (1.6)	114.1 (5.3)	94.9 (1.3)
1983	156.8 (9.3)	121.7 (0.5)	125.2 (5.9)	106.0 (0.4)
1984	180.2 (11.5)	136.7 (0.9)	143.5 (7.1)	119.6 (0.6)
1985	203.0 (13.2)	147.4 (1.7)	161.3 (8.4)	129.4 (1.2)
1986	196.9 (11.5)	136.0 (3.4)	159.5 (7.2)	119.1 (2.4)
1987	182.8 (13.2)	134.1 (7.6)	146.5 (8.0)	115.8 (6.1)
1988	204.1 (21.8)	156.4 (13.8)	157.8 (14.1)	134.3 (11.9)
1989	199.3 (19.8)	148.4 (18.9)	162.7 (16.8)	130.6 (18.7)
K^e	106.8 (17.6)		107.1 (29.6)	

YEAR	$\Gamma = 0.05$; $M_b = 0.3$		$\Gamma = 0.05$; $M_b = 0.2$	
	2+ BIOMASS	EXP BIOMASS	2+ BIOMASS	EXP BIOMASS
1978	131.1 (2.0)	111.4 (1.9)	112.8 (1.1)	100.0 (1.3)
1979	142.7 (2.7)	119.4 (0.9)	122.5 (1.5)	106.7 (0.6)
1980	145.8 (2.6)	113.8 (0.7)	125.9 (1.5)	101.0 (0.5)
1981	132.9 (2.6)	105.5 (0.9)	113.4 (1.5)	92.3 (0.6)
1982	127.4 (3.0)	111.2 (0.8)	106.5 (1.6)	96.7 (0.6)
1983	139.1 (3.2)	122.1 (0.2)	116.2 (1.7)	106.4 (0.2)
1984	156.0 (3.8)	135.3 (0.5)	131.4 (2.0)	118.6 (0.2)
1985	171.7 (4.6)	145.0 (1.3)	145.3 (2.5)	127.7 (0.8)
1986	172.2 (4.5)	135.4 (3.2)	146.7 (2.6)	118.4 (2.3)
1987	161.9 (6.5)	134.4 (7.0)	136.2 (4.4)	116.0 (5.7)
1988	173.0 (11.9)	154.9 (12.4)	142.6 (9.0)	133.5 (11.0)
1989	182.1 (16.2)	149.5 (19.1)	153.7 (14.6)	131.2 (18.8)
K^e	136.9 (45.6)		138.2 (69.2)	

(Table 10.13 Continued)

YEAR	$\Gamma = 0.02$; $M_b = 0.3$		$\Gamma = 0.02$; $M_b = 0.2$	
	2+ BIOMASS	EXP BIOMASS	2+ BIOMASS	EXP BIOMASS
1978	128.0 (0.3)	114.3 (0.6)	111.3 (0.2)	101.7 (0.5)
1979	138.1 (0.4)	120.6 (0.4)	120.3 (0.3)	107.4 (0.3)
1980	141.3 (0.4)	114.5 (0.3)	123.7 (0.3)	101.4 (0.3)
1981	128.8 (0.4)	106.6 (0.3)	111.4 (0.3)	92.9 (0.3)
1982	123.0 (0.5)	112.4 (0.2)	104.4 (0.3)	97.4 (0.3)
1983	134.1 (0.6)	122.4 (0.0)	113.9 (0.4)	106.6 (0.1)
1984	149.4 (0.8)	134.8 (0.4)	128.3 (0.5)	118.3 (0.2)
1985	163.4 (1.3)	143.9 (1.2)	141.2 (0.8)	126.9 (0.7)
1986	165.4 (2.3)	134.9 (3.1)	143.4 (1.6)	118.0 (2.3)
1987	156.2 (4.9)	134.6 (6.7)	133.6 (3.7)	116.2 (5.6)
1988	164.7 (9.4)	154.3 (11.8)	138.7 (8.1)	133.2 (10.7)
1989	177.3 (15.1)	149.8 (19.2)	151.4 (14.1)	131.4 (18.9)
K^e	225.2 (58.2)		240.5 (78.7)	
MSY	95.5 (6.8)		88.3 (3.7)	

Table 10.14 : Recruitment ($N_{y,1}$) estimates in millions and estimates of hake consumption by hake in '000 tons with their corresponding C.V.s (expressed as a percentage) in parenthesis obtained from the applications of the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure.

YEAR	$\Gamma = 0.075$; $M_b = 0.3$		$\Gamma = 0.075$; $M_b = 0.2$	
	RECRUITMENT	PREDATION	RECRUITMENT	PREDATION
1978	3638 (46)	180.7 (56)	1790 (42)	85.0 (55)
1979	3932 (36)	246.7 (48)	2285 (43)	128.0 (53)
1980	4063 (33)	265.7 (45)	2685 (38)	148.9 (48)
1981	4692 (33)	271.9 (45)	2690 (39)	139.8 (49)
1982	4471 (38)	246.6 (49)	2164 (41)	111.9 (51)
1983	5447 (42)	336.9 (54)	2919 (48)	157.5 (56)
1984	6448 (43)	460.6 (56)	4192 (41)	241.5 (53)
1985	4940 (42)	419.2 (57)	3214 (39)	224.6 (52)
1986	5094 (44)	406.1 (60)	3345 (38)	219.5 (51)
1987	7045 (54)	481.3 (75)	4568 (46)	248.8 (60)
1988	2369 (50)	240.3 (78)	1577 (48)	125.3 (67)
1989	253 (49)	66.0 (83)	158 (46)	34.8 (77)

YEAR	$\Gamma = 0.05$; $M_b = 0.3$		$\Gamma = 0.05$; $M_b = 0.2$	
	RECRUITMENT	PREDATION	RECRUITMENT	PREDATION
1978	1214 (33)	40.6 (58)	762 (24)	21.7 (53)
1979	1320 (39)	56.2 (59)	781 (30)	29.7 (54)
1980	1522 (38)	65.3 (56)	895 (30)	34.6 (52)
1981	1686 (34)	66.2 (53)	1034 (26)	35.6 (49)
1982	1437 (33)	55.3 (54)	877 (24)	28.8 (49)
1983	1687 (42)	71.3 (59)	922 (31)	34.8 (53)
1984	2215 (48)	100.5 (63)	1178 (37)	48.6 (56)
1985	1983 (49)	101.4 (63)	1037 (38)	50.0 (56)
1986	2121 (50)	101.3 (65)	1123 (37)	50.0 (55)
1987	2395 (58)	103.1 (76)	1265 (40)	48.9 (61)
1988	912 (64)	57.3 (84)	479 (48)	27.9 (68)
1989	103 (52)	16.6 (91)	57 (48)	8.6 (79)

(Table 10.14 Continued)

YEAR	$\Gamma = 0.02$; $M_b = 0.3$		$\Gamma = 0.02$; $M_b = 0.2$	
	RECRUITMENT	PREDATION	RECRUITMENT	PREDATION
1978	654 (9)	7.9 (43)	493 (7)	5.0 (46)
1979	583 (11)	10.1 (43)	430 (10)	6.5 (45)
1980	641 (12)	11.6 (41)	470 (10)	7.5 (43)
1981	788 (10)	12.5 (39)	596 (9)	8.1 (42)
1982	704 (9)	10.5 (40)	536 (8)	6.6 (42)
1983	683 (12)	12.3 (41)	483 (10)	7.5 (43)
1984	782 (14)	15.9 (42)	552 (12)	9.8 (44)
1985	646 (15)	15.9 (42)	448 (13)	9.9 (44)
1986	731 (16)	16.1 (42)	506 (14)	10.0 (44)
1987	885 (18)	16.5 (46)	619 (15)	9.9 (47)
1988	320 (24)	9.6 (53)	225 (21)	5.9 (53)
1989	38 (28)	3.1 (64)	27 (26)	1.9 (65)

Table 10.15 : Model parameter and variable estimates, and estimated C.V.s (expressed as a percentage) obtained from five applications of the Butterworth-Andrew observation error estimator to the data for Cape hake off the South African west coast. Biomass units are '000 tons.

(a) "Base case" application

PARAMETER	ESTIMATE	C.V.	95% CONFIDENCE INTERVAL	
r	0.379	12.5	0.294	0.481
q x 10 ⁻³	14.075	11.6	11.057	17.543
K	1438.	9.9	1190.	1756.
Ω	0.522	13.1	0.409	0.667
B _{MSY}	719.	9.9	595.	878.
B ₈₉	604.	9.0	508.	724.
B ₈₉ /K	0.420	7.0	0.368	0.484
B ₈₉ /B _{MSY}	0.841	7.0	0.735	0.969
MSY	136.3	2.7	128.5	143.1
-2lnL	-25.94			
σ	0.12			

(b) B₁₉₁₇/K estimated

PARAMETER	ESTIMATE	C.V.	95% CONFIDENCE INTERVAL	
r	0.379	0.1	0.378	0.379
q x 10 ⁻³	14.017	2.8	13.180	14.753
K	1439.	0.2	1436.	1444.
Ω	0.517	7.2	0.442	0.591
B ₁₉₁₇ /K	1.120	5.1	1.104	1.419
B _{MSY}	720.	0.2	718.	722.
B ₈₉	611.	5.8	540.	686.
B ₈₉ /K	0.425	5.6	0.376	0.475
B ₈₉ /B _{MSY}	0.849	5.6	0.752	0.949
MSY	136.3	0.1	136.1	136.6
-2lnL	-25.93			
σ	0.13			

(Table 10.15 Continued)

(c) Fox form for the surplus production form

PARAMETER	ESTIMATE	C.V.	95% CONFIDENCE INTERVAL	
r	1.310	6.6	1.138	1.483
q x 10 ⁻³	12.097	7.6	10.377	13.981
K	1927.	5.5	1738.	2165.
Ω	0.438	8.6	0.371	0.518
B _{MSY}	709.	5.5	639.	797.
B ₈₉	681.	5.5	610.	763.
B ₈₉ /K	0.354	4.8	0.321	0.389
B ₈₉ /B _{MSY}	0.961	4.8	0.872	1.057
MSY	122.8	1.9	118.0	127.2
-2lnL	-32.40			
σ	0.10			

(d) Survey data ignored

PARAMETER	ESTIMATE	C.V.	95% CONFIDENCE INTERVAL	
r	0.379	13.2	0.282	0.488
q x 10 ⁻³	14.205	12.2	10.857	17.857
K	1437.	10.6	1177.	1807.
B _{MSY}	718.	10.6	588.	903.
B ₈₉	585.	9.3	487.	712.
B ₈₉ /K	0.407	7.5	0.352	0.475
B ₈₉ /B _{MSY}	0.815	7.5	0.703	0.951
MSY	136.3	2.9	127.3	143.4
-2lnL	-55.34			
σ	0.12			

(Table 10.15 Continued)

(e) CPUE ignored

PARAMETER	ESTIMATE	C.V.	95% CONFIDENCE INTERVAL	
r	1.354	8.0	0.938	1.563
K	489.	10.7	430.	679.
Ω	0.766	> 1000	-	-
B_{MSY}	245.	10.7	215.	339.
B_{89}	415.	11.5	366.	550.
B_{89}/K	0.848	8.6	0.805	0.852
B_{89}/B_{MSY}	1.696	8.6	1.609	1.704
MSY	165.7	1.4	158.6	167.8
$-\ln L$	22.84			

MANAGEMENT PROCEDURES FOR CAPE HAKE AND BALEEN WHALE RESOURCES

by

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Part II

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CHAPTER 11 - SPECIFICATION OF PERFORMANCE MEASURES, OPERATING MODELS AND TRIALS

The summary statistics used to measure the performance of a management procedure for a simulation trial are:

- a) the mean (non-discounted) total catch over a 20-year management period,
- b) the standard deviation of the total catch distribution,
- c) the mean final depletion at the end of a 20-year management period,
- d) the standard deviation of the final depletion distribution,
- e) the mean of the lowest depletions,
- f) the 5th smallest lowest depletion, and
- g) the mean and median of the distribution of the annual percentage change in catch.

The effects of cannibalism are incorporated using the predation model detailed in Chapter 10. Movement of fish across the stock area is modelled by diffusion. A model based on the assumption that the distribution of effort across the fishing grounds is governed by abundance and area-specific factors is developed. Age- and length-structured operating models are specified and their "base case" parameter values given. The data generated by these operating models are:

- 1) catches-by-mass,
- 2) CPUE,
- 3) survey biomass estimates,
- 4) recruitment indices,
- 5) catches-at-age, and
- 6) mass-at-age data.

A large number of robustness trials are specified in addition to a set of "base case" trials.

11.1 Introduction

In order to determine how well a management procedure is able to achieve the objectives specified by the management authority (see Section 1.3) within the context of a simulation study, it is necessary to be able to describe these objectives operationally. A difficulty that arises is that one procedure may perform better than another if performance is measured using one statistic, but would not if performance was measured by another (similar) statistic. For example, Punt and Butterworth (1990) found that the ranking of production model-estimation procedures changed if the median rather

than the mean of the distributions of the absolute value of the relative difference between estimates of management quantities and their true values was used as a performance measure. For this reason, the outcome of the selection process to be applied cannot be regarded as entirely objective, because it may depend to some extent on the particular performance statistics chosen.

