Exploration of the effectiveness of alternative management responses to variable recruitment

E. A. Fisher, S. A. Hesp & N. G. Hall

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OBJECTIVES:

(1) Develop a generic management strategy evaluation framework capable of generating (through simulation) age composition data and results from catch curve analyses for species with a range of different life history characteristics (including different levels of recruitment variability), displaying these for decision-making in a weight-of-evidence framework (based on that currently being used in Western Australia), accepting the resulting decisions and simulating the consequences of those decisions.

(2) Explore and advise on the effectiveness of alternative scientific, management and communication approaches to the types of data generated for species with a range of life history characteristics (including different levels of recruitment variability) considered by the management strategy evaluation framework, based on the results of scenario testing and the use of Monte Carlo methods.

NON-TECHNICAL SUMMARY

OUTCOMES ACHIEVED TO DATE

A management strategy evaluation (MSE) program has been developed that is widely applicable to fish species for which assessments rely on mortality estimation using age composition data. The program combines the results of catch curve and per recruit analyses with subjective information for other risk factors to provide an overall risk assessment for the stock. Several computer workshops have been held to explore the effectiveness of the MSE program for conveying stock status information. Workshop participants were presented with a range of fishery scenarios and asked to decide how to manage fish stocks given their perceived states of those stocks and available management controls. Analyses of data produced during those workshops demonstrated that the program effectively communicates stock assessment information to people with limited background in fisheries science. The project has provided researchers a useful tool for informing the design of age-based monitoring programs. The effectiveness of sampling procedures and catch curve analyses for monitoring fish stocks has been shown to vary markedly, depending on the biology of fish species (including level of recruitment variability). Researchers and managers are provided information, based on our Monte Carlo simulations, about how various factors influence the effectiveness of different management controls. The program is now freely available at http://www.cffrfisheriesmodelling.net/mse.htm.

This study has produced a generic (management strategy evaluation, or MSE) computer program that can be used to assess the likely effectiveness of alternative management options for a fish stock. The program is applicable in situations where data of the type collected during a standard biological study of a fish stock are available. Such data include values for a range of parameters, such as those describing growth and the reproductive biology of a species, and age composition data (numbers at each age) determined from fish samples. The model is well suited to recreational scalefish fisheries and some small-scale commercial fisheries for scalefish.

Our MSE model consists of an “operating model” (for a single fish species in a single area) which simulates a range of processes within the fish population and keeps track of the numbers of fish of each sex, at each length and age over time. It also contains an “observation model” which generates age and length composition sample data (with error), an “assessment model” which produces information about the state of the fish stock based on estimates of mortality, and a decision-making model that simulates how changes to management are chosen.

The model can simulate data for fish species with differing biological characteristics, such as longevity and level of recruitment variability (that is, variation in numbers of fish born in different years that survive to become large enough to enter into the fishery). The relative impacts of several management controls commonly used for scalefish line fisheries (bag and boat limits, minimum legal lengths for retention, temporal and spatial closures, catch quotas and effort reductions) are simulated, taking into account post-release mortality and, for several controls, level of fisheries compliance.

The operating model estimates the effects of any existing management controls on the stock given a specified initial exploitation state. After a specified period of time (simulation projection period), it then assesses the effects of any changes to the initial management.

The assessment model produces a range of stock assessment information, including values from mortality and per recruit analyses. The program then combines this with additional subjective information about various potential risk factors to the stock, in what is sometimes referred to as a “weight-of-evidence” framework. This approach is based on the one currently being adopted by the Department of Fisheries, Western Australia, for managing data-limited scalefish fisheries.

A key focus of the project was to design an effective, user-friendly interface for the MSE program. A series of workshops involving undergraduate university students, and researchers and managers were held during the project. In these workshops, participants interacted with the software by “pulling the various management levers” according to their views as to which management actions were most appropriate, given their perceived state of the fish stock after assessing information provided by the program.

The management decisions made by the students who participated in the workshops were logged by the computer and then analysed. The results demonstrated that the students, who had limited background in fish stock assessment, made both predictable and logical decisions. As would be expected if they understood the information provided by the program, students made different decisions depending on the biology of the fish species in question, and the level to which a fish stock was experiencing fishing pressure. The results suggest that the program could be an effective tool for communicating stock assessment information to people without a strong background in fisheries science.

