FRDC FINAL REPORT

SUSTAINABILITY OF SMALL-SCALE,
DATA POOR COMMERCIAL FISHERIES:
DEVELOPING ASSESSMENTS,
PERFORMANCE INDICATORS AND
MONITORING STRATEGIES FOR
TEMPERATE REEF SPECIES

Philippe E. Ziegler, Malcolm Haddon,
and Jeremy M. Lyle

September 2006

FRDC Project No. 2002/057

Australian Government

Fisheries Research and
Development Corporation

National Library of Australia Cataloguing-in-Publication Entry

Ziegler, Philippe Eric.
Sustainability of small-scale, data poor commercial fisheries : developing assessments, performance indicators and monitoring strategies for temperate reef species.

Bibliography.
Includes index.
ISBN 1 86295 341 4.

1. Fishery management - Australia. 2. Fisheries - Production contol - Australia. 3. Fish stock assessment - Australia - Data processing. I. Haddon, Malcolm, 1953- . II. Lyle, Jeremy. III. Tasmanian Aquaculture and Fisheries Institute. Marine Research Laboratories. IV. Fisheries Research and Development Corporation (Australia). V. Title. (Series : FRDC Project report ; no. 2002/057).

338.37270994

Published by the Marine Research Laboratories – Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Private Bag 49, Hobart, Tasmania 7001.

E-mail: Jeremy.Lyle@utas.edu.au Ph. (03) 6227 7277 Fax: (03) 6227 8035

The Fisheries Research and Development Corporation plans, invests in and manages fisheries research and development throughout Australia. It is a statutory authority within the portfolio of the federal Minister for Agriculture, Fisheries and Forestry, jointly funded by the Australian Government and the fishing industry.

FRDC FINAL REPORT

SUSTAINABILITY OF SMALL-SCALE,
DATA POOR COMMERCIAL FISHERIES:
DEVELOPING ASSESSMENTS,
PERFORMANCE INDICATORS AND
MONITORING STRATEGIES FOR
TEMPERATE REEF SPECIES

Philippe E. Ziegler, Malcolm Haddon, and Jeremy M. Lyle

September 2006

FRDC Project No. 2002/057

Tasmanian Aquaculture and Fisheries Institute (TAFI)

TABLE OF CONTENTS

NON TECHNICAL SUMMARY

2002/057 Sustainability of small-scale, data poor commercial fisheries: developing assessments, performance indicators and monitoring strategies for temperate reef species

PRINCIPAL INVESTIGATOR: Dr J.M. Lyle

ADDRESS: University of Tasmania
Tasmanian Aquaculture and Fisheries Institute
Private Bag 49
Hobart TAS 7001
Telephone: 03 6227 7277 Fax: 03 6227 8035

OBJECTIVES:

1. Develop appropriate and meaningful performance indicators for sedentary reef-dwelling species, using banded morwong and wrasse as models.
2. Determine minimum information requirements for the effective stock assessment of these species.
3. Develop appropriate model frameworks for testing the performance indicators developed for these fisheries.

NON TECHNICAL SUMMARY:

The development of live fish markets in Australia during the early 1990s created a strong demand for temperate reef species, specifically banded morwong and wrasse. Prior to this, these species had little commercial value. Although these fisheries have tended to be small-scale (<200 tonnes per annum), they have imposed large increases in fishing pressure on the reef fish communities, and while there has been much work on tropical species, our knowledge of how temperate reef species respond to fishing pressure is comparatively poor.

Low value, combined with the implications of a high level of spatial structuring in the populations, pose considerable difficulties for stock assessment. Specifically, the quantity of data necessary to conduct robust assessments based on, for example age- and spatially-structured models, cannot typically be justified financially. A need was therefore identified to explore suitable and meaningful stock assessment methods and monitoring strategies for such species.

This project involved five main components; a review of performance measures including their application in data-poor situations; a survey of live-fish industry

participants; detailed analysis of available fishery and biological characteristics; development of an operating model and evaluation of the suitability of biological performance measures for stock assessment; and finally management strategy evaluation based on fishery data and simple decision rules. Banded morwong in Tasmania was selected as a case study since considerable information about the fishery and biological characteristics were available.

Maximum yield target aimed at maintaining the spawning biomass at levels considered sustainable in the long term are used in high-value and well-researched fisheries around the world. If data sources are limited, simple catch and effort analysis for trends or, with enough contrast in the data, surplus-production models represent common assessment tools. Performance of these measures in reference years and some form of maximum sustainable yield (MSY) are the most widely used reference points.

The industry survey highlighted several key factors relevant to the use of fishery data for stock assessment, including data reliability (due to deliberate mis-reporting and misinterpretation of reporting requirements) and the influence of external factors (seal interactions and market forces) on catch and effort over time. In using these data a series of adjustments were made to the catch history and where possible included (*e.g.* seal interactions) in the standardization of catch rates. The survey also revealed that only part of the distributional depth range of banded morwong was fished, giving rise to the notion of a depth refuge. In developing the operating model, this aspect, along with on/off-shore movement dynamics, was explored.

A range of basic assessment tools involving catch and catch rate trend analyses and changes in key biological indicators were examined but none provided clear indications about stock status. Catch and catch rate indicators suggested that, initially at least, the fishery had impacted the stocks but that there had been relative stability in both measures in recent years. Such stability was not, however, reflected in trends for key biological indicators. In practice, catch rate stability could be based in part on the serial depletion of spatially-structured populations (noting that the spatial scale of catch and effort reporting [½ degree blocks] does not match the scale of the populations [individual reefs]).

Per-recruit and catch curve analyses suggested that sustainable fishing mortality levels are low and that current fishing pressure may be too high, though there was considerable variability in the data. All investigated biological measures indicated that the fishery had impacted on the stocks. Some changes, such as the decrease in the median age for females from around 20 to 7 years, appeared dramatic. However, age structure information, particularly for females, indicated that the representation of old fish was still relatively high, supporting the notion that there is residual biomass still available to be fished.

Biologically, banded morwong turned out to be a particularly complex species, exhibiting an unexpected degree of adaptiveness and flexibility. There was an unanticipated but significant population response to the assumed stock depletion, involving an acceleration of individual growth rates over time coupled with a decrease in the size and age of maturity. Given the life history characteristics of the species, especially its extreme longevity (living in excess of 90 years in Tasmania), these changes were surprisingly strong, and raised the question as to whether they were a density-dependent response or due to changing environmental conditions.

In full knowledge of the potential spatial mismatch of stock dynamics and available data, a compromise was needed for the development of an operating model that would simulate some spatial structuring, but not greatly exceed the limitations of data quality and quantity. The approach taken was to develop two types of operating model, a spatially simple one-region model and a spatially more complex five-region model based on stock assessment regions. High model uncertainty proved to be the main limitation of the operating models. This was largely unavoidable due to the lack of data in some crucial parameters; *e.g.* recruitment variability, regional distribution of biomass, and biomass distribution and movement rates between fished onshore and unfished offshore areas. Because of the lack of information over the spatial scale of the fishery, the models were over-parameterized. Nevertheless, some general conclusions about the stock status could be made. Most scenarios tested in the five-region model indicated high harvest rates or fishing mortalities over an extended period of the fishery. Only in recent years were these parameters predicted to be reduced to lower and more sustainable levels, within the range of internationally recommended biomass reference points. All scenarios tested by the model predicted relatively low estimates of mature biomass in at least some regions.

Despite model uncertainty, this study did provide some valuable insights into assessment and management strategies. Simple biological measures such as median age or size, age and sex ratios performed poorly as fishery performance measures and were found to be non-discriminatory as indices of mature biomass or stock status. Recruitment variability was only partly responsible for the poor performance. Even with low recruitment variability, the relationships between most of the potential performance measures and mature biomass were generally too imprecise for a useful estimation of stock status. This was the case when the measures were used singly or in combination. The imprecision of biological population properties as stock performance measures, together with uncertainty due to sampling errors and the often poor spatial representation of biological samples, severely restrict the use of biological properties in stock assessments of reef fish species. At best, such measures could be applied as qualitative indices and, if there was consistency in the direction of change between measures over time and/or space, indicate by weight of evidence that changes in the populations had occurred. Life history characters can provide insight into the level of productivity to be expected from a species, but such information alone may not be informative about stock status and whether harvest levels are sustainable.

Management strategy evaluation, by contrast, revealed that simple decision rules based on measures derived from catch rate data could potentially act to control catches and achieve a preferred or targeted level of catch rates or mature biomass. This was a promising result given that catch and effort data are often only the primary data source available for small-scale fisheries. However, more exploratory work would be necessary to determine how robust such catch rate management strategies would be to alternative possible realities (alternative operating model structures).

In data poor fisheries where there is an array of disparate information, a strategy of weight-of-evidence may well be the most productive approach to discerning the stock status. It is critical though to have clear and unambiguous management objectives, reference limits and decision rules and these will need to be defined by fishery managers and industry.

OUTCOMES ACHIEVED

The main outcome of this project has been the development of a modelling framework that has allowed the value of different performance measures for the assessment and management of spatially structured populations to be tested. In data-poor situations basic biological characterisation, while not necessarily informative about stock status, is important in evaluating the susceptibility of a species to over-fishing. Through the application of a management strategy evaluation we have revealed that simple decision rules based on measures derived from catch rate analyses can potentially achieve a preferred or targeted level of mature biomass; at least it can avoid depletion at a large scale. In this regard, priority needs to be given to achieving a high level of commercial data quality, including appropriate spatial reporting, and the development of clearly defined management objectives and decision rules. In Tasmania commercial logbooks are being modified to include greater spatial resolution and provision for catch verification. Progress is also being made to develop more explicit performance measures for key scalefish species, including banded morwong.

This study has highlighted the difficulties of managing and assessing spatially structured fish stocks, especially those low value fisheries that are typically data poor. Of critical importance with such fisheries is the development of explicit and detailed management objectives towards which each fishery can be managed. The generic objectives common to many fishery management plans are inadequate when faced with the problems inherent in spatially complex fish stocks. The recognition of this need for specific objectives is the outcome, but the objectives themselves should be generated by the fishery managers in consultation with the fishers and other stakeholders.

In relation to Tasmania, a significant and direct outcome of this project has been a review of the management of the banded morwong fishery, with a Departmental paper recently tabled to the Scalefish Fishery Advisory Committee (August 2006) proposing the introduction of output controls (individual quotas) coupled with spatially-explicit performance indicators. This paper resulted from an industry-management forum held early in August 2006 at which the implications of the stock assessment and catch projections derived from this project were considered. Progress is also being made to develop more explicit performance measures for other key scalefish species in Tasmania and it is hoped that these will be presented as part of the 2006 Scalefish Fishery assessment report.

This project has also raised the awareness by resource managers and industry of the importance of meaningful (and accurate) spatial information relating to catch and effort. Accordingly, the Tasmanian general fishing logbook is being revised to provide for greater spatial resolution in catch and effort reporting (particularly in relation to inshore reef areas), as well as catch verification via catch disposal declaration.

KEYWORDS: Banded morwong, wrasse, stock assessment, fishers' knowledge, performance measures, management strategy evaluation.

ACKNOWLEDGEMENTS

We gratefully acknowledge the many commercial fishers who allowed us to sample their catches over the past decade and assist in the collection of biological samples. Several past and present TAFI staff, in particular Ray Murphy, Graeme Ewing, Tim Debnam and Sean Tracey were responsible for the collection and processing of much of the biological material that underpins this project.

Acknowledgements are also due to Alexander Morison and Dr David Smith for providing access to fishery and biological information for banded morwong and wrasse (FRDC Project No. 1997/128) from Victoria. Mr Morison also provided valuable comments on the project report.

The assistance of Paul Bozinis, Ken Smith and Sue Smith in collecting biological samples from Victoria, and the work of staff of the Central Ageing Facility in Queenscliff in providing age estimates for samples from Victoria, is gratefully acknowledged.

The enthusiasm and time provided by the live-fish fishers in Tasmania and Victoria to discuss their industry and knowledge of the target species is greatly appreciated.

BACKGROUND

The development of markets for live fish in the early 1990s saw a rapid expansion of fisheries based on inshore temperate reef-dwelling species off south-eastern Australia. The principal species involved are banded morwong (*Cheilodactylus spectabilis*), blue-throat wrasse (*Notolabrus tetricus*) and purple wrasse (*N. fucicola*). Gillnets are used to capture banded morwong while wrasse are targeted by line fishing and/or fish traps, with a limited by-catch from gillnets.

Banded morwong fishery

In Tasmania, banded morwong catches rose from less than 10 tonnes per annum prior to 1992/93 to a reported peak of 145 tonnes in 1994/95, although the reliability of this figure is in question. Reported catches then declined steadily to just 34 tonnes in 1999/2000. Both effort and catch rates also declined and, significantly, the mean annual catch rate in 1999/2000 was at its lowest historic level, giving rise to serious concern over the status of the stocks and sustainability of the fishery (Lyle and Hodgson 2001). Since then catches have fluctuated between 45-55 tonnes and, depending upon region, catch rates have exhibited either a slight improvement or a continued decline (Lyle *et al.* 2005).

The Victorian fishery operated at a low level (about 10 tonnes p.a.) prior to its closure in February 2000. The unusual step of closure was taken due to concerns expressed by licence-holders, resource managers and conservation groups about the long-term sustainability of the fishery. It was then re-opened in September 2000 as a Developmental Fishery and recent catches have fluctuated between 9-11 tonnes p.a. (Anon. 2004a).

Tasmania and Victoria have introduced a range of management controls, including limited entry (there are 29 licensed banded morwong operators in Tasmania and currently two permit holders in Victoria), maximum and minimum size limits and a two-month (March and April) spawning season closure. In addition, a 50 fish per day catch limit applies in Victoria. Despite extensive management intervention there remains uncertainty whether even the current reduced catch levels are sustainable. Catch and catch-rate management triggers have been exceeded for several years in Tasmania but in the absence of more robust performance measures the most appropriate management responses remain unclear.

Wrasse fishery

Tasmanian wrasse catches peaked at 180 tonnes in 1994/95 but have since ranged between 70-90 tonnes p.a. (Lyle *et al.* 2005). Victorian catches have fluctuated between 40-60 tonnes p.a. since the late-1990s (Anon. 2004a) while in South Australia there has been a growing interest in this fishery, with catches expanding to 40-50 tonnes p.a. in recent years.

Although both purple and blue-throat wrasse are targeted in Tasmania, fishers do not tend to distinguish between species in catch returns and thus the relative contribution of either species is unknown. By contrast, in Victoria blue-throat wrasse is the dominant species caught in all but the west coast region, where purple wrasse are the main species captured (Anon. 2004b). Reporting of catches at the species level remains an issue in both Tasmania and Victoria.

Management of the live wrasse fishery in Tasmania is through limited entry (there are currently 58 wrasse licence-holders) with minimum size limits currently in place. In November 2001, the minimum size limit for wrasse was increased (from 28 to 30 cm) and the maximum size limit (43 cm) removed following a review of the Scalefish Fishery Management plan. Operators without wrasse licences are restricted to a daily by-catch limit of 30 kg and are not permitted to sell the product on the live fish market. Similar arrangements apply in Victoria, with 51 Ocean (wrasse) fishery access licences issued and by-catch limits for other operators. Management of the Victorian fishery is currently under review, with excess (latent) effort identified as a major problem and consideration being given to implementing spatial management (zoning) (Anon. 2004b). Specific wrasse licences do not apply in South Australia, with all holders of Marine Scalefish licences having access to the species, indicating potential for further expansion of the fishery. Size limits do not apply in South Australia.

Biological considerations

Banded morwong and wrasse are site attached species that display evidence of population structuring at a spatial scale of individual reefs. Biologically, however, they are very different and, therefore, the populations are likely to respond differently to the effects of fishing.

Banded morwong are long-lived with maximum ages well over 80 years in Tasmania (Murphy and Lyle 1999)[1]. Slower growth rates and smaller maximum sizes for female banded morwong result in a high vulnerability of females, while the faster growth rates and large sizes attained by males mean that they grow through the fishery's keyhole size limit more quickly than females. The implications of this on the sustainability of the fishery have not been fully evaluated.

The two wrasse species are moderately long-lived (up to 20 years), with similar habits but contrasting social and sexual systems. Purple wrasse are gonochoristic (sex determination occurs very early in development) while blue throat wrasse are protogynous hermaphrodites, *i.e.* they change sex from female to male. This sex change process is a function of social structure and size or age, and by virtue of legal size limits and market size preferences male blue-throat wrasse are more vulnerable to the fishery than females. Smith *et al*. (2003) used per-recruit analyses that incorporated sex change and noted that high fishing pressure resulted in a marked reduction in the proportion of male biomass in blue-throat wrasse populations.

[1] Subsequent sampling has revealed individuals with estimated ages of over 90 years.

Implications for management and assessment

Both banded morwong and wrasse are site attached as adults and display evidence of population structuring at a spatial scale of individual reefs. As a consequence, localised over-fishing, or serial depletion, constitutes a potential risk. In such fisheries, which are small scale and based on sedentary stocks with spatially-structured populations, conventional data intensive assessment techniques cannot be justified because the cost of data collection is high, especially given the need to account for potential spatial heterogeneity of the resource. In the case of banded morwong, the long life of the fish also suggests that any stock recovery is likely to be very slow if a stock is depleted. This gives rise to the question of what biological and/or fishery indicators, if any, are appropriate and meaningful and hence what data are required to monitor performance and address issues of sustainability.

NEED

The development of live fish markets in Australia during the early 1990s created a strong demand for temperate reef species, specifically banded morwong and wrasse. Prior to this, these species had little commercial value apart from use as bait by rock lobster fishers. The new market demand has resulted in large increases in fishing pressure directed at the reef fish communities and while there has been much work on tropical species, our knowledge of how temperate reef species respond to fishing pressure is comparatively poor.

Although banded morwong and wrasse have vastly different life history characteristics (early life history, reproductive strategies, and age and growth), both are relatively sedentary and exhibit population structuring at a small spatial scale. Nevertheless, management of coastal fisheries tends to be conducted at a large scale. In Tasmania, for example, the management of these stocks relates to the whole State. The general mismatch between the spatial scales of fishery management, fishing operations and fish population structure implies there is considerable potential for localised depletion, and hence for serial depletion of the resources.

In Tasmania, steady declines in catch and catch rates for banded morwong have led to concerns that fishing has already negatively impacted stocks. In Victoria, a more controlled approach to the development of the fishery has been taken, with an initial developmental phase to be followed by a review to ascertain long-term sustainability.

Although key fishery indicators, catches and catch rates (analysed at state-wide or regional scales) have remained relatively stable in recent years for wrasse, there are anecdotal reports of localised depletions from both Tasmania and Victoria. Furthermore, because of their protogynous hermaphroditic development, the selective removal of adult male blue-throat wrasse may have the potential to impact significantly on egg production, even if female spawner biomass is adequate.

There is therefore a need to develop robust management strategies that might include stock assessments, appropriate performance measures and monitoring strategies if these species are to be managed sustainably. However, being small-scale fisheries based on sedentary, spatially-structured populations, an innovative approach to fishery and biological monitoring and data analyses is required.

OBJECTIVES

1. Develop appropriate and meaningful performance indicators for sedentary reef-dwelling species, using banded morwong and wrasse as models.
2. Determine minimum information requirements for the effective stock assessment of these species.
3. Develop, where possible, appropriate model frameworks for testing the management strategies developed for these fisheries.

CHAPTER 1: ASSESSMENT AND MANAGEMENT OF SPATIALLY-STRUCTURED, SITE ATTACHED, FISH POPULATIONS

1.1 Introduction

A fishery can be considered data poor if insufficient information is available to produce a defensible stock assessment. This definition is more than a simple statement about the quantity or quality of data and focuses instead on the purpose for which data is collected. Haddon *et al*. (2005) suggest that insufficient information for an assessment can arise because:

1. a fishery is new or developing and a time series of information has yet to be collected;
2. sufficient specific information is unavailable irrespective of how the long fishery has operated (extensive data collection is difficult to justify for low-value fisheries);
3. data collection tends to focus on the target species so bycatch or byproduct fisheries constitute another common data poor category, even though they may constitute a significant proportion of a target fishery's catch; and
4. an assumption of spatial homogeneity has been adopted for fisheries that are spatially or otherwise structured.

Recognition of the last category is poorly represented in the published literature but is potentially a major reason for fisheries to be classified as data poor. For a stock to have spatial structure implies that it is made up of a large number of inter-connected but relatively independent and geographically separate sub-populations (one form of meta-population). Where a stock has spatial structure, simple data collections from one or two locations may not be representative of the whole and thus could not validly be used to assess the whole (*e.g.* Orensanz and Jamieson 1998; Parma *et al*. 2003). A spatially-structured fishery seems likely to cause extra problems when few data are available, because under such circumstances formal stock assessments are typically limited to simple models. Unfortunately, there do not appear to be any simple or standard spatial models that can be used in stock assessment. Furthermore, spatial structuring of a stock can occur in many ways, making it difficult to produce a simple solution to the complexities present in the class of fisheries studied here.

Many reef fish populations demonstrate spatial structuring, with limited movement between reefs after settlement (*e.g.* McCormick and Choat 1987; Jones 1988; Newman and Williams 2001). Banded morwong, blue-throat wrasse, and purple wrasse are such species, supporting small but recently developed fisheries in south-eastern Australia. Banded morwong has been the subject of a previous research project (Murphy and Lyle 1999), and of more recent fishery-independent catch samples, which have provided biological data on size and sex composition, age and growth, and maturity. In Tasmania and Victoria the only data generally available for the finfish fisheries are the legally required commercial catch and effort data, which are collected relative to an array of statistical reporting blocks. As such, banded morwong constitute a relatively data-rich species among a sea of data poor fisheries and because of the availability of this data, the species was chosen as an example temperate reef fish with which to examine a range of alternative management strategies. In particular, the objective of this study was to test an array of performance measures for fisheries, which usually only have commercial catch and effort statistics, sometimes of questionable integrity, plus limited and occasional collections of biological information. One of the primary questions was to determine whether an occasional collection of such biological data could be used to improve attempts at stock assessment.

1.2 Stock Assessment Methods

For the purposes of this report formal stock assessment is taken to mean mathematical modelling of stock dynamics in a manner aimed at determining the status of the stock and whether current catch or effort levels are sustainable. This does not include analyses such as yield-per-recruit, which although useful do not ensure sustainability.

The development of formal stock assessment methodologies has followed a number of divergent paths. Early work on age-structured stock assessment models was produced by various Russian scientists (Baranov 1918; Derzhavin 1922; both in Ricker 1971). Indirectly this led to the development of what became statistical Catch-at-AGE ANalysis (CAGEAN) also known as Integrated Analysis and Virtual Population Analysis (VPA) (Beverton and Holt 1957; Paloheimo 1958; Murphy 1965; Gulland 1965 – in Garrod 1967). This in turn led to the simpler Cohort Analysis (Pope 1972). All of these methods were inherently deterministic and each cohort was treated as a completely independent entity. This meant that no indication of any uncertainty associated with the analysis was produced (Megrey 1989).

Non-deterministic analyses only became possible when fishing mortality was represented as the combination of an age-specific and year-specific coefficient. This separation of the two mortality components (the so-called separability assumption), made it possible to reduce the total number of parameters defining a model and opened the way to modern statistical Catch-at-age analysis. Doubleday (1976) provides a description of the separability assumption but credits a number of unpublished manuscripts for the idea (Megrey 1989). The introduction of the “separability” assumption set such stock assessment models into an optimization framework using least squared residuals and eventually maximum likelihood, and more recently still Bayesian statistics. This permitted and led to stock assessments that included estimates of the uncertainty associated with the predicted outcomes of each analysis.

A second path in the history of stock assessment methodologies led to formal stock assessments that related to the total stock dynamics and were relatively simple line fitting exercises that assumed each fishery was in equilibrium with the stock being fished (Schaeffer 1954, 1957). Unfortunately, in situations where catch rates fall as stocks decline, the assumption of equilibrium biases estimates of sustainable catches high. In the 1950s and 1960s, stock assessments were expected to generate estimates of safe levels of catch and/or the effort required to attain that catch. Importantly, the notion that these estimates were uncertain and might be extremely imprecise was essentially ignored. Such models include the simpler equilibrium surplus production models and original yield-per-recruit models. It is quite possible that these overly simple and biased models contributed to a number of highly publicised fishery collapses.

The first important advance came from moving away from the assumption that fished stocks were necessarily deterministic or in equilibrium with the exploitation they were experiencing. This shift, along with the advent of the more complex, often age-structured models discussed above, was a major improvement but the aim was still to produce a single estimate of the optimal catch or effort levels that would drive the fishery to its optimum level of production. Little or no attention was paid to the issue of uncertainty in the estimation of the measures that were the basis of management advice.

Eventually, in the 1980s and early 1990s, methods were developed that led not only to estimates of parameters with management value (such as permissible catch levels) but permitted estimates of the precision of the estimates (one way of describing uncertainty) (Fournier and Archibald 1982; Pope and Shepherd 1982; Deriso *et al*. 1985). Unfortunately, at the time, many management organizations were unaware of how to use management advice couched in terms that included estimates of precision. One worry for assessment scientists became that management decisions would be made that related more to the upper or lower bounds of estimates of catch and effort instead of the median values taking into account the relative precision of the estimates. There was a protracted debate about how best to characterize uncertainty with the questions centring on whether to use Bayesian or Frequentist methods (Maximum Likelihood methods). This discussion was well intentioned but not especially productive. The key outcome was the recognition of uncertainty (Francis 1992). Because this wasn't being recognized by management agencies at that time, rather than debate how best to characterize uncertainty more effort should have been directed at informing the managers on how best to use the uncertain estimates.

The next major step forward was the development of "Risk Assessments" aimed at determining the implications of applying a particular harvest strategy into the future (Francis 1992). This entailed developing methods for projecting stock assessment models into the future under constant management conditions but possibly stochastic recruitment. A key development in risk assessments was in producing probabilistic statements regarding the likelihood of different outcomes (*e.g.* in five years time there is an X% chance that the spawning stock biomass will be greater than at present if catches are restricted to Y tonnes), which led to improvements in the understanding of risk and uncertainty in stock assessments.

Management strategy evaluation (MSE) is the latest step in the evolution of single species stock assessment methodology. This is not a new way of assessing fished stocks but rather an approach used to compare the relative effectiveness of different management strategies (which include data collection, stock assessment, and decision rules for how to use the outcomes from the stock assessment). Management strategy evaluation (Punt 1992; De la Mare 1996; Butterworth *et al*. 1997; Smith *et al*. 1999; Punt *et al*. 2002) will be discussed in more detail below (see Chapter 7).

While there is always room for new stock assessment methods, the principal problem now is deciding which stock assessment method to apply in a particular situation. There are stock assessment methods that range from the simple to the very complex and each has its set of assumptions and its requirements for data. Having said that, it must be noted that there are no standard stock assessment models available that include detailed spatial structure as a major component. There have been numerous instances where spatial structure has been included in stock assessments but there is no general method available for doing this. Generally, the inclusion of spatial details in any assessment is limited by the availability of information. With data poor fisheries this is an even bigger problem and spatial structure is usually ignored.

1.3 The Scale of Data Aggregation in Stock Assessment Models

Most formal stock assessment methods involve some form of mathematical model of the dynamics of an exploited species and each has a set of assumptions. Even the simplest of line-fitting models make assumptions about what constitutes a meaningful scale of aggregation of data. For example, non-equilibrium surplus production models (Polacheck *et al*. 1993; Haddon 2001) aggregate all age-classes and individuals into one group and just follow the dynamics of total stock biomass through time ignoring any spatial details.

The decision concerning which model or models to use with a given fishery is affected by a multitude of factors. For data poor fisheries, the most obvious factor is the availability of information. If the age-structure of the catch is not available then age-structured stock assessment models can still be used but fitting data to such models must become, by definition, indirect. Thus, an age-structured surplus production model can be fitted solely to catch and catch-rate data but there would only be value in doing this if there were ancillary data such as length-frequency information or occasional age-structure information to add to the base data.

The types of data used for stock assessment comes in many forms and includes:

- Catches and catch rates (ideally standardized) used in most stock assessment models;
- Length-distribution of catch – can be used in age- and stage-based models; not an index of relative abundance;
- Age-structure of the catch – can be used in age-structured models; not an index of relative abundance;
- Tagging data – depending on tagging survey design can be useful for selectivity, growth, movement, stock size, and harvest rate;
- Survey estimates of relative or absolute biomass; and
- Discards.

The types of fish stock assessment models that are available include:

- Surplus production models (both simple and age-structured);
- Depletion models (similarities with surplus production models, tend to relate to single years but can be multi-year);
- Age-structured models (CAGEAN, VPA, Integrated Analysis; follows cohorts);
- Delay-difference models (simplified age-structured models, knife edge selectivity); and
- Stage-based models (emphasizes growth and recruitment, used with species that cannot be aged).

In the Tasmanian fishery for banded morwong the only data available are unverified commercial catch and effort reports and some occasional fishery independent observations of sex ratio, size distribution, and age-structure from, at most, three locations along the east coast. It is always possible to summarize the available data and consider any trends visible. However, when the spatial structuring assumed to occur in the natural population is considered, the possibility of drawing valid general conclusions about the stock is compromised. The primary problem of concern is that the commercial catch effort data is available only with half degree (30 by 30 nm) statistical block resolution and the fishery independent sampling is very sparse.

Serial depletion is often difficult to detect in large-scale data because it can occur despite stable catch rates. Thus, if fishers deplete a small area to a point where their catch rates seriously decline, they move their operation to regain better catch rates. Such movements only need to be relatively minor depending on the degree of site attachment of the fish species and may remain hidden in the coarse scale of reporting. Until depletion levels become extreme, catch rates in fisheries prone to serial depletion become more dependent upon fisher behaviour than upon stock size. Serial depletion of sub-populations can be a real problem in site-attached fish such as banded morwong and wrasse, but at present there is no way of detecting this other than verbal reports from fishers.

1.4 The Dynamic-Pool Assumption

Ideally and traditionally, all stock assessment models operate at the level of particular biological stocks (Russell 1931). As with many concepts in ecology and particularly in fisheries, the definition of the term "stock" is by no means clear. In the context of stock assessment models, a stock is considered to be a self-contained reproductive unit or "a group with unimpeded gene flow" (Pitcher and Hart 1982). Each major stock is mainly isolated from every other stock (Harden-Jones 1968). However, often the term *stock* is used simply as a convenient reference to the management unit being considered. As Royce (1984, p.215) put it: "….*stock* is used to mean an exploited or management unit. It will be obvious that the ideal definition of *stock* is that of a single inter-breeding population, but this condition is so rarely demonstrable, either because of scanty data or because of the rarity of isolated interbreeding populations, that *stock* must be more or less arbitrarily defined."

In general, stock assessment models assume that immigration and emigration are in balance and that the recruitment dynamics being described relate to a single stock. There are examples where the implications of this assumption of a unified stock have been explored (Punt 2003) but such examples are rare. In the case of banded morwong, the species has a relatively long pelagic larval life of approximately 6 months. This is long enough for significant mixing around the coast of Tasmania. Thus, while adult sub-populations may be relatively discrete due to behavioural reasons of site attachment, the prolonged mixing assumed to be experienced by the larval phase would ensure a single genetic population. Banded morwong around Tasmania (and possibly beyond) are thus likely to form a single biological stock.

Unfortunately, associated with the assumed stock structure and pervading all non-spatial stock assessment models is the assumption that each stock forms a dynamic pool. This means that removals from one part of the stock will have an impact on the productivity of the whole stock. Time-lags between the removals and their effects may be introduced, but whenever the impact is felt it is felt evenly across the whole stock. In effect, the dynamics of the stock are not affected by the geographical distribution of the component individuals; they all contribute to the same dynamic pool. This dynamic pool assumption denies the possibility of serial or local depletions occurring within a single stock.

The dynamic pool assumption is a reasonable approximation to reality when a fishery relates to a species that is not site attached and moves freely around its geographical distribution, when fishing effort is distributed across the full range of the stock, and when the time scale over which the fishery is assessed is long relative to the movement dynamics of the fish and the fleet. Unfortunately, whenever these conditions are not met, as for example with banded morwong and other site attached reef species such as wrasse, then standard stock assessment models are likely to be inappropriate. In principle, it would be possible to generate a stock assessment model with sufficient spatial detail to capture the real dynamics of such stocks. However, in practice, the geographical scale required would be so fine that with current data collection methods it would not be possible to characterize the state of the fishery adequately and any assessment could be at best only approximate and at worst, by ignoring localised depletions, completely misleading.

The dynamic pool assumption is a serious though generally disregarded flaw in current stock assessment models. The assumption is most seriously inconsistent with spatially-structured populations of site attached species, and by ignoring spatial structure any stock assessment model is likely to be overly optimistic in its predictions. It would be incapable of detecting serious declines in stock abundance until they became extreme and would increase the risk of stock damage and even stock collapse.

1.5 Management Objectives for Spatially-Structured Stocks

Managing the exploitation of living marine resources is best attempted if the management is aimed at achieving one or more explicit objectives. Unfortunately the objectives of fisheries management are often only stated in broad and imprecise terms and where more than one objective is given they can even be inconsistent. It is common to include objectives aimed at including the notion of sustainable exploitation, economic efficiency, and, less commonly, social aims such as avoiding the concentration of access rights into only a few hands. For example, the management of all commercial scalefish fisheries in Tasmania falls under the Tasmanian Scalefish Fishery Management Plan (DPIF 1998). There are no formal targets defined for the fishery, and the objectives towards which the fishery is managed are only loosely stated. The stated purpose of the Living Marine Resources Act is to achieve sustainable development of living marine resources but in effect, the management objectives are stated in such a manner that their interpretation is unclear:

- Maintain biomass and fish recruitment at sustainable levels by restricting the level of fishing effort that can be applied by the scalefish fishery;
- Sustain yield and economic returns by optimising the yield and/or per-unit value from fish being recruited to the fishery;
- Mitigate any adverse interactions that result from competition between different fishing methods or sectors, and interactions with other users of the marine environment;
- Maintain or provide reasonable access to fish stocks for recreational fishers;
- Minimise the environmental impact of scalefish fishing methods;
- Ensure economic returns to the community; and
- Ensure the production of the highest quality, hygienic product for local and international markets.

Numerous strategies have been adopted in attempts to achieve such objectives. To measure the success of these different strategies requires that the objectives decided upon need to be more explicit and precise in their meaning. For example, the notion of sustainable exploitation or sustainable development remains obscure even today because there are so many possible interpretations that can be placed on the unqualified term “sustainable”. In practice, it would appear that the real interest is in maximizing the yield from a fishery while retaining the ability to take a harvest well into the future; often with the secondary aim of generating stable catches each year. The primary aim of fishery management appears to be to define a fishing operation or management strategy that attains a high or maximal level of profits while maintaining the stocks that give rise to the fishery at levels that ensure the continuance of the fishery. The old, flawed idea of the maximum sustainable yield attempted to capture this notion but failed to provide the means of achieving the aim (Larkin 1977). The primary problem for fisheries management has been to achieve this optimization in the presence of high levels of uncertainty.

Fishery management objectives effectively identify the outcomes desired for and from each given fishery. Of course there are many different ways of achieving the same outcome when managing natural resources. And the problem of deciding on a particular management strategy becomes even more complicated when the available information is limited.

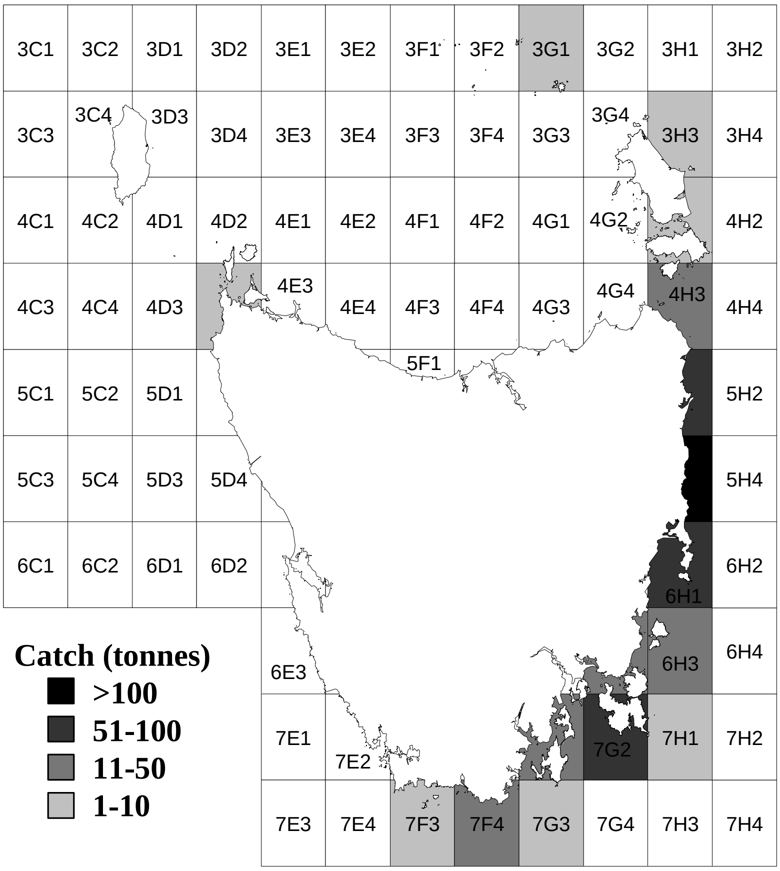

Fig. 1.1: Map of Tasmania with 30 nm fishing blocks and total catch summed over the period 1995/96 to 2003/04.

Most management objectives ignore spatial structuring. Thus, while there may be an objective that includes or requires ecological sustainable development, this provides no useful guidance on how to manage a spatially-structured fishery appropriately. Once the geographical distribution of the stock and of fishing effort are considered, the possible outcomes are much more numerous and complex. The potential complexities can be illustrated with the example of banded morwong in Tasmanian waters (Fig. 1.1). While the species is reported as being distributed in all areas around Tasmania, including the west coast (Yearsley *et al.* 1999), commercial catches have been mainly taken from the east coast. The relative isolation, regular rough weather, and difficulty of transporting live fish across such distances undoubtedly limit successful fishing for banded morwong off the west coast. When the notion of sustainability is considered in the context of the spatial distribution of the fishery, then what is meant exactly by sustainable becomes unclear. With the long larval phase it is possible that even if the majority of the east coast mature biomass was removed, the stock could recover from the spread of larvae from unfished areas on the west coast (though recovery would most likely take many years and possibly decades). The question becomes one of whether adult populations in all areas must be fished sustainably or are other options acceptable.

The possibilities include banded morwong to be fished sustainably:
1. at the scale of Tasmania (implying that depletions at the scale of the 30 nm fishing blocks and below could occur);
2. at the level of the east and west coast (implying that within the east coast depletions at the scale of the 30 nm fishing blocks and below could occur);
3. at the level of each of the 30nm fishing blocks (implying that depletions at the scale of populations within a block could occur);
4. at the level of populations within each 30 nm fishing block.

The four strategies with their specific requirements for spatial management are precautionary to different degrees and not all can be realised. Strategy 1 which manages the fishery at the largest geographical scale of the whole of Tasmania, could hardly be said to be precautionary, since it completely neglects the issue of serial depletion. But this strategy is probably the one that best characterizes the current management of banded morwong in Tasmania. On the other hand, strategy 4 is the most precautionary, but with the current scale of data collection it would be impossible to manage banded morwong at the scale of individual populations. Only the first three management strategies could be put into practice.

While, in principle, assessment scientists should not pass public judgement on the appropriateness of management objectives, the exact objective towards which it is desired to manage such fisheries needs to be elucidated before sound stock assessment advice can be produced. The forgoing discussions and arguments clearly identify the lack of policy development with regard to spatial structured fish stocks. Assuming it is accepted that the government is responsible for setting objectives to manage their fisheries, then it is not acceptable or appropriate for these objectives to be vague or ill-defined. It must be recognized that the classic fishery performance measures that use catches and catch rates relate to fisheries having far less complex spatial dynamics. Until there has been debate and decision concerning the objectives for the management of spatially structured coastal reef fisheries, the great uncertainty involved with such fisheries implies that the only responsible response by scientists is to produce highly precautionary advice to managers.

CHAPTER 2: DESCRIPTION AND APPLICATION OF PERFORMANCE MEASURES

2.1 Introduction

The aim of this chapter is to present an overview of the performance measures and reference points commonly used in fisheries around the World. Many of the methods described require extensive sets of commercial and/or biological data and are therefore not suitable for data-poor situations that typically occur in new, developing or small-scale fisheries. This means that the range of suitable assessment models and appropriate management advice applicable to these fisheries is severely restricted. We also summarise performance measures and risk analysis associated with uncertainty in parameter estimates, model processes, and model structure.

The second part of this chapter examines current examples of stock assessment, performance measures and reference points applied to selected fisheries from around the World, focussing on small-scale fisheries for medium to long-lived and potentially spatially-structured species.

Since this project aims to explore the minimum stock assessment and suitable fishery performance measures for the management of data-poor fisheries, we concentrate on a single-species approach. Recently there have been increasing calls for fisheries assessments to move towards multi-species and ecosystem models rather than remain with single species models (Garcia and Cochrane 2005). However, the complexity reached in ecosystem models is much greater than with single-species approaches, and their data requirements are typically also much greater. Thus, multi-species and ecosystem modelling are not viable options when attempting to generate management advice in data-poor situations.

2.2 Fishery Harvest Strategies

Fisheries management systems typically consist of a management cycle for decision making based on results from formal stock assessments, which are interpreted through the use of a set of, possibly implicit, decision rules. This management cycle is often annual and ideally is set within a policy framework with defined management objectives, reference points, fishery performance measures and management reviews (*e.g*. Sainsbury *et al*. 2000; Punt *et al*. 2001a; Punt *et al*. 2001b).

In this context, performance measures and reference points have been defined as (after Caddy and Mahon 1995; Sainsbury *et al*. 2000; Fig. 2.1):

Performance indicator: A quantity that can be measured and used to follow changes in the status of a system component that is assumed to relate to sustainability; *e.g*. the ratio of the current spawning biomass to the unexploited spawning biomass;

Performance measure: The value of a performance indicator in relation to a reference point that can be used to measure management performance against an objective; *e.g.* the probability that the stock is currently above the reference point.

Reference point: The value of a performance indicator that corresponds to some agreed trigger for management action; *e.g.* 35% of unexploited biomass B_0;

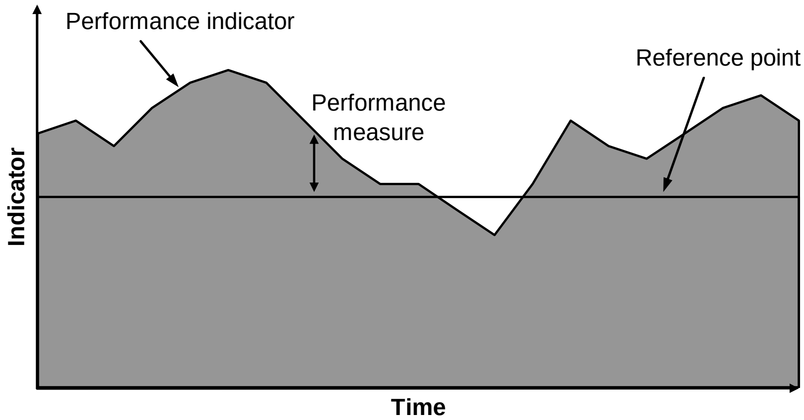

Fig. 2.1: Schematic graph showing the relationship between performance measures, target reference points, and limit reference points (after Sainsbury *et al*. 2000). Because the reference lines in this diagram are precise this fails to capture the uncertainty in any estimate.

Fishery performance indicators are therefore those statistics that permit a formal determination of whether, and to what extent, the fisheries management applied to a stock has achieved the objectives set for the fishery. However, a fishery statistic only then becomes a performance indicator once detailed reference points have been defined and thus performance can be measured. This is very loosely handled in the fisheries literature where almost any statistic is described as performance indicators without a formal definition of reference points. It could be argued that the distinction between performance indicator and measures bears the danger of confusing the differences between a simple fishery statistic and a performance indicator.

Commercial wild fisheries may be managed in a range of different ways to achieve stated management objectives. Each different way is often referred to as a harvest strategy and relates to whether management is attempting to control either fishing effort or catch. So-called “input” controls attempt to manage directly any quantity or quality of fishing effort. Such controls include regulating the duration of the fishing season, closed areas, gear restrictions, vessel restrictions, fishing license restrictions, and related mechanisms (Morison 2004). The alternatives make up “output” controls, which are aimed at limiting catch directly and these include any kind of quota or catch limit and qualitative measures such as size limits. The harvest strategies that have been identified generally relate to these two alternative approaches to management, although variations are possible which combine facets of both.

There are three main categories of harvest strategies:

1. Constant Catch Strategy

As is obvious from its name, such approaches try to control the annual catch taken from a stock. This is the strategy behind fisheries managed by quotas; long-lived relatively predictable species are best suited to this kind of management.

2. Constant Fishing Mortality

This is based on the idea of taking a constant proportion of the available biomass each season. Obviously, this requires knowledge of how much biomass is available and is somewhat less sensitive to life-history characteristics.

3. Constant Escapement Strategy

This harvest strategy relies on allowing a constant biomass to not be caught (to escape fishing mortality); the amount is that deemed to be required to generate the required recruitment in subsequent years. Because of variation in total available biomass neither the yield nor the harvest rate can be expected to be constant under a constant escapement strategy. This strategy has been used with some salmon and squid stocks.

In many fisheries the management in place is insufficient to control either effort or catch. These fisheries can be considered effectively open access even though they may not fit the classic definition. This has always led eventually to unsustainable practices and cannot be considered as a real option for management. For example, the Tasmanian banded morwong fishery is limited by 29 fishing licenses and additional controls. While only about half this number of licenses are fully active, all of the 29 fishers or licensees could choose to fish. This effectively means that the current catches and effort are not limited by management but rather by external factors such as markets.

2.3 Performance Indicators

Assuming there are objectives towards which a fishery is being managed, it should be possible to determine how well the management is performing relative to those objectives. Any measurable quantity or statistic relating to the operation of a fishery may be used as a measure of performance; ideally such measures will relate directly to an explicit objective or target.

Yield Y, fishing mortality F and stock biomass B form the basis of most fishery performance measures and reference points related to the target species (Caddy and Mahon 1995) and are, at a given time t, related to each other through the classic catch equation:

$$Y_t = H_t B_t \quad \text{or} \quad Y_t = qE_tB_t$$

where q is catchability and E is fishing effort. The harvest rate H_t is related to the instantaneous fishing mortality rate F by:

$$F = -\text{Ln}(1-H).$$

Thus, the control of fishing mortality or yield (catch) influences stock biomass and vice versa. Many reference points are based on measures of fishing mortality, yield and biomass; *e.g.* the fishing mortality which produces on average the maximum sustainable yield (MSY); the fishing mortality which maximises the yield-per-recruit; or the spawning stock biomass which produces a desired level of recruitment. There is another suite of economic reference points, such as Maximum Economic Yield, which will not be explored here.

Specifying the performance measure as either a fishing mortality rate or biomass level for the stock has a number of advantages and disadvantages (Table 2.1). Because of this, using a combination of a maximum fishing mortality, a biomass level below which the maximum allowable fishing mortality rate is reduced, and an absolute minimum biomass limit is now implemented in many fisheries to provide protection against overfishing (Rosenberg and Restrepo 1996).

Table 2.1: Advantages and disadvantages of performance measures based on fishing morality *F* and biomass levels (after Rosenberg and Restrepo 1996).

	PM based on fishing mortality *F*	PM based on biomass level *B*
Advantages	• *F* relates to the act of fishing and therefore can be controlled directly by management. • There is theoretical and empirical basis for the selection of maximum *F*, *e.g.* for recruitment over-fishing. • Only relatively sparse fishery data and life-history characteristics information on the stock is required. • A maximum *F* can prevent the stock from depletion due to fishing in the long term. • In fisheries near the limit, a maximum *F* can be gradually reduced, avoiding a total closure of the fishery.	• The biomass is more directly linked to recruitment than *F*. • Minimum biomass levels provide a guide for management of stocks that are already depleted. • During periods of adverse environmental conditions, a minimum biomass level provides a seed stock for possible recovery when conditions are more favourable.
Dis-advantages	• A maximum *F* that is appropriate over a middle range of biomass levels may not be appropriate at biomass extremes, *i.e.* no increased protection when the stock is in poor condition and no development of full fishery potential if estimated with data from a period of light fishing. • *F* may require adjusting when environmental conditions and life-history characteristics are changing.	• Specifying minimum biomass levels is often difficult and data-intensive. • The lack of observations over the full range of stock conditions can cause the minimum biomass to be poorly estimated. • Closing a fishery below a biomass limit and opening it above results in highly variable fishery yields and economic problems.

In many situations alternative performance measures are needed because neither fishing mortality nor stock biomass can be estimated due to insufficient data. If a logbook system is in place, trends in catch per unit effort (CPUE) can be used as a proxy for stock biomass. This approach relates catch rates to stock biomass assuming that catchability q is constant or known to vary in a particular manner. Biological population parameters including the mean size or age and the proportion of mature animals, can also serve as performance measures. But in the absence of a formal stock assessment, their meaning in the context of stock status may be uncertain particularly with confounding effects on such parameters arising from recruitment variability. Thus, while any management advice produced when using biological performance measures may be designed to move the fishery towards an agreed target, there is no guarantee that this will ensure sustainability.

Other performance indicators relating to non-target species and bycatch, or environmental issues such as area fished or damage caused to the habitat by the fishing gear, are rarely useful in data-poor fisheries. Often, these indicators would be even more difficult to quantify than population parameters of the species themselves.

2.4 Reference Points

Reference points specify a certain value of a performance measure that represents a state of the fishery or the population that is believed to be useful for the management of the stock in question (FAO 1993). Clearly defined management objectives are therefore very important in order to establish reference points as part of any set of control rules (Caddy and Mahon 1995; Caddy 1998; Caddy and Cochrane 2001).

Reference points can relate to an absolute value (*e.g.* $F_{0.1}$, see below for definition), a hypothetical virgin state of the population (*e.g.* $F_{35\%}$), or to a value of the performance measure in some reference years (*e.g.* $F_{1972\text{-}85}$). In cases where a trend in some performance measure is being considered, *e.g.* an observed decline in catch rates between years, there can be some implied limit reference point defining what is considered to be an unacceptable rate of decline (*e.g.* Lyle *et al*. 2005). For such indirect performance measures even if a workable target reference point is defined (*e.g.* catch rate in 1985), there would be no guarantee that the target would necessarily imply the stock was being fished sustainably.

Reference points are commonly placed into two categories relating to their function either as target or limit reference points (Caddy and Mahon 1995):

Target Reference Point (TRP): indicates the value of a performance measure that corresponds to some agreed limit of the fishery and/or resource which is considered to be desirable and at which management action, whether during development or stock rebuilding, should aim.

Limit or Threshold Reference Point (LRP): indicates the value of a performance measure that corresponds to some agreed target of the fishery and/or resource which is considered to be undesirable and which management action should avoid.

Early usage of reference points tended to provide imprecise statements so that a LRP might have been to keep the instantaneous fishing mortality rate below F_{MAX}, while a TRP might have been to obtain a spawning stock biomass that would give rise to the maximum sustainable yield (Y_{MSY}). Attention is now paid, however, to the uncertainty inherent in stock assessments so that now a limit reference point might be that there is a probability of <10% that the ratio of the current spawning biomass to the unexploited spawning biomass is below 20%; a target reference point might be a 50% probability that the ratio of current spawning biomass to the unexploited spawning biomass is greater than 50%.

Management approaches based on a single target reference point (TRP) or limit reference point (LRP) have proven vulnerable to overfishing once a reference point has been exceeded. This has often been at least partly because of uncertainty in estimating the status of the fishery relative to the reference point. It is now recommended that a combination of TRPs and LRPs are used in a system of explicit harvest control rules (Caddy and Mahon 1995; Rosenberg and Restrepo 1996; Caddy 1999; Hilborn *et al*. 2002). These harvest control rules should specify a maximum fishing mortality rate for a stock in healthy condition, some strategy for reducing fishing mortality as biomass falls below the target level of stock biomass, and a lower absolute biomass threshold below which fishing must be stopped. In such a system, the TRP plays the role of a ‘first line of defence’, below which fishing intensity is reduced to minimise the probability that the LRP will be reached.

The choice of what reference points to apply in the management of a particular fishery depends on several factors. Reference points which are based on estimates that include a wider range of information about the stock are often considered to be more robust and therefore more effective in achieving their goal than reference points which are based only on little information. However, often the available data about the fish species and stocks limits the range of possible reference points and relatively simple reference points have to be chosen. Sometimes less precise or even empirical reference points are to be preferred if they are better understood by industry and management. In data-poor fisheries when the status of the fishery and the stock is highly uncertain and information sources are insufficient to reliably estimate fishing mortality and biomass levels, a combination of multiple alternative reference points relating to different aspects of the stocks and fishing mortality may be ‘safer’ than applying a single consideration (Caddy 1999). Under these circumstances, management advice can be produced using a weight-of-evidence approach that considers all sources of available information and whether each source indicates the stock status is healthy or not.

A variety of reference point criteria have been proposed for the sustainable exploitation of fish stocks. They refer to maximum yield, maximum biological production, yield-per recruit, spawner-per-recruit, the stock-recruitment relationship, alternative biological population parameters, and combinations of the above. Comprehensive reviews can be found in Caddy and Mahon (1998), Caddy (1998), and Gabriel and Mace (1999).

While some reference points relate to commercial catch information or biological data alone, many, including some MSY measures and yield-per-recruit analyses, need either an independent estimate of stock biomass or fishing mortality to be fully useful. Because such estimates are often difficult to obtain when there is little data and can introduce additional uncertainty into the assessment process, such approaches may not be useful in data-poor fisheries.

2.5 Maximum Yield Targets

If the target of a fishery is maximizing the yield or catch, this target is ideally estimated by use of a stock assessment model. This can be even a simple assessment such as that produced by a surplus production model, which should be possible in many data-poor situations. However, where no reliable catch rate information is available, approaches that do not require a formal assessment and only consider catch histories can still lead to defensible advice about catch targets (Annala 1993). The terminology of yield targets tends to be couched in terms that relate to fishing mortality targets which are aimed at achieving a particular stock size that should, in theory, give rise to a particular yield. This practice should not obscure the fact that yield targets or recommended catch levels can be based almost purely on reported catch data, with the uncertainties inherent is such voluntary data.

The concept of Maximum Sustainable Yield (MSY) was widely used as a target reference point for single species fisheries by fisheries managers in the 1960s and 1970s; it was even included in the fisheries legislation of many countries (*e.g.* New Zealand's Fisheries Act 1983). MSY represents the highest point of the classic surplus production curve describing the relationship between the annual fishing effort by the fishing fleet and the yield that should result if that particular fishing mortality level were maintained until equilibrium were reached (Schaefer 1954, 1957). The Schaefer models are based on the idea that maximum fish harvest can be extracted at the population level where population growth is fastest. However, the original models assumed equilibrium conditions, *i.e.* each level of fishing effort has an equilibrium sustainable yield or surplus production, and if the fishing effort is changed the stock is assumed to move immediately to a different stable biomass with is associated surplus production. Based on these unrealistic assumptions, equilibrium models are no longer recommended in stock assessment.

To overcome the equilibrium assumption, process and observation error estimators have been developed for non-equilibrium surplus-production models (Punt 1990, 1995; Hilborn and Walters 1992; Polacheck *et al*. 1993; Haddon 2001). Applying both error types simultaneously and in a completely flexible way has proven to be difficult. Using observation errors has advantages to focussing on process errors since they appear to better reflect the circumstances underlying the observations (Ludwig and Walters 1989). Simple non-equilibrium surplus production models can be applied with limited data. For example, Lyle *et al*. (2005) used such a model to describe the population dynamics of southern calamary off the East Coast of Tasmania when there were only seven data points. While management advice with respect to target harvest rates was produced to achieve a desired level of spawning biomass, it was admitted that the analysis was prone to extreme uncertainty and, in reality, the management guidance provided by such simplistic assessments should be considered critically.

Due to uncertainty in the estimation of MSY, brought about by recruitment variability and by asymmetrical biological responses to overfishing and 'underfishing', there is a high risk of stock overfishing if MSY is used as a Target Reference Point (TRP). Instead, the use of MSY as a LRP has been recommended and allows for choosing a more cautious TRP (Caddy and McGarvey 1996). The FAO Code of Conduct (FAO 1995) indicates that MSY conditions should now be regarded as representing a limiting condition that should not be exceeded when speaking in terms of fishing mortality (*i.e.* remain below F_{MSY}) or allowed to fall below when speaking in terms of stock size (*i.e.* keep the stock above B_{MSY}). In fact, MSY conditions are now considered to constitute the minimum conditions for stock recovery (Caddy 1998).

When used as targets F_{MAX} and F_{MSY} have been seriously criticised as sustainability measures (Larkin 1977; Sissenwine 1978; Hilborn and Walters 1992). As an alternative to F_{MSY} Doubleday (1976) suggested that fishing at the effort level corresponding to $^2/_3$ of the effort that gives rise to MSY ($\frac{2}{3}F_{MSY}$) would be more sustainable. While the risk of stock collapse would be significantly reduced, still around 80% of the MSY could be harvested. Although a more conservative measure, $\frac{2}{3}F_{MSY}$ is somewhat arbitrary and the same estimation uncertainties apply to it as for F_{MSY}. Simulations of Mace (1988) and Francis (1992) indicate that the appropriate factor to multiply MSY varies between about 0.6 and 0.9 depending on various parameters, particularly the steepness of the assumed stock-recruitment relationship.

In New Zealand fisheries, the notion of MSY has been interpreted as Maximum Constant Yield (MCY) and Current Annual Yield (CAY). These are used as more conservative target reference points than the classic MSY (Annala 1993). MCY is a constant catch strategy similar to MSY and is based on the idea of taking the same catch from a fishery year after year (Fig. 2.2). MCY is defined as the maximum constant catch that is estimated to be sustainable with an acceptable level of risk at all probable future levels of biomass; the notion was developed to complement the introduction of a quota management system into New Zealand fisheries. MCY implies much lower levels of fishing mortality and catch than MSY and depends to a certain extent on the current state of the fish stock and the recruitment variability of the species. In developed fisheries with adequate data to fit a population model, $\frac{2}{3}F_{MSY}$ has been suggested as a suitable approximation to F_{MCY} (Annala 1993). However, as shown in Fig. 2.3, it might be better to use $F_{0.1}$.

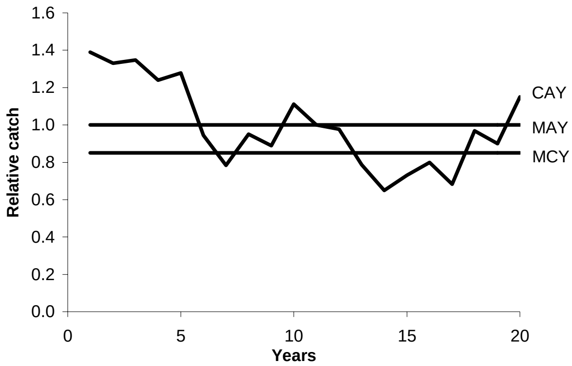

Fig. 2.2: The relationship between Maximum Constant Yield (MCY), Current Annual Yield (CAY) and Maximum Average Yield (MAY) (after Annala *et al.* 2001)

The Current Annual Yield (CAY) strategy applies a constant fishing mortality rather than a constant catch, and can be seen as a dynamic interpretation of MSY. Because fish populations fluctuate in size from year to year, it is necessary to alter the catch every year to get the best yield from a fishery and to keep the fishing mortality constant. CAY is the one-year catch level calculated by applying a reference fishing mortality F_{ref} to an estimate of the fishable biomass present during the next fishing year. F_{ref} is the level of instantaneous fishing mortality that, if applied every year, would maximise the average yield from the fishery and is often set equal to $F_{0.1}$ (see below). The Maximum Average Yield (MAY), another interpretation of the MSY concept, is obtained by applying a CAY over a number of years and thus is the maximum long-term annual catch (Annala *et al*. 2001). Without relatively precise knowledge of stock biomass, the MCY strategy is probably the best practical method of maximising average yield (Table 2.2).

Alternative possibilities for providing advice about yield or catch limits exist in the absence of sufficient data to run even a simple model (Annala 1993). Insufficient data includes cases where the available data exhibits only minor or no trends through time so that simple models would be inappropriate. Under such circumstances stock assessment models require a great deal of biological information, such as age-structure through time, and related material. Instead, if only catch and effort data are available then management advice can be produced assuming the average catch over a relatively stable period in the fishery is representative of what the stocks can sustain (Table 2.3).

Table 2.2: Estimation of yield reference points requiring at least a simple stock assessment model to provide a biomass estimate.

Assessments:	
MSY = $0.5MB_0$ or xMB_0	Gulland and Boerema 1973; Gulland 1983
MSY = $c_1B_0 = c_2B'_0$	Beddington and Cooke 1983
MSY = $0.5ZB_c = 0.5(Y_c + MB_c)$	Cadima, in Troadec 1977
MSY = $rK/4$	Equilibrium and non-equilibrium models
MCY = $0.25F_{0.1}B_0$ or MCY = $0.25MB_0$	Beddington & Cooke 1983; Mace 1988; Francis 1992; Annala 1993
MCY = $0.5F_{0.1}B_{av}$ or MCY = $0.5MB_{av}$	Annala 1993
CAY = $\dfrac{F_{ref}}{F_{ref}+M}(1-e^{-(F_{ref}+M)})B_{beg}$	Annala 1993
Parameters: M = natural mortality; c_1, c_2 = constants; Z = total mortality: Y_c = current yield; $F_{0.1}$ see below (yield-per-recruit); F_{ref} = reference fishing mortality; B_0 = total virgin biomass; B'_0 = virgin recruited biomass; B_c = current biomass; B_{av} = average historic recruited biomass; B_{beg} = biomass at start of the year; r = population growth rate; K = maximum population size	
Advantages: Relative simple methods.	
Disadvantages: Some methods with severe assumptions including equilibrium conditions. Often low accuracy or great uncertainty in estimates.	
Comments: The common approximation of MSY = $0.5MB_0$ by Gulland and Boerema (1973) has been shown to often overestimate MSY (Beddington and Cooke 1983; Getz *et al*. 1987; Mace 1988). Gulland (1983) proposed an alternative equation with the constant x that can be related to the growth and mortality characteristics of the stock. Beddington and Cooke (1983) proposed a modification of Gulland's generalized model distinguishing biomass estimates referring to the total virgin biomass B_0 from those referring to the virgin recruited biomass B'_0. The constants c_1 and c_2 are provided for given values of age at recruitment, natural mortality M and the von Bertalanffy growth parameter K. The MSY estimator proposed by Cadima's (in Troadec 1977) is unbiased only when either the stock is virgin (and the estimator becomes equivalent to Gulland's estimator for virgin stocks), or the stock is fished at MSY at the time of the biomass survey (Garcia *et al*. 1989). The MSY of non-equilibrium models can be interpreted as an average, long-term expected potential yield if the stock is fished optimally The MCY estimates are more conservative than the MSY estimates. The appropriate factor to multiply $F_{0.1}B_0$ or MB_0 may be somewhat higher or lower than 0.25 (Beddington and Cooke 1983; Mace 1988; Francis 1992) and depends primarily on the steepness of the assumed stock-recruitment relationship. The MCY based on B_{av} (Annala 1993) is estimated over several years of historic biomass. The CAY returns the highest yield of the fishery while recognising fluctuations in stock size, since catches are altered every year (Annala 1993). But the Baranov catch equation is data-intensive since it requires an estimate of the stock biomass at the beginning of the fishing year.	

Table 2.3: Estimation and yield reference points without production models.

Assessment: MCY = cY_{av} Annala (1993)
Parameters: Y_{av} = average catch; c = natural variability factor
Assumptions: Selected yield measures are representative for the status of the stock. Catches may have to be standardised for changes in catchability.
Advantages: Simple method. Estimates of yield are often available.
Disadvantages: Assumes near-equilibrium conditions. Selection of an appropriate period may be difficult. Estimates of natural mortality and maximum life expectancy of the fish may not be available in a data-poor situation. May be unsuitable for long-lived species .
Comments: Familiarity with stock demographics and the history of the fishery is needed to determine the appropriate period on which to base the estimates of Y_{av}. The period chosen will depend on the catch, fishing mortality F and fishing effort time series, the management regime, and the lifespan of the species. The period should not contain systematic changes in catch, F or effort, and be equal to at least half the exploited lifespan of the fish. The natural variability factor c ranges between 0.6 and 1.0 and is based on computer simulations which showed that the MCY for a stock is inversely related to the degree of natural variability in its abundance (Mace 1988). It is assumed that a stock with higher natural mortality will have fewer age-classes and suffer greater fluctuations in biomass. Therefore, the higher natural mortality and variability in stock size, the lower the value for c (for values see Annala *et al*. 2001). If the catch data are from a period when the stock was fully exploited (*i.e.* fishing mortality near the level that produces MAY), the method should provide a good estimate of MCY. In this case, $Y_{av} \approx$ MAY. If the population was under-exploited the method gives a conservative estimate of MCY. Due to its relatively parsimonious data requirements this is a suitable method when catch and effort data lacks contrast and thus no production model can be fitted. It is widely used in stock assessments in New Zealand (Annala *et al*. 2001).

Maximising the overall yield combines all aspects of population productivity, recruitment, growth and mortality, but ignores details such as age and size at recruitment. This has led to the use of analytical, age-structured models based on detailed population dynamics. The simplest type of these models is the yield-per-recruit (YPR) analysis which uses information on average individual growth, natural mortality and vulnerability to fishing (Beverton and Holt 1957).

Initially, F_{MAX}, the level of fishing mortality for a given size at first capture that maximises the average yield from each recruit entering the fishery, has been used in stock assessments (Fig. 2.3). However, for many species there is no clear maximum to the curve of YPR against fishing mortality. There seems also little doubt that F_{max} generally exceeds F_{MSY} substantially except for strong density-dependence in the population, and that fishing at this rate over an extended period of time will deplete the spawning stock and reduce future recruitment (Deriso 1982; Clark 1991; Mace 1994). Under certain circumstances, F_{MAX} may be useful as a limit reference point (Mace and Sissenwine 1993; Mace 1994).

In practice, instead of F_{MAX}, F_{MSY}, or even ⅔F_{MSY}, many fisheries around the world are now being managed using an $F_{0.1}$ strategy (pronounced "F zero point one"; Hilborn and Walters 1992). The value of $F_{0.1}$ is determined numerically by finding the fishing mortality rate at which the slope of the YPR curve is 10% of the slope at the origin (Fig. 2.2). It should be noted that the $F_{0.1}$ strategy is *ad hoc* and has no theoretical justification except that empirically it appears to be more conservative or risk averse than F_{MAX} to departures from the assumptions of the yield-per-recruit analyses. For a relatively small loss in yield, one often gains a great deal in stock resilience to poor recruitment years and other sources of uncertainty. It may be the case that even the $F_{0.1}$ strategy is insufficiently conservative and this can only be determined by further experience with more fisheries. Hilborn and Walters (1992, p. 461) concluded that: "$F_{0.1}$ policies may be one of the most significant changes in fisheries harvesting practice since the earlier widespread acceptance of MSY. They are significant not because of any theoretical breakthrough, or any intrinsic elegance, but simply because they provide a replacement for F_{MAX} and MSY and appear to often be robust".

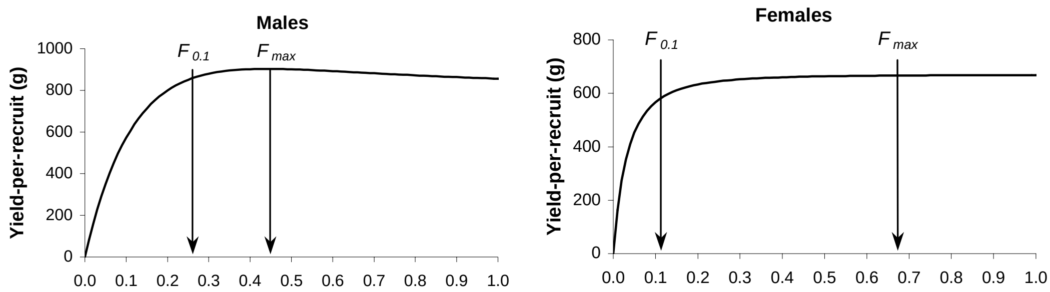

Fig. 2.3: Yield-per-recruit curves for male and female banded morwong (after Lyle *et al*. 2004). Note that for males $F_{0.1}$ is approximately the same as ⅔F_{MAX}, whereas for females it is only about 0.18F_{MAX}. Doubleday's (1976) suggestion may have worked for males but not for females. This indicates an advantage of $F_{0.1}$ over either ⅔F_{MAX} or ⅔F_{MSY}.

The benefits of the inherent simplicity and low data requirements of the methods based on production models are reduced by severe restrictions in their valid use. Precision, accuracy and appropriateness under a range of circumstances remain to be tested for many of these methods. There are numerous assumptions implied when such simple methods are used and these are often ignored. In short, care is needed in the use of such methods.

The most complex and data-intensive reference points refer to the stock-recruitment (S-R) or spawning biomass-per-recruit (SPR) criteria. They address the relationship between the spawners or the eggs they produce and the recruitment of young fish to the fished or spawning population and are based on the idea that successive generations must replace each other on average in order for fish stocks to persist. Recruitment overfishing is therefore understood to occur when the spawning biomass produced on average by a year class is lower than the spawning biomass of its parents.

To determine the optimal rate of exploitation, the stock-recruitment relationship and the current stock and/or fishing mortality status needs to be known. The stock-recruitment criteria will therefore rarely be useful in data-poor situations. Shepherd (1982) is generally credited with the development and application of SPR reference points, and extensive discussions of SPR can be found in Sissenwine and Shepherd (1987), Sissenwine *et al.* (1988) and Mace and Sissenwine (1993).