The four measures which have been chosen in this study to quantify the management objectives listed in Section 1.3 are:

- a) Total catch (ΣC) = $\sum_{y=1990}^{2009} C_y$
- b) Depletion at the end of 20 years multiplied by 100 (B_{2009}^e/K^e)
- c) Lowest depletion during the 20 year period multiplied by 100:
 $(B/K)_{\min}^e = \text{MIN}(B_y^e/K^e : y = 1990, \dots, 2009)$
- d) Variability of catches:

$$v = 100 \frac{\sum_{y=1990}^{2009} |C_y - C_{y-1}|}{\sum_{y=1990}^{2009} C_y}$$

where B_y^e is the exploitable biomass during year y , and K^e is the exploitable component of the average pristine biomass.

The rationale for the choice of a 20 year management period was that this is sufficiently long to ensure that the initial conditions should not influence the final results excessively. Selection of a longer period would have resulted in impractical computer time requirements. The decision not to discount future catches was made primarily to be consistent with ICSEAF's specification of an operating model and performance measures for the hake resource off

northern Namibia, for which future catches were not discounted (ICSEAF 1990). In addition, with a rapidly increasing population size, southern Africa's demand for protein is likely to increase markedly in the next 20 years. It thus seems socially inappropriate to give greater weight to immediate compared to future catches, suggesting a discount rate near zero.

The selection of a 20 year management period is equivalent to setting the discount rate to infinity after 20 years. The final depletion statistic was chosen to ensure that no management procedure could perform "well" by achieving a high total catch through exterminating the resource in the final (20th) year, as would be appropriate if the discount rate was indeed infinite after the end of the 20 year period. The lowest depletion measure is of interest because, although resource extinction is unlikely to take place within 20 years, the resource could be driven to levels at which compensatory mechanisms (that might inhibit resource recovery from overexploitation) could start having an effect. The lowest depletion measure can be inspected to determine whether the population drops to such levels. The measure used to quantify inter-annual catch variability was selected because it can be interpreted easily (it is the average annual percentage change in the catch over the 20 year period).

In order to estimate the distributions of the four performance measures, an operating model [defined as "the best representation of the true situation which it is possible to construct by means of a probability model" (Linhart and Zucchini 1986)] was constructed to represent the "true" dynamics of the resource. A number R_{\max} of sets

of artificial data were then produced, based on the parameter values and the error structure of the operating model. (Computer time restrictions dictated a "base case" value of 100 for R_{\max} .) Each management procedure was then permitted to "manage" these R_{\max} simulated resources for 20 years. At the end of each simulation, the values of the four performance measures were computed [i.e. $\Sigma C(R,M)$, $(B_{2009}/K^e)(R,M)$, $(B/K)_{\min}^e(R,M)$ and $V(R,M)$ - note the dependence of the performance measures on the simulation number (R) as well as the management procedure (M)]. A total of 400 quantities were thus evaluated for each management procedure for each simulation trial. These 400 quantities provided a Monte-Carlo representation of the distributions of the four performance measures. It would have been extremely difficult to attempt to deal with these distributions in their totality, so that this study has only considered the following eight summary statistics, the formulae for which already incorporate the choice $R_{\max} = 100$ (the acronym for each quantity is given in parentheses):

$$[\text{Mean } \Sigma C] \quad - \quad \frac{1}{100} \sum_{R=1}^{100} \Sigma C(R,M)$$

$$[\text{S.d. } \Sigma C] \quad - \quad \frac{1}{99} \sum_{R=1}^{100} \left[\Sigma C(R,M) - \frac{1}{100} \sum_{R'=1}^{100} \Sigma C(R',M) \right]^2$$

$$[P_f] \quad - \quad \frac{1}{100} \sum_{R=1}^{100} (B_{2009}/K^e)(R,M)$$

$$[\text{S.d. } P_f] \quad - \quad \frac{1}{99} \sum_{R=1}^{100} \left[(B_{2009}/K^e)(R,M) - \frac{1}{100} \sum_{R'=1}^{100} (B_{2009}/K^e)(R',M) \right]^2$$

$$[\text{Mean } P_{\text{low}}] - \frac{1}{100} \sum_{R=1}^{100} (B/K)_{\min}^e(R,M)$$

[5%P_{low}] - the 5th lowest $(B/K)_{\min}^e(R,M)$ value

$$[\text{Mean } V] - \frac{1}{100} \sum_{R=1}^{100} V(R,M)$$

[Med V] - Median{ V(R,M) }

The median, as well as the mean of the V distribution was reported because, in a few of the simulations, the total catch was near-zero. This could result in an outlying high value of V(R,M) which could heavily influence the mean of the V distribution. However, the median of the distribution is not influenced by such outliers. In cases in which the operating model considers two species, the biomasses in the above formulae relate to both species combined.

In order to ease the comparisons in this thesis, the Mean ΣC and s.d. ΣC statistics have been expressed as percentages of the values of these statistics achieved by the PROK-SC-F1 procedure for the "base case" single-species trials (see Section 12.1).

In standard multiple-criteria decision making theory [e.g. Keeney and Raiffa (1976)], the best management procedure will be that which maximizes some function (the utility function) of the eight summary statistics chosen. However, a number of problems arise when attempting to define a utility function. In particular, there are difficulties associated with the time required to construct such a function (which may require considerable input from decision makers), and with the problems associated with making value judgements in a

context-free environment (Stewart *et al.* 1990). For these reasons, no attempt will be made in this thesis define a utility function.

The operating models considered here attempt to represent as much of the "real world" as possible (given the limitations of the available data). The factors determining the dynamics of the hake resource off the South African west coast and its exploitation pattern are very complex, and include biological, economic, and social components. Some of the main biological processes which determine the dynamics of this (and indeed any) fish population are:

- a) age-/length- and sex-structure,
- b) tissue growth,
- c) stock-recruitment relationship,
- d) competition,
- e) predation,
- f) movement across the stock area (spatial effects), and
- g) environmental effects on any or all of the above.

Processes related to the determination of the fishing mortality on the population are:

- a) the harvesting strategy (i.e. the algorithm used to set the TAC),
- b) age-/length- and sex-specific selectivity,
- c) fishing strategies (i.e. the distribution of effort across the fishing grounds),
- d) discarding, and

- e) technological interactions (e.g. co-operative fishing practices, by-catch in a fishery directed at another species, etc).

Two classes of operating model have been considered here. One class considers the length- and the age-structure of the resource, while the other considers only the age-structure. Before these models are developed, the models used to represent predation/cannibalism and spatial effects will be described. The two operating model classes will then be detailed, along with their "base case" parameter values. A description of the initial conditions for each of the R_{\max} simulations will then be given, followed by a description of how the data used by the management procedures will be generated. Finally, the robustness trials are detailed. When presenting the predation and spatial models and the two operating model classes, the two hake species have been assigned the indices $s=1$ (*M. capensis*) and $s=2$ (*M. paradoxus*). Summation over the s -index thus indicates summation over both hake species.

The operating models are described below in complete generality. However, in order to determine the consequences of the various effects investigated, only one effect at a time has been added to a "base case" operating model in most instances. This means that some of the specifications given below (e.g. natural mortality which is related to the spatial structure of the population) are not actually considered in the simulations.

A list of all of the 109 management procedures considered in this study is given in Table 11.1. Appendix 11.A contains the algorithm used to construct the acronyms used to represent these procedures.

11.2 The predation model

The two models developed to apply the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure (Equations 10.15, 10.18 and 10.19) have been used here. The "base case" parameter values selected for the predation model are as follows:

$$M_b = 0.3\text{yr}^{-1}$$

$$L_{50}^P = 57.6\text{cm}$$

$$\delta^P = 19.6$$

$$R_f = 0.251$$

$$\sigma_w = 0.324$$

$$\Gamma = 0.005\text{yr}^{-1}(\text{'000 tons})^{-1}.$$

The basal rate of natural mortality (M_b) was set at 0.3yr^{-1} because the estimates of exploitable biomass provided by the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure were closer to their production-model equivalents when M_b was fixed at this value (rather than 0.2yr^{-1}). The values of the parameters R_f and σ_w were obtained from the results of the application of the $M_b=0.3\text{yr}^{-1}$ "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure for the control value $\Gamma = 0.05$ (see Table 10.8). Γ has been taken to 0.005 instead of $0.05\text{yr}^{-1}(\text{'000 tons})^{-1}$ because the operating model biomasses are approximately 5-10 times larger than those provided by the VPA. Thus, to keep natural mortality at levels

comparable to the $\Gamma=0.05$ application of the "M-cannibalistic" VPA, Γ had to be reduced appropriately. Some of the robustness trials (see Section 11.7) consider the effects of different values for Γ .

11.3 Spatial effects

The spatial factors affecting a fishery can be divided into two types: those related to the distribution of the resource across the ocean and those related to the distribution of fishing effort across the fishing grounds. Note that, in general, the fishing grounds will not cover the complete area in which the resource is found.

The South African west coast has been divided into 12 subareas (two latitude zones x six depth zones). The boundaries and areas of each of these subareas are given in Table 11.2. The coast has been divided into northern and southern latitude zones (see Figure 11.1) because historic (prior to 1962) fishing effort was aimed almost exclusively at what has been defined as the southern latitude zone (A.I.L. Payne, SFRI, pers. commn).

11.3.1 Fish distribution and movement

Quantitative investigations of fish distribution patterns are very few (a notable exception to this is Caddy (1975)). There is, however, a large amount of qualitative information on fish distribution for hake. For example, as detailed in Chapter 2, *M. paradoxus* is found in deeper water than *M. capensis* and older/larger fish are generally found in deeper water than younger/smaller fish. As the only data available from which to determine the movement patterns that give

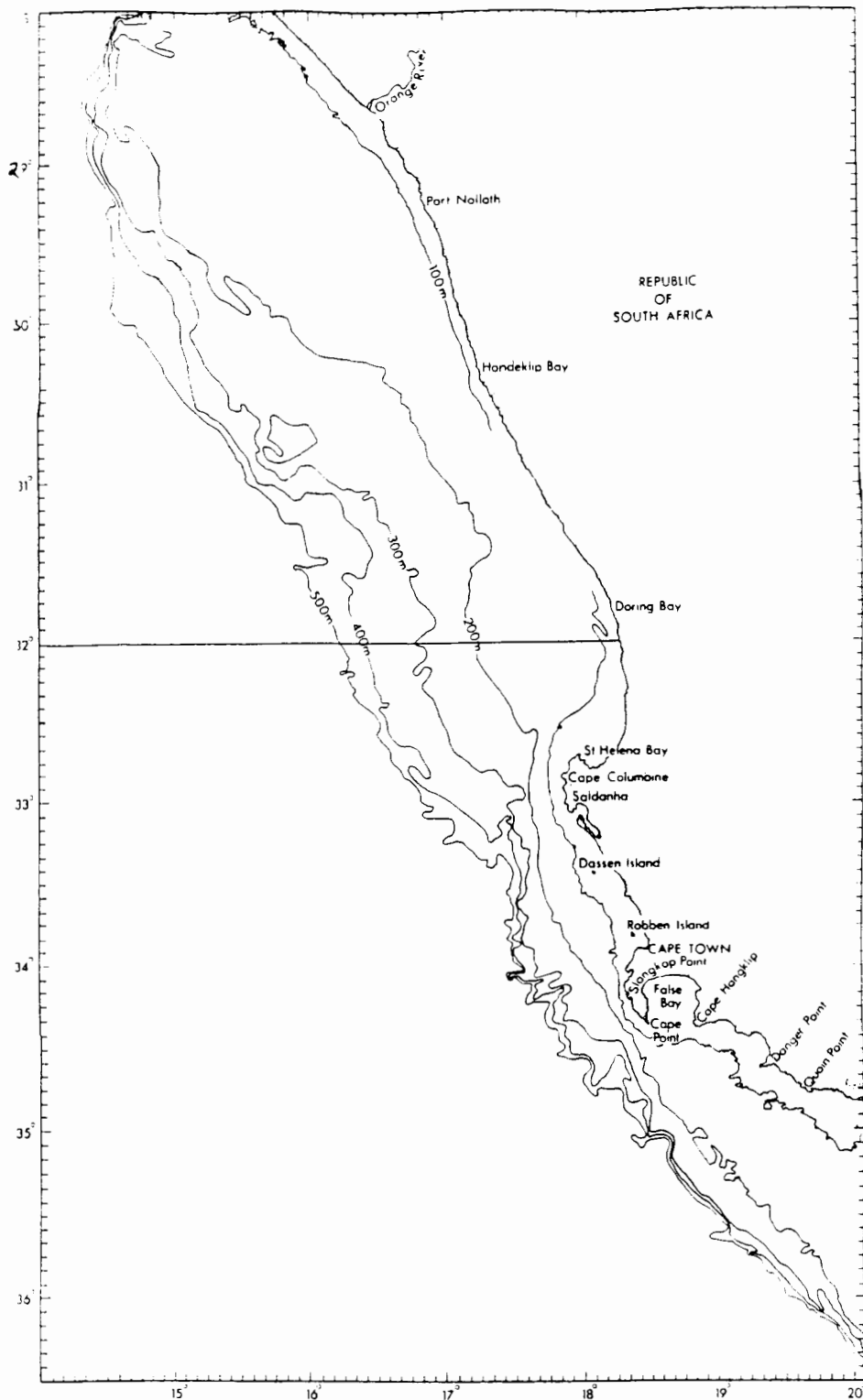


Figure 11.1 : Map of the South African west coast illustrating how it has been divided into 6 depth zones and 2 latitude zones.

rise to these distributions are the estimates of numbers-at-age by depth zone and species obtained from the research cruises (Tables 4.8 and 4.9 provide such estimates obtained using the algorithm given in Appendix 4.B), a simple diffusion-like model has been used here.