We used MSE to explore the effectiveness of different sampling strategies and assessment methods for two species with different biology, and assuming different levels of recruitment variability for each species. The sample size required to reliably estimate total mortality varied between species and type of analysis used. One commonly used type of analysis (linear catch curve analysis) was shown to underestimate total mortality for a short-lived species, regardless of sample size. Another type of analysis (relative abundance analysis), which takes variability in recruitment into account, overestimated total mortality for a long-lived species, unless the sample size was relatively large. Some recommendations are provided as to how the reliability of these types of analyses might be improved. Recruitment variability was shown to lead to less precise mortality estimates, which, in turn, is likely to impact on the precision of any stock assessment advice given to managers.

We also used the model to explore the effectiveness of different management controls for different fish species. Depending on a range of factors, some controls were more effective than others for reducing mortality. Post-release mortality was shown to be particularly important for influencing the effectiveness of several of the controls. Although, on average, recruitment variability is not likely to impact on the effectiveness of management, at any particular time, the consequences of applying some controls is likely to be more variable when recruitment variability is high.

The MSE software has been developed in Visual Basic.net, employing shell procedures to AD Model Builder for catch curve analysis procedures (for rapid optimisation). The program is freely available for download as an .exe file from http://www.cffrfisheriesmodelling.net/mse.htm, together with the various data input files required to run the program. No additional software (other than a standard Windows operating environment) is required for running the MSE program. Instructions on how to download and run the program are provided in Chapter 2.

Program development and improvement of the MSE model will continue for at least the next 2.5 years during a FRDC project that commenced in November 2010. During this new project, a spatial component will be built into the model to enable more robust assessment of overall mortality in populations of fish species which exhibit size-related offshore movements.

KEYWORDS: Management strategy evaluation, age-based monitoring, mortality estimates, scenario testing, Monte Carlo analysis, recruitment variability.

ACKNOWLEDGEMENTS

We wish to express our gratitude to Dr James Scandol for reviewing our initial project application. Dr Scandol provided us with many excellent ideas on developing effective tools for communicating stock assessment information and for using MSE to explore the effectiveness of alternative science and management options. We thank Drs Brent Wise and Brett Molony (Department of Fisheries, WA), Kane Moyle (RecFishWest) and Richard Stevens and Felicity Horn (WAFIC) for valuable feedback and discussion during the project. We appreciate the advice of Dr Fiona Valesini (Murdoch University) and Prof Bob Clarke (Plymouth Marine Laboratory, UK) on the use of multivariate statistics for analysing data produced during scenario testing student workshops. Thanks to the many others from the Department of Fisheries, RecFishWest, WAFIC and Murdoch University for kindly attending our computer workshops on MSE and providing us with feedback on the program. Further thanks to students at Murdoch University for allowing us to use the data they produced during scenario testing workshops. We thank the FRDC for giving us the opportunity to undertake this study.

CHAPTER 1: General introduction

BACKGROUND

The management strategy evaluation (MSE) approach

It is now widely understood that past failures to adequately account for the many uncertainties inherent in fisheries has been a major contributing factor to the often poor performance of fisheries management (*e.g.* Ludwig *et al.*, 1993; Hilborn, 1997). However, substantial progress has been achieved towards developing stock assessment approaches that account for such uncertainties (Butterworth *et al.*, 1997; Cooke, 1999; Peterman, 2004), made possible by rapid advances in computer technology (Schnute *et al.*, 2007). One approach, in particular, has received considerable attention among fisheries stock assessment scientists. Referred to as management strategy evaluation (MSE) or harvest strategy evaluation, this approach involves using computer simulation to assess the likely performance of alternative options for managing fisheries resources (Smith, 1994; Punt, 2008). MSE has been recognised as a valuable tool for helping managers and other fishery stakeholders reach agreement when formulating new management plans for fisheries (Smith, 1993; Schnute *et al.*, 2007).