SPR is usually expressed as a percentage of the SPR under unfished conditions (*i.e.* at virgin spawning biomass B_0) and is designated as %SPR or %SSBR. The fishing mortality which produces any particular %SPR is designated as $F_{\%SPR}$ or just $F_{\%}$, *e.g.* $F_{30\%}$.

2.6 Other Targets

The total mortality Z experienced by a stock due to all causes of death can be used as reference point without being separated into fishing and natural mortality. Unfished populations are dominated by relatively large and old individuals, whose contribution to biological production (growth, yield, and deaths due to predation) is lower than that of younger individuals. These younger fish are expected to gradually come to dominate a population with increasing exploitation. Thus, there may be a mortality level F_{MBP} or Z_{MBP}, at which the Maximal Biological Production (MBP) is obtained from the stock (Caddy and Csirke 1983). The MBP can be estimated using a time series of catch per unit effort data applied in a Schaefer production model assuming equilibrium conditions.

Given a sample of numbers-at-age in commercial catches it is also possible to apply a classic catch curve analysis (Beverton and Holt 1957) to obtain an estimate of total mortality Z. If a catch curve is used and there is an estimate of the natural mortality of the species, the ratio of natural to fishing mortality has been suggested as a potential LRP. As a rough guide it is often stated that F should not be allowed to become greater than two times M (Gulland and Boerema 1973).

2.7 Comparison of Reference Points

Target reference points: For northern hemisphere demersal fish, Clark (1991) showed that a yield of at least 75% of MSY is possible as long as the spawning biomass is maintained between 20-60% of the unfished level. This can be achieved by a fishing mortality that will reduce the spawner per recruit (SPR) to around 35% of the unfished level, *i.e.* $F_{35\%}$. Clark (1991) found that this fishing mortality will often be close to M and $F_{0.1}$, when recruitment and maturity coincide. A slightly lower fishing mortality rate of $F_{40\%}$ was later recommended to reduce the frequency of episodes of low spawning biomass, especially with serially correlated recruitment variation (Clark 1993). $F_{40\%}$ is the fishing mortality that will lead the stock eventually to attain a spawning stock size that is 40% of the unfished spawning stock. Mace (1994) recommended that $F_{40\%}$ be adopted as a target fishing mortality, but that it be adjusted to accommodate any known or assumed degree of density dependence in the S-R relationship. Especially for long-lived stocks with low resiliency, such as some of the Pacific Coast rockfishes *Sebastes spp.*, the $F_{40\%}$ strategy can result in very low levels of recruitment. A higher target for SPR should then be adopted, *e.g.* $F_{50\%}$ or $F_{60\%}$, though obviously with some reduction in yield (Clark 2002).

A robust alternative harvest strategy for fish stocks with relatively constant equilibrium abundance would be to use a purely biomass-based strategy that maintains spawning biomass at around 40% of the unfished level (Clark 1991). This strategy is resilient against uncertainty in the S-R relationship and parameter estimates of natural mortality but it can be difficult to determine the unfished biomass. Furthermore, quota recommendations will be sensitive to changes in estimates of unfished and present biomass.

Limit reference points: Mace and Sissenwine (1993) advocated the use of 20% SPR as a recruitment overfishing limit for North Atlantic stocks believed to have average resilience, and at least 30% SPR for little-known stocks. Mace (1994) supported these recommendations and suggested that these results may also be applied to stocks outside the North Atlantic. These studies complement earlier theoretical and empirical work by Goodyear (1977, 1980, 1989, 1993), Gabriel *et al*. (1989) and Clark (1991) that has resulted in estimates of spawner-per-recruit becoming the most common basis for recruitment overfishing reference points in U.S. fishery management plans. Most definitions use either 20% SPR or 30% SPR as the recruitment overfishing reference point (Rosenberg *et al*. 1993).

2.8 Simple Performance Measures

In many cases, lack of informative data prevents the explicit estimation of reference points related to YPR or SPR. As alternative options yield targets proxies can be developed that are based on the type and quality of data that are available and which relate to levels from reference years (Table 2.4). Proxies for fishing mortality include the mean or median size or age in catches and measures of fishing effort, those for biomass include changes in commercial catch per unit effort (CPUE) or research surveys. The currently observed levels can be compared to target levels derived from years where stocks were believed to be only lightly exploited.

The choice of suitable reference levels or reference periods and changing rates is often contentious and arbitrary. It depends not only on the knowledge and understanding of the fishery and biological parameters of the stocks, but also on negotiations between management, industry and research. Of course, if a period has been selected erroneously as being representative for light exploitation, perhaps because of under-reporting of catches, then it is likely that the stock will suffer stronger than intended depletion. Such effectively arbitrary reference periods may need to be regularly reviewed and confirmed.

Due to their inherent simplicity these performance indicators and reference points need to be treated with care. Results from single assessments may be misleading and ambiguous. For example, a decreasing mean size could indicate either high fishing mortality, strong recruitment pulses, or non-representative data. Ideally, a combination of different measures should be used instead (Caddy 1999), using many data sources simultaneously. In addition, trends across time should be considered more valuable than single observations.

Table 2.4: Alternative targets and reference points with few data requirements.

* *Catch, effort and catch per unit effort (CPUE)*
Assessment: Catch, effort or CPUE do not exceed or fall below limits set based on reference years or a pre-defined rate of change within a given time period.
Data requirements: Time series of catch and effort data. Catch rates should be standardised for changes in catchability related to changes in fishing efficiency or effort distribution.
Advantages: Estimates of catch and effort are often available.
Disadvantages: Catch and catch rates are often not representative for the status of the stock due to changes in catchability and fishery. Indeed, especially for highly spatially-structured stocks they can often be uninformative or misleading.
Comments: Due to the limited data requirements, this method is frequently chosen when catch and effort data lack contrast and thus no production model can be fitted. However, the analysis is often of limited value on its own and should be used in combination with other criteria.

* *Mean or median size*
Assessment: Mean or median size in catches does not fall below a reference size.
Data requirements: Time series of size-frequency distributions in current and reference catches. Preferably also an estimate of size at maturity, or a YPR / SPR analysis.
Advantages: Can be quite easily determined by market sampling, and could be used as an indicator for the current fishing mortality in relation to F_{max} or $F_{0.1}$.
Disadvantages: It requires unbiased data on catch size-frequency distribution (high-grading and discarding can be a problem). The interpretation of results can be ambiguous, *e.g.* a decreasing mean size can indicate strong recruitment or high fishing mortality.
Comments: If size at maturity is constant under different levels of fishing pressure, the mean size criterion can be related to the size at first maturity, or a YPR / SPR analysis. *E.g.* the exploitation rate should be such that the average size of fish caught is greater than the average size at maturity to allow a minimum amount of spawning (Caddy and Mahon 1995).

* *Weight at size or age at maturity*
Assessment: Weight at size or age at maturity in catches does not change compared to a reference range.
Data requirements: Time series of weight, size, age and maturity of individuals in catches.
Advantages: This approach targets the spawning stock biomass.
Disadvantages: Changes in weight at age and size at maturity are probably only detected when stock size changes dramatically (*e.g.* from lightly to highly-exploited stocks), and large sample sizes are needed to detect these effects. Changes may be very subtle and gradual with a considerable time lag. Maturity may be difficult to determine, especially outside the reproductive season.
Comments: Observed changes could also be caused by processes other than density-dependence, *e.g.* environmental changes.

Table 2.4 continued.

* *Proportion of mature individuals in catches*
Assessment: Proportion of mature individuals in stock does not fall below agreed percentage of mature fish in the catch.
Data requirements: Time series of frequencies of mature and immature individuals in catches.
Advantages: This approach targets the spawning stock biomass.
Disadvantages: Depletion of large or old fish can go undetected, *i.e.* size or age-specific increase of fecundity is not accounted for. Maturity may be difficult to determine, especially outside the reproductive season.
Comments: Only applicable if recruitment to fishery is at a smaller size than onset of maturity.

* *Sex ratio*
Assessment: Sex ratio in catches does not change compared to a reference level.
Data requirements: Time series of sex ratios in catches.
Advantages: Often simple to determine.
Disadvantages: Biology of the species must be well understood to interpret results, as they can be ambiguous. Only applicable if the species exhibits sex-specific catchability due to different sizes or behaviour between males and females.

2.9 Performance Measures, Uncertainty and Risk Assessment

Target reference points, limit reference points, uncertainty, and risk assessment are all important for the appropriate use and implementation of performance measures in fisheries stock assessment and management. An extensive literature exists on the issues involved in developing and using these assessment and management tools (see reviews by Caddy and Mahon 1995; Rosenberg and Restrepo 1996; Francis and Shotton 1997).

When performance measures and the current status of a fishery are estimated in a stock assessment, it is desirable that the uncertainty about each estimate be quantified and used to analyse the risk, *i.e.* to calculate the probability of achieving the desired or undesired outcome. In such an analysis, the risk should also include a measure of severity of the undesirable event (*i.e.* ideally there would be a low risk of very undesirable events occurring).

Uncertainty in fishery stock assessment derives from different sources (Caddy and Mahon 1995; Francis and Shotton 1997), including:

- measurement uncertainty of observed quantities such as the catch or biological parameters;
- process uncertainty, such as underlying stochasticity in the population dynamics, *e.g.* the variability in recruitment;
- model uncertainty by mis-specification of the model structure; this is most often ignored despite being of fundamental importance.
- estimation uncertainty, such as inaccuracy and imprecision in the estimation of abundance or fishing mortality rate;

- implementation uncertainty, *e.g.* from the inability to exactly achieve a target harvest strategy by a specific management policy; and
- institutional uncertainty, from the lack of well-defined social, economic and political objectives in fisheries management, or problems arising from interaction between individuals and groups contributing to the management process.

The process of dealing with risk in fisheries management has two distinct stages; firstly formulating advice for fisheries managers through stock and risk assessment; secondly dealing with the ways in which managers use that advice to make decisions (Francis and Shotton 1997). Risk assessment is usually performed by applying a stock assessment model to observed commercial and biological observations. Based on the measurement and process uncertainty of the input data, the likelihood of an outcome and the associated risk, *e.g.* probability that the stock remains above a specified threshold level $P(B_{curr} > B_{threshold})$, can be estimated by Monte Carlo simulations and other resampling techniques such as bootstrapping. Thus, the performance measure summarizes all information and is typically statistical in nature, *i.e.* relates to probabilities, expected values, standard deviations etc.

Typically in data-poor fisheries, estimates of fishing mortality or stock biomass are associated with high levels of uncertainty. This increases the risk of overfishing *i.e.* overshooting the reference level, or conversely reduces the acceptable level of the current fishing mortality. To be precautionary and to reduce the risk of overshooting the limit fishing mortality, any recommended target fishing mortality should be located at a lower fishing mortality rate when uncertainty is great due to the poor quality of the available information (Caddy and Mahon 1995). However, the definition of an acceptable risk remains difficult and should depend to a certain degree on the specific fishery. Some precautionary generalisations may be made, such as the definition proposed by Francis (1993) in which the level of harvesting is considered safe if the spawning stock biomass is above a desired stock at least 90% of the time.

Risk management should include a definition of the quantities to be estimated by the risk assessment as well as clearly-defined pre-negotiated decision rules given the results of this assessment. Following the precautionary approach as outlined in the FAO Code of Conduct for Responsible Fisheries (FAO 1995), it has been found useful to define decision rules (also named harvest control rules) with one or more LRPs or a pair of LRP and TRP (Rosenberg and Restrepo 1996). Once one or a series of LRPs indicate overfishing, a pre-agreed management action is triggered. This is maintained or reinforced until the resource recovers to a specified level, after which the exploitation rate may be increased slightly. The management action should be automatic and is ideally agreed to in advance by the resource users and their representatives. Where there is inadequate data for establishing decision rules, setting a pair of LRP and TRP can be an arbitrary procedure. It may be possible to set a single LRP that corresponds to serious but not catastrophic conditions and then base a TRP on estimates of variance and probability of overshoot (Caddy and McGarvey 1996).

2.10 Stock Assessments, Performance Measures and Reference Points for Data-Poor Fisheries

The particular stock assessments and reference points used in fisheries around the world generally reflect the availability of data for each fishery. If data sources are limited, simple catch and effort analysis for trends or, with enough contrast in the data, surplus-production models represent common assessment tools.

Many coastal reef fisheries around the world are comparable to those for banded morwong and wrasse in south-eastern Australia, *i.e.* they are small-scale, multi-species, often exploit spatially-structured stocks, and are typically data-poor. Examples of the characteristics of selected fisheries from Australia, New Zealand, South Africa, USA and the Mediterranean including information on the life history of the species, the assessment method and reference points are summarised in Table 2.5. The selection focuses on mainly resident temperate reef fish species with medium to long life spans of greater than 15 years.

Analysis of commercial catch and effort data is the assessment method for many coastal fisheries around Australia, mainly due to the lack of data other than catch and effort. In Tasmania, most scalefish fisheries are assessed through similar measures of trends in catch and effort irrespective of their species specific life-history characteristics such as growth parameters and stock dynamics. Unreliable commercial information can create problems when interpreting the assessment. A further problem is the fact that fishery objectives, reference points, and decision rules leading to management actions are often not specified.

Surplus-production models are applied in some fisheries where longer time series with contrast in the data are available, *e.g.* for snapper in Shark Bay, Western Australia. Due to the high value of snapper, the stock assessment in New Zealand is generally more comprehensive, yet variable between eight individual stock assessment regions, reflecting the quantity and type of fishery and biological data available. The assessment and estimation of the Maximum Constant Yield (MCY) ranges from a corrected measure of the average yield, to a surplus-production model and finally a detailed statistical catch-at-age structured model (Annala *et al*. 2001). The latter includes a risk assessment with model projections of the stock development into the future.

Simple catch and effort analysis or an assessment of (unspecified) biological indicators is also performed for some US and Mediterranean coastal fisheries, where insufficient data restricts more comprehensive assessments. Surplus-production models have proven to be disappointing as stock assessment method for most Mediterranean fisheries, because of, *inter alia*, the lack of time series and contrast in the data, and the multi-species character of most fisheries (Oliver 2002).

Where possible, catch and effort information is replaced by YPR and preferably SPR analyses. South Africa and the USA use a SPR analysis as the basis for defining recruitment overfishing reference points. In the South African line-fishery, 40-50%

SPR and 25-39% SPR are common TRPs and LRPs, respectively. In the USA, most definitions use either $F_{30\%}$ or $F_{40\%}$ as TRP for recruitment overfishing and 20-30% SPR as LPR for overfished biomass (*e.g.* snapper and grouper in the South Atlantic and Gulf of Mexico). For fish species with low resilience such as rockfish, the TRP can be higher, *e.g.* $F_{50\%}$. These reference fishing mortality and biomass levels are typically not determined by a S-R relationship for the particular stock, but are based on theoretical work (*e.g.* Goodyear 1977, 1980, 1989, 1993; Gabriel *et al*. 1989; Clark 1991). Assessments of current fishing mortality or stock biomass levels remain therefore the main problem. The size and age composition of the catch is often used to estimate fishing mortality by catch curve analysis, but due to spatial structuring of the fished population this method rarely provides comprehensive and representative samples.

Only in relatively few cases, such as the recreational icon species tautog (*Tautoga onitis*) in the USA, is enough data generated from fishery-dependent and independent surveys to conduct a VPA. In some fisheries of the Western Mediterranean, Length Cohort Analysis (LCA), as a simplification of the VPA, is combined with the simple catch and effort analysis and represents currently the most widely used method. The availability of reliable data, however, remains the primary shortcoming in the all these stock assessments.

2.11 Other Fisheries

Assessment methods applied to coastal reef fisheries are generally representative of managed fisheries in general, *i.e.* similar assessments and reference points are applied to all fisheries within a country or management area depending on the availability of necessary data (Table 2.6).

In most States of Australia, analysis for trends in catch and effort is the first approach, followed by surplus-production models, YPR, SPR and statistical catch-at-age population models. There are generally no legal requirements for a minimum stock assessment or performance of the fisheries as there are in the USA (see below). Correspondingly, reference points and management actions are rarely defined or pre-negotiated, even for the large South East Fisheries (SEF) within Commonwealth waters (although for the SEF this is in the process of changing).

This contrasts with fisheries in New Zealand, South Africa and the USA. In New Zealand, fish stocks are assessed following a strict schema that defines the assessment given a certain range of available commercial and biological data. The estimation of Maximum Constant Yield (MCY) is defined for new fisheries, for fisheries with an estimate of historical biomass, for fisheries with yield estimated with or without a surplus-production model, and for fisheries where a stochastic population model can be fitted (Annala *et al*. 2001). While the latter is the only method that allows some specification of the risk associated with an MCY, its implementation requires further development for the definition of stock collapse and what probability of collapse is acceptable. The same was required in fisheries for anchovies and sardines in South Africa for which a management strategy evaluation has been conducted (Cochrane *et al*.

1998). However, as in Australia, complex stock assessment models are uncommon in South Africa and most other fisheries lack a comprehensive assessment.

In the USA, the Magnuson-Stevens Fishery Conservation Management Act (amended by the Sustainable Fisheries Act; NMFS 1996) requires that every exploited species be assessed. The recruitment overfishing reference points are defined in such a way that the rate or level of fishing mortality should not jeopardise the capacity of a fishery to produce the MSY on a continuing basis. While for well-known species, SPR levels for target and limit reference points are usually defined around $F_{30\%}$ as TRP and 20% SPR as LRP, the legal requirement becomes a problem for stocks with insufficient data available to conduct SPR analysis. Trends in catch and effort, average recent catch levels as an approximation of MSY, and various biological indicators are then used instead.

The management advice for the ICES fisheries has made extensive use of reference points related to YPR (Hildén 1993). During the 1970s and 1980s, fisheries advice was largely based on F_{MAX} and $F_{0.1}$. In 1987, two additional reference points F_{med} and F_{high} with mainly indicative character were introduced. F_{med} is the fishing mortality, at which the stock on average replaces itself, while F_{high} is defined as the level of fishing mortality that leaves 90% of the recruitment points in a stock-recruitment plot above the line through the origin. Hildén (1993) investigated the success and failure of the application of these reference points based on the management recommendations for 54 stocks given in 1978 and 1990. There were about equal numbers of successes and failures and a considerable number of status quo, however, the effectiveness of the reference points were difficult to distinguish from other fisheries management objectives, such as simply reaching consensus, maximising catches, or short-term economic considerations. Thus, while reference points like F_{MAX}, $F_{0.1}$ and F_{med} have been often referred to and found to be useful, they are difficult to implement. Since 1998, ICES has managed stock and fisheries with B_{pa} and F_{pa} (*pa* stands for precautionary approach) which incorporate error in the estimation of *F* and SSB (ICES 2002). If a stock is estimated to be above B_{pa} or below F_{pa}, then there is a high probability that it will be above a limit B_{lim} or below a limit F_{lim}, the safe biological limits for the stock and fishery.

The Commission of the Conservation of Antarctic Marine Living Resources (CCAMLR) has used $F_{0.1}$ as one of the first elements in its management policy for finfish species (see CCAMLR's Approach to Management on http://www.ccamlr.org), but for some species now uses escapement of the spawning stock as the criterion to determine the allowable level of fishing mortality. Stochastic stock assessment models are in place called Generalised Yield Model (GYM) for finfish or Krill Yield Model (KYM) for krill, accounting for estimates of current or pre-fishing biomass, effects of previous catches on the stock recruitment fluctuations and uncertainty in demographic parameters. This allows using model projections to evaluate the effects of different catch levels on stock biomass.

Table 2.5: Overview over stock assessments and reference points of fisheries for temperate coastal reef fish species.

Fishery	Life history	Location	Assessment	Reference point	Source
Banded Morwong (*Cheilodactylus spectabilis*) Purple Wrasse (*Notolabrus fucicola*) Blue-throated Wrasse (*Notolabrus tetricus*)	Temperate reef Long-lived (>15y) Resident *C. spectabilis*: Gonochoristic *N. fucicola*: Gonochoristic *N. tetricus*: Protogynous	Australia, TAS	Catch and effort analysis Age and size composition of catch YPR (if possible)	Trigger point for management action: Catch, effort and CPUE levels from reference years Catch rate change over 30% Significant change in size composition Unacceptably high bycatch	Tasmanian Scalefish Fishery Policy Document (Anon. 1998)
		Australia, SA	Catch and effort analysis, size/age composition of catch, YPR (if possible)	Unspecified	South Australia Fisheries Act 1982 (Anon. 1982)
		Australia, VIC	Catch and effort analysis, size/age composition of catch, YPR, SPR (if possible)	Unspecified	Parliament of Victoria: Inquiry into Fisheries Management - Second Report (Environment and Natural Resources Committee 2002)
Snapper *(Pagrus auratus)*	Temperate Long-lived (>20y) Coastal migrant with spawning aggregations	Australia, SA	Catch and effort analysis, size/age composition of catch, YPR	Unspecified	South Australia Fisheries Act 1982 (Anon. 1982)
		Australia, VIC	Catch and effort analysis, size/age composition of catch, YPR, SPR	Unspecified	Parliament of Victoria: Inquiry into Fisheries Management - Second Report (Environment and Natural Resources Committee 2002), Coutin (2000) Snapper
		Australia, NSW	Catch and effort analysis, size/age composition of catch, YPR	Unspecified	NSW Fisheries Status Report 2001/2002 (Kennelly and McVea 2003), Fishery Management Strategies Performance Report 2004 (Anon. 2004c)
		Australia, QLD	Catch and effort analysis	Unspecified	Queensland fisheries resource: current condition and recent trends 1988-2000 (Williams 2002)
		Australia, WA	Surplus production model	TRP: $F = F_{MSY}$	Status of the Fishery Report 2002/2003 (Penn *et al.* 2003)
		New Zealand	Surplus production model YPR Age-structured population model in certain regions (using total catch, CPUE, growth, natural mortality, length-weight, selectivity, recruitment indices, SST, catch at age, tagging estimates of absolute biomass)	TRP: - If $B < B_{MSY}$: MCY = CSP, CSP = current surplus production If $B > B_{MSY}$: MCY = MSY, If no estimate of B: MCY = cY_{av} - CAY with F_{max} if estimate of B Model projections: Stock increase $P(B_{2020} > B_{1999})$ Stock rebuild $P(B_{2020} > B_{MSY})$ Expected status $E(B_{2020} / B_{MSY})$	Annala *et al.* (2001)

Table 2.5 continued.

Fishery	Life history	Location	Assessment	Reference point	Source
Linefisheries for *e.g.*: Silver kob (*Argyrosomus inodorus*) Seventy-four (*Polysteganus undulosus*) Red steenbras (*Petrus rupestris*)	Temperate reef Long-lived (>20y) Resident or coastal migrant	South Africa	SPR for reference points Age/length key, size composition of catch, or mark-recovery for estimates of *F* Assessment every *x* years (x = 0.5 age_{max})	TRP: 40% SPR LRP: 25% SPR	South African Linefish Status Reports (Mann 2000), Linefish Management Protocol (Penney *et al*. 1997, Griffiths *et al*. 1999)
Rockfish (*Sebastes spp.*)	Many temperate Most long-lived (>20y) Relative resident	USA, West coast	Estimates of *F* and *B* from SPR or formal stock assessment when available Otherwise biological indicators	TRP: F = F50% LRP: F = F25% or 50% of BMSY or (unspecified) biological indicators	NOAA's Report to Congress on the Status of U.S. Fisheries - 2001 (NMFS 2002); California's Living Marine Resources: Status report (Leet *et al*. 2002)
Sheephead wrasse (*Semicossyphus pulcher*)	Many temperate Long-lived (>20y) Mainly resident Protogynous	USA, West coast	Catch and effort analysis or biological indicators	TRP: MSY = 0.5Yav (for data-poor species)	NOAA's Report to Congress on the Status of U.S. Fisheries - 2001 (NMFS 2002), Pacific Fisheries Management Council (1998)
Tautog (*Tautoga onitis*)	Temperate Long-lived (>20y) Small seasonal migration Gonochoristic	USA, East coast	VPA	TRP: F = F40%	Fishery Management Plan for Tautog (Stirratt *et al.* 2002)
Snapper, Grouper etc. *e.g. Epinephelus spp.*	Temperate / subtropical Often long-lived (>20y) High site fidelity with spawning aggregations Many protogynous	USA, South Atlantic USA, Gulf of Mexico	Estimates of *F* and *B* from SPR or formal stock assessment when available Otherwise biological indicators	South Atlantic: TRP: F = F40% to F30% LRP: 30% SPR Gulf of Mexico: TRP: F = F40% to F30% LRP: 20% SPR	NOAA's Report to Congress on the Status of U.S. Fisheries - 2001 (NMFS 2002)
Mediterranean coastal fisheries (*e.g.* Mullet *Mullus spp.*)	Temperate Many long-lived	Mediterranean	Catch and effort analysis, YPR, Length cohort analysis (LCA) / VPA Direct estimates of *B* from acoustic, egg production and trawl surveys	Unspecified	Oliver (2002), Lleonart (1999)

Table 2.6: Overview over stock assessments and reference points of selected fisheries around the world.

Location	Fishery	Assessment	Reference point	Source
Australia, Tasmania	Marine scalefish and squid	Catch and effort analysis	Trigger point for management action: Catch, effort and CPUE levels from reference years Catch rate change over 30% Significant change in size composition Unacceptably high bycatch	Tasmanian Scalefish Fishery Policy Document (Anon. 1998)
Australia, SA	Marine scalefish	Catch and effort analysis, size/age composition of catch, YPR (if possible)	Unspecified	South Australia Fisheries Act 1982 (Anon. 1982)
Australia, Victoria	Marine Scalefish	Catch and effort analysis, size/age composition of catch, YPR, SPR	Unspecified	Parliament of Victoria: Inquiry into Fisheries Management - Second Report (Environment and Natural Resources Committee 2002), Coutin (2000) Snapper
Australia, NSW	Marine Scalefish *e.g.* - Trap and Line fishery - Estuary finfish and ocean haul	Catch and effort analysis, size/age composition of catch, YPR	Unspecified	NSW Fisheries Status Report 2001/2002 (Kennelly and McVea 2003), Fishery Management Strategies Performance Report 2004 (Anon. 2004c)
Australia, QLD	Coral reef line fishery and rocky reef line fishery	Analysis of catch and effort	Unspecified	Queensland fisheries resource: current condition and recent trends 1988-2000 (Williams 2002)
Australia, WA	Marine Scalefish *e.g.* - West and south coast Estuarine Fisheries - Shark Bay scalefish - Northern demersal	Analysis of catch and effort, surplus production model, YPR, SPR	LRP: Catch, effort and CPUE levels from reference years If YPR: $F = F_{0.1}$ If SPR: $F = F_{30\%}$ TRP: If production model: $F = F_{MSY}$	Status of the Fishery Report 2002/2003 (Penn *et al.* 2003)
Australia, NT	Coastal Line and Net Fisheries	Analysis of catch and effort, preliminary age-structure and stock reduction analysis	Unspecified	Zeroni (2003)
Australia, Commonwealth waters	(a) Patagonian toothfish, blue grenadier (SEF) (b) SBT, orange roughy and eastern gemfish (SEF), school shark (c) Broadbill swordfish, 14 other species of SEF, deepwater flathead and bight redfish (GAB)	(a) Formal stock assessment (b) Formal stock assessment (c) Analysis of catch and effort	(a) TRP: 40% SPR LRP: 20% SPR (b) Rebuilding stocks towards TRP of 30% or 40% SPR (c) Catch, effort and CPUE levels from reference years, catch in relation to TAC	Status of fish stocks managed by the Australian Government (Caton and McLoughlin 2004).

Table 2.6 continued.

Location	Fishery	Assessment	Reference point	Source
New Zealand	Marine Scalefish	Catch and effort analysis, surplus production model, YPR, SPR, age-structured population model	TRP: (a) New fisheries: MCY = $0.25F_{0.1}B_0$ or $0.25MB_0$ (b) Exploited fisheries with estimates of historic B: MCY = $0.5F_{0.1}B_{av}$ or $0.5MB_{av}$ (c) Fisheries with surplus production model: MCY = 2/3MSY If $B_{curr} < B$ to sustain 2/3MSY: MCY = 2/3CSP (CSP = current surplus production) (d) CPUE without surplus production model: MCY = cY_{av} (c = natural variability factor) (e) Stochastic population model Where adequate stock biomass data: CAY from Baranov catch equation with *e.g.* $F_{0.1}$	Annala *et al.* (2001)
South Africa	(a) Linefisheries (b) Anchovies and sardines	(a) SPR; and age/length key, size composition of catch, or mark-recovery for estimates of F (b) Operating model	(a) TRP: 40-50% SPR LRP: 25-39% SPR (b) Combination of: - P(SB < 20% SB_0 within any 20y period) < defined risk level - Average annual catch within reference levels - Interannual catch variability within reference levels - Average SB levels over 20y above reference levels	South African Linefish Status Reports (Mann 2000) Linefish Management Protocol (Penney *et al*. 1997, Griffiths *et al*. 1999), Cochrane *et al*. (1998)
USA	All marine scalefisheries	(a) Estimates of F and B from formal stock assessment *e.g.* VPA (few species) (b) Biological parameters where no quantitative analysis possible (most species) (c) Catch and effort analysis where limited information available (few species)	(a) Various levels for TRP and LRP, *e.g.* TRP: $F = F_{30\%}$ LRP: 20% SPR (b) Unspecified (c) Unspecified	NOAA's Report to Congress on the Status of U.S. Fisheries - 2001 (NMFS 2002), and *e.g.* California's Living Marine Resources: Status report (Leet *et al.* 2002)
NAFO, ICES	Marine Scalefish	Age-structured models and S-R relationship over a significant period of time	LRP: and and $_{pa}$ and $_{pa}$ (pa stands for precautionary approach) to account for uncertainty in the assessment TRP: SSB above B_{pa} and below F_{pa} (rarely defined and used)	ICES 2001
CCAMLR	Scalefish, *e.g.* Patagonian toothfish	Generalised Yield Model (GYM) to estimate spawning stock levels	LRP: Select the lower fishing level γ of - γ_1 so that P(SB < 20% of pre-exploitation median level over a 20y harvest period) <10% - γ_2 so that median escapement in SB over 20y period = 75% (or 50% if no predation on stock) of pre-exploitation median level	Understanding CCMLAR's approach to management (Kock 2000)

2.12 Concluding Remarks

It is difficult to estimate how well the many reference points listed here have achieved their goals and prevented overfishing of spawning biomass. There are several reasons for this:

- The uncertainty in the data does often not permit a determination of the stock status with a precision sufficient to detect changes;
- Effective management actions have not been in place to ensure limitation of fishing mortality to the required levels; and
- Reference points have not been in place long enough to register any significant effects on the stocks.

Available data for many developing or small-scale fisheries are so limited that there is often no alternative to using only the most rudimentary quantitative analysis. In these situations, sound decision rules are not possible or credible. However, the analysis might point in a general direction and future research priorities may be the key outcome (Hilborn and Peterman 1996).

In the meantime, advice for appropriate management actions can be supported by educated guesses for parameter estimation through a meta-analysis of data and estimates from species with taxonomic and regional affiliations to the target species in a particular fishery. This would also assist in determining what is likely to be the most sensitive component for the assessment. If preliminary management actions are going to be implemented before appropriate data become available, then those actions should be set up as part of an experiment with adequate monitoring (Hilborn and Peterman 1996). There has been a rise in the use of meta-analyses to produce at least some management advice but, as with all these indirect methods, care needs to be taken when interpreting any results. Simple taxonomic similarity does not guarantee ecological similarity and this seems to be forgotten in the enthusiasm for finding at least some information about a fished species (FishBase 2000).

CHAPTER 3: SURVEY OF LIVE-FISH INDUSTRY PARTICIPANTS

3.1 Introduction

Fishers knowledge represents a source of 'non-scientific' information that is often overlooked or under-valued by researchers and managers when attempting to understand the dynamics of a fishery or target species. Catch and effort information alone cannot adequately depict the dynamics of a fishery or the stocks, especially where external factors such as market developments, changes in gear, fishing practices and management have influence. For developing and data-limited fisheries, fisher's information has the potential to be very instructive when attempting to synthesise available information for stock assessment or for hypothesis testing.

3.2 Methods

A semi-structured questionnaire designed to collect information about industry perceptions and experiences in the live-fish fisheries for banded morwong and wrasse was developed in consultation with industry. The questionnaire was split into the following sections

- Profiling information - years of industry experience, dependence on the live fishery, licences held, etc.
- Resource status – general perceptions about current resource status and changes that may have occurred over time.
- Fish behaviour – general observations about fish behaviour, especially in regards to interpretation of fishery data.
- Catch and catch rates – trends in catch and catch rates over time and spatially.
- Commercial catch and effort data quality – reliability of commercial catch and effort information.
- Key industry developments – developments including changes in gear and fishing behaviour and their potential impacts on catch and effort.
- Management – appropriateness of current management strategies.

Key live-fish operators in Tasmania and Victoria were identified based on their current and previous involvement (> 3 years) in the banded morwong and/or wrasse fisheries. A sample of fishers was contacted and interviewed, sample selection being undertaken giving regard to achieving wide regional coverage. All interviews, mostly involving individuals but occasionally in groups, were conducted in person by the Principal Investigator between March and August 2003.

3.3 Results and Discussion

A total of 17 Tasmanian fishers, representing 12 separate fishing operations and three processors, and 9 Victorian operators participated in the survey. Overall, 10 fishing operations involving banded morwong (8 Tasmanian and 2 Victorian) and 14 involving wrasse (7 Tasmanian and 7 Victorian) were represented in the survey, noting that some operators targeted both banded morwong and wrasse. All operators who were contacted participated fully in the survey, with interviews generally lasting 1.5-2 hours in duration. A profile of respondent's years of experience in the live-fish fishery is presented in Table 3.1. While not necessarily representative of the wider industry, the greater representation of fishers with >10 years experience in the Tasmanian sample reflects the fact that the live-fish fishery developed initially in Tasmania and then expanded into Victoria (and more recently into South Australia).

Table 3.1: Years of involvement in the live fish fishery (at the time of interview), based on fishing units

Fish processors have been excluded.

Experience (yrs)	Tasmania	Victoria
3-5	5	2
5-10	2	6
10+	5	1
Total	12	9

3.3.1 Banded morwong - Tasmania

Resource status

Fishers generally acknowledged that the fishery had impacted on the stocks, primarily due to unregulated fishing activity that occurred early in the history of the fishery in the early to late 1990s. Poor fishing and handling practices resulted in considerable losses of live product and fishing pressure was heavy as operators sought to capitalise prior to the introduction of restrictive management arrangements.

Several fishers noted that while they had observed localised depletions, stocks appeared to have recovered and there was consensus that catch rates had generally stabilised or even improved in most areas off the east coast of Tasmania since about 2001. There was, however, acknowledgement that there were now fewer 'exceptional' days (based on catches) than in the earlier years of the fishery. In addition it was noted that there had been recent expansion of the fishery off the south coast and north-east around Flinders Island into areas that had not been traditionally fished heavily in the past. Resource status in these areas was unknown.

Many operators reported relatively high abundances of undersized fish in their catches from all regions over the past couple of years and interpreted this as a further indicator of healthy stock condition.

Overall, there was a general perception that current catch levels were sustainable, implying that there was limited if any potential for expansion of the fishery based on resource sustainability beyond 40-50 tonnes per year.

Fish behaviour

It was widely noted that individual reefs could be fished down but when left alone for several months, catch rates would recover. Most fishers were of the opinion that there was replenishment of stocks onto the shallow fishing grounds, probably the result of movement of fish in from adjacent areas, including deeper water. Such fish could be distinguished from 'resident' stock by paler colour and poorer body condition. It was also noted that fish move into the shallow reefs seasonally to spawn.

Limited fishing at depths greater than about 30 m was reported, but fishers were aware that banded morwong occur at greater depths and those who had fished the deeper reefs confirmed that catches tended to be dominated by large fish, supporting the notion of size structuring by depth. One operator suggested that habitat rather than depth may be a key factor, *i.e.* small to medium fish tend to occupy habitats that include boulders and thick macroalgal cover whereas larger over-size fish favour more open and lower profile reef habitat.

Catch and catch rates

Fishers noted that catch rates varied seasonally, being lowest during the colder months, and were linked to the activity levels of the fish. Peak catches and catch rates were consistently recorded during January/February, prior to spawning, and then in May immediately following the reopening of the fishery. It was noted that spent fish tended to be easily stressed and were vulnerable to high rates of holding mortality.

Environmental factors such as moon phase, state of the tide, and sea conditions were considered to influence catch rates, though fisher's perceptions on the nature of the relationships between individual factors and catch rates was not consistent.

Seals were consistently identified as the major factor impacting catch and catch rates. It was noted that interactions with seals, resulting in loss of catch and damage to gear, had increased especially since about 1998 and was particularly severe during 2000. Once seals locate nets they damage the catch (often just eating the viscera) or remove fish from the net (often observed throwing fish, again rarely consuming the whole fish). Fishers have adapted their fishing practices to minimise these impacts, mainly by spreading the nets over wide areas and, when interactions occur, moving gear away from affected sites or even terminating a days fishing. Certain areas off the Tasman Peninsular and St Helens were identified as particular problem areas. The consequences of seal interactions has been an effective reduction in catch and catch rates due to direct losses of fish and reduced opportunities to fish in desired locations. Thus, despite increased seal interactions, the recent stability in catch rates in relation to the landed product led fishers to generally conclude that stocks have remained relatively robust and healthy.

Markets were also acknowledged to be an important factor influencing catch and effort. During the early development of the fishery the market was not generally limiting. Several processors, including some who transported live-fish by tanker trucks directly to the mainland markets, competed to purchase product, despite significant wastage due to poor handling. Since the early 2000s the Tasmanian processing sector has consolidated to three main processors and there has been a slight contraction in market demand. The SARS outbreak in 2003 reduced demand and some product replacement to other live fish species has occurred. Effectively, during periods of high fish availability fishers are now subject to processor imposed weekly catch limits.

Another factor having some bearing on daily catches levels is the onboard holding capacity of the fishing operation itself. For instance, dinghy operators are generally limited to between 50-100 kg of live product.

Key industry developments

Limited changes in gear and fishing practices have occurred over the history of the fishery. Mesh size has increased slightly, from 133 mm (5¼ inch) to 140 mm (5½ inch), and there had been an increase in mesh drop from about 15 to 25 meshes. There was a perception that the deeper nets were more effective, being less likely to roll-up than the shallower nets. The change in mesh size was not considered to have been a significant factor in terms of catch rates or the size range of fish captured.

Following the introduction of limited entry in 1998 there was phase of licence transfers that initially disrupted the catching sector as new participants entered the fishery and this may have contributed, with growing seal interactions, to the observed fishery-wide decline in catch rates. Subsequently, overall catch rates are likely to have been influenced by increased experience in the fishery.

The increase in size limits introduced in 1998 was not considered to have had a significant impact on catch rates, with the loss of access to the smaller sizes compensated for by access to fewer but heavier fish. The primary impact was on beach price, with fish larger than about 1.5 kg receiving a lower unit price.

Commercial catch and effort data quality

Fishers highlighted their opinion that data quality was especially poor prior to 1998. In particular, over-reporting of catches was considered to have occurred as fishers anticipated that catch history would be used in the allocation of licences. There were also concerns about the reporting of seal impacts in catch and effort data of recent years, since most operators did not consistently report fishing activity on days of heavy seal impacts

Several issues were also identified in relation to catch and effort reporting. Many fishers noted that they only reported that portion of the catch that was sold live, while mortalities were not taken into account. As the extent of such mortalities appeared to vary between fishers and seasonally and has varied over time (generally declining as fishers become more experienced in fish handling), the effect on reported catch estimates as opposed to actual fishery induced mortality is uncertain. Most fishers had on occasions experienced large losses and regularly loose a small percentage of their catch.

Effort can be analysed in terms of fishing days or the quantity of gear used each day. The latter measure is clearly more sensitive as an indicator of fishing intensity. However, with few exceptions, it was apparent that there has been considerable confusion about reporting requirements in the Tasmanian General Fishing Logbook regarding how effort (gear and effort units) should be recorded, making it difficult to justify using this measure of effort. Some fishers noted that they routinely reported the quantity of net carried but not necessarily lifted each day, and rather than average soak time fishers either reported total hours fished or number of net sets undertaken.

Data quality and reporting consistency, along with the impacts of seal interactions, are important factors that will affect the interpretation of fishery dependent information in this fishery.

Management

There was good support for the two month spawning closure though some fishers believed that the closure should commence in mid-February rather than at the beginning of March. The rationale given was that in some areas and in some years spawning commenced earlier than March and this would provide more comprehensive protection, especially when such fish are vulnerable to handling stress and mortality. There was strong acceptance of the size limits, though fishers and processors noted that the market preferred smaller fish and a lower minimum size limit would better suit the market.

There was consensus that the fishery could not support all 29 licence holders as viable live-fish operations, and the removal of excess capacity was identified as a priority along with verification of catch returns. Leasing of licences was seen by several fishers as a threat to the fishery as lease holders had no long-term stake in the fishery, being motivated by the desire to maximise short-term returns. This resulted in poor fishing practices and wastage, which still remain issues for the industry.

3.3.2 Banded morwong - Victoria

Resource status

The fishery is effectively limited to eastern Victoria (generally east of Lakes Entrance), reflecting the distribution and abundance of banded morwong in coastal waters off Victoria. Prior to the implementation of the developmental fishery in 2000, fishing was unregulated and considered to have impacted the stocks. The fishery currently has only two operators endorsed to fish (both interviewed) with stringent catch and reporting requirements in place. Each fisher effectively works over a wide area of coast and, with the reduction of effort by other fishers, catches and catch rates have stabilised or even improved in recent years.

Fish behaviour

Generally similar comments to Tasmania.

Key industry developments

Fishers initially used relatively small mesh gillnets (100-115 mm) and fish as small as about 28 cm were marketed, with some large daily catches reported. More recently, Victorian fishers have adopted and modified fishing methods developed in Tasmania, though both considered that they were still in a learning phase in relation to the fishery. Catch rates had generally improved over time but it was acknowledged that this might be partly linked to increased experience.

Prices paid to fishers in Victoria were substantially higher ($20-22 per kilo ex-vessel) than in Tasmania ($12-13 per kilo), reflecting differences in the cost of transportation to markets (mainly Sydney).

Catch and catch rates

A daily trip limit of 50 fish has been applied as part of the licensing requirements and was reported as being regularly achieved. To this end, catch rate data may be of limited value, at least until fish abundances are reduced and the trip limit ceases to be a limiting factor.

The recent loss of productive fishing grounds with the introduction of several marine protected areas off eastern Victoria was expected to impact on the fishery in the future. It was anticipated that areas that remained open to fishing would be subject to increased fishing pressure, increasing the risk of localised depletions. Unlike Tasmania, seal interactions were generally considered to be a minor issue at this stage.

Management imposed trip limits, rather than market demand, was perceived to be a limiting factor on catches for the Victorian operators.

Commercial catch and effort data quality

A specific banded morwong logbook was introduced in 2000 and reporting requirements were well understood by the fishers and data considered to be reliable. Greater spatial resolution of catch and effort reporting and provision for reporting catch numbers including discards represents an improvement over logbook data provided in the Tasmanian fishery.

Management

Seasonal spawning closure and size limits match those introduced in Tasmania. Banded morwong stocks were acknowledged to be more limited than those in Tasmania but the fishery could support the current number of operators on a sustainable basis. Of concern to the fishers was the fact that the fishery was classified as developmental and there was a need to formalise its status to allow for long-term planning and commitment.

3.3.3 Wrasse - Tasmania

Resource status

The Tasmanian wrasse fishery targets purple and blue-throat wrasse. Which species is captured is determined by the habitat fished; purple wrasse favour boulders and macroalgal cover whereas blue-throat wrasse tend to be found on broken or cobble bottom with light algal cover. Traps and lines are used to fish for purple wrasse, blue-throat wrasse are taken mainly by line fishing. Regionally, blue-throat wrasse were reported to dominate catches taken off the north-west, north, and central east coasts whereas purple wrasse dominate catches from the north-east and south-east coasts.

Early in the development of the fishery, and prior to the introduction of restrictive management arrangements, poor fishing and handling practices resulted in considerable losses with heavy fishing pressure applied to the stocks. All fishers interviewed agreed that localised depletion of wrasse stocks has occurred but, because of the large minimum size limit (300 mm total length), there are significant quantities of undersized fish present in the fished populations. In practice if an individual reef is overfished (from an economic perspective), the population will remain healthy because of the biomass of sub-legal but mature fish that remain. Operators fish over large areas and tend to return to specific areas on a rotational basis, generally once every 3-12 months. In this way a reef site can be fished hard with a large proportion of the legal-sized biomass selectively harvested. Growth of fish was believed to be the major contributor to renewing the biomass of fished stocks. It was acknowledged, however, that while current catches and catch rates were lower than those recorded at the start of the live-fish fishery they had more or less stabilised since the late 1990s.

Fish behaviour

Fishers reported that large blue-throat wrasse (males) tend to be more aggressive than smaller individuals and are caught first, the average size of fish generally declining during the fishing operation. Large males also tend to occur in deeper water, suggesting some size structuring by depth. There was considered to be little evidence of hierarchy for purple wrasse, at least based on the size structure of the catch. One fisher reported observing spawning aggregations of purple wrasse off the east coast, a phenomenon recently recorded by researchers (Neville Barrett, TAFI; pers. comm.).

North and east coast fishers noted that a proportion of the undersized blue-throat wrasse (< 300 mm) catch were males, implying that there may have been a biological response to fishing pressure in some areas in the form of an earlier sex transition.

Held in captivity, blue-throat wrasse males tend to be aggressive so some fishers practise ‘detoothing’ to reduce damage. It is common practice also to hold fish in the dark to reduce stress and antagonistic interactions. Purple wrasse on the other hand, do not exhibit aggressive behaviour with conspecifics.

Catch and catch rates

It was generally acknowledged that market demand represented an important limiting factor for live-fish wrasse catches. Markets had contracted in recent years, influenced by competition from Victoria and product replacement.

Water clarity was identified as a major determinant of fishing success, with lowest catch rates when water is murky. Fishers noted no strong evidence for seasonal variability in catch rates, though fish in spawning condition tend to be vulnerable to handling mortality.

Seals were not a significant problem in the wrasse fishery.

Key industry developments

A number of fishers commenced using traps to capture wrasse but subsequently switched to line fishing. Traps, whilst effective for purple wrasse, tend to include a significant by-catch of undersized fish whereas line fishing with large hooks tends to select for legal-sized fish.

Processors noted a market preference for banded morwong, while wrasse were not considered a premium product. Of the two wrasse species, blue-throat wrasse were favoured by buyers because of their appearance (colour) but purple wrasse were more robust and better suited to live-fish transport and holding. Reflecting this, processors preferred purple wrasse and tended to limit the quantities of blue-throat wrasse purchased.

Most fishers interviewed emphasised that to remain viable they needed to be an integrated live-fish operation with banded morwong or diversified, targeting a range of other species as they became seasonally available. Relatively low average daily catches (typically <50 kg) and low ex-vessel prices (7-8/kg) meant that the wrasse fishery was economically marginal.

Commercial catch and effort data quality

There was a strong sense that catches prior to the late 1990s were substantially over-stated, linked to fishers expectations about future access to the live-fish fishery. Subsequent catch and effort data quality is variable and the survey established clear problems with the way effort was reported due mainly to interpretation of reporting requirements. Importantly, species are not routinely differentiated in catch returns. Most fishers reported line fishing effort correctly, *i.e.* number of lines fished and total fishing time, but trap effort was rarely reported correctly. The measure of trap effort (number of trap lifts and average soak time) was often reported as number of traps carried and either number of sets or total fishing time. Catches were invariably reported as quantities sold live, mortalities were not taken into account.

The main wrasse processors expressed concerns that total reported production was overstated and that live fish market was probably about 50 tonnes per annum.

Management

The minimum size limit of 300 mm for all wrasse species has been well accepted by industry, although it was noted that purple wrasse from the south and west coasts tend to attain smaller sizes compared with east coast populations. In these areas, a reduction in the minimum size limit to 290 mm would be beneficial to industry. The dropping of an upper size limit for wrasse had created some marketing difficulties, with the live-fish markets reluctant to take fish larger than about 1.5 kg. Amongst the processors there was support for the re-introduction of an upper size limit (note in 2001 the lower size limit for wrasse was raised from 280 to 300 mm and the upper size limit of 430 mm was dropped).

There was considerable concern about the number of wrasse licences issued in Tasmania and the level of latent effort. In practice few wrasse fishers have been active in recent years.

3.3.4 Wrasse - Victoria

Resource status

Blue throat wrasse are the dominant species taken in the Victorian fishery, accounting for 95-100% of the catch taken between Mallacoota in the east and Warrnambool in the west. Purple wrasse also locally known as saddle wrasse become increasingly important west of Port Fairy, and dominate (about 80%) of the catch from around Portland. All fishing is undertaken with lines, including some limited use of droplines.

During the mid-1990s there was a high degree of interest in the developing live-fish fishery and a relatively large number of operators reported fishing activity. Participation in the wrasse fishery was reported to have declined markedly since the early 2000s, with probably less than 10 fulltime fishers active at the present time.

Fishers reported that localised depletions of both species had occurred as a result of heavy fishing pressure though undersized fish were plentiful. However, fishable stocks recovered relatively quickly mainly due to growth. Fishers operate over wide areas of coastline, generally with only a small number of other fishers working the same region. Fishing pressure is thus dispersed, reducing the effects of fishing on individual reefs. Experienced fishers noted that high catch rates were achieved at the start of the live-fish fishery and that while catch rates had been somewhat lower in recent years, they had remained stable for several years, implying sustainability after the initial fish-down.

Fish behaviour

Generally similar comments to Tasmania.

Catch and catch rates

Generally similar comments to Tasmania.

Sea conditions in the exposed waters off Victoria appeared to be an important limiting factor in determining opportunities to fish for wrasse. The market was not considered to be an important limiting factor although processors were noted to be unreliable in collecting fish on occasions.

The recent loss of productive fishing grounds with the introduction of marine protected areas throughout Victoria was expected to impact on the fishery in the future. It was anticipated that areas that remained open to fishing would be subjected to increased fishing pressure.

Key industry developments

Apart from development of improved fish handling procedures and some experimentation with droplines and different hook types (several fishers used barbless circle hooks), there have been limited changes in the wrasse fishery.

Market preference is for fish over 290 mm, which is larger than the minimum legal length of 280 mm for blue-throat wrasse. Most fishers noted that they applied self-imposed minimum size limits of 290-300 mm for both wrasse species although no size limits apply for purple wrasse.

Prices paid for wrasse to fishers in Victoria were generally higher ($8-12/kg ex-vessel) than in Tasmania ($7-8/kg), reflecting differences in the cost of transportation to markets (mainly Melbourne). It was noted that product from Tasmania had created gluts on the market in the past but this situation had not been a major issue in recent years.

Commercial catch and effort data quality

The primary concern about data quality surrounded false reporting of catches, motivated from an expectation that management restrictions based on catch history may be introduced in the future. Many fishers believed that non-active operators were reporting catches and that if catch history was to be used in any way, catch verification was essential.

Management

Latent effort was identified as an issue of concern as was the over-statement of catches by non-active fishers. Following high initial interest in the fishery, associated with the introduction of licences, it was noted that in recent years the number of active or dedicated wrasse fishers had declined substantially.

Future management of the Victorian fishery, including licence transferability and size limits were identified as important issues. Some operators were in favour of transferability whereas others were opposed, at least until some of the latent effort had been removed from the industry. In relation to size limits, there was support for introducing a minimum size limit for purple wrasse, or wrasse in general. The latter approach would simplify management arrangements and potentially reduce fishing mortality due to recreational catches of undersized wrasse. It was asserted that recreational fishers generally do not recognise female blue-throat wrasse as the same species as the males. Recognising the market imposed size limit of at least 290 mm, an increase in the current blue-throat wrasse size limit had some support from fishers.

3.4 Summary - Key Implications for Resource Assessment

Banded morwong - Tasmania:

- Commercial data quality has been variable over the history of the fishery, in part due to deliberate misreporting but also due to misinterpretation of reporting requirements.
- Catch and catch rates have become increasingly influenced by factors external to fish availability, including level of seal interactions and more recently, market requirements.
- While there has been some recent spatial expansion in the fishery into non-traditional areas, catches and catch rates have generally stabilised over the past 2-3 years within the main area of the fishery.
- Resilience of individual fishing grounds to fishing pressure appears to be influenced by a combination of new recruits entering the stock and replenishment of stock from other areas, *e.g.* deeper unfished reefs.
- From an industry perspective, the fishery is sustainable at current catch levels which themselves are linked to market demand.

Banded morwong - Victoria:

- The Victorian fishery is still in a development phase, though opportunities for further expansion are likely to be limited.
- Data quality since 2000 was considered reliable but catch rate information was influenced by the imposition of catch limits.
- The loss of access to productive fishing due to the implementation of MPAs is likely to result in some concentration of fishing effort in open areas and place pressure on stocks.

Wrasse - Tasmania:

- Commercial data quality has been variable over the history of the fishery especially prior to 1998, in part due to deliberate misreporting but also due to misinterpretation of reporting requirements.
- Catch returns do not routinely distinguish species and thus species based trends in fishery indicators are confounded.
- Resilience of individual fishing grounds to fishing pressure appears to be influenced mainly by new recruits (growth) entering the fishery.
- From an industry perspective, the current catches are sustainable with catches being linked to market demand, but there is significant excess capacity in the industry.

Wrasse - Victoria:

- Commercial data quality has been variable over the history of the fishery, largely due to deliberate misreporting of catch and effort but also due to misinterpretation of reporting requirements.
- Catch returns do not routinely distinguish species, but because of spatial structure of the catches some inferences at the species level can be made with considerable confidence.
- Resilience of individual fishing grounds to fishing pressure appears to be influenced mainly by new recruits (growth) entering the fishery.
- From an industry perspective, the current catches are sustainable, but there is significant excess capacity in the industry.

CHAPTER 4: BANDED MORWONG: A CASE STUDY FOR A SMALL-SCALE TEMPERATE REEF FISH FISHERY

4.1 Introduction

Banded morwong was chosen as an example species for this project due to the relatively large amount of information available over a significant portion of the fishery (Table 4.1). A previous FRDC-project (1995/145, Murphy and Lyle 1999) investigated different aspects of the biology and fishery of the species, including the development of a protocol for ageing, age-at-length described by a standard von Bertalanffy growth function, maturity, yield-per-recruit and egg-per-recruit analyses, size and age composition, mesh selectivity, and movement. Improved understanding of the fishery and the target species from this last study led to some management changes, the most significant being to raise the legal size limits to provide increased protection to mature females before they become vulnerable to the fishery. The project did not conduct a stock assessment by itself, but investigated the impact of fishing on the catch per unit effort (CPUE), and the size and age compositions at selected locations. In the early 2000s biological monitoring resumed at the same sites along the east coast of Tasmania and has proved important in quantifying changes in the stocks.

Banded morwong is a rocky reef species distributed from around Sydney, south to eastern Victoria and around Tasmania (Gomon *et al*. 1994). They also occur in New Zealand waters where they are found down to about 50 m, with females and juveniles inhabiting the relatively shallow sections of the reef and males tending to dominate deeper reef regions (McCormick 1989a). On many southern Tasmanian reefs large changes in depth occur over short distances, suggesting depth stratification of the population may be less pronounced than that described from New Zealand. There is no information on the stock structure of banded morwong and thus the relationships of populations throughout the range are unknown. With the long pelagic larval stage, however, it seems possible that there is a single genetic stock off eastern Tasmania.

Regular simple stock assessments started in 1998 with analyses of trends in catch, effort and catch rates conducted at state-wide as well as regional scales (Lyle and Jordan 1999). However, the interpretation of trends in catch, effort and catch rates was severely compromised by the limited data quality, primarily due to the lack of verification of commercial logbook returns (see Chapter 3), and suspected weak representation of stock status due to the large spatial scale of reporting and factors impacting the fishery dynamics rather than the stock. In the 2003 assessment a series of biological indicators were included for the first time using results from this study, including size and age composition, sex ratios, catch curve and yield and spawner biomass-per-recruit analyses (Lyle *et al*. 2004).

In this chapter, the current management system, fishery data and life-history characteristics of banded morwong are presented and discussed. Data presented here constitute the primary inputs into the stock modelling described in Chapter 5.

Table 4.1: Fishery and biological data available for the assessment of banded morwong in Tasmania.

'mm' is monthly, 'dd' is daily, x means data available, 1° implies one degree blocks (60 nm), ½° implies half degree blocks (30 nm).

Data type		89/90-93/94	94/95	95/96	96/97	97/98	98/99	99/00	00/01	01/02	02/03	03/04
Fishery												
Catch	temporal	mm	mm	dd	dd	dd	dd	dd	dd	dd	dd	dd
	spatial	1°	1°	½°	½°	½°	½°	½°	½°	½°	½°	½°
Effort				x	x	x	x	x	x	x	x	x
Biological												
Age composition	St. Helens									x	x	
(incl. size, sex, and	Bicheno			x	x				x	x	x	x
maturity)	Tasman			x	x				x	x	x	
Length composition	St. Helens		x	x	x				x	x	x	
(some sexed)	Bicheno		x	x	x	x			x	x	x	x
	Tasman		x	x	x	x			x	x	x	

4.2 Current Management

The live-fish fishery for banded morwong is a coastal gillnet fishery off eastern Tasmania, with catches up until 2002/03 concentrated off the central and south-east coasts (Fig. 4.1). In more recent years significant catches have come from the more northern blocks around Flinders Island. A preliminary examination of commercial catch data for 2004/05, however, has indicated that production from this region had again declined to minor levels.

Despite low production and value, currently less than AU$ 0.7million per year, the fishery is highly regulated with an array of management regulations, which was introduced in a sequence aiming at ensuring a sustainable fishery. In 1994, minimum and maximum size limits (330 and 430 mm fork length) were introduced to protect large adults and permit spawning prior to recruitment to the fishery. These sizes had the advantage that they matched market size requirements, so compliance was not considered a major problem. In 1995, a two-month closed season during the peak spawning period (March and April) was introduced and has remained in place since that time. As well as offering protection to the spawning stock, this again had the advantage that since spawning fish exhibit poor survivorship when being transported live, compliance was again not considered a major problem.

In 1996, interim licensing arrangements were implemented, with around 90 live-fish endorsements issued, allowing access to banded morwong and wrasse. These interim live-fish endorsements were replaced in 1998 by a specific banded morwong licence. There are currently 29 license holders in Tasmania. Also in 1998, both size limits were increased by 30 mm, after it became apparent that they offered minimal protection to mature females. Despite the increase, the lower limit was still set close to the size of 50% maturity at that time (Murphy and Lyle 1999).

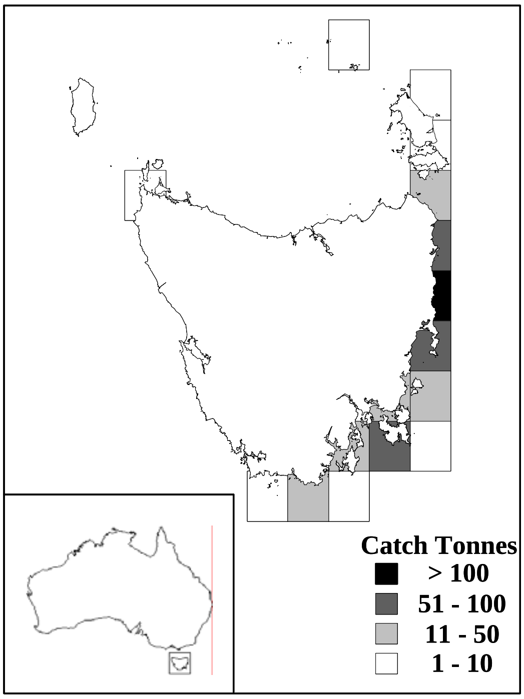

Fig. 4.1: Map of Tasmania indicating total catches of banded morwong over 1 tonne by ½ degree fishing block summed over the period 1995/96 - 2003/04.

Under the Tasmanian Scalefish Fishery Management Plan (DPIF 1998), a series of fishery performance measures or indicators are applied generically to all species. For routine assessments, annual summaries of catch, effort and catch rates are evaluated against a set of effectively arbitrary threshold levels (Lyle *et al*. 2004). These threshold levels or trigger points have been defined as levels or rates of change that are considered to be outside the normal variation of the stocks and the fishery. However, the reference years used to define "normal" variation were also arbitrary and cannot be considered either normal or to represent periods of constant effort or stable conditions; rather they represent previous experience and need to be interpreted with that in mind. The thresholds are reached when one or more of the following criteria are met:

- Total catch, fishing effort or catch rates are, to a set degree for each species, outside levels observed during given reference years (1994/95 to 1997/98 for banded morwong);
- Total catch either declines or increases in a year by more than 30% from the previous year;

These threshold levels relate to the fishery and not the stock. Performance measures relating to the stock that can be used to characterize the status of the stock are:

- A significant change occurs in biologically important characteristics, *e.g.* size or age composition, size at maturity; and
- Any other indications of stock stress.

These four classes of performance measure provide a framework against which the status of the fishery is assessed and, if deemed necessary, flag the need for management action. However, while the catch and effort threshold levels are clearly although arbitrarily defined, the other definitions remain vague and leave much room for interpretation. In addition, apart from the implementation of a review process, management responses are not formally defined if threshold levels are reached; there are no agreed decision rules within the fishery.

4.3 Fishery Characteristics

Since juvenile and adult banded morwong are largely site attached (see below), populations on individual reefs will remain relatively discrete and therefore catch and catch rate trends should ideally be evaluated at this spatial scale. However, for practical reasons, primarily the spatial resolution of the data (½ degree fishing blocks), analyses have been undertaken at the regional or block level for the main fishing areas. Regions have been defined as north east coast, including Flinders Island (3F2, 3F4, 3G1, 3G2, 3G3, 3G4, 3H3, 4G2, 4G4, 4H1, 4H2, 4H3, 4H4), St Helens (5H1), Bicheno (5H3 & 6H1), Maria (6H3 & 6G4) and Tasman (7G2 & 7H1) (refer Fig. 1.1 for map of fishing blocks). Collectively, catches from these regions have averaged over 90% of the total banded morwong production taken each year since the mid-1990s.

The assessment of catch and effort trends has been restricted to an examination of gillnet (known as graball) data, which account for around 99% of the total catch of banded morwong. Fishers are required to report the quantity of net lifted and the average set duration but interviews with fishers have established that many incorrectly report the quantity of net carried on board rather than lifted and either report average set duration, total fishing time or number of sets (Chapter 3). Because of identified problems in the way many fishers have interpreted effort reporting requirements in logbooks, effort and catch rates have also been defined as days fished and kilograms per day, respectively.

4.3.1 Catch

The ‘live fish’ fishery for banded morwong began in the early 1990s. All holders of a fishing licence (vessel) were able to take this species and, as a result, there was a dramatic increase in effort directed at banded morwong (and wrasse). Reported landings of banded morwong increased from 7 tonnes in 1991/92 to over 145 tonnes in 1993/94, though the latter figure is considered to be highly unreliable. Between 1994/95 and 1999/00, reported catches declined steadily from over 100 tonnes to just 34 tonnes, before recovering to between 45-55 tonnes p.a. (Fig. 4.2). Industry reports suggested that a generally tighter domestic market has played a role in the most recent reduction of catches (Chapter 3).

Reports at a daily temporal scale and smaller regional scale only became available from 1995/96 onwards. Catches generally declined in most areas between the mid to late 1990s, with the exception of St. Helens where catches expanded up until 1997/98 before declining (Fig. 4.2). Since 1999/2000, catches from most regions recovered slightly and either have remained relatively stable (Tasman), declined (St Helens and Bicheno) or increased (north east coast and Maria). Expansion of the fishery into the north east coast in 2002/03 (particularly around Flinders Island) meant that state-wide production was maintained to some extent by increased effort in areas that have been lightly exploited in the past.

Results of the National Recreational Fishing Survey indicated that the recreational catch of banded morwong in 2000/01 was low, at around one tonne (Lyle 2005). This is consistent with estimated recreational gillnet catch levels from the latter part of the 1990s (Lyle 2000) and confirms that the recreational catch relative to the commercial fishery is small.

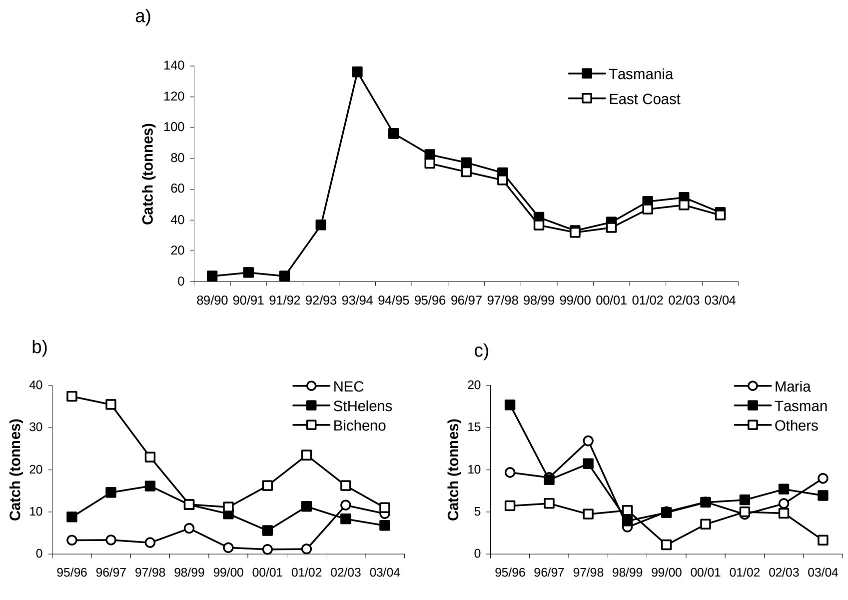

Fig. 4.2: Annual reported catch of banded morwong (tonnes): (a) state-wide and east coast reported catches; (b) north east coast (NEC), St Helens and Bicheno catches; and (c) Maria, Tasman and remaining regions (Other).

4.3.2 Fishing effort

Total effort (days fished) declined initially to about half the mid-1990s level by 1999/2000, then increased steadily to 2002/03 but fell in 2003/04 to a level equivalent to 63% of that expended in the mid-1990s (Fig. 4.3). By contrast, gear usage (100m net hours), although considered an unreliable metric because of confusion over reporting requirements by industry, increased prior to the introduction of the management plan in 1998 but has declined since to less than 40% of mid-1990s level. This implies that early in the development of the fishery, progressively more gear was deployed on average for each day fished prior to gear restrictions introduced as part of the management plan, and that since about 2000, fishers appear to be setting less gear for each day fished. There are numerous industry reports of increasing levels of seal interference with nets over time that have meant that affected fishers have often resorted to fishing with less gear or doing fewer sets each day to reduce losses to seals (see Chapter 3).

Regionally, the most conspicuous trends in effort (days fished) have included general declines in the Maria and Tasman regions. There was also an initial decline in effort in the Bicheno region that was followed by a recovery to mid-1990s levels by 2001/02. By 2003/04, however, effort had declined again, to about 55% of the mid-1990s level. Effort in the St Helens region generally expanded to peak in 2001/02 and 2002/03 but fell in 2004/05 to a level equivalent to that for the late 1990s. Finally, after a general decline in effort off the north east coast there was a sharp increase in activity during 2002/03 and 2003/04 (Fig. 4.3).

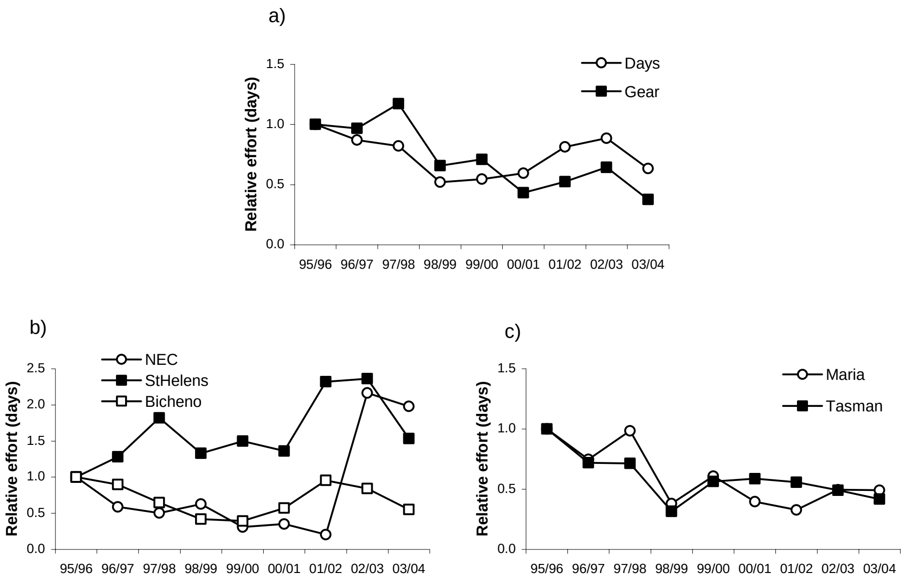

Fig. 4.3: Banded morwong gillnet effort relative to 1995/96 levels: (a) state-wide relative effort based on days fished (Day) and gear units (Gear); (b) relative effort (days fished) in the north east coast (NEC), St. Helens and Bicheno regions; and (c) relative effort (days fished) in the Maria and Tasman regions.