0-year-old *M. capensis* are assumed to move into the 0-100m and 100-200m depth zones immediately after birth. Similarly 0-year-old *M. paradoxus* are assumed to move into the 100-200m and 200-300m depth zones at this time. Fish are then assumed to move progressively out to greater depths as they get older/larger. The per capita rate at which fish move from one subarea to another is assumed to be independent of abundance and time, and merely a function of age and the subareas between which movement is taking place. For simplicity, it has been assumed that in each year, fish either move between the northern and southern latitude zones within the same depth zone or to an adjacent deeper depth zone within the same latitude zone. They can, of course, also remain in the subarea in which they were at the start of the year. No account is taken in the model of either recruitment fluctuations or the fact that natural mortality is a function of depth zone (due to the distribution of different length-classes of hake). While both of these factors are likely to lead to some bias in the estimates obtained, this crude representation is probably sufficient for the purposes for which this model is constructed.

The movement rates between different depth zones have been estimated by minimizing the quantity:

$$SS = \sum_{y=1986}^{1989} \sum_{a=1}^7 \sum_{D=1}^5 (\text{logit}\{Q_{a,D,y}^s\} - \text{logit}\{\hat{Q}_{a,D}^s\})^2 \quad (11.1)$$

where $Q_{a,D,y}^s$ is the observed fraction of fish of species s aged a years in depth zone D during year y ,

$\hat{Q}_{a,D}^s$ is the model-predicted fraction of fish of species s aged a years in depth zone D during year y (as movement is assumed to be independent of time, $\hat{Q}_{a,D}^s$ is not dependent on year),

Σ' is summation over those depth zones for which it is assumed that $\hat{Q}_{a,D}^s$ is non-zero, and

$\text{logit}(X) = \ln\{X/(1-X)\}$.

The summation in Equation (11.1) has been taken over ages 1 to 7 even though data for ages 0 to 12 are given in Tables 4.8 and 4.9. This is because the estimates of numbers-at-age for age 0 and for ages 8-12 are very imprecisely determined.

$\hat{Q}_{a,D}^s$ is determined by assuming that all 0-year-old fish occur in two of the six depth zones. This is not an unreasonable assumption (see Tables 4.8 and 4.9). The location of the 0-year-old *M. capensis* and 0-year-old *M. paradoxus* are as detailed above. Thus for *M. capensis* ($s=1$) the distribution of 0-year-olds is modelled by:

$$\hat{Q}_{0,0-100}^s = \hat{\psi}_{0,0-100}^s ; \quad \hat{Q}_{0,100-200}^s = (1 - \hat{\psi}_{0,0-100}^s) \quad (11.2)$$

where $\hat{\psi}_{a,D}^s$ is the model-predicted probability that a fish of species s and of age $a-1$ does not move from depth zone D to depth zone $D+1$ during the year [where $a=-1$ refers to the

distribution of fish at birth, and zone D+1 refers to the next deeper zone after zone D].

The values of $\hat{Q}_{a,D}^s$ for $a > 0$ can then be calculated from the balance equation:

$$\hat{Q}_{a,D}^s = (1 - \hat{\psi}_{a-1,D-1}^s) \hat{Q}_{a-1,D-1}^s + \hat{\psi}_{a-1,D}^s \hat{Q}_{a-1,D}^s \quad D > 1 \quad (11.3)$$

$$\hat{Q}_{a,1}^s = \hat{\psi}_{a-1,1}^s \hat{Q}_{a-1,1}^s \quad D = 1$$

where $D = 1$ corresponds to 0-100 for *M. capensis* and 100-200 for *M. paradoxus*.

Given the limited amount of data available to estimate movement rates, it is necessary to model the parameters $\psi_{a,D}^s$ as parsimoniously as possible. Four alternative parameterizations of $\psi_{a,D}^s$ have been considered (in all four parameterizations, $\psi_{a,D}^s$ has been assumed to be dependent on species).

- a) $\psi_{a,D}^s$ is age and depth zone invariant,
- b) $\psi_{a,D}^s$ is a function of age alone,
- c) $\psi_{a,D}^s$ is a function of depth zone only, and
- d) $\psi_{a,D}^s$ is the product of an age-effect and a depth zone-effect.

In order to use a logit transformation (see Equation 11.1), it is necessary that none of the values to be transformed are zero or unity. The approach used to eliminate zeros from the data (there were no ones) was to replace them by 1% of the smallest non-zero value for the corresponding age. Predicted values of zero (i.e. the top right-hand corner of the Q matrix - see Table 11.4) were omitted from the

sum of squares calculation of Equation (11.1), because the effects of entries fixed at zero should not influence the determination of the free parameters of the model.

Of the four parameterizations considered, the most general submodel provides the best fit to the data (see Table 11.3). In order to perform the tests in Table 11.3, it has been assumed that each non-zero entry in the predicted Tables is an independent data point. Thus, the application of the log-likelihood ratio test is not strictly valid because these data points are not all independent. However, the applications of this test do provide indications of the relative quality of the four fits. One rather surprising feature of the results of Table 11.3 is that the fit of the five parameter submodel (depth zone effects only) has a lower sum of squares than the fit of the eight parameter submodel (age effects only). This suggests the depth zone effects alone explain far more of the variance (particularly for *M. paradoxus*) than age-effects alone. Nevertheless, as the results of Table 11.3 suggest that incorporating both effects into the model provides the best fit, the operating model has been based on the product model.

Table 11.4 contains the predicted fractions of each age-class in the 0-500m depth range by species and depth zone (i.e. the \hat{Q} matrices). Table 11.5 contains the actual data used to fit the submodels and also gives the residual matrices ($\text{logit}\{Q_{a,D,y}^s\} - \text{logit}\{\hat{Q}_{a,D}^s\}$). Table 11.6 contains the estimates of the model parameters.

At face value, the fits of the best submodel to the data appear to be reasonably satisfactory, because the model estimates of the depth zone in which the maximum number of fish in a particular age-class are found, are usually close to the observed values (compare the data matrices in Table 11.5 with the predicted matrices in Table 11.4). Furthermore, the residuals of the fits (Table 11.5) appear reasonably random.

For *M. capensis*, the observations indicate that very low percentages of fish of this species are found in water deeper than 400m (Table 11.5), and the model predictions (Table 11.6) reflect this by setting ψ values for $D = 300-400$ very close to unity. The parameter estimates for this species suggest that the fish generally do not change depth zones, but rather remain where they are (most age and depth zone effects are close to unity). Exceptions to this are ages 2 and 4, for which 97% and 70% respectively of the fish are estimated to change depth zone during the year. Movement is almost entirely age-related and independent of depth, except for the 0-100m depth zone for which the depth zone effect is somewhat different (0.945).

The picture for *M. paradoxus* is more complex, with depth zone effects playing a greater role. This is because a large fraction of age 4+ fish move to the 500m+ depth zone each year (the depth zone effect for the 400-500m depth zone is 0.091 - Table 11.6). Note that the fractions given in Table 11.4 refer only to fish in the 0-500m depth range, because the 500m+ depth zone to which the fish move has not been incorporated in the tabulation.

Extending this approximation to take account of the two latitudinal zones, the proportion of fish of species s aged a years which moves from subarea A to subarea A' , $T_a^{s,A,A'}$, is modelled by:

$$T_a^{s,A,A'} = \begin{cases} [1 - \hat{\psi}_{a,D(A)}] & \text{if subareas } A \text{ and } A' \text{ lie in the same} \\ & \text{latitude zone and subarea } A' \text{ is one} \\ & \text{depth zone deeper than subarea } A \\ \lambda_z \hat{\psi}_{a,D(A)} & \text{if subareas } A \text{ and } A' \text{ lie in the same} \\ & \text{depth zone and subarea } A \text{ lies in} \\ & \text{latitude zone } z \\ 0 & \text{otherwise} \end{cases} \quad (11.4)$$

where $D(A)$ denotes the depth zone of subarea A .

This formalism assumes that the proportion of fish which moves between the two latitude zones is a fraction of the fish which do not move to a deeper depth zone. The assumption that the fraction is independent of species and age has been made because there are no data to quantify the extent of longshore movement. "Base case" values for λ_z of 0.2 and 0.4 for the southern and northern latitude zones respectively have been assumed, although there is no quantitative basis for this choice. For this reason, a robustness trial investigating the sensitivity of the results to a different value for λ_z has been performed (see Section 11.7.4). The value of λ_z for the Northern latitude zone is twice that for the Southern latitude zone so that (at equilibrium) the pristine biomass in the Southern zone is twice that in the Northern, even though the area of the latter is slightly larger (see Table 11.2). This difference in pristine biomasses is consistent with the perception that the density in the

Northern zone is naturally lower than that in the Southern (A.I.L. Payne, SFRI, pers. commn).

Thus the approach adopted is fairly simple, because the available data make it impossible to estimate the parameter values needed to model further complexity.

11.3.2 Distribution of effort across the fishing grounds

The distribution of effort across the fishing grounds has seldom been investigated quantitatively for fisheries [exceptions to this are Caddy (1975), Hilborn and Ledbetter (1979) and Hilborn and Walters (1987)]. A large number of factors influence this distribution. Some of these are:

- a) effort goes where fishing is best (i.e. effort responds to catch rate),
- b) there is exploratory effort,
- c) there are minimum acceptable catch rates,
- d) fishermen seek out particular sizes of fish which are more profitable or required for the market at a particular time,
- e) the topography of certain (possibly high density) regions is such that fishing is not possible there (for example, off the South African west coast, there are some rocky areas on which trawlers cannot operate even though long-lining has shown that large numbers of hake may be found there), and
- f) fishermen may fish in certain regions only because they happen to be *en route* to their preferred fishing grounds (this is the case for trawlers based on the South African

west coast, which may trawl off this coast whilst *en route* to the fishing grounds on the south coast).

It is not possible to model the distribution of effort across the fishing grounds in great detail due to lack of data. However, it has been suggested (Hilborn and Ledbetter 1979) that the net result of all these factors can be summarized by three alternative hypotheses.

- a) Fishermen always fish in the same traditional patterns. Hence the best predictor of the distribution of next year's fishing effort is the average of the distributions of all previous years' fishing effort.
- b) Fishermen move to maximize CPUE. Hence the best predictor of the distribution of next years fishing effort is the next year's predicted CPUE by area.
- c) Each area has different costs and revenue levels. Fishermen operate to maximize profits and so tend to find an appropriate balance between factors a) and b).

Hilborn and Ledbetter (op. cit.) argue that hypothesis c) is the most appropriate for the British Columbia salmon fishery. This would also seem likely to be the case for the hake fishery off the South African west coast, because CPUE is not constant across the fishing grounds (see Table 11.7) but fishermen have nevertheless expanded their operations (to include regions where CPUE is lower) in the past.

Let V^A be the combination of all factors (other than abundance) which determine the distribution of total effort across the fishing grounds (for example, distance from port and topography). V^A needs to be a

function of time as well as subarea, because the distribution of effort has changed markedly over time because some factors (for example, distance from port and bottom depth) have become less important. Approximately, the effort applied in subarea A can be modelled by:

$$E_Y^A = (V_Y^A \tilde{B}_Y^A / \sqrt{H^A}) / \sum_{A'} (V_Y^{A'} \tilde{B}_Y^{A'} / \sqrt{H^{A'}}) E_Y^{\text{tot}} = D_Y^A E_Y^{\text{tot}} \quad (11.5)$$

where E_Y^A is the effort applied in subarea A during year y,
 V_Y^A is the (abundance independent) desirability of subarea A as a function of time,
 \tilde{B}_Y^A is the exploitable biomass (i.e. both species combined) in subarea A at the start of year y,
 H^A is the area of subarea A (see Table 11.2).
 E_Y^{tot} is the total effort applied during year y.

Equation (11.5) is only an approximation to hypothesis c). In particular, if V^A is independent of area, formula (11.5) does not collapse to hypothesis b) as would be required of an exact representation of hypothesis c). The $\sqrt{H^A}$ factor in Equation (11.5) is a compromise between two extreme perceptions of how "abundance" should be measured. Assume that there are only two subareas and also assume that their desirabilities are the same; now consider the problem of allocating a fixed quantity of effort between them.

- a) If the two subareas contain the same biomass but have different areas (and hence different fish densities and catch rates), intuitively most of the effort should be allocated to

the subarea with the highest density (i.e. "abundance" can be equated with "density").

- b) If the two subareas have the same initial densities but one has a higher biomass, optimal performance would be achieved by allocating more effort to the subarea with the larger biomass (i.e. "abundance" can be equated with "biomass").

In order to obtain reasonable performance in both of these cases, "abundance" has been taken to be the geometric mean of density ($=B_Y^A/H^A$) and biomass ($=B_Y^A$).