As outlined by Smith *et al.* (1999), MSE involves

(1) clearly specifying the management objectives to be achieved,

(2) turning the specified objectives into quantitative performance indicators that can be readily measured,

(3) specifying alternative strategies for managing the fishery,

(4) evaluating the effectiveness of each management strategy using Monte Carlo simulation methods, and

(5) effectively communicating the results of the evaluation to decision-makers in a way that lays bare the trade-offs in performance of alternative strategies across the specified management objectives.

An important feature of the MSE approach is that, when assessing the effectiveness of alternative management strategies, MSE attempts to consider the whole management system, from monitoring strategies, through to stock assessment and implementation of management (Dichmont *et al.*, 2006). Furthermore, MSE emphasises the need to identify and model key uncertainties during the evaluation process and to determine how these might influence the robustness of alternative strategies for satisfying the management objectives (Smith, 1993; Smith *et al.*, 1999). To determine which management strategies are likely to be most successful, performance measures (analogous to reference points used in stock assessments) are commonly specified as quantifiable targets or limits (Sainsbury *et al.*, 2000).

The MSE framework

The main characteristic of the MSE simulation framework is that it distinguishes between the true state of a natural system and that perceived through monitoring and assessment (Kell *et al.*, 2005; Dichmont *et al.*, 2006). The true system is represented by an operating model, which simulates the dynamics of the fish stock and its fishery, as well as interactions between them (Butterworth & Punt, 1999; Rademeyer *et al.*, 2007). An operating model needs to be sufficiently complex to capture the key aspects of the simulated system and to reflect the underlying reality, but must also allow the consequences of contrasting hypotheses about the true dynamics of the system to be evaluated (Kell *et al.*, 2007). This is especially important in situations where relevant data may be limited or lacking, which is true for many recreational fisheries and minor commercial fisheries.

The various processes contributing to management of a fisheries resource are collectively referred to as “management strategies” (Sainsbury *et al.*, 2000; Dichmont *et al.*, 2006). In general, a management strategy consists of

(1) an observation model that simulates monitoring and collection of data from the fish stock and its associated fishery,

(2) an assessment model which analyses the data to assess the state of the fish stock relative to specified reference points, and

(3) a decision rule that is used to adjust management, given the perceived state of the resource based on the stock assessment information.

The latter element of the management strategy also commonly includes an implementation model that simulates the effects of chosen management actions on the resources and the fishery (Sainsbury *et al.*, 2000). An outline of the typical framework for MSE and its core components is illustrated in Fig. 1.1.

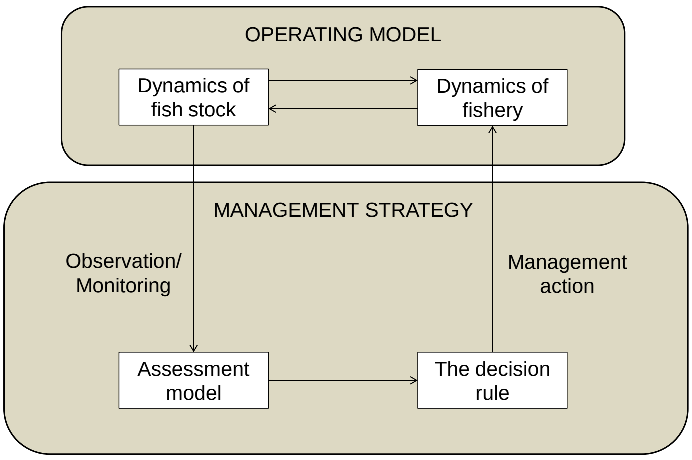


Fig. 1.1. Schematic overview of the MSE model framework and the different components of the operating model and the management strategy. Modified from Kell *et al.* (2005) and Dichmont *et al.* (2006).

MSE and dealing with uncertainty

MSE can be applied to address all major sources of uncertainty in fisheries, including process, observation, parameter, model and implementation error (Francis & Shotton, 1997). For example, process error resulting from variability in annual recruitment of fish species can be considered as deviations about an expected mean value estimated using some form of stock-recruitment relationship (Punt & Smith, 1999; Kraak *et al.*, 2008). Observation error can be accounted for in simulations by including various sources of uncertainty about how well sample data represent the real population (*e.g.* Polacheck *et al.*, 1999), or, as is more common, by describing measurement uncertainty associated with inaccurate and imprecise recording of data, such as for the ages of fish (Fulton *et al.*, 2