4.3.3 Catch rates

When calculating catch rates, the arithmetic mean of catch rates rarely describes the data accurately. Often catch rate data are log-normally distributed and the geometric mean of all valid individual daily catch records should be calculated instead. The geometric mean is calculated as the nth root of the product of the *n* individual rates y_i:

$$GM_{\bar{y}} = \sqrt[n]{\prod y_i} \tag{4.1}$$

This is equivalent to computing the arithmetic mean of the logarithm of each catch rate, and then taking the exponent:

$$GM_{\bar{y}} = \exp\left[\frac{1}{n}\left(\sum \ln\left(y_n\right)\right)\right] \tag{4.2}$$

Declines in catch and effort between 1995/96 and 1999/2000 were accompanied by a steady fall in the State-wide catch rates (Fig. 4.4), giving rise to serious concerns about the sustainability of the fishery. Since 1999/2000 catch rates have, however, risen to above 1995/96 levels, with a stronger increase evident for catch per gear unit than catch per day. Regional catch rates indicate that after initial falls, there have been recent increases in the Maria and Tasman regions, to levels comparable to or higher than during the reference period from 1995/96 to1997/98 (Fig. 4.4). By contrast, catch rates for the St. Helens and Bicheno regions have remained comparatively low (< 70% of the minimum reference levels).

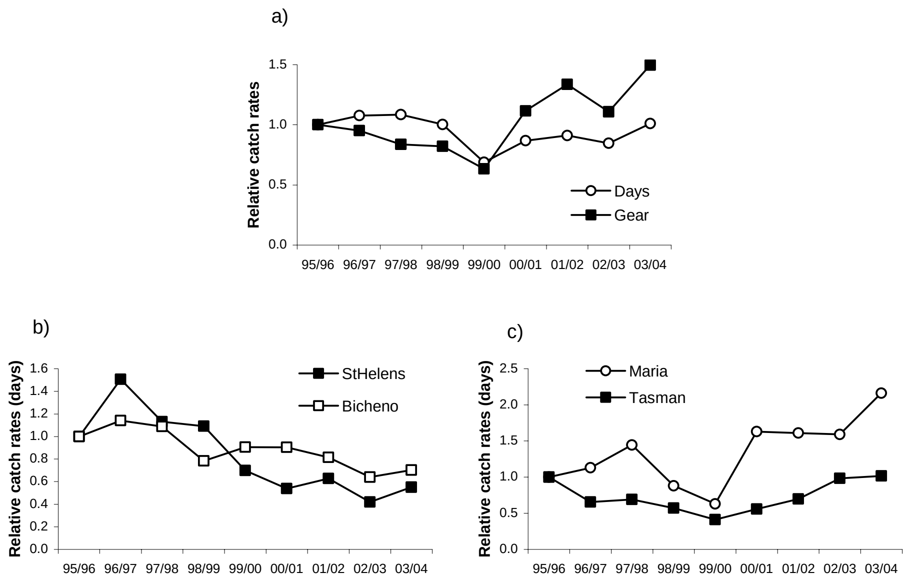

Fig. 4.4: Geometric means of banded morwong gillnet catch per unit effort (CPUE) relative to 1995/96 levels: (a) state-wide CPUE based on days fished and gear units; (b) relative CPUE (days fished) in the St. Helens and Bicheno regions; and (c) relative CPUE (days fished) in the Maria and Tasman regions.

As an improvement to using the simple geometric mean of catch rates as a stock indicator, catch rates have been standardised using generalized linear models (GLM) to reduce the impact of obscuring effects such as region, depth, season or skipper (Kimura 1981, 1988). However, while standardisation is preferred to the geometric mean, there remains no guarantee that a relation exists between the standardised catch rates and stock size, as other factors may have effects on changes in biomass that are unaccounted for by the statistical model.

Standardisation of catch rates was conducted for an annual time scale, at both a state-wide scale and for four separate fishing regions along the east coast (Table 4.2). The data was selected with respect to skippers who had reported catches for at least two years and who had had a median catch of at least one tonne of banded morwong across all years present in the fishery. These restrictions selected data that accounted for 77% of the total catch reported since 1995/96.

The GLMs were fitted to different combinations of various factors for which information were available, *viz*. skipper, vessel, fishing block, depth zone fished (<10 m, 10-20 m, 20-30 m, and >30 m), bimonthly period, and seal interference. A bimonthly period rather than month was included as a temporal factor because there would have been too few records each month to give reliable results. Due to the annual spawning season closure in March and April, only five bimonthly categories were investigated. Seal interference was included into the analysis, but it turned out not to be a very influential factor. Reporting of seal interference (in the catch returns seal interference is reported as 'occurrence') appeared to be very inconsistent, and fishing trips with seal interference and very low catch are often not reported at all (Chapter 3). In any case, a report of seal interference did not in any way allow quantification of the severity of the interaction in terms of lost catch or impact on fishing activity.

Standardised catch rates for banded morwong were fitted to natural log-transformed catch rate data (assuming a lognormal distribution), using a normal distribution family with an identity link. All models were fitted using a forward approach by stepwise addition of each factor starting with the time-step. Some interaction terms between various factors were also considered, but these were limited to combinations for which sensible interpretations could be ascribed. The optimal model was chosen based on minimization of the Akaike's and the Bayesian Information Criterion (AIC and BIC; Burnham and Anderson 1998).

Standardisation suggested a greater degree of stability by comparison with unstandardised catch rates (Table 4.2 and Fig. 4.5). The analyses generally support industry perceptions that fishing has impacted on the stocks, but that in recent years catch rates have stabilized or even improved.

Table 4.2: Generalized linear models (GLM) for the catch rates of banded morwong across the whole east coast of Tasmania, and in the separate St. Helens, Bicheno, Maria and Tasman regions.

Region		Model	Variation described
Whole East Coast	Ln cpue =	Constant + year + vessel + bimonth + seals + block + depth + skipper + block*seals	44.7%
St. Helens	Ln cpue =	Constant + year + vessel + seals + bimonth + block + block*seals	50.9%
Bicheno	Ln cpue =	Constant + year + vessel + bimonth + seals + block	49.3%
Maria	Ln cpue =	Constant + year + bimonth + seals	40.9%
Tasman	Ln cpue =	Constant + year + bimonth + skipper + seals + depth + skipper*seals	49.9%

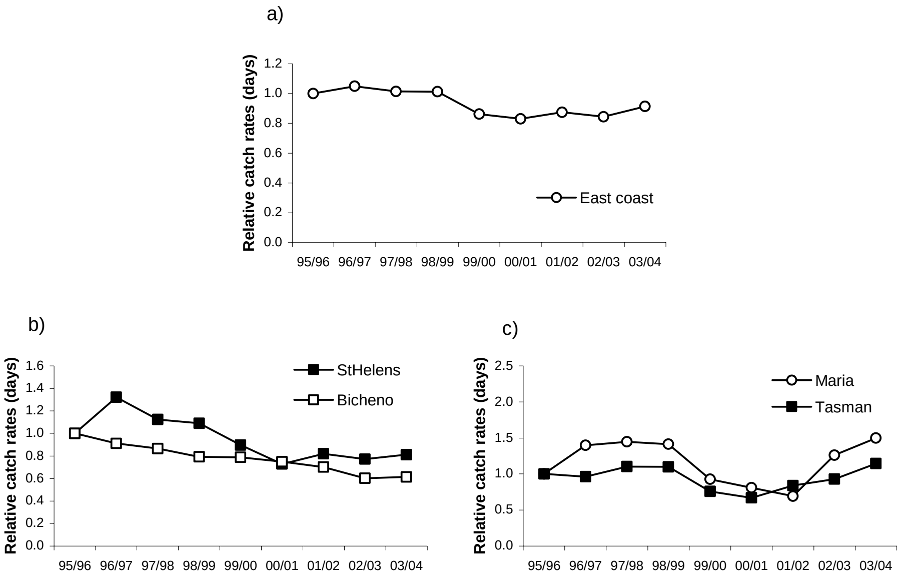

Fig. 4.5: Standardised catch per unit effort (CPUE) relative to 1995/96 levels: (a) East coast CPUE (days fished); (b) relative CPUE (days fished) in the St. Helens and Bicheno regions; and (c) relative CPUE (days fished) in the Maria and Tasman regions. Compare with Fig. 4.4.

4.4 Gillnet Mesh Selectivity

Selection curves quantify the relative capture probability of fish of a given size. Gillnet selectivity is normally expressed as a function of fish length, although the fish girth is more directly related to the capture process of gillnets and would be therefore a better measure in gillnets selectivity analyses. Gillnet selectivity for banded morwong was estimated indirectly by fishing simultaneously the same type of gillnet with different variants of mesh size.

Data for gillnet mesh selectivity collected by Murphy and Lyle (1999) around the Tasman Peninsula and Bruny Island in south-east Tasmania were reanalysed and extended to include previously un-analysed data for 140 mm mesh size (the standard mesh size used by most commercial fishers). Gillnet fishing trials using five different mesh sizes *viz*. 89, 105, 114, 133 and 140 mm (3½ to 5½ inches), were conducted between August 1995 and May 1997 and summarised for each 'summer' (November to April) and 'winter' (May to October) season. Sampling was stratified, with three recognised depth strata (shallow 0-10 m, medium 11-20 m, and deep 21-30 m). However, because the deep stratum was under-represented in the samples due to limited suitable rocky habitat below about 20 m depth, data from the medium and deep stratum had to be amalgamated. Catches of each season and depth strata (shallow, medium/deep) were standardised for effort and then amalgamated over all seasons and depths due to low sample sizes in numbers of fish captured.

Mesh selectivity was analysed for 2-cm size classes using the SELECT method (Share Each Length class's Catch Total; Millar 1992; Millar and Fryer 1999). To estimate gillnet selectivity, assumptions or inferences about the form of the selection curve, the fishing intensity, and the population size distribution need to be made (Millar and Holst 1997). The form of the selection curve relates to the retention of fish in the mesh. Normal and gamma selection curves with a spread proportional to mesh size ('geometrical similarity') were used in log-linear models as described by Millar and Holst (1997). The relative fishing intensity $p_j\,(l)$ of length l fish in a gillnet j is the probability that a fish of length l contacts this gear type given it contacts the combined gear. If catches of each mesh size are standardised for effort, the relative fishing intensity is a measure of the relative power of each mesh type to capture a fish. This could be related to mesh size, and here each selection curve was fitted twice, first under the assumption of equal fishing power of gillnets and then again assuming fishing power to be proportional to mesh size. The model deviances from the gamma selection curve were not influenced by the fishing power assumption, because the offset is confounded with parameters already in the model. Finally, no form of population size distribution was assumed and the abundance of fish in length class l is fitted in the model.

The gamma selection curve provided the best fit (Fig. 4.6), probably because many large fish are retained in the net mainly by wedging and tangling rather than by gilling:

$$s_t = \left(\frac{l_t}{\alpha k m}\right)^{\alpha} e^{\left(\alpha - \frac{l_t}{km}\right)} \tag{4.3}$$

where s_t is the selectivity and l_t the length of age class t, m is the mesh size of the nets used, and α and k are the selectivity parameters.

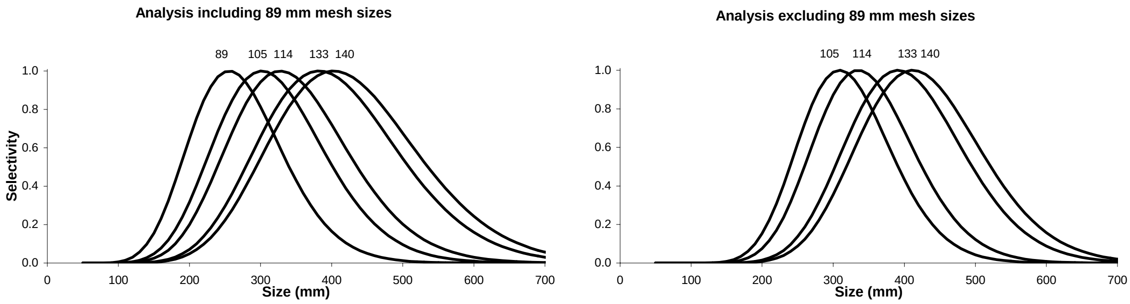

Fig. 4.6: Gillnet mesh selectivity for banded morwong including and excluding 89 mm mesh data. Gamma selection curves for mesh sizes 89, 105, 114, 133 and 140 mm.

The model deviances indicated a lack of fit for all models (Table 4.3). Particularly catches of the 89 mm nets provided poor fits due to a bimodal distribution of the size sample, possibly caused by low sample sizes. Omitting the 89 mm mesh data, estimated parameter for the spread of the curves slightly decreased (Fig. 4.6). However, the improvement in the prediction of the size frequencies for all mesh sizes was small (Fig. 4.7).

Table 4.3: Different fits of selectivity models using both Normal and Gamma distributions to describe the spread of selectivity.

The model deviance is the likelihood ratio goodness of fit.

Model		Equal fishing powers		Fishing power α mesh size	
	df	Parameters	Model deviance	Parameters	Model deviance
All mesh sizes:					
Normal	70	$(k_1, k_2) = (0.2373, 0.0103)$	265.5	$(k_1, k_2) = (0.2749, 0.0094)$	268.0
Gamma	70	$(\alpha, k_2) = (15.45, 0.0186)$	236.8	$(\alpha, k_2) = (15.45, 0.0186)$	236.8
Omitting 89mm:					
Normal	52	$(k_1, k_2) = (0.2745, 0.0056)$	177.8	$(k_1, k_2) = (0.2939, 0.0053)$	179.0
Gamma	52	$(\alpha, k_2) = (22.87, 0.0129)$	156.5	$(\alpha, k_2) = (22.87, 0.0129)$	156.5

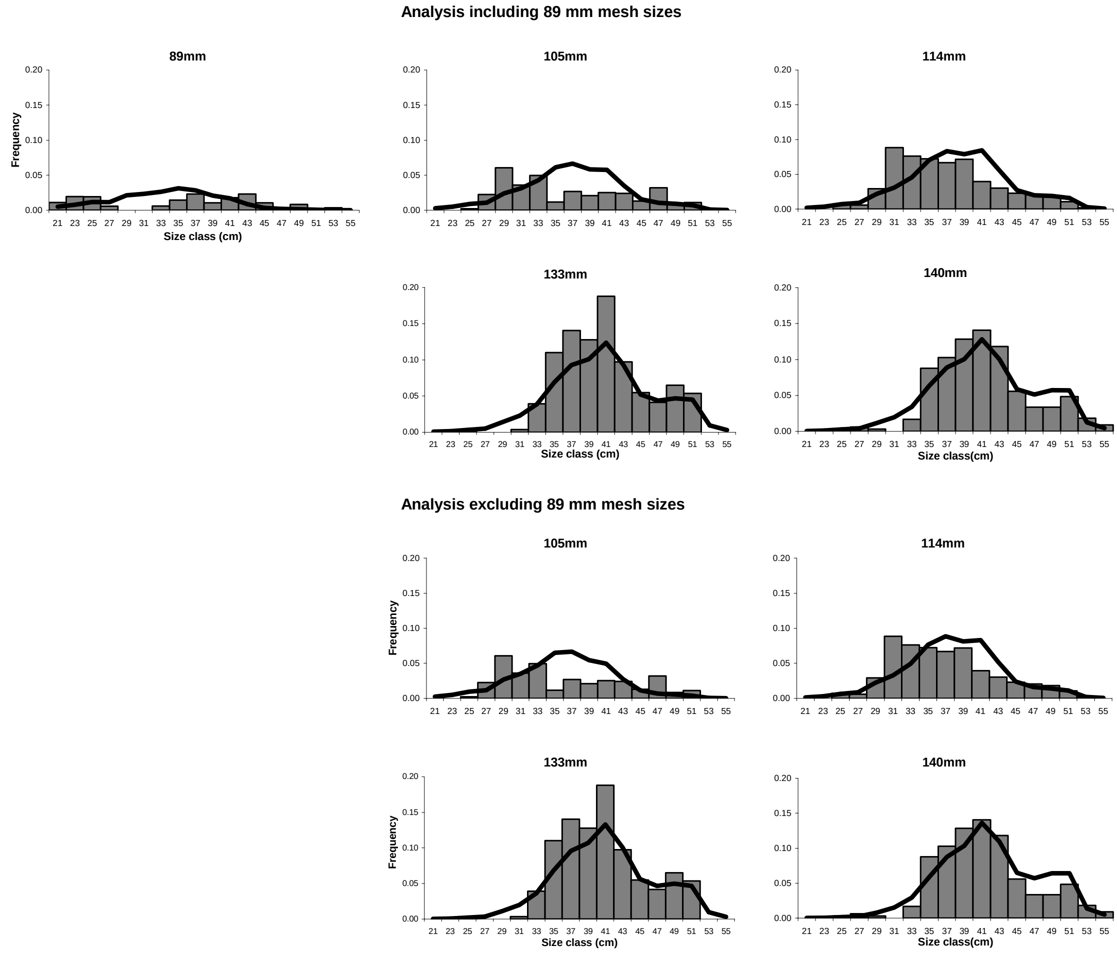

Fig. 4.7 Observed (grey bars) and predicted black line) banded morwong frequencies including and excluding 89 mm mesh data in 2cm size classes for gillnet mesh sizes 89, 105, 114, 133 and 140 mm.

4.5 Biological Characteristics

4.5.1 Biological monitoring

Biological monitoring for age and maturity studies generally occurred during the spawning periods in March and April in the Bicheno region between 1995/96 and 2003/04, in the Tasman region between 1995/96 and 2002/03 and in the St. Helens region between 2001/02 and 2002/03 (Table 4.4, Fig. 4.8). The samples from the mid-1990s derived from a previous FRDC-project on the impact of fishing on inshore temperate reef species (Murphy and Lyle 1999). All samples came from the eastern shore of Tasman Peninsula, from Bicheno or St. Helens. The reefs around St. Helens were only sampled in two years, while only Bicheno was sampled in 2004. Additional size composition data were available from commercial catch sampling undertaken in 1994/95 and 1997/98.

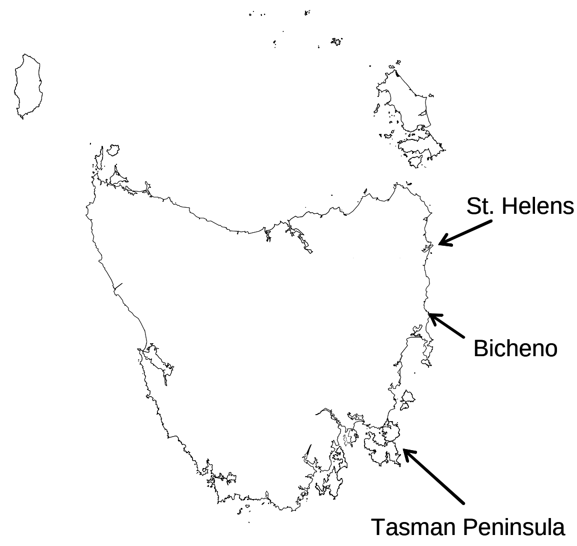

Fig. 4.8: East coast sites for banded morwong biological samples.

Chartered commercial vessels using commercial fishing gear took all spawning season samples. The fishers participating in the sampling, the fishing gear used and the reefs targeted by the fishing were similar across all years sampled. Gillnets of 20-100 m length with mesh sizes of 133-140 mm were usually set in 10-30 m depth on reefs and hauled after a few hours. All fish were sexed, length (fork length FL in mm) and weight (in grams) determined and the age, gonad weight and maturity stage were estimated.

Table 4.4: Sample sizes of the age and maturity studies in the Tasman, Bicheno and St. Helens regions.

Year is year of collection; F is female; M is male

	Tasman			Bicheno			St Helens			
Year	**F**	**M**	**Total**	**F**	**M**	**Total**	**F**	**M**	**Total**	**Total**
1996	130	98	228	150	107	257				485
1997	138	71	209	196	57	253				462
2001	128	103	231	145	72	217				448
2002	78	92	170	102	127	229	98	107	205	604
2003	116	98	214	106	147	253	82	95	177	644
2004				173	210	383				383
Total	590	462	1052	872	720	1592	180	202	382	3026

4.5.2 Size composition

The size structure of gillnet caught banded morwong has changed since the late 1990s, with a greater representation of smaller fish in current catches (Fig. 4.9-4.11). Between 1994/95 and 2003/04, size frequency distributions for males and females in the Tasman and Bicheno regions exhibited declines in modal lengths. Data for St. Helens are noisy (mainly due to small sample sizes) and no clear trend is detectable.

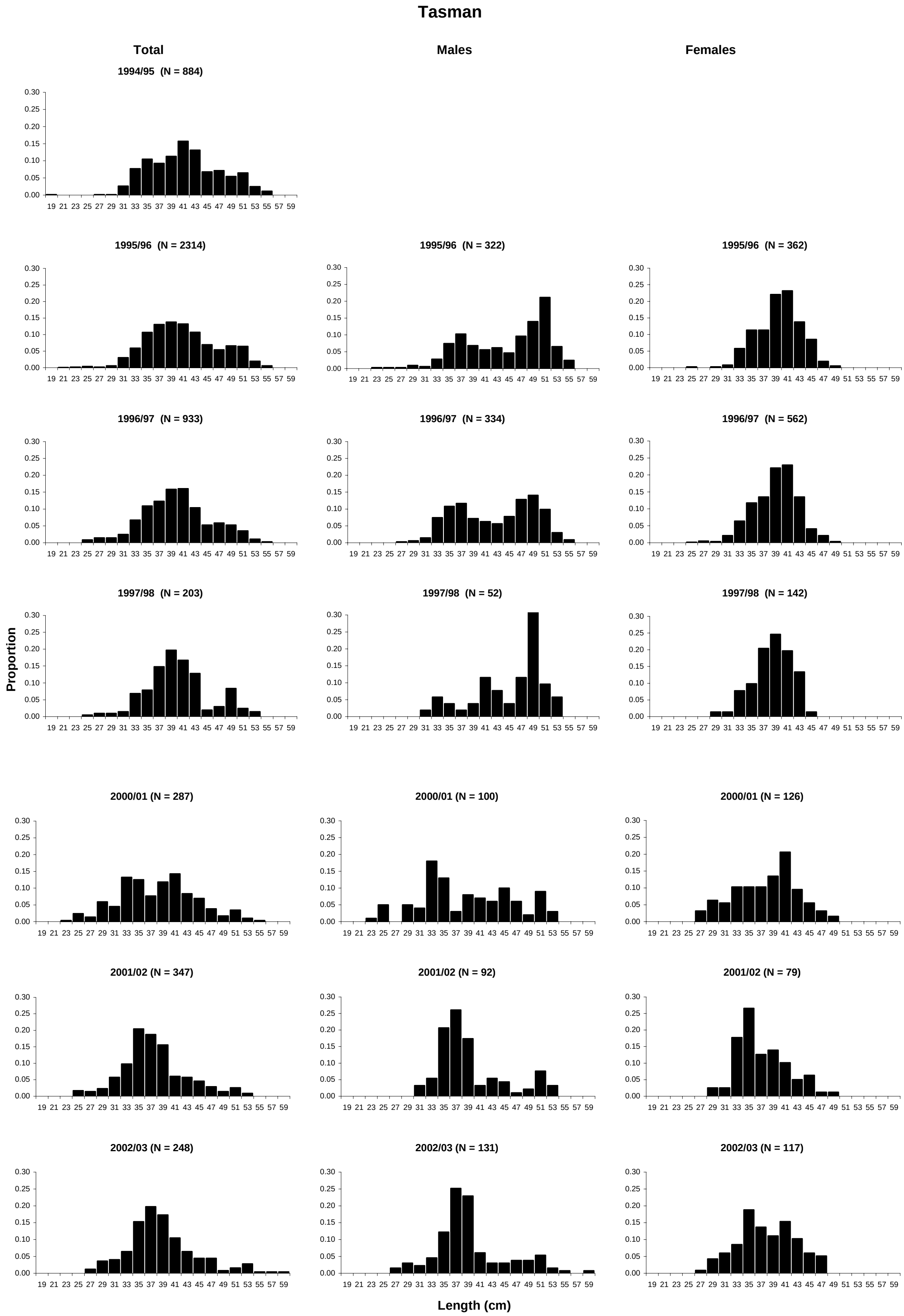

Fig. 4.9: Annual size composition of total sample, male and female banded morwong in catches of the Tasman Peninsula fishing region between 1994/95 and 2002/03. Relative frequencies in 2 cm bins (values given denote mid-point).

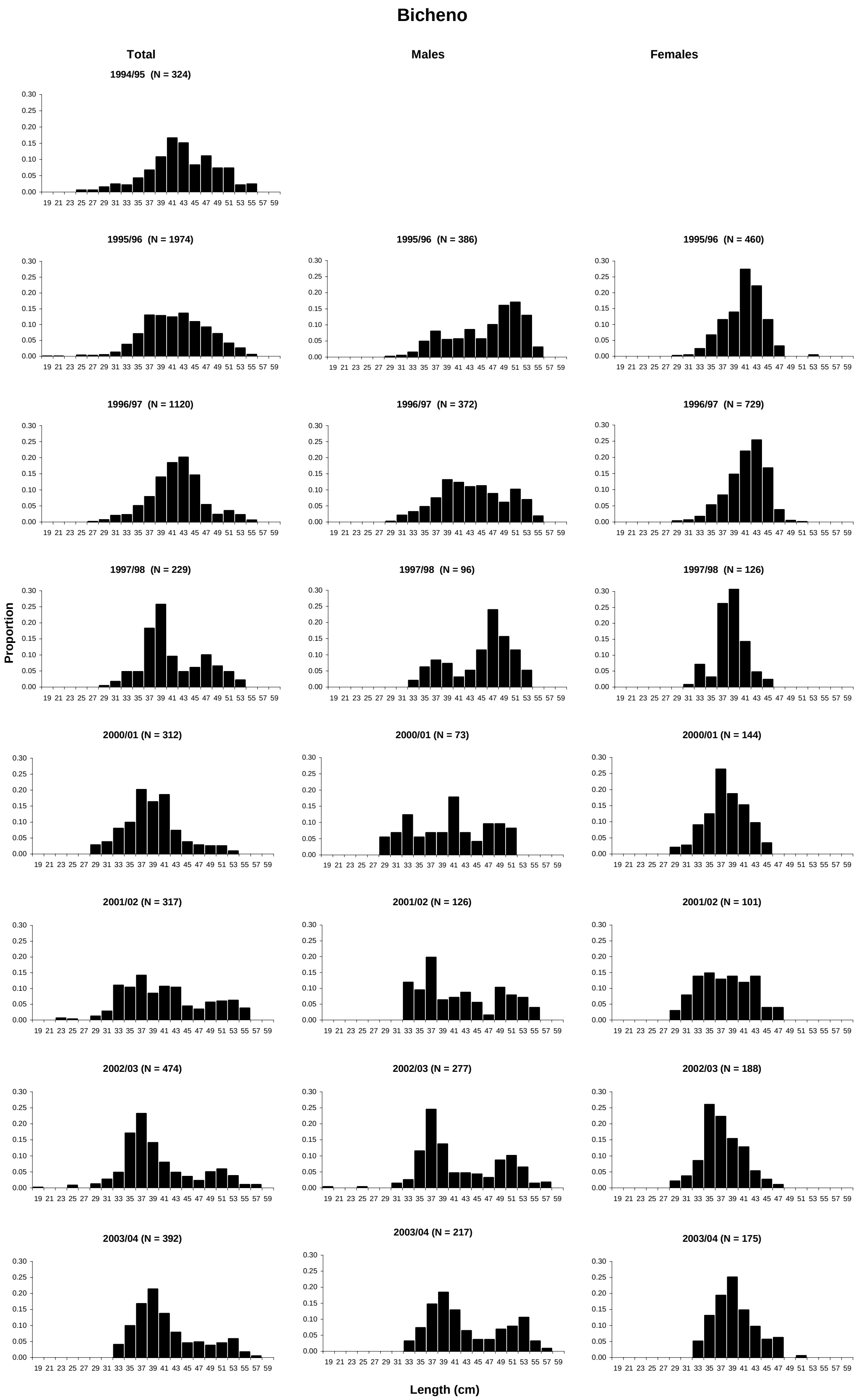

Fig. 4.10: Annual size composition of total sample, male and female banded morwong in catches of the Bicheno fishing region between 1994/95 and 2003/04. Relative frequencies in 2 cm bins (values given denote mid-point).

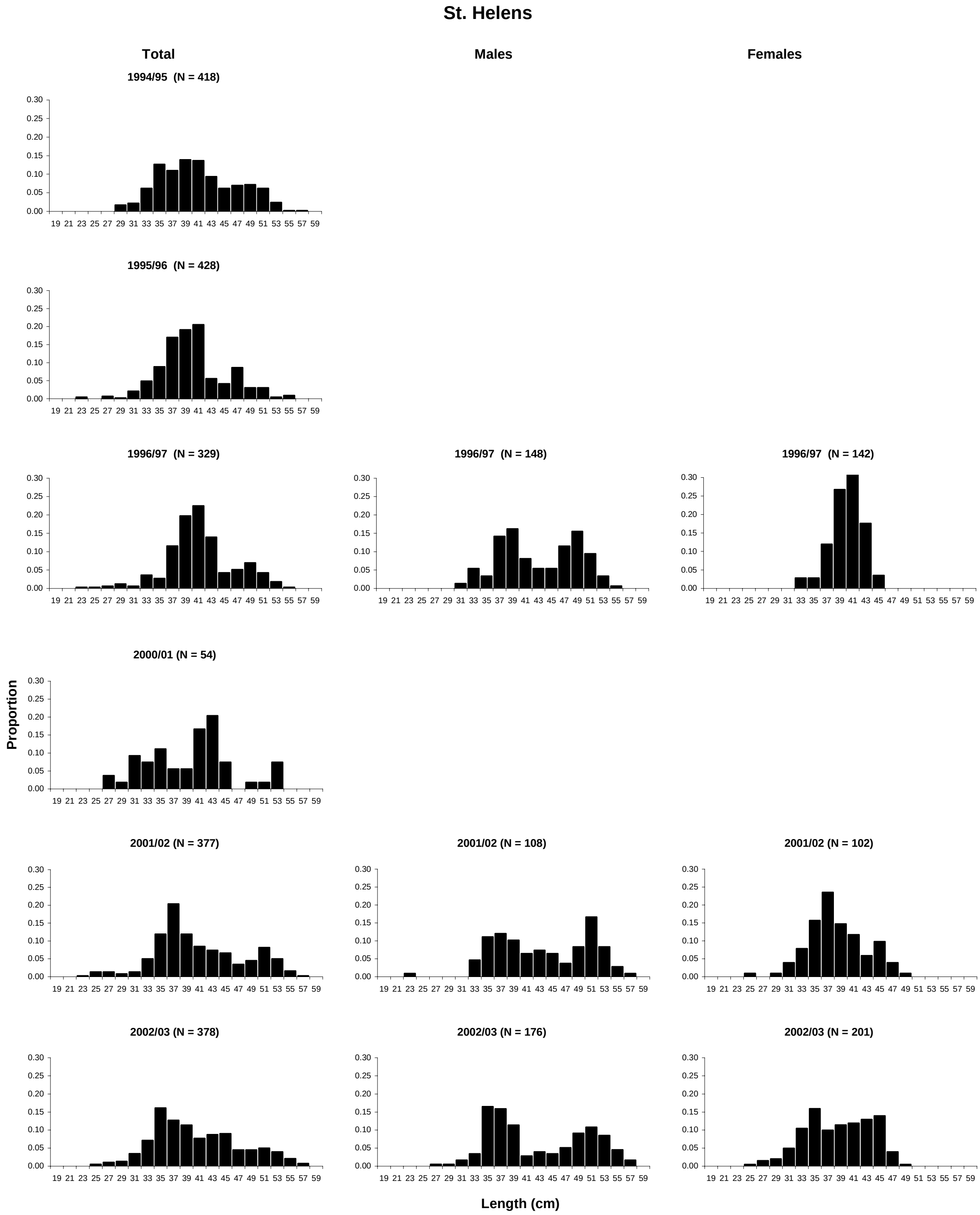

Fig. 4.11: Annual size composition of total sample, male and female banded morwong in catches of the St. Helens fishing region between 1994/95 and 2002/03. Relative frequencies in 2 cm bins (values given denote mid-point).

Male size compositions have been typically bimodal in each of the years sampled, however the relative heights of the modes have changed over time. In the Tasman and Bicheno regions, the dominant size class dropped from around 51 cm in the mid 1990s to around 37 cm in the most recent samples. By contrast, female size compositions have been generally uni-modal, though the position of the mode has also fallen from 41 cm to around 35 cm with size distributions increasingly skewed towards a greater representation of smaller size classes.

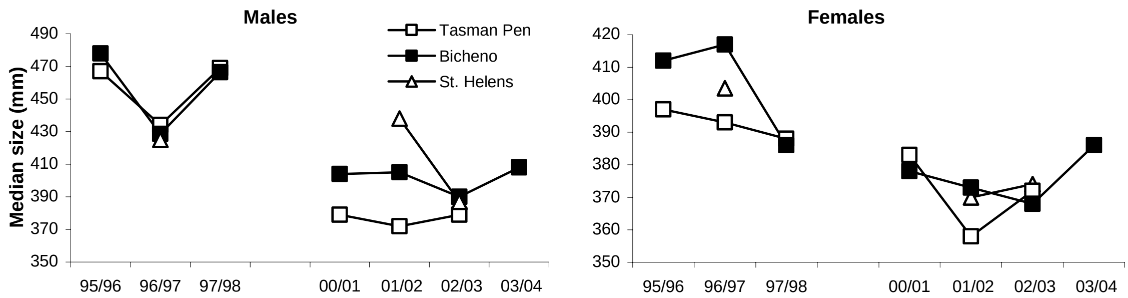

Fig. 4.12: Median size of male and female banded morwong in catches of the Tasman, Bicheno and St. Helens regions.

Changes in size structure are reflected in changes in median size (a more appropriate measure than average size; Fig. 4.12). While median sizes varied considerably during the 1990s for males, they have generally declined to around 385 mm in recent years in all regions. For females, the downward trend from around 405 to 375 mm has been more consistent over time and was most pronounced in the Bicheno region. Interestingly, in that region, median lengths for both males and females increased by 18 mm in 2003/04, the reason for this shift is unclear.

In practice, changes in mesh sizes may have masked even stronger changes in the size composition over time. In the mid 1990s most gillnets used by fishers had mesh sizes of 133 mm whereas by the latter part of the 1990s most fishers had switched to mesh sizes of around 140 mm, which are more selective for larger fish (refer section 4.4).

While decreases in fish size are consistent with expected impacts of fishing on the stocks, the biological significance of these observations is not clear. Firstly, levels of change that are considered either tolerable or significantly severe have not been defined. Secondly, relative size compositions are not useful for distinguishing between the effects of the removal of larger fish and/or increases in the numbers of smaller fish through recruitment. And thirdly, besides fishing mortality which includes handling mortality and seal-induced mortality, seasonal changes in availability of fish associated with inter-annual variability in timing of spawning and variability in the timing of sample collection may have also contributed to the observed changes.

4.5.3 Age composition

Age was determined and validated by analysis of otolith structure (Murphy and Lyle 1999). The age of banded morwong was estimated by counting opaque growth zones in thin transverse sections of sagittal otoliths in transmitted light. Validation by use of oxy-tetracycline (OTC) marking of otoliths in tagged fish and bomb-radiocarbon methods (Kalish 2001) indicated that otolith rings represent annual increments.

Age composition of population samples based on spawning season surveys indicate substantial changes during the period from 1996 to 2004. For instance, in the Bicheno region, males older than 10 years have been proportionally less abundant since 2000/01 compared to the samples taken during the mid 1990s (Fig. 4.13). Reductions were particularly evident for males aged 10-15 years, age classes that would have been exposed to fishing pressure for some years. The same trend was apparent in the Tasman region, however, males older than 10 years have been generally scarce throughout all years sampled. Because males grow rapidly through the legal-size keyhole, most males are only susceptible to fishing mortality between the ages of 4-10 years assuming that all non-legal sized fish are released and survive.

In contrast, females recruit to the fishery at around 4-5 years of age and typically remain vulnerable for the remainder of their lives (Fig. 4.14). Fishing has had a marked impact on female age structure in the Tasman and Bicheno regions. While there were still old females over 50 years present even in the latest samples, their relative contribution had decreased significantly compared to the mid 1990s, and females up to 6 years now dominate catches. These data indicate that the fishery has become increasingly dependent on new recruits.

Age composition data also provides evidence for recruitment variability, with the ability to track some strong and weak cohorts over time. For instance, in both the Tasman and Bicheno regions and for both sexes, a relatively strong cohort of 3-year olds can be tracked from 2000/01 to 4 year olds in 2001/02 and 5 year olds in 2002/03 (Figs 4.13 and 4.14). Conversely, in 2000/01, 5-year olds of both sexes appeared to be poorly represented in the Tasman region and this relatively weak cohort was evident as a trough in the age structure as 6 and then 7 year olds in subsequent years. In the Bicheno samples, this cohort was also comparatively rare and could be tracked though to 2003/04 when, as 8-year olds, very few individuals were sampled relative to the older age classes. Unfortunately, age composition data alone only permits a judgement about relative recruit numbers but not about absolute numbers, so it is not possible to ascertain whether, for example, the strength of the 3-year old cohort in 2000/01 was large, average or even poor in relation to levels of pre-exploitation recruitment. The growing reliance of the fishery (and population) on new recruits means that the extent of recruitment variability will have significant impacts on stock size and structure and will increasingly influence parameters such as the median size and age of the catch.

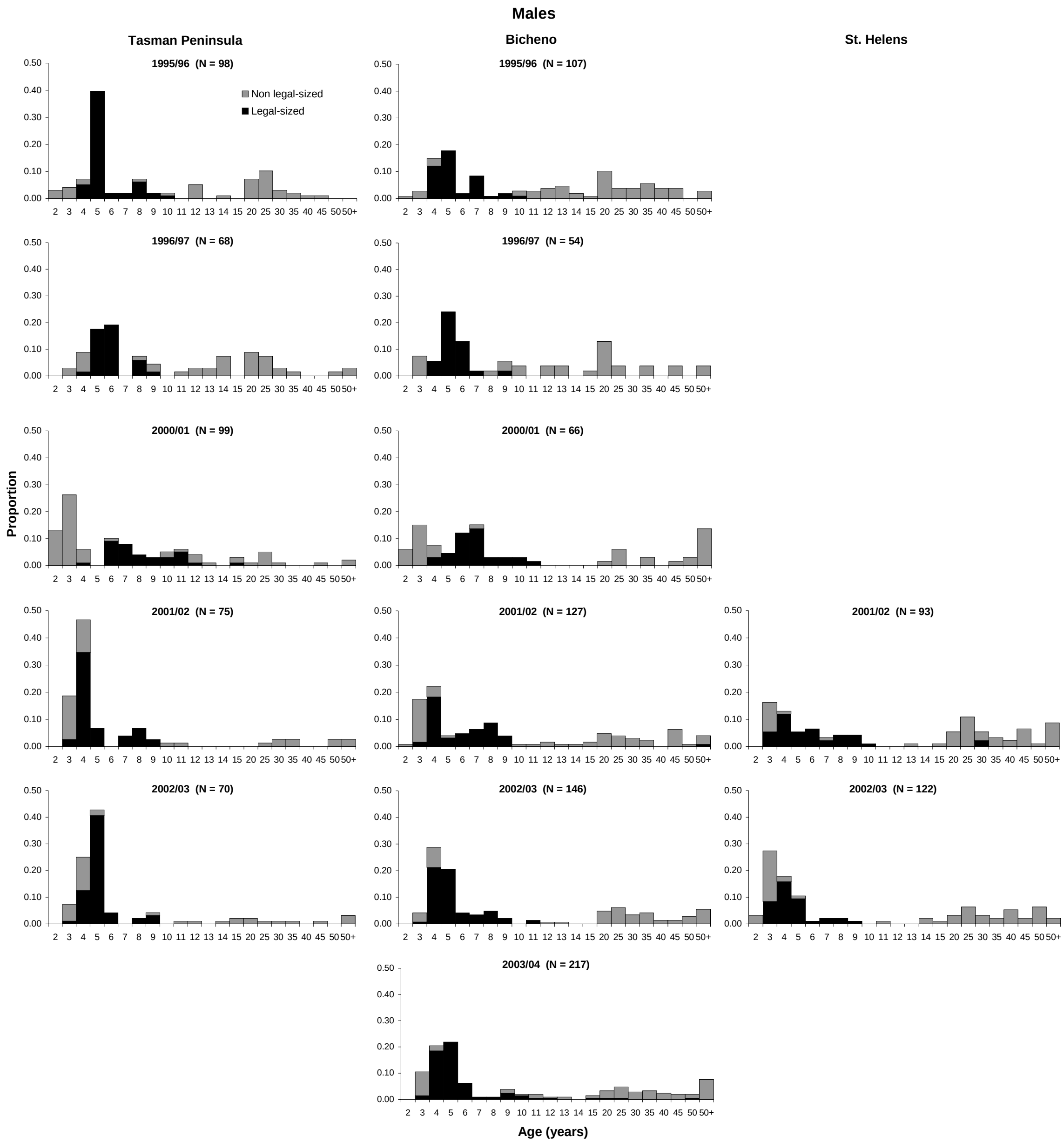

Fig. 4.13: Relative age composition of male banded morwong in catches of the Tasman, Bicheno and St. Helens fishing regions between 1995/96 and 2003/04. Black bars refer to legal-sized fish, grey bars to non-legal-sized (undersized and oversized) fish. Relative frequencies in 5-year classes (values given denote upper limit).

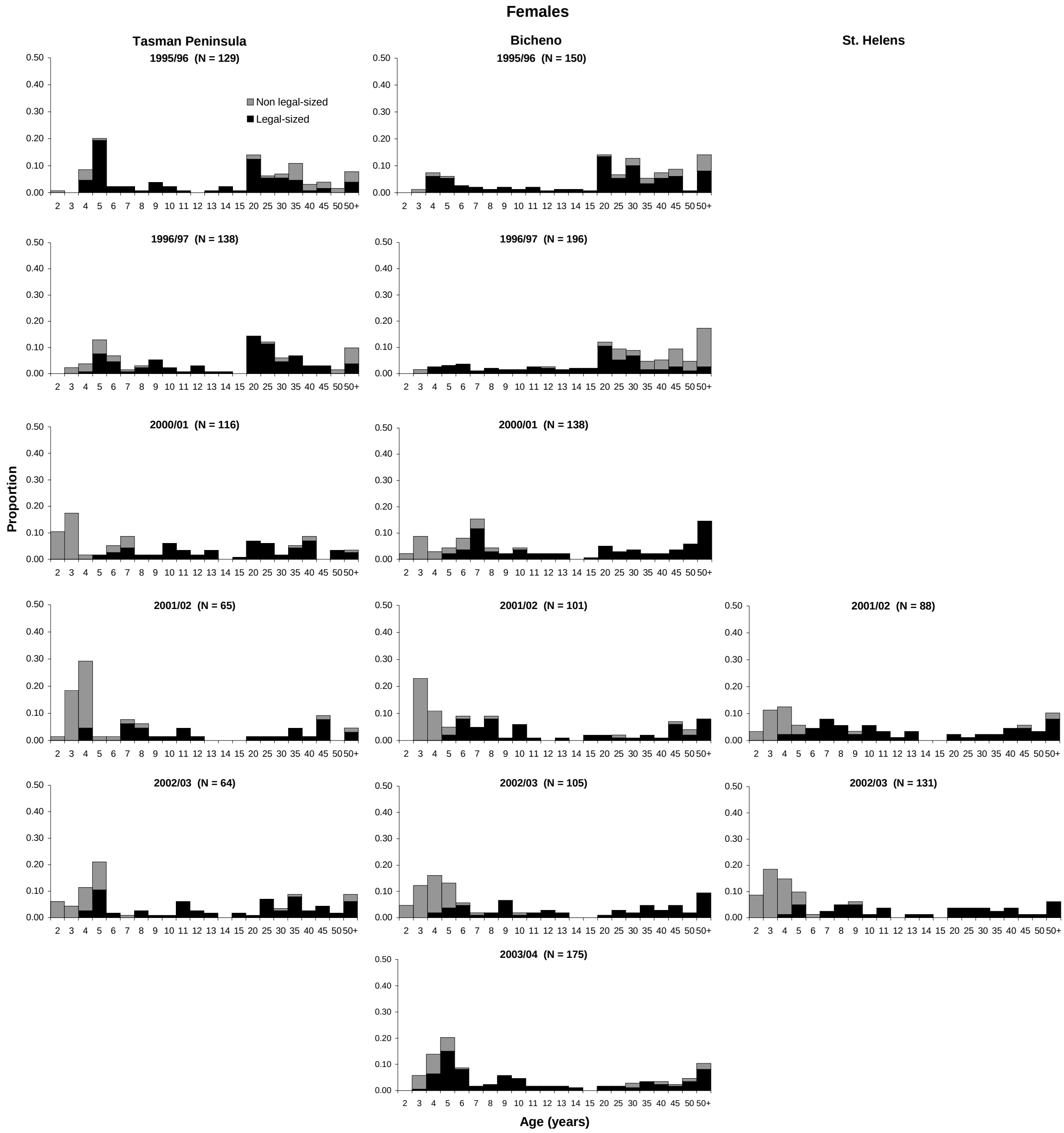

Fig. 4.14: Relative age composition of female banded morwong in catches of the Tasman, Bicheno and St. Helens fishing regions between 1995/96 and 2003/04. Black bars refer to legal-sized fish, grey bars to non-legal-sized (undersized and oversized) fish. Relative frequencies in 5-year classes (values given denote upper limit).

Changes in age composition are reflected in the trends based on median ages (Fig. 4.15). While the median age of males has decreased only slightly (with the exception of St. Helens), the median age of females has fallen dramatically from around 20 to 7 years since the mid-1990s, a trend that is reflected in both Bicheno and Tasman regions.

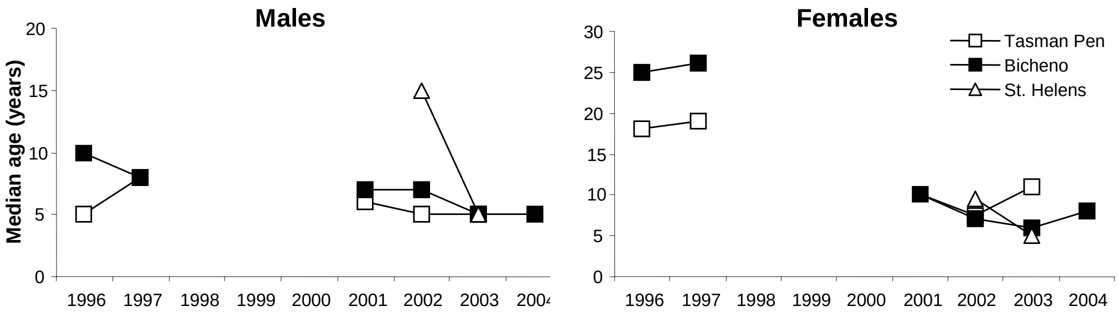

Fig. 4.15: Median age of male and female banded morwong in catches of the Tasman, Bicheno and St. Helens regions. The numbers of observations in each case can be seen in Figs 4.13 and 4.14. There was no data available for 1998 to 2000.

4.5.4 Growth

From the previous study into banded morwong, the species had been recognized as having an unusual combination of high longevity, yet fast initial growth and early maturity (Murphy and Lyle 1999). Because growth and maturity strongly affect the dynamics of fished populations and their response to fishing, considerable effort has been invested into the re-examination and determination of these life-history patterns in the present study.

There are now samples of length at age data available for describing growth from the Tasman and the Bicheno regions in the years 1996, 1997, 2001, 2002, and 2003, and in 2004 for Bicheno. Developing and comparing growth functions by year and region proved difficult due to the combination of relatively small sample sizes and the large number of age classes represented in each population, resulting in small numbers of fish in each age class. In addition, younger animals, mainly 3 to 10 year olds, consistently dominated the samples, while older animals were generally rare.

Murphy and Lyle (1999) fitted a standard von Bertalanffy growth function (VB) to the data from 1996 and 1997. However, this resulted in overestimation of size in very young fish and negatively biased estimates of the hypothetical length of zero t_0. In the analyses conducted in the present study these effects were counteracted in two ways. Firstly, a sample of 10 small fish, which had settled around November 1996 and were estimated to be around half a year old (Wolf 1998), was added to the age samples of all years. These young fish were assumed to represent the average size of very young fish in each year and functioned as an anchor for the growth functions, such that t_0 did not deviate substantially from zero.

The standard VB fitted poorly to length at age data with the anchor samples resulting in highly skewed residuals, and was therefore replaced by a two-phase VB growth function, in such a way that growth follows a different VB function in each phase, with a discontinuity at the transition age between the two phases (Hearn and Polacheck 2003):

$$L_t = \begin{cases} L_{\infty 1}(1 - e^{(-K_1(t - t_{01})}) + \varepsilon & t \le t_{trans} \\ L_{\infty 2}(1 - e^{(-K_2(t - t_{02})}) + \varepsilon & t > t_{trans} \end{cases} \tag{4.4}$$

where the L_t is the length at age t, L_∞ is the average maximum length for the species, K is the Brody growth coefficient, t_0 is the age at a hypothetical length of zero, and ε is a normal random residual (with mean zero and standard deviation σ). The transition age t_{trans} is the age at which growth switches from the lower to the upper VB growth function.

Including the age 0.5 fish meant that the optimum model fit was often found at a low transition age; in some models, it occurred at the lowest possible age of 2 or 3 years. This resulted in highly uncertain estimates of the lower VB, because it was fitted to only a small number of data points (Fig. 4.17). To avoid this problem, a decision was made to require a minimum of 10 data points when estimating the transition age.

To improve the precision of the description of growth it would have been sensible to obtain larger sample sizes by amalgamating samples from across regions and/or years. However, before samples could be amalgamated it was necessary to determine whether samples from Tasman and Bicheno regions implied equivalent growth. Similarly, it was necessary to determine whether the samples from different years were sufficiently similar that they could be validly combined. Eventually it became clear that the description of growth for banded morwong across the years 1996 to 2004 was far more complicated than expected, because changes had occurred to the younger year classes through the eight year period.

In a first step, growth was compared between the Tasman and Bicheno samples. Samples were found to differ between regions in 1997 and 2001 for both males and females, and in 2003 for females only (Likelihood Ratio Test; Kimura 1980; Haddon 2001). In both, 2001 and 2003, only $L_{\infty 2}$ of the upper growth function was significantly different between years, while in the 1997 sample the whole 2-phase VB growth function differed between the regional samples. This was expressed by significant differences in the parameters pairs $L\infty_1$ and K_1, and K_2 & t_{zero2} for males and females, *i.e.* $L_{\infty 2}$ of the upper growth function was similar between the region for males and only weakly significantly different as a pair with K_2 for females.

Table 4.5: Likelihood ratio tests for 2-phase von Bertalanffy growth functions (VB) between the Bicheno and Tasman region by year.

Shown are results from χ^2 tests. 'Coincident' means all parameters are identical between regions, '=VB_1' and '=VB2' means all parameters of the lower or upper VB are identical between regions, '= L∞1' etc means this parameter is identical between regions. Significant differences are marked in bold. For 1997, pairs of identical parameters were also tested.

			Coincident	=VB_1	=VB_2	=$L\infty_1$	=K_1	=t_{zero1}	=$L\infty 2$	=K_2	=t_{zero2}
		df	7	3	3	1	1	1	1	1	1
Males	1996	χ^2	12.59								
		p	0.083								
	1997	χ^2	56.27	28.40	33.44	0.08	2.08	1.77	0.05	0.44	0.02
		p	**0.000**	**0.000**	**0.000**	0.781	0.150	0.184	0.827	0.505	0.880
	2001	χ^2	31.25	2.71	28.99	2.70	2.48	1.31	14.01	0.43	2.62
		p	**0.000**	0.439	**0.000**	0.100	0.116	0.252	**0.000**	0.513	0.106
	2002	χ^2	1.54								
		p	0.981								
	2003	χ^2	3.36								
		p	0.850								
Females	1996	χ^2	11.97								
		p	0.102								
	1997	χ^2	142.5	16.43	131.2	0.77	3.31	2.32	1.19	0.14	0.15
		p	**0.000**	**0.001**	**0.000**	0.380	0.069	0.127	0.275	0.707	0.698
	2001	χ^2	23.84	1.37	22.61	0.32	0.04	0.00	4.18	0.43	0.10
		p	**0.001**	0.712	**0.000**	0.572	0.849	0.951	**0.041**	0.513	0.755
	2002	χ^2	9.20								
		p	0.239								
	2003	χ^2	65.43	2.06	63.88	1.98	1.24	0.50	22.92	0.62	0.04
		p	**0.000**	0.560	**0.000**	0.160	0.265	0.480	**0.000**	0.431	0.836

Table 4.5 cont.			=$L\infty_1$ & K_1	=$L\infty_1$ & t_{zero1}	=K_1 & t_{zero1}	=$L\infty_2$ & K_2	=$L\infty_2$ & t_{zero2}	=K_2 & t_{zero2}
		df	2	2	2	2	2	2
Males	1997	χ^2	26.80	2.41	2.22	1.57	0.05	13.72
		p	**0.000**	0.300	0.329	0.456	0.976	**0.001**
Females	1997	χ^2	15.76	2.37	3.35	7.50	1.69	18.36
		p	**0.000**	0.305	0.187	**0.024**	0.429	**0.000**

Average sizes of males and females older than 20 years, when fish are believed to have reached their maximum size and do not grow any further, varied substantially (Fig. 4.16). Often this was at least partly due to low sample sizes, some being as small as 7 individuals. An ANCOVA indicated that average maximum lengths for males did not differ between the regions and had increased only slightly over time (years: t = 2.015, p = 0.045). For females in the Tasman region however, the maximum sizes in the samples had increased over the years (interaction year*Tasman: $t = 3.784$, $p < 0.0005$). This was not the case in the Bicheno region.

This increase in the average maximum size of females in the Tasman region was surprising. Size-selective fishing pressure up to 1999 could have removed smaller females from the population and resulted in a bias towards larger females. But the increase appeared only to have occurred in the Tasman region and not around Bicheno, where the bulk of the catch was taken in the mid 1990s. Under the current size limits, virtually all females older than 20 years are targeted by the fishery and are subject to similar gear selectivity. Thus such a trend would not be expected to intensify. A greater shift to larger mesh sizes could have occurred in the Tasman region and brought about this unexpected result, but this was not detected in the interviews with fishers, Chapter 3). For males, size-selective removal is unlikely, because the majority of males grow through the size limits within a few years. Fishing pressure could bias the average maximum size for males only if we assume that slow-growing fish also reach smaller maximum sizes.

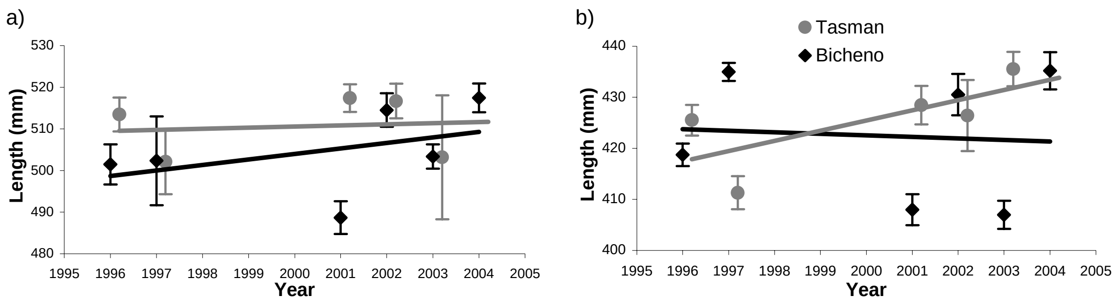

Fig. 4.16: Average length with SE and regression lines of all (a) male and (b) female banded morwong over the age of 20 years from Tasman Peninsula (grey) and Bicheno (black).

Rather than reflecting actual changes in the population, we assume that the variability in the estimates of average maximum sizes was more likely to be caused by the effects of small sample sizes. This resulted in low numbers for those age classes needed to reliably estimate L_∞ of the upper growth function. Based on the assumption of a sampling artefact, we pooled regions in all years, including those where L_∞ of the upper growth function was found to be the only differing factor. Pooling the fish samples from 1997 clearly violated the results from the regional comparison. Length at age showed quite distinct differences between the two regions, with generally larger males and females of all ages in the Bicheno sample compared to the Tasman sample. But based on the regional similarities of the growth functions in the other four sampling years, it seemed more likely that the 'real' growth function did not differ between the regions, and that other factors, such as small inconsistencies in the sampling regime or sampling locations, may have led to the observed result. False positives are to be expected when making multiple comparisons. Pooled length at age plots and the corresponding 2-phase VB growth functions are presented in Fig. 4.17 and parameter values of the growth functions in Table 4.6, including those for regional and pooled samples in 1997. Because of the regional differences found in 1997, comparisons below are also provided by region when 1997 is involved.

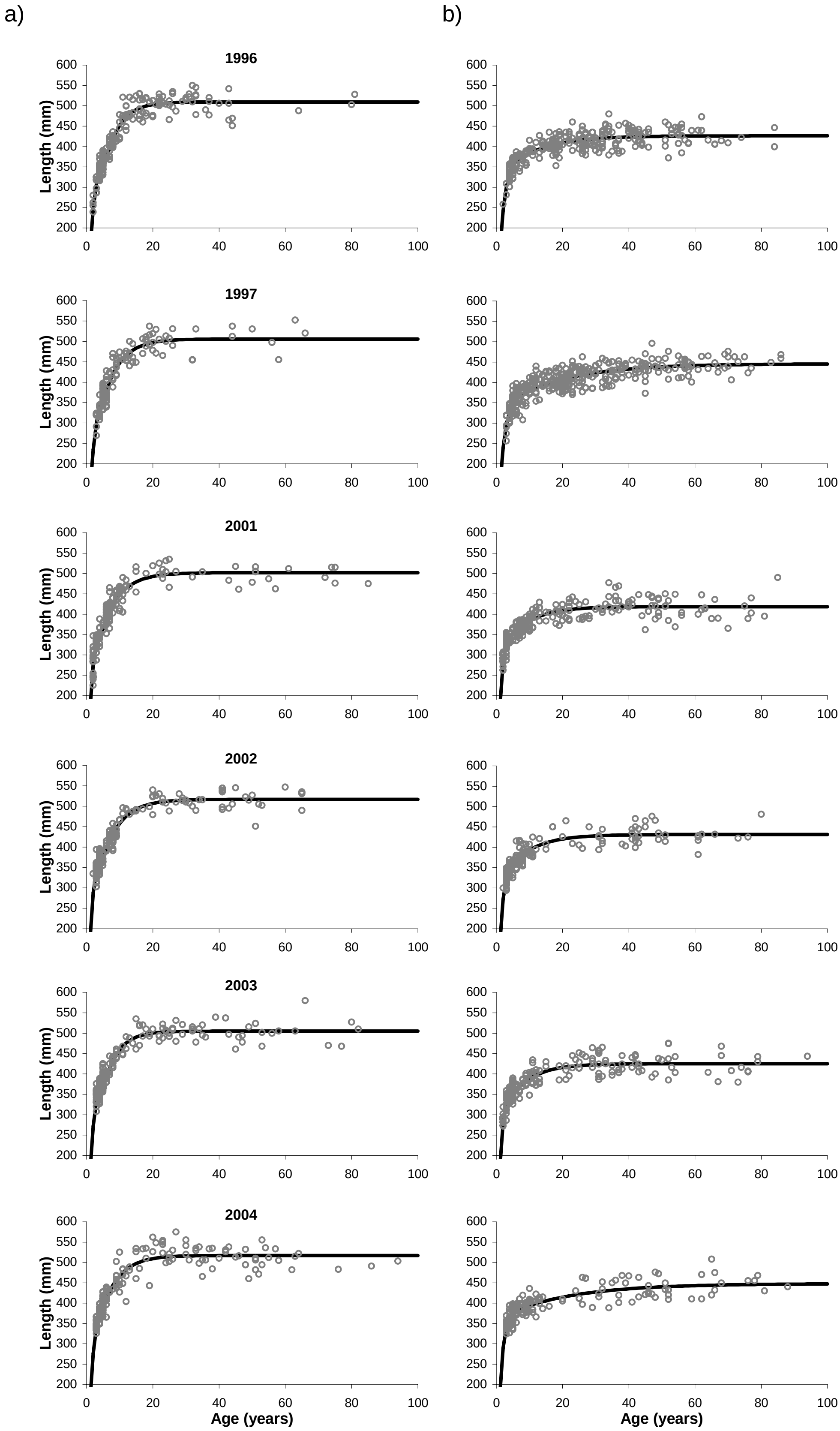

Fig. 4.17: Length at age for (a) male and (b) female banded morwong between 1996 and 2004. Grey open circles indicate individual fish, the black lines represent the 2-phase von Bertalanffy growth functions.

Table 4.6: Parameters of the 2-phase von Bertalanffy growth functions by sex and year, for combined samples from Tasman and Bicheno in all years and regional samples in 1997.

Ta is Tasman, Bi is Bicheno, L_∞ is the average maximum length for the species, K is the Brody growth coefficient, t_0 is the age at a hypothetical length of zero, and sigma is the standard deviation describing the variation around the growth curve (derived in the maximum likelihood fitting process).

		Parameters								
		$L_{\infty 1}$	K_1	t_{zero1}	sigma$_1$	$L_{\infty 2}$	K_2	t_{zero2}	sigma$_2$	Transition age
Males	1996	386.3	0.58	0.25	13.57	509.40	0.22	0.14	21.37	5
	1997	404.9	0.48	0.21	18.31	505.8	0.19	-1.31	24.10	5
	1997 Ta	418.3	0.38	0.15	11.06	506.0	0.17	-1.61	20.43	5
	1997 Bi	390.1	0.68	0.29	12.98	504.5	0.18	-2.44	22.04	5
	2001	372.2	0.80	0.31	12.76	500.3	0.18	-2.55	21.29	5
	2002	384.0	0.82	0.32	14.03	516.8	0.18	-2.12	17.37	5
	2003	400.0	0.67	0.29	15.48	504.9	0.23	-0.54	18.40	5
	2004	418.3	0.63	0.29	12.89	517.0	0.19	-2.50	25.98	5
Females	1996	371.9	0.60	0.24	12.47	426.3	0.09	-15.55	19.20	5
	1997	368.0	0.60	0.24	16.24	445.7	0.05	-25.11	20.67	5
	1997 Ta	399.3	0.40	0.15	14.32	439.0	0.05	-26.67	18.57	5
	1997 Bi	375.8	0.63	0.26	10.83	449.0	0.05	-30.28	15.51	5
	2001	358.2	0.91	0.33	12.48	418.7	0.13	-9.21	18.97	5
	2002	362.8	0.82	0.31	11.09	431.3	0.12	-9.93	20.03	5
	2003	362.7	0.88	0.32	14.55	425.1	0.14	-7.75	20.78	5
	2004	376.2	0.88	0.33	14.67	447.7	0.05	-34.19	20.31	5

In the likelihood ratio tests for males, comparing the growth functions between 1996-1997 found significant differences when regions were considered, but were effectively coincident for pooled samples (Table 4.7). The same results were obtained for samples between 1997-2001, except that it was only the Tasman region that exhibited significant differences between 1997 and 2001. Between 2001-2002 and 2003-2004, for the pooled samples both phases of the 2-phase growth function were significantly different and in particular L_∞ of the upper VB. As for males, both phases of the growth functions between 2002-2003 were effectively coincident for females. However, for all other year-year comparisons, the complete 2-phase growth functions exhibited significant differences between pairs of years, based on different parameter combinations from the L_∞ of the upper VB only (*e.g.* 2001-2002) to almost all parameters of both VBs (1997 in the Tasman region versus the pooled sample in 2001) (Table 4.7).

Table 4.7: Likelihood ratio tests for 2-phase von Bertalanffy growth functions (VB) between subsequent years for samples of combined regions and regional samples in 1997. Significance levels were reduced to = 0.25 due to multiple testing.

Shown are results from χ^2 tests. 'Coincident' means all parameters are identical between years, '=VB$_1$' and '=VB$_2$' means all parameters of the lower or upper VB are identical between years, '= L∞_1' etc means this parameter is identical between years. Significant differences are marked in bold.

			Coincident	=VB$_1$	=VB$_2$	$=L\infty_1$	$=K_1$	$=t_{zero1}$	$=L\infty 2$	$=K_2$	$=t_{zero2}$
		df	7	3	3	1	1	1	1	1	1
Males	96 vs 97	χ^2	4.92								
		p	0.670								
	96 vs 97Ta	χ^2	23.83	13.27	11.05	1.14	3.24	1.77	0.18	1.81	1.75
		p	**0.001**	**0.004**	**0.011**	0.286	0.072	0.183	0.672	0.179	0.186
	96 vs 97Bi	χ^2	29.73	11.95	18.55	1.80	0.07	0.00	0.04	0.17	1.70
		p	**0.000**	**0.008**	**0.000**	0.180	0.790	0.967	0.841	0.676	0.192
	97 vs 01	χ^2	12.48								
		p	0.086								
	97Ta vs 01	χ^2	47.25	26.60	23.21	2.44	9.22	5.56	0.25	0.02	0.45
		p	**0.000**	**0.000**	**0.000**	0.118	**0.002**	**0.018**	0.615	0.875	0.504
	97Bi vs 01	χ^2	12.97								
		p	0.073								
	01 vs 02	χ^2	32.44	16.09	17.08	0.95	0.03	0.09	8.69	0.02	0.17
		p	**0.000**	**0.001**	**0.001**	0.329	0.860	0.768	**0.003**	0.902	0.677
	02 vs 03	χ^2	14.45								
		p	0.044								
	03 vs 04	χ^2	48.49	28.17	21.55	4.57	0.33	0.00	9.75	1.45	2.13
		p	**0.000**	**0.000**	**0.000**	0.033	0.563	0.981	**0.002**	0.228	0.144
Females	96 vs 97	χ^2	21.19	0.76	20.46	0.06	0.00	0.00	14.35	4.48	2.26
		p	**0.003**	0.858	**0.000**	0.813	0.996	0.963	**0.000**	0.034	0.133
	96 vs 97Ta	χ^2	54.20	9.41	45.74	1.18	2.80	1.50	2.85	5.52	2.42
		p	**0.000**	**0.024**	**0.000**	0.278	0.094	0.221	0.092	**0.019**	0.120
	96 vs 97Bi	χ^2	64.18	2.28	62.18	0.04	0.06	0.07	21.84	4.27	3.84
		p	**0.000**	0.517	**0.000**	0.835	0.800	0.788	**0.000**	0.039	0.050
	97 vs 01	χ^2	53.20	14.60	39.55	0.48	4.69	2.32	35.11	13.32	9.08
		p	**0.000**	**0.002**	**0.000**	0.489	0.030	0.128	**0.000**	**0.000**	**0.003**
	97Ta vs 01	χ^2	75.24	34.81	44.45	3.69	15.37	7.98	8.80	13.82	8.63
		p	**0.000**	**0.000**	**0.000**	0.055	**0.000**	**0.005**	**0.003**	**0.000**	**0.003**
	97Bi vs 01	χ^2	95.32	3.64	92.71	1.25	2.73	1.61	52.52	13.22	12.07
		p	**0.000**	0.302	**0.000**	0.263	0.098	0.205	**0.000**	**0.000**	**0.001**
	01 vs 02	χ^2	25.66	0.62	25.08	0.19	0.40	0.14	10.53	0.03	0.02
		p	**0.001**	0.893	**0.000**	0.664	0.525	0.704	**0.001**	0.860	0.882
	02 vs 03	χ^2	7.19								
		p	0.409								
	03 vs 04	χ^2	43.18	18.57	25.76	3.90	0.00	0.03	16.24	8.49	8.70
		p	**0.000**	**0.000**	**0.000**	0.048	1.000	0.863	**0.000**	**0.004**	**0.003**

The results from the likelihood ratio tests reflected a trend towards increased growth rates in recent years mainly in the younger age classes (Fig. 4.18). This growth acceleration was most pronounced for 3-year old males and females (Fig. 4.19 and Table 4.8). Each year, 3-year old males and females were on average over 5 mm larger than the 3 year old males and females of the previous year. This trend decreased gradually for older age classes and ceased to be significant for 10-year old fish.

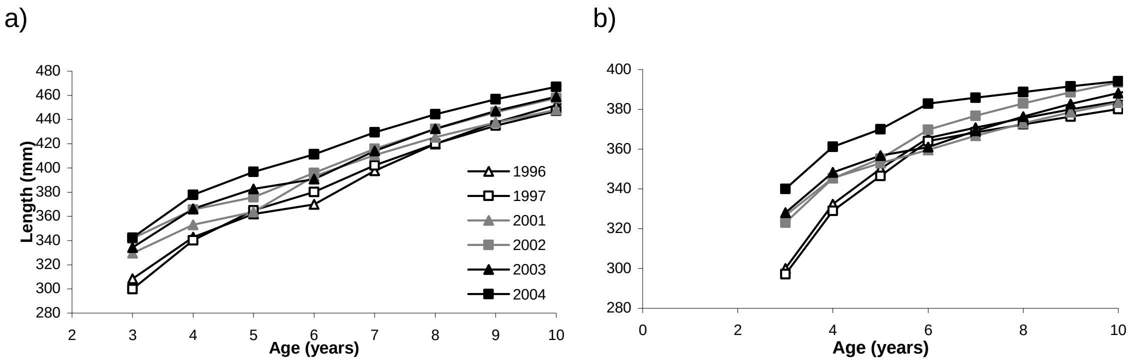

Fig. 4.18: 2-phase von Bertalanffy growth functions for 3 to 10-year old (a) male and (b) female banded morwong between 1996 and 2004.