The V_Y^A are defined only up to a multiplicative constant (if all the V_Y^A are multiplied by a constant, the values of D_Y^A are unchanged). It is possible, by assuming that the CPUE for a subarea is an index of the density in that subarea, to obtain estimates for the parameters V_Y^A [again up to a multiplicative constant] by means of the formula:

$$V_Y^A = E_Y^A \sqrt{H^A} / (C/E)_Y^A \quad (11.6)$$

Application of formula (11.6) requires that data be available for all subareas for all years. Unfortunately, many of the requisite data do not exist (either at all, or if they do, not in a computer readable format). What are available are the raw catch and effort data (hence catch and effort by subarea) for 1989. These data were used to obtain V_Y^A for 1989 (see Table 11.7). The remainder of the V_Y^A 's were inferred by noting that the fishery expanded from the Cape Peninsular fishing grounds to cover the entire continental shelf with the introduction of the foreign fleets in 1962. Furthermore, prior to 1962

(approximately), most of the fishing was restricted to depths of less than 400m (A.I.L. Payne, SFRI, pers. commn). For simplicity therefore, V_Y^A has been modelled as follows:

$$V_Y^A = \begin{cases} 0 & \text{if } y < 1963 \text{ and } A \text{ is in the Northern} \\ & \text{latitude zone or } y < 1963 \text{ and } A \text{ is} \\ & \text{deeper than 400m} \\ 0 & \text{A is deeper than 500m} \\ V_{1989}^A & \text{otherwise} \end{cases} \quad (11.7)$$

This approach is obviously a gross simplification of the real situation, but should be sufficient to mimic the major features of the changes in the spatial exploitation pattern for Cape hake off the South African west coast.

11.4 The operating models

In constructing the two operating models, three important assumptions were made to keep the algebra relatively simple, so that the computation time did not get impractically long.

The first assumption involves ignoring the sex-structure of the population. This means, for example, that the number of births (see Equation 11.13) is a function of the total mature biomass rather than of its female component only. Violations of this assumption are unlikely to be of much consequence, however, as the values of the stock-recruitment relationship parameters are selected to achieve the desired starting conditions, and taking account of sex-structure would probably result in only slight alterations of these values.

The second simplifying assumption is that $F_{Y,a} = S_{Y,a} \cdot F_Y$. While this will not lead to substantial error if F_Y is small, the possibility exists that, if F_Y is large enough, almost the entire population in a subarea may be exterminated (even for small $S_{Y,a}$). This is clearly unrealistic because a small $S_{Y,a}$ reflects, in part, the fact that some fish are not available to the fishing gear. A more appropriate way of modelling availability would be to consider two components of the population: one which is available to fishing and another which is not. Fish would then tend to move from the non-available to the available component as they become older. However, the implementation of such an approach would lead to substantially increased computation times.

The third assumption is that all movement between subareas occurs at the end of the year. This assumption is unlikely to result in substantial error, unless fishing mortality in some of the subareas is sufficiently high that the numbers-at-age at the end of the year in those subareas are markedly lower than the corresponding average numbers-at-age during the year. This is because fish from other subareas could have "fed" the heavily exploited subarea(s) during the year had such movement been possible, and thus kept the biomass reasonably steady.

In formulating the operating models, it has been assumed for convenience that the C.V. of a log-normal distribution is well approximated by the standard deviation of the associated normal distribution. While this approximation is accurate if the standard

deviation is small, some error is introduced in cases when the standard deviation is large.

The values of the operating model parameters were obtained from the biological analyses (Chapter 2), the applications of *ad hoc* tuned VPA model-estimation procedures (Chapter 10) and the applications of the Butterworth-Andrew observation error estimator (Chapter 10).

11.4.1 The age-structured operating model

The "base case" operating model is one in which only the age-structure of the resource is taken into account explicitly. This selection was made because of the computer time requirements when using the length-structured operating model (see Section 11.4.2 below).

Basic dynamics:

$$N_{y+1,a+1}^{s,A} = N_{y,a}^{s,A} e^{-z_{y,a}^{s,A}} \left[1 - \sum_{A'} T_a^{s,A,A'} \right] + \sum_{A'} N_{y,a}^{s,A'} e^{-z_{y,a}^{s,A'}} T_a^{s,A',A} \quad (11.8)$$

where $N_{y,a}^{s,A}$ is the number of fish of species s aged a years in subarea A at the start of year y ,

$z_{y,a}^{s,A}$ is the total mortality on fish of species s aged a years in subarea A during year y ,

$T_a^{s,A,A'}$ is the proportion of fish of species s aged a years that move from subarea A to subarea A' "during" (actually modelled as "at the end of") the year (see Equation 11.4),

$A = 1, 2, \dots, 12,$

$a = 0, 1, 2, \dots, a_{\max} (=9),$ and

$y = 1917, 1918, \dots, n.$

Total mortality:

$$Z_{y,a}^{s,A} = M_{y,a}^{s,A} + F_{y,a}^{s,A} \quad (11.9)$$

where $M_{y,a}^{s,A}$ is the natural (including predation) mortality on fish of species s aged a years in subarea A during year y [computed using an appropriate extension of Equation (10.18), which takes spatial effects into account],

$F_{y,a}^{s,A}$ is the fishing mortality on fish of species s aged a years in subarea A during year y :

$$F_{y,a}^{s,A} = S_{y,a}^s F_y^A$$

$$F_y^A = D_y^A q_y E_y^{\text{tot}} / H^A \quad (11.10)$$

$$\tilde{D}_y^A = (V_y^A e^{\epsilon_{y,E}^A} \tilde{B}_y^A / \sqrt{H^A}) / \sum_{A'} (V_y^{A'} e^{\epsilon_{y,E}^{A'}} \tilde{B}_y^{A'} / \sqrt{H^{A'}})$$

$$\epsilon_{y,E}^A \sim N(0; \sigma_E^2)$$

The H^A factor in Equation (11.10) allows for the fact that catch rate is proportional to density by taking account of the fact that the areas of the various subareas are not identical,

F_y^A is the fully selected fishing mortality in subarea A during year y (fishing mortality on fish for which $S_{y,a}^s \rightarrow 1$),

\tilde{D}_y^A is the desirability of subarea A during year y (note that

a stochastic component has been added to formula (11.5) to make allowance for "errors" in assessing where the "best" fishing will be found during the year],

V_Y^A is the component of the desirability of subarea A in year y which is independent of "abundance" (see Section 11.3.2),

\tilde{B}_Y^A is the exploitable biomass in subarea A at the start of year y,

H^A is the area of subarea A (see Table 11.2),

$\tilde{S}_{Y,a}^s$ is age-specific selectivity for species s in year y ($S_{Y,a}^s$ is the deterministic age-specific selectivity function for species s - $\tilde{S}_{Y,a}^s$ includes a stochastic component - see Equation 11.15),

q_y is the catchability coefficient - assumed in the "base case" operating model to be independent of time, and

E_y^{tot} is the total effort applied in year y.

The parameter σ_E is set equal to 0.05. This is an educated guess. (Though a value for this parameter could be estimated from data once these become available for years after 1989.) One of the robustness trials (see Section 11.7.4) investigates the sensitivity of the results to the value of σ_E . The value of the parameter q is set equal to 1. This value can be selected arbitrarily. All of the model-estimation procedures considered treat CPUE as a relative index of abundance, so that the absolute value of q has no effect on the performance of the management procedures (which, of course, do not "know" that it is equal to 1).

Exploitable biomass:

The exploitable biomass at the start of year y :

$$\tilde{B}_y^s = \sum_A \tilde{B}_y^{s,A} = \sum_A \sum_s \sum_a w_a^s S_{y,a}^s N_{y,a}^{s,A} \quad (11.11)$$

where w_a^s is the mass of a fish of species s aged a years at the start of the year:

$$w_a^s = 0.5 (w_{a+1/2}^s + w_{a-1/2}^s)$$

$w_{a+1/2}^s$ is the mass of a fish of species s aged a years in the middle of the year (see Table 11.7).

The exploitable biomass in the middle of year y :

$$B_y^e = \sum_A \sum_s \sum_a w_{a+1/2}^s S_{y,a}^s N_{y,a}^{s,A} e^{-Z_{y,a}^{s,A}/2} \quad (11.12)$$

Births:

$$N_{y,0}^s = e^{\epsilon_{y,r}^s - \sigma_r^2/2} \alpha SB_y^s (\beta + SB_y^s) \epsilon_{y,r}^s \sim N(0; \sigma_r^2) \quad (11.13)$$

$$SB_y^s = \sum_A \sum_a f_a^s N_{y,a}^{s,A}$$

where α, β are (Beverton-Holt) stock-recruitment relationship parameters,

SB_y^s is the spawner biomass of species s at the start of year y , and

f_a^s is the fecundity index of a fish of species s aged a

years.

For cases in which the operating model considers more than one subarea, the total recruitment is divided equally across each subarea in which 0-year-old fish are found (see Section 11.3.1).

In their specifications for an operating model for the hake resource off Northern Namibia, both Punt (1988) and ICSEAF (1990) obtained the values for the parameters α and β (see Equation 11.13) by requiring that MSY , B_{MSY}^e/K^e and K^e were pre-specified values. However, it was not possible to use this approach here, because the estimates of current depletion corresponding to the values of α and β selected in this way may be very different from those predicted by the production model or "M-independent-of-age" *ad hoc* tuned VPA model-estimation procedures. Using such parameter values would result in simulations which do not mimic the CPUE trend for the hake resource off the South African west coast particularly well. The approach used instead was to select values for α and β so that the relative increase in the exploitable biomass over the past nine years was equal to a specified value and that the current biomass matched a specified value.

Two alternative scenarios are considered for each trial. These correspond to obtaining the biomass increase rate and the current biomass from the results of the "M-independent-of-age" *ad hoc* tuned VPA and from the production model assessment, and are as follows.

- a) The "VPA" scenario - $B_{1989}^e = 150\ 000$ tons; exploitable biomass relative increase = 38.9%.
- b) The "production model" scenario - $B_{1989}^e = 604\ 000$ tons;

exploitable biomass relative increase = 54.9%.

Note that this selection means that for each trial, simulations are carried out for two sets of stock-recruitment relationship parameters.

Consideration of the variability of the recruitment estimates obtained from the VPA analyses [Tables 10.5 and 10.13] suggests that σ_r must be less than 0.2. However a value of σ_r of this size seems unrealistically low to describe the extent of recruitment variability for a species such as hake (J.G. Shepherd, pers. commn), so that a value of 0.3 was chosen for this parameter for the "base case" trials. Three of the robustness trials investigate the sensitivity of results to the value of this parameter (see Section 11.7.1).

Catch-by-mass:

$$C_{y.} = \sum_a \sum_A \sum_s w_{a+1/2}^s F_{y,a}^{s,A} N_{y,a}^{s,A} [1 - \exp(-Z_{y,a}^{s,A})] / Z_{y,a}^{s,A} \quad (11.14)$$

where C_y is the catch-by-mass in year y .

Selectivity:

Selectivity as a function of age is given by the formula:

$$S_{y,a}^s = [1 + e^{-(a-a_{r,y}^s)/\delta_r^s}]^{-1} \quad (11.15)$$

$$\tilde{S}_{y,a}^s = S_{y,a}^s e^{\epsilon_{y,s}} \quad \epsilon_{y,s} \sim N(0; \sigma_s^2)$$

where δ_r^s determines the width of the selectivity ogive for species

s, and

$a_{r,y}^s$ is the age-at-50%-selectivity for species s in year y:

$$a_{r,y}^s = \begin{cases} a_{r,1}^s & Y < Y_1 \\ a_{r,1}^s + (a_{r,2}^s - a_{r,1}^s)(Y - Y_1)/(Y_2 - Y_1) & Y_1 \leq Y \leq Y_2 \\ a_{r,2}^s & Y > Y_2 \end{cases}$$

$a_{r,1}^s$ is the age-at-50%-selectivity for species s for year Y_1 and before, and

$a_{r,2}^s$ is the age-at-50%-selectivity for species s for year Y_2 and after.

Thus the model assumes that the age-at-50%-selectivity has changed linearly over part of the time. This is an attempt to represent the trend in the age-at-50%-selectivity which was assumed in some previous age-structured assessments of the Cape hake resource off the South African west coast [e.g. Bergh (1986)]. Note that the *ad hoc* tuned VPA assessments performed in Chapter 10 used only data for recent years (1978-1989), and were not concerned with the possibility of such a change. In instances where a trend in age-at-50%-selectivity has been argued to have taken place, the changes occurred before that period. The values chosen for Y_1 (1964) and Y_2 (1978) are based on suggestions made by Bergh (1986). The "base case" operating model incorporates a temporally invariant age-at-50%-selectivity. However, one of the robustness trials (see Section 11.7.1) considers the possibility of temporal trends in the age-at-50%-selectivity.

The values for the selectivity function parameters depend on the structure of the operating model. If the operating model incorporates

spatial effects, both species have the same selectivity function ($a_{r,y} = 1.83\text{yr}$ and $\delta_r = 0.27$). These values are obtained by fitting a logistic curve to the selectivities-at-age obtained from the results of the "M-cannibalistic" *ad hoc* tuned VPA model-estimation procedure (see Chapter 10). The effect of different levels of fishing mortality in different subareas (the result of the subarea-specific desirabilities and densities) is to cause the overall age-specific fishing mortality pattern to be different from that which would occur for a homogeneous population. Specifically, the age-at-50%-selectivity is larger than 1.83yr for *M. capensis*, and lower than this for *M. paradoxus*.