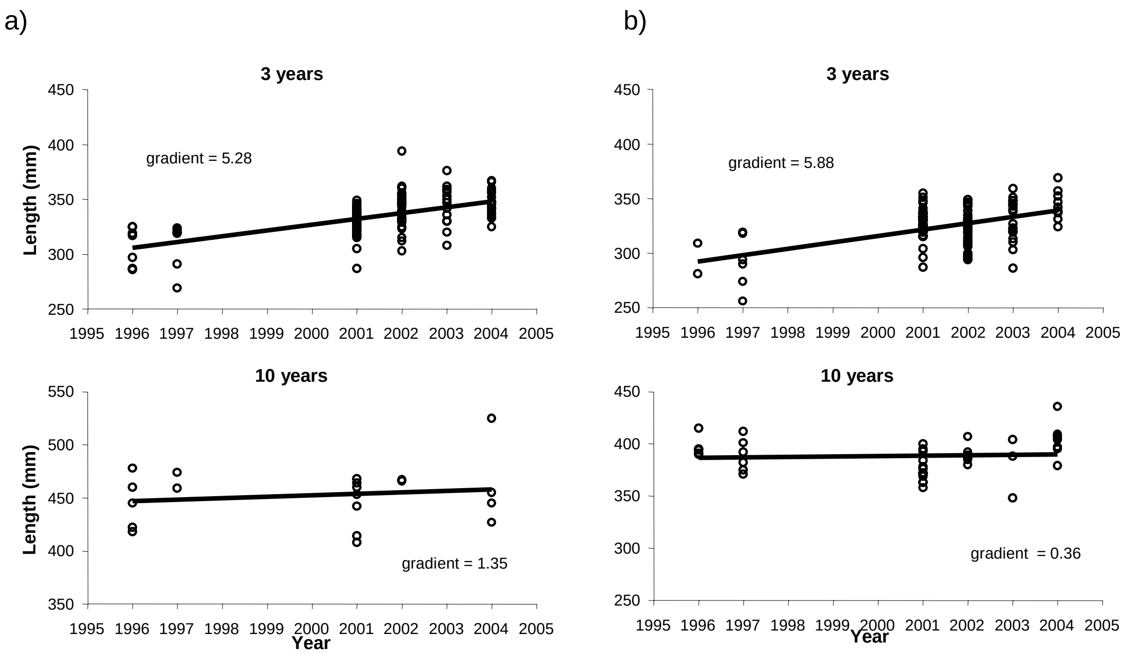

Fig. 4.19: Regression analysis of annual trends for 3 and 10-year old (a) male and (b) female banded morwong. Note different units on the y-axis for 10-year old males.

Table 4.8: Regression analysis of annual trends for year classes 3 to 10 years with sample size N, gradient and regression statistics.

	Males				Females			
Age (years)	**N**	**Gradient**	***t*-value**	***p***	**N**	**Gradient**	***t*-value**	***p***
3	120	5.28	7.99	<0.0001	103	5.88	5.99	<0.0001
4	215	4.06	9.60	<0.0001	122	2.78	6.92	<0.0001
5	213	3.80	11.79	<0.0001	145	2.25	5.59	<0.0001
6	71	3.96	4.82	<0.0001	73	1.80	2.56	<0.05
7	48	3.19	2.83	<0.01	57	2.42	2.19	<0.05
8	47	1.39	1.49	0.14	41	2.36	2.16	<0.05
9	37	3.55	4.07	<0.0005	43	1.73	2.72	<0.01
10	20	1.35	0.66	0.52	42	0.36	0.37	0.71

These changes in growth were not caused by only a single fast growing cohort. They were consistent over time and reflected in all cohorts recruiting to the fishery in recent years (Fig. 4.20). Cohorts in the samples from 1996 and 1997 were all on average smaller at age than cohorts recruiting to the fishery in the 2000s. The average size of the different cohorts seemed to converge towards older years.

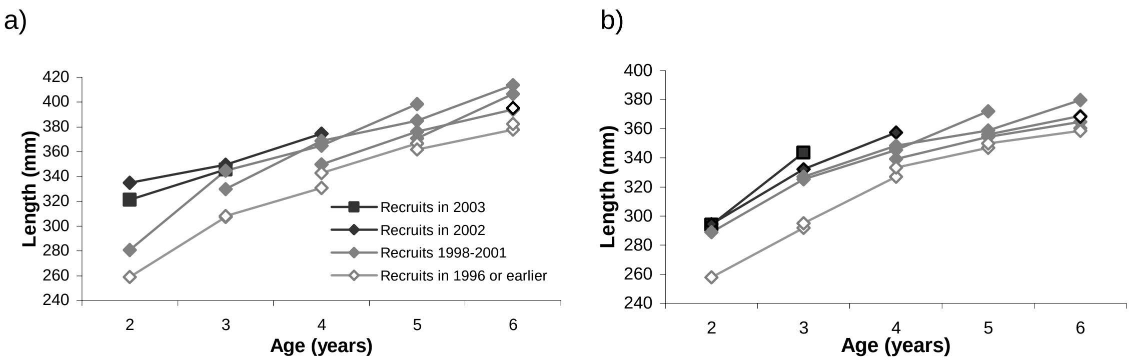

Fig. 4.20: Growth cohorts of (a) male and (b) female banded morwong. Shown is the average size of each age cohort. Fish recruiting to the fishery at 2 years of age up to 1996 (open diamonds), 1998-2001 (grey diamonds), 2002 (black diamonds) and 2003 (black squares).

4.5.5 Spawning and maturity

In Tasmanian waters, banded morwong come into spawning condition between mid to late February and early May, with the size distribution of oocytes in the ovaries indicating they are serial spawners (Murphy and Lyle 1999). Individuals have been found to be highly territorial, spawning on the same reef over several years (McCormick 1989b). The eggs and larvae are concentrated on the surface and considerable numbers of *Cheilodactylus spp*. larvae have been caught some distance off the shelf break of eastern Tasmania, suggesting a pelagic stage that is distributed in offshore waters (B. Bruce, CSIRO; pers. comm.). Juveniles appear in shallow water on rocky reefs and tide-pools between September and December after a pelagic phase of around 4-6 months (Wolf 1998).

To determine sexual maturity in females, a logistic function with binomial error distribution was fitted to the size at maturity (pooled in 1 cm size classes) or age at maturity by:

$$\mu_{t,y} = \frac{e^{(a_y + b_y l_t)}}{1 + e^{(a_y + b_y l_t)}} \qquad (4.5)$$

where μ_t is the proportion of age class t in year y that is sexually mature, l_t is the length at age t, and a and b are the maturity parameters. Ovaries were staged on a scale from 1 to 7, whereas stages 1-3 are considered to be immature, stages 4-6 mature and stage 7 is resting (Murphy and Lyle 1999).

Because sample sizes of immature females were generally too low to compare maturity between regions within years, data were pooled into the periods 1996-1997, 2001 and 2002-2003. Typically, only relatively small numbers of immature females were captured, substantially increasing uncertainty in the model fits. In the 2004 sample, just 3 immature females were captured so data for that year were not used in the analysis.

Females tended to be smaller at maturity around Tasman Peninsula than in Bicheno in 1996-97, but similar in size in the early 2000s (Fig. 4.21). Differences in age at maturity were minor between the two regions in all years considering the small sample sizes. No comparison tests were conducted because of relatively low numbers of immature individuals and related uncertainty of the shape and steepness of the logistic function.

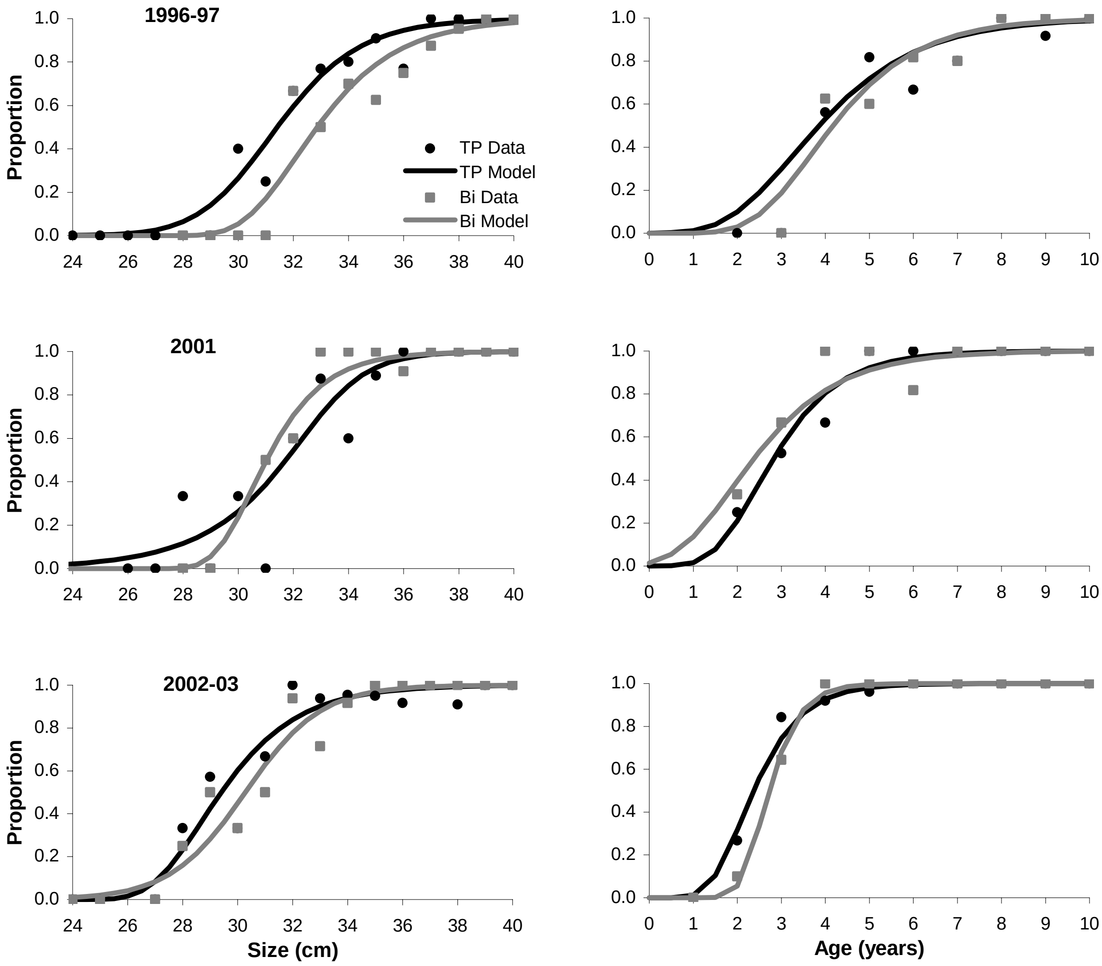

Fig. 4.21: Proportion of mature female banded morwong by size class (cm) and age (years) for 1996-97, 2001 and 2002-03. Observed data from Tasman Peninsula (TP, black diamonds) and fitted logistic function (black lines), and data from Bicheno (Bi, grey squares) and fitted logistic function (grey lines).

Both, size and age at maturity in females have decreased substantially from about 32 cm and 4-5 years of age to about 30 cm and 2-3 years of age over the observed periods between 1996-1997 and 2002-2003 (Fig. 4.22). The decreases in size at maturity occurred earlier in Bicheno, where maturity in 2001 was intermediate between the earlier and later samples. In contrast, the logistic function fitted to size data from the Tasman Peninsula in 2001 was still similar to that of 1996-97. Maturity not only occurred at earlier ages, but also more rapidly as indicated by the increasing steepness of the logistic function over the years.

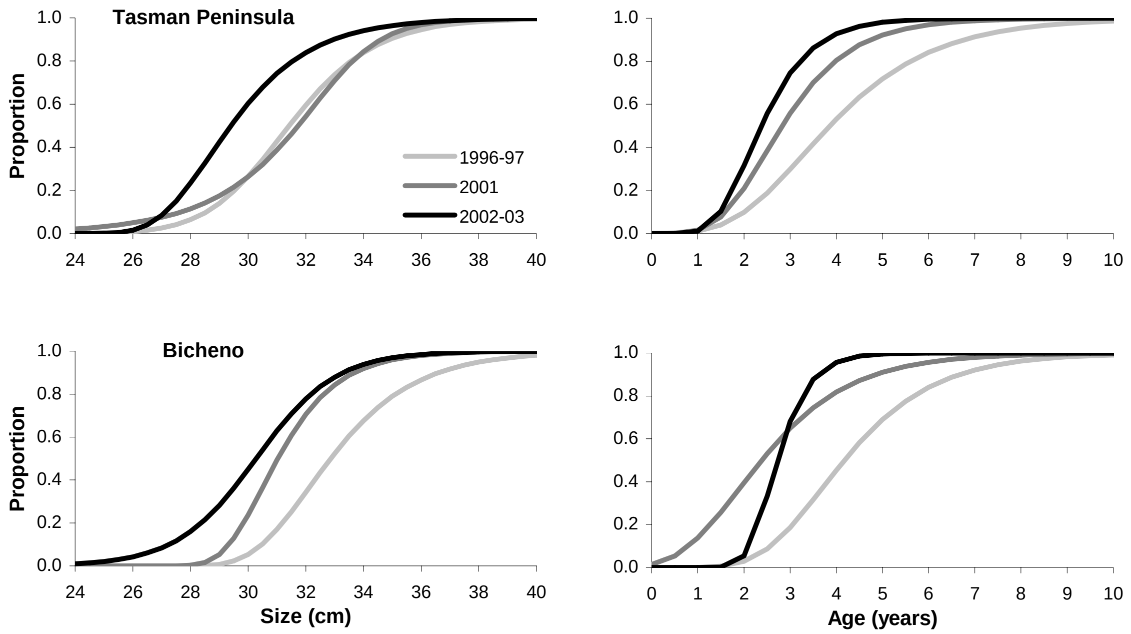

Fig. 4.22: Proportion of mature female banded morwong by size class (cm) and age (years) for Tasman Peninsula and Bicheno. Fitted logistic function to data from 1996-97 (light grey lines), 2001 (dark grey lines) and 2002-03 (black lines).

4.5.6 Sex ratio

Sex ratios based on spawning season surveys in the Bicheno and Tasman fishing regions have shifted from female dominated (up until 2001) to roughly equal numbers of males and females in the more recent samples (Fig. 4.23). The proportion of females in the legal-sized catch has dropped consistently in both regions, at least partly caused by the greater selective fishing pressure on females.

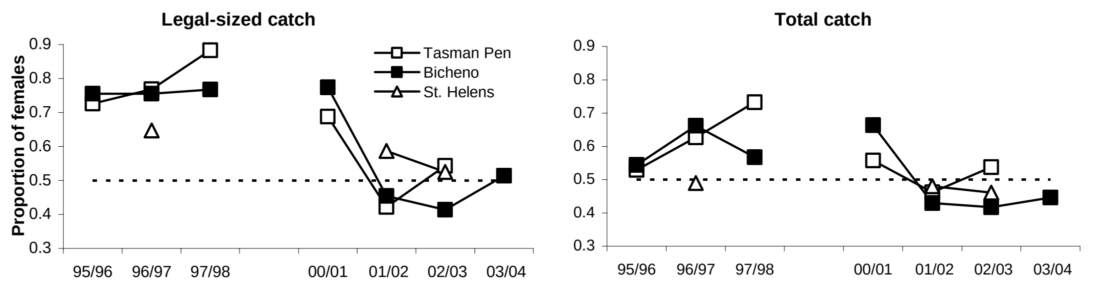

Fig. 4.23: Proportion of female banded morwong in the legal-sized and total catch in the Tasman, Bicheno and St. Helens fishing regions. Dotted lines represent a sex ratio of 1:1. The gap in the diagram represents two years.

The sudden shift in sex ratios between 2000/01 and 2001/02 in both Bicheno and Tasman regions is difficult to explain but small sample sizes and other sampling issues may have contributed, possibly mediated through some behavioural mechanism on behalf of the fish. For instance, activity patterns and hence vulnerability may change at different times within a spawning season. Consistency in ratios since 2001/02, however, lends support for a genuine change in population sex ratios. Sex ratio changes were also apparent in the total sample but, reflecting the influence of the substantial numbers of undersized recruits of both sexes, were less extreme when compared with the legal sized catch.

4.5.7 Catch curve analyses

Catch curve analysis has been used to estimate total mortality Z. The age-frequency samples have been corrected for gillnet mesh selectivity (see Section 4.4) to convert them into estimates of the age composition in the population. Given the longevity of banded morwong, spatial structuring of the whole stock, and low available sample sizes, data may not be entirely representative of the populations at a regional scale. In order to increase sample sizes and reduce noise in the data, information from the Tasman and Bicheno regions have been pooled. The catch curves were based on ages classes between the first recruited age of 4 years, or 2 and 3 years if they led to a steeper gradient, and up to 14 years, omitting age classes for which there were no data. This analysis revealed a considerable range of estimates of Z, namely 0.16 for males and 0.22 and 0.12 for females in 1995/96 and 1996/97, respectively, and a range of 0.20-0.43 for males and 0.12-0.33 for females in the 2000/01 to 2003/04 samples (Table 4.9, Fig. 4.24). Since over half of the reported catch had already been taken by 1996, the fishery would have impacted stocks even the earliest samples. Population age-frequency plots also highlight the apparent dichotomy between trends for younger (to 14 years) and older age classes for both sexes and the reduction of older individuals, particularly females, in the population in latter years (Fig. 4.24).

By comparison with many other fisheries, total mortality rates are low, but given the life-history characteristics of the species, natural mortality is also expected to be very low. Based on the approximations of Hoenig (1983) and Sparre *et al*. (1989), natural mortality M was estimated at 0.05, implying that fishing mortality F (total mortality minus natural mortality) exceeded 0.1 in the mid 1990s and has been higher than 0.2 in recent years. As a guideline, a limit reference point has been suggested of F being no more than two times M, which has clearly been exceeded in banded morwong from the sampled sites.

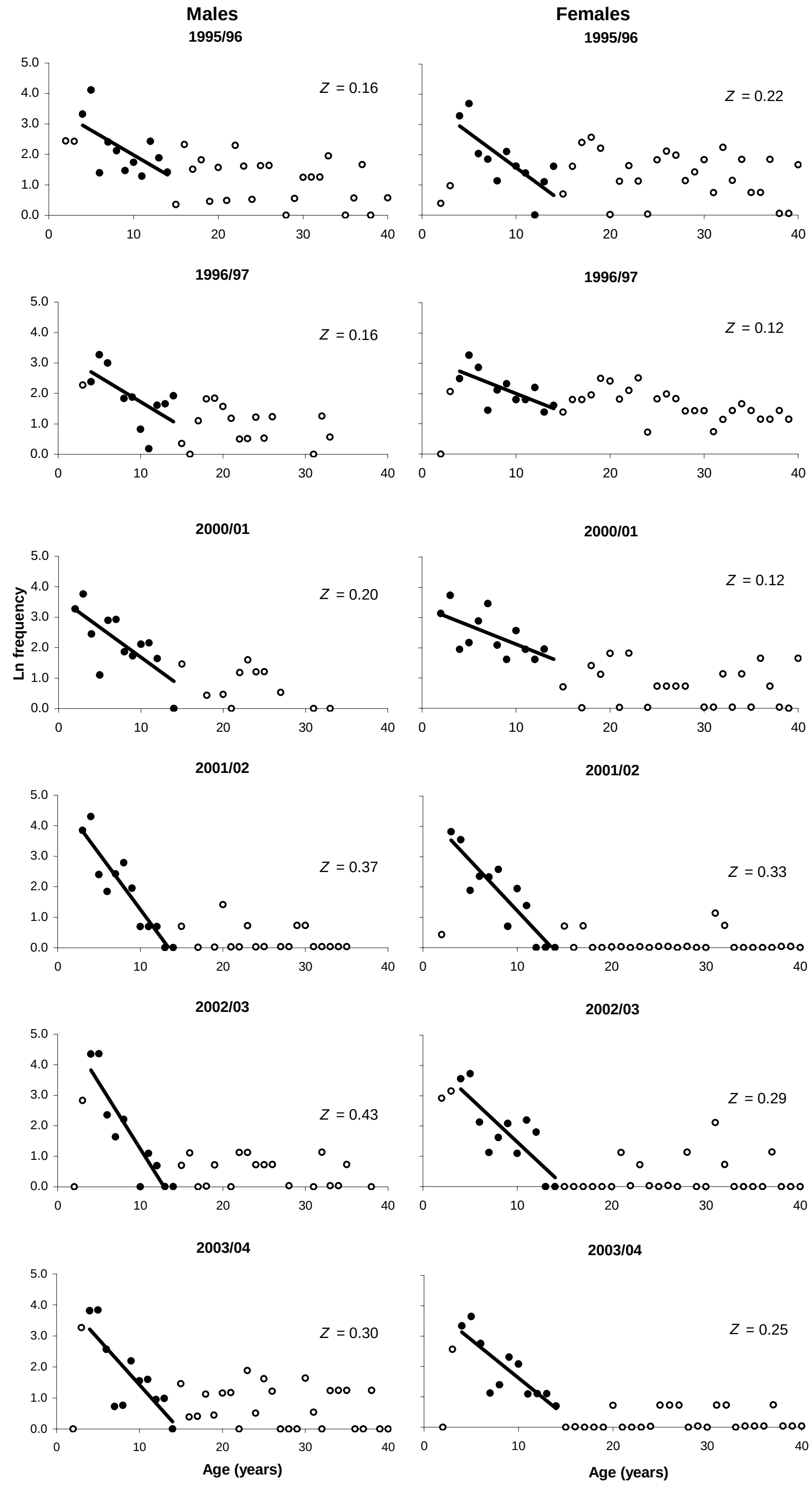

Fig. 4.24: Catch curves for male and female banded morwong in the Tasman Peninsula and Bicheno regions (pooled sites) between 1995/96 and 2003/04. The log-transformed and selectivity-corrected age-frequencies are based on direct age estimates. Data between 4 years (fish fully selected to fishing gear) or 2 and 3 years if they led to a steeper gradient and 14 years (first zero catches) has been used for mortality estimation (filled circles).

4.5.8 Yield-per-recruit and spawning biomass-recruit analyses

Yield-per-recruit and spawning biomass-per-recruit analyses have been performed using the current size limits (360-460 mm) and assumed natural mortality $M = 0.05$. These analyses highlight the dilemma faced when determining fishing reference points for species with strong sex-based differences in growth parameters. For instance, the reference level $F_{0.1}$ from the yield-per-recruit analysis was estimated as 0.25 for males but just 0.11 for females (Fig. 4.25). Because males remain vulnerable to the fishery on average only between 4 and 10 years of age, they can sustain much higher levels of fishing mortality than females. The curves were indiscriminant around their maximum values resulting in large values for F_{max} particularly for females. This is one of the reasons why F_{max} should be avoided for management advice.

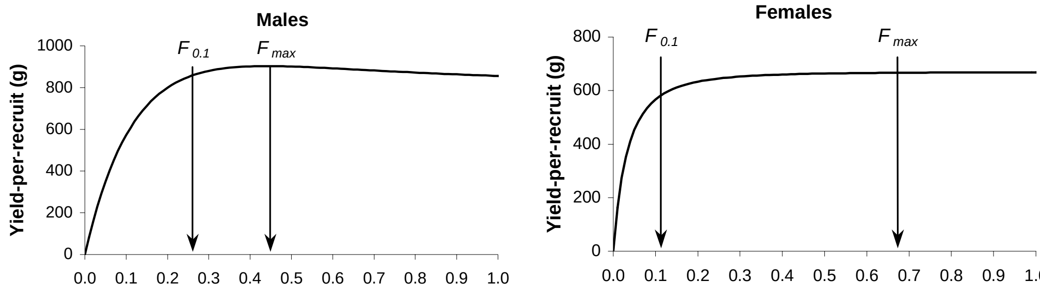

Fig. 4.25: Yield-per-recruit curves for male and female banded morwong. Results shown for size limits of 360-460 mm and natural mortality $M = 0.05$.

While $F_{0.1}$ has often been used as a target to minimize growth overfishing, it does not take recruitment variation into account (*e.g.* Clark 1991, 1993; Mace and Sissenwine 1993) and so does not necessarily prevent recruitment overfishing. To address recruitment overfishing, the fishing mortality rate $F_{30\%}$ that reduces spawning biomass-per-recruit to 30% of the unfished level, has been suggested as a limit reference point to avoid recruitment overfishing, and $F_{40\%}$ has been suggested as a target reference point for stocks where little is known about the stock-recruitment relationships and resilience (Clark 1993, 2002; Mace and Sissenwine 1993; Mace 1994). Using ovary weight-at-size as a proxy for annual fecundity, the spawning biomass-per-recruit curve drops fast with increasing fishing mortality, and hence fishing mortalities are low for both reference levels, with $F_{40\%} = 0.07$ and $F_{30\%} = 0.12$ (Fig. 4.26). This analysis suggests that only low fishing mortalities, close to $F_{0.1}$ for females, are sustainable. At mortality rates $F > 0.2$, relative spawning biomass falls below 20%.

A precautionary approach would favour reference mortality rates for recruitment overfishing of females as the basis for management advice. However, since it is impractical to manage the fishery differentially based on sex, this level of fishing mortality would provide only about 50% of the theoretical maximum yield for males.

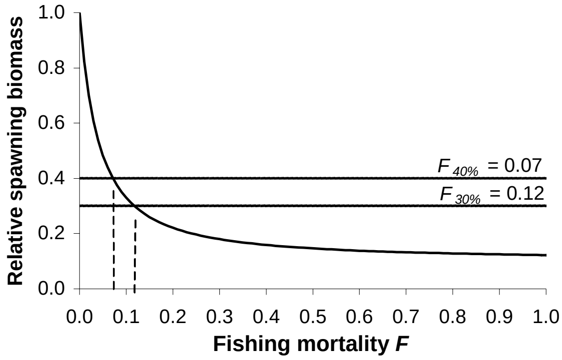

Fig. 4.26: Spawning biomass-per-recruit curve for female banded morwong. Results shown for size limits of 360-460 mm and natural mortality $M = 0.05$.

Increased minimum size limits would of course provide more effective protection to the female spawning biomass. The increase in the minimum size limit from 330 to 360 mm in 1998 represented a compromise between biological results from yield- and spawning biomass-per-recruit analyses and economic considerations (Murphy and Lyle 1999). Due to a market preference for small fish, any higher minimum size limits would have a severe negative impact on access to the live-fish markets.

Based on estimates of F inferred by catch curve analysis (section 4.5.7), it would appear that reference points $F_{0.1}$, $F_{30\%}$ and $F_{40\%}$ derived from yield-per-recruit and spawning biomass-per-recruit analyses for females have been exceeded in recent years.

4.5.9 Weight-at-length

The weight-at-length relationship is described by:

$$W_t = aL^b e^{\varepsilon} \qquad (4.6)$$

where the a and b coefficients define the power relationship between length and weight (Fig. 4.27) and e^{ε} is the log-normal residual error structure.

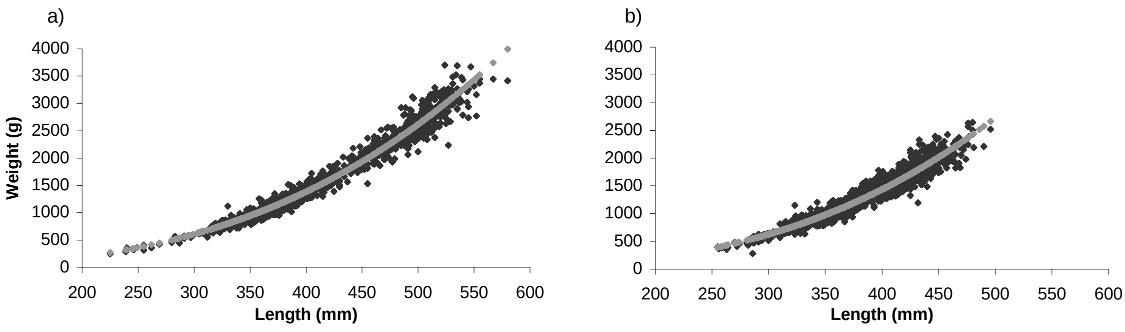

Fig. 4.27: Length-weight relationship of (a) male and (b) female banded morwong in Tasmania.

The relationships for males and females were similar, although females do not reach the large size of males and the likelihood ratio test indicated sex-based differences. These differences could have been a function of the spawning season, since the total weight-at-length of males was slightly smaller the total weight-at-length of females, but slightly larger than the somatic weight-at-length of females (Table 4.9).

Table 4.9: Parameters of weight-at-length relationship and results from likelihood ratio test comparing total weight-at-length of males with total weight-at-length of females and somatic weight-at-length of females.

Likelihood ratio test was performed for individuals 255-480 mm length only.

		Parameters		Likelihood ratio test	
		a	b	χ^2	p
Males	Total weight	0.0373	2.852		
	Total weight (255-480 mm)	0.0268	2.9415		
Females	Total weight	0.0356	2.875	112.3	<0.00001
	Somatic weight	0.0429	2.805	155.3	<0.00001

4.5.10 Movement rates

For the analysis of movement, tag-recapture data for banded morwong from the Tasman Peninsula and St. Helens, which had been collected by Murphy and Lyle (1999), were reanalysed. The convoluted coastline of the Tasman Peninsula was divided into small blocks of between 1 and 4 km in length. The distance off shore was not deemed important as all fish were tagged in < 30 m of water. If a fish was tagged and recaptured in the same block or had only moved to an adjacent block, the direct distance was used as an estimate of movement. For longer distances moved, the direct distance between the midpoints of each block was used to calculate the distance between adjacent blocks. This is similar to the distance of the smoothed coastline between the midpoints. The minimum distance moved between tagging and recapture was then estimated as the cumulative sum of distance between block midpoints. In contrast, the coastline around St. Helens is relatively straight and direct distances between tagging and recapture location were used as an estimate of minimum distance moved.

The distance moved was largely independent of the time at large (Fig. 4.28). Since many of the fish were tagged and recaptured on research trips, or by a major contributing fisher who moved between the areas with a similar annual pattern, data points peak at around 350 and 700 days.

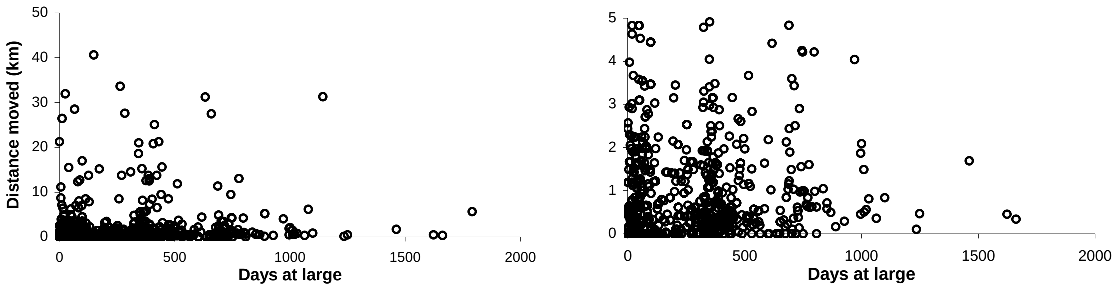

Fig. 4.28: Distance moved versus days at large for banded morwong. Right-hand figure is a close-up for distances up to 5km.

All data were combined and a negative exponential function used to model the cumulative frequencies of undirected distances moved:

$$P_r = 1 - (Y \times \exp^{(-r^a)})^d \qquad (4.7)$$

where P_r is the probability of moving distance r, and Y, a and d are parameters for the exponential function.

While the model described the data reasonably well for both areas, it over-estimated the number of movement events greater than about 4kms for both areas, more so in the St Helens region (Fig. 4.29). Despite that bias, 95% of all movements were less than 5km (model) and 8.5 km (data) in both regions.

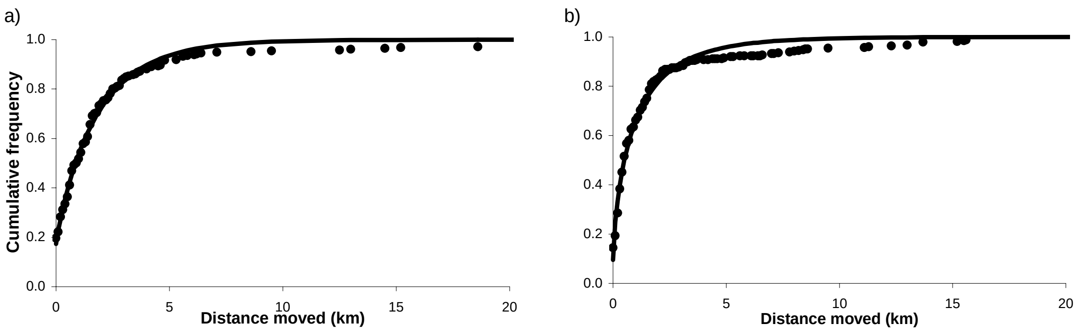

Fig. 4.29: Cumulative frequency versus distance moved (km) for (a) Tasman Peninsula and (b) St. Helens. Data points of the few individuals which have moved more than 20 km are not shown.

4.6 Discussion

Fished species, which are known or suspected of being spatially-structured, often violate many of the assumptions typically used in stock assessment models (*e.g.* Parma *et al*. 2003). They can have heterogeneous stock structure, spatial gradients in life-history characteristics such as growth or size at maturity, and diverse stock-recruitment relationships. If the fishery is also small-scale and low in value, then extensive data collection and assessments at the appropriate spatial scales are difficult to justify. Reliable but simple estimators of stock status together with management reference points that take into account the sedentary character and the specific life-history characteristics of the species are therefore needed. These can be based on simple biological indicators, singly or as a suite of indicators (Caddy and Mahon 1995; Caddy 1998; Caddy 1999). For example, Die and Caddy (1997) proposed a single reference point for total or fishing mortality which considers the size at first capture in relation to mean size at maturity to allow spawning before capture. Multiple reference points are generally considered 'safer' than applying a single option especially when information sources are highly uncertain and insufficient. Caddy (1999) proposed a semi-quantitative approach where the fisheries management cycle incorporates a resource 'traffic light' system indicating the state of the fishery on a multiple reference point board.

4.6.1 Fishery characteristics

In our analyses for banded morwong, a range of basic assessment tools have been examined, but none have provided clear indications about stock status (see also Ziegler *et al.* in press). Catch and effort analyses indicate recent stability but such stability could be based in part on the serial depletion of spatially-structured populations and/or fish down of accumulated biomass of a large number of age classes. In effect, the catch and effort data suggest stability in the fishery but are, in fact, generally uninformative about stock condition. Unfortunately, this apparent stability does not necessarily indicate that the current levels of exploitation are sustainable and a similar stability is not observed in a set of selected biological indicators. There are several limitations related to the use of fishery dependent data. These include the masking effects on localised changes in abundance arising from the geographical expansion of the fishery within and outside of original fishing grounds, the limited insights that catch rates provide into stock status for patchy, site-attached species such as banded morwong, and issues of data quality of commercial catch returns. More specifically:

- State-wide catches have been maintained to some extent by an expansion of fishing effort from traditional fishing grounds on the east coast to new areas, with a significant increase in catches off the north east coast since 2002/03.
- Due to the limited movement of banded morwong, their populations are spatially-structured and may even differ from one reef to the next. At the same time, fishers operate over relatively wide areas of coast to maintain catches. Because a fishing block, the spatial scale for reporting, is likely to encompass catches from many, potentially independent, populations of banded morwong, localized or serial depletions may not be detected.

- Catches and catch rates ignore the fact that banded morwong can grow very old (over 95 years) and catches cannot distinguish between yield due to new recruits entering the fishery from yield from the fish-down of accumulated biomass by successively removing the older fish.
- Industry members affirm the view that both the frequency of seal interactions and the quantities of fish lost to seals have increased over the history of the fishery, and that seal interactions are considered to be a more significant factor influencing the downturn in catch and effort than variation in stock abundance.
- Industry representatives also noted that new participants had entered the fishery (and some experienced operators had exited the fishery) during the late 1990s and early 2000s and this was likely to have depressed catch rates. Over the past few years there has been some constancy in terms of participants in the fishery, with experienced operators accounting for the bulk of the catch. Their experience in the fishery will undoubtedly result in higher catch rates than those achieved by inexperienced operators.
- Over the past few years the domestic market for Tasmanian banded morwong has remained flat, partly due to competition from other live product (including banded morwong from Victoria). As a result, processors influence the level of effort and catch as they seek to match market requirements. To this end, fishers often operate under what are effectively weekly catch quotas and this undoubtedly impacts on effort.
- Fishers emphasized that the reliability of the commercial catch and effort data was probably poor up until about 2000. In addition, a number of issues were identified in relation to catch and effort reporting, which affect any interpretation of fishery-dependent information. They include:

  (i) considerable confusion about how gear units should be recorded;
  (ii) fishers tend to report only the portion of the catch that was sold live and do not include mortalities (noting that handling of the catch has improved and hence mortality rates declined over time); and
  (iii) most operators do not consistently report activity on days of heavy seal impacts.

The role of external factors such as climate change and recovery in seal numbers around Tasmania, and consequent increased predation on banded morwong caught in fishing nets and in the wild population, are potentially confounding factors that may have also impacted on banded morwong stocks. There is little doubt that the increased incidence of seal interactions over the history of the fishery has resulted in substantial incidental mortality of banded morwong and impacted on fisher behaviour and catch rates.

4.6.2 Biological characteristics

Per-recruit and catch curve analyses suggest that sustainable fishing mortality levels are low but current fishing pressure may be too high, though there is considerable variability in the data. All investigated biological measures indicate that the fishery has impacted on the stocks. Some of these changes, such as the decrease of median age for females from around 20 to 7 years, appear dramatic. However, age structure information, particularly for females, indicates that the representation of old fish is still relatively high, supporting the notion that there is residual biomass still available to be fished. Changes in the character of the stock are to be expected with exploitation, however, at sustainable levels of exploitation it would be expected that the biological measures would attain some form of dynamic equilibrium.

We also found some changes with increasing growth rates of the younger cohorts and, significantly, maturation at an earlier age and smaller size. Young fish of both sexes grew faster in the early 2000s and females attained maturity at an earlier age and smaller size that in the mid 1990s. These changes imply that maturation does not occur at fixed sizes or ages for banded morwong and that this species is far more flexible in its life history characteristics than expected.

Although changes in the biological characteristics of populations are to be expected with exploitation (*e.g.* Buxton 1993; Harris and McGovern 1997; Helser and Almeida 1997), the magnitude and speed of the observed changes are unexpected given the longevity of banded morwong. This fish species is unusual with fast growth initially and early maturity, and would be an outlier when added to the relationship between maximum age and age at first maturity determined from 432 fish species by Froese and Binohlan (2000). The population dynamics of banded morwong may therefore be more comparable to other fast-growing rather than long-lived species. However, the observed rapid changes in growth and maturity can be considered as unusual even for a fast-growing species,

Unfortunately, limited sample sizes of very small and young fish (up to 3-year olds) resulted in great uncertainty in the estimation of the extent of the changes in growth and maturity. Where sample sizes were relatively robust (*viz*. ages 4 to 10), growth increases were evident across each of the age classes, although the extent of the increase declined with age. This observation is consistent with the general slowing of growth towards an asymptote but the trends visible also indicate that the changes in growth are a recent phenomenon and the effects on older age classes are yet to be seen.

Uncertainty related to the exact shape and steepness of the maturity ogive did not permit statistical comparison of curves. However, good sample sizes of females 330 mm or larger and 4 years or older indicated that there had been substantial changes in the proportion of mature females. While the whole logistic curve for size at maturity generally shifted to smaller sizes, the age at maturity function also increased in steepness indicating that within the population the transition from first to full maturity in females is now more rapid, with 50% maturity occurring at just 3 years of age compared with 4 years during the mid-1990s.

A range of factors may influence growth, and age and size at maturity of a population, including environment characteristics such as water temperature (*e.g.* Brett 1979), changes in population density (Helser and Almeida 1997; Godø and Haug 1999; Lorenzen and Enberg 2001), and genetic changes due to size-selective fishing (Imsland and Jónsdóttir 2002; Grift *et al*. 2003; Olson *et al*. 2005). All three mechanisms provide possible explanations for the observed changes of growth and maturity in banded morwong. However, their significance for the stock assessment and interpretation for management is quite different.

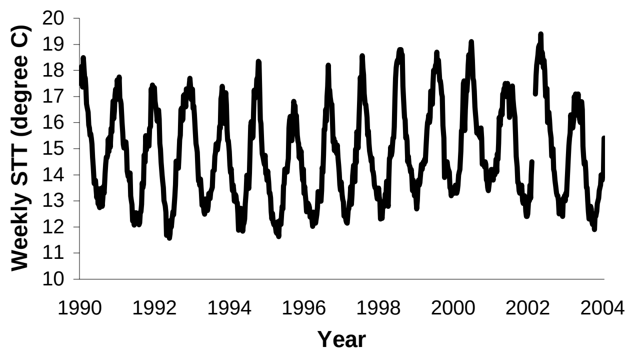

Fig. 4.30: Weekly Sea Surface Temperature (SST) from NOAA satellite AVHRR imagery at the East coast of Tasmania (42.5S 148.5E) between 1990 and 2005.

The effects of water temperature on fish growth have been studied extensively (*e.g.* Brett 1979). The relationship between temperature and growth is usually nonlinear with maximum growth at intermediate levels. In Tasmania, there is evidence of an approximate 10-year cycle in westerly wind strength (Thresher 1994). This is consistent with a general increase of weekly water temperatures along the East coast of Tasmania since the mid 1990s reaching a maximum around 2000/01 (Figs. 4.30 and 4.31). For instance, between September 1995 and February 2001, weekly Sea Surface Temperatures (SST) during spring (September to November) and summer (December to February) increased on average by 0.32ºC and 0.39ºC per year, respectively. Consistently higher water temperatures in recent years could have potentially contributed to the observed changes in growth and maturity and also may continue to do so, considering the fact that it takes a few years for a fish to become fishable. If water temperature is indeed an important driver, then growth and maturity may be expected to slowly return to the levels observed in the mid 1990s since water temperatures have now started to cool down (Fig. 4.30). Once quantified, a proven relationship between the physical environment and patterns in growth and maturity could be used as a proxy for changes of life-history characteristics when performing future stock assessment.

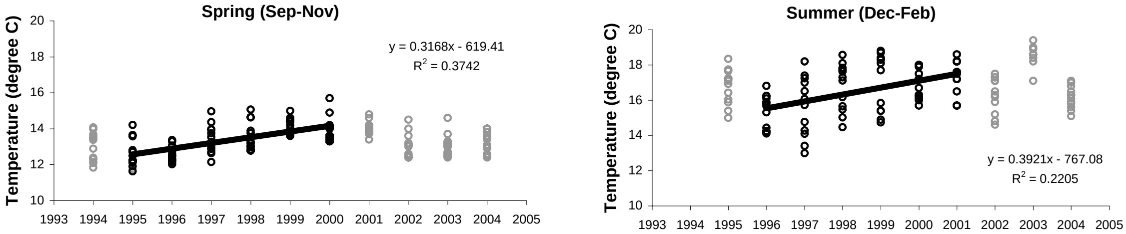

Fig. 4.31: Weekly SST at the East coast of Tasmania (42.5S 148.5E) between 1990 and 2005 and regression (black open circles and line) for spring (Sep-Nov) and summer (Dec-Feb; 1996 refers to summer 1995/96 etc.).

Density-dependent growth can also result in faster growth of individuals at smaller stock size due to decreased competition for food or habitat and has been suggested as a mechanism for some fish species (Helser and Almeida 1997; Godø and Haug 1999; Lorenzen and Enberg 2001). If this were the case for banded morwong, then the observed changes in growth and maturity would be indicative of a substantial decrease in population size. Particularly where stock assessments do not provide clear answers due to high model and parameter uncertainty, substantial changes in life-history characteristics can act as an additional indicator for stock stress.

The underlying process of such a density-dependent effect is unclear, but for banded morwong it could involve a combination of complex size-dependent social behaviour and competition for food and shelter. Banded morwong populations on a reef area are sex and size-structured. Assuming they behave around Tasmania as they do around New Zealand then after recruiting into the shallow parts of a reef, individuals tend to migrate to deeper water as they grow. This trend is more accentuated for males, resulting in a dominance of large males in deeper areas, while females are more frequent in shallower parts (McCormick 1989a). This distribution pattern differs slightly during the spawning season, when mature females also move towards deeper reefs. Fish smaller than 200 mm maintain small reef areas exclusive of similar-sized conspecifics while larger individuals are ignored (Leum and Choat 1980). Larger fish swim further and cover larger foraging areas. These areas tend to overlap between individuals, reflecting a lesser degree of exclusive feeding areas and site-associated aggressive behaviour between large individuals. However, adult males exhibit defensive territorial behaviour towards conspecifics of similar size, defending patches of reefs at least during the spawning season (McCormick 1989b).

Banded morwong are microcarnivors with both sexes relying heavily on gammarid amphipods, ophiuroids and polychaetes (McCormick 1998). Larger fish feed on larger prey than small fish due to structural changes of the mouth and feeding apparatus, but the gross taxonomic composition of the diet remains similar. In New Zealand, an abrupt change in feeding at around 250 mm standard length (SL) was found, when ophiuroids started to feature predominantly in the diet of many individuals. This change coincided with the start of gametogenesis (McCormick; unpublished data). One implication of the prey size difference is that, although juveniles and adults feed on the same prey, they effectively exploit different components of the resource, this, in conjunction with the territorial behaviour to similar sized fish, minimising intraspecific competition between different-sized fish.

Both males and females may benefit from reduced intraspecific competition and increased food availability when stock levels are reduced. Lower competition for establishment and defence of territories and numbers of intraspecific interactions may allow adult males to invest more energy into growth, maturation and mating. Reduced suppression of onset of maturity of small females by the presence of larger females, and intense pair-wise mating interactions between males and females (McCormick 1989b) could also play a role in the earlier onset of maturity for females.

A third mechanism that might have driven the changes in growth and size at maturity relates to possible genetic changes due to size-selective fishing. Increased mortality of larger fish in a population due to fishing can select for fish that mature at a younger age and smaller size, because delaying maturation decreases the probability of successful reproduction. For such a genotypic change to occur, a non-random fishing mortality has to apply in respect to the life-history characteristics, *e.g.* selection of a particular size range. Such a genetic change due to fishing pressure has been suggested for a number of species including Atlantic cod (*Gadus morhua*; Imsland and Jónsdóttir 2002, Olson *et al*. 2005) and North Sea plaice (*Pleuronectes platessa*; Grift *et al*. 2003). Once a genotypic shift has occurred, long recovery times are expected (Law 2000).

There are, however, several reasons indicating that a genotypic change in growth and maturity is unlikely for banded morwong. Firstly, the time span of just 5-10 years over which the changes occurred appears extremely short for a genotypic change. This is the equivalent to less than two generations and much less than generally advocated as being a minimum period required to observe evolutionary changes, although changes in growth have been observed within as few as 3-4 generations (Conover and Munch 2002). Secondly, while slow-growing males are more susceptible to fishing and more likely to be removed, because they remain longer within the size limits range of 360-460 mm, females are fished throughout their size range. Size-selection would therefore be expected to favour slower-growing females, which do not reach the size limits. Only during a short period in the fishery, between 1995 and 1998, were size limits smaller than the current limits and may have selected for fast-growing individuals. Thirdly, banded morwong larvae settle after a 6-month pelagic phase to coastal reefs. This long oceanic phase is likely to decouple the reef areas where larvae originate, from those areas where the larvae settle (Bruce *et al*. 2000). Thus, larvae settling on a particular reef could be a mix of fish originating from reefs with both high and low levels of exploitation, masking potential genotypic effects. Despite these arguments, a genotypic component to the observed changes in growth and maturity cannot be fully rejected. It would however, imply an extremely high fishing mortality with a substantial decrease in population numbers of banded morwong from many if not most areas around Tasmania.

Probability reaction norm analysis has been suggested as an approach to distinguish the underlying factors when changes in maturation and growth rates occur simultaneously (Stearns and Koella 1986; Heino *et al.* 2002). This analysis consists of generating an array of functions that characterise the size-dependent maturation probabilities at all relevant ages (Heino *et al.* 2002). Using this approach, Olson *et al*. (2005) have shown that changes in age and size at maturation of Atlantic cod observed over 30 years were likely to have been based on an evolutionary shift towards younger ages and smaller sizes, and are not confounded by concomitant phenotypic changes in growth or survival. In contrast, size at maturity of 5-year old male smallmouth bass (*Micropterus dolomieu*) was found to be a product of phenotypic plasticity, driven by environmentally based differences in growth rates (Dunlop *et al*. 2005). The probability reaction norm analysis for North Sea plaice indicated that size and age at maturity had significantly shifted towards younger age and smaller length over 40 years (Grift *et al*. 2003). Short-term fluctuation from plastic responses were found to be superimposed on a persistent long-term trend resulting from a genetic shift and higher body growth. Unfortunately, because reaction norms require an array of functions characterising the size-dependent maturation probabilities at all relevant ages, this approach requires far greater sample sizes for each age class than what was available in this study. Thus, we cannot quantify the contribution of factors leading to the observed changes in growth and maturity. Nevertheless, variation in phenotypic plasticity based on density-dependent processes together with an increase in water temperature seem more probable than a detectable genotypic reaction. Continuously falling water temperatures over the next years may help in estimating the relative importance of the physical environment in future samples of growth and maturity

The unexpected changes in growth rates and size and age of maturation would be uninformative for a stock assessment, if environmental factors such as increased average water temperatures were the main driver. However, if these differences do in fact prove to be density-dependent or caused by fishery selection, they would suggest a significant reduction in stock size and a precautionary approach to future management would be required.

4.6.3 Final conclusions

The need for reliable but simple estimators of stock status together with management reference points that take into account the sedentary character and the specific life-history characteristics of the species is apparent from the presented data. Because catch and effort are strongly influenced by external factors such as management changes, seal interactions and market dynamics, the current management system in Tasmania based on reference years and trends in catch, effort and catch rates, refers mainly to changes within the fishery, rather than the stocks. For catch rates to detect changes in the stock status, their spatial resolution of data collection would also needs to match that of the population dynamics. Rather than the relatively large-scale data reporting at ½ degree blocks, a much finer scale would be needed to significantly reduce the risk of masked serial depletion.

Information about the character or relative abundance of populations in the deeper reef areas or potential mixing rates with the shallower areas is also missing. If large males concentrate at greater depths as suggested by McCormick (1989a), they may receive further protection from fishing through a ‘depth refuge’, since fishing rarely occurs at depths over 25 m. Any females found in deeper water will also be less vulnerable to capture. Anecdotal reports suggest that fish in the deeper waters do in fact tend to be large specimens and fishers contend that there is replenishment of shallower fishing grounds throughout the year, possibly due to movement of fish from deeper water. This gives rise to the suggestion that a portion of the population may be protected from fishing by a depth refuge. Through mixing this could account for the continued presence of very old individuals in even the most recent research samples, despite the level of fishing pressure that has been exerted on the stocks. Unfortunately, we have no information about the structure or relative abundance of populations in the deeper reef areas or potential mixing rates with the shallower areas to evaluate the potential importance of depth refuges on the stocks. Fishing surveys of such areas and an understanding of the size and distribution of suitable deep reef habitat relative to the shallow fished reef areas could prove informative in evaluating the potential importance of depth refuges.

In Chapter 5 we discuss the development of an operating model that utilizes available biological and fishery information for banded morwong in an attempt to represent the status of the stocks around Tasmania.

CHAPTER 5: DEVELOPMENT OF THE OPERATING MODEL STRUCTURE

5.1 Introduction

Management strategies provide the most complete characterization of how particular fisheries are managed. A full description of a management strategy would include the data collection details, the stock assessment approach used, and the set of decision rules, usually based around performance measures that are used to generate management advice from the results of the assessment. Management strategy evaluation (MSE, Chapter 7) is a technique designed to permit the comparison of alternative management strategies when searching for combinations of stock assessment methods and performance measures that would be robust to a wide range of possible realities. Such a search is necessary because the natural uncertainty that arises when assessing any fishery stems, at least in part, from a lack of knowledge of the true dynamics of the underlying fish stocks. Essentially, the technique requires that a detailed model is constructed which is then used to artificially generate data of the type that is typically obtained from the fishery concerned. This detailed model is denoted the 'operating model'. It is this artificially generated data that is analysed by the different management strategies, and by comparing the assessment with the known pseudo-"reality" defined by the operating model it is possible to determine how well each management strategy is performing. For the MSE to be successfully applied to the particular fishery concerned, the operating model must be a credible representation of the biology and stock dynamics of the fish species and therefore requires substantial amounts of biological information.

Repeated biological sampling of banded morwong along the Tasmania east coast provided information about crucial life-history parameters such as growth, maturity, and large-scale movement rates (Chapter 4). From these surveys it became evident that the spatial structure of banded morwong stocks would form the major challenge for the development of the operating model. Because banded morwong rarely move between reefs and generally move less than 5 km, even over a period of years, populations are likely to differ at a reef to reef scale. This fine-scale spatial structure of the population dynamics was not represented in the commercial and biological data available for the development of the operating model. Commercial catch and effort data are reported at the scale of 30 nm fishing blocks and this large scale of data acquisition is likely to mask any processes at the scale of the reef, such as serial depletion and the shift of fishing effort when searching for the best fishing grounds. The biological samples were even more localized being collected at only a few spot locations and it is uncertain how representative they are of other sites/reefs off the east coast of Tasmania.

In full knowledge of the potential spatial mismatch of stock dynamics and available data, a compromise was needed for the development of an operating model that would simulate some spatial structuring, but not greatly exceed the limitations of data quality and quantity. The approach taken here was to develop two types of operating model, a spatially simple 1-region model and a spatially more complex 5-region model based on the stock assessment regions. These two models of spatial structure promised to deliver some insights into the limitations inherent when representing the stock dynamics with a spatially-inappropriate operating model.

In addition to the available biological information, both the 1-region and 5-region models required a range of assumptions to be made to complete the required knowledge of biological characteristics, resulting in the operating models being over-parameterized (especially the 5-region model). Under such circumstances an array of sensitivity analyses are required to test the impact of the assumptions. Usually, in an MSE the impacts of the variability inherent in the various processes on management strategy performance are explored in detail. Two major sources of variation were immediately apparent, *viz*. the usual recruitment variability and the less commonly investigated model uncertainty. Because the spatial structure and associated dynamics were so poorly known, it was necessary to set up a balanced design of alternative combinations of operating model designs and compare the outcomes of trying to assess data from the various alternatives.

5.2 Methods

5.2.1 Model structure

An age- and spatially-structured operating model was developed and fitted to biological and commercial catch and effort data for banded morwong between 1990 and 2004 from the east coast of Tasmania. A detailed description of the model structure can be found in Appendix 3.

A statistical catch-at-age population model was constructed (Quinn and Deriso 1999; Haddon 2001) that sub-divided the available biomass into discrete populations within regions. This allowed for spatial structure in the population dynamics and fishing mortality. Different scenarios of stock sub-division were tested to simulate the effects of spatial structuring on the fish stock and its assessment. All scenarios treated the east coast of Tasmania banded morwong population as a single biological stock, even though the population may have been sub-divided into a meta-population. This was supported by the knowledge that there is an extended pelagic larval stage (up to 6 months) suggesting that, despite the lack of adult movement, larval mixing would homogenize the population genetically.

The operating model simulated either a single or five banded morwong region(s) along the east coast of Tasmania following the fishing blocks and regions used in the stock assessment (Fig. 5.1). The number of regions used in the model was a compromise between maximum spatial resolution, data availability and fishery dynamics. In addition, the data was often so limited and noisy that blocks had to be pooled in order to obtain a workable set of data. The decision to combine two fishing blocks along the coast into the Bicheno and Tasman regions was based on the fishery dynamics, with sections of both blocks often being fished within a single day.

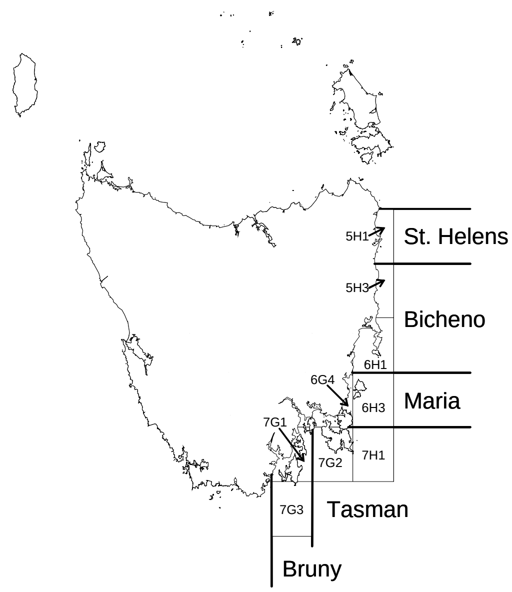

Fig. 5.1: Tasmanian east coast with fishing blocks and the 5 regions along the east coast of Tasmania used in the operating model. Some catch from the Flinders Island region was taken in 2002 and 2003 but catches from this region have since declined again and the area was not included in the modelling.

5.2.2 Biomass distribution

The fishery-dependent and independent data from the separate assessment regions was insufficiently informative to permit an estimate of the biomass in each region individually. Therefore, the biomass was estimated for the overall east coast and then distributed among the regions. This increased the stability of each regional biomass estimate, especially where the available data was limited and noisy such as that from the Bruny region. The area of reef habitat or fished reef area was used as a proxy for the relative biomass in each region, assuming a close link between fish density and reef habitat (inclusive of high profile, medium profile, low profile and patchy reef habitat types). Habitat maps were used for the Bruny, Tasman and Maria regions, where estimates of total habitat area down to a depth of 40 m were available (SEAMAP Tasmania Project, http://www. utas.edu.au/tafi/seamap/).

To supplement the mapping information and to estimate the reef area in areas without habitat maps, banded morwong fishers were asked to identify on charts the reefs that are fished for banded morwong or that were believed to be suitable fish habitat. These markings were read into the ArcView GIS program to estimate the total habitat area in each region (Table 5.1). Most fishers seemed to overestimate the area of fished reefs, as in the Maria region, where the fisher's estimate of 70 km^2 was substantially higher than the mapped area of 45 km^2. Estimates in the Bicheno and St. Helens regions were also assumed to be overstated. For a best estimate of the reef area, these fisher's estimates were adjusted to an arbitrary 80% of the original estimates. The reef area in the Bruny region fished by banded morwong fishers is significantly smaller than the total reef area, presumably due to unproductive bottom or different habitat structure particularly in the south of the region. The actual reef area relevant to the fishery was therefore limited to that in the northern part of the region. This process was assumed to result in a best estimate of reef areas in each of the five regions (Table 5.1).

Table 5.1: Estimated reef areas (km2) taken which are fished or believed to be suitable habitat for banded morwong, including best estimates of reef areas.

The % column relates to the proportion of the whole coast considered.

Region	Habitat Mapping		Fisher's estimates		Best estimate		Proportion
	Fished (<25 m)	Total (<40 m)	Fished (<25 m)	Total (<40 m)	Total	%	Onshore (<25m)
St. Helens				77	62 [4]	0.16	
Bicheno				244	195 [4]	0.51	
Maria		45		70	58 [3]	0.15	
Tasman	33	47	26		47 [2]	0.12	0.70
Bruny	35	45			23 [1]	0.06	0.78

[1] 50% of reef area, since the southern parts are not fished and seen by fishers as unsuitable habitat
[2] Mapped reef area
[3] Mean of reef area mapped and estimated by fisher
[4] 80% of reef area estimated by fisher

Given the high degree of uncertainty in the parameter estimates of regional distribution of fish biomass, three different scenarios of regional biomass distribution were examined in the operating model, with the contribution of the Bicheno region ranging from 30-65% and equal contributions from the Tasman, Maria and St. Helens regions at levels that complemented that in Bicheno (see Table 5.3).

The mismatch between the depth range of the species distribution and that of the fishery added complexity and uncertainty to the model structure. In order to minimize effects of barotrauma and thus maximize fish survival, fishing for banded morwong is largely restricted to depths of less than about 25 m. Because the distribution of banded morwong extends beyond the depth of the live-fish fishery, there is the potential for an unfished component of the stock in a depth refuge. The model allowed for this by specifying a fished population onshore in depths up to 25 m and an unfished population offshore in depths greater than 25 m with movement occurring between them.

In the Tasman and Maria regions, where depth contours were available for the mapped reefs, approximately 70-80% of the reef habitat was found onshore within depths of 0-25 m, and 20-30% offshore within depths of 25-40 m (Table 5.1). The extent of reef areas beyond 40 m depth was unknown. For the operating model, three different scenarios were examined, ranging from all biomass in the onshore population and available to the fishery to only 50% onshore.

These biomass distributions were combined with different movement rates. Two types of movement were distinguished in the model; (i) movement alongshore between regions and (ii) movement between onshore and offshore populations. Tag-recapture data indicated low movement rates between regions (Chapter 4). For the operating model, 0.1% of all fish in a region were assumed to move to the adjacent region each year. Movement is modelled as occurring at the end of each year and is restricted to mature fish.

Movement rates between onshore and offshore populations could not be determined from tag-recapture data due to a limit to the precision of information with respect to location and depth. Instead, these movement rates were an input parameter in the model. Movement between onshore and offshore populations is a combination of mobility, m, defined as the proportion of the mature population that becomes vagrant or mobile and becomes capable of shifting from each population to adjoining populations, combined with π_i, the proportion of habitat or biomass in each population (see Appendix 3). Thus, the movement rate from population p into population p+1 can be represented as $m\pi_p$. Population p retains $1-m\pi_{p+1}$ of its total, but gains $m\pi_p$ of the neighbouring population p+1. If the proportion of habitat is equal (*i.e.* 50:50) then the movement rate equals the mobility, however, if the proportional distribution of the population deviates from 50:50 then the movement rates will become asymmetric. Different mobility rates ranging from 100% to only 25% of all fish being mobile each year were tested in the operating model. Table 5.3 shows an overview of all model scenarios examined.

The model operated on an annual time step and was fitted to 15 years of commercial catch and effort data from the earliest catch reports available in 1989/90 to 2003/04 inclusive. It was assumed that each region and population was at a pre-exploitation equilibrium with the corresponding age and sex structure prior to 1989. While the live-fish fishery only started in that year, mainly lobster fishers had previously taken banded morwong for use as bait in their traps. Although the quantity taken each year is unknown, it was believed, from discussions with rock lobster fishers, to have been low compared to that taken by the targeted fishery in the mid 1990s.

The fishing year was defined to start on the 1st of May and finish on the 30th of April the following calendar year in order to bring it in line with the spawning season and corresponding closure for both the commercial and recreational fisheries between 1st of March to 30th April. For the purposes of the operating model, 1995/96 is taken to be 1996, 1996/97 to be 1997 and so on. The biological catch sampling also fitted into this dating scheme as it occurred during the spawning closure.

5.2.3 Biological components of the operating model

Biological characteristics such as growth, age at maturity and natural mortality were assumed to be the same in each region and known without error. Length at age was modelled by a sex-specific 2-phase von Bertalanffy growth function (Chapter 4). The growth patterns were simplified and a single growth function used across all regions in any given year. Three periods with different specific growth and maturity curves were used; these were 1990-1998, 1999-2001, and 2002-2004. To achieve this, all available data were pooled by sex across regions and for the periods 1996-97, 2001, and 2002-03, respectively. Sex-specific maximum sizes were assumed to be constant across all years. To represent this, all data for males and females older than 20 years were pooled by sex and used in all model fits, such that all upper VB functions would converge. The growth and maturity data from 1996-97 were attributed to the operating model years 1990-98, that from 2001 was applied to 1999-2001 and that from 2002-03 was applied to 2002-04. In this way, the increased growth rates and decreased size and age at maturity found during the period (Chapter 4) were incorporated into the model.

Selectivity at age or the number of fish at a given age available to the fishery varied between males and females due to the combination of differences in growth pattern by sex and the keyhole size limits (Fig. 5.2). Females recruit to the fishery at the age of 4-6 years and typically remain vulnerable to fishing for the rest of their lives, while males grow through the size limits within a few years, mainly between 4-10 years. This means that the sex ratio is sensitive to changes in the age structure caused by either heavy exploitation and/or strong recruitment.

The selectivity at age function was determined from the length at age function in conjunction with growth variation, the gear selectivity and the keyhole size limits. Mesh selectivity and vulnerability were assumed constant between sexes; the former based on the similar body shape of males and females, the latter assuming similar behaviour of males and females, in particular similar swimming activity within their home range and larger-scale movement between depth strata and regions. This is probably an over-simplification given the population structuring by sex and size and associated movement patterns observed by McCormick (1989a and 1989b).

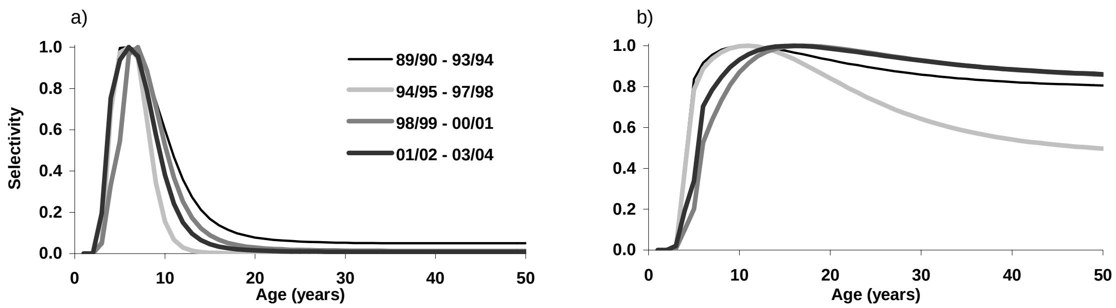

Fig. 5.2: Selectivity at age for (a) male and (b) female banded morwong for mesh size 133 mm, 330-480 mm size limits and growth derived from 96/97 sampling (light black line); mesh size 137 mm, 330-430 mm size limits and growth derived from 96/97 sampling (heavy light grey line); and mesh size 137 mm, 360-460 mm size limits and growth derived from 2001 sampling (heavy dark grey line) and 2002-03 sampling (heavy black line).

Because growth accelerated and the size limits have changed over the years of the study, four different selectivity periods were distinguished (Fig. 5.2). No size limits were in place until 1994 and the mesh size used was predominantly 133 mm. These mesh sizes rarely catch fish smaller than 330 mm and, due to a lack of demand for larger fish, those over approximately 480 mm were usually returned. This dynamic was reflected in the keyhole size limit of 330-430 mm fork length that was introduced for the fishing season 1994/95. In addition, fishers started to use gillnets with 140 mm mesh size as well as those with 133 mm mesh size. Thus, for all periods with size limits, an average mesh size of 137 mm was assumed for the modelling. The size limits were revised in 1998 and minimum and maximum sizes were increased by 30 mm to 360-460 mm for the 1998/99 fishing year. The growth acceleration found in the catch samples between 2001 and 2003 impacted on the selectivity curve and was also assumed to have started in 1999.

Recruitment of two year old fish was fitted in the model and occurred at the start of each year with an equal ratio of males and females. With a long larval phase of around 4-6 months, recruitment was assumed to be uniform along the coastline and only dependent on the reef habitat available for settlement, *i.e.* recruitment in each region was assumed to be proportional to its area. All recruitment in the model occurred to onshore populations, because juveniles are predominantly found in shallow waters and recruitment to shallow waters is followed by a gradual outward migration with increasing size (Leum and Choat 1980; McCormick 1989). Three different scenarios with a range of recruitment variability from little variation ($\sigma = 0.1$) to large variation ($\sigma = 0.6$) were examined (Table 5.3).

5.2.4 Harvest components of the operating model

Commercial catch

Catch and effort data for the operating model is based on commercial catches, the recreational fishery for banded morwong in Tasmania is very small (Lyle 2005).

Commercial catch analysis has been based primarily on catch and effort information provided by compulsory logbook returns to the Tasmanian Department of Primary Industries and Water (DPIW). Prior to 1995, catch returns were based on monthly landings by species and one degree fishing blocks fished were reported. Subsequent catch returns provide daily summaries of fishing operations, including method, location (based on ½ degree fishing blocks), fishing depth, effort, catch weights and whether seal interference had occurred. All catch returns are self-reported and unverified, and therefore their accuracy is uncertain.

Fishers and processors generally believe that catches were substantially over-stated before the introduction of live-fish endorsements in 1996 and that data quality was variable at least up until about 2000 (Chapter 3). Particularly, the very large catches reported in 1993/94 and 1994/95 appear inflated. In discussion and agreement with banded morwong processors and key fishers, a more plausible catch history was estimated for the east coast. Three data sources were evaluated:

- Fishers and processors repeatedly and independently estimated the peak market capacity (landings of live and dead fish) during the early 1990s to between 60 and 70 tonnes per annum.
- The logbook returns have been adjusted by manually reducing unlikely high catch returns particularly in the early 1990s to more realistic levels (Fig. 5.3). This correction reduced the peak of total catches in 1993/94 from 131 to 90 tonnes, but had little effect on catches after 1994/95. Because only extreme catch returns were addressed, these corrected logbook totals are still likely to be inflated compared to the real landings. However, fish mortality was reported to be comparatively high during this period due to poor practices of handling and maintaining fish after capture in tanks and cauffs.
- Processor returns were estimated based on the monthly receipts of a major live-fish processor extrapolated for the assumed activity of other known Tasmanian processors (Fig. 5.3). The resulting estimates are, however, likely to underestimate the true live catch of banded morwong up until 1998/99 since unknown quantities of product was transported directly to the mainland via tanker trucks. In addition, this analysis does not account for handling mortality or product sold dead or used as bait.

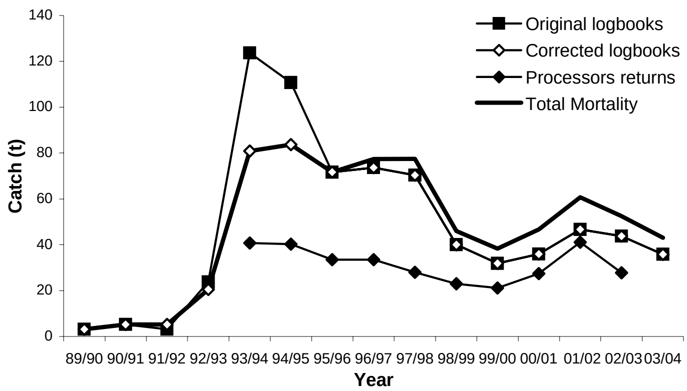

Fig. 5.3: Banded morwong catch history used in the operating model based on unadjusted logbook returns (filled squares), corrected returns (open diamonds), extrapolated processor returns (filled diamonds), and estimated total mortality of banded morwong (heavy line) between 1989/90 and 2003/04.

In developing an agreed catch history, the corrected logbook returns were assumed to be the best approximation of total fishing-related mortality, implying that over-reporting roughly equalled handling mortality and that both have been significantly reduced in recent years.

Since the early 1990s seals have become a growing problem for the industry, with a substantial impact on the fishing operation and increased incidental mortality through damage and loss of fish from the nets. In contrast to the handling mortality, some of this mortality occurs largely 'out of sight' and is therefore very difficult to quantify. A scaling factor to account for seal-related mortality was added to the corrected logbook returns, continuously increasing since 1996/97 and peaking in 2000-2002 (Table 5.2).

Table 5.2: Assumed seal mortality factor in the operating model.

Year	Factor	Year	Factor
1996/97	1.05	2000/01	1.30
1997/98	1.10	2001/02	1.30
1998/99	1.15	2002/03	1.20
1999/00	1.20	2003/04	1.20

Annual catches were estimated from the corrected logbook returns. For catches taken prior to 1994/95 the one degree fishing blocks do not correspond with the regional definitions applied in this study. Specifically fishing blocks 5H, 6H and 7G covered two regions and catches between 1989/90 and 1993/94 needed to be assigned to St. Helens/Bicheno, Bicheno/Maria Island, and Tasman Peninsula/Bruny Island regions respectively. Because fishing had started earlier in the Bicheno region than in the St. Helens region, a gradually increasing proportion of the catches of block 5H from 0% in 1991/92 to 17% in 1994/95 was assigned to St. Helens. Maria Island was assigned 49% of block 6H, and Bruny Island 23% of block 7G, based on their catch proportions between 1994/95 and 1997/98.

Catch rates

Standardised catch rates since 1995/96 were used as input variables in the operating model (section 4.3.3). Because of identified problems in the way many fishers have interpreted effort reporting requirements in logbooks (Chapter 3), effort and catch rates were defined as days fished and kilograms per day, respectively.

Catchability

Catchability was estimated by the model from the relationship of observed catch rates and exploitable biomass. The catchability coefficient $\hat{q}$ was thereby assumed to be constant and each annual $\hat{q}_y$ to be only an estimate of the overall $\hat{q}$. However, the banded morwong catch and effort data indicated that catchability differed between 1996-1999 and 2000-2004, and thus two catchability coefficients were estimated. This change could have been caused by a combination of new management regulations implemented in late 1998 and increasing interactions with seals around this period. The former resulted in some restructuring of the fishing fleet, restrictions on the amount of gear used and increased in the size limits. As a result of growing interactions with seal there were substantial changes in fishing practices and losses to seal predation.

5.2.5 Uncertainty of model and parameter estimates

The model was conditioned on commercial catch data and fitted using maximum log likelihood methods on observed standardised catch rates, catch age-composition, and sex ratios. Contributions to the log likelihood of the model fits to catch rate, sex ratio and age composition data was weighted with inverse proportion to their respective variation (*i.e.* less weight to the more variable). Estimated model parameters, 30 in total, were the equilibrium age composition at the start of the first year in the population along with recruitment levels in each year of the fishery.

Uncertainty in relation to spatial resolution of the banded morwong stocks was addressed by examining a range of models based on 1 or 5 regions, and an onshore population only or a combination of onshore and offshore populations with movement in between (Table 5.3). Three different levels of onshore/offshore distribution of biomass and four levels of mobility rates were tested. For the models with 5 regions, the regional biomass distribution was addressed by three different scenarios which were based on the relative habitat area estimates. In total there were 27 possible model scenarios in the 1-region model and 81 model scenarios in the 5-region model. Each of these were considered and compared. The model fits were deterministic for any given scenario and compared based on their log likelihood values.

Instead of a pre-defined stock-recruitment relationship such as the Beverton-Holt relationship, recruitment of 2 year olds, which was fitted in every year, was constrained by a penalty term contributing to the log likelihood. This penalty term was related to the recruitment variability, which was varied from low variability ($\sigma = 0.1$) to high variability ($\sigma = 0.6$).

Table 5.3: Alternative Operating Model configurations: Spatial structure, parameters and parameters tested.

Model structure	Parameters	Parameter values
1 or 2 populations	Recruitment variability	$\sigma = 0.1, 0.2, 0.6$
(1 region with on/offshore)	Biomass onshore	BtOn = 1.0[1], 0.75, 0.5
	Movement rates on/offshore	Move = 1.0, 0.75, 0.5, 0.25
5 or 10 populations	Regional biomass distribution[2]	A: 0.1 - 0.65 - 0.1 - 0.1 - 0.05
(5 regions with on/offshore)		B: 0.15 - 0.5 - 0.15 - 0.15 - 0.05
		C: 0.2 - 0.3 - 0.2 - 0.2 - 0.1
	Recruitment variability	$\sigma = 0.1, 0.2, 0.6$
	Biomass onshore	BtOn = 1.0[1], 0.75, 0.5
	Movement rates on/offshore	Move = 1.0, 0.75, 0.5, 0.25

[1] Movement rates not applicable
[2] Proportion of regional biomass in the St. Helens, Bicheno, Maria, Tasman and Bruny regions

5.3 Results

5.3.1 One-region model

The quality of the model fits was largely determined by the parameters set for both the recruitment variability and depth distribution of biomass combined with movement rates between onshore and offshore populations (Table 5.4). Higher recruitment variability improved the fit of the models to available data. This was particularly due to an improved representation of the age composition. The results from model fits with recruitment variability $\sigma = 0.2$ were intermediate or very similar to those with $\sigma = 0.1$ or 0.6. Therefore, results from recruitment variability $\sigma = 0.2$ are usually not presented.

With each given recruitment variability, the best fit was generated when the biomass onshore was set to 50% combined with the lower movement rates of 0.5. However, overall intermediate depth biomass distributions and movement rate scenarios were very similar (Table 5.4). While the models closely matched the changes in catch rates, none of the model scenarios managed to describe the sudden and substantial drop of sex ratios from around 0.75 to 0.5 in the early 2000s (Figs. 5.4 and 5.5). Given this a judgement concerning which of these scenarios is best or most plausible, seemed inappropriate despite (significant) differences in the log likelihood.

Table 5.4: The maximum likelihood contributions for each data source when the model was fitted to the observed data in the 1-region model for all scenarios tested.

Different levels of biomass onshore (BtOn), movement rates between onshore and offshore populations, and levels of recruitment variation σ with the contribution of catch rates (CE), sex ratio (Sex) and age structure (Age) to the objective function and the value of the objective function itself (Total). Best model (with smallest log likelihood) in bold.

BtOn	Movement rate	Sigma = 0.1				Sigma = 0.2				Sigma = 0.6			
		CE	**Sex**	**Age**	**Total**	**CE**	**Sex**	**Age**	**Total**	**CE**	**Sex**	**Age**	**Total**
1	**N/A**	-48.7	-14.0	181.8	125.9	-51.4	-14.5	158.3	108.3	-56.6	-14.7	150.2	84.3
0.75	**1**	-48.7	-14.0	181.8	125.9	-51.4	-14.5	158.3	108.3	-56.6	-14.7	150.2	84.3
0.75	**0.75**	-48.7	-14.0	181.8	125.9	-51.4	-14.5	158.3	108.3	-56.6	-14.7	150.2	84.3
0.75	**0.5**	-48.7	-14.0	181.8	125.9	-51.4	-14.5	158.3	108.3	-56.6	-14.7	150.2	84.3
0.75	**0.25**	-49.8	-13.3	185.2	127.1	-53.6	-13.9	163.6	110.2	-55.8	-14.3	153.6	88.5
0.5	**1**	-50.9	-13.2	180.6	123.7	-54.7	-13.8	162.6	108.3	-55.2	-14.3	153.8	89.9
0.5	**0.75**	-50.8	-13.1	180.3	123.0	-54.6	-13.7	160.3	106.6	-59.5	-14.0	152.6	84.2
0.5	**0.5**	-51.1	-12.9	176.7	119.9	-55.0	-13.5	158.6	104.5	**-63.0**	**-13.7**	**153.8**	**82.3**
0.5	**0.25**	-51.6	-12.2	187.9	128.8	-54.0	-12.8	171.7	116.1	-56.2	-13.2	162.9	98.0

The optimum model fits to the age composition data were obtained when high recruitment variability was assumed (Fig. 5.6). However, the large variations in age-structure between years, which is why large recruitment variability was required, may have been at least partly an artefact of the relatively small sampling sizes. Again, with any given recruitment variability, the fit of the model varied little between the different onshore biomass and movement rate scenarios (Table 5.4).

The number of males older than 15 years ('16+'age class) were badly overestimated by all models (Fig. 5.6). This could have been caused by a poor estimate of the selectivity of the gear for these large fish. Alternatively, the fact that adult males tend to move less than females (McCormick 1989a) may make them less susceptible to capture than females, and this distinction was not included in the model. In addition, most males in the plus group are oversized and usually released after capture. Because the model has no discard mortality function, the expected numbers of these males may be inflated. Notwithstanding this, being oversized and male, these fish had no effect on estimates of both the (female) mature biomass and harvest rates.

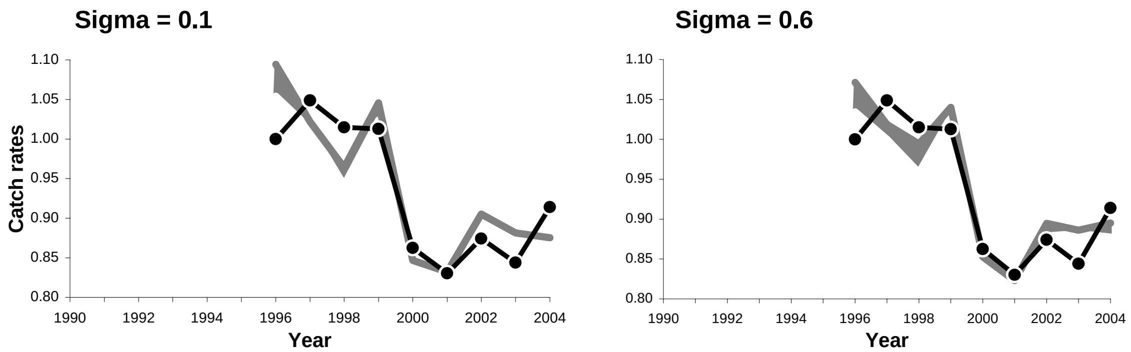

Fig. 5.4: Catch rates for recruitment variability $\sigma = 0.1$ and 0.6: Observed standardised catch rates (black circles and heavy line) and the full range of model predictions (grey) for all onshore biomass and movement rate scenarios in the 1-region model between 1990-2004. Which biomass and movement rate scenario is used clearly has only minor effects relative to the stock dynamics implied by the catch rates.

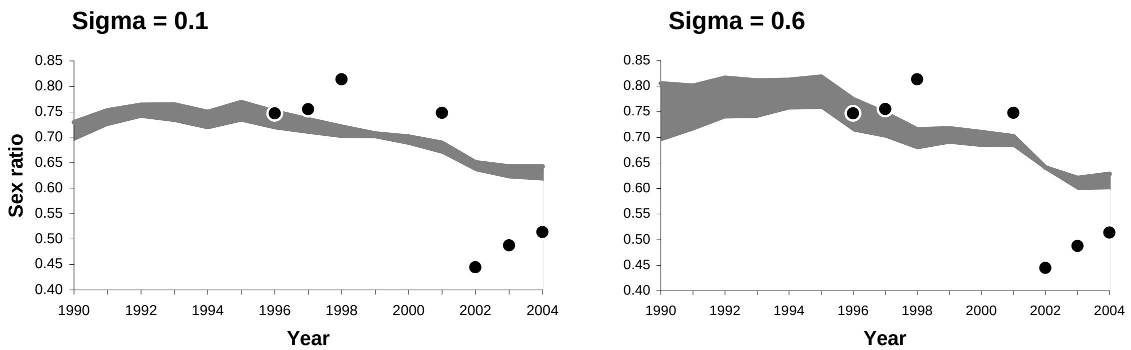

Fig. 5.5: Sex ratio as proportion of females in commercial catch for recruitment variability $\sigma = 0.1$ and 0.6. Observed sex ratios (black circles) and range of model predictions (grey) for all onshore biomass and movement rate scenarios in the 1-region model between 1990-2004.

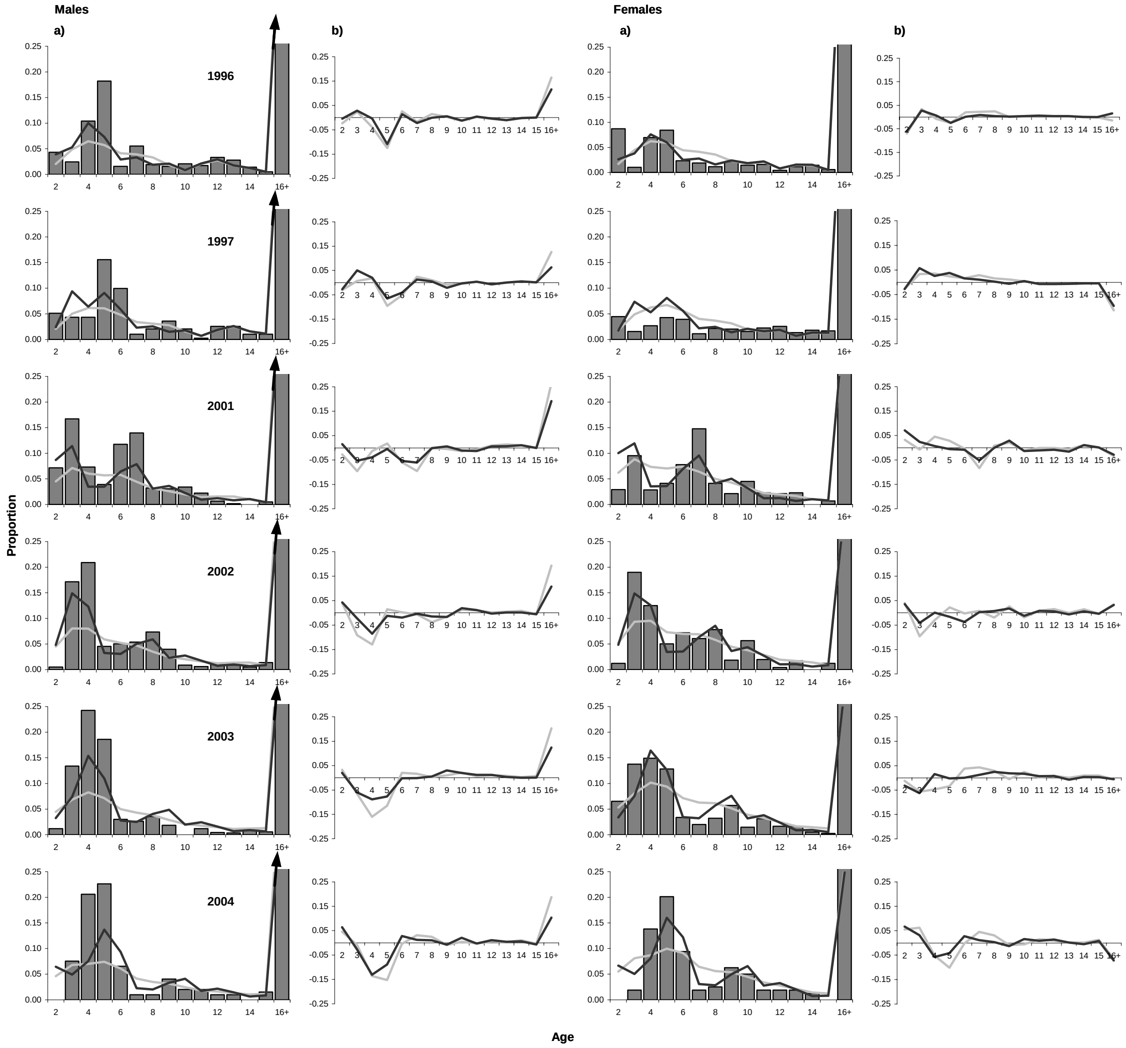

Fig. 5.6: Age composition in relative proportions of male and female banded morwong in biological samples from 1996, 1997, and 2001-2004 in the 1-region model with all biomass onshore and therefore available to the fishery: (a) Observed data (columns) and model fits for recruitment variability $\sigma = 0.1$ (grey line) and $\sigma = 0.6$ (black line), and (b) model residuals. Proportions limited to 0.25, which has cut the predicted proportion of 16+ males and females in all the graphs (see the residual plot for a more realistic representation).

Estimates of mature biomass and harvest rates were surprisingly similar between the recruitment scenarios, but varied considerably between the different onshore biomass and movement rate scenarios (Table 5.5). This was mainly a reflection of the amount of fish available to the fishery in the onshore population, with higher estimates for mature biomass, $MatB_{90}$ and $MatB_{04}$ and onshore harvest rates when more biomass was offshore protected from the fishery. The estimates of the ratio of mature biomass in 2004 relative to 1990, $MatB_{04/90}$, varied between 0.51 and 0.68, and estimates of H_{04} onshore between 0.10 and 0.16. Total H_{04} of the onshore and offshore populations followed the reverse trend because of the protected offshore population and estimates varied between 0.05 and 0.10.

Table 5.5: Model estimates for all onshore biomass (BtOn) and movement rate scenarios in the 1-region model for recruitment variability σ = 0.1, 0.2 and 0.6.

Results of mature biomass (in tonnes) in 1990 ($MatB_{90}$) and 2004 ($MatB_{04}$), biomass levels in 2004 relative to virgin levels in 1990 ($MatB_{90/04}$), harvest rate in 2004 for the onshore population (H_{04} onshore) and the total onshore and offshore populations (H_{04} total), and geometric mean of annual recruits (in numbers) between 1990-2004.

Bt On	Movement rate	$MatB_{90}$ (t)	$MatB_{04}$ (t)	$MatB_{04/90}$ (t)	H_{04} (onshore)	H_{04} (total)	Recruits
Sigma = 0.1							
1	N/A	645	345	0.54	0.10	0.10	84764
0.75	1	676	387	0.57	0.12	0.09	90003
0.75	0.75	695	401	0.58	0.11	0.09	90696
0.75	0.5	734	431	0.59	0.11	0.08	92734
0.75	0.25	784	456	0.58	0.10	0.08	91720
0.5	1	771	479	0.62	0.14	0.07	99124
0.5	0.75	819	541	0.66	0.13	0.07	105439
0.5	0.5	918	623	0.68	0.11	0.06	112155
0.5	0.25	1112	714	0.64	0.10	0.05	107930
Sigma = 0.2							
1	N/A	675	343	0.51	0.10	0.10	77784
0.75	1	713	384	0.54	0.12	0.09	82786
0.75	0.75	719	397	0.55	0.12	0.09	84091
0.75	0.5	744	415	0.56	0.11	0.08	85005
0.75	0.25	796	445	0.56	0.10	0.08	85086
0.5	1	772	453	0.59	0.15	0.08	89954
0.5	0.75	858	534	0.62	0.13	0.07	98732
0.5	0.5	953	618	0.65	0.11	0.06	104554
0.5	0.25	1059	663	0.63	0.10	0.06	98874
Sigma = 0.6							
1	N/A	685	348	0.51	0.10	0.10	73092
0.75	1	721	377	0.52	0.12	0.09	75051
0.75	0.75	754	402	0.53	0.12	0.09	78586
0.75	0.5	743	402	0.54	0.11	0.09	77655
0.75	0.25	759	422	0.56	0.11	0.08	78813
0.5	1	779	439	0.56	0.16	0.08	81024
0.5	0.75	824	501	0.61	0.14	0.07	89594
0.5	0.5	882	536	0.61	0.13	0.07	89453
0.5	0.25	997	627	0.63	0.11	0.06	91098

Mature biomass had fallen in all scenarios since 1990, mainly due to the high catches in the mid 1990s, and stabilised since 1998 with only a slight decreasing trend since (Fig. 5.7). This is despite (or thanks to) to increased productivity in the population due to higher growth rates and earlier maturity of young fish combined with a reduced overall catch.

Harvest rates were similar for all recruitment variability scenarios and peaked around 1998. Harvest rate reached between 0.19 and 0.29 in the fished population, which is equivalent to F = 0.21 to 0.34, and total harvest rate across both the onshore and offshore population reached 0.10 to 0.19, equivalent to F = 0.11 to 0.21. Model estimates of fishing mortality tend to be higher than spawning biomass-per-recruit analysis estimates for $F_{40\%}$ = 0.07 or even $F_{30\%}$ = 0.12 (Chapter 4). Only recent harvest rates appear to be within the ranges of $F_{40\%}$ or $F_{30\%}$.

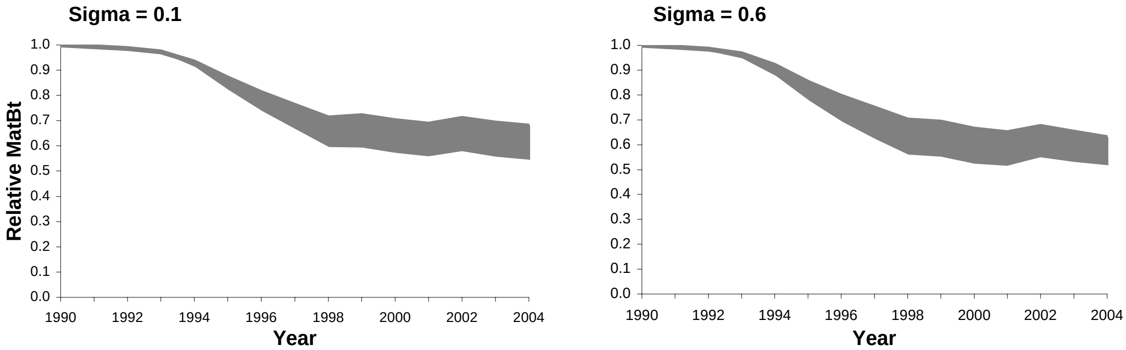

Fig. 5.7: Range of mature biomass estimates relative to the levels in 1990 for recruitment variability σ = 0.1 and 0.6, and all onshore biomass and movement rate scenarios in the 1-region model between 1990-2004.

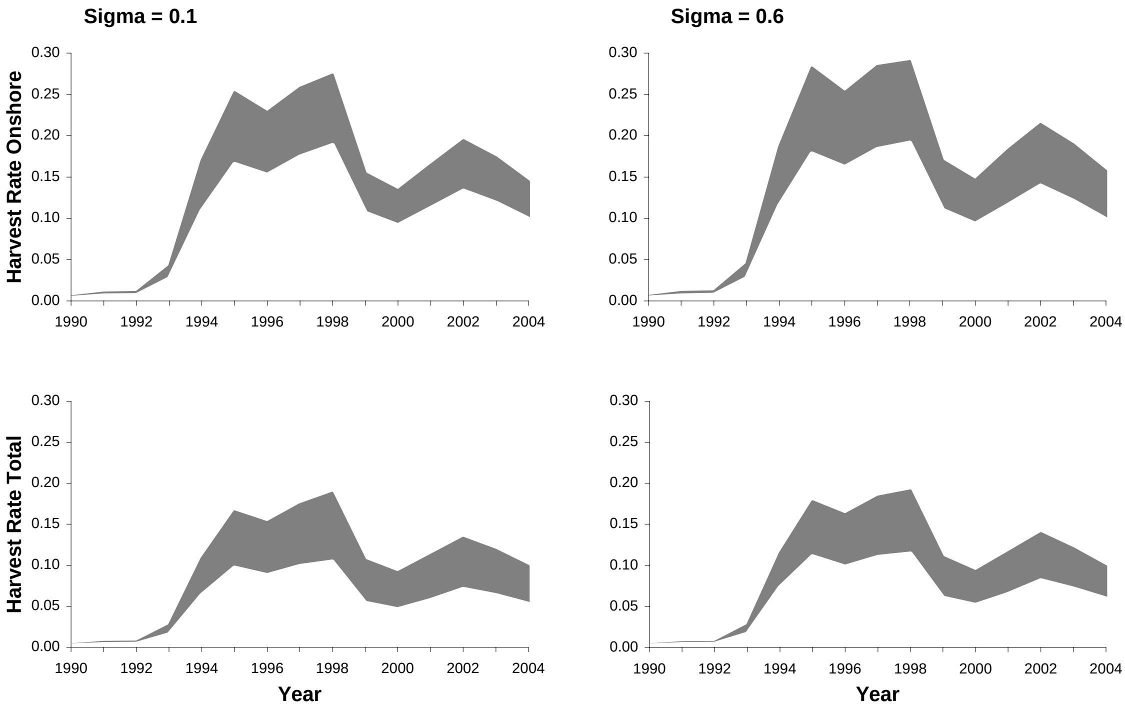

Fig. 5.8: Range of harvest rate estimates in the onshore population and the total onshore and offshore stock for recruitment variability σ = 0.1 and 0.6, and all onshore biomass and movement rate scenarios in the 1-region model between 1990-2004.

Annual recruitment residuals were fitted by the model in the context of the variation permitted as controlled by the penalty term for recruitment variability. In all scenarios, recruitment variation appeared to increase with decreasing population size over the years (Fig. 5.9).

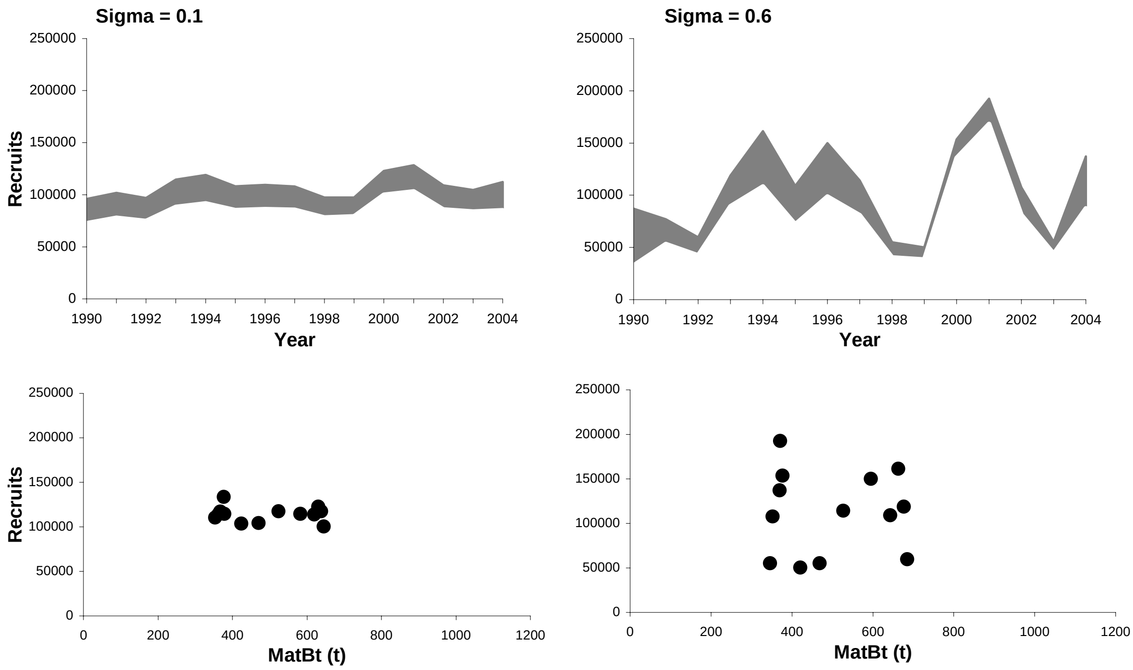

Fig. 5.9: Range of annual total recruitment estimates (in numbers) for recruitment variability $\sigma = 0.1$ and 0.6, and all onshore biomass and movement rate scenarios in the 1-region model between 1990- 2004; and stock-recruitment relationship between mature biomass (in tonnes) in year *y* and recruits (in numbers) in year *y*+2 for scenario with 50% biomass onshore and movement rates of 0.5 (best model).

5.3.2 Five-region model

When the spatial resolution of the model is increased from one region to five regions, different regional trends in model estimates emerged. Again, higher recruitment variability improved the model fits mainly due to better representation of the age composition (Table 5.6).

Different regional biomass scenarios fitted the data with varying success. Scenario C, which has the most balanced distribution of regional biomass amongst the regions, always performed best (Table 5.6). In comparison to the 1-region model, the onshore biomass and movement rate scenarios had stronger influence on the fit to the age composition. Scenarios with a high proportion of the biomass onshore generally performed better than those with little biomass onshore. The best models were generated when the biomass onshore was set to 100% or at 75% combined with the lower movement rates of 0.25. The results of the nine different combinations of onshore biomass and movement rate are illustrated as ranges on subsequent graphs to help identify their contributions to the overall uncertainty surrounding the model outcomes.

Some important data characteristics were poorly simulated by all model scenarios including scenario C. While the log likelihood indicated that the model fits to catch rates and sex ratios were similar between all tested scenarios, the graphs support this only for catch rates (Fig. 5.10, scenarios A and C with recruitment variability $\sigma = 0.6$ are presented here as examples). The substantial drop of females in the early 2000s and resulting lower sex ratios in the catch, however, is not equally represented by all regional biomass models (Fig. 5.11). For example, while in scenario A the predicted sex ratio decreases to approximately similar levels as the observed levels in the St. Helens and Tasman regions, it remains high in the Bicheno regions. In scenario C, the reverse is the case. The fit to the age composition was again relatively poor in all scenarios, yet best in the scenarios with high recruitment variability (Fig. 5.12). Given these poor fits at regional levels, a definite choice for the best or most likely model seemed again unjustified.

The dynamics of the spatial model are primarily determined by the trade-off or balance between new recruitment and movement from offshore to onshore populations. Not surprisingly, as recruitment variability is allowed to increase the importance of onshore movement from unfished areas reduces in importance. This process is obscured by the biomass distribution scenario. When the biomass is most evenly distributed (scenario C), the quality of fit is greatly influenced by the immediately available biomass (*i.e.* the onshore biomass). The optimum model fits (within 1.94 of each other; Haddon, 2001) for the lowest permitted recruitment variability are found with 75% of biomass onshore and mobility rates of 0.25 or 0.5. As recruitment variability is allowed to increase, the optimum conditions of biomass distribution and mobility alter to 100 or 75% onshore and either no mobility or only 0.25 (Table 5.6). The differences between scenarios A and B illustrate that with the mostly strongly aggregated distributions (scenario A) recruitment variation has little effect relative to the biomass distribution (all recruitment variation options indicate best fits with only 50% of biomass onshore and 0.25 mobility). While in Scenario B, intermediate between A and C, as recruitment variability is permitted to rise to 0.6 one of the optimum model fits now relates to a biomass distribution of 75% onshore with 0.25 mobility. These interactions between the various influences (regional biomass distribution, onshore/offshore biomass distribution, recruitment variability, and mobility) cannot be clarified further without much more information because of the over-parameterization of the operating model. The temptation is to select the overall optimum model fit but without further evidence concerning recruitment variability and biomass distributions to move around the model uncertainty selecting the optimum model purely on quality of fit would not necessarily be correct.

Table 5.6: The maximum likelihood contributions for each data source when the model was fitted to the observed data in the 5-region model for all scenarios tested.

Different levels of biomass onshore (BtOn), movement rates between onshore and offshore populations, and levels of recruitment variation σ with the contribution of catch rates (CE), sex ratio (Sex) and age structure (Age) to the objective function and the value of the objective function itself (Total). Results for recruitment variability σ = 0.1, 0.2, and 0.6, and regional biomass distribution of scenario A: 0.1 - 0.65 - 0.1 - 0.1 - 0.05; scenario B: 0.15 - 0.5 - 0.15 - 0.15 - 0.05; and scenario C: 0.2 - 0.3 - 0.2 - 0.2 - 0.1. Best model (smallest log likelihood) in bold, when two model fits are within 1.94 of each other then by likelihood ratio tests they would not be considered statistically different.

		Sigma = 0.1				Sigma = 0.2				Sigma = 0.6			
Bt On	Movement rate	CE	Sex	Age	Total	CE	Sex	Age	Total	CE	Sex	Age	Total
Scenario A													
1	N/A	-136.1	-38.7	152.2	-13.7	-137.9	-40.3	133.2	-31.0	-138.6	-42.2	126.5	-50.6
0.75	1	-135.9	-38.0	150.1	-16.0	-137.3	-39.6	134.1	-30.9	-137.8	-41.7	127.8	-48.3
0.75	0.75	-136.1	-38.1	147.1	-19.1	-137.6	-39.7	131.2	-34.1	-138.1	-41.8	125.1	-51.4
0.75	0.5	-136.5	-38.2	143.7	-22.8	-138.0	-39.8	127.6	-38.1	-138.8	-41.8	122.3	-54.9
0.75	0.25	-136.9	-38.4	141.7	-25.0	-138.5	-40.0	125.1	-40.9	-139.3	-41.8	119.9	-58.0
0.5	1	-136.2	-36.9	152.5	-14.7	-136.9	-38.4	140.6	-25.8	-137.4	-40.4	136.5	-38.7
0.5	0.75	-137.0	-37.1	145.0	-22.8	-138.0	-38.6	133.7	-34.0	-138.3	-40.6	129.2	-47.0
0.5	0.5	-138.1	-37.5	136.8	-32.0	-139.3	-39.0	124.9	-44.0	-139.7	-40.8	120.8	-57.2
0.5	0.25	**-138.9**	**-38.2**	**132.1**	**-37.2**	**-140.3**	**-39.6**	**118.4**	**-51.0**	**-140.8**	**-41.3**	**114.0**	**-65.4**
Scenario B													
1	N/A	-138.1	-44.2	134.0	-38.3	-140.4	-46.0	114.5	-57.1	-141.8	-47.7	108.7	-77.0
0.75	1	-137.6	-42.9	135.3	-36.8	-139.4	-44.7	119.2	-52.6	-140.6	-46.5	114.1	-69.6
0.75	0.75	-138.0	-42.9	132.4	-39.9	-139.8	-44.7	116.3	-55.8	-140.9	-46.4	111.4	-72.7
0.75	0.5	-138.5	-42.7	128.9	-43.5	-140.4	-44.5	112.7	-59.6	-141.6	-46.2	108.2	-76.4
0.75	0.25	-139.2	-42.7	126.5	-46.1	-141.1	-44.3	109.7	-62.9	**-142.3**	**-45.9**	**105.0**	**-80.0**
0.5	1	-137.3	-40.9	142.8	-29.3	-138.2	-42.4	131.2	-40.5	-139.0	-44.2	127.7	-53.0
0.5	0.75	-138.5	-40.8	136.3	-36.7	-139.5	-42.2	124.8	-48.1	-140.3	-43.9	121.3	-60.5
0.5	0.5	-139.8	-40.6	128.3	-45.2	-141.1	-42.0	116.4	-57.4	-142.0	-43.5	112.9	-70.1
0.5	0.25	**-141.3**	**-41.1**	**122.6**	**-51.7**	**-142.6**	**-42.3**	**108.7**	**-65.8**	**-143.5**	**-43.6**	**104.8**	**-79.8**
Scenario C													
1	N/A	-151.8	-46.7	122.2	-66.4	**-153.5**	**-47.7**	**103.4**	**-84.3**	**-154.4**	**-48.6**	**97.4**	**-102.3**
0.75	1	-150.9	-45.8	125.8	-62.7	-152.0	-46.7	110.0	-77.4	-152.8	-47.8	104.9	-92.8
0.75	0.75	-151.3	-45.7	123.2	-65.5	-152.4	-46.7	107.5	-80.3	-153.2	-47.8	102.5	-95.6
0.75	0.5	**-151.4**	**-45.3**	**120.1**	**-68.1**	-152.6	-46.3	104.2	-83.3	-153.5	-47.4	99.3	-98.7
0.75	0.25	**-151.4**	**-44.8**	**117.6**	**-69.5**	**-152.6**	**-45.7**	**101.1**	**-85.3**	**-153.3**	**-46.7**	**96.0**	**-101.1**
0.5	1	-149.0	-44.1	137.1	-50.3	-149.3	-44.9	125.5	-60.6	-149.7	-46.0	121.3	-72.2
0.5	0.75	-149.7	-43.7	131.4	-56.1	-150.2	-44.6	119.8	-66.7	-150.5	-45.6	115.8	-78.2
0.5	0.5	-149.6	-42.8	124.0	-61.9	-150.3	-43.7	111.8	-73.4	-150.8	-44.7	107.8	-85.4
0.5	0.25	-149.6	-42.4	117.5	-66.6	-150.4	-43.2	103.4	-80.2	-150.8	-44.2	99.0	-93.6

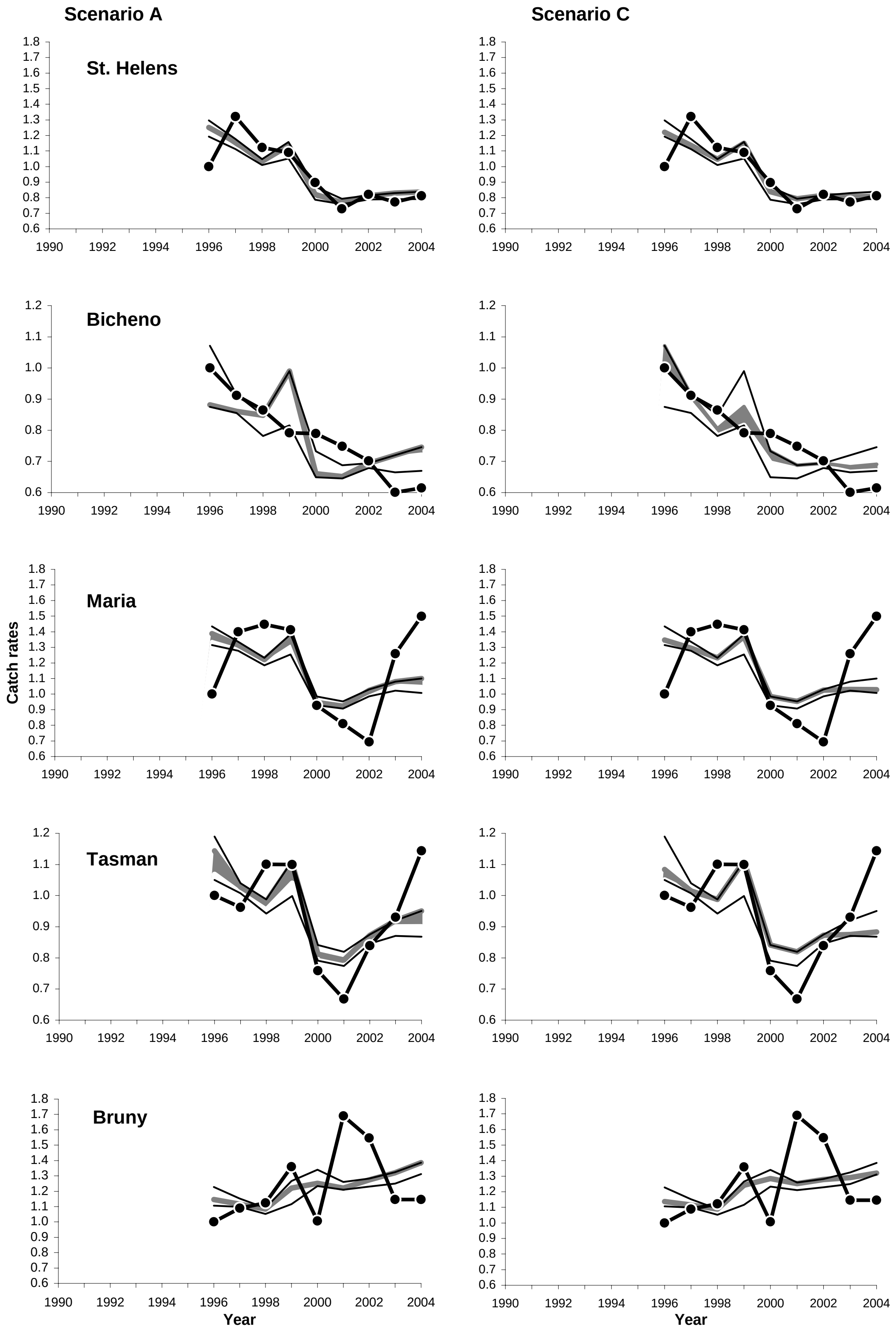

Fig. 5.10: Catch rates for regional biomass scenarios A and C and recruitment variability $\sigma = 0.6$ in the 5 regions between 1990-2004: Observed standardised catch rates relative to the levels in 1995 (black circles and heavy black lines), range of model predictions (grey) of all nine onshore biomass and movement rate scenarios for regional biomass scenarios A and C, and minimum and maximum estimates of all regional biomass, onshore biomass and movement rate scenarios (light black lines). Note varying catch rate scales.

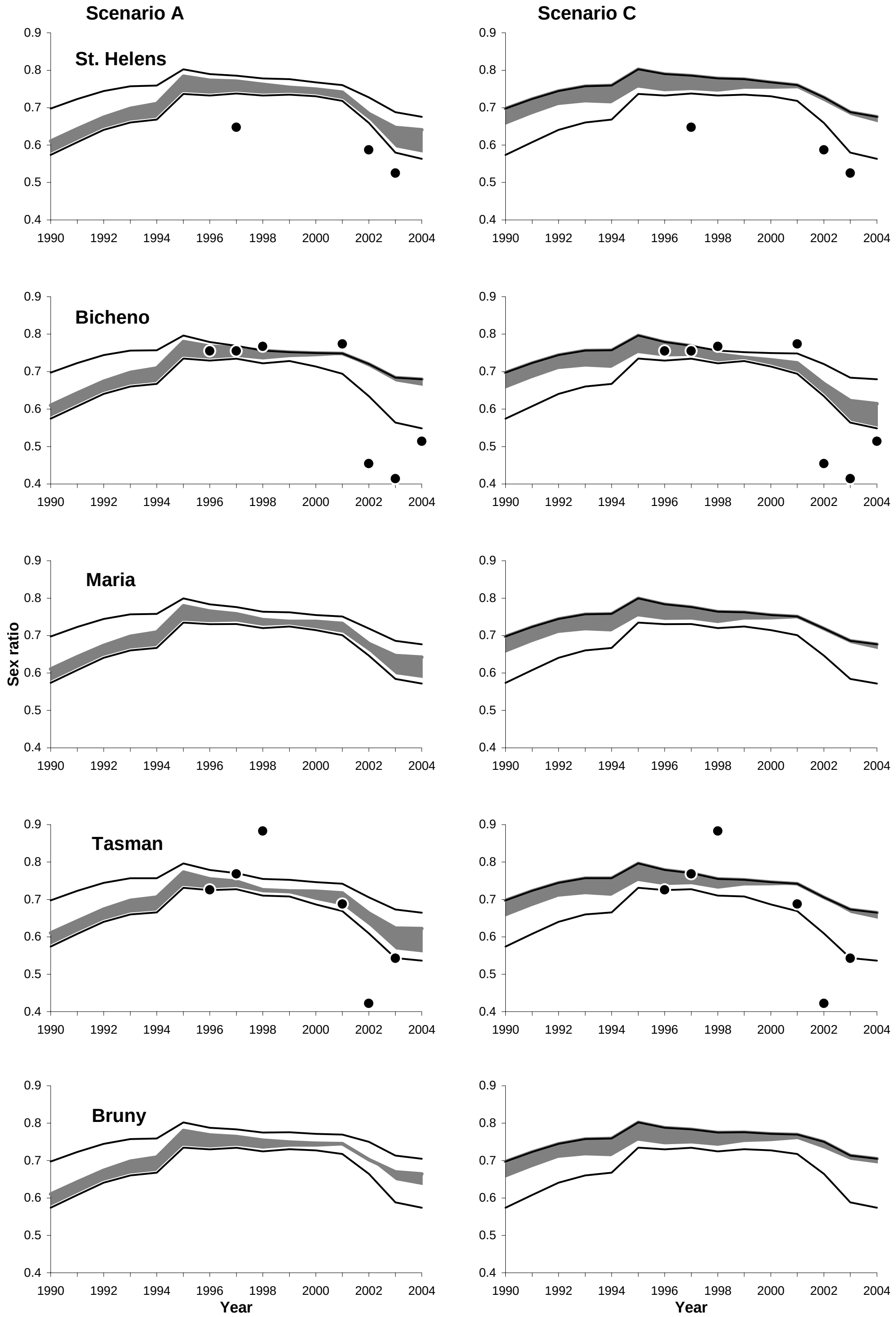

Fig. 5.11: Sex ratios as proportion of females in commercial catch for regional biomass scenarios A and C and recruitment variability $\sigma = 0.6$ in the 5 regions between 1990- 2004: Observed sex ratios (black circles), range of model predictions (grey) of all onshore biomass and movement rate scenarios for regional biomass scenarios A and C, and minimum and maximum estimates of all regional biomass, onshore biomass and movement rate scenarios (light black lines). There were no sex ratio observations from the Maria and Bruny regions.

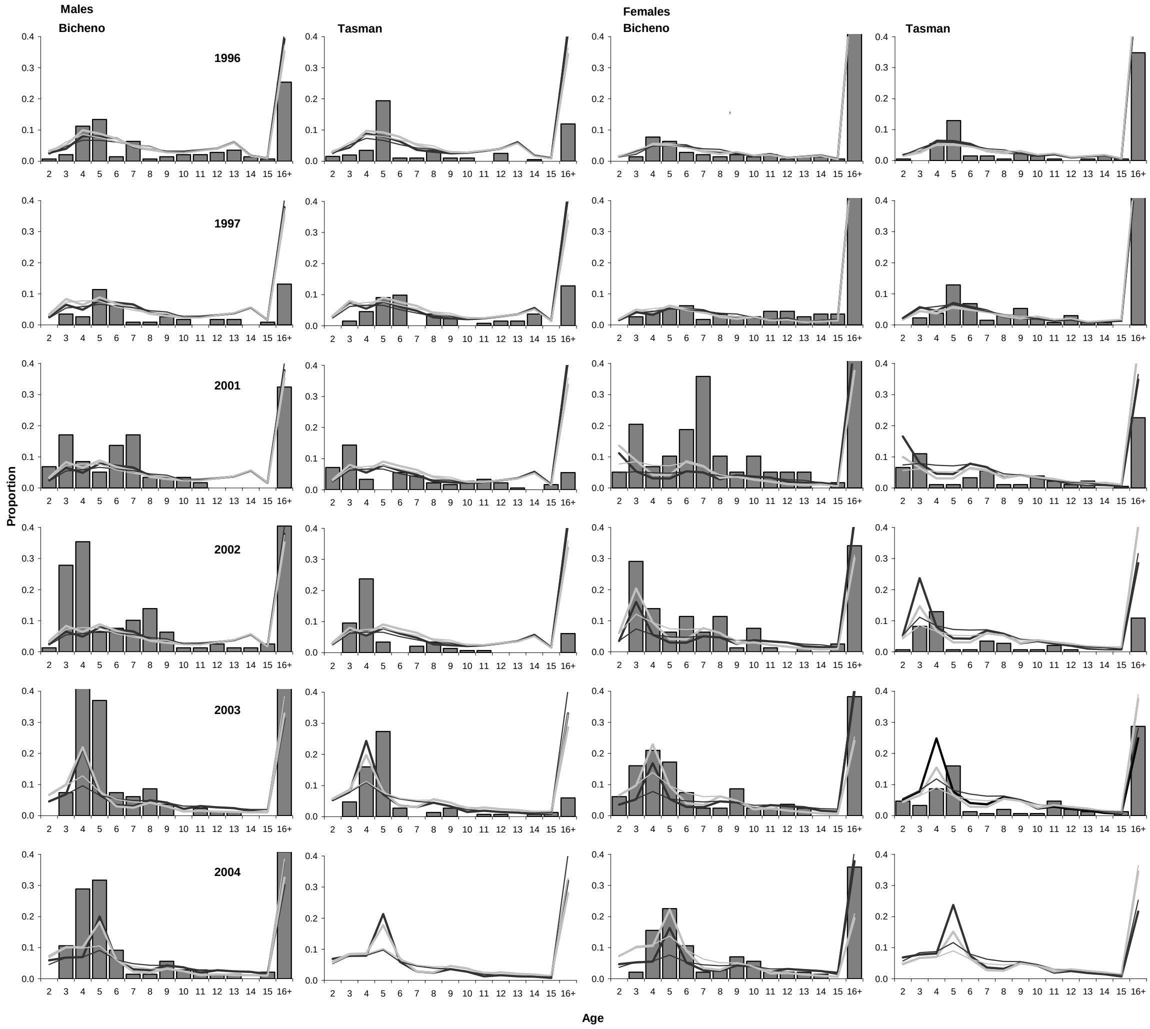

Fig. 5.12: Age composition in relative proportions of male and female banded morwong in biological samples from 1996, 1997, and 2001-2004 in the Bicheno and Tasman regions of the 5-region model with all biomass onshore and therefore available to the fishery: Observed data (grey columns) and model fits for regional biomass scenarios A with recruitment variability $\sigma = 0.1$ (light black line) and $\sigma = 0.6$ (heavy black line), and regional biomass scenarios C with recruitment variability $\sigma = 0.1$ (light grey line) and $\sigma = 0.6$ (heavy grey line). Proportions limited to 0.4. No samples were taken in the Tasman region in 2004.

The total mature biomass ratio, $MatB_{04/90}$ and H_{04} differed widely depending on the regional biomass scenario (Table 5.7, Figs. 5.13 and 5.14). Only results from recruitment variability $\sigma = 0.6$ are presented because these models always provided a better overall model fit to observed data. In fact, regional biomass distribution contributed far more to the uncertainty surrounding model outcomes than the different onshore biomass and movement rate scenarios (Figs. 5.13 and 5.14).

The lack of real data on the regional biomass distribution was extremely influential on the predicted model outcomes; the uncertainty surrounding all model estimates was high. Within the three regional biomass distribution scenarios the allocation to Bicheno varied the most, from 65% in scenario A to 30% in scenario C, which reflected the fact that a significant proportion of the total yield from the fishery came from this region. Because of this variation between scenarios the range of estimates of $MatB_{04/90}$ was most pronounced in the Bicheno region, ranging from 0.33 to 0.88 (Table 5.7, Fig. 5.13). Differences in the range of estimates of $MatB_{04/90}$ were less extreme in the other regions, but still ranged considerably from 0.42 to 0.72 in the St. Helens, Maria and Tasman regions, and from 0.49 to 0.81 in the Bruny region where only a small proportion of the biomass was attributed. Harvest rate varied also substantially with H_{04} onshore from 0.02 to 0.22 in the Bicheno region, from 0.07 to 0.20 in the St. Helens, Maria and Tasman regions, and from 0.02 to 0.07 in the Bruny region (Table 5.7, Fig. 5.13). Total H_{04} of the onshore and offshore population was obviously lower due to the added biomass by the offshore population.

Table 5.7: Ranges of model estimates for all onshore biomass and movement rate scenarios in the 5-regions with recruitment variability = 0.6.

Model outcomes for each region, the sum of all regions (Total), and for the 1-region model as comparison (1 Region). Results of mature biomass (in tonnes) in 1990 (MatB90) and 2004 (MatB04), biomass levels in 2004 relative to virgin levels in 1990 (MatB90/04), harvest rate in 2004 for the onshore population (H04 onshore) and the total onshore and offshore populations (H04 total), and geometric mean of annual recruits (in numbers) between 1990 and 2004. Regional biomass distribution in scenario A: 0.1 - 0.65 - 0.1 - 0.1 - 0.05; scenario B: 0.15 - 0.5 - 0.15 - 0.15 - 0.05; scenario C: 0.2 - 0.3 - 0.2 - 0.2 - 0.1.

				Region				
		St. Helens	**Bicheno**	**Maria**	**Tasman**	**Bruny**	**Total**	**1 Region**
$MatB_{90}$ (t)	A	121 to 223	787 to 1448	121 to 223	121 to 223	61 to 111	1210 to 2227	685 to 997
	B	112 to 194	372 to 646	111 to 194	111 to 194	37 to 65	743 to 1291	
	C	174 to 289	261 to 433	174 to 289	174 to 289	87 to 145	869 to 1445	
$MatB_{04}$ (t)	A	68 to 145	666 to 1195	74 to 151	62 to 141	45 to 85	913 to 1716	348 to 627
	B	49 to 112	212 to 427	55 to 118	46 to 110	18 to 39	379 to 805	
	C	100 to 192	86 to 222	106 to 198	98 to 190	64 to 110	454 to 913	
$MatB_{04/90}$	A	0.56 to 0.69	0.83 to 0.88	0.61 to 0.72	0.51 to 0.66	0.74 to 0.81	0.75 to 0.82	0.51 to 0.63
	B	0.44 to 0.59	0.57 to 0.68	0.49 to 0.63	0.42 to 0.58	0.49 to 0.63	0.51 to 0.64	
	C	0.57 to 0.67	0.33 to 0.51	0.61 to 0.70	0.56 to 0.66	0.74 to 0.79	0.52 to 0.63	
H_{04} onshore	A	0.10 to 0.13	0.02 to 0.03	0.10 to 0.14	0.09 to 0.14	0.02 to 0.03	0.04 to 0.06	0.10 to 0.16
	B	0.13 to 0.19	0.06 to 0.09	0.13 to 0.20	0.12 to 0.19	0.05 to 0.07	0.09 to 0.14	
	C	0.07 to 0.12	0.12 to 0.22	0.08 to 0.13	0.07 to 0.11	0.01 to 0.02	0.08 to 0.14	
H_{04} total	A	0.05 to 0.10	0.01 to 0.02	0.06 to 0.11	0.05 to 0.11	0.01 to 0.02	0.02 to 0.04	0.06 to 0.10
	B	0.07 to 0.14	0.03 to 0.06	0.07 to 0.15	0.07 to 0.14	0.02 to 0.05	0.05 to 0.10	
	C	0.04 to 0.07	0.06 to 0.15	0.04 to 0.08	0.04 to 0.07	0.01 to 0.01	0.04 to 0.08	
Recruits	A	13720 to 20489	89183 to 133181	13720 to 20489	13720 to 20489	6860 to 10245	137205 to 204894	73092 to 91098
	B	11825 to 16565	39416 to 55218	11825 to 16565	11825 to 16565	3942 to 5522	78832 to 110436	
	C	16497 to 22820	24746 to 34229	16497 to 22820	16497 to 22820	8249 to 11410	82485 to 114098	

The region-specific impact of the regional biomass scenarios becomes obvious when the trajectories of mature biomass and harvest rates are considered (Fig. 5.13 and 5.14). Due to the relative nature of the biomass distribution, estimates of $\text{MatB}_{04/90}$ and H_{04} in one or more regions were relatively low for any given scenario. Scenario A was most optimistic, while in the remaining scenarios low mature biomass levels and high harvest rate levels were estimated in the St. Helens, Maria and Tasman regions in scenarios B, or in the Bicheno region in scenario C.

The reason for the lowest impact of fishing on the stocks in scenario A was probably due to the highest overall recruitment and therefore productivity (Table 5.7 and Fig. 5.15). Recruitment variability was also substantially higher than in the other scenarios.

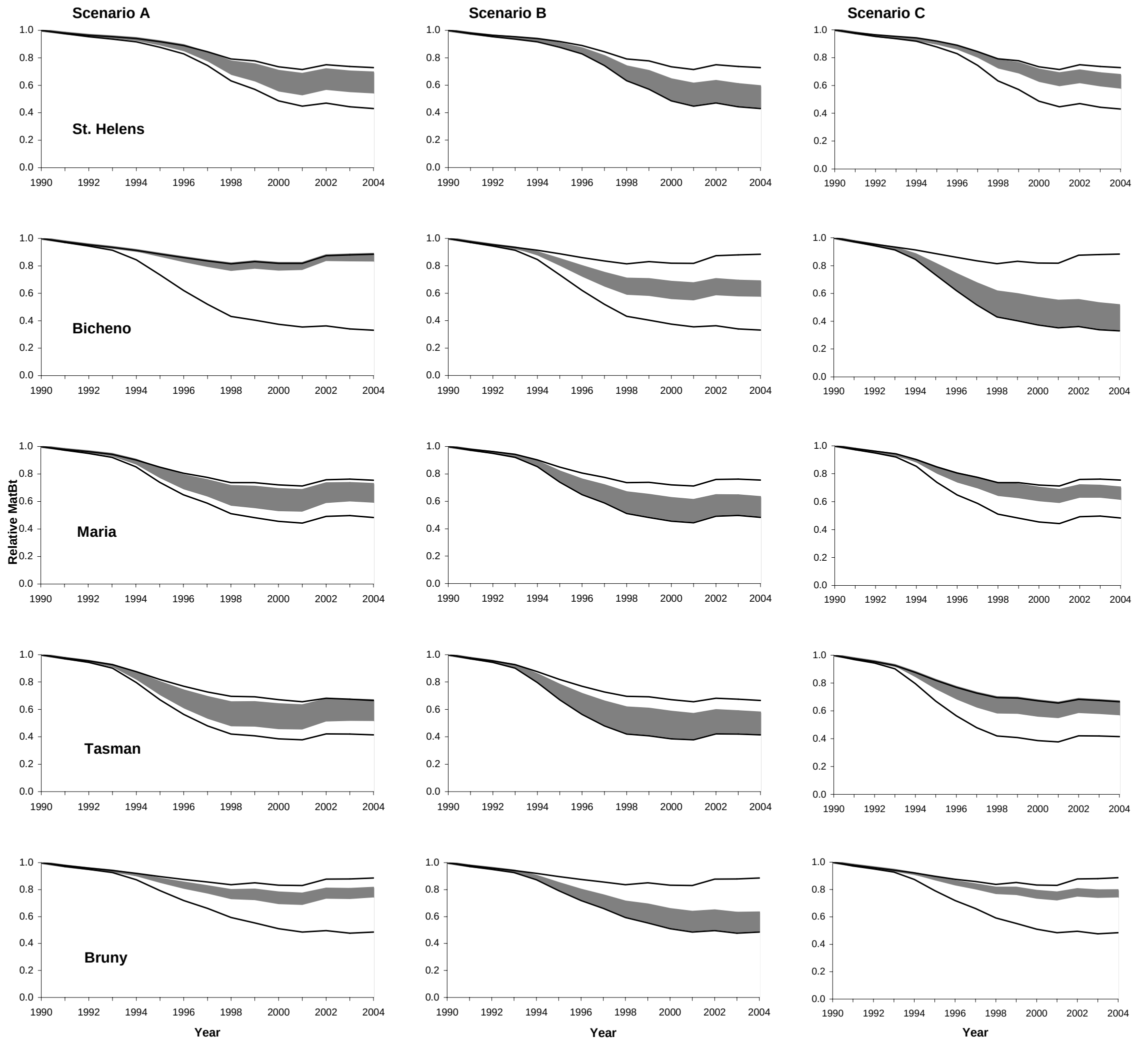

Fig. 5.13: Range of mature biomass estimates (grey) of all onshore biomass and movement rate scenarios relative to the levels in 1990 for regional biomass scenarios A, B, and C and recruitment variability σ = 0.6 between 1990- 2004, and minimum and maximum estimates of all regional biomass, onshore biomass and movement rate scenarios within a region (light black lines).

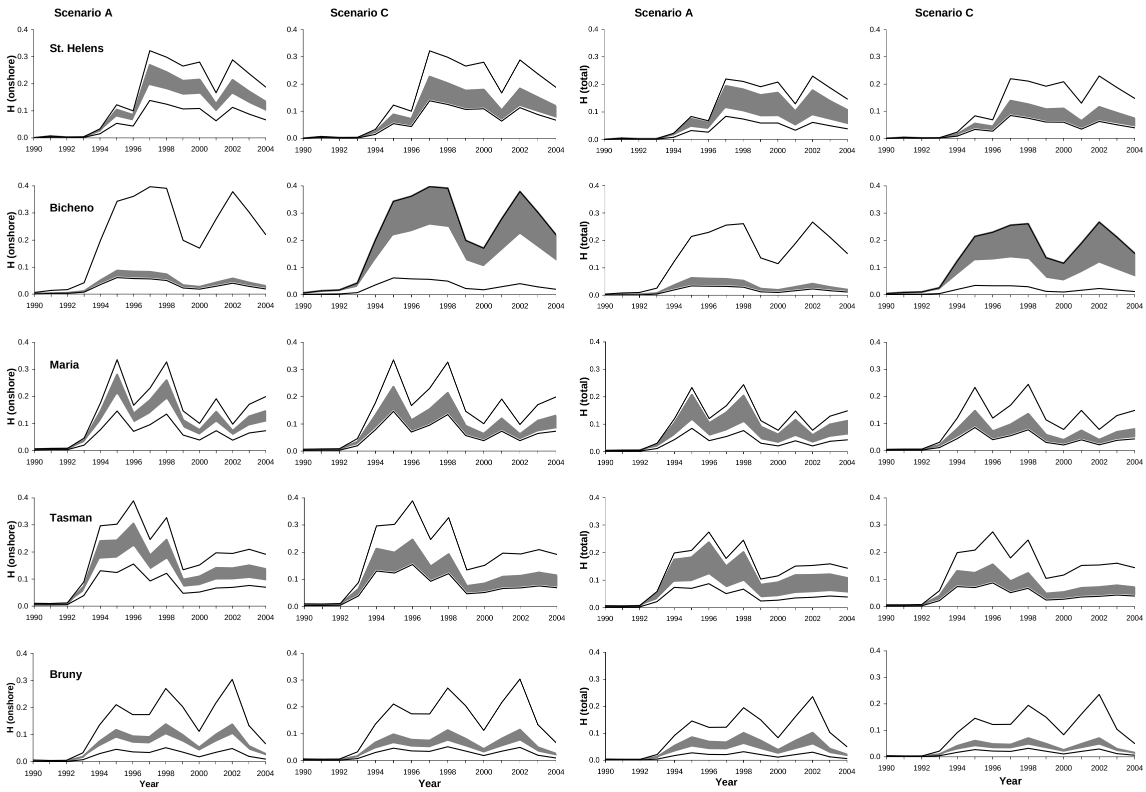

Fig. 5.14: Range of harvest rate estimates (grey) in the onshore population (left two columns) and the total onshore and offshore stock (right most two columns) for all onshore biomass and movement rate scenarios for regional biomass scenarios A and C and recruitment variability $\sigma = 0.6$ in the 5 regions between 1990- 2004. The minimum and maximum estimates of all regional biomass, onshore biomass and movement rate scenarios within a region are illustrated as fine black lines.

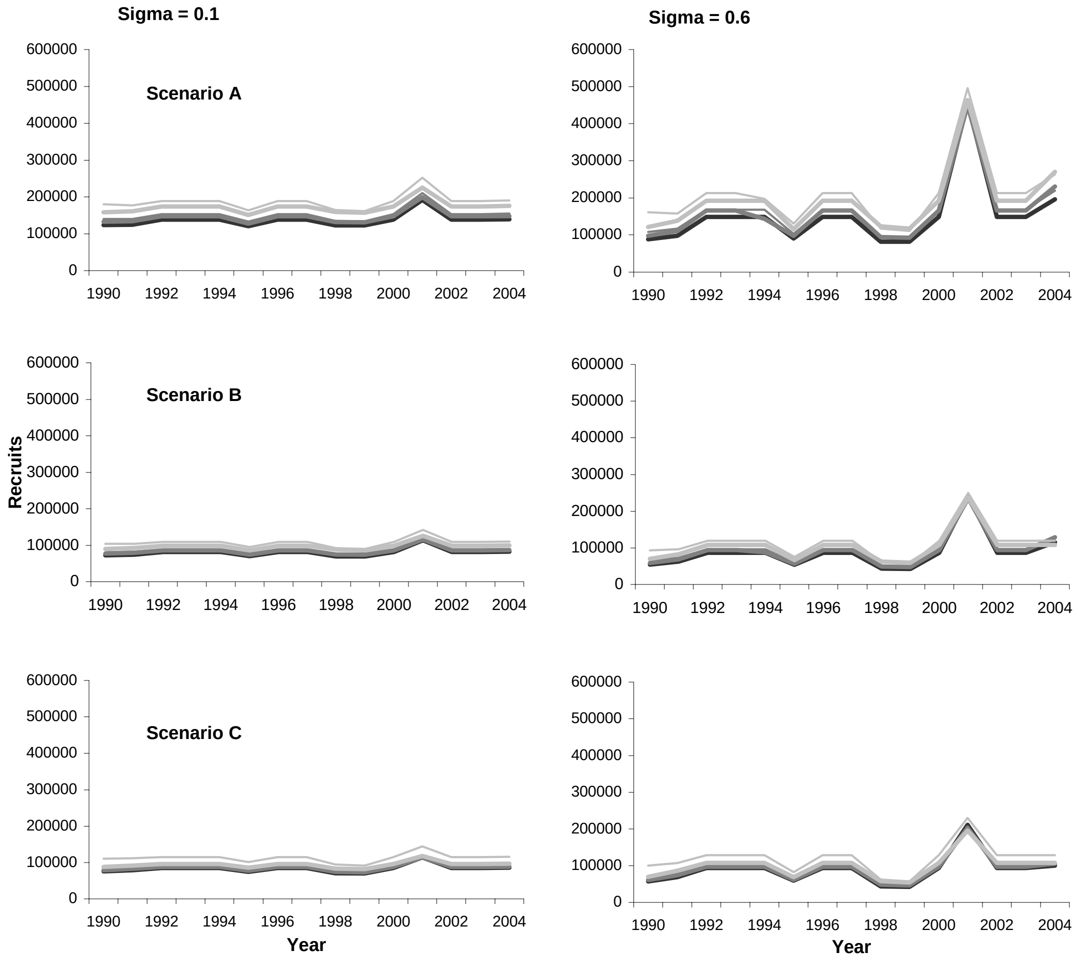

Fig.5.15: Range of annual total recruitment estimates (in numbers) for recruitment variability $\sigma = 0.1$ and 0.6 with regional biomass distribution A, B and C for the 5-region model between 1990- 2004.

5.4 Discussion

An operating model needs to simulate the stock dynamics and other processes in a realistic manner to be a reliable representation of a fish stock. This applies to both the basic biology of the species and the fishery dynamics, where fishery data are used to characterise the stocks, *e.g.* catch rates as indices for biomass. When this is achieved the operating model can be used to explore the sensitivity of model outcomes to different sources of uncertainty inherent in a stock assessment and management strategy. In this study, however, analyses have been restricted to the consideration of model uncertainty only, namely regional and onshore-offshore biomass distribution, movement rates, and recruitment variability. The addition of further variability in the form of observation and/or process error was deemed uninformative on top of the high level of uncertainty that applied to the current model.

Ostensibly, a relative wealth of data was available to characterise key aspects of the biology of banded morwong, especially when considering the small scale and low value of the fishery in Tasmania. Biological monitoring allowed a description of the life-history characteristics such as growth, maturity and movement, and indeed revealed unexpected changes in some of these characteristics over time (Chapter 4). This type of information is important for the characterisation of any fish species, as well as the identification of potential stock risks and performance measures. Nevertheless when attempting to fit the operating model to the biological data, particularly sex ratio, and to a lesser extent the age composition data, the model proved unreliable. As noted in Chapter 4, comparatively small sample sizes coupled with spatial structuring within the population may have influenced the more general applicability of the results. That is to say, biological samples were typically derived from a number of individual reefs within a region and since banded morwong show little movement between reefs or beyond 5 km distance, reef-by-reef diversity in factors such as age composition and sex ratios will influence the characteristics of the combined regional sample.

It is also doubtful whether catch rates, used as the main relative index of abundance (a proxy for available biomass), were spatially representative, although in the different way to the biological monitoring. Catch and effort are reported at a spatial resolution that is much lower than that at which the fishery affects the stocks. Because serial depletion would be masked by this mis-match of scale, catch rates may have appeared more stable than they would have been had they been able to be analysed at a finer spatial scale. Furthermore, the accuracy and reliability of the catch rate data is in question in terms of (i) reporting accuracy; (ii) use of daily rather than gear usage based catch rates; and (iii) the impact of external factors such as seals and markets on catches. For the above reasons the application of catch rates as an index of abundance in this case is likely to introduce uncertainty into the model.

The 1 and 5-region models were developed in the full knowledge that they did not adequately represent the spatial structure of the stocks. But within the limitations of available information and recognising the degree of over-parameterization they provide some useful insights into banded morwong populations. While it was possible to test various scenarios, the overall poor fit of the models to available data limited its utility in the identification of optimal scenarios. The 1-region model was, by definition, unable to detect differences and potentially critical levels of mature biomass or harvest rates at regional levels. In the 5-region model, very different trajectories of mature biomass and harvest rates were predicted in the different scenarios tested and any of the regions could have been subject of heavy fishing pressure and resulting low mature biomass. The level of uncertainty also differed between the 1 and 5-region model. Uncertainty in the 1-region model, deriving mainly from the model structure relating to onshore biomass and movement rate levels, was modest for estimates of mature biomass, but quite large for harvest rates. The uncertainty around the predicted mature biomass and harvest rates in the 5-region model depended strongly on what assumption was used concerning the regional distribution of the stock biomass. For example, the onshore biomass and movement rate scenarios had little influence on the mature biomass trajectories in scenario A, but added substantial uncertainty in scenario C (20% in 2004 - see Figs 5.13 and 5.14). Nevertheless, the regional biomass distribution was, in all cases, the main contributor to the overall uncertainty surrounding model predictions.

The results from this operating model can be interpreted in the light of a formal stock assessment for banded morwong. Unfortunately, model uncertainty is not necessarily smaller compared to the simple analyses of commercial catch and effort data and biological monitoring (Chapter 4). In fact, the model rather reflects the uncertainty of the underlying data and uncertainty over the dynamics and spatial structuring of the stock. Fitting the model to real data did not greatly reduce uncertainty derived from the variety of assumptions about the possible regional and depth biomass distributions. Nevertheless, some general conclusions about the stock status can be made. Most scenarios tested in the 5-region model indicated high harvest rates or fishing mortalities over an extended period of the fishery. Only in recent years were these parameters predicted to have been reduced to lower and more sustainable levels within the range of the mature biomass reference points of $F_{40\%}$ or $F_{30\%}$ (Chapter 4; Mace and Sissenwine 1993; Mace 1994). All scenarios tested by the model also predicted relatively low estimates of mature biomass in at least some regions. In absolute terms, the mature biomass levels in 2004 did not particularly decrease, *i.e.* they were higher than 30% or 40% of the virgin mature biomass. However, these estimates are likely to be best-case scenarios because of the potential for masking of serial depletion due the low spatial resolution of catch rates. But if this and the high uncertainty of the results are taken into account, there is a considerable risk that the mature biomass overall, in many regions or on many individual reefs is in a critical state.