The values for the selectivity function parameters for the single species (no spatial effects) operating model are obtained by fitting a logistic curve to selectivities-at-age for the "base case" 'M-independent-of-age' *ad hoc* tuned VPA model-estimation procedure. The resultant estimates are $a_{r,y} = 1.67\text{yr}$ and $\delta_r = 0.29$. The corresponding values for the two-species operating model are chosen so that the age-at-50%-selectivity for *M. capensis* (species 1) is larger than that for *M. paradoxus* (species 2), and so that the average of the two selectivity curves is similar to the curve for the "base case" VPA assessment referenced above. The resultant estimates are $a_{r,y}^1 = 3\text{yr}$, $\delta_r^1 = 0.5$, $a_{r,y}^2 = 1.5\text{yr}$ and $\delta_r^2 = 0.25$.

The value chosen for σ_s is 0.2. This selection was made to be consistent with ICSEAF's specification of an operating model for the hake resource in Northern Namibia (though their selection of this value was itself somewhat arbitrary).

Fecundity

The fecundity index is given by:

$$f_a^s = \begin{cases} w_a^s & \text{if } a > a_m^s \\ 0 & \text{otherwise} \end{cases} \quad (11.16)$$

where a_m^s is the age-at-50%-maturity of species s .

The selection of 4 years for a_m^s was made by converting the length-at-50%-maturity for both species and sexes combined (Table 2.4) into an age using the corresponding estimated growth curve. Estimates of a_m^s which were obtained by applying this approach to the maturation data for both sexes combined, but for each species separately, also suggest ages-at-50%-maturity of approximately 4 years for each species.

11.4.2 The length-structured operating model

The length-structured operating model has been investigated because it considers length as well as age. This means that it is possible to generate catch-at-age and hence mass-at-age data in a manner which corresponds more closely to the real situation than is possible for the age-structured operating model. This is because it is possible to model the stochasticity involved in the collection and processing of data more directly by using a length-based model. Unless otherwise specified, the values of the parameters for the length-structured operating model are the same as those for the "base case" age-structured operating model.

Basic dynamics:

$$N_{y+1,a+1,\ell}^{s,A} = 0.25J_{y+1,a+1,\ell-7}^{s,A} + 0.5J_{y+1,a+1,\ell-8}^{s,A} + 0.25J_{y+1,a+1,\ell-9}^{s,A} \quad (11.17)$$

$$J_{y+1,a+1,\ell}^{s,A} = N_{y,a,\ell}^{s,A} e^{-Z_{y,\ell}^{s,A}} [1 - \sum_{A'} T_a^{s,A,A'}] + \sum_{A'} N_{y,a,\ell}^{s,A'} e^{-Z_{y,\ell}^{s,A'}} T_a^{s,A',A}$$

where $N_{y,a,\ell}^{s,A}$ is the number of fish of species s aged a years in length-class ℓ in subarea A at the start of year y ,

$Z_{y,\ell}^{s,A}$ is the total mortality during year y on fish of species s in subarea A which were in length-class ℓ at the start of year y ,

$T_a^{s,A,A'}$ is the proportion of fish of species s aged a years which move from subarea A to subarea A' "during" the year (assumed for simplicity and because of lack of data to be a function of age rather than of length),

$A = 1, 2, \dots, 12$,

$\ell = 1, 2, \dots, n_\ell (=100)$,

$a = 0, 1, 2, \dots, a_{\max} (=9)$, and

$y = 1917, 1918, \dots, n$.

For simplicity, it has been assumed that growth is linear (8cm per year on average) and that both hake species grow at the same rate each year. The effects of stochastic growth have been incorporated into the operating model by allowing mixing between length-classes. The values 0.25, 0.5 and 0.25 have been chosen semi-arbitrarily because there are no data (from, say tagging experiments) which could be used to obtain estimates of the length-class mixing rates. The

values selected result in predicted length-at-age distributions which are not inconsistent with the observed distributions (i.e. the spread of lengths about the mean lengths predicted by this model is not inconsistent with the actual data).

Total mortality:

$$Z_{y,\ell}^{s,A} = M_{y,\ell}^{s,A} + F_{y,\ell}^{s,A} \quad (11.18)$$

where $M_{y,\ell}^{s,A}$ is the natural (including predation) mortality during year y on fish of species s in subarea A which were in length-class ℓ at the start of that year [computed using an appropriate extension of Equation (10.15), which takes spatial effects into account],

$F_{y,\ell}^{s,A}$ is the fishing mortality during year y on fish of species s in subarea A which were in length-class ℓ at the start of that year:

$$F_{y,\ell}^{s,A} = \tilde{S}_{y,\ell}^s F_y^A$$

$$F_y^A = \tilde{D}_y^A q_y E_y^{\text{tot}} / H^A \quad (11.19)$$

$$\tilde{D}_y^A = (V_y^A e^{\epsilon_{y,E}^A} \tilde{B}_y^A / \sqrt{H^A}) / \sum_{A'} (V_y^{A'} e^{\epsilon_{y,E}^{A'}} \tilde{B}_y^{A'} / \sqrt{H^{A'}})$$

$$\epsilon_{y,E}^A \sim N(0; \sigma_E^2)$$

$\tilde{S}_{y,\ell}^s$ is length-specific selectivity for species s in year y ($S_{y,\ell}^s$ is the deterministic length-specific selectivity function for species s - $\tilde{S}_{y,\ell}^s$ includes a stochastic component - see Equation 11.24).

Exploitable biomass:

The exploitable biomass at the start of year y :

$$\tilde{B}_y^A = \sum_A \tilde{B}_y^A = \sum_A \sum_s \sum_\ell \sum_a w_\ell^s S_{y,\ell}^s N_{y,a,\ell}^{s,A} \quad (11.20)$$

where w_ℓ^s is the mass of a fish of species s in length-class ℓ :

$$w_\ell^1 = 0.005047 \bar{\ell}^{3.113} \quad (\text{kg})$$

$$w_\ell^2 = 0.006157 \bar{\ell}^{3.046} \quad (\text{kg}).$$

The exploitable biomass in the middle of year y :

$$B_y^e = \sum_A \sum_s \sum_\ell \sum_a \tilde{w}_\ell^s S_{y,\ell}^s N_{y,a,\ell}^{s,A} e^{-Z_{y,\ell}^{s,A}/2} \quad (11.21)$$

where \tilde{w}_ℓ^s is the mean mass (during the year) of a fish of species s which was in length-class ℓ at the start of the year.

Births:

$$N_{y,0}^s = e^{\epsilon_{y,r}^s - \sigma_r^2/2} \alpha SB_y^s / (\beta + SB_y^s) \quad \epsilon_{y,r}^s \sim N(0; \sigma_r^2) \quad (11.22)$$

$$SB_y^s = \sum_A \sum_\ell \sum_a f_\ell^s N_{y,a,\ell}^{s,A}$$

where α, β are (Beverton-Holt) stock-recruitment relationship parameter values,

SB_y^s is the spawner biomass of species s at the start of year y , and

f_ℓ^s is the fecundity index of a fish of species s in length-

2.

In cases in which the operating model incorporates more than one subarea, the total recruitment is divided equally between each subarea in which 0-year-old fish are found. The recruitment in each subarea is then distributed equally across the length-classes in which 0-year-olds are found. The values for α and β are calculated as for the age-structured operating model.

Catch-by-mass:

$$C_Y = \sum_a \sum_A \sum_l \sum_s \tilde{w}_l^s F_{Y,l}^{s,A} N_{Y,a,l}^{s,A} [1 - \exp(-Z_{Y,l}^{s,A})] / Z_{Y,l}^{s,A} \quad (11.23)$$

where C_Y is the catch-by-mass in year y .

Selectivity

Selectivity as a function of length is given by the formula:

$$S_{Y,l}^s = [1 + e^{-((l-l_r^s)/\delta_r^s)}]^{-1} \quad (11.24)$$

$$\tilde{S}_{Y,l}^s = S_{Y,l}^s e^{\epsilon_{Y,s}} \quad \epsilon_{Y,s} \sim N(0; \sigma_s^2)$$

where δ_r^s determines the width of the selectivity ogive for species s ,
and

l_r^s is the length-at-50%-selectivity for species s .

A value for σ_s of 0.05 was chosen so that the average C.V. for the selectivity of an age-class is similar to that for the "base case" age-structured operating model.

The values of the length-specific selectivity function parameters: $\ell_r^1 = \ell_r^2 = 22.68\text{cm}$ and $\delta_r^1 = \delta_r^2 = 2.192$ were obtained by converting the values of the "base case" age-specific selectivity function parameters into length, using the assumed growth curve.

Fecundity:

The fecundity index is given by:

$$f_\ell^s = [1 + \exp(-(\bar{\ell} - \ell_m^s) / \delta_m^s)] w_\ell^s \quad (11.25)$$

where ℓ_m^s is the length-at-50%-maturity for species s , and δ_m^s is a parameter which determines the width of the maturation ogive for species s .

The values for the maturation ogive parameters: $\ell_m^1 = 39.38\text{cm}$, $\delta_m^1 = 0.134$, $\ell_m^2 = 42.16\text{cm}$ and $\delta_m^2 = 0.124$ were taken from the results of the analyses reported in Table 2.4.

11.5 Setting up the simulation trials

The initial conditions for each trial (for year $y=1917$) correspond to a resource whose biomass is drawn from the distribution about its average pre-exploitation level which would be expected to result from the assumed level of random recruitment fluctuation. Such a situation would seem to be realistic for the hake resource off the South African west coast, because any catches prior to 1917 would have had a negligible effect on the resource biomass. Unlike previous analyses of this sort for the Cape hake fisheries [e.g. Punt (1988)], the biomass was not set exactly at its average pre-exploitation level, as

this may lead to improved performance for procedures which assume that $B_{1917}=K$.

The numbers-at-age (and length) for each of the R_{\max} Monte-Carlo data sets at the start of year $y=1917$ were thus generated as follows.

- a) From arbitrary starting values for the numbers-at-age (and length, when necessary), the model described above was projected forward deterministically and with zero catches (i.e. $\epsilon_{y,r}^s = \sigma_r^2/2; F_y=0$) until the biomass of the resource (i.e. both species combined) reached equilibrium.
- b) The resource was then projected forward for 20 more years with no catches, but with stochastically fluctuating recruitment (i.e. $\epsilon_{y,r} \neq 0; F_y=0$).
- c) The resultant numbers-at-age (length) after the 20 years were taken to be the numbers-at-age (length) at the start of 1917.

The numbers-at-age (and length) at the start of management (1990) are generated by projecting the numbers-at-age (and length) at the start of 1917 forward using Equation (11.7) [or Equation (11.17)]. If, during this process, the resource is rendered extinct, the simulation is aborted. When performing the projections of the operating model, catches from 1917 to 1989 are set equal to their historical values, and those subsequent to 1989 are set to the TACs estimated by the management procedure under consideration (i.e. TACs set are assumed always to be taken exactly).

11.6 Generation of data

The data which are generated by the operating models, and which can therefore be used by management procedures are:

- a) catch data,
- b) effort/CPUE data,
- c) survey biomass data,
- d) recruitment indices,
- e) catch-at-age data, and
- f) mass-at-age data.

Data types a)-d) are generated in exactly the same manner for both classes of operating model, but because it incorporates length- as well as age-structure, the length-structured operating model can generate the catch-at-age data in a manner which more closely resembles the real situation than is possible for the age-structured operating model.

11.6.1 Catch data

The reported catch-by-mass data (C_y^{obs}) are available for every year from 1917 until year $n-1$ (where n ranges from 1990 to 2009). The reported catches for the years prior to 1954 are subject to log-normally distributed error with a C.V. of 5% to make allowance for the errors inherent in the estimation of total catches from logbooks by Chalmers (1976). This method of generating catch-by-mass data thus assumes that there was no discarding in the past. One of the robustness trials (see Section 11.7.2) involves investigating the

performance of management procedures for a situation in which discarding has actually taken place.

11.6.2 Effort/CPUE data

Effort data are available for all years from 1955 until year $n-1$. The effort in year y , E_y^{obs} , generated by the operating model is calculated as follows:

$$E_y^{obs} = E_y^{tot} e^{\epsilon_{y,q} - \sigma_q^2/2} \quad \epsilon_{y,q} \sim N(0; \sigma_q^2) \quad (11.26)$$

where E_y^{tot} is the total effort applied in year y , and

σ_q is the observation error standard deviation - taken to be 0.16 in the "base case" operating model. This value was obtained from consideration of the standard deviations of the residuals of the fishing mortality *versus* fishing effort relationships for the "base case" 'M-independent-of-age' *ad hoc* tuned VPA model-estimation procedure (~0.2) [Table 10.6], and the standard deviation of the residuals of the production model fit to the CPUE data (~0.12) [Table 10.15].

11.6.3 Survey biomass data

Survey biomass estimates of abundance are available twice a year, starting in the middle of 1983. The mid-year survey biomass estimates are generated using the formula:

$$(B_y)^e^{obs} = \Omega B_y^e e^{\epsilon_{y,u} - \sigma_u^2/2} \quad \epsilon_{y,u} \sim N(0; \sigma_u^2) \quad (11.27)$$

and the begin-year survey biomass estimates using the formula:

$$\hat{B}_Y^{obs} = \Omega \hat{B}_Y e^{\epsilon_{Y,u} - \sigma_u^2/2} \quad \epsilon_{Y,u} \sim N(0; \sigma_u^2) \quad (11.28)$$

where Ω is the bias of the survey biomass estimates - taken to be 0.5 on the basis of the results of the application of the Butterworth-Andrew observation error estimator [Table 10.15], and

σ_u is the "C.V." of the survey biomass estimates - taken to be 0.21 in the "base case" operating model. This value reflects the combined effects of sampling error (C.V.s typically of the order 0.15 - see Table 4.7) and other factors which contribute to the precision of the survey biomass indices [e.g. weather, skipper performance (even for a planned survey grid, catch rates obtained depend on the skill of the skipper at the time of setting the net), area coverage, and movement, etc] - also taken (somewhat arbitrarily) to be 0.15.