In addition, these model results are based on substantial changes in some life-history characteristics of banded morwong. With faster growing young individuals and earlier maturing females (and maybe also males) the fish stocks are now much more productive than at the start of the fishery and even during the mid 1990s (Chapter 4). The effect of this increased productivity was magnified by the model predictions of larger recruitment pulses in recent years. Predicted recent stability in the fish biomass and the fishery may therefore be founded mainly on an increase in young fish, rather than sustainable use of old fish. Both, the changes in life-history characteristics and increased recruitment variability coincided with predicted decreasing population size. It is therefore more likely that they are mainly a density-dependent response or selection of these characteristics by fishing, indicating heavy exploitation rather than simply being caused by environmental changes such as increasing water temperatures (refer Chapter 4). The substantial changes in the life-history characteristics were unexpected for a long-lived species such as banded morwong and demonstrate also the importance of repeated biological monitoring over time to get detailed knowledge on the biology of the species when developing an appropriate operating or stock assessment model.

In the following chapters this operating model is used to investigate if simple measures derived from fishery-independent or dependent information can be used to estimate stock status or manage the stock to given objectives. In particular, simulations explore the potential of biological properties as stock performance measures, and compare the performance of different scenarios where decision rules were based on catch rate indicators.

CHAPTER 6: THE VALUE OF BIOLOGICAL PROPERTIES AS PERFORMANCE MEASURES

6.1 Introduction

In the process of creating the model description of the Tasmanian banded morwong population and its fishery, the fine-scale spatial structuring exhibited by the species challenged many of the usual assumptions in stock assessment modelling. The major problem identified was our ignorance of the spatial distribution of the stock biomass and the related spatial dynamics (Chapter 5).

The use of this assessment model as an operating model in Monte Carlo simulations or management strategy evaluations (MSEs) generally incorporates a detailed consideration of the effects of numerous sources of variation and uncertainty. These include observation error, process error, management decision implementation error, estimation error, institutional uncertainty, and model uncertainty (see Chapter 2; and Caddy and Mahon 1995). In the banded morwong case, the lack of available data on the spatial dynamics of the stock meant that all multi-region models were highly over-parameterized. Thus, the major source of uncertainty for the banded morwong simulations is model uncertainty. In a typical simulation analysis, available data is used to find the optimal fitting model and then different degrees of variation are applied to aspects of the operating model's structure to test how robust the various model outcomes are to such uncertainties. Because of the massive model uncertainty, it was decided instead that, at least initially, the banded morwong simulations would consist of setting up a balanced array of plausible options for the model structure and using that array to determine whether various performance measures provided useful information about the stock's status.

Fisheries management of data-poor fisheries is typically based on commercial fisheries data such as trends or thresholds in catch, effort and catch rates (see Chapter 2). However, there can be severe problems in relation to commercial data. Catch rate information deriving from compulsory logbook returns is often inaccurate due to deliberate or accidental misreporting of catch or effort, the spatial representation of the fishing activity may be inappropriate to the processes driving the dynamics, and the same management approach is often applied generically to many fish species independent of their biological characteristics.

Biological properties of fished populations, collected from fishery-dependent or independent monitoring could address some of these problems. A controlled survey design means that the data sampling regime is known and structured. And biological properties would certainly better consider the particular set of life-history characteristics of the species such as longevity, growth and maturity. The spatial representation could impose a problem, since limited budgets typically restrict the number of locations sampled. However, despite the poor spatial representation, biological measures have the potential to either replace or at least supplement a stock analysis based on commercial catch and effort data (Chapter 4). Nonetheless, the interpretation can remain ambiguous if the dynamics of biological properties are poorly understood.

In this chapter, we investigate the quantitative characteristics of some biological properties of banded morwong populations and explore their potential use as measures of stock performance. The capacity of biological performance measures derived from the age and size composition is explored by use of Monte Carlo simulations.

6.2 Methods

As outlined in Chapter 5, an operating model was developed to provide the means of comparing the efficiency and effectiveness of alternative fishery performance indicators and different management strategies (data collection, assessment method, plus decision rules; Chapter 7). A detailed description of the model structure and the projection is provided in Appendices 3 and 4. The model of the population dynamics was designed not only to fit available data but to use the parameter estimates obtained to generate forward projections. The operating model included a component describing the fishery dynamics (the harvest) and a management component regulating the fishery.

To assess the value of biological properties as performance measures of the stock status, the model was used to project the stock and fishery forward for twenty years under constant harvest scenarios. The underlying stock dynamics were known from the operating model (the ‘real population’) and were compared with biological properties either from a perfect fishery-independent representation (estimated by using gear selectivity only) or a simulated fishery-independent sample with different sample sizes (estimated by using gear selectivity and random sampling).

6.2.1 Biological components of the operating model

Only the 5-region operating model was used in the simulations. A major source of model uncertainty was the unknown distribution of banded morwong stock biomass among the modelled five East coast regions. The three scenarios A, B and C corresponding to the spatial distribution of biomass as described in Chapter 5 were used for comparison (Table 6.1), whereby scenario A had performed worst and scenario C best. For each regional scenario an intermediate scenario for the combination of onshore biomass (75% of the biomass onshore) and movement rates (0.5) between the onshore and offshore population was selected. This scenario performed only slightly poorer than the best models, and its specifications were seen as a suitable compromise between those of the best models which had either 100% biomass onshore or 75% biomass onshore and movement rates of 0.25 (Chapter 5). Each regional scenario was tested at two levels of recruitment variability $\sigma = 0.1$ and 0.6. The computations were therefore limited to a total of only six different scenarios (Table 6.1).

The model parameters and recruitment residuals for each of the six scenarios were determined by fitting the model to the historical biological and commercial data available from the fishery. They were then used to project the populations in each region from the pre-exploitation equilibrium in 1990 forward to the start of 2005. By applying random variation in recruitment and accounting for the reported catches from 1990-2004, the age structure for the start of 2005 was obtained. It was from 2005 that the given constant catch strategy was first applied.

Table 6.1: Characteristics of the six model scenarios evaluated.

Regional biomass relates to proportion of total biomass located in the St. Helens, Bicheno, Maria, Tasman and Bruny regions, respectively.

Regional biomass	Recruitment variability σ	Biomass onshore	Movement rate
A: 0.10 - 0.65 - 0.10 - 0.10 - 0.05	0.1	0.75	0.5
	0.6	0.75	0.5
B: 0.15 - 0.50 - 0.15 - 0.15 - 0.05	0.1	0.75	0.5
	0.6	0.75	0.5
C: 0.20 - 0.30 - 0.20 - 0.20 - 0.10	0.1	0.75	0.5
	0.6	0.75	0.5

The recruitment levels used in each case were based on the geometric mean and standard deviation of the predicted recruitment levels derived from fitting the model to the historical period (Appendix 4). In addition, because the recruitment variability appeared to increase with reducing stock biomass, the standard deviation was modified to be density-dependent and influenced by the population numbers in the previous year. Environmental influences during the larval phase may be important in determining the inter-annual variability of a recruitment pulse at settlement. However, the extent of the survival of any such pulse after settlement and subsequent recruitment to the fishery as 2 year olds in year y was assumed to be influenced by the existing population. Population density was approximated by the population numbers in year y-1. The range of recruitment variability in the projections was limited to the estimated range from the historical period. This limitation may have resulted in a cap on very large recruitment events at low population size, effectively reducing the productivity of the stocks and resulting in more conservative rebuilding scenarios. Recruitment was also assumed to occur under any circumstances, *i.e.* even when there was virtually no standing biomass left in the model populations. While this can be seen as an overly-optimistic assumption, recruitment could well be supplemented from populations outside the fishery areas, since the long pelagic larval phase of around 6 months most likely decouples recruitment sources and sinks (Chapter 4). Overall, there is great uncertainty about the recruitment dynamics and hence a wide range of inherent variability was used (σ = 0.1 and σ = 0.6).

All biological characteristics, as estimated for 2004, including natural mortality, movement, growth and maturity, were used to project the fishery and stock forward from 2005 to 2024. Although the changes in growth and maturity observed in the stock from 1996-2004 were assumed to have been initiated through density-dependent mechanisms (Chapter 4), these mechanisms and any potential delays between changes in biomass and the biological response were unknown and therefore not simulated.

Because of the changes to growth and maturity the projections were undertaken with the fish stock in its most productive growth and maturity state. Thus, any stock recovery within the model population may be over-optimistic and needs to be treated with caution, since increasing stock size would be expected to reverse the effects on growth and maturity.

6.2.2 Fleet dynamics in the operating model

Little is known about the fleet dynamics or whether there has been any previous redirection of effort. Such re-direction of effort is possible since some fishers are sufficiently mobile that they could fish different regions to follow better catch rates. In addition, new fishers may lease or buy a license, and there is latent effort of license holders who previously have not fished or have only taken small catches, that could be targeted at regions with promising catch rates. However, in the absence of reliable data the spatial distribution of catch and effort during the projected fishery dynamics were assumed to reflect that of historical times. Changes of total catch or effort translated into proportional increase or decrease of catches or effort in each region. No relative effort was assumed to be shifted from one region to another, even in the case of catch rates dropping in one region relative to others. This assumption was taken because it is likely to mirror the behaviour of some established fishers who are based in specific regions and contribute a substantial share of the overall catch.

To estimate the total fishing-induced mortality or ‘real catch’, the reported or allowable catch was multiplied by a reporting and seal mortality bias. While over-reporting may occur at times, it was assumed that fishers generally under-report the catch, *e.g.* by reporting only landed live catch but not mortalities. Therefore, the allowable catch was multiplied by a randomly selected normally distributed factor $N(1.05, 0.1)$, *i.e.* a normal distribution with mean 1.05 and standard deviation $\sigma = 0.1$. This distribution was mostly greater than 1.0 (indicating under-reporting) but could become less than 1.0 indicating a less common over-reporting event. A second normally distributed multiplication factor $N(1.15, 0.05)$ was used to account for fishing-induced mortality from seal interactions. Thus, the total fishing-induced mortality was generally higher than the reported or allowable catch but could be also sometimes lower.

Assuming no major changes in fishing practices or fish behaviour, catchability estimated from historical data was used in the model projections to estimate catch rates from exploitable biomass estimates.

6.2.3 Stock assessment using biological properties

The selection of suitable biological performance measures depends strongly on the life-history and fishery characteristics of the fished species. For example, sex ratios could be used if males and females have a different catchability or vulnerability towards the fishing gear based on some physical or behavioural characteristics. This is the case in banded morwong due to the combination of sex-specific differences of growth and the keyhole selectivity, but would also apply to blue-throated wrasse where males are more aggressive than females towards line fishing gear and minimum size limits and sex change (female to male) mean that males are more susceptible to fishing than females.

For banded morwong, a suite of biological measures that can be estimated from population sampling was selected and their potential and suitability for use as stock status proxies investigated. These measures included sex ratio (proportion of females in the catch), median age, age ratio (proportion of fish under 16 years old in the catch) and total mortality Z (as estimated from catch curves). In addition, the median size was calculated from the size composition of each sample. To generate a realistic size composition, the length L of each individual in a sample was estimated based on the parameters L_∞, K and t_0 of the 2-phase von Bertalanffy growth function (for both the lower and upper functions in each sex), multiplied by a randomly selected normally distributed factor $N(L, \sigma)$, *i.e.* based on the estimated standard deviations σ of the observed age-length relationships (Chapter 4). In all cases the biological measures were considered for males and females separately.

The biological measures were tested in two stages. Firstly, we tested the theoretical power of the biological measures as indices for mature biomass. For this, the actual values of all measures in year y, as predicted in a perfect fishery-independent sampling, were estimated without measurement error from the age and size composition of the real population (*i.e.* the operating model) adjusted by gear selectivity. These measures were then compared with an index for stock status expressed as the ratio of mature biomass in year y relative to the mature biomass from the near-virgin state of the fish stocks in 1990. Relationships between biological measures and mature biomass were then investigated in graphical plots for their potential value in characterizing the stock status.

Secondly, we tested how well biological measures can be estimated from samples. Thus, the predicted values of these biological properties were compared to estimates sampled from the operating model at different sample sizes. These samples were generated by randomly sampling individuals of the simulated population according to gear selectivity. Results from different sample sizes of 150, 250, 400 and 1000 fish were compared.

To test for relationships over a wide range of biological measures and mature biomass levels, a fishery projection was chosen which resulted in a significant depletion of the stocks, *i.e.* a constant removal of 80 tonnes fish per year over the 20 years of the projection. The analysis was restricted to the Bicheno region because a great range of predicted biomass levels occurred in this region under the different regional biomass scenarios A, B and C. An 80 tonnes coast-wide catch translates to about 29 tonnes from Bicheno region (about 36.3% of total, equivalent to the proportional catch in 2004).

Unfortunately, results from the first 6 years of the model period between 1990-1995 had to be excluded since biological monitoring only started in 1996 and no age data was available to fit any cohorts aged 10 years and older in 1990. Thus, the performance of the biological measures could not be compared at relatively high levels of mature biomass close to virgin biomass.

6.3 Results

6.3.1 Stock assessment using biological indicators

Despite heavy fishing with a constant removal of 80 tonnes of fish per year over the 20 years of the projection (equivalent to 29 tonnes from Bicheno), mature biomass in the Bicheno region was not reduced below 50% of its virgin levels in scenario A. Only in scenarios B and C was the relative mature biomass reduced to less than 10% of its virgin levels.

Actual predicted values of biological properties and corresponding relative mature biomass levels calculated from the simulated population (the operating model) are presented in Figs. 6.1 to 6.6 for the three regional biomass scenarios A, B and C and the two recruitment variability levels $\sigma = 0.1$ and 0.6. In each case these data are based upon 200 simulations, which was sufficient to map most of the variation expressed.

Trends were more distinct for low relative to high recruitment variability. This is not unexpected, because the low recruitment variability scenario is closer to a constant dynamics state and thus is intrinsically a far more stable system.

There were no obvious trends for any of the biological measures in males, even at low recruitment variability. A constant catch scenario had only unpredictable and minor effects on all of the biological measures considered. Because of the small number of fishable age classes in the male population fishing had only a relatively minor impact over the period of the simulation. However, the mature biomass of females was generally strongly reduced over the same period. Thus, biological measures for males and mature biomass of females did not correlate well.

The power to discriminate between different stock sizes using any of the biological measures for females was greatly reduced in biomass scenario A (Figs. 6.1 and 6.2). Only with decreasing levels of relative mature biomass did any trends in these measures became more obvious. In scenarios B and C both the sex ratios and the age ratio in females gave information with regard to the relative stock biomass level. At worst it could exclude possibilities (Figs. 6.3 to 6.6).

Neither median age, median size, nor estimates of *Z*, were robust to model uncertainty in relation to recruitment variability in any of the three regional biomass scenarios (Figs. 6.1 to 6.6). The spread of the relative mature biomass with respect to all the biological measures considered was broad and indistinct in all scenarios. Only for catch curve estimates of Z was there some indication that at low recruitment variation, low and high biomass levels could be separated when mortality estimates were low (Fig. 6.5). Because females exhibited more contrast than males in the results for all three biomass distribution scenarios and for all potential biological performance measures, further analyses concentrated on sex ratios and the biological properties for females.

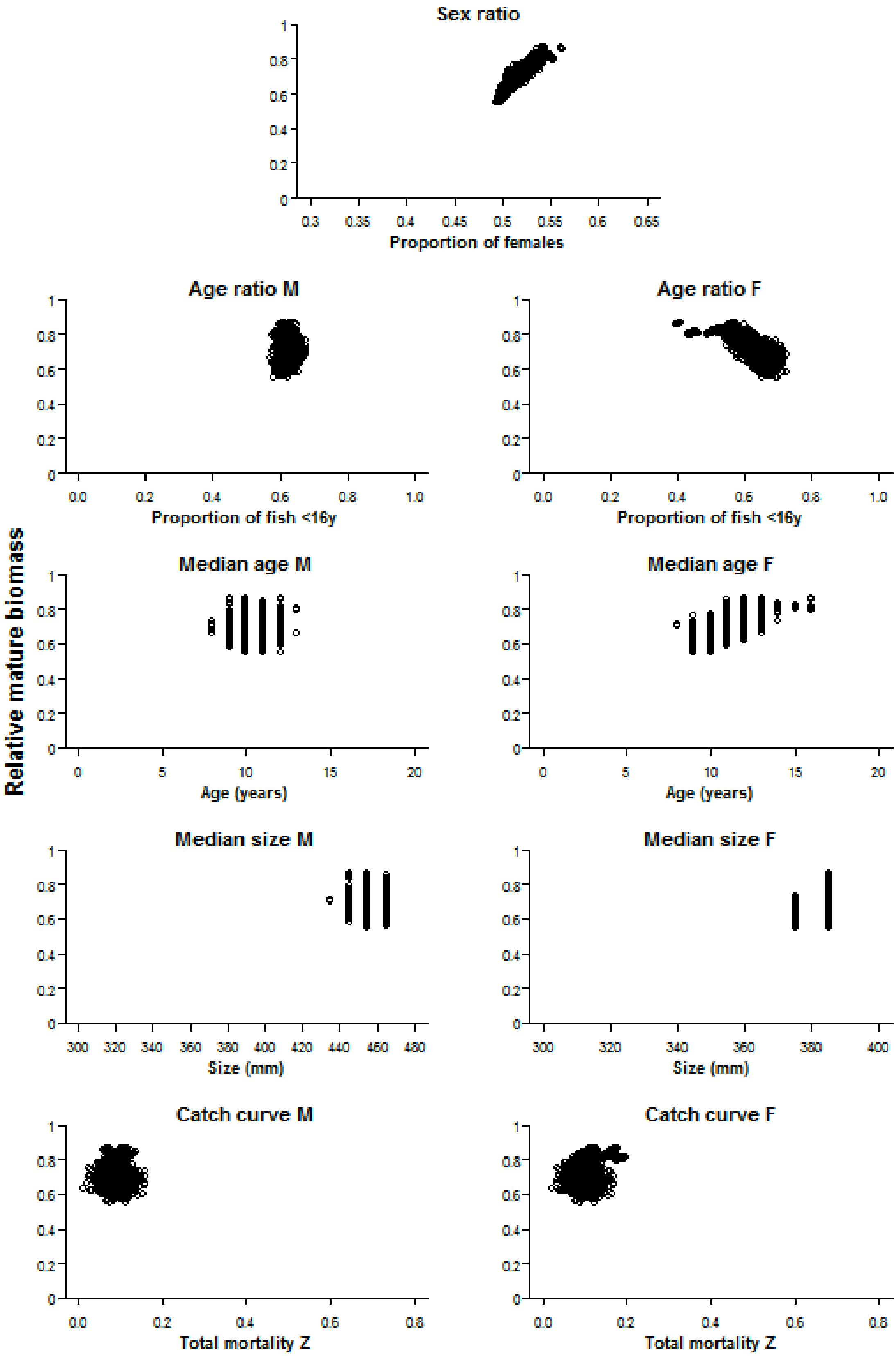

Fig. 6.1: Relative mature biomass versus biological properties of the populations for regional biomass scenario A and recruitment variability $\sigma = 0.1$. Results shown are based on 200 simulations.

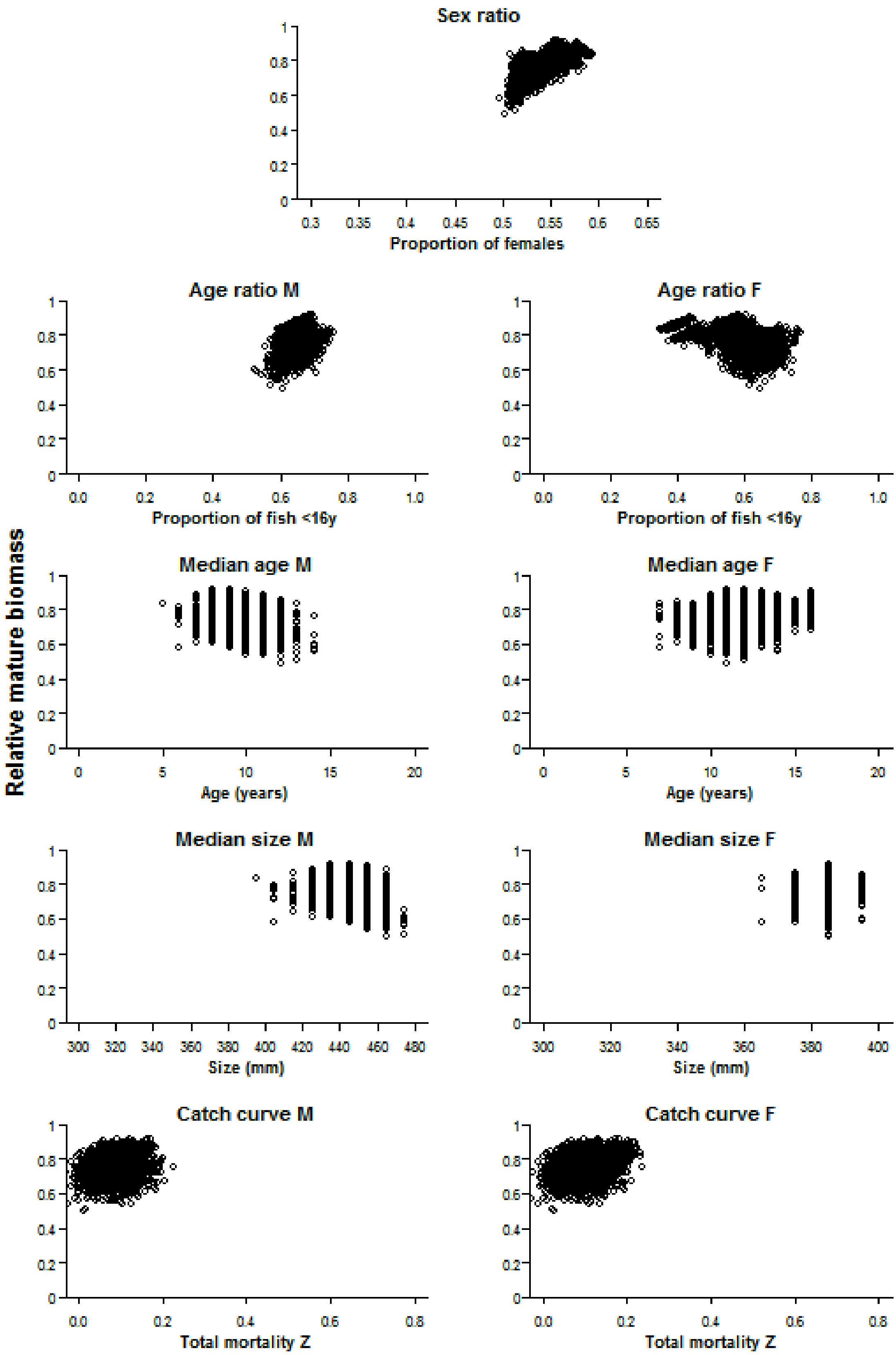

Fig. 6.2: Relative mature biomass versus biological indicators for regional biomass scenario A and recruitment variability $\sigma = 0.6$. Results shown are based on 200 simulations.

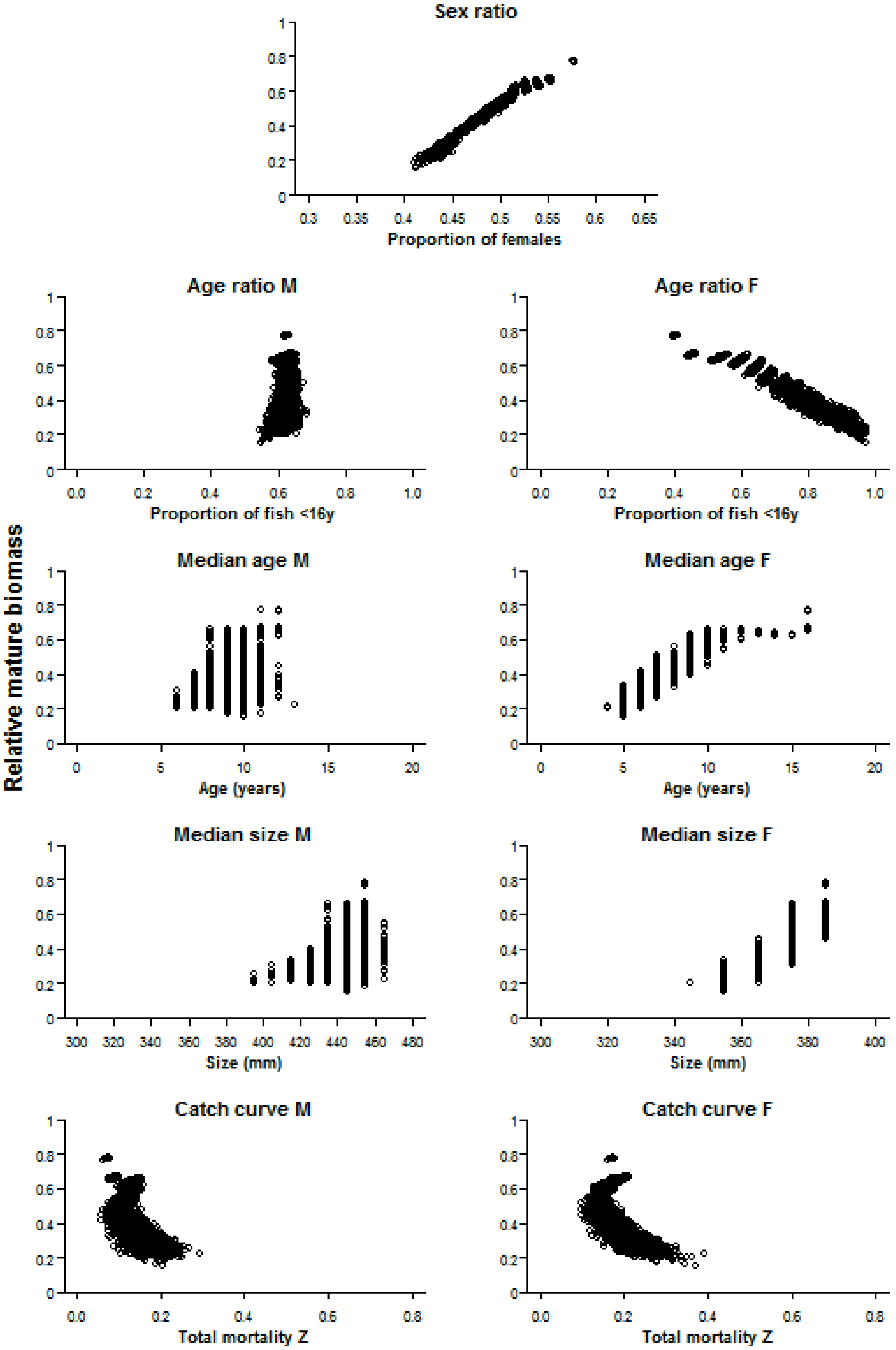

Fig. 6.3: Relative mature biomass versus biological indicators for regional biomass scenario B and recruitment variability $\sigma = 0.1$. Results shown are based on 200 simulations.

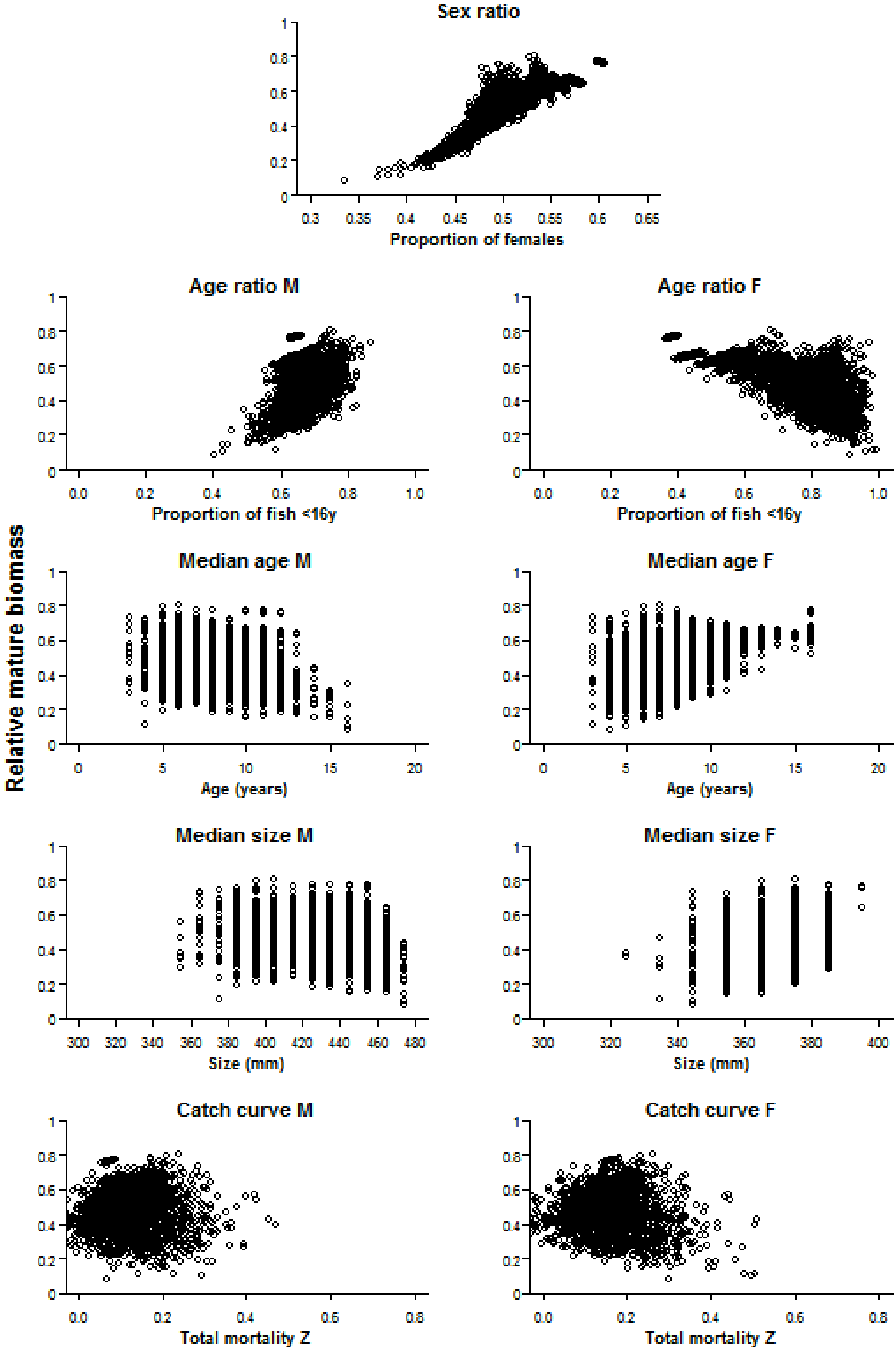

Fig. 6.4: Relative mature biomass versus biological indicators for regional biomass scenario B and recruitment variability $\sigma = 0.6$. Results shown are based on 200 simulations.

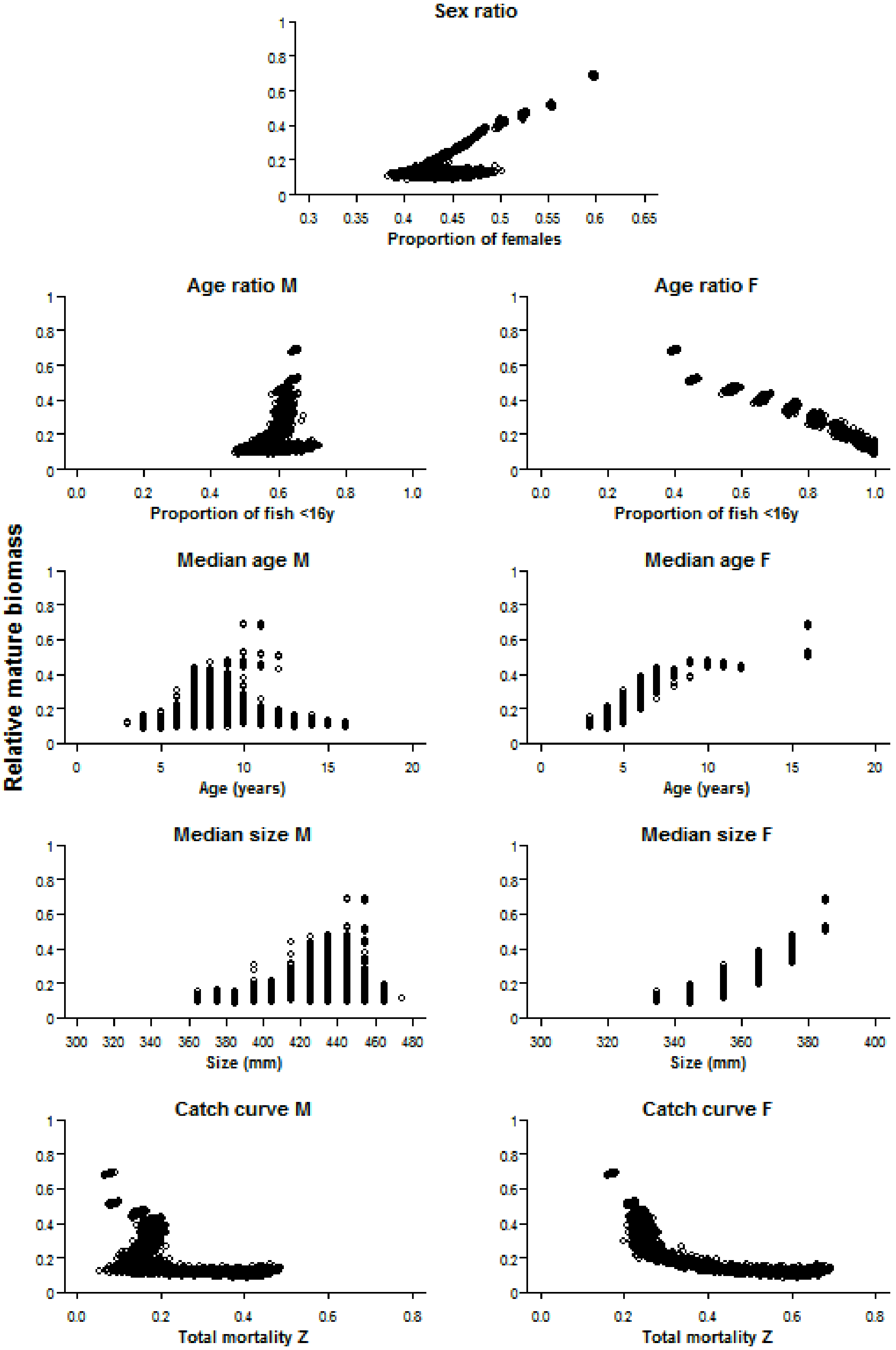

Fig. 6.5: Relative mature biomass versus biological indicators for regional biomass scenario C and recruitment variability $\sigma = 0.1$. Results shown are based on 200 simulations.

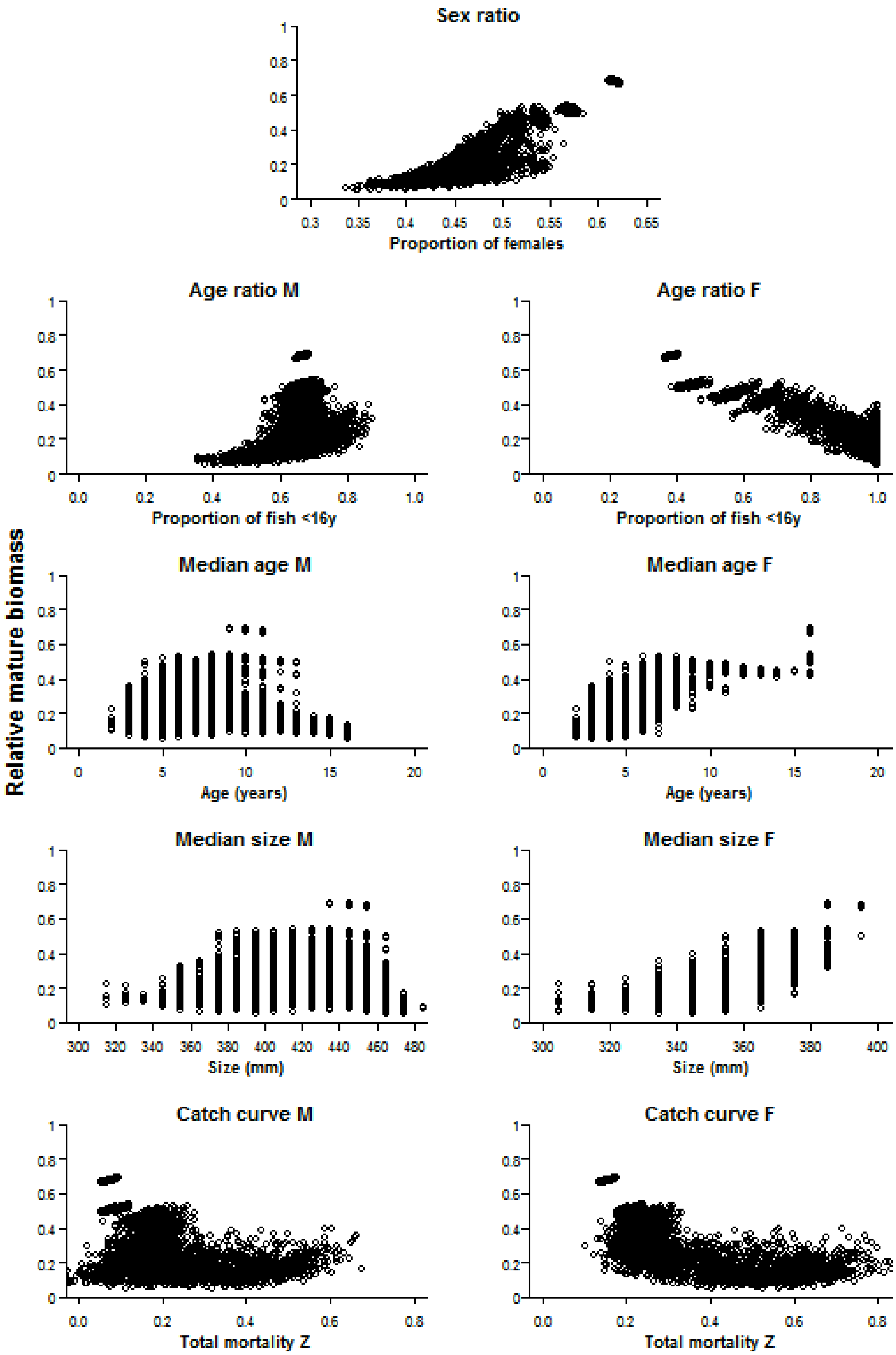

Fig. 6.6: Relative mature biomass versus biological indicators for regional biomass scenario C and recruitment variability $\sigma = 0.6$. Results shown are based on 200 simulations.

The results from each of the three regional biomass scenarios for females were combined to get an overview of the available range of possible mature biomass and corresponding biological property levels. The results confirm that the relationships between sex ratio and all other biological properties for females and relative mature biomass were tighter with low rather than high recruitment variability (Fig. 6.7). But the correlations between any of the properties and mature biomass, while best at low performance measure levels, were generally weak even with low recruitment variability.

Sex ratios generally increased with increasing relative biomass. But low levels of sex ratios were ambiguous even with low recruitment variability, since sex ratios could vary from 0.45-0.5 even in a virtually depleted stock state. For a sex ratio of 0.5, as observed in recent samples from the Bicheno or Tasman regions, any biomass level from near-depletion to 0.5 or 0.8 of unfished biomass for low or high recruitment variability respectively, was possible. Thus, medium and high sex ratios did not discriminate any particular stock state but low sex ratios could be informative (Fig. 6.7). A sex ratio of 0.45 indicated that the relative stock biomass could not be more than about 40%.

The measures for age ratio, median age and median size behaved similarly. All provided the best relationship to mature biomass at low biomass levels, but the relationship lacked discriminatory power with high recruitment variability. For example, mature biomass seemed to be correlated to median age at lower levels from 3-9 years with low recruitment variability. However, for high recruitment variability the range of relative mature biomass (with a maximum of 1.0) at any given median age generally exceeded 0.5. Thus, a median age of 7 years which was commonly found in the samples from the Bicheno region during recent years, could be related to any relative mature biomass from 0.1-0.85. Interestingly, high median ages, in this case the maximum model median age of 16 years, could be found in populations which were reduced to less than 0.4 of its virgin biomass.

Similarly, the relationship between median size and mature biomass was reasonably tight at low recruitment variability, but relatively insensitive at high recruitment variability (Fig. 6.7). Only at the lower end of the range, small sizes indicated a significant reduction in biomass. This occurred when basically all fishing relied on recruits of the same year and virtually no biomass from older fish was left available. However, these size classes were far below the lowest levels observed in the wild population, *i.e.* 360-370 mm. At these levels, the model indicated no trend but rather a large range of biomass levels between near-depletion and 0.7 of virgin biomass.

While the actual recruitment variability levels in the wild population were unknown, the age composition data indicated that considerable variability is likely (Chapter 4). Thus, in order to be precautionary and provide conservative estimates of mature biomass, the scenarios with high recruitment variability and therefore high uncertainty seemed to be more relevant for this analysis.

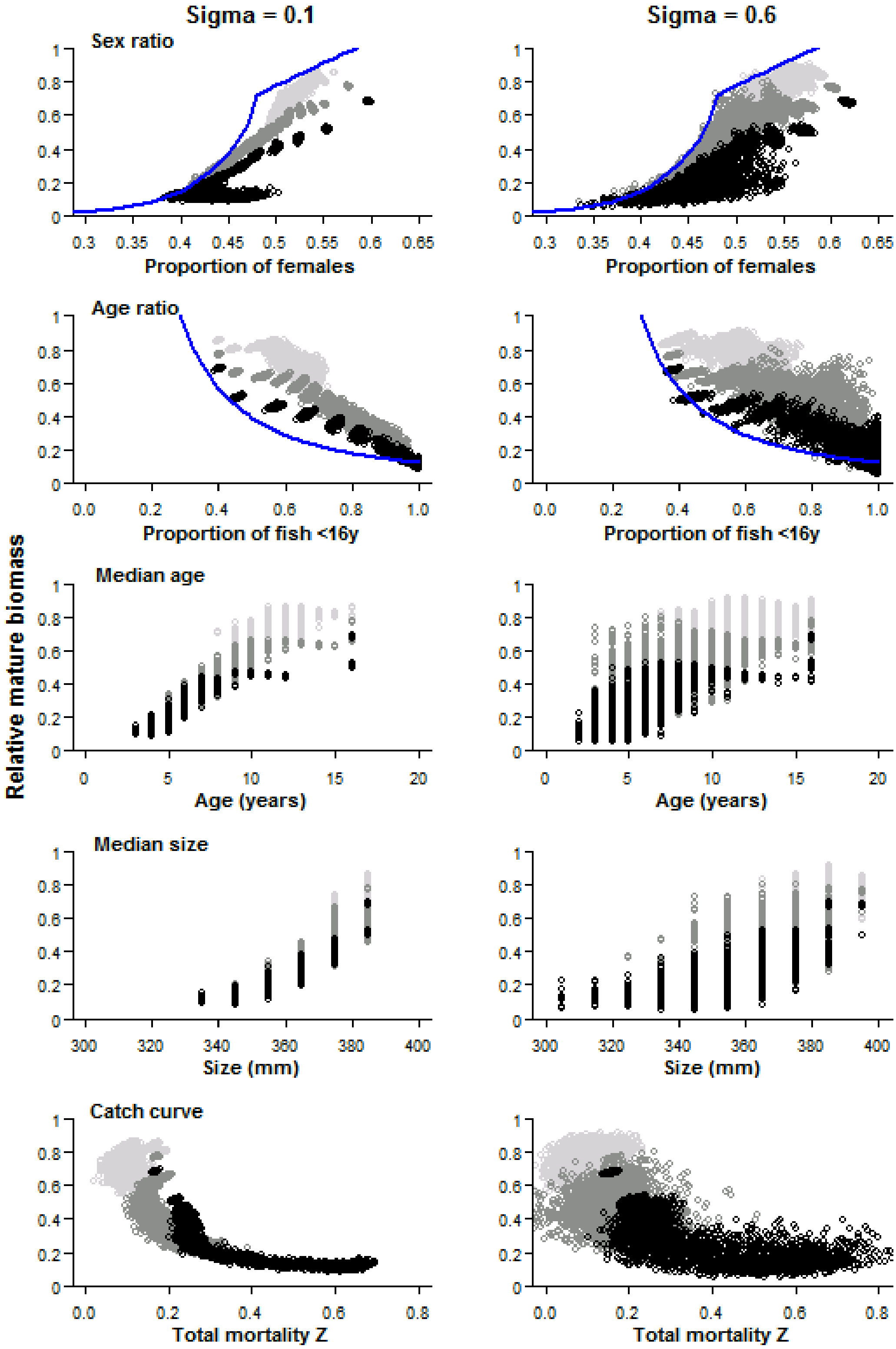

Fig. 6.7: Relative mature biomass versus biological properties used as performance measures for females with recruitment variability $\sigma = 0.1$ and 0.6 and all regional biomass scenarios (A: light grey, B: dark grey, C: black). The bold line in the sex ratio graph describes the maximum estimate, the bold line in the age ratio graph the minimum estimate of relative mature biomass (see text for estimation). Results shown are based on 200 simulations.

Since any single biological property was unable to provide a precise estimate of relative mature biomass, combinations of measures were explored. The sex ratio and the age ratio within the female population appeared the only promising measures, since only they had distinct maximum or minimum limits. Functions representing maximum and minimum limits of biomass were fitted to the maximum and minimum values for all observed levels of the sex ratio and age ratio measures. The maximum limit of biomass estimates based on the sex ratio measure (x_{sr}) was given by an exponential and a linear function:

$$\widehat{B}_{SP}^{upper} = \begin{cases} 0.00009883\exp^{(18.3027x_{sr})} & \text{for } x \leq 0.48 \\ 2.7237x_{sr} - 0.5832 & \text{for } x > 0.48 \end{cases} \tag{6.1}$$

The minimum limit of biomass estimates based on the age ratio measure (x_{ar}) for females was given by a single power function:

$$\widehat{B}_{SP}^{lower} = 0.1213x_{ar}^{-1.6893} \tag{6.2}$$

Unfortunately, the predictive power of the combination of minimum and maximum estimates of relative mature biomass was generally low and depended on the specific levels of sex ratio and age ratio. Compared to the real relative mature biomass of the simulated population (Fig. 6.8), the precision of the biomass estimation, *i.e.* the difference between the maximum and minimum estimation of biomass based on sex and age ratio, was poor overall (Fig. 6.9). The only exceptions were situations with a simultaneous high ratio of young females and low age ratio. This was the case when mature biomass was at near-depletion. At higher levels of relative mature biomass, the biological measures predicted stock status only imprecisely and often only a large range of biomass was predicted. Particularly at biomass levels which are most important to stock assessment, when mature biomass has been reduced to 0.2-0.4 of the unfished levels, predictive power was low. For example, for real biomass levels between 0.2-0.3 of virgin biomass, in only about 30% of all simulations the estimated biomass ranges were lower than 0.2, while in all other cases the estimated ranges were higher and at times over 0.6 (Fig. 6.10). For real biomass levels between 0.3-0.4 of virgin biomass, the minimum biomass range predicted by the biological indicators was 0.2, and the situation was worse for all higher real biomass levels.

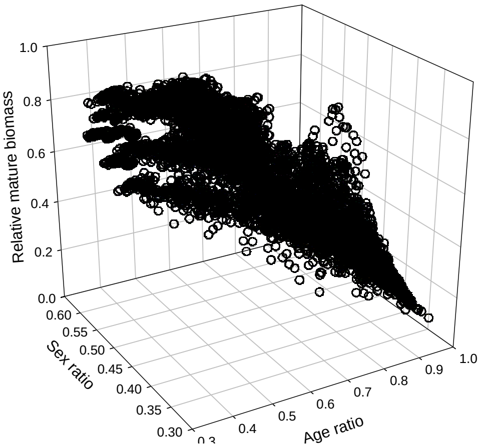

Fig. 6.8: Known relative mature biomass compared to the sex ratio and age ratio measures. Results shown are based on only 50 simulations for greater visual clarity.

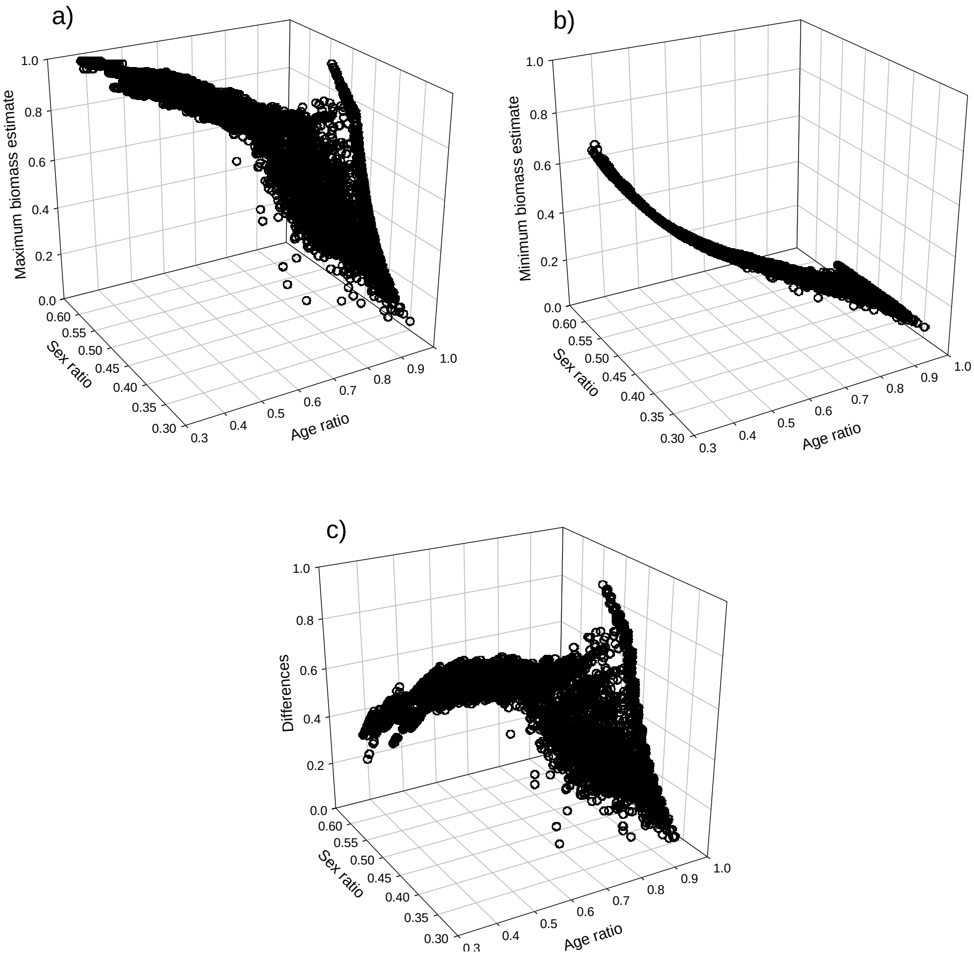

Fig. 6.9: (a) Maximum and (b) minimum estimate of relative mature biomass based on the sex ratio and age ratio measures; and (c) the difference (precision) between the minimum and maximum estimates of biomass. Results shown are based on only 50 simulations for greater visual clarity.

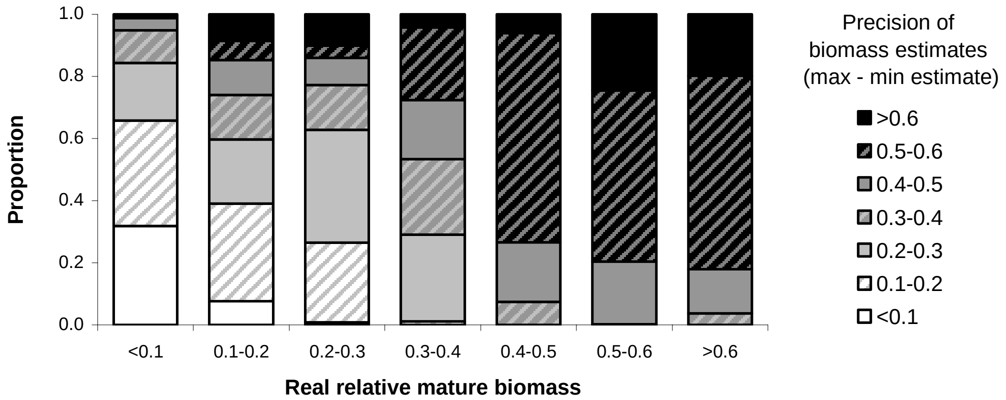

Fig. 6.10: Precision of estimation of relative mature biomass: Relative proportion of all simulations which resulted in different levels of precision for the estimated biomass range (estimated as difference between maximum and minimum estimates) for a given level of real biomass relative to virgin biomass. Proportions scaled to 1 for each level of real relative mature biomass.

6.3.2 Effects of sampling size on estimates of biological properties

Predicted performance measures obtained as perfect fishery-independent representation of the simulated population (operating model) and observed measures based on random samples thereof were highly correlated for sex ratio and all biological properties for females (Table 6.3 and Fig. 6.11). As expected, the correlation improved with increasing sample size, indicated by decreasing the standard deviations and increasing R^2. However, very high and, for the banded morwong fishery, unrealistic sample sizes of 1000 and more individuals were required for a reliable estimate of sex ratios.

Observed median ages were strongly biased for predicted ages above 9 years in females, but were more closely correlated for younger ages. This pattern was stronger in males, although the majority of estimates seemed to follow a linear trend between the estimate and sample size, and the correlation improved substantially with larger sample sizes. Large median sizes in males were also poorly correlated, while the correlation for sizes smaller than 380 mm was relatively tight (Fig. 6.11).

Table 6.3: Regression results between observed and predicted biological properties for males and females from different sample sizes (see Fig. 6.11).

Property and Sample size (N)	Males				Females			
	Slope	Intercept	StDev	R^2	Slope	Intercept	StDev	R^2
Sex ratio								
N = 150					0.9987	0.0007	0.0407	0.57
N = 250					0.9998	0.0000	0.0315	0.70
N = 400					1.0005	-0.0002	0.0250	0.78
N = 1000					1.0004	-0.0001	0.0157	0.90
Age ratio								
N = 150	1.0026	-0.0015	0.0559	0.44	0.9994	0.0005	0.0469	0.94
N = 250	0.9995	0.0003	0.0433	0.57	1.0000	-0.0001	0.0360	0.96
N = 400	0.9955	0.0032	0.0341	0.68	1.0001	-0.0002	0.0285	0.98
N = 1000	1.0032	-0.0021	0.0214	0.84	0.9997	0.0002	0.0180	0.99
Median age								
N = 150	0.8642	1.4422	2.1935	0.49	0.9650	0.3181	1.3871	0.88
N = 250	0.9047	1.0046	1.7880	0.61	0.9779	0.2177	1.1210	0.92
N = 400	0.9248	0.7571	1.4883	0.70	0.9862	0.1336	0.9287	0.95
N = 1000	0.9604	0.4029	1.0190	0.84	0.9935	0.0690	0.6532	0.97
Median size								
N = 150	0.9025	40.8400	18.2564	0.64	0.9891	3.8422	6.4750	0.85
N = 250	0.9309	29.0842	14.8698	0.74	0.9840	5.8624	5.5722	0.88
N = 400	0.9498	21.1031	12.2733	0.82	0.9886	4.1903	4.8321	0.91
N = 1000	0.9720	12.0454	8.5379	0.90	0.9878	4.5175	3.7820	0.94

Total mortality Z derived by the catch curve analysis was strongly underestimated, except for small $Z < 0.2$. Observed Z was biased downwards and generally peaked at a level well below the real level, but this downward bias decreased slightly with sample size. While real maximum total mortality was predicted to be as high as $Z = 0.90$, the observed Z was 0.33 and 0.38 for males and females with $N = 150$, 0.44 and 0.38 with $N = 250$, 0.51 and 0.43 with $N = 400$, and 0.62 and 0.54, respectively with $N = 1000$. The source of this major bias when using catch curves remains unclear, but may be related to the extended age-structure of the banded morwong and the fact that the analysis is restricted only to ages generally below 16 years.

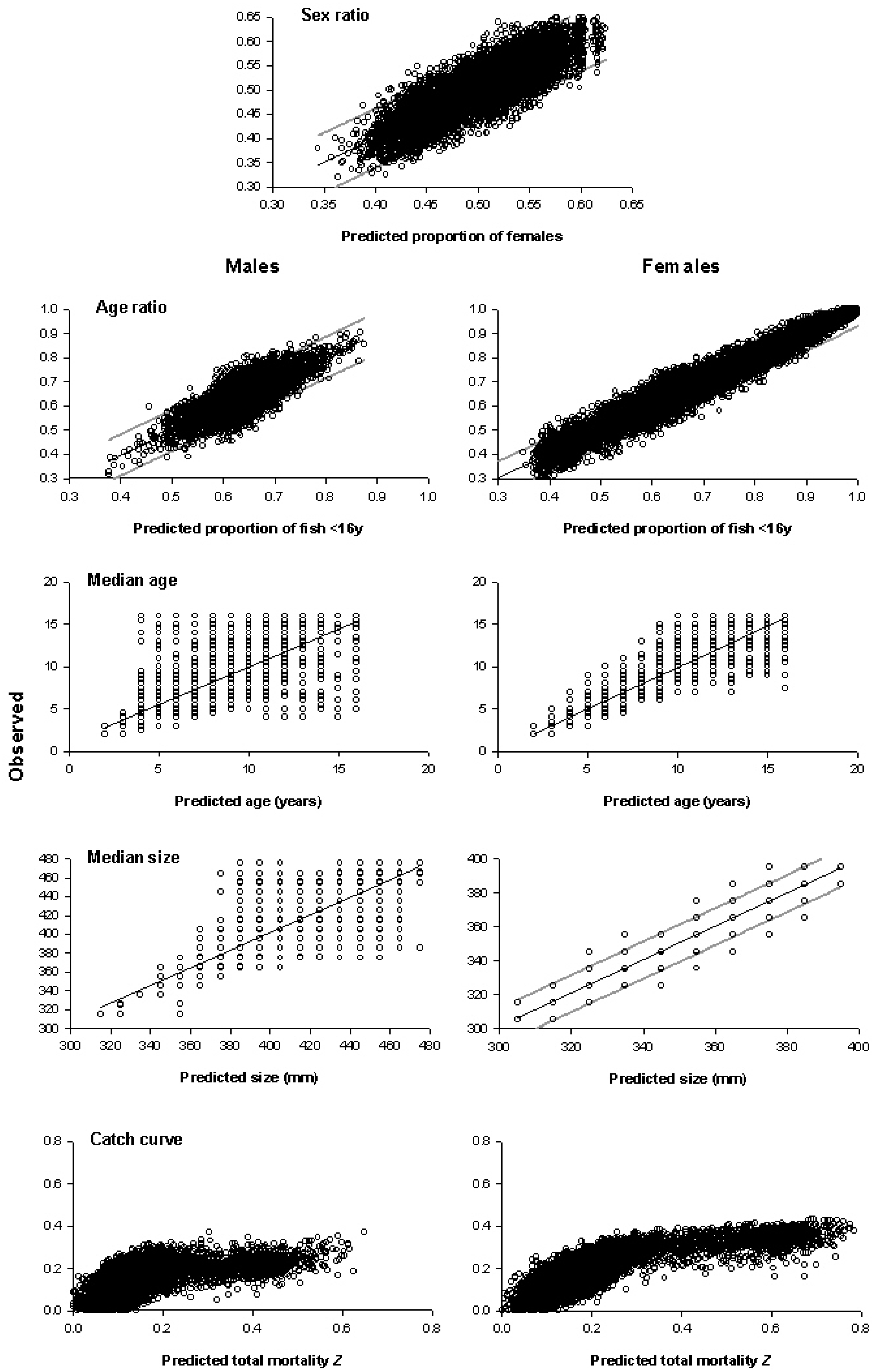

Fig. 6.11: Observed (sampled) versus predicted (real) biological indicators with regression line and 95% confidence intervals based on an observed sample size of $N = 250$. Results shown are based on only 50 simulations for increased visual clarity.

6.4 Discussion

Biological population properties generally appear unable to act as stock performance measures or contribute to a quantitative stock assessment for data-poor species such as banded morwong. The correlations between any of the tested biological properties and the relative mature biomass, both derived from the simulated population, were weak and often ambiguous. There were, however, sex-based differences. The sex ratio and the potential measures from females (age ratio, median age and median size) appeared to be more suitable than the same properties for males. This is not surprising, given the small number of ages that the fishery impacts the male population, compared to females which enter the fishery at a young age and typically remain vulnerable for the rest of their lives.

Recruitment variability was only partly responsible for the poor performance. Even with low recruitment variability $\sigma = 0.1$, the relationships between most of the potential performance measures and mature biomass were generally too imprecise for a useful estimation of stock status. In addition, the performance of most biological measures varied strongly over their range, with *e.g.* medium to high median age and median size correlating poorly to mature biomass.

Assuming high recruitment variability, the precision of the correlations between the biological population properties and relative mature biomass decreased substantially. Given that real recruitment variability is unknown but appears to be significant (Chapter 4), the more conservative or cautious relationship should be assumed for a quantitative assessment. This renders any use of a single biological property as generally too imprecise.

The same was also the case when biological measures were tested in combination for their power to predict relative mature biomass. The sex ratio and the age ratio of females seemed to be the only promising measures because any given level of sex and age ratio had a distinct maximum and minimum of relative mature biomass, respectively. But the uncertainty of the biomass estimate, *i.e.* the difference between the maximum and minimum biomass estimate, was typically too large for such an estimate to be useful in a quantitative stock assessment. There was the possibility that some combinations would exclude some possibilities but this would be only a minor benefit in an assessment.

Sampling error added to the uncertainty in the biomass estimates based on the biological properties. Naturally, the uncertainty was reduced as sample sizes increased. But sampling error was high, particularly for estimates of the sex ratio, with a sample size of $N = 250$, which was typical in recent sampling events from the East coast of Tasmania. At such sample sizes, substantial sampling biases from the fishing operation or the fish distribution and behaviour would have to be expected as well.

The imprecision of biological population properties as stock performance measures, together with uncertainty due to sampling errors and the often poor spatial representation of biological samples, severely restrict the use of biological properties in stock assessments of reef fish species. Where no comprehensive assessment model can be constructed, biological properties are unable to replace an assessment based on commercial catch and effort data, or, in fact, even quantitatively supplement this assessment, because their interpretation can be highly ambiguous. Nevertheless, biological data from fishery-independent or fishery-dependent sampling are valuable for a qualitative assessment. These data should be collected and used to determine the life-history characteristics of the fished species and inform the stock assessment about potential risks and trends in the fish populations.

In data poor fisheries where there is an array of disparate information a strategy of weight-of-evidence appears to be the most productive approach to discerning the stock status. For example, the fact that the growth and size-at-maturity has changed over recent years in banded morwong indicates some large changes have occurred in the population. This is counter to the relative stability in the catch rates that have been observed recently in the commercial fishery.

CHAPTER 7: MANAGEMENT STRATEGIES USING DECISION RULES BASED ON CATCH RATES

7.1 Introduction

Commercial catch and effort data is often the only data source in data-poor fisheries and therefore commonly used as the base for stock assessment (Chapter 2). Such assessments typically include analyses of catch, effort and catch rate trends and/or comparisons of current levels in relation to reference levels that may trigger a management response (*e.g.* Lyle *et al.* 2005). However, because the reference levels and the type of management response are often unspecified or only vaguely specified, the assessments are predominantly descriptive without clear recommendations and the management or any management arrangements may remain inactive.

In this chapter, we examine if catch and effort information, in particular catch rate data, can be used in a stock assessment and management to lead to desirable management outcomes. Unlike biological population properties which can be directly compared to different levels of stock status (Chapter 6), catch rate data are only relative measures of stock biomass, and their dynamics depend on the combination of past and present catch and available biomass. It is therefore not possible to perform a meaningful comparison directly between catch rates and mature biomass levels.

Catch rate measures need to be tested in a framework of decision rules specifying management objectives and strategies. This is best done in a management strategy evaluation (MSE; Smith 1994). An MSE compares the performance of different management strategies through simulated time. Thus, to examine whether catch rates can be used as performance measures, clear management objectives, reference limits and management responses need to be defined. Similar to the Monte Carlo simulations in Chapter 6, an operating model forms the base of an MSE and needs to be able to realistically represent and simulate the dynamics of the fishery and the stock.

The interpretation of the performance of any set of decision rules obviously depends on the particular management objective(s) selected for a fishery. The objectives used here are only vaguely defined as to stabilise or rebuild the current catch rates in the fished stock, implying a stabilisation and rebuilding of the mature biomass. If different strategies performed comparably, preference would be given to those that were robust to different realities (as defined by the operating model) and those that would return higher catches. These objectives allowed for at least a qualitative comparison of the different management rules. However, to implement working management controls, objectives need to be properly defined and, ideally, negotiated between the fisheries management and industry (*e.g.* Lane and Stephenson 1999). This process still needs to be undertaken for all fisheries in Tasmania, since the objectives declared in the Tasmanian Scalefish

Management Plan (Anon. 1998) are relatively vague and have not been negotiated between the stakeholders of the fisheries.

7.2 Methods

7.2.1 General methods

The model used for the analyses considered in this chapter was the same as that described in Chapter 6. Descriptions of the model specifications and initiation algorithm can be found in Sections 6.2.1 to 6.2.4 and Appendices 3 and 4.

As in Chapter 6, model uncertainty was the major source of uncertainty for the banded morwong simulations. Therefore, the simulations consisted of setting up a balanced array of plausible options for the model structure and using that array to determine whether the various decision rules were able to achieve their management outcomes.

7.2.2 No management controls: constant catch scenarios

As a base case, and representing the current management arrangements (licences, gear controls and size limits), scenarios without explicit decision rules were investigated in a harvest strategy evaluation. In these scenarios, reported catch was assumed to be constant over the projection period, representing a form of unadjusted allowable catch, where the management control is not altered over the projected years even when catch rates and effort change substantially. Largely based on the current and historical catch, four different catch scenarios and their impact on catch rates, the effort required to achieve the catch, and relative mature biomass were examined.

A *status quo* scenario with a total annual catch of 36 tonnes was compared with two scenarios where 2/3 and 4/3 of the *status quo* were taken, *i.e.* 24 tonnes and 48 tonnes respectively. Finally, an extreme scenario of 80 tonnes annual catch was investigated. This latter catch scenario is comparable to historically high catches and has relevance given the intention by industry to explore overseas markets for banded morwong (and other live-fish).

Catches or effort were assumed to follow the management decision, *i.e.* the total catch allowed was taken in all years. Only when the catch could not be taken, because of insufficient exploitable biomass, was the catch reduced so that the harvest rate would not exceed 0.95. Reported effort or catches were assumed to be generally under-reported compared to the actual fish removed from the stocks or killed in the operation because of barotrauma or seals (Chapter 6). However, this applied only to legal-sized fish, while all non-legal fish were assumed to be returned live to the sea.

Again, all three regional biomass scenarios A, B, and C (see Chapter 5) with recruitment variability $\sigma = 0.1$ and 0.6 were investigated for a scenario with 75% of the biomass onshore and a movement rate of 0.5.

7.2.3 Decision rules based on catch rates

The operating model was modified to include a management regime that imposed control over the fishery through modification of allowable reported catch. The decision rules implemented were hypothetical in the sense that the fishery management currently uses no catch control at all. Decision rules could equally well have been applied to effort but controlling catch provides outcomes that are more easily understood and so only catch related decision rules were explored.

Management decision rules were based on performance measures derived from commercial fishery data, in particular standardised catch rates. Two types of performance measure were examined, with a lower and upper reference limit defined for each (Table 7.1). The first measure was the gradient of catch rates across the previous four years. If this gradient exceeded either the lower limits of -0.1 or -0.05 or the upper limits of 0.05 or 0.1, then catch level changes were instigated. The second measure was related to threshold catch rates with a lower limit of 0.6 or 0.8 and an upper limit of 1.0 or 1.2, below and above which catch level changes were implemented. Both measures were applied either separately or in combination.

Table 7.1: Characteristics of the evaluated catch rate decision rule scenarios with the reference measures and lower and upper limits.

Catch rate gradients are estimated over the previous 4 years, catch rate thresholds are the current level of standardised catch rates. The management response is a multiplication factor on current catch levels when the lower or upper reference limit was triggered. The different combinations of decision rule led to a total of eight different scenarios being compared.

Reference measure	Reference limits		Management response (multiplication factor)	
	Lower	Upper	Lower	Upper
Catch rate gradient	-0.05	0.05	1 + gradient	1 + gradient
	-0.1	0.1	1 + gradient	1 + gradient
Catch rate gradient	-0.05	0.05	0.8	1.1
	-0.1	0.1	0.8	1.1
Catch rate threshold	0.8	1.2	0.8	1.1
	0.6	1.0	0.8	1.1
Catch rate gradient & threshold	-0.05 or 0.8	0.05 or 1.2	0.8	1.1
	-0.05 or 0.6	0.05 or 1.0	0.8	1.1

By comparing the performance measure from the stock assessment with these reference limits, the management response took one of these three states (Table 7.1):

- If the stock assessment resulted in a measure less than the lower limit, the reported catch was reduced by a specified value, *e.g.* current catch * 0.8;
- If the stock assessment resulted in a measure higher than the upper limit, the reported catch was increased by a specified value, *e.g.* current catch * 1.1;
- If the stock assessment resulted in a measure between the lower and upper limit, the reported catch remained the same as in the previous year.

For the catch rate gradient measure only, two types of management response were tested; (i) a response proportional to the gradient of the catch rates from the previous 4 years with x = (1+ gradient) as the multiplication factor for the catch when the gradient was outside the reference limit range, and (ii) a pre-defined constant response with either a catch decrease by 0.8 when the catch rate gradient or levels fell below the lower reference limit, or a catch increase by 1.1 when the catch rate gradient or levels exceeded the upper reference limit (Table 7.1).

In reality, management decisions are rarely implemented in the following year since data collection, stock assessment and consequent management decisions tend to be part of a rather lengthy process. Thus, the model allowed for a delay between the year y for which the stock assessment was undertaken (in which data was collected), and the year when the management decision was implemented. This delay was typically 2 years. In addition, after a change in the management regime in year y+2, no further change in management regime could be implemented in the following year to simulate a period when the impact of the management change is being assessed. Thus, even when the situation which had triggered the management response (*e.g.* low catch rate levels) in year y may have become worse in the following year y+1, catch or effort levels were assumed to remain stable in year y+3 and no further management changes could be instigated until y+4.

Four initial catch levels of 24, 36, 48, and 80 tonnes were compared to test the ability of the management response to find a balance between catch removal and levels of catch rates and mature biomass in such a way that mature biomass would stabilise at current levels or indeed increase in the long term. While catches of 80 tonnes were assumed to be too high to maintain the current biomass, these scenarios focussed on the ability and response time within the 20-year period to reduce catches to levels at which falling catch rates and biomass trends would be reversed.

7.3 Results

7.3.1 Constant catch scenarios

The harvest strategy evaluations were based on the forward projections of four different constant catch scenarios of 24, 36, 48 and 80 tonnes per annum. Figs. 7.1 to 7.4 summarise the results for the reported and total fishing-induced mortality or 'real catch', and for catch rates relative to 1995, effort required to achieve the catch, and mature biomass relative to the virgin state in 1990 for the regional biomass scenarios A and C and recruitment variability σ = 0.1 and 0.6. Regional biomass scenario B produced intermediate results and is not shown.

Generally, trends were similar for both levels of recruitment variability σ = 0.1 and 0.6, although observed variation in catch rates, effort and relative mature biomass was slightly higher for the latter. The two regional biomass scenarios A and C represented extremes in the biomass attributed to the Bicheno region, with 65% of the total biomass in scenario A, 50% of the biomass in scenario B and only 30% of the biomass in scenario C. Depending on the regional biomass distribution the catch removal impacted the regions differently, in particular the Bicheno region. In contrast, the trends in catch rates, effort and mature biomass from all combined regions were surprisingly similar. This clearly illustrates how an overall stock assessment can obscure regional trends. For example, with an 80 tonne harvest strategy and regional biomass scenario C, catch rates dropped much more quickly and biomass was depleted earlier in the Bicheno region than indicated by an assessment of the overall data (Fig. 7.4).

The effects on catch rates, effort, and relative mature biomass gradually increased with increasing catch. With a constant catch scenario of 24 tonnes (equivalent to an average of 29 tonnes total mortality when seal mortality and other sources are included), relative catch rates, effort and relative mature biomass remained stable over the whole projected period of 20 years (Fig. 7.1). When 36 tonnes were taken annually (equivalent to an average of 44 tonnes total mortality), catch rates and mature biomass slightly decreased over time, while the effort required to achieve the catch increased (Fig. 7.2). With a constant catch scenario of 48 tonnes (equivalent to an average of 58 tonnes total mortality), catch rates fell continuously over the 20 years projection to levels well below the current catch rate levels (Fig. 7.3). Mature biomass also continued to decline and the effort required to support the catch increased substantially in all regions except in the Bicheno region with high relative biomass in scenario A.

As a result of a constant (intended) removal of 80 tonnes, catch rates and mature biomass dropped to very low levels within 10 years and did not recover during the remaining period (Fig. 7.4). In fact, the 80 tonnes catch often could not be taken due to lack of sufficient biomass. Rather, the total fishing-induced mortality or real catch stabilised at only 70 tonnes, and often a substantial part of the existing biomass was taken annually. In reality, this is unlikely to occur since the fishery would become economically unsustainable well before such low biomass levels were reached due to the enormous effort required to achieve the catch. But the scenario clearly demonstrated that the stocks could not tolerate such high catches over a prolonged period.

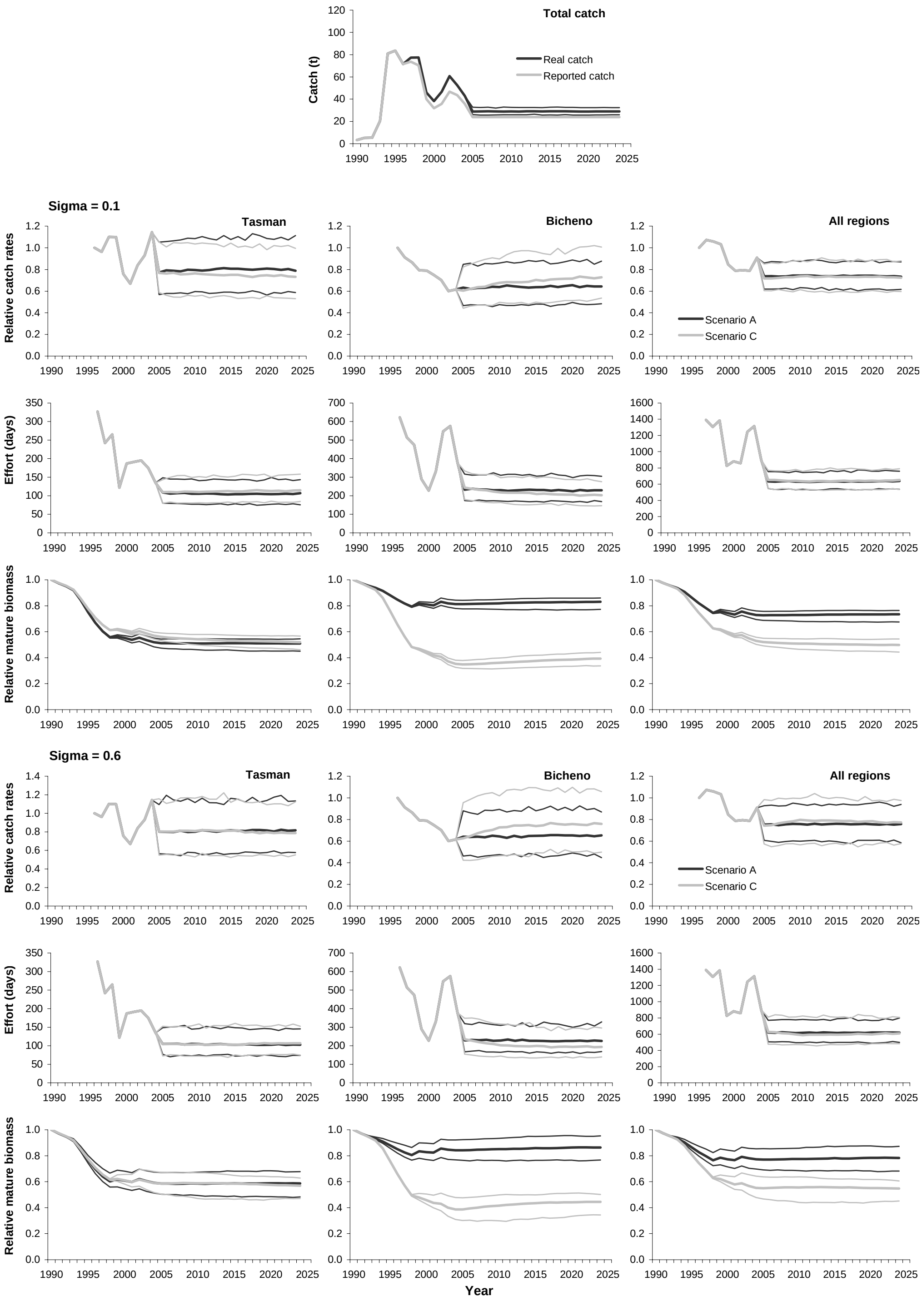

Fig. 7.1: Constant catch simulation with 24 tonnes total reported catch for Tasman and Bicheno region and all regions combined from 2005-2025. Historical data used for 1990-2004. Catch taken (top graph) with reported catch (grey line) and total fishing-induced mortality ('real catch', black lines) which is the reported catch multiplied with a bias and random variation for reporting and seal or other fishing-induced mortalities. Observed catch rates and effort, and mature biomass for recruitment variability $\sigma = 0.1$ and 0.6. Catch rates relative to levels in 1995 (catch rates for all regions combined is the catch-weighed geometric mean of all regions); effort in days needed to achieve the removals; and mature biomass relative to virgin levels in 1990 for regional biomass scenarios A (black lines) and C (grey lines) with median (bold lines) and 95% confidence intervals. Scenarios presented are for 75% onshore biomass and movement rates of 0.5. Results shown are based on 500 simulations.

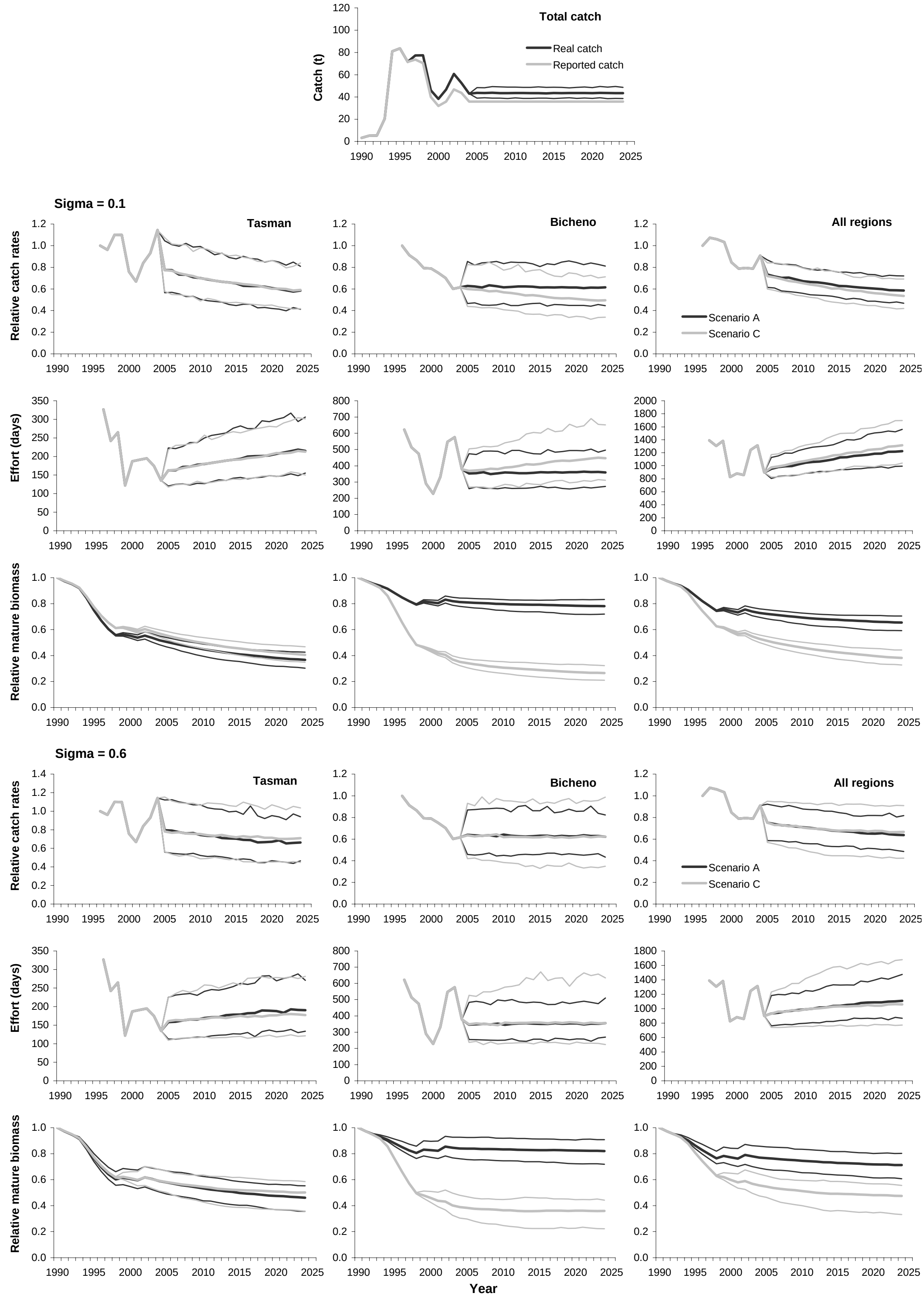

Fig. 7.2: Constant catch simulation with 36 tonnes total reported catch for Tasman and Bicheno region and all regions combined from 2005-2025. See legend to Fig. 7.1 for more details.

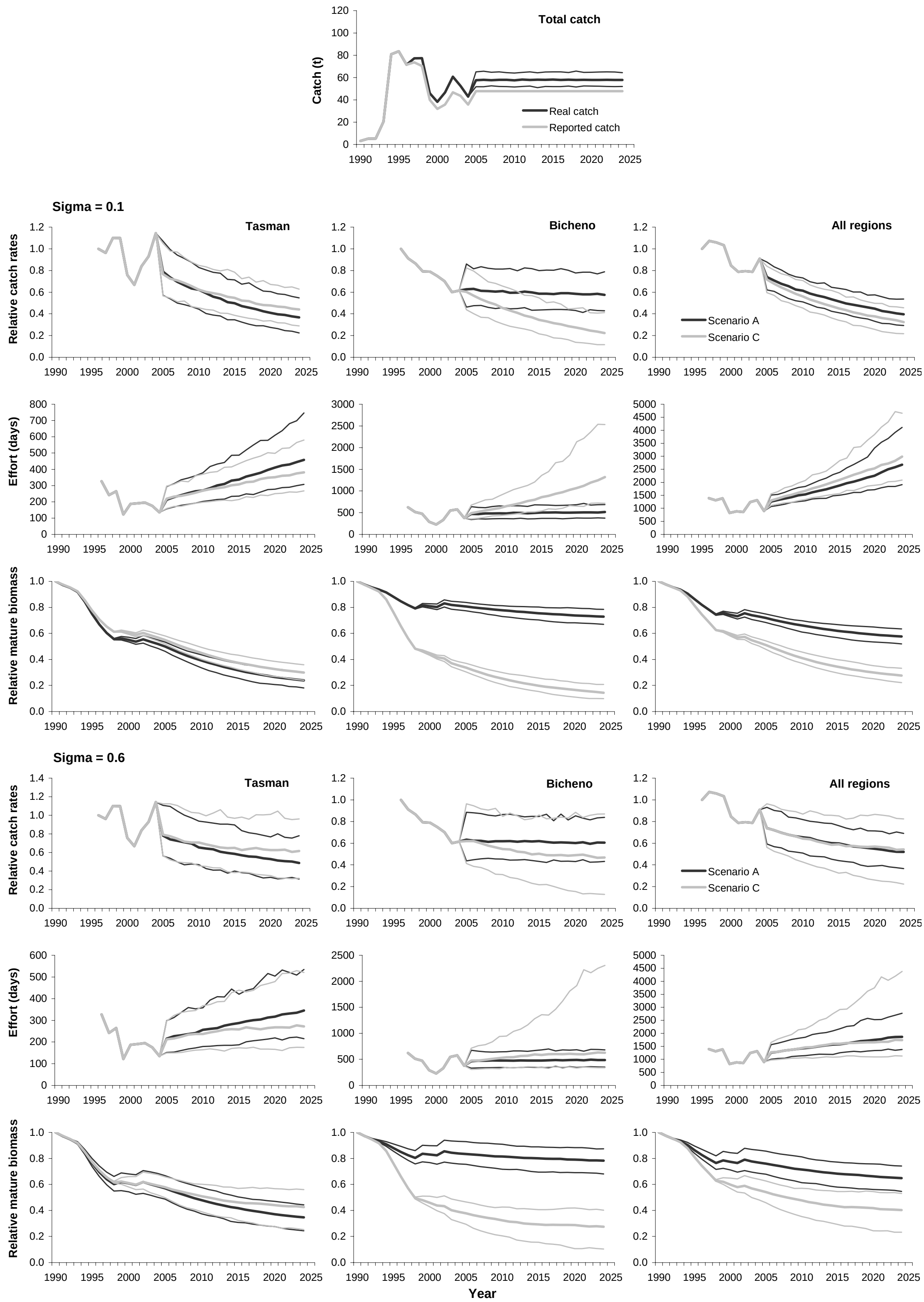

Fig. 7.3: Constant catch simulation with 48 tonnes total reported catch for Tasman and Bicheno region and all regions combined from 2005-2025. See legend to Fig. 7.1 for more details.

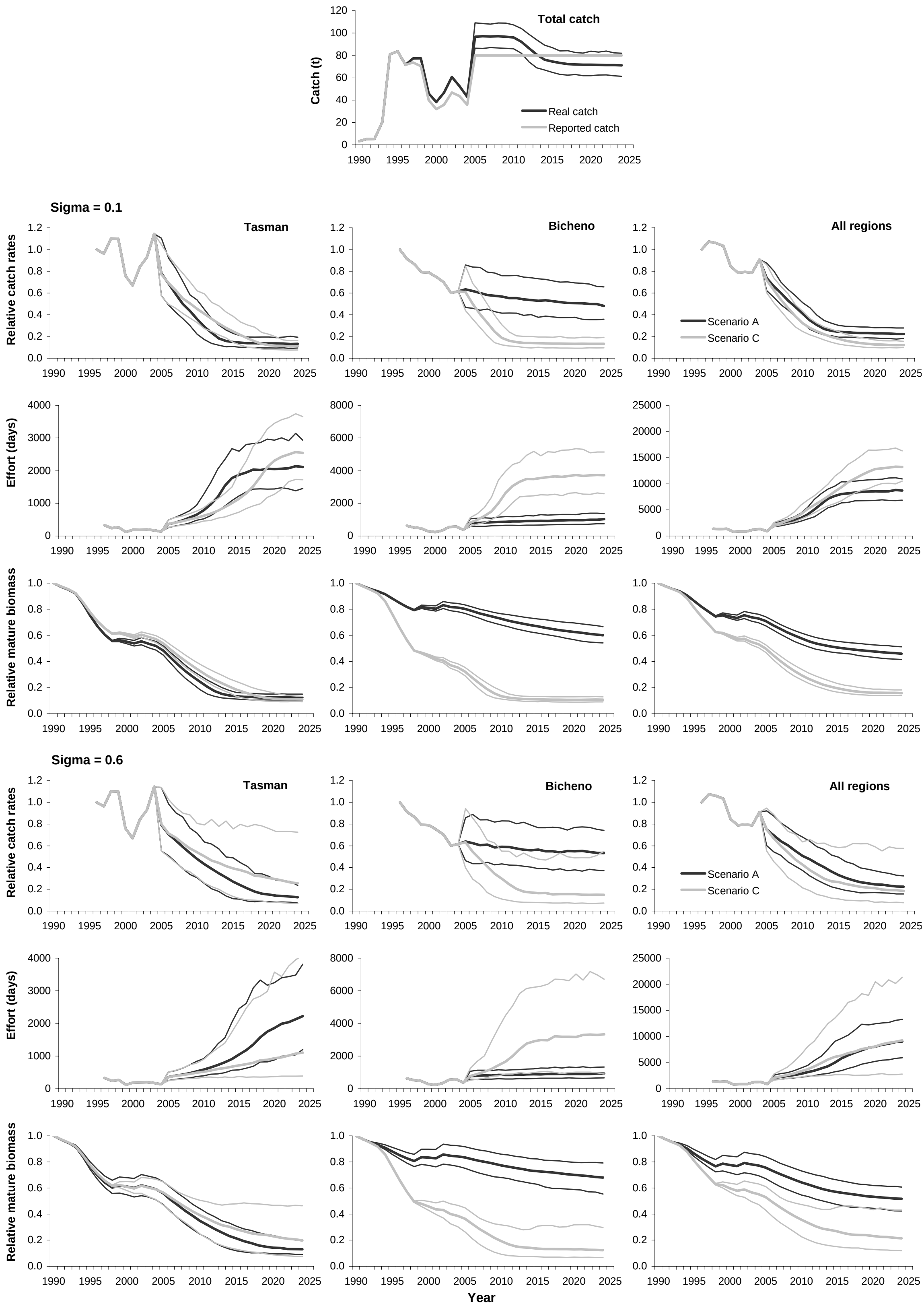

Fig. 7.4: Constant catch simulation with 80 tonnes total reported catch for Tasman and Bicheno region and all regions combined from 2005-2025. See legend to Fig. 7.1 for more details.

7.3.2 Decision rules based on catch rates

Trends of observed catch, catch rates, and effort, *i.e.* without the effects of reporting bias and seal and other fishing-induced mortality, were very similar to the 'real' measures from the simulated populations, when these effects were considered, and thus only the former are presented (Figs. 7.5 to 7.8). Levels of catch rates, effort and biomass differed for regional biomass scenarios A and C. With overall estimated mature biomass being smaller in scenario C (Chapter 4) fishing impacted more intensely on the relative mature biomass in scenario C compared to scenario A. But the two scenarios resulted in similar trends for the overall population when all regions were pooled and thus only results from scenario C are presented.

The upper reference limits were rarely exceeded. Under all scenarios catch rate gradients or the catch rate levels tended to remain below the levels triggering an increase in catches. Thus, the ability of these upper limits to allow a controlled expansion of the fishery's catch was not tested.

The decision rules based on the gradient of catch rates against years prompting a response proportional to the gradient proved too insensitive to stabilise or rebuild catch rates and mature biomass when medium and high catches were taken from the stock (scenario (a) in Figs 7.5 to 7.8). Scenarios with low initial catch levels of 24 tonnes were non-informative, since catch rates remained stable over the 20 years. Neither the tighter gradient reference limits of -0.05 and 0.05 nor the larger limits of -0.1 and 0.1 was ever triggered and thus catch levels were not altered. Relative mature biomass also remained stable over the 20 years of projection. With initial catch levels of 36 tonnes, catch rates and relative mature biomass continued to decrease, although only slightly. At initially high catches of 80 tonnes the proportional management response resulted only in a minor reduction of catches to an average of 50-60 tonnes, and the reduction was slow. This was not enough to rebuild catch rates and mature biomass, both of which stabilised at low levels.

The pre-defined management response based on a fixed increase or decrease of catches performed better than the proportional response, maybe because the management response was more responsive or sensitive to exhibited changes (scenario (b) in Figs 7.5 to 7.8). In particular, the high initial catch scenario of 80 tonnes dropped faster and stabilised at 40-50 tonnes. While catch rates and biomass levels again reached low levels, there was a clear tendency for slow rebuilding. The scenarios with the gradient limits of -0.05 and 0.05 generally performed better than those with limits -0.1 and 0.1, *e.g.* catch rates and mature biomass were stabilised with initial catch of 36 tonnes with the former reference limits, while they continued to drop with the latter limits.

Both lower catch rate thresholds of 0.6 and 0.8 triggered strong responses and strongly reduced catches to 10-20 tonnes (scenario (c) in Figs 7.5 to 7.8). Because relative catch rates prior to the projection were around 0.7, the strategy with a limit of 0.8 reduced even the small initial catches of 24 tonnes, while the strategy with a limit of 0.6 rarely interfered. With medium or high initial catch levels, this strategy resulted in frequent catch reductions with either lower reference limit. This allowed strong rebuilding of catch rates and relative mature biomass to near-current levels within the 20-year projection period, even with high initial catch levels.

The strategy of combining catch rate gradient limits of -0.05 and 0.05 or -0.1 and 0.1 and catch rate thresholds of 0.6 and 1.0 achieved similar results to those of the catch rate limits of 0.6 and 1.0 on their own (scenario (d) in Figs 7.5 to 7.8). By reducing catches earlier they were stabilised at slightly higher levels of 20-30 tonnes, however the performance in terms of rebuilding catch rates and mature biomass levels was slightly reduced. The strategy with the tighter gradient limits generally resulted in slightly higher catches in the final year.

The total catch taken and, inversely, the catch variation expressed as the coefficient of variation over 20-year period reflected the relative catch levels in the last year (Table 7.2 for initial catches of 80 tonnes). If maximising the total catch taken over the 20-year period and minimising the catch variation between years were added as secondary fishery-related management objectives, then the decision rule of combining catch rate gradient limits and catch rate thresholds (scenario d) would have been considered as best strategy.

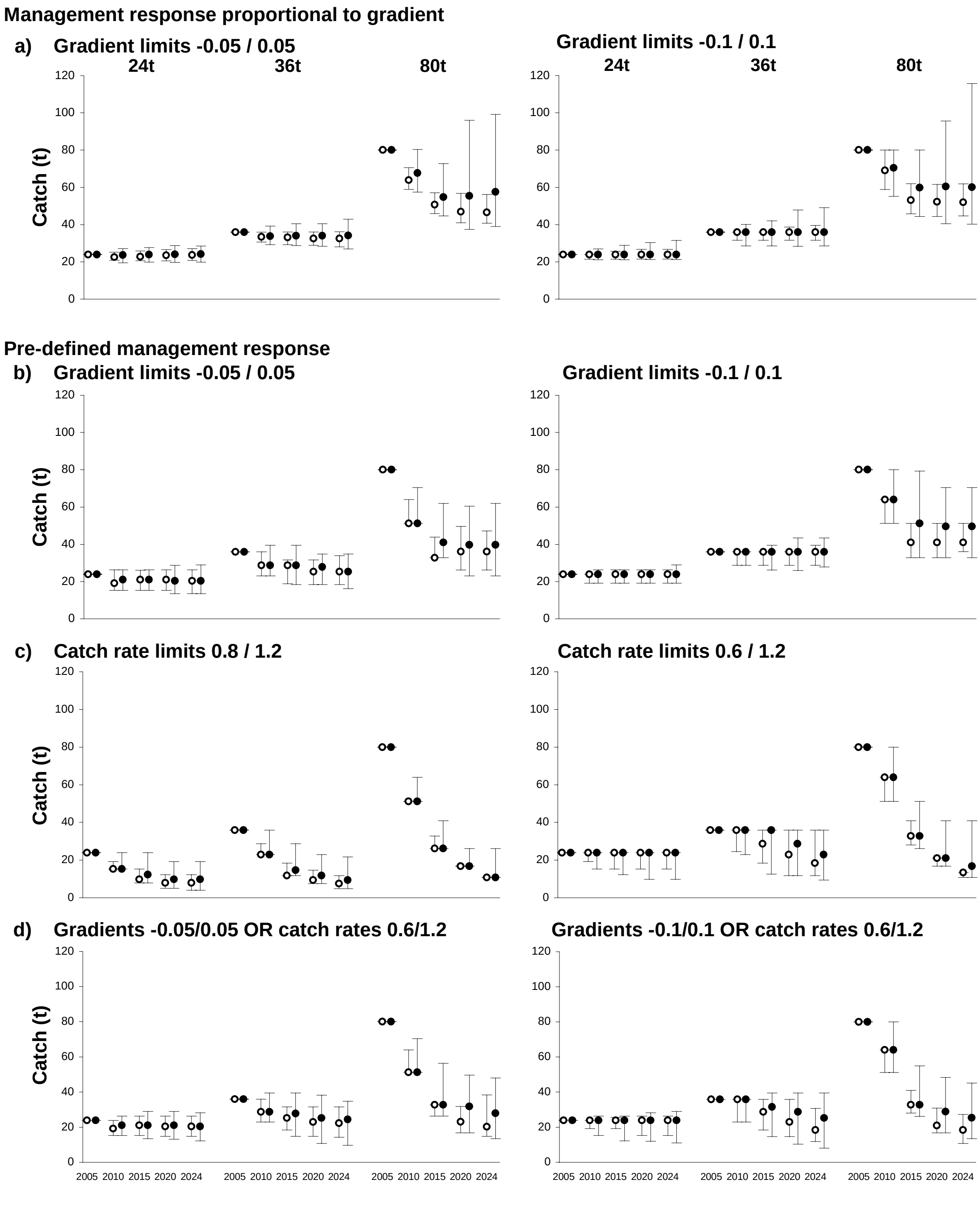

Fig. 7.5: Projections of observed catch (tonnes). With median and 95% confidence intervals for recruitment variability $\sigma = 0.1$ (open circles) and $\sigma = 0.6$ (filled circles) in 5y-steps and for observed initial catches of 24, 36 and 80 tonnes under different decision rule and reference limit scenarios. (a) Management response, *i.e.* catch increase or decrease, is proportional to catch rate gradient once the gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (b) to (d) the management response is pre-defined as a decrease of catch by 0.8 or an increase by 1.1 once (b) the catch rate gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (c) the catch rates are outside the reference range of 0.8 to 1.2 or 0.6 to 1.0; and (d) the catch rate gradient is outside the reference range of -0.05 to 0.05 or the catch rates are outside the reference range of 0.6 to 1.0. Results shown are for all regions pooled and regional biomass scenario C and based on 250 simulations.

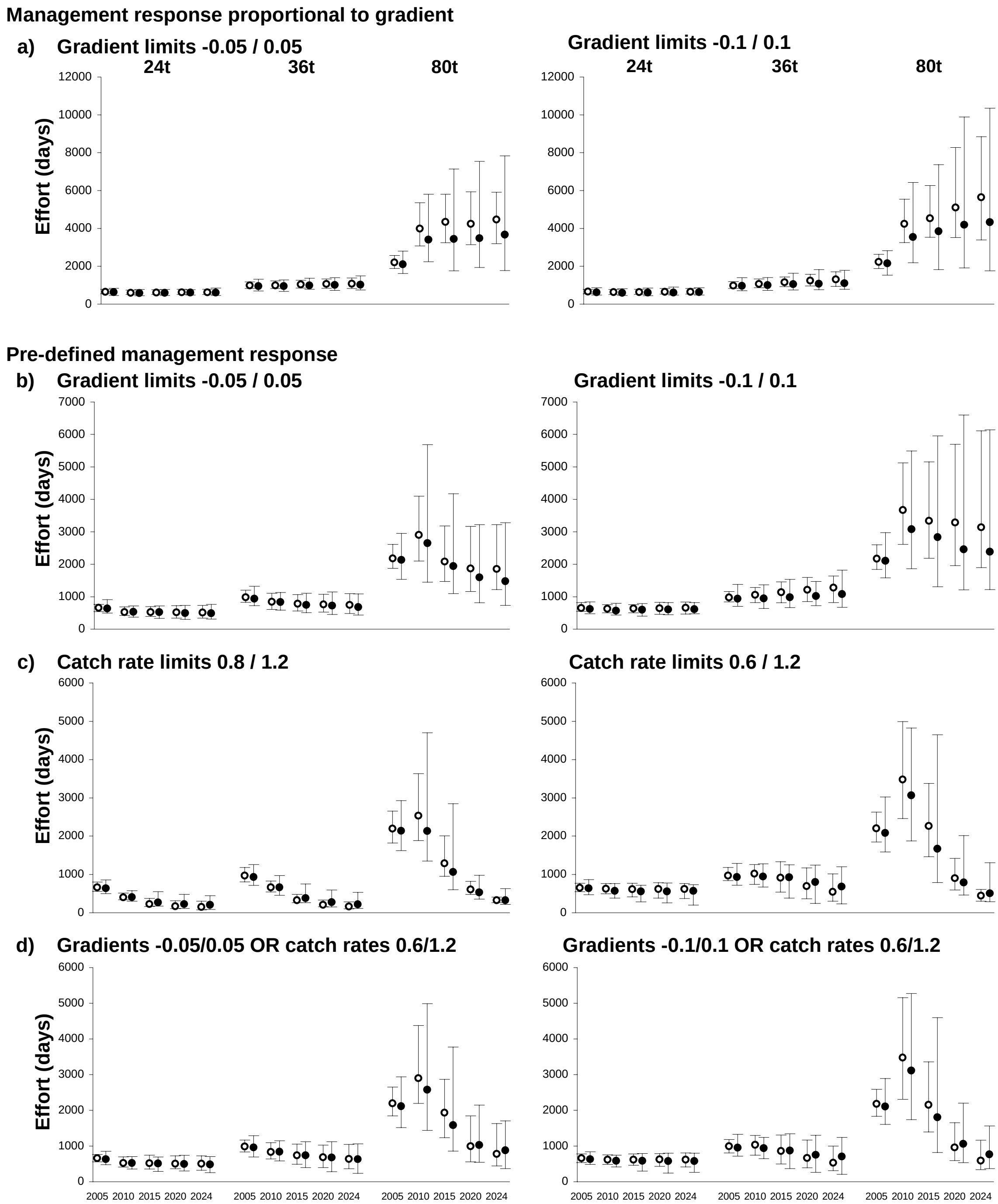

Fig. 7.6: Projections of effort (days) required to achieve the catch. With median and 95% confidence intervals for recruitment variability $\sigma = 0.1$ (open circles) and $\sigma = 0.6$ (filled circles) in 5y-steps and for observed initial catches of 24, 36 and 80 tonnes under different decision rule and reference limit scenarios. (a) Management response, *i.e.* catch increase or decrease, is proportional to catch rate gradient once the gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (b) to (d) the management response is pre-defined as a decrease of catch by 0.8 or an increase by 1.1 once (b) the catch rate gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (c) the catch rates are outside the reference range of 0.8 to 1.2 or 0.6 to 1.0; and (d) the catch rate gradient is outside the reference range of -0.05 to 0.05 or the catch rates are outside the reference range of 0.6 to 1.0. Results shown are for all regions pooled and regional biomass scenario C and based on 250 simulations.

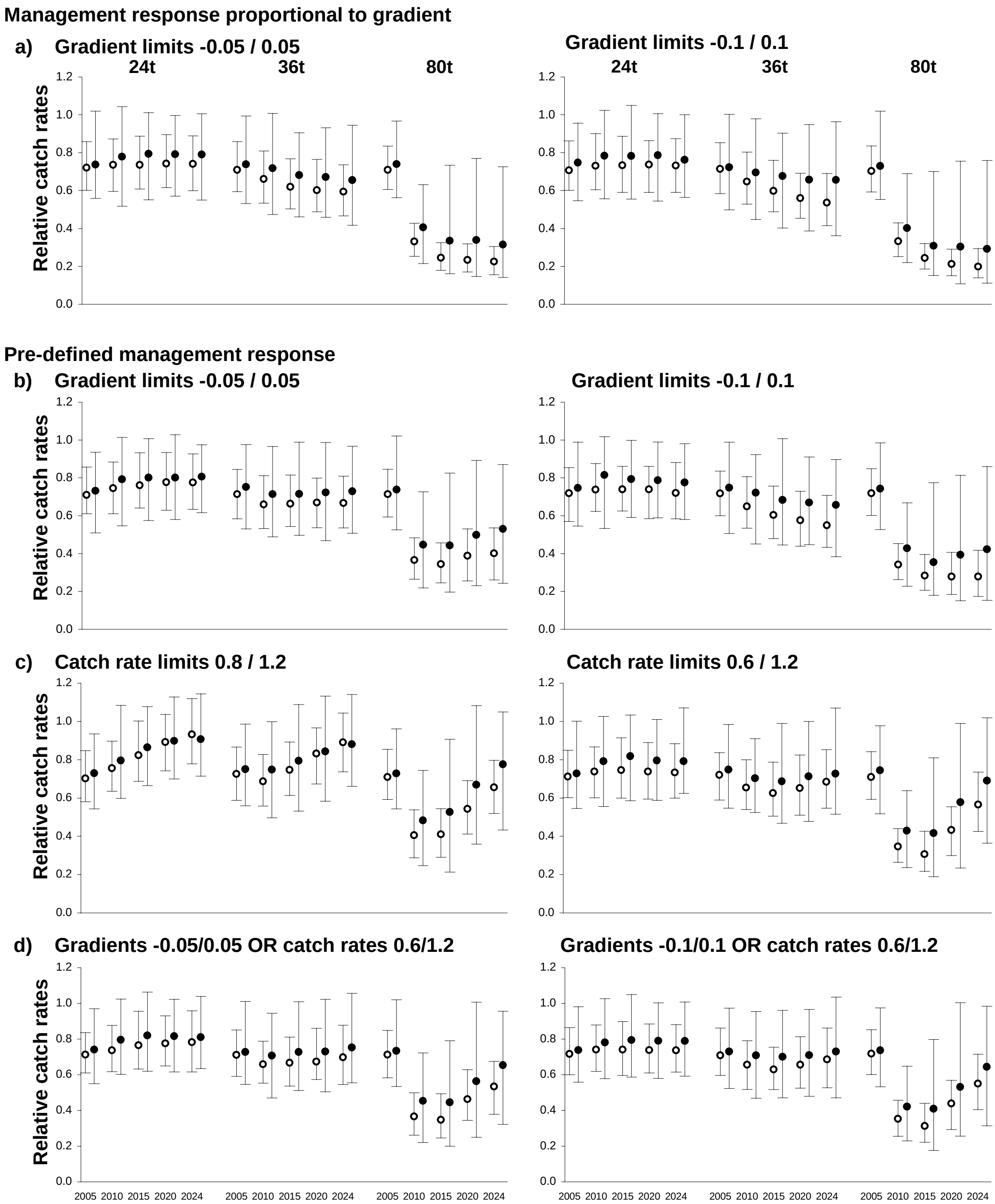

Fig. 7.7: Projections of catch rates relative to levels in 1995. With median and 95% confidence intervals for recruitment variability $\sigma = 0.1$ (open circles) and $\sigma = 0.6$ (filled circles) in 5y-steps and for observed initial catches of 24, 36 and 80 tonnes under different decision rule and reference limit scenarios. (a) Management response, *i.e.* catch increase or decrease, is proportional to catch rate gradient once the gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (b) to (d) the management response is pre-defined as a decrease of catch by 0.8 or an increase by 1.1 once (b) the catch rate gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (c) the catch rates are outside the reference range of 0.8 to 1.2 or 0.6 to 1.0; and (d) the catch rate gradient is outside the reference range of -0.05 to 0.05 or the catch rates are outside the reference range of 0.6 to 1.0. Results shown are for all regions pooled and regional biomass scenario C and based on 250 simulations.

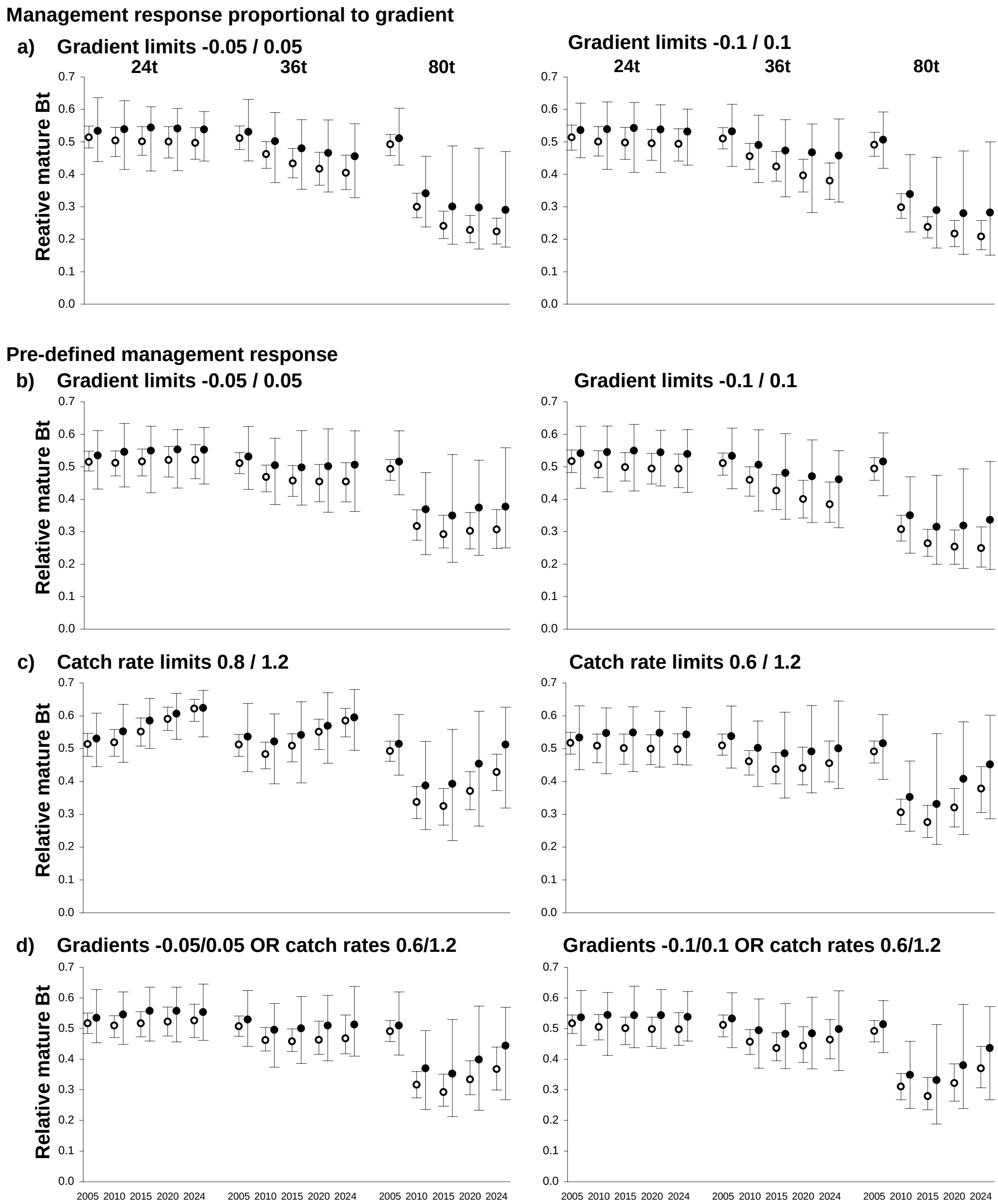

Fig. 7.8: Projections of mature biomass relative to virgin levels in 1990. With median and 95% confidence intervals for recruitment variability $\sigma = 0.1$ (open circles) and $\sigma = 0.6$ (filled circles) in 5y-steps and for observed initial catches of 24, 36 and 80 tonnes under different decision rule and reference limit scenarios. (a) Management response, *i.e.* catch increase or decrease, is proportional to catch rate gradient once the gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (b) to (d) the management response is pre-defined as a decrease of catch by 0.8 or an increase by 1.1 once (b) the catch rate gradient is outside the reference range of -0.05 to 0.05 or -0.1 to 0.1; (c) the catch rates are outside the reference range of 0.8 to 1.2 or 0.6 to 1.0; and (d) the catch rate gradient is outside the reference range of -0.05 to 0.05 or the catch rates are outside the reference range of 0.6 to 1.0. Results shown are for all regions pooled and regional biomass scenario C and based on 250 simulations.

Table 7.2: Catch, catch rate and relative mature biomass results for observed initial catches of 80t.

With median and 95% confidence intervals (given as range *e.g.* 60-82) for recruitment variability $\sigma = 0.1$ and $\sigma = 0.6$ under different decision rule and reference limit scenarios. Measures for catch included the mean and minimum catch (Min) over the 20-year period, the catch in the last year of the projection (2024), total catch and the coefficient of variation of catch (CV) over the 20-year period, Measures for catch rates and relative mature biomass include the minimum levels and levels in the last year of the projection (2024). See Fig. 7.5 for detailed description of the decision rules (a) to (d). Results shown are for all regions pooled and regional biomass scenario C and based on 250 simulations.

	Catch					**Catch rates**		**Relative MatBt**	
	Mean	Min	2024	Total	CV	Min	2024	Min	2024
Sigma 0.1									
a) Proportional response with gradient limits of:									
-0.05/0.05	70	56	56	1398	3.5	0.22	0.22	0.22	0.22
	60-82	47-69	48-70	1205-1632	3.7-2.9	0.16 - 0.3	0.16-0.3	0.19-0.26	0.19-0.26
-0.1/0.1	73	63	63	1466	2.4	0.20	0.20	0.21	0.21
	62-88	52-76	52-78	1244-1753	2.8-1.9	0.14 - 0.29	0.14-0.29	0.17-0.25	0.17-0.26
b) Pre-defined response with gradient limits of:									
-0.05/0.05	57	41	44	1146	7.2	0.32	0.40	0.29	0.31
	46-71	30-55	34-58	925-1420	8.1-5.7	0.23 - 0.45	0.26-0.54	0.25-0.35	0.25-0.37
-0.1/0.1	65	51	52	1291	5.0	0.27	0.28	0.25	0.25
	53-80	38-66	42-66	1056-1601	5.8-3.4	0.17 - 0.39	0.17-0.42	0.19-0.3	0.19-0.31
c) Pre-defined response with catch rate limits of:									
0.8/1.2	43	13	13	869	17.2	0.38	0.66	0.32	0.43
	39-52	12-15	12-15	775-1031	15.4-19.2	0.26 - 0.5	0.52-0.8	0.26-0.37	0.37-0.48
0.6/1.0	51	16	16	1015	16.8	0.29	0.57	0.27	0.38
	42-61	12-18	12-18	837-1213	15.5-18.3	0.22 - 0.41	0.43-0.74	0.23-0.32	0.31-0.44
d) Pre-defined response with gradient limits or catch rate limits of:									
-0.05/0.05 or 0.8/1.2	51	26	26	1013	13.0	0.32	0.53	0.29	0.37
	41-64	16-42	16-48	828-1286	14-9.9	0.23 - 0.45	0.38-0.67	0.24-0.34	0.3-0.44
-0.1/0.1 or 0.6/1.0	51	21	21	1026	15.6	0.30	0.55	0.28	0.37
	42-64	13-34	13-35	838-1282	14.8-12.8	0.21 - 0.42	0.4-0.72	0.23-0.33	0.31-0.44

Table 7.2: Continued

	Catch					Catch rates		Relative MatBt	
	Mean	Min	2024	Total	CV	Min	2024	Min	2024
Sigma 0.6									
a) Proportional response with gradient limits of:									
-0.05/0.05	76	66	70	1519	1.8	0.31	0.31	0.29	0.29
	59-104	44-89	47-119	1171-2088	3.8-1.1	0.13 - 0.61	0.14-0.73	0.16-0.45	0.18-0.47
-0.1/0.1	79	71	73	1588	1.2	0.29	0.29	0.28	0.28
	60-111	45-95	51-146	1206-2210	3.3-1.4	0.11 - 0.65	0.11-0.76	0.15-0.45	0.15-0.5
b) Pre-defined response with gradient limits of:									
-0.05/0.05	62	48	48	1235	5.2	0.42	0.53	0.34	0.38
	45-86	27-73	28-75	909-1714	8.8-2.6	0.19 - 0.73	0.24-0.87	0.2-0.48	0.25-0.56
-0.1/0.1	71	59	59	1410	3.1	0.35	0.42	0.31	0.34
	52-97	37-83	39-87	1047-1937	6-1	0.15 - 0.67	0.15-0.86	0.18-0.47	0.18-0.52
c) Pre-defined response with catch rate limits of:									
0.8/1.2	44	14	14	886	16.4	0.48	0.78	0.38	0.51
	39-60	12-31	12-32	780-1209	15.4-13.7	0.21 - 0.73	0.43-1.05	0.22-0.51	0.32-0.63
0.6/1.0	54	21	21	1073	14.5	0.38	0.69	0.32	0.45
	40-77	13-48	13-48	807-1530	15-8.2	0.19 - 0.59	0.36-1.02	0.21-0.46	0.29-0.6
d) Pre-defined response with gradient limits or catch rate limits of:									
-0.05/0.05 or 0.8/1.2	55	33	33	1091	9.8	0.43	0.65	0.35	0.44
	41-79	16-60	16-60	819-1587	13.4-4.7	0.2 - 0.7	0.32-0.96	0.21-0.49	0.27-0.57
-0.1/0.1 or 0.6/1.0	56	29	30	1114	11.8	0.36	0.64	0.32	0.44
	41-79	15-55	15-55	830-1577	13.9-5.5	0.16 - 0.63	0.31-0.98	0.19-0.46	0.27-0.57

7.4 Discussion

The management strategies used in these simulations were extremely simple and required only catch rate data. The assessment was based on a standardisation of catch rates and a simple regression of these catch rates against year. This information was then combined with a given set of decision rules based on reference limits to generate a management response. A range of these very minimalist management strategies were tested, each consisting of a specific combination of catch rate assessment, decision rule and management response.

Despite being very simple, the management strategies that use decision rules based on catch rates were clearly preferable to the harvest strategies only, *i.e.* the scenarios involving no real management control over catch. Depending on the particular arrangement of the chosen decision rule they were able to control and adjust catch levels if there were signs of undesired changes or trends in catch rates (or biomass). As long as the decision rules were sufficiently sensitive to the range of changes in observed catch rates, the reference limits in the decision rules could be adjusted to maintain catch rates and rebuild exploitable biomass if enough time was available. The harvest strategy evaluation, on the other hand, was only informative to some degree by indicating the maximum catch levels of 24-36 tonnes which on average should not be exceeded if the management objective is to maintain or rebuild catch rates (or mature biomass) at current levels. However, by its nature it is inflexible and did not highlight any advisable response when higher than desired catches were taken.

The interpretation of the performance of the different decision rules obviously depended on the management objective. The objectives used here only vaguely described the intention of the management strategy to stabilise or rebuild catch rates or mature biomass. To achieve these objectives, only reference limits were defined, while the target was implied as the range between the reference limits. Nevertheless, it was considered sufficient and appropriate for a qualitative comparison of the performance of the different management scenarios. Before decision rules can be applied to the real fishery, clear and unambiguous management objectives and reference limits need to be defined by the fishery managers and industry. This can be a lengthy process, particularly when separate fishery sectors with specific interests exploit the same fish stocks (*e.g.* Mapstone *et al.* 2004).

Particular focus was given here on the ability of a management strategy to reduce the deliberately high catch levels of 80 tonnes and reverse decreasing trends in catch rates and mature biomass. The high catch scenario was seen as realistic, because the potential development of overseas markets would without doubt lead to substantial increases in effort within the fishery. Presently (2005), the total catch is, to a large degree, market-limited, *i.e.* the processors dictate and limit the catch of the fishers to match what they can sell. Fishers would certainly increase their catch should the markets expand, and a management strategy is required that could cope with such a rapid expansion.

For the management objective to maintain or rebuild catch rates (and implicitly mature biomass) as soon as possible following any major decline, tight reference limits and thus a high sensitivity to change, combined with strong management responses appeared to achieve the highest catch rate and mature biomass at intermediate and high catch levels.

The management strategy (c) with relatively high catch rate limits (0.8) was considered to be the preferred strategy if rebuilding catch rates and biomass were the only objectives. However, this was achieved by reducing the initially high catch levels of 80 tonnes to very low levels (10-20 tonnes).

If additional fishery management objectives of maximising catches were also considered, the management strategy (d) with the tightest reference limits for the catch rate gradient (-0.05 and 0.05) and the relatively low catch rate limits (0.6) appeared best. Because maximising biomass and catch simultaneously is not possible as they contradict each other, a compromise would be necessary. This management strategy compensated an immediate sharp decline in catches leading to a short-term loss with relatively high final catch levels or a long-term gain. Excessively high catch levels of

80 tonnes were reduced to medium levels (20-30 tonnes) achieving an intermediate total catch and catch variation over the 20-year simulation period, while at the same time catch rates and mature biomass substantially rebuilt. This strategy appeared to be more responsive and attained the crude objectives better than the strategy (b) with gradient limits only because it had a stronger rebuilding capacity. It was also better than the strategy (c) with catch rate limits only because the faster catch reduction meant higher levels of catches overall and in the final years, while the rebuilding capacity was only slightly reduced. Nevertheless, what compromise between catch and biomass levels is acceptable by a specific management strategy remains to be negotiated in detail by fisheries management and industry in the management objectives.

In particular, the management strategy with the tightest reference limits for the catch rate gradient (-0.05 and 0.05) and the relatively low catch rate limits (0.6) was considered to be the preferred strategy. This management strategy compensated with an immediate sharp decline in catches leading to a short-term loss with relatively high final catch levels or a long-term gain. Excessively high catch levels of 80 tonnes were reduced to medium levels (20-30 tonnes), while at the same time catch rates and mature biomass substantially rebuilt. This strategy appeared to be more responsive and attained the crude objectives better than the strategy with gradient limits only because it had a stronger rebuilding capacity. It was also better than the strategy with catch rate limits only because the faster catch reduction meant higher levels of catches in the final years.

Decision rules using a management response proportional to the catch rate gradient were too insensitive to stop the decrease in catch rates and mature biomass when initial catch levels were high. These proportional catch corrections were generally lower than the reductions by the constant catch control of 0.8. Thus, rather than the proportional corrections themselves, the extent of the catch reductions could have been inappropriate, and a catch control system $1 + x$*gradient with $x > 1$ could result in higher catch rate and biomass levels. Alternatively, the range of acceptable gradients could have been made narrower. This would have increased the sensitivity of the decision rules to changes and lead to more frequent management responses.

Decision rules with proportional multiplication factors on current catch may be also more difficult to negotiate with and be understood by industry, and therefore be less acceptable, than a catch control with a constant factor. Both types of decision rules would need initial negotiations about the reference limits. Under a decision rule using a proportional response, the results from catch rate analysis are likely to be disputed whenever the gradient is close to or exceeds the reference limits. In a struggling fishery, this could occur whenever a stock assessment is performed and management response is possible. Decision rules based on defined reference limits, on the other hand, only need initial negotiations about the reference limits and the level of increase or decrease of the catch. Once these decisions are taken, the results from the catch rate analysis would seldom be disputed except when the gradient is in the vicinity of the reference limits. The consequences in case of a management change are also clear and foreseeable, and this can be important for fishers in the planning of their long-term future fishing activity.

We did not test management controls with different levels of catch reductions, *e.g.* a catch multiplication factor of 0.9 or 0.7 instead of 0.8 as applied here. A control with a weaker catch reduction of 0.9 would probably behave similarly to the tested proportional management control since the gradient triggering a management control was generally between -0.1 and -0.05. On the other hand, a stronger reduction of 0.7 is likely to stabilise or rebuild catch rates and biomass faster. The definition of the management control levels in combination with the levels of the reference limits will result in a balance between frequency and severity of management interference. What balance is preferred may vary depending on the type and size of the fishery and should also be part of the negotiations about the management objectives between management and industry.

A major disadvantage of a management strategy based on catch rates and related catch and effort information is the common lack of appropriate, accurate and reliable data (Chapter 3). It is generally very difficult to agree on and implement decision rules if the underlying information cannot be trusted. Thus, should catch rate data be used in a management strategy, it must be assured that the logbooks return unambiguous and clearly-structured information suitable for the fishery, that the logbook data is verified and that the spatial resolution of catch and effort returns are appropriate for the stock and fishery structure of the fished species (Lyle *et al.* 2005). Furthermore, as a fishery matures and data quality improves, the comparability of catch rate information over time will change and influence both catch rate trends (gradient) and the true relative catch rate. This is the case in Tasmania, where a major revision of the logbooks is being implemented to increased spatial resolution and the potential for verification of catches through association of catch and catch disposal records. There is also the issue of effort creep, for example through gear and technological improvements, that can influence catch rates but is only sometimes dealt with in the standardisation of catch rates.

Controlling the catch directly would require a change in the management system for scalefish fisheries in Tasmania, and many other states, where small scale and low value fisheries do not justify the cost of implementing quantitative output control systems. Where this is not possible, the catch could be controlled indirectly by input controls such as effort controls or gear regulations combined with relatively close monitoring. The direct effects of using effort responses have not been tested, but could be used in a similar way as the catch responses. The catch could also be limited indirectly by setting the lower and upper size limits alone or in combination with gear regulations in such a way that a specific proportion of the mature female population never becomes vulnerable to the fishery. A similar situation is already in place for purple wrasse, where the size limit is such that females (and males) have a considerable number of years to spawn before they become vulnerable to the fishery.

Management objectives other than the catch rate performance, such as economic success of the fishery, and biodiversity or ecosystem functioning have not been considered in this study and cannot therefore be commented upon. This investigation explored the potential of different management strategies to address simple management objectives relating to catch rates and implied biomass performance. It is now up to the fishery management and the fishing industry to define a suitable set of such objectives that could be applicable to the banded morwong fishery. Having developed an operating model the relative performance of alternative management strategies can be explored once such objectives have been defined. But in a first step, we have shown here that it is possible to meet some fishery objectives with relative simple sets of decision rules and management controls. Based on these findings, we recommend that the process of defining objectives and management strategies should be initiated to replace the current non-specific processes with tools similar to those developed here.

CHAPTER 8: GENERAL DISCUSSION

8.1 Introduction

Fisheries that target temperate reef fish species in Australia tend to be small-scale and can be classified as data-poor. Low value, combined with the implications of a high level of spatial structuring in the populations, pose considerable difficulties for stock assessment. Specifically, the quantity of data necessary to conduct robust assessments based on, for example age- and spatially-structured models, cannot typically be justified financially.

A need was identified to explore suitable and meaningful stock assessment methods and monitoring strategies for such species. Banded morwong in Tasmania was selected as an example species since there was an unusually large amount of information available about the fishery and biological characteristics, which had been gathered over much of the fishery history. Catch data since the start of the fishery in about 1990, catch rate data since 1995, and biological sampling data from the mid 1990s, allowed the construction of an age- and spatially-structured stock assessment model. This model was used as an operating model within a simplified management strategy evaluation framework to explore the relative effectiveness of different performance measures, assessment methods, and simple management strategies.

Banded morwong turned out to be a particularly complex species, exhibiting an unexpected degree of adaptiveness and flexibility. Limited movement by the adult fish meant that the stock is spatially structured, with populations differing, possibly between individual reefs, and thereby violating the dynamic-pool assumption generally held by assessment models (this assumption holds that spatial details do not significantly effect the stock dynamics). Related to this, there has been and continues to be a mis-match between the spatial scale of the populations and reporting of commercial catch and effort statistics that has the potential to mask the impacts of serial depletion of stocks. In addition, biological monitoring revealed an unanticipated but significant population response to the assumed stock depletion involving an acceleration of individual growth rates over time coupled with a decrease in the size and age of maturity. Given the life history characteristics of the species in Tasmania, especially its extreme longevity, these changes were surprisingly strong, increasing the complexity of the model and raising the question as to whether they were a density-dependent response or due to changing environmental conditions.

High model uncertainty proved to be the main limitation of the operating model (Chapter 5). This was largely unavoidable due to the spatial mismatch between assessment and stock dynamics, and the lack of data in some crucial population parameters; *e.g.* recruitment variability, regional distribution of biomass, and biomass distribution between fished onshore and unfished offshore areas. Because of the lack of information over the spatial scale of the fishery (a lack that most fisheries would also experience) the model ended by being badly over-parameterized. Despite model uncertainty, this study did provide some valuable insights into assessment and management strategies. Simple biological measures such as median age or size, age and sex ratios proved highly ambiguous and uncertain as indices of mature biomass or stock status (Chapter 6). This was the case when the measures were used singly or in combination. At best, such measures could be applied as qualitative indices and, if there was consistency in the direction of change between measures over time and/or space, indicate by weight of evidence that changes in the populations had occurred. Unfortunately, such information alone may not be informative about stock status and, importantly, whether harvest levels are sustainable. Life history characters can provide insights into the level of productivity to be expected from a species, but the simple biological characters such as sex ratio or median age performed poorly as fishery performance measures.

Management strategy evaluation, by contrast, revealed that simple decision rules based on measures derived from catch rate data could potentially act to control catches and achieve a preferred or targeted level of catch rates or mature biomass (Chapter 7). This was a promising result given that catch and effort data are often only the primary data source available for small-scale fisheries. However, more exploratory work would be necessary to determine how robust such catch rate management strategies would be to alternative possible realities (alternative operating model structures).

8.2 Assessment of Small-scale, Data-poor Fisheries

If the spatial structure of stocks is to be adequately represented in a stock assessment, large quantities of fishery and biological data would be required. However, what is possible in principle for small-scale fisheries is unlikely in practice since much less information would typically be available than in the banded morwong case. Therefore decisions need to be made regarding what sampling and assessment strategy has the potential to provide the best cost to benefit outcome. Results of this study provide some guidance on how to assess fish species under such situations. Three key steps form part of the process: determination of key life history traits; effective utilisation of fishers' knowledge; and development of management decision rules based on catch rate analysis.

Life history characteristics

Relevant life history information, including size and age composition, growth, and size and age at maturity, should, ideally, be determined. The structure and size of the fishery, the motivation of individual fishers and available financial resources will influence whether fishery-independent or dependent monitoring program would achieve more representative samples. In the event that such a program cannot be justified then meta-analysis of related species would represent a minimum first step to characterise the likely productivity and resilience of the target species.

While information from biological sampling may not be directly useful for quantitative assessment of the stock status (Chapter 6), it can reveal general trends over a range of measures (and time or space if enough samples are available) to indicate qualitative changes in the character of the stocks. Interpretation, however, may be difficult and ambiguous due to the low spatial representation of the stocks and noise in the data. For banded morwong, it was found that sample sizes necessary to reliably estimate biological measures such as sex ratio or age ratio effectively, exceeded those that were achievable within the constraints of the resources available to sample the fishery.

Information gained from biological sampling has the potential to assist in evaluating the susceptibility of a species to over-fishing, *i.e.* the likelihood that the fishing activity under current management arrangements severely reduces the mature biomass of females (or males if the population is sperm-limited due to the selective removal of males as may be the case for blue-throat wrasse). Biological information can also highlight possible alternative management options (*e.g.* size limits, spatial or seasonal closures, gear restrictions). In situations where there is little or no information to support a formal stock assessment, classification of fish species based on life-history characteristics can be used to provide a framework to consider appropriate management options (King and McFarlane 2003, Winemiller 2005). For example, using King and McFarlane's (2003) criteria, banded morwong could be classified as being intermediate between a periodic and an opportunistic strategist. Banded morwong is long-lived with a low degree of variability in overall abundance, where annual recruitment is only a fraction of the spawning biomass, but it is not slow growing and late maturing (all characteristics of a periodic strategist). At the same time it is fast-growing, early maturing and has strongly fluctuating recruitment, but not a short life span and high variability in overall abundance (all characteristics of a opportunistic strategist). Each strategy has its own set of recommended management options, *e.g.* maintaining the age composition by spatial refuge or size limits for the periodic strategist, and maintaining critical minimum spawning biomass by close monitoring and in-season management response for the opportunistic strategist. Thus, biological sampling cannot be used to provide direct fishery performance measures but can be used to provide warnings about potential limitations when exploiting different species. For example, it is because the size of maturity in purple wrasse is known to be well below the minimum legal length that there are few concerns about the fishery for that species (which contrasts with the situations with banded morwong).

Given relationships between life-history characteristics, such as age at first maturity and maximum age, and size at maturity and maximum size, some parameters can be estimated if necessary with a reasonable degree of certainty based on meta-analyses (Froese and Binohlan 2000; Punt et al. 2005). Interestingly, banded morwong appears to be an exception in that, while it conforms well based on size at maturity relative to maximum size, it is a clear outlier in the general relationship between age at first maturity and maximum age for many fish species (Froese and Binohlan 2000). This highlights the importance of at least a preliminary study of key biological characteristics. Of course, any other biological information of life-history characterisation, such as movement rates, fish behaviour, and habitat extent (whether based on formal mapping or fisher knowledge), to provide proxies for the distribution of biomass at different spatial scales would contribute to a better understanding of the species.

As demonstrated for banded morwong, however, it may not be enough to simply characterise life-history parameters once. This is particularly true where strong depletion is expected, or where changes in environmental conditions have been experienced. Repeated sampling may be necessary, although given the low value of these species and thus the restricted financial resources to research them, there may be need to be a trade-off between re-sampling previously assessed species versus collection of base-line information for previously un-assessed species.

Where it is not feasible to collect the data for the species itself, life-history characteristics from closely related species can be informative. This can be useful for identifying susceptibility (*e.g.* long-lived or slow maturing), but caution needs to be exercised since biological characters are not always preserved within families or even genera and so may provide misleading information. For example, the red morwong (*Cheilodactylus fuscus*), a closely related species to banded morwong, also has a long life-expectancy of around 40 years (Lowry 2003) but males and females of red morwong grow to similar sizes and thus would not exhibit the strong sex-based differences in selectivity and vulnerability to a fishery as are found in banded morwong.

Utilisation of fishers' knowledge

Fishers' knowledge represents a source of 'non-scientific' information that has the potential to be both very instructive about fishery dynamics and stock condition and very misleading. Basic information about industry processes such as fleet dynamics, changes in the dynamics of the fishery over time, and impact of external forces (*e.g.* seal interactions and markets) are a crucial part in the understanding and assessment of any fishery. Fishers' information can be gathered though informal interactions or surveyed using a more structured questionnaire approach (Chapter 3). Collection of such information represents a cheap and rapid alternative to more costly scientific surveys (Poizat and Baran 1997) and has often been overlooked or under-valued by researchers and managers when attempting to understand the dynamics of a fishery or target species.

A range of information can be collected, including the profiling details (*e.g.* fishing experience), key industry developments and impacts on catch and effort (market developments, changes in gear, fishing practices), perceptions about resource status and fish behaviour, and general management issues. In relation to banded morwong, fisher information highlighted the fact that the fishery only impacts part of the distributional range of the species, there is in effect a depth refuge, implying some degree of stock protection and potential for replenishment. Clearly any shift in fishing pressure into deeper water will have implications for the stocks and needs to be monitored.

Fisher knowledge can also assist in the interpretation of fishery data, including factors to be considered in catch rate standardisation. In addition, it may highlight disjunctions between what fishers know and what they report, *e.g.* misreporting of catches and effort. In the case of banded morwong and wrasse in Tasmania, it was revealed that catch and effort information have been subject to intentional and unintentional misreporting that has impacted on the utility of these data for stock assessments. Furthermore, the influence of external factors such as seal interactions and market influences on catch and effort became apparent.

Finally, it is important that fisher information is collected and synthesised within the context of the actual experience of the fisher, for instance in relation to the areas and time-frame in which the fisher has operated in the fishery.

Stock assessment and decision rules based on catch rates

In the absence of informative biological performance measures that can be used to develop of robust quantitative stock assessment models, we propose consideration of a management decision rule framework based on catch rate analysis. A prerequisite for this approach is the routine collection of reliable catch and effort data.

Catch rates need to be standardised with respect to key influencing factors, including those highlighted by fishers, although this will often be only a basic standardisation, *i.e.* not properly accounting for factors such as effort creep. The main limitation of an assessment based on catch rates is data quality. As in the case of banded morwong and indeed most scalefish species in Tasmania, data has a low reliability and does not match the spatial scale of the stock dynamics. Thus, priority needs to given to ensure the quality of the fishery data:

- Fishery-specific logbooks are desirable with clear and unambiguous reporting requirements (fishers' interpretation of reporting requirements can be 'tested' simply through formal or informal interactions).
- Catch returns need to be verified and while independent observer coverage may not be justifiable, catches could be matched against catch disposal records.
- Measures of effort need to be appropriate and should relate to the quantities of gear fished at each site either on a shot by shot or, at least, daily basis.
- Potential mismatches between the spatial scale of catch and effort reporting and stock dynamics need to be minimised. Where a mismatch is likely to occur or remain even at fine-scale reporting, such as for sedentary species, the assessment and management will need to account for this when interpreting catch rates. It is hoped that technological solutions may be found to counter this obvious lack of quality data.

Without explicit management objectives and targets it is difficult to determine whether a fishery is being managed successfully. This is especially the case in fisheries for highly spatially structured species where different levels of depletion could exist in different geographical regions. To be meaningful, any assessment should be part of a management system that has clear management objectives, well defined reference points or targets, and agreed upon decision rules for how to respond to significant changes in the stock or fishery.

8.3 Application to Banded Morwong in Victoria

The Victorian fishery for banded morwong is small compared to the Tasmanian fishery and in a developmental state with very few participants (Chapter 3). Catches are restricted to a daily limit of 50 fish, which is apparently achieved regularly. Detailed catch and effort information has only been available since 2000 and biological monitoring undertaken from four locations between 2001-2003, only two of which were repeat samples in each of the three years.

Although it was initially intended to incorporate the Victorian data into the development of an operating model, the paucity of commercial data (two operators and limited time series) and high level of inconsistency in available biological data precluded this. For instance, age composition proved highly variable between years (*e.g.* strong cohorts in one year were not apparent in subsequent years) (Fig. 8.1) resulting in inconsistent trends in biological measures such as sex ratios, age ratio or median age. Nevertheless, the Victorian situation can be used to test the implementation of the recommendations proposed in this study.

The biological data for Victoria illustrates three important points. Firstly, data inconsistency between years demonstrates limitations in the use of such data for stock assessments, even if the use were to be only qualitative. Secondly, there are substantial differences between some key biological characteristics of banded morwong from Tasmania and Victoria, highlighting the importance of sampling biological data, even from relatively close geographical locations. For example, growth of banded morwong in Victoria is slower initially and maximum sizes attained are substantially smaller than in Tasmania (Fig. 8.2 and compare Fig. 4.17). These differences have important implications for the stocks since Victoria has the same keyhole legal size limit as Tasmania, *viz*. 360-460 mm. Thirdly, basic biological characterisation can inform on potential vulnerability of a species to over-fishing.

As indicated in Fig. 8.1, the age at which females first become vulnerable to the fishery is substantially older (7+ years) than in Tasmania (4 years, Fig.4.14), implying greater opportunity for spawning to occur prior to recruitment to the fishery. Furthermore, the fact that many females probably do not attain the minimum legal size implies that the size limits are a more risk adverse management measure than the same limits used in Tasmania. While males, due to slower growth rates and smaller maximum sizes, are more vulnerable to and remain longer in the fishery than in Tasmania, it is evident that a significant proportion do eventually exceed the upper size limit. Thus, as long as there is sufficient ‘escapement’, the prospect of a sperm shortage would appear to be a low risk. However, based on experiences in Tasmania, with substantial changes in some key population parameters, there remains a need to periodically reassess growth characteristics and in particular size and age at maturity. Rather than improving productivity in the face of exploitation as in Tasmania, a faster growth rate in response to depletion would increase the proportion of females that are vulnerable to the fishery in Victoria.