11.6.4 Recruitment indices

For simplicity, recruitment indices are only generated at the start of each year:

$$\hat{N}_{Y,0}^{obs} = \Omega_2 \hat{N}_{Y,0} e^{\epsilon_{Y,u} - \sigma_u^2/2} \quad \epsilon_{Y,u} \sim N(0; \sigma_u^2) \quad (11.29)$$

where Ω_2 is the bias of the recruitment index (as all management procedures which use the recruitment index treat it as a

relative index of recruitment, the absolute value of Ω_2 is unimportant so it was set arbitrarily to 0.1), and σ_u is the C.V. of the recruitment index - taken to be 0.52 in the "base case" operating model. This value has been chosen because the survey estimates of 0-year-abundance typically have C.V.s of 0.5. The remainder of σ_u is made up by the effects of non-sampling contributions (C.V. = 0.15 assumed above).

11.6.5 Catch-at-age/Mass-at-age data

Catch-at-age and mass-at-age data are generated for every year from 1978 to year $n-1$. The actual details of the generation process depend on whether the operating model is age-structured or length-structured. One of the robustness trials investigates the effect of generating catch-at-age data for the period 1964 to $n-1$. The choice of 1964 corresponds to the first year in which length frequency data were collected, and hence the first year for which it would be possible, in principle, to estimate catch-at-age data.

11.6.5.1 Age-structured operating model

$$C_{y,a}^{\text{obs}} = G \cdot C_{y,a} e^{\epsilon_{y,a,c} - \sigma_{y,a,c}^2/2} \quad (11.30)$$

where $C_{y,a}^{\text{obs}}$ is the estimate made of the catch (in number) of fish aged a years during year y ,

C_y^{obs} is the reported catch-by-mass in year y (see Section 11.6.1),

$C_{y,a}$ is the "true" catch (in numbers) of fish aged a years during year y ,

$$\epsilon_{y,a,c} \sim N(0; \sigma_{y,a,c}^2)$$

$$G = C_Y^{\text{obs}} / \left(\sum C_{y,a} \frac{-\text{obs}}{w_{y,a}} e^{\epsilon_{y,a,c} - \sigma_{y,a,c}^2/2} \right)$$

$$\sigma_{y,a,c}^2 = \sigma_c^2 \left[\sum_{a=0}^{a_{\text{max}}} C_{y,a} / (a_{\text{max}}+1) / C_{y,a} \right]^{\beta'}$$

Note that $C_{y,a}$ is not known to an assessor, who has to use the $C_{y,a}^{\text{obs}}$ data in assessment calculations. Provided β' is positive, this formulation ensures that, in a given year, the variances for catches-at-age which are small will be larger than those for catches-at-age which are large. This mimics the effects of sampling error. This approach is very simple and does not take length- and spatial-effects into account when generating catch-at-age data.

The adjustment factor, G , is chosen to ensure that observed values of catch-at-age always correspond to the same fixed reported catch mass. The "base case" parameter values are $\beta' = 1.0$ and $\sigma_c = 0.4$. These choices were made because they correspond to the selections made by ICSEAF (1990) in their specification of the error structure for catch-at-age data. This selection was based on mimicking the levels of variability typical of catch-at-age data.

As a rough check to see whether the value assumed for σ_c in the "base case" operating model (0.4) is realistic, a numbers-at-age matrix was calculated by projecting the numbers at the start of each cohort predicted by the "base case" application of the "M-independent-of-age" *ad hoc* tuned VPA [see Table 10.3(a)] forward, assuming that

fishing-mortality-at-age can be divided in an age- and a year-effect (the separability assumption). The selectivity-at-age vector was taken from Table 10.7 and the fully-selected fishing mortalities were taken from Table 10.2(a). M was taken to be 0.3yr^{-1} . From this estimated N -matrix, an estimated catch-at-age matrix was computed. Values for $\sigma_{a,c}$ were then estimated by finding the standard deviation of the differences between the logarithms of the observed and predicted catches-at-age.

The elements of the resultant $\sigma_{a,c}$ vector are smaller than those predicted from Equation 11.30. However, the estimates of $\sigma_{a,c}$ were calculated assuming that the predicted catches-at-age were independent. This is not the case, because the numbers-at-age at the start of each cohort, the selectivities-at-age and the fully-selected fishing mortalities are estimated parameters. Making an *ad hoc* correction for the estimation of these parameters to the values of $\sigma_{a,c}$ results in values of $\sigma_{a,c}$ which are somewhat larger than those predicted by Equation 11.30. These results suggest that the use of $\sigma_c=0.4$ is not unrealistic.

11.6.5.2 Length-structured operating models

In contrast to the purely empirical approach used for the age-structured operating model to generate catch-at-age and mass-at-age data, the generation of these data for the length-structured operating model is far more realistic, because information on the "true" length- as well as age-distribution of catches is available. The estimated catches-at-age are generated using the formula:

$$C_{Y,a}^{obs} = \sum_{\ell} n_{Y,a,\ell}^{obs} f_{Y,\ell}^{obs} C_Y^{obs} / w_Y^{-obs} \quad (11.31)$$

where $C_{Y,a}^{obs}$ is the estimate made of the catch (in number) of fish aged a years during year y ,

$n_{Y,a,\ell}^{obs}$ is the estimate made of proportion of fish in year y in length-class ℓ which are aged a years,

$f_{Y,\ell}^{obs}$ is the estimate made of the proportion of fish in the catch which fall into length-class ℓ in year y , and

C_Y^{obs} is the reported catch-by-mass in year y (see Section 11.6.1), and

w_Y^{-obs} is the estimate made of the mean mass of a fish in year y .

In practice (see Chapter 4), the estimates required to apply Equation (11.31) are derived from samples collected on research surveys and of commercial catches. These data have been generated here in a manner that mimics this collection process as closely as possible.

The estimates of the proportion of the catch falling into each length class in year y ($f_{Y,\ell}^{obs}$) are generated by normalizing an estimated overall catch length frequency for year y :

$$f_{Y,\ell}^{obs} = f_{Y,\ell}^T / \sum_{\ell} f_{Y,\ell}^T \quad (11.32)$$

where $f_{Y,\ell}^T$ is the overall catch length-frequency generated for year y .

The overall length-frequency for year y is the sum of the length-frequency samples "collected" during that year. Each of these

individual length-frequency samples is generated from the catch taken in one subarea using the following algorithm.

- a) The subarea (A), from which the sample used to provide the length-frequency data was collected, is generated from a multinomial distribution with a probability of selecting area A of:

$$P_A = \frac{E_Y^A}{\sum_{A'} E_Y^{A'}} \quad (11.33)$$

[i.e. it is assumed that areas in which intensive fishing takes place are more likely to be sampled, as the length-frequency data are collected from the commercial vessels.]

- b) The length-frequency data are then generated from independent multinomial random variables. The probability of a randomly selected fish falling into length-class ℓ is taken to be:

$$P_\ell = \frac{C_{Y,\ell}^A}{\sum_{\ell'} C_{Y,\ell'}^A} \quad (11.34)$$

where $C_{Y,\ell}^A$ is the number of fish in length-class ℓ at the start of year y , which were caught in subarea A during that year.

This approach to generating the overall catch length-frequency data takes account of the fact the individual length-frequency samples usually overrepresent some length-classes because fish are not homogeneously distributed over the fishing grounds. The number of samples used to construct the actual annual overall catch length-frequencies, and the total number of fish sampled each year, are time dependent. The historic values of these parameters are given in Table

11.9. The number of samples collected, as well as the number of fish measured, in each year in the future is taken to be the average over the last three years (Table 11.9). As there are no data on the number of fish measured per sample, it is assumed that the same number of fish are measured for each sample.

The estimated proportion of ages within a length-class during year y , $n_{y,a,l}^{obs}$, is estimated from the results of the direct surveys. Because these cruises cover most of the fishing grounds, the problems associated with spatial effects are reduced. Hence, in order to generate these data, the following algorithm is applied.

- a) The "true" age-length key for the catch in year y ($n_{y,a,l} = C_{y,a,l} / \sum C_{y,a,l}$) is computed.
- b) The number of fish sampled per 10cm length-class is generated from a normal distribution. The normal distributions used are given in Table 11.10. These distributions were calculated from the actual length data for the fish collected for the purpose of ageing during the period 1986-1989. The numbers of fish aged per 1cm length-class are then generated by sampling from the number aged in the corresponding 10cm length-class, assuming random uniform selection across each 10cm length-class. [Note that it is not possible to assume that the average number of fish per 1cm length-class are measured each year, because for some 10cm length-classes, the average measured over each 1cm length-class is less than 1 - see Table 11.10.1]

c) Ages are then assigned to each fish by generating ages from a multinomial distribution with probabilities $n_{y,a,\ell}$.

The estimate of the mean mass of a fish in year y is given by:

$$\bar{w}_y^{\text{-obs}} = \sum_{\ell} f_{y,\ell}^{\text{obs}} \left(\sum_s \tilde{w}_{\ell}^s C_{y,\ell}^s / \sum_s C_{y,\ell}^s \right) \quad (11.38)$$

[Note that $\bar{w}_y^{\text{-obs}}$ is an estimate and not an exact value because an estimated quantity ($f_{y,\ell}^{\text{obs}}$ - the estimate of the proportion of the catch in length-class ℓ) is used in its calculation.]

The masses-at-age are then generated using the formula:

$$\bar{w}_{y,a}^{\text{-obs}} = \sum_{\ell} n_{y,a,\ell}^{\text{obs}} f_{y,\ell}^{\text{obs}} \left(\sum_{\ell} \tilde{w}_{\ell} C_{y,\ell} / \sum_{\ell} C_{y,\ell} \right) / \sum_{\ell} n_{y,a,\ell}^{\text{obs}} f_{y,\ell}^{\text{obs}} \quad (11.36)$$

11.7 The robustness trials

In order to determine the robustness of the management procedures to variations in the parameter values and functional forms adopted for the "base case" simulation trials (which involve neither cannibalism nor spatial effects), a number of robustness trials were performed in which these parameter values and functional forms were altered. The replacement of the age-structured by the length-structured operating model is treated as one of these robustness trials, so that all of the changes in parameter values and functional forms are specified relative to the "base case" two-species age-structured operating model (BC2). The values of the stock-recruitment relationship parameters α and β for the robustness trials in which population

dynamics parameters have been altered (trials B1, B8, and B9), were determined as for the "base case" operating model (see Section 11.4.1). The "base case" single-species age-structured operating model will be referred to as BC1.

Due to computer time restrictions, it was possible to consider only a very small subset of all plausible variations to the operating model parameter values and functional forms. The variations considered here were selected on two grounds: those which were considered to be most likely to have a substantial impact on the performance of the management procedures; and those which might be able to explain why certain management procedures performed badly for the "base case" trials while others did not. (Note that this latter set are not "robustness" trials in the strict sense of the term, because often they are deliberately "easier" than the corresponding "base case" trial. Their intent is to determine which of the various effects included in that trial is causing problems to the management procedure. However, for simplicity of reference, the qualification "robustness" is also used to categorize these "explanatory" trials.)

The sections that follow first detail robustness trials for the no cannibalism/ no spatial effects operating model, and then go on to specify similar trials for an operating model with cannibalism (only) and further a model with spatial effects (only).

11.7.1 Basic model related robustness trials

- 1) The stock-recruitment relationship has a Ricker instead of a Beverton-Holt form (see Equation 11.13) (Trial B1):

$$N_{Y,0}^s = e^{\epsilon_{Y,r}^s - \sigma_r^2/2} \alpha S_{Y,r}^s e^{-\beta S_{Y,r}^s} \quad \epsilon_{Y,r}^s \sim N(0; \sigma_r^2)$$

- 2) The recruitment residuals are correlated (see Equation 11.13).

$$\epsilon_{Y,r} = \sigma_1 \epsilon_{Y-1,r} + \sqrt{1 - \sigma_1^2} \epsilon'_{Y,r} \quad \epsilon'_{Y,r} \sim N(0; \sigma_r^2)$$

Two cases are considered:

$$\sigma_1 = 0.1 \text{ - Trial B2.}$$

$$\sigma_1 = 0.4 \text{ - Trial B3.}$$

- 3) The extent of recruitment variability is altered from the "base case" value $\sigma_r = 0.3$ to:
- $\sigma_r = 0.4$ - Trial B4
- $\sigma_r = 0.2$ - Trial B5
- $\sigma_r = 0.7$ - Trial B6.
- 4) The resource is exactly at its average pre-exploitation size in 1917 - Trial B7.
- 5) The age-at-50%-selectivity is assumed to have decreased over time - $a_{r,1964} = 4\text{yrs}$, $a_{r,1978} = 1.83$ (Equation 11.15) - Trial B8. The selection of these values for the age-at-50%-selectivities was not data-based, unlike most of the other selections in this thesis. However, these parameters are such that, if changes in the age-at-50%-selectivity have a detrimental effect on performance this

should be evident from the results. [The rationale for considering a decrease in the age-at-50%-selectivity is that recent increases in the CPUE might reflect a change in strategy to exploit younger fish previously ignored (for market related reasons, perhaps), rather than a recovery in the biomass overall.]