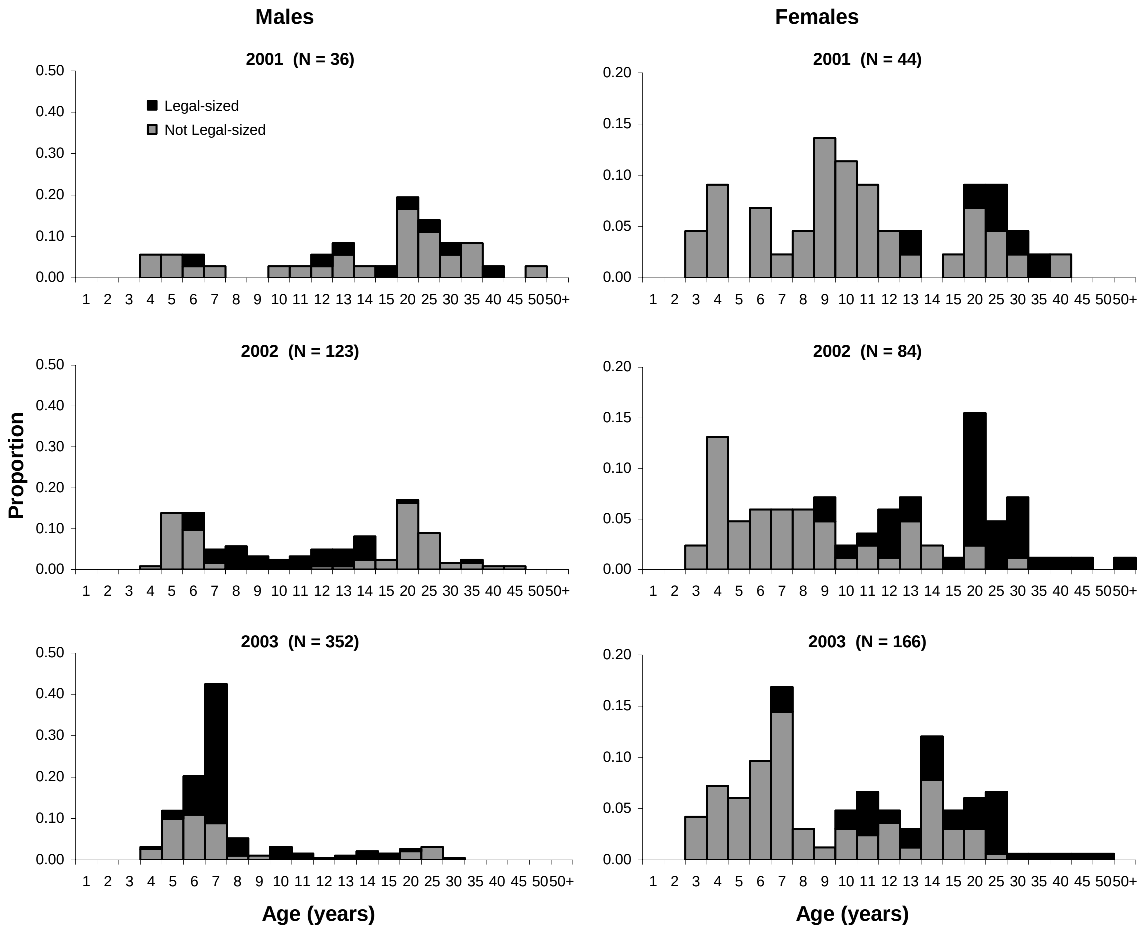

Fig. 8.1: Relative age composition of male and female banded morwong in combined catches of Cape Conron and Mallacoota in Victoria between 2001 and 2003. Black bars refer to legal-sized fish, grey bars to non-legal-sized (undersized and oversized) fish. Relative frequencies in 5-year classes for ages 16-50 years (values given denote upper limit) and pooled for over 50 years (50+). The oldest animal found was 53 years old.

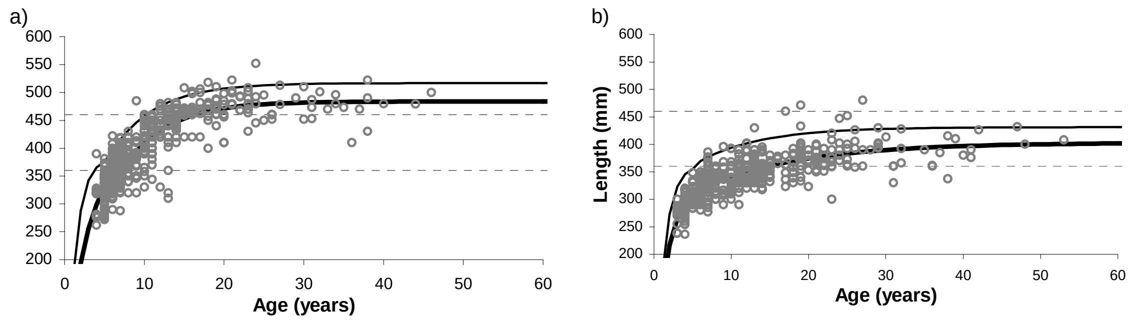

Fig. 8.2: Length at age for (a) male and (b) female banded morwong in Victoria (all years 2001-03 and sites pooled). Grey open circles indicate individual fish, the heavy black lines represent the 2-phase von Bertalanffy growth functions, and the thin black lines represent the 2-phase von Bertalanffy growth functions fitted to data from 2002 in Tasmania. The dotted lines are the size limits (same as in Tasmania).

In relation to the long-term management of the Victorian fishery, the fishery has the advantage of an effective limit on overall effort (with just two licensed operators), implicit controls on catch (based on daily trip limits and seasonal closures) and a specific catch and effort logbook, the reporting requirements for which are well understood by industry (although data are not subject to verification). While the time series of catch and effort data available is limited, catch rate data can be resolved to relatively small spatial-scales and thus may be informative about localised changes in abundance. Furthermore, the establishment of three marine protected areas within the area of the fishery (Cape Howe, Point Hicks and Beware Reef) and the prohibition of gillnetting in adjacent NSW coastal waters provides a formalized level of protection to the overall spawning stock of banded morwong, though resultant displacement of fishing effort into open areas may still result in localised over-fishing.

Using the MSE framework for Tasmanian banded morwong, there would appear to be considerable benefits to developing a series of simple decision rules based on measures derived from the catch rate data as part of any long-term management strategy for this fishery. The precise detail of any such decision rules would need to be negotiated with the stakeholders.

8.4 General Recommendations

A major and perhaps the first task of any management process should be to clearly define the *management objectives* for the fishery (*e.g.* Stephenson and Lane 1995; Mapstone *et al.* 2005). Such objectives should be negotiated between resource managers and stakeholders to maximise their acceptance and increase cooperation by involved stakeholders. In most data-poor fisheries no clear management objectives have been defined. It is possible that this lack of objectives stems from a recognition that most of the common types of management targets and objectives require a formal assessment if they are to be achieved; and this would be too expensive in such low value fisheries. What is required is a set of management objectives and associated targets that could be achieved with minimal data requirements and simple assessments.

Current management arrangements are typically inadequate for complex fisheries with highly spatially-structure populations and thus s*patially explicit management objectives* are required. In Tasmania, management objectives address ‘sustainability’ on a state-wide or possibly regional scale but do not address the question of how to respond to potentially conflicting indicators, *e.g.* when a regional assessment indicates heavy depletion but state-wide assessment does not. This demonstrates there is also a need for sets of decision rules that define what actions (management levers) to pursue should such conditions occur.

High uncertainty in stock assessment requires a *precautionary management* approach, with more conservative reference points than when the stock status is better known (Caddy and Mahon 1995). Applied to the catch rate decision rules used here, this principle would refer to the levels of catch rate (gradient and/or threshold) nominated.

Higher target or limit catch rates would be considered more precautionary than lower catch rate levels. If reference years are to be selected great care needs to be used in their selection. Of equal importance is the agreement of the industry involved who must accept the management actions that may be required in response to apparent changes in the condition of the stock.

Reef dwelling fish species are often found to exhibit spatial structuring of their populations. That is, potentially, the mature population on each reef can operate almost independently of other reefs; this is what makes the possibility of serial depletion such a threat for these species. It is the case that in data-poor situations the level of assessment and of possible spatial management is often at a coarser scale than the spatial structuring of fish species being exploited. This implies a degree of ignorance about the stock status that, in turn, implies the need for a precautionary management approach, but in practice no simple solution has been found for management to deal with this problem. As indicated in the Victorian situation, the establishment of a network of closed areas (*e.g*. marine protected areas) has the potential to provide some protection to spawning biomass but management based on closures alone cannot guarantee long-term sustainability of a fishery. For species that have relatively long planktonic larval phases, such as banded morwong and wrasse (Welsford 2003), benefits from larval dispersal into areas open to fishing may occur. In addition there may be replenishment of fished areas due to movement from adjacent formally closed or unfished areas, though mixing rates are likely to be low for typically site-attached species. Set against potential benefits is the impact of effort displacement from the closed areas. Without reductions in effort (or catch) commensurate with reduction in fishable area, the resultant concentration of fishing activity into the remaining open areas may lead to overfishing and in extreme cases eventual fishery collapse (Haddon *et al*. 2003).

Unfortunately, *qualitative assessments* generally do not lead to management change, even if there is a case for strong depletion of the stocks. Only a fishery collapse gives a signal strong enough to lead to substantial reductions in fishing mortality (Hildén 1993). This is the fishery's equivalent to closing the door after the horse has bolted. Such a lack of action stems from the assessment not being persuasive enough and nor clear how bad the situation is and, therefore, how stringent the management response should be. Individual biological measures are often too ambiguous to define stock status and there may be alternative explanations for the observed changes. Consistency in trends and 'weight of evidence' from an array of different biological and fishery measures (Caddy 1999) may strengthen the qualitative assessment of a fishery. But as in the case of banded morwong, despite substantial changes in many of the measures investigated, no significant management changes have been instigated. In part, this had been due to the overly optimistic approach generally seen in fisheries management, in addition the lack of defined management targets and decision rules means bad news can be ignored or explained away. If a situation can be explained by some positive development, *i.e.* strong recruitment pulses in recent years, then this is perceived as being more likely than a potential negative explanation, *i.e.* over-exploitation of the stocks. Without a quantitative assessment, combined with a set of strict decision rules, it is highly unlikely that an effective and consistent management response will ever be achieved.

In particular, for banded morwong, it is recommended that the management strategy evaluation initiated in this study be expanded to explore the responsiveness of catch rate information to a wider array of possible realities. For such simple decision rules to work they must be robust to a wide range of possible levels of production and confusion from the true fishery. If the decision rules, explored in Chapter 7, prove to be robust to a wide range of simulated realities, then it is recommended that:

- some target catch rates be selected;
- that some way of obtaining verified catch and effort information be arranged (a specialized log-book and catch verification);
- that a set of decision rules be agreed to which imply management measures that will lead to changes in effort or catch; and
- that an annual system of fishery review centred around the simple assessment and decision rules be developed and implemented in the management plan for this species.

BENEFITS

Fishers and resource managers will benefit from this study through improved understanding of the risks and challenges involved in managing for the long-term sustainable development of fisheries based on temperate reef fisheries. More broadly it is anticipated that there will be benefits for industry and resource managers in other jurisdictions in terms of management and assessment of other small-scale fisheries. Spatial issues in fisheries management have traditionally been met through zonation of fisheries or closures. However, for low value species spatial issues are often ignored. This project has demonstrated that even for such data poor fisheries it may be necessary to introduce or search for suitable management objectives that will work to ensure a sustainable and economically viable fishery. Before such management can be achieved there is a need to clearly define the objectives towards which it is desired to manage each fishery.

Temperate reef fisheries tend to be based on species with relatively complex life histories, which can make the species susceptible to serial depletion. This complexity must be recognized if management is to proceed. This implies that there may be hard decisions to be made with regards to catch levels and access if these fisheries are to be managed rather than simply monitored.

FURTHER DEVELOPMENT

Focus of this project was on banded morwong from Tasmania, primarily because sufficient biological information was available to construct detailed operating models. As the project developed it became increasingly apparent that the species was far more complex than anticipated, including exhibiting changes in population characteristics. This resulted in increased model complexity. High model uncertainty proved to be the main limitation of the operating models. This was largely unavoidable due to the lack of data in some crucial parameters; *e.g.* recruitment variability, regional distribution of biomass, and biomass distribution and movement rates between fished onshore and unfished offshore areas. Habitat extent has been applied as a proxy for relative biomass distribution in the model. Anecdotal information provided by fishers was used where habitat mapping information was not available. The SEAMAP project currently being undertaken by TAFI will, over time, provide detailed habitat maps for the entire coast-line of Tasmania and, as more information becomes available, the models can be updated. In relation to distribution and movement patterns over small spatial scales (along shore and on/off shore) the recently funded project "Spatial management of reef fisheries and ecosystems: Understanding the importance of movement" (FRDC 2004/002) is expected to provide some important insights that should result in refinement of the operating models.

There is a clear need to improve the spatial resolution and quality of the catch and effort data and to this end the Tasmanian Department of Primary Industries and Water is currently modifying the general fishing catch returns. The new logbook is scheduled for introduction in 2007 and will include provision for reporting retained and released catch of live-fish, more detailed spatial information, and links to catch disposal information. Overtime, these data should become increasing valuable for assessment.

Further development of the management strategy evaluation initiated in this study is recommended to explore the responsiveness of catch rate information to a wider array of possible realities. For such simple decision rules to work they must be robust to a wide range of possible levels of production and confusion from the true fishery.

Only limited data was available for banded morwong from Victoria and there was a high level of inconsistency in biological parameters. While there may be value in modifying the model to more closely match the Victorian situation, model uncertainty is likely to be even higher than for Tasmania, in part because of data inconsistency but also because of missing data. Despite this, it is clear that the characteristics of banded morwong growth in Victoria confer greater protection to females and current management restrictions imply a more risk adverse situation than in Tasmania. Similarly, although limited biological information was available for the wrasse species, conservative size limits that provide several years of protection to mature females (Smith *et al*. 2003; Ewing 2004) may be a sufficient management measure to provide adequate protection to spawning stocks. This fortunate combination of biological reality with the practice of the fishery may provide guidance as to a positive direction to move in the future; positive that is for stock sustainability but not necessarily for the success of all fisheries. It is possible that the biology of a species in a particular part of its distribution may preclude anything except a very small fishery from being imposed. This is the issue that is demanding that explicit objectives for small scale, data poor fisheries be developed.

PLANNED OUTCOMES

Planned outcomes included a re-evaluation of management options based on improved assessments for temperate reef fish stocks, provision of appropriate performance indicators, recommendation of key data requirements for effective on-going monitoring, and development of a cost-effective approach for the on-going assessment of the banded morwong and wrasse fisheries.

The main outcome of this project has been the development of a modelling framework to test the value of different performance measures for the assessment and management of spatially-structured populations. In data-poor situations basic biological characterisation, while not necessarily informative about stock status, is important in evaluating the susceptibility of a species to over-fishing. Through the application of a management strategy evaluation we have revealed that simple decision rules based on measures derived from catch rate analyses can potentially achieve a preferred or targeted level of mature biomass. In this regard, priority needs to be given to achieving a high level of commercial data quality, including appropriate spatial reporting, and the development of clearly defined management objectives and decision rules.

This study has highlighted the difficulties of managing and assessing spatially structured fish stocks, especially those low value fisheries that are typically data poor. Of critical importance with such fisheries is the development of explicit and detailed management objectives towards which each fishery can be managed. The generic objectives common to many fishery management plans are inadequate when faced with the problems inherent in spatially complex fish stocks. The recognition of this need for specific objectives is the outcome, but the objectives themselves should be generated by the fishery managers in consultation with the fishers and other stakeholders.

In relation to Tasmania, a significant and direct outcome of this project has been a review of the management of the banded morwong fishery, with a Departmental paper recently tabled to the Scalefish Fishery Advisory Committee (August 2006) proposing the introduction of output controls (individual quotas) coupled with spatially-explicit performance indicators. This paper resulted from an industry-management forum held early in August 2006 at which the implications of the stock assessment and catch projections derived from this project were considered. Progress is also being made to develop more explicit performance measures for other key scalefish species in Tasmania and it is hoped that these will be presented as part of the 2006 Scalefish Fishery assessment report.

This project has also raised the awareness by resource managers and industry of the importance of meaningful (and accurate) spatial information relating to catch and effort. Accordingly, the Tasmanian general fishing logbook is being revised to provide for greater spatial resolution in catch and effort reporting (particularly in relation to inshore reef areas), as well as catch verification via catch disposal declaration. The revised logbook version is scheduled for introduction during 2007.

CONCLUSION

Fisheries that target temperate reef fish species in Australia tend to be small-scale and data-poor. Low value, combined with the implications of a high level of spatial structuring in the populations, pose considerable difficulties for stock assessment. Specifically, the quantity of data necessary to conduct robust assessments based on, for example age- and spatially-structured models, cannot typically be justified financially.

The specific objectives of this study were to develop appropriate and meaningful performance indicators, determine minimum information requirements for stock assessment and develop model frameworks for testing performance indicators for small-scale fisheries based on sedentary, spatially-structured populations. Banded morwong in Tasmania was selected as an example species since there was a considerable amount of information available about the fishery and biological characteristics. This allowed the construction of age- and spatially-structured stock assessment models which were used as operating models within a simplified management strategy evaluation framework to explore the relative effectiveness of different performance measures, assessment methods, and simple management strategies.

Banded morwong turned out to be a particularly complex species, exhibiting a unexpected degree of adaptiveness and flexibility. Limited movement by the adult fish meant that the stock is spatially structured, with populations differing, possibly between individual reefs, and thereby violating the dynamic-pool assumption generally held by assessment models (this assumption holds that spatial details do not significantly effect the stock dynamics). Related to this, there has been and continues to be a mis-match between the spatial scale of the populations and reporting of commercial catch and effort statistics that has the potential to mask the impacts of serial depletion of stocks. In addition, biological monitoring revealed an unanticipated but significant population response to the assumed stock depletion involving an acceleration of individual growth rates over time coupled with a decrease in the size and age of maturity. Given the life history characteristics of the species in Tasmania, especially its extreme longevity, these changes were surprisingly strong, increasing the complexity of the model and raising the question as to whether they were a density-dependent response or due to changing environmental conditions.

High model uncertainty proved to be the main limitation of the operating models. This was largely unavoidable due to the spatial mismatch between assessment and stock dynamics, and the lack of data in some crucial population parameters; *e.g.* recruitment variability, regional distribution of biomass, and biomass distribution between fished onshore and unfished offshore areas. Because of the lack of information over the spatial scale of the fishery (a lack that most fisheries would also experience) the model ended by being badly over-parameterized. Despite model uncertainty, this study did provide some valuable insights into assessment and management strategies. Simple biological measures such as median age or size, age and sex ratios performed poorly as fishery performance measures and were found to be non-discriminatory as indices of mature biomass or stock status (Objectives 1 and 2). This was the case when the measures were used singly or in combination. At best, such measures could be applied as qualitative indices and, if there was consistency in the direction of change between measures over time and/or space, indicate by weight of evidence that changes in the populations had occurred. Unfortunately, such information alone may not be informative about stock status and, importantly, whether harvest levels are sustainable. Life history characters can provide insights into the level of productivity to be expected from a species, but the simple biological characters such as sex ratio or median age performed poorly as fishery performance measures.

Management strategy evaluation, by contrast, revealed that simple decision rules based on measures derived from catch rate data could potentially act to control catches and achieve a preferred or targeted level of catch rates or mature biomass (Objective 3). This was a promising result given that catch and effort data are often only the primary data source available for small-scale fisheries. However, more exploratory work would be necessary to determine how robust such catch rate management strategies would be to alternative possible realities (alternative operating model structures).

REFERENCES

Annala, J.H. (1993) Fishery assessment approaches in New Zealand's ITQ system. pp. 791-804 .*In:* G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke and T.J. Quinn II (eds.) *Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations.* Alaska Sea Grant College Program Report No. **93-02**.

Annala, J.H., Sullivan, K.J., O'Brien, C.J. and N.W.M. Smith (2001) Report from the Fishery Assessment Plenary, May 2001: stock assessments and yield estimates. New Zealand Ministry of Fisheries. 515p.

Anon. (1982) Fisheries act 1982. Parliament of South Australia.

Anon (1998) Scalefish fishery: policy document. Tasmanian Department of Primary Industry and Fishery. 91p.

Anon. (2004a) Fisheries Victoria commercial fish production bulletin 2004. Primary Industries Research Victoria, Queenscliff, Victoria. 33p.

Anon. (2004b) Victorian commercial wrasse fishery discussion paper - future management arrangements. Department of Primary Industries, Victoria. Fisheries Management Paper No. **17.** 12p.

Anon. (2004c) Fishery Management Strategies Performance Report 2004. NSW Department of Primary Fisheries. 20p.

Baranov, F.I. (1918) On the question of the biological basis of fisheries. *Nauchnye Issledovaniya Ikhtiologicheskii Instituta Izvestiya* **1**: 81-128. (Translated from Russian by W.E. Ricker).

Beddington, J.R. and J.G. Cooke (1983) The potential yield of fishstocks. *FAO Fisheries Technical Papers No.* **T242**. 47p.

Beverton, R.J.H., and S.J. Holt (1957) On the dynamics of exploited fish populations. U.K. Ministry Agriculture, Fisheries and Food. *Fish. Invest.* (Ser. 2) No. **19**. 533p.

Brett, J.R. (1979) Environmental factors and growth. pp. 599-675. *In:* W.S. Hoar, D.J. Randall and J.R. Brett (eds.) *Fish physiology, Vol. 8*. Academic Press, New York.

Bruce, B., Bradford, R., Griffin, D., Gardner, C. and J. Young (2000) A synthesis of existing data on larval rock lobster distribution in southern Australia. *FRDC Final Report, Project No.* **96/107**. 57p.

Burnham, K.P, and D.R. Anderson (1998) *Model selection and inference. A practical information-theoretic approach*. Springer Verlag, New York. 353 p.

Butterworth, D.S., Cochrane, K.L. and J.A.A. De Oliveira (1997) Management procedures: A better way to manage fisheries? The South African experience. pp. 83-90. *In*: E.K. Pikitch, D.D. Huppert and M.P. Sissenwine (eds.) *Global trends: fisheries management*. Bethesda: American Fisheries Society.

Buxton, C.D. (1993) Life-history changes in exploited reef fishes on the east coast of *S. Afr. Environ. Biol. Fishes* **36**: 47-63.

Caddy, J.F. (1998) A short review of precautionary reference points and some proposals or their use in data-poor situations. *FAO Fish. Tech. Pap*. **379**. 30p.

Caddy, J.F. (1999) Deciding on precautionary management measures for a stock based on a suite of limit reference points (LRPs) as a basis for a multi-LRP harvest law. *NAFO Sci. Counc. Stud*. **32**: 55-68.

Caddy, J.F. and K.L. Cochrane (2001) A review of fisheries management past and present and some future perspectives for the third millennium. *Ocean Coast. Manage.* **44**: 653-682.

Caddy, J.F. and J. Csirke (1983) Approximations to sustainable yield for exploited and unexploited stocks. *Oceanogr. Trop.* **18**: 3-15.

Caddy, J.F. and R. Mahon (1995) Reference points for fishery management. *FAO Fish. Tech. Pap*. **347**. 83p.

Caddy, J.F. and R. McGarvey (1996) Targets of limits for the management of fisheries? *N. Am. J. Fish. Manage.* **16**: 479-487.

Caton, A. and K. McLoughlin (2004) Status of fish stocks managed by the Australian Government. Australian Government Department of Agriculture, Fisheries and Forestry, Bureau of Rural Sciences.

Clark, W.G. (1991) Groundfish exploitation rates based on life history parameters. *Can. J. Fish. Aquat. Sci*. **48**: 734-750.

Clark, W.G. (1993) The effect of recruitment variability on the choice of a target level of spawning biomass per recruit. pp. 233-246. *In:* G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke and T.J. Quinn II (eds.) *International symposium on management strategies for exploited fish populations*. Alaska Sea Grant College Program Report No. **93-02**.

Clark, W.G. (2002) $F_{35\%}$ revisited ten years later. *N. Am. J. Fish. Manage*. **22**: 251-257.

Cochrane, K.L., Butterworth, D.S., De Oliveira, J.A.A. and B.A. Roel (1998) Management procedures in a fishery based on highly variable stocks and with conflicting objectives: Experiences in the South African pelagic fishery. *Rev. Fish Biol. Fisheries* **8**: 177-214.

Conover, D.O. and S.B. Munch (2002) Sustaining fisheries yields over evolutionary time scales. *Science* **297**: 94-96.

Coutin, P. (2000) Snapper 1998. Marine and Freshwater Resources Institute, Queenscliff, Victoria. *Fisheries Victoria Assessment Report No.* **19**.

De la Mare, W.K. (1996) Some recent developments in the management of marine living resources. pp. 599-616. *In:* R.B. Floyd (ed) *Frontiers of Population Ecology*. CSIRO Publishing, Melbourne, Australia.

Deriso, R.B. (1982) Relationship of fishing mortality to natural mortality and growth at the level of maximum sustainable yield. *Can. J. Fish. Aquat. Sci.* **39**: 1054-1058.

Deriso, R.B., Quinn, T.J. and P.R. Neal (1985) Catch-age analysis with auxiliary information. *Can. J. Fish. Aquat. Sci*. **42**: 815-824.

Derzhavin, A.N. (1922) See Ricker 1971.

Die, D.J. and J.F. Caddy (1997) Sustainable yield indicators from biomass: are there appropriate reference points for use in tropical fisheries? *Fish. Res.* **32**: 69-79.

Doubleday, W.G. (1976) A least-squares approach to analysing catch at age data. *Res. Bull. Internat. Comm. Northwest Atl. Fish.*, **12**: 69-81.

Dunlop, E.S., Shuter, B.J. and M.S. Ridgway (2005) Isolating the influence of growth rate on maturation patterns in the smallmouth bass (*Micropterus dolomieu*). *Can. J. Fish. Aquat. Sci.* **62**: 844-853.

Environment and Natural Resources Committee (2002) Inquiry into Fisheries Management - Second Report. Parliament of Victoria.

Ewing, G.P. (2004) Spatial and temporal variation in growth and age composition of the temperate wrasse *Notolobrus fucicola* in Tasmania waters. MSc thesis, University of Tasmania, 117p.

FAO (1993) Reference points for fishery management: their potential application to straddling and highly migratory resources. *FAO Fisheries Circular No.* **864**. 52pp.

FAO (1995) Precautionary approach to fisheries. Part 1: Guidelines on the precautionary approach to capture fisheries and species introductions. *FAO Fisheries Technical Papers, Rome, FAO, No.* **350/1**. 52p.

FAO (1996) Precautionary approach to fisheries. Part 2: Scientific papers. *FAO Fisheries Technical Papers, Rome, FAO, No.* **350/2**. 210p.

FAO (1999) Indicators for sustainable development of marine capture fisheries. *FAO Technical Guidelines for Responsible Fisheries No.* **8**. 68p.

Fournier, D.A. and C.P. Archibald (1982) A general theory for analyzing catch at age data. *Can. J. Fish. Aquat. Sci*. **39**: 1195-1207.

Francis, R.I.C.C. (1992) Use of risk analysis to assess fishery management strategies: A case study using orange roughy (*Hoplostethus atlanticus*) on the Chatham Rise, New Zealand. *Can. J. Fish. Aquat. Sci*. **49:** 922-930.

Francis, R.I.C.C. (1993) Monte Carlo evaluation of risk for biological reference points used in New Zealand assessments. pp.221-230. *In:* S.J. Smith, J.J. Hunt and D. Rivard (eds.) *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Pub. Fish. Aquat. Sci.* **120**.

Francis, R.I.C.C. and R. Shotton (1997) "Risk" in fisheries management: a review. *Can. J. Fish. Aquat. Sci.* **54**: 1699-1715.

Froese, R. and C. Binohlan (2000) Empirical relationships to estimate asymptotic length, length at first maturity and length at maximum yield per recruit in fishes, with a simple method to evaluate length frequency data. *J. Fish Biol.* **56**: 758-773.

Froese, R. and D. Pauly (2005) FishBase. *World Wide Web electronic publication on* http://www.fishbase.org. (Version 06/2005).

Gabriel, W.L. and P.M. Mace (1999) A review of biological reference points in the context of the precautionary approach. pp 34-45. *In:* V.R. Restrepo (ed.)

Providing Scientific Advice to Implement the Precautionary Approach Under the Magnuson-Stevens Fishery Conservation and Management Act. Key Largo, Florida, *NOAA Tech. Memo.* **NMFS-F/SPO-40**.

Gabriel, W.L., Sissenwine, M.P. and W.J. Overholtz (1989) Analysis of spawning stock biomass per recruit: an example for Georges Bank haddock. *N. Am. J. Fish. Manage.* **9**: 383-391.

Garcia, S.M. and K.L. Cochrane (2005) Ecosystem approach to fisheries: a review of implementation guidelines. *ICES J. Mar. Sci.* **62**: 311-318.

Garcia, S.M., Sparre, P. and J. Csirke (1989) Estimating surplus production and maximum sustainable yield from biomass data when catch and effort time series are not available. *Fish. Res.* **8**: 13-23.

Garrod, D.J. (1967) Population dynamics of the Arcto-Norwegian cod. *J. Fish. Res. Board Can.* **24**: 145-190. (Appendix contains Gulland 1965).

Getz, W.M., Francis, R.C. and G.L. Swartzman (1987) On managing variable marine fisheries. *Can. J. Fish. Aquatc. Sci.* **44**: 1370-1375.

Godø, O.R. and T. Haug (1999) Growth rate and sexual maturity in cod (*Gadus morhua*) and Atlantic halibut (*Hippoglossus hippoglossus*). *J. Northwest Atl. Fish. Sci.* **25**: 115-123.

Gomon, M.F, Glover, J.C.M. and R.H. Kuiter (1994) *The fishes of Australia's south coast*. State Printer, Adelaide, Australia. 992 pp.

Goodyear, C.P. (1977) Assessing the impact of power plant mortality on the compensatory reserve of fish populations. pp. 186-195. *In:* W. Van Winkle (ed.) *Proceedings of the Conference on Assessing the Effects of Powerplant Induced Mortality on Fish Populations*. Pergamon Press, N.Y.

Goodyear, C.P. (1980) Compensation in fish populations. pp. 253-280. *In:* C.H. Hocutt and J.R. Stauffer Jr. (eds.) *Biological Monitoring of Fish*. Lexington Books, D.C. Heath and Co., Lexington, MA.

Goodyear, C.P. (1989) Spawning stock biomass per recruit: the biological basis for a fisheries management tool. *ICCAT Working Document* **SCRS/89/82**. 10p.

Goodyear, C.P. (1993) Spawning stock biomass per recruit in fisheries management: foundation and current use. pp. 67-81. *In:* S.J. Smith, J.J. Hunt and D. Rivard (eds.) *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Pub. Fish. Aquat. Sci.* **120**.

Griffiths, M.H., Attwood, C. and R. Thomson (1999) New management protocol for the South African linefishery. Marine and Coastal Management, Department of Environmental Affairs and Tourism, South Africa. 12p.

Grift, R.E., Rijnsdorp, A.D., Barot, S., Heino, M. and U. Dieckmann (2003) Fisheries-induced trends in reaction norms for maturation in North Sea plaice. *Mar. Ecol. Prog. Ser.* **257**: 247-257.

Gulland, J.A. (1965) Estimation of mortality rates. Annex to Arctic Fisheries Working Group Report (meeting in Hamburg, January 1965). International Council for the Exploration of the Sea, Document 3 (mimeo), Copenhagen. (Appendix in Garrod 1967. Cited in Megrey 1989).

Gulland, J.A. (1983) Fish *stock assessment. A manual of basic methods*, John Wiley & Sons, Chichister.

Gulland, J.A. and L.K. Boerema (1973) Scientific advice on catch levels. *Fish. Bull*. **71**: 325-335.

Haddon, M. (2001) *Modelling and Quantitative Methods in Fisheries*. CRCPress / Chapman & Hall. Boca Raton. 406p.

Haddon, M., Ziegler, P.E., Lyle, J. and P. Burch (in press) Using a spatially-structured model to assess the Tasmanian fishery for banded morwong (*Cheilodactylus spectabilis*). *In:* Assessment and management of new and developed fisheries in data-limited situations. 21st Wakefield Fisheries Symposium, Anchorage, Alaska 2003.

Haddon, M., Buxton, C., Gardner, C. and N. Barrett (2003) Modelling the effect of introducing MPAs in a commercial fishery: a rock lobster example. Pp 428-436. in *Aquatic Protected Areas.* (*eds*) Beumer, J.P., Grant, A.,, and D.C. Smith. Proceeding of the World Congress on Aquatic Protected Areas, Cairns, Australia, August 2002. p.689.

Harden-Jones, F.R. (1968) *Fish Migration*. Edward Arnold, London.

Harris, P.J. and J.C. McGovern (1997) Changes in the life history of red porgy, *Pagrus pagrus*, from the southeastern United States, 1972-1994. *Fish. Bull*. **95**: 732-747.

Hearn, W.S. and T. Polacheck (2003) Estimating long-term growth-rate changes of southern bluefin tuna (*Thunnus maccoyii*) from two periods of tag-return data. *Fish. Bull.* **101**: 58-74.

Heino, M., Dieckmann, U. and O.R. Godø (2002) Measuring probabilistic reaction norms for age and size at maturation. *Evolution* **56**: 669-678.

Helser, T.E. and F.P. Almeida (1997) Density-dependent growth and sexual maturity of silver hake in the north-west Atlantic. *J. Fish. Biol*. **51**: 607-623.

Hilborn, R., Parma, A. and M. Maunder (2002) Exploitation rate reference points for West Coast rockfish: Are they robust and are there better alternatives? *N. Am. J. Fish. Manage.* **22**: 365-375.

Hilborn, R. and R.M. Peterman (1996) The development of scientific advice with incomplete information in the context of the precautionary approach. pp. 77-101 in: *FAO Fisheries Technical Paper No*. **350**, Part 2, Precautionary approach to fisheries. Part 2: Scientific papers. Prepared for the Technical Consultation on the Precautionary Approach to Capture Fisheries (Including Species Introductions). Rome, FAO, 210 pp.

Hilborn, R. and C.J. Walters (1992) *Quantitative Fisheries Stock Assessment: Choice, Dynamics, and Uncertainty*. Chapman & Hall, London.

Hildén, M. (1993) Reference points for fisheries management: the ICES experience. pp. 59-65. *In:* S.J. Smith, J.J. Hunt and D. Rivard (eds.) *Risk evaluation and* *biological reference points for fisheries management*. *Can. Spec. Pub. Fish. Aquat. Sci.* **120**.

Hoenig, J.M. (1983) Empirical use of longevity data to estimate mortality rates. *Fish. Bull.* **82**: 898-903.

ICES (2002) Report on the study group on the further development of the precautionary approach to fishery management. ICES, Copenhagen, ICES CM 2002/ACME:10. 157pp.

Imsland, A.K. and Ó.D.B. Jónsdóttir (2002) Is there a genetic basis to growth in Atlantic cod? Fish and Fisheries **3**: 36-52.

Jones, G.P. (1988) Ecology of rocky reef fish of north-eastern New Zealand: a review. *NZ J. Mar. Freshwater Res.* **22**: 445-462.

Kalish, J.M. (2001) Use of the bomb radiocarbon chronometer to validate fish age. Fisheries Research and Development Cooperation, Canberra, Australia. *Final Report FRDC Project* **93/109**. 384p.

Kennelly, S. and T. McVea, (2003) Status of fisheries resources 2001/02. NSW Fisheries, Cronulla Fisheries Centre, Cronulla, NSW. 296p.

Kimura, D.K. (1980) Likelihood methods for the von Bertalanffy growth curve. *Fish. Bull.* **77**: 765-776.

Kimura, D.K. (1981) Standarized measures of relative abundance based on modelling log(c.p.u.e.), and their application to pacific ocean perch (*Sebastes alutus*). *J. Con. Int. Explor. Mer* **39**: 211-218.

Kimura, D.K. (1988) Analyzing relative abundance indices with log-linear models. *N. Am. J. of Fish. Manage*. **8**: 175-180.

King, J.R. and G.A. McFarlane (2003) Marine fish life history strategies: applications to fishery management. *Fish. Manage. Ecol.* **10**: 249-264.

Kock, K.-H. (2000) Understanding CCMLAR's approach to management, The Convention on the Conservation of Antarctic Marine Living Resources (CCMLAR), Hobart. 47p.

Lane, D.E. and R.L. Stephenson (1999) Fisheries-management science: a framework for the implementation of fisheries-management systems. *ICES J. Mar. Sci.* **56**: 1059-1066.

Larkin, P.A. (1977) An epitaph for the concept of maximum sustainable yield. *Trans. Am. Fish. Soc*. **106**: 1-11.

Law, R. (2000) Fishing, selection, and phenotypic evolution. *ICES J. Mar. Sci.* **57**: 659-668.

Leet, W.S., Dewees, C.M., Klingbeil, R. and E.J. Larson (2001) Living marine resources: a status report, California Department of Fish and Game, Resource Agency.

Leum, L.L. and J.H. Choat (1980) Density and distribution patterns of the temperate marine fish *Cheilodactylus spectabilis* (Cheilodactylidae) in a reef environment. *Mar. Biol.* **57**: 327-337.

Lleonart, J. (1999) Precautionary approach and Mediterranean fisheries. CIESM Workshop. Kerkennah, Tunisia. 13p.

Lorenzen, K. and K. Enberg (2002) Density-dependent growth as a key mechanism in the regulation of fish populations: evidence from among-population comparisons. *Proc. Biol. Sci. B* **269**: 49-54.

Lowry, M. (2003) Age and growth of *Cheilodactylus fuscus*, a temperate rocky reef fish. *New Zealand J. Mar. Freshwater Res.* **37**: 159-170.

Ludwig, D. and C.J. Walters (1989) A robust method for parameter estimation from catch and effort data. *Can J. Fish. Aquat. Sci.* **46**: 137-144.

Lyle, J.M. (2005) 2000/01 survey of recreational fishing in Tasmania. Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, *Tech. Rep. Ser. No.* **24**. 97p.

Lyle, J.M. and K. Hodgson (2001) Tasmanian scalefish fishery assessment - 2000. Tasmanian Aquaculture and Fisheries Institute *Tech. Rep. No.* **19**. 70 p.

Lyle, J.M. and A.R Jordan (1999) Tasmanian scalefish fishery assessment - 1998. Tasmanian Aquaculture and Fisheries Institute, *Fish. Assess. Rep*. 88p.

Lyle, J.M., Haddon, M., Ziegler, P.E. and S.R. Tracey (2005) Tasmanian scalefish fishery - 2004. Tasmanian Aquaculture and Fisheries Institute, *Fish. Assess. Rep*. 79p.

Lyle, J.M., Ziegler, P.E., Haddon, M., Tracey, S.R. and P. Burch (2004) Tasmanian scalefish fishery - 2003. Tasmanian Aquaculture and Fisheries Institute, *Fish. Assess. Rep*. 88p.

Mace, P.M. (1988) The relevance of MSY and other biological reference points to stock assessments in New Zealand. *New Zealand Fish. Assess. Res. Doc.* **88/30**. 41p.

Mace, P.M. (1994) Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. *Can. J. Fish. Aquat. Sci.* **51**: 110-122.

Mace, P.M, and M.P. Sissenwine (1993) How much spawning per recruit is enough? *Can. Spec. Publ. Fish. Aquat. Sci*. **120**: 101-118.

Mann, B.Q. (2000) South African Linefish Status Reports. Oceanographic Research Institute, Durban, South Africa. 262p.

Mapstone, B.D., Davies, C.R., Little, L.R., Punt, A.E., Smith, A.D.M., Pantus, F., Lou, D.C., Williams, A.J., Jones, A., Ayling, A.M., Russ, G.R. and A.D. McDonald (2004) The effects of line fishing on the Great Barrier Reef and evaluations of alternative potential management strategies. CRC Reef Research Centre, Townsville, Australia. *CRC Reef Research Centre Tech. Rep. No.* **52**. 205p.

McCormick, M.I. (1989a) Spatio-temporal patterns in the abundance and population structure of a large temperate reef fish. *Mar. Ecol. Prog. Ser*. **53**: 215-225.

McCormick, M.I. (1989b) Reproductive ecology of the temperate reef fish *Cheilodactylus spectabilis* (Pisces: Cheilodactylidae). *Mar. Ecol. Prog. Ser.* **55**: 113-120.

McCormick, M.I. (1998) Ontogeny of diet shifts by a microcarnivorous fish, *Cheilodactylus spectabilis*: relationship between feeding mechanics, microhabitat selection and growth. *Mar. Biol.* **132**: 9-20.

McCormick, M.I. and J.H. Choat (1987) Estimating total abundance of a large temperate reef fish using visual strip-transects. *Mar. Biol.* **96**: 469-478.

Megrey, B.A. (1989) Review and comparison of age-structured stock assessment models from theoretical and applied points of view. *Am. Fish. Soc. Symp.* **6**: 8-48.

Millar, R.B. (1992) Estimating the size-selectivity of fishing gear by conditioning on the total catch: The SELECT (Share Each LEngthclass's Catch Total) model. *J. Am. Stat. Soc.* **87**: 962-968.

Millar, R.B. and R.J. Fryer (1999) Estimating the size-selection curves of towed gears, traps, nets and hooks. *Rev. Fish Biol. Fish.* **9**: 89-116.

Millar, R.B. and R. Holst (1997) Estimation of gillnet and hook selectivity using log-linear models. *ICES J. Mar. Sci*. **54**: 471-477.

Morison, A.K. (2004) Input and output controls in fisheries management: a plea for more consistency in terminology. *Fish. Manage. Ecol.* **11**: 411-413.

Murphy, G.I. (1965) A solution o the catch equation. *J. Fish. Res. Board Can.* **22**: 191-202.

Murphy, R.J. and J.M. Lyle (1999) Impact of gillnet fishing on inshore temperate reef fishes, with particular reference to banded morwong. Tasmanian Aquaculture and Fisheries Institute, FRDC Final Rep. **95/145**. 136p. http://www.utas.edu.au/tafi/PDF_files/GillnetFRDCfinalreport.pdf

Newman, S.J. and D.McB. Williams (2001) Spatial and temporal variation in assemblages of Lutjanidae, Lethrinidae and associated fish species among mid-continental shelf reefs in the central Great Barrier Reef. *Mar. Freshwat. Res.* **52**: 843-851.

NMFS (1996) Magnuson-Stevens Fishery Conservation and Management Act. *NOAA Tech. Memo.* **NMFS-F/SPO-23**. 121p.

NMFS (2002) Annual report to congress on the status of U.S. fisheries - 2001. U.S. Department of Commerce, NOAA, National Marine Fisheries Service, Silver Spring, MD. 142p.

Oliver, P. (2002) Methods for assessing Mediterranean fisheries. Scientific Advisory Committee for the General Fisheries Commission for the Mediterranean (SAC-GFCM). 22p.

Olsen, E.M., Lilly, G.R., Heino, M., Morgan, M.J., Brattey, J. and U. Dieckmann (2005) Assessing changes in age and size at maturation in collapsing populations of Atlantic cod (*Gadus morhua*). *Can J. Fish. Aquat. Sci.* **62**: 811-823.

Orensanz, J.M.L. and G.S. Jamieson (1998) The assessment and management of spatially structured stocks: an overview of the North Pacific Symposium on Invertebrate Stock Assessment and Management. pp. 441-459. *In:* G.S. Jamieson and A. Campbell (eds.) *Proceedings of the North Pacific Symposium* *on Invertebrate Stock Assessment and Management*. *Spec. Publ. Fish. Aquat. Sci.* **125**.

Pacific Fishery Management Council (1998) Environmental assessment / regulatory impact review for Amendment 11 to the Pacific Coast groundfish fishery management plan. Pacific Fishery Management Council, *NOAA No.* **NA87FC0008**. 58p.

Paloheimo, J.E. (1958) A method of estimating natural and fishing mortalities. *J. Fish. Res. Board Can* **15**: 749-758.

Parma, A.M., Orensanz, J.M.L., Elías, I. and G. Jerez (2003) Diving for shellfish and data: incentives for the participation of fishers in the monitoring and management of artisanal fisheries around southern South America. pp. 8-29. *In*: S.J. Newman, D.J. Gaughan, G. Jackson, M.C. Mackie, B. Molony, J. St. John and P. Kailola (eds.) *Towards sustainability of data-limited multi-sector fisheries*. Department of Fisheries, Perth, Western Australia. *Fish. Occ. Publ.* **6**.

Penn, J.W., Fletcher, W.J. and F. Head (2003) Status of the fisheries report 2002/2003. Western Australian Department of Fisheries, Fisheries Research Division. 235p.

Penney, A.J., Griffiths, M.H. and C.G. Attwood (1997) Management and monitoring of the South African marine line fishery. *SANCOR Occasional Report No.* **3**. 87p.

Pitcher, T.J. and P.J.B. Hart (1982) *Fisheries Ecology*. Croom Helm Ltd. London. 414p.

Poizat, G. and Baran, E. (1997) Fishermen's knowledge as background information in tropical fish ecology: a quantitative comparison with fish sampling results. *Environ. Biol. Fishes* **50**: 435-449.

Polacheck, T., Hilborn, R. and A.E. Punt (1993) Fitting surplus production models: Comparing methods and measuring uncertainty. *Can. J. Fish. Aquat. Sci.* **50**: 2597-2607.

Pope, J.G. (1972) An investigation into the accuracy of virtual population analysis using cohort analysis. *Res. Bull. Intern. Comm. Northwest Atl. Fish*. **9**: 65-74.

Pope, J.G. and J.G. Shepherd (1982) A simple method for the consistent interpretation of catch-at-age data. *J. Con. Int. Explor. Mer* **40:** 176-184.

Punt, A.E. (1990) Is $B_1 = K$ an appropriate assumption when applying an observation error production-model estimator to catch-effort data? *S. Afr. J. Mar. Sci.* **9**: 249-259.

Punt, A.E. (1992) Selecting management methodologies for marine resources, with an illustration for southern African hake. *S. Afr. J. Mar. Sci.* **12**: 943–958.

Punt, A.E. (1995) The performance of a production-model management procedure. *Fish. Res.* **21**: 349-374.

Punt, A.E. (2003) The performance of a size-structured stock assessment method in the face of spatial heterogeneity in growth. *Fish. Res.* **65**: 391-409.

Punt, A.E., Cui, G.R. and A.D.M. Smith (2001) Defining robust harvesting strategies, performance indicators and monitoring strategies for the SEF. Fisheries

Research and Development Cooperation, Canberra, Australia. *Final Report FRDC Project* **98/102**. 170p.

Punt, A.E., Smith, A.D.M. and G.R. Cui (2001) Review of progress in the introduction of management strategy evaluation (MSE) approaches in Australia's South East Fishery. *Mar. Freshwater Res.* **52**: 719-726.

Punt, A.E., Smith, A.D.M. and G. Cui (2002). Evaluation of management tools for Australia's South East Fishery 1. Modelling the South East Fishery taking account of technical interactions. *Mar. Freshwat. Res.* **53**: 615-629.

Punt, A.E., Smith, D.C. and M.T. Koopman (2005) Using information for 'data-rich' species to inform assessments of 'data-poor' species through Bayesian stock assessment methods. Primary Industries Research Victoria, Queenscliff. Draft FRDC Report **2002/094**. 170p.

Quinn, T.J. II. and R.B. Deriso (1999) *Quantitative Fish Dynamics*. Oxford University Press, Oxford. 542p.

Ricker, W.E. (1971) Derzhavin's biostatistical method of population analysis. *J. Fish. Res. Board Can.* **28**: 1666-1672.

Ricker, W.E. (1975) Computation and interpretation of biological statistics of fish populations. *Bull. Fish. Res. Board Can*. 382p.

Rosenberg, A.A. and S. Brault (1993) Choosing a management strategy for stock rebuilding when control is uncertain. pp. 243-249. *In:* S.J. Smith, J.J. Hunt and D. Rivard (eds.) *Risk evaluation and biological reference points for fisheries management. Can. Spec. Pub. Fish. Aquat. Sci.* **120**.

Rosenberg, A.A. and V.R. Restrepo (1996) Precautionary management reference points and management strategies. FAO Fisheries Technical Papers, Part 2: Scientific Papers. No. **350/2**.

Royce, W.F. (1984) *Introduction to the Practice of Fishery Science*. Academic Press Inc., London. 428p.

Russell, E.S. (1931) Some theoretical considerations on the 'overfishing' problem. *J. Cons. Int. Expl. Mer* **6**: 3-20.

Schaefer, M.B. (1954) Some aspects of the dynamics of populations important to the management of the commercial marine fisheries. *Bull. Inter-Am. Trop. Tuna Comm.* **1**: 25-56.

Schaefer, M.B. (1957) A study of the dynamics of the fishery for yellowfin tuna in the Eastern Tropical Pacific Ocean. *Bull. Inter-Am. Trop. Tuna Comm*. **2**: 247-285.

Sainsbury, K., Smith, A.D.M. and H. Webb (2000) Current use and recommendations for future development of sustainability indicators to measure performance of Australian fisheries against ESD objectives. CSIRO Marine Research, Australia, & Fisheries Research and Development Corporation, Project No. **98/168**. 111p.

Shepherd, J.G. (1982) A versatile new stock-recruitment relationship for fisheries, and the construction of sustainable yield curves. *J. Cons. Int. Expl. Mer* **40**: 67-75.

Sissenwine, M.P. (1978) Is MSY an adequate foundation for optimum yield? *Fisheries* **3**: 22-24.

Sissenwine, M.P., Fogarty, M.J. and W.J. Overholtz (1988) Some fisheries management implications of recruitment variability. pp. 129-152. *In:* J.A. Gulland (ed.) *Fish Population Dynamics.* John Wiley & Sons.

Sissenwine, M.P. and J.G. Shepherd (1987) An alternative perspective on recruitment overfishing and biological reference points. *Can. J. Fish. Aquat. Sci.* **44**: 913-918.

Smith, A.D.M. (1994) Management strategy evaluation - the light on the hill. pp 249-253. *In:* D.A. Hancock (ed.) *Population dynamics for fisheries management*. Australian Society for Fish Biology, Perth, Western Australia,.

Smith, A.D.M., Sainsbury, K.J. and R.A. Stevens (1999). Implementing effective fisheries management systems - management strategy evaluation and the Australian partnership approach. *ICES J. Mar. Sci.* **56:** 967–979.

Smith, D.C., Montgomery, I., Sivakumaran, K.P., Krusic-Golub, K., Smith, K. and R. Hodge (2003) The fisheries biology of bluethroat wrasse (*Notolabrus tetricus*) in Victorian waters. Marine and Freshwater Institute, Victoria, Australia, and Fisheries Research and Development Corporation, Australia, Report No. **97/128**. 88p.

Sparre, P., Ursin, E. and S.C. Venema (1989) Introduction to tropical fish stock assessment. Part 1. Manual. *FAO Fish. Tech. Rep.* **306/1**. 337p.

Stearns, S.C. and J.C. Koella (1986) The evolution of phenotypic plasticity in life-history traits: predictions of reaction norms for age and size at maturity. *Evolution* **40**: 893-913.

Stephenson, R.L. and D.E. Lane (1995) Fisheries management science: a plea for conceptual change. *Can. J. Fish. Aquat. Sci.* **52**: 2051-2056.

Stirratt, H., Caruso, P., Lazar, N., Simpson, D. and F.W. Steimle (2002) Addendum III to the Fishery Management Plan for Tautog. Atlantic States Marine Fisheries Commission, *Fishery Management Report No.* **25c**.

Thresher, R.E. (1994) Climatic cycles may help explain fish recruitment in south east Australia. *Aust. Fish.* **53**: 20-22.

Troadec, J.-P. (1977) Fishing and assessment of stocks. *FAO Fisheries Circulations* **701**: 1-8.

Welsford D.C. (2003) Interpretation of otolith microstructure in the early life- history stages of two temperate reef wrasses (Labridae). *Mar Fresh Res* **54**:69-75

Williams, L.E. (2002) Queensland's fisheries resources: current condition and recent trends 1988-2000. Queensland Department of Primary Industries, Brisbane. 180p.

Winemiller K.O. (2005) Life history strategies, population regulation, and implications for fisheries management. *Can. J. Fish. Aquat. Sci.* **62**: 872-885.

Wolf, B. (1998) Update on juvenile banded morwong in Tasmania. *Fishing Today* **11(4)**: 30-31.

Yearsley, G.K., Last, P.R. and R.D. Ward (1999) *Australian Seafood Handbook. Domestic Species*. CSIRO Marine Research Hobart. 461p.

Zeroni, P. (2003) Fishery status reports 2003. Northern Territories Department of Business, Industry and Resource Development. *Fishery Report No.* **78**.

Ziegler, P.E., Lyle, J., Haddon, M. and P. Burch (in press) Developing assessments and performance indicators for a small-scale temperate reef fish fishery. *In:* Assessment and management of new and developed fisheries in data-limited situations. 21st Wakefield Fisheries Symposium, Anchorage, Alaska 2003.

APPENDIX 1: INTELLECTUAL PROPERTY

This is not applicable to this project.

APPENDIX 2: STAFF

Dr Jeremy Lyle, Tasmanian Aquaculture and Fisheries Institute, Marine Research Laboratories

Assoc. Prof. Malcolm Haddon, Tasmanian Aquaculture and Fisheries Institute, Marine Research Laboratories

Dr Philippe Ziegler, Tasmanian Aquaculture and Fisheries Institute, Marine Research Laboratories

Alexander Morison, Primary Industries Research Victoria, Queenscliff

Dr David Smith, Primary Industries Research Victoria, Queenscliff (currently CSIRO Marine and Atmospheric Research, Hobart)

APPENDIX 3: DESCRIPTION OF OPERATING MODEL

The population dynamic in the operating model is represented by an age-structured model, described in the following.

A3.1 Basic population dynamics

Numbers at age were described by the standard age-structured model equations modified to account for multiple populations and by an expression summarizing movement between those populations:

$$\hat{N}^{p,s}_{t,y+1} = \begin{cases} \pi_p R^s_{t_{\min},y+1} & t = t_{\min} \\ N^{p,s}_{t-1,y} e^{-\left(Z^{p,s}_{y,t}\right)} & t_{\min}+1 \le t \le t_{\max-1} \\ N^{p,s}_{t_{\max}-1,y} e^{-\left(Z^{p,s}_{y,t}\right)} + N^{p,s}_{t_{\max},y} e^{-\left(Z^{p,s}_{y,t}\right)} & t = t_{\max} \end{cases} \qquad \text{(A3.1)}$$

where

$\hat{N}^{p,s}_{t,y+1}$	is the predicted numbers of fish in population *p* of sex *s* and age *t*, in year *y*+1,
π_p	is the proportion of available recruits to be found in population *p*,
$R^s_{t_{\min},y+1}$	is the fitted recruitment of sex *s* of age t_{min},
t_{min}, t_{max}	are the minimum and maximum age group, with the latter referred to as the plus age-group because it combines ages $t_{\max}$ and all older ages that are not modelled explicitly,
$Z^{p,s}_{y,t}$	is the total mortality of fish in population *p* of sex *s* at age *t* in year *y*, with fishing mortality occurring only in inshore populations :

$$Z^{p,s}_{t,y} = \begin{cases} M + s^s_t \hat{F}^p_y & \text{if } p = \text{ onshore population} \\ M & \text{if } p = \text{ offshore population} \end{cases} \qquad \text{(A3.2)}$$

s^s_t	is the sex, *s*, and age, *t*, specific selectivity
M	is the instantaneous rate of natural mortality assumed constant through all ages,
$\hat{F}^p_y$	is the estimated fully selected instantaneous rate of fishing mortality in population *p* in year *y*.

Equation (A3.1) was combined with the equation for movement. Movement was assumed to occur separately between onshore and offshore populations and between populations adjacent to each other alongshore. Movement was generally restricted to mature fish and acted at the end of each year to generate the final numbers at age in each population.

Between onshore and offshore populations, movement was assumed to be a combination of mobility rate and the relative proportion of suitable habitat into which the animals can move:

$$\hat{N}^{s,p}_{t,y+1} = \left(1 - m\pi_{p+1}\right)\mu_{t,y} N^{s,p}_{t,y+1} + m\pi_p \mu_{t,y} N^{s,p+1}_{t,y+1} \qquad \text{(A3.3)}$$

where

m	is the proportion of the mature population that becomes vagrant or mobile and becomes capable of shifting from each population to adjoining populations,
π_p	is the proportion of habitat/biomass in each population p,
$\mu_{t,y}$	is the proportion of age class t in year y that is sexually mature. The potential for variation among years was included because this was observed in Tasmanian populations of banded morwong.

Thus, the movement rate from population p into the neighbouring population p+1 can be represented as $m\pi_p$. Population p retains $1\text{-}m\pi_{p+1}$ of its total and gains $m\pi_p$ of population p+1. If the proportion of habitat is equal (*i.e.* $\pi_p = 0.5$) then the movement rate equals the mobility, however, if the proportional distribution of the population deviates from 50:50 then the movement rates will become asymmetric. Thus, this approach to describing movement includes both the propensity to move within a population and the area over which it can spread. A fish may begin to move and its probability of settling in one of the available areas is related to the relative area inhabited by the two populations.

In models with more than one region, consisting of a set of onshore and offshore populations, movement occurred alongshore between all neighbouring populations onshore and all neighbouring populations offshore. Outer regions were assumed to be in equilibrium with adjacent areas not represented in the model and thus, no net movement occurred along the outer borders. Movement was assumed to be a combination of mobility rate and a constant movement rate alongshore:

$$\hat{N}^{s,p}_{t,y+1} = \begin{cases} N^{s,p}_{t,y+1} - mk\mu_{t,y}N^{s,p}_{t,y+1} + mk\mu_{t,y}N^{s,p+2}_{t,y+1} & p = 1,2 \\ N^{s,p}_{t,y+1} - 2mk\mu_{t,y}N^{s,p}_{t,y+1} + mk\mu_{t,y}N^{s,p+2}_{t,y+1} & 3 \le p \le p_{\max} \\ \qquad + mk\mu_{t,y}N^{s,p-2}_{t,y+1} & \\ N^{s,p}_{t,y+1} - mk\mu_{t,y}N^{s,p}_{t,y+1} + mk\mu_{t,y}N^{s,p-2}_{t,y+1} & p = p_{\max}-1, p_{\max} \end{cases} \tag{A3.4}$$

where

k is the movement rate alongshore.

Maturity at age is described by a logistic model:

$$\mu_{t,y} = \frac{e^{(a_y + b_y l_t)}}{1 + e^{(a_y + b_y l_t)}} \tag{A3.5}$$

where

$\mu_{t,y}$ is the proportion of age class t in year y that is sexually mature

a_y, b_y are the maturity parameters in year y

Given maturity at age and knowledge of numbers in each population and sex, the mature or spawning biomass in year y after removing half of the annual natural and fishing mortality was determined using:

$$\hat{B}^{p}_{S,y} = \sum_{t=t_{\min}}^{t_{\max}} \sum_{p=1}^{NPops} \mu_{t,y} W^{s}_{t,y} \hat{N}^{p,s}_{t,y} e^{-Z^{p,s}_{t,y}/2} \tag{A3.6}$$

where

$\hat{B}^{p}_{S,y}$ is the predicted spawning biomass in population p in year y,

$\hat{N}^{p,s}_{t,y}$ is the number of fish in population p of age t in year y where the sex s is female,

$W^{s}_{t,y}$ is the weight at length for sex s at age t in year y,

t_{min}, t_{max} is the minimum and maximum age group (plus-group).

The predicted exploitable biomass was defined as the fishable biomass in onshore populations in year y after removing half of the annual natural and fishing mortality using:

$$\hat{B}_{E,y}^{p}=\sum_{s=1}^{2}\sum_{t=t_{\min}}^{t_{\max}} W_{t,y}^{s} s_{t,y}^{s} N_{t,y}^{p,s} e^{-Z_{t,y}^{p,s}/2} \qquad \text{(A3.7)}$$

where

$\hat{B}_{E,y}^{p}$	is the exploitable biomass in year y where population p is onshore
$s_{t,y}^{s}$	is the selectivity of age class t for sex s in year y.

A3.2 Growth

Growth is described in terms of length at age and weight at length by a 2-phased von Bertalanffy equation:

$$L_{t,y}^{s}=\begin{cases} L_{\infty 1,t}^{s}\left(1-e^{-K_{1,y}^{s}\left(t-t_{01,y}^{s}\right)}\right)+\varepsilon_{y}^{s} & t \le t_{trans}^{s} \\ L_{\infty 2,t}^{s}\left(1-e^{-K_{2,y}^{s}\left(t-t_{02,y}^{s}\right)}\right)+\varepsilon_{y}^{s} & t > t_{trans}^{s} \end{cases} \qquad \text{(A3.8)}$$

where

$L_{t,y}^{s}$	is the length at age t in year y for sex s
$L_{\infty,t}^{s}$	is the average maximum length for the species in year y for sex s
K_{y}^{s}	is the Brody growth coefficient in year y for sex s
$t_{0,y}^{s}$	is the age at a hypothetical length of zero in year y for sex s
t_{trans}^{s}	is the age transition between growth function 1 and growth function 2 for sex s
ε_{y}^{s}	is a normal random residual in year y for sex s.

The weight at length relationship is described by:

$$W_{t,y}^{s}=a_{s}(L_{t,y}^{s})^{b_{s}} \qquad \text{(A3.9)}$$

where

$W^{s}_{t,y}$	is the weight at length for sex *s* at age *t* in year *y*,
a_s, b_s	are the coefficients define the power relationship between length and weight.

A3.3 Selectivity

Mesh selectivity was estimated using the SELECT method (Share Each Length class's Catch Total; Millar 1992; Millar and Fryer 1999). It is described by the gamma selection function (rather than the normal selection function, indicating that many large fish are retained in the net mainly by wedging and tangling rather than by gilling):

$$r_l = \left(\frac{l}{\alpha k m}\right)^{\alpha} e^{\left(\alpha - \frac{l_t}{km}\right)} \tag{A3.10}$$

where r_t is the mesh selectivity and l_t the length of age class *t*, *m* is the mesh size of the nets used, and α and *k* are the selectivity parameters. Mesh selectivity was not thought to be influenced by sex (Murphy and Lyle 1999).

Sex-specific selectivity $s^{s}_{t,y}$ at age *t* in year *y* for the lower and upper size limits is estimated as the relative selectivity at age:

$$s^{s}_{t,y} = \frac{s^{s}_{t,y}}{\max(s^{s}_{t,y})} \tag{A3.11}$$

and approximated by summing up sex-specific selectivity at age and size $s^{s}_{t,l,y}$ by 5 mm intervals *l* between 0 and 600 mm:

$$s^{s}_{t,y} = \sum_{l=lower}^{upper} r_l s^{s}_{t,l,y} * \sum_{l=0}^{600} r_l s^{s}_{t,l,y} \tag{A3.12}$$

where:

$$s^{s}_{t,l,y} = \int_{l}^{l+1} N(L^{s}_{t,y}, \sigma^{s}_{t,y}) \Delta l \tag{A3.13}$$

$s^{s}_{t,y}$	is the selectivity of age class t for sex s in year y,
$s^{s}_{t,l,y}$	is the selectivity of age class *t* and 0.5 cm size interval *l* for sex *s* in year *y*, estimated from the von Bertalanffy growth function $L^{s}_{t,y}$ and its standard deviation $\sigma^{s}_{t,y}$.

A3.4 Fishing mortality, catch and catch rates

The fishing mortality rate for each age class in population p is defined in terms of the fully selected instantaneous fishing mortality rate $\hat{F}_y^p$ in year y combined with the selectivity for each age class t and sex s:

$$F_{y,t}^{p,s} = s_{t,y}^s \hat{F}_y^p \tag{A3.14}$$

where

$s_{t,y}^s$ is the relative selectivity of age class t for sex s in year y in population p,

$\hat{F}_y^p$ is the fitted fully-selected fishing mortality in population p and year y.

The predicted catch in onshore populations in each year y was defined as the sum of the predicted catch at age multiplied by the weight at age:

$$\hat{C}_y^p = \sum_{s=1}^{2} \sum_{t=t_{\min}}^{t_{\max}} W_{t,y}^s \frac{F_{t,y}^{p,s}}{F_{t,y}^{p,s} + M} N_{t,y}^{p,s} \left(1 - e^{-\left(M + F_{y,t}^{p,s}\right)}\right) \tag{A3.15}$$

where

$\hat{C}_y^p$ is the predicted catch in year y where population p is onshore. All fishing is assumed to occur instantaneously in the middle of the year.

The predicted catch rates were determined by:

$$\hat{I}_y^p = \hat{q}_p \hat{B}_{E,y}^p \tag{A3.16}$$

where

$\hat{I}_y^p$ is the predicted catch rates in population p and year y,

$\hat{q}_p$ is the predicted catchability in population p, determined by a closed form of the equation using observed catch rates assuming that the catchability coefficient is a constant and each $\hat{q}_p$ is only an estimate of the overall $\hat{q}_p$ with lognormal error:

$$\ln(\hat{q}_p) = \frac{\sum_{y=1996}^{2004} \ln\left(\frac{I_y^p}{B_{E,y}^p}\right)}{n} \tag{A3.17}$$

I_y^p is the observed catch rates in population p and year y,

n is the number of years with catch rates observations between 1996 and 2004.

APPENDIX 4: DESCRIPTION OF MSE MODEL

A4.1 Projection

Each simulation run consisted of a historical period from 1990-2004 and a projected period of 20 years from 2005-2024.

A4.2 Recruitment

To initialise the age structure for each simulation run at the start of the projection when a management strategy is applied, the populations within each region was projected from the pre-exploitation equilibrium in 1990 to the start of 2005 with random variation in recruitment and under subject of the reported catches from 1990-2004. Recruitment in each year was based on the geometric mean of the fitted recruitment parameters and the density-dependent standard deviation σ_R of recruitment:

$$R^{s}_{t_{\min},y} = GM(\overline{R}^{s}_{t_{\min},90-04})e^{\varepsilon_{R,90-04}} \qquad \varepsilon_{R,y} \sim N(0,\sigma^2_R) \qquad \text{(A4.1)}$$

where

$R^{s}_{t_{\min},y}$	is the geometric mean (GM) of the fitted recruitment by sex s of age t_{min},
σ^2_R	is the density-dependent parameter that determines the extend of annual variation in recruitment.

The standard deviation used in the projections was considered to be density-dependent and influenced by the population numbers in the previous year y-1. To estimate the relationship between population numbers and recruitment standard deviation, the 15 historical years were ranked by population numbers and split in four groups with the last group consisting of only 3 years. The geometric recruitment mean of each group was estimated and a regression calculated. This regression represented the density-dependent effect of population numbers on recruitment variability. The range of recruitment variability was limited to the estimated range from the historical period. This limitation capped very large recruitment events at low population size, effectively reducing the productivity of the stocks and resulting in more conservative rebuilding scenarios. On the other hand, recruitment was assumed to occur under any circumstances, *i.e.* even when there was virtually no standing biomass left in the model populations.

A4.3 Catch, catch rates and effort

With catch as the unit which was controlled by management, the total fishing-induced mortality or 'real catch' in projected years was calculated from the reported catch modified by the reporting seal or bycatch bias by:

$$\hat{C}_y^p = C_y^p L_y D_y e^{\varepsilon_{L,y}} e^{\varepsilon_{D,y}} \quad \varepsilon_{L,y} \sim N(0,\sigma_L^2), \quad \varepsilon_{D,y} \sim N(0,\sigma_D^2) \qquad \text{(A4.2)}$$

where

$\hat{C}_y^p$	is the predicted total catch in year y where population p is onshore. All fishing is assumed to occur instantaneously in the middle of the year,
C_y^p	is the observed reported catch in year y where population p is onshore,
L_y	is the logbook reporting bias,
σ_L^2	is the parameter that determines the extend of annual variation in logbook reporting rates,
D_y	is the discard and seal-induced mortality in year y where population p is onshore,
σ_D^2	is the parameter that determines the extend of annual variation in discarding and seal-induced mortality,

The predicted catch rates were again determined by Equation (A3.16), and total predicted effort estimated by:

$$\hat{E}_y^p = \frac{\hat{C}_y^p}{\hat{I}_y^p} \qquad \text{(A4.3)}$$

where

$\hat{E}_y^p$	is the total predicted fishing effort in population p in year y,
$\hat{I}_y^p$	is the predicted catch rates in population p and at the start of year y,

Reported effort is then determined from the total predicted effort by:

$$E_y^p = \frac{\hat{E}_y^p}{L_y e^{\varepsilon_{L,y}}} \quad \varepsilon_{L,y} \sim N(0,\sigma_L^2) \qquad \text{(A4.4)}$$

where

E_y^p	is the observed reported effort in year y where population p is onshore,
$\hat{E}_y^p$	is the predicted total effort in year y where population p is onshore,
L_y	is the logbook reporting bias,
σ_L^2	is the parameter that determines the extend of annual variation in logbook reporting rates.

Observed catch rates were determined by dividing reported catch by reported effort:

$$I_y^p = \frac{C_y^p}{E_y^p} \tag{A4.5}$$

Catch, catch rates and effort were calculated on a regional base, and summarised (catch and effort) or the catch-weighted geometric mean taken (catch rates) to estimate the results for all regions combined.

In simulations with reported effort as management control unit (not presented here), the predicted total effort, predicted catch rates, predicted and reported catches and observed catch rates were estimated in the reverse order.