- 6) The length-structured rather than age-structured operating model is used - Trial B9.

11.7.2 Data-related robustness trials

The data-related robustness trials involve making changes to the way in which the data used by the management procedures are generated (see Section 11.6). The "base case" operating model for these trials is the two-species age-structured operating model which does not incorporate spatial effects or cannibalism.

- 1) The C.V. of the pre-1954 catches (see Section 11.6.1) is increased to 20% - Trial D1.
- 2) Catches prior to 1960 are underreported by 50%, and catches after 1960 by 10% [this is not to mimic possible discarding - see 3) below - but rather to investigate the effects of possible misreporting] - Trial D2.
- 3) Discarding takes place. This is effected by 'discarding' 80% of the catch of 0-year-olds and 50% of the catch of 1-year-olds prior to calculating the reported catches-by-mass and catches-at-age - Trial D3.

4) The extent of the variability in the fishing effort-fishing mortality relationship (Equation 11.26) is greater than in the "base case" operating model (for which $\sigma_q = 0.16$):

a) $\sigma_q = 0.2$ - Trial D4

b) $\sigma_q = 0.4$ - Trial D5.

5) q is time-dependent:

$$a) \quad q_y = \begin{cases} \exp[0.02(y-1970)] & \text{if } y > 1970 \\ 1 & \text{otherwise} \end{cases} \quad \text{- Trial D6}$$

$$b) \quad q_y = \begin{cases} \exp[0.1(y-1970)] & \text{if } y > 1970 \\ 1 & \text{otherwise} \end{cases} \quad \text{- Trial D7}$$

$$c) \quad q_y = \begin{cases} \exp[0.02(1970-y)] & \text{if } y > 1970 \\ 1 & \text{otherwise} \end{cases} \quad \text{- Trial D8}$$

$$d) \quad q_y = \begin{cases} \exp[0.1(1970-y)] & \text{if } y > 1970 \\ 1 & \text{otherwise} \end{cases} \quad \text{- Trial D9}$$

The particular parameter value choices are essentially arbitrary, because there are as yet no analyses on possible increases in vessel efficiency on which to base them. However, these four trials capture the essential problem of the catchability coefficient exhibiting a trend with time. This is important, as such an effect could markedly influence estimates of resource status (and recent recovery) based on CPUE trends.

6) The extent of variability in the survey biomass estimates is greater than the $\sigma_u = 0.21$ chosen for the "base case" operating model - $\sigma_u=0.3$ (Equations 11.27 and 11.28) - Trial D10.

- 7) The error structure of the catch-at-age data (Equation 11.30) is altered (note that $\beta' = 1$ and $\sigma_C = 0.4$ for the "base case"):
- a) $\beta' = 0$ - Trial D11 - no dependence of the precision with which an element of the catch-at-age matrix is determined upon its size.
 - b) $\beta' = 2$ - Trial D12 - substantial dependence of catch-at-age variability upon size
 - c) $\sigma_C = 0$ - Trial D13 - no error in measuring the catch-at-age data
 - d) $\sigma_C = 0.7$ - Trial D14 - substantially increased catch-at-age data variability.
- 8) Catch-at-age data are available from 1964 (rather than only 1978) until year (n-1) - Trial D15.

11.7.3 Robustness trials related to the predation model

The "base case" operating model for the trials involving cannibalism is one which does not consider spatial effects and models only one species. The trials are as follows.

- 1) The "base case" for this scenario (i.e. using the "base case" values of the cannibalism parameters - see Section 11.2) - Trial P1.
- 2) The predation mortality per unit predator biomass is varied:
 - a) $\Gamma = 0.0005$ - Trial P2
 - a) $\Gamma = 0.0025$ - Trial P3
 - a) $\Gamma = 0.02$ - Trial P4

a) $\Gamma = 0.05$ - Trial P5

- 3) The basal natural mortality rate is assumed to be 0.2yr^{-1} - Trial P6.
- 4) The "base case" size preference function is replaced by one in which predation is relatively heavier on older fish. ($R_f \rightarrow 2R_f$) - Trial P7.
- 5) The size preference function is widened. ($\sigma_w \rightarrow 2\sigma_w$) - Trial P3.

11.7.4 Robustness trials related to the spatial model

- 1) The "base case" for this scenario (i.e. using the "base case" values of the spatial submodel parameters - see Section 11.3) - Trial S1.
- 2) The values of inter-latitude zone movement rates (λ_z) are increased from 0.2/0.4 to 0.3/0.6 respectively - Trial S2.
- 3) The desirabilities of each subarea are the same within each year (but may change from one year to the next). This is achieved by setting V_{1989}^A to 1 in Equation (11.10) - Trial S3.
- 4) All subareas in the depth range 0-500m are exploited in all years and all have the same desirability. This is achieved by setting V_y^A to 1 for all years and subareas in the depth range 0-500m - Trial S4.

- 5) The efficiency of the fleet in finding the "best" fishing grounds is reduced by increasing σ_E to 0.2 - see Equation (11.10) - Trial S5.

Table 11.1 : A list of the 109 management procedures considered in this thesis. The details of the model-estimation procedures are given in Sections 6.4, 7.4, 7.6, 8.3 and 9.5, while descriptions of the harvesting strategies are given in Chapter 9. The algorithm used to construct the acronyms is given in Appendix 11.A.

Production-model based management procedures

Model-estimation Procedure	Harvesting strategies
PROK-SC	$f_{0.1}, MAC_{0.1}, RY$
PROE-SC	$f_{0.1}, MAC_{0.1}, RY$
PROK-FO	$f_{0.1}, MAC_{0.1}, RY$
PROE-FO	$f_{0.1}, MAC_{0.1}, RY$
PCOK-SC	$f_{0.1}, MAC_{0.1}, RY$
PCOK-FO	$f_{0.1}, MAC_{0.1}, RY$
PSOK-SC	$f_{0.1}, MAC_{0.1}, RY$
PSOK-FO	$f_{0.1}, MAC_{0.1}, RY$
PRTE-SC	$f_{0.1}, MAC_{0.1}, RY$
PRTE-FO	$f_{0.1}, MAC_{0.1}, RY$
PROK-S1	$f_{0.1}, MAC_{0.1}, RY$
PROK-RI	$f_{0.1}, MAC_{0.1}, RY$
PROK-PT	$f_{0.1}, MAC_{0.1}, RY$
PROK-S2	$f_{0.1}, MAC_{0.1}, RY$
PROK-S3	$f_{0.1}, MAC_{0.1}, RY$

(Table 11.1 Continued)

Partially age-structured management procedures

Model-estimation Procedure	Harvesting strategy
DE-BH	$f_{0.1}$
DE-RI	$f_{0.1}$

Age-structured management procedures

Model-estimation Procedure	Harvesting strategies
AD1	$F_{0.1}, F_{\text{status-quo}}$
AD2	$F_{0.1}, F_{\text{status-quo}}$
AD3	$F_{0.1}, F_{\text{status-quo}}$
AD4	$F_{0.1}, F_{\text{status-quo}}$
AD5	$F_{0.1}, F_{\text{status-quo}}$
AD6	$F_{0.1}, F_{\text{status-quo}}$
AD7	$F_{0.1}, F_{\text{status-quo}}$
AD8	$F_{0.1}, F_{\text{status-quo}}$
IN1	$F_{0.1}, F_{\text{status-quo}}$
IN2	$F_{0.1}, F_{\text{status-quo}}$
IN3	$F_{0.1}, F_{\text{status-quo}}$
IN4	$F_{0.1}, F_{\text{status-quo}}$
IN5	$F_{0.1}, F_{\text{status-quo}}$
IN6	$F_{0.1}, F_{\text{status-quo}}$
SH1	$F_{\text{status-quo}}$
SH2	$F_{\text{status-quo}}$

Table 11.2 : Areas in Nm^2 of 10 of the 12 subareas making up the South African west coast (see Figure 11.1). No areas are given for the two 500+m depth-zones because it is assumed that no fishing takes place there.

Depth Zone(m)	Northern Zone (28° - 32°S)	Southern Zone (32° - 36°S)
0-100	1 271	2 800
100-200	8 793	5 351
200-300	5 133	3 393
300-400	1 921	2 296
400-500	1 673	1 347
500+	-	-

Table 11.3 : Application of the log-likelihood ratio test (Draper and Smith 1966) to fits of submodels of model (11.3) to data on the fraction of fish by age and species in each depth zone. All submodels are compared to the submodel in which the probability of moving to an adjacent deeper depth zone is dependent on the product of an age effect and a depth zone effect. SS refers to the minimum value of the sum of squares function (Equation 11.1) encountered by the non-linear minimization routine.

M. capensis

EFFECTS MODELLED	SS	No parameters	F
Age and Depth zone	727.2	13	-
Age effect only	876.9	8	7.8
Depth zone effect only	841.2	5	3.7
Neither	1853.6	1	24.5

M. paradoxus

EFFECTS MODELLED	SS	No parameters	F
Age and Depth zone	190.9	13	-
Age effect only	307.9	8	16.9
Depth zone effect only	241.4	5	4.5
Neither	461.9	1	16.2

Table 11.4 : Estimates of the fraction of fish occurring in each depth zone making up the 0-500m depth range by age (i.e. the \hat{Q} matrices). Blank entries indicate fractions assumed to be zero. Note that *M. paradoxus* is assumed to be found only in water deeper than 100m.

M. capensis

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
0	0.92	0.08			
1	0.85	0.15	0.00		
2	0.02	0.83	0.14	0.00	
3	0.02	0.82	0.15	0.00	0.00
4	0.01	0.26	0.63	0.11	0.00
5	0.01	0.25	0.63	0.11	0.00
6	0.00	0.25	0.63	0.12	0.01
7	0.00	0.24	0.63	0.12	0.01

M. paradoxus

AGE	100- 200m	200- 300m	300- 400m	400- 500m
0	0.68	0.32		
1	0.46	0.48	0.06	
2	0.15	0.50	0.32	0.03
3	0.00	0.17	0.51	0.32
4	0.00	0.13	0.59	0.28
5	0.00	0.10	0.64	0.25
6	0.00	0.07	0.62	0.30
7	0.00	0.05	0.58	0.36

Table 11.5 : Observed fractions of fish in the 0-500m depth range by depth zone and age (i.e. the Q matrices) and the residuals of the fit of the best submodel to these data. Entries left blank indicate values left out of analysis.

(a) *M. capensis* - January 1986

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.66	0.34	0.00	0.00	0.00	-1.1	1.1			
2	0.30	0.53	0.16	0.00	0.00	2.9	-1.5	0.1		
3	0.01	0.50	0.48	0.00	0.00	-0.5	-1.5	1.6	-1.9	
4	0.00	0.24	0.74	0.02	0.00	-3.4	-0.1	0.5	-1.8	-1.2
5	0.00	0.27	0.66	0.07	0.00	-2.0	0.1	0.1	-0.5	-1.5
6	0.00	0.34	0.40	0.25	0.00	-0.7	0.5	-0.9	0.9	-0.8
7	0.00	0.37	0.28	0.35	0.00	-0.5	0.6	-1.5	1.4	-0.9

July 1986

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.46	0.54	0.00	0.00	0.00	-1.9	1.9			
2	0.22	0.75	0.03	0.00	0.00	2.5	-0.5	-1.6		
3	0.00	0.55	0.43	0.01	0.00	-1.9	-1.3	1.4	2.8	
4	0.00	0.19	0.67	0.13	0.00	-2.1	-0.4	0.2	0.3	-4.5
5	0.00	0.14	0.48	0.38	0.00	-1.0	-0.7	-0.6	1.6	-2.8
6	0.00	0.12	0.40	0.47	0.01	-4.4	-0.8	-0.9	1.9	0.1
7	0.00	0.19	0.38	0.42	0.00	-5.0	-0.3	-1.0	1.7	-0.8

(Table 11.5 Continued)

January 1987

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.68	0.32	0.00	0.00	0.00	-1.0	1.0			
2	0.39	0.60	0.01	0.00	0.00	3.3	-1.2	-3.1		
3	0.10	0.77	0.13	0.00	0.00	1.6	-0.3	-0.2	0.1	
4	0.00	0.50	0.50	0.00	0.00	-1.5	1.1	-0.5	-3.6	-3.9
5	0.00	0.30	0.68	0.02	0.00	-3.1	0.2	0.2	-1.7	-2.7
6	0.00	0.27	0.64	0.09	0.00	-1.7	0.1	0.1	-0.3	-1.9
7	0.00	0.38	0.43	0.18	0.00	-0.9	0.7	-0.8	0.5	-1.3

July 1987

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.88	0.12	0.00	0.00	0.00	0.3	-0.3			
2	0.73	0.26	0.01	0.00	0.00	4.7	-2.6	-3.3		
3	0.04	0.71	0.24	0.01	0.00	0.6	-0.7	0.6	2.3	
4	0.00	0.45	0.45	0.10	0.00	-2.2	0.8	-0.7	0.0	-4.6
5	0.00	0.28	0.47	0.24	0.00	-0.8	0.2	-0.6	0.9	-0.3
6	0.00	0.23	0.49	0.27	0.00	-0.8	-0.1	-0.6	1.0	-0.9
7	0.00	0.22	0.50	0.28	0.00	-0.8	-0.2	-0.5	1.1	-1.2

(Table 11.5 Continued)

February 1988

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.89	0.11	0.00	0.00	0.00	0.3	-0.3			
2	0.59	0.40	0.01	0.00	0.00	4.1	-2.0	-3.2		
3	0.04	0.74	0.22	0.00	0.00	0.6	-0.5	0.5	-0.8	
4	0.00	0.43	0.55	0.02	0.00	-0.9	0.8	-0.3	-1.9	-3.3
5	0.01	0.28	0.46	0.25	0.00	0.0	0.2	-0.7	1.0	-4.1
6	0.00	0.18	0.35	0.47	0.00	-1.0	-0.4	-1.1	1.9	-1.2
7	0.00	0.25	0.23	0.52	0.00	-0.7	0.0	-1.8	2.1	-1.1

August 1988

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.78	0.22	0.00	0.00	0.00	-0.5	0.5			
2	0.67	0.31	0.01	0.00	0.00	4.5	-2.4	-2.6		
3	0.05	0.47	0.44	0.04	0.00	0.9	-1.7	1.5	3.7	
4	0.01	0.12	0.66	0.22	0.00	-0.1	-1.0	0.1	0.8	-2.4
5	0.00	0.14	0.59	0.27	0.00	-0.7	-0.7	-0.1	1.0	-4.8
6	0.00	0.24	0.49	0.26	0.00	-0.7	0.0	-0.5	1.0	-0.8
7	0.00	0.38	0.36	0.25	0.00	-0.6	0.7	-1.1	0.9	-1.0

July 1989

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.68	0.31	0.01	0.00	0.00	-1.0	1.0			
2	0.71	0.20	0.09	0.00	0.00	4.6	-3.0	-0.6		
3	0.35	0.52	0.12	0.01	0.00	3.2	-1.5	-0.3	2.6	
4	0.05	0.44	0.31	0.20	0.00	2.2	0.8	-1.3	0.7	-0.2
5	0.00	0.29	0.47	0.24	0.00	-0.9	0.2	-0.6	0.9	-5.0
6	0.00	0.32	0.51	0.17	0.00	-1.1	0.3	-0.5	0.4	-1.2
7	0.00	0.25	0.49	0.26	0.00	-0.6	0.0	-0.6	1.0	-1.0

(Table 11.5 Continued)

(b) *M. paradoxus* - January 1986

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.24	0.69	0.07	0.00	0.0	-1.0	0.9		
2	0.00	0.11	0.62	0.25	0.01	0.0	-0.3	0.5	-0.4	
3	0.00	0.20	0.35	0.35	0.10	0.0	4.1	1.0	-0.7	-1.5
4	0.00	0.01	0.28	0.33	0.37	0.0	1.7	1.0	-1.1	0.4
5	0.00	0.00	0.23	0.45	0.31	0.0	0.3	1.0	-0.8	0.3
6	0.00	0.00	0.15	0.60	0.25	0.0	0.4	0.8	-0.1	-0.3
7	0.00	0.00	0.05	0.70	0.24	0.0	-0.1	0.0	0.5	-0.6

July 1986

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.26	0.59	0.15	0.00	0.0	-0.9	0.4		
2	0.00	0.01	0.53	0.38	0.08	0.0	-2.9	0.1	0.3	
3	0.00	0.00	0.20	0.53	0.27	0.0	-0.5	0.2	0.1	-0.3
4	0.00	0.00	0.06	0.70	0.24	0.0	-1.5	-0.9	0.5	-0.2
5	0.00	0.00	0.06	0.62	0.32	0.0	-1.1	-0.6	-0.1	0.3
6	0.00	0.00	0.06	0.44	0.49	0.0	-0.5	-0.1	-0.7	0.8
7	0.00	0.00	0.03	0.32	0.64	0.0	-0.6	-0.5	-1.1	1.1

(Table 11.5 Continued)

January 1987

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.59	0.38	0.02	0.00	0.0	0.5	-0.4		
2	0.00	0.18	0.69	0.11	0.02	0.0	0.2	0.8	-1.3	
3	0.00	0.03	0.48	0.31	0.18	0.0	1.8	1.5	-0.8	-0.7
4	0.00	0.00	0.42	0.32	0.25	0.0	0.0	1.6	-1.1	-0.1
5	0.00	0.00	0.31	0.41	0.28	0.0	0.5	1.3	-0.9	0.1
6	0.00	0.00	0.35	0.35	0.30	0.0	1.1	1.9	-1.1	0.0
7	0.00	0.00	0.12	0.39	0.49	0.0	0.7	0.9	-0.8	0.5

July 1987

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.53	0.40	0.07	0.00	0.0	0.3	-0.3		
2	0.00	0.04	0.56	0.34	0.06	0.0	-1.5	0.2	0.1	
3	0.00	0.00	0.22	0.51	0.26	0.0	-0.5	0.3	0.0	-0.3
4	0.00	0.00	0.05	0.55	0.39	0.0	-1.8	-1.0	-0.1	0.5
5	0.00	0.00	0.01	0.59	0.40	0.0	-2.4	-2.2	-0.2	0.7
6	0.00	0.00	0.00	0.58	0.41	0.0	1.4	-3.0	-0.2	0.5
7	0.00	0.00	0.00	0.44	0.54	0.0	2.0	-2.5	-0.6	0.7

(Table 11.5 Continued)

February 1988

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.68	0.30	0.01	0.00	0.0	0.9	-0.7		
2	0.00	0.19	0.47	0.33	0.02	0.0	0.3	-0.1	0.0	
3	0.00	0.02	0.56	0.27	0.15	0.0	1.5	1.8	-1.0	-1.0
4	0.00	0.00	0.34	0.58	0.07	0.0	-1.0	1.2	0.0	-1.6
5	0.00	0.00	0.17	0.71	0.12	0.0	-0.7	0.5	0.3	-0.9
6	0.00	0.00	0.06	0.51	0.43	0.0	-0.5	-0.2	-0.5	0.5
7	0.00	0.00	0.06	0.50	0.44	0.0	0.0	0.1	-0.4	0.3

August 1988

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.40	0.52	0.07	0.00	0.0	-0.2	0.2	0.1	-6.4
2	0.00	0.01	0.24	0.62	0.13	0.0	-2.7	-1.1	1.2	1.5
3	0.00	0.00	0.02	0.54	0.44	0.0	-1.0	-2.1	0.1	0.5
4	0.00	0.00	0.00	0.28	0.71	0.0	-4.7	-4.1	-1.3	1.9
5	0.00	0.00	0.00	0.48	0.52	0.0	-0.6	-3.4	-0.7	1.1
6	0.00	0.00	0.01	0.57	0.43	0.0	-2.8	-2.5	-0.2	0.5
7	0.00	0.00	0.00	0.62	0.38	0.0	-2.9	-2.8	0.1	0.1

July 1989

AGE	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m	0- 100m	100- 200m	200- 300m	300- 400m	400- 500m
1	0.00	0.50	0.45	0.04	0.00	0.0	0.2	-0.1		
2	0.00	0.01	0.40	0.48	0.11	0.0	-2.5	-0.4	0.7	
3	0.00	0.00	0.16	0.58	0.25	0.0	0.0	-0.1	0.3	-0.3
4	0.00	0.00	0.02	0.89	0.09	0.0	-2.0	-2.0	1.7	-1.3
5	0.00	0.00	0.05	0.91	0.04	0.0	-1.5	-0.8	1.7	-2.1
6	0.00	0.00	0.09	0.87	0.04	0.0	-0.9	0.1	1.4	-2.3
7	0.00	0.00	0.06	0.91	0.03	0.0	-0.6	0.1	2.0	-2.8

Table 11.6 : Estimates of the age- and depth-zone- effects obtained by fitting model (11.3) to the data given in Table 11.5 for submodel d) (i.e. the product model) for $\psi_{a,D}^S$.

EFFECT	<i>M. capensis</i>	<i>M. paradoxus</i>
Age 0	1.000	1.000
Age 1	1.000	1.000
Age 2	0.030	0.486
Age 3	1.000	0.039
Age 4	0.298	0.652
Age 5	0.986	0.711
Age 6	0.992	0.666
Age 7	1.000	0.610
Depth 0-100m	0.945	
Depth 100-200m	0.987	0.678
Depth 200-300m	1.000	0.809
Depth 300-400m	0.989	1.000
Depth 400-500m	0.995	0.091

Table 11.7 : Estimates of $v^A(1989)$ obtained from the raw catch and effort data for 1989 using Equation (11.6). Note that no estimates of $v^A(1989)$ have been obtained for the 500m+ subareas because the operating models assume that no fishing takes place in these subareas.

Subarea	Catch (tons)	Effort ('000 hours)	CPUE (t/h)	Area (nm ²)	$v^A(1989)$ x1000
NORTH					
=====					
0-100m	5.4	0.0058	0.917	1271	0.18
100-200m	95.6	0.027	3.464	8793	0.08
200-300m	1481.8	1.258	1.177	5133	14.91
300-400m	2213.7	0.874	2.531	1921	7.88
400-500m	8359.3	1.455	5.745	1673	6.20
500m+	170.9	0.031	5.508	-	
TOTAL	12326.6	3.651			
SOUTH					
=====					
0-100m	0.0	0.0	-	2800	0.00
100-200m	1132.8	0.429	2.637	5351	2.22
200-300m	17925.9	4.399	4.075	3393	18.53
300-400m	27813.2	4.895	5.682	2296	17.98
400-500m	19466.7	3.254	5.982	1347	14.82
500m+	1563.4	0.299	5.214	-	
TOTAL	67902.0	13.276			

Table 11.8 : The mean mass in mid-year of fish aged $a+1/2$ years (i.e. $w_{a+1/2}$). Units are kg.

AGE	<i>M. capensis</i>	<i>M. paradoxus</i>	Both species
0	0.016	0.013	0.013
1	0.068	0.058	0.061
2	0.196	0.182	0.190
3	0.454	0.440	0.452
4	0.853	0.795	0.827
5	1.406	1.267	1.341
6	2.124	1.886	2.005
7	2.988	2.614	2.785
8	4.141	3.345	3.694
9	5.280	4.228	4.735
10	6.935	4.697	5.586

Table 11.9 : Number of samples collected to construct annual length-frequency histograms and the total number of fish measured each year (source: R.W. Leslie, SFRI, pers. commn). The row 'AVE' is the average number of samples and fish measured over the most recent three years.

Year	No Measured	No Samples	Year	No Measured	No Samples
1973	10134	11	1982	23007	40
1974	5021	6	1983	42970	68
1975	6493	18	1984	35232	64
1976	23772	26	1985	25833	47
1977	21530	20	1986	34441	60
1978	66387	73	1987	40247	70
1979	15149	28	1988	34218	60
1980	23721	36	1989	31864	55
1981	27424	52	AVE	35443	62

Table 11.10 : Means and standard deviations of the number of fish measured per 10cm length-class. The values were calculated from the actual length data for the fish collected for the purpose of ageing over the period 1986-1989.

Length-class	Mean	S.D.
11-20cm	144.5	79.9
21-30cm	234.8	68.7
31-40cm	242.0	70.5
41-50cm	209.5	53.9
51-60cm	198.0	39.1
61-70cm	189.3	27.4
71-80cm	113.0	34.6
81-90cm	41.0	20.3
91-100cm	4.5	3.3

APPENDIX 11.A : ACRONYMS FOR MANAGEMENT PROCEDURES USED IN THIS
THESIS

In order to simplify the presentation of the results, the names of the management procedures have been replaced by acronyms. Most acronyms consist of two parts. The first part represents the model-estimation procedure and the second the harvesting strategy.

1) Production-model based management procedures

All of the acronyms for the production-model based management procedures consist of three mnemonics separated by dashes. The first mnemonic always begins with the letter 'P'. The next letter is 'R' if both CPUE and survey data are used in the estimator, 'C' if CPUE data alone are used and 'S' if survey data alone are used. The next letter is 'O' for an observation error estimator or 'T' for a total least squares estimator. The final letter in the first mnemonic is 'K' if B_{1917} is assumed to be equal to the average pristine level (K), or 'E' if B_{1917}/K is an estimable parameter.

The second mnemonic reflects the particular form of the surplus production function used:

Fox	-	FO
Pella-Tomlinson	-	PT
Ricker	-	RI
Schaefer	-	SC
Shepherd - (1987)	-	S1
Shepherd - (1982)	-	S2
Shepherd - (1982)-like	-	S3

and the third mnemonic represents the harvesting strategy. The options considered in this thesis are:

$f_{0.1}$	- F1
$f_{0.2}$	- F2
$MAC_{0.1}$	- MC1
$MAC_{0.2}$	- MC2
Replacement Yield	- RY

2) Partially age-structured management procedures

Only two partially age-structured management procedures are considered in this thesis (see Section 8.3). These are:

- Beverton-Holt form of the stock-recruitment relationship (DE-BH)
- Ricker form of the stock-recruitment relationship (DE-RI)

3) Fully age-structured management procedures

All of the acronyms for the fully age-structured management procedures consist of two mnemonics separated by a dash. The first of these represents the estimator (see Tables 7.1 and 7.2), while the second represents the harvesting strategy considered:

$F_{0.1}$	- F1
$F_{\text{status-quo}}$	- FSS

41 Short-cut methods

Only two short-cut methods are considered in this thesis. They are the SH1 (SHOT) and SH2 (DROP) methods. Both of these approaches are detailed in Section 9.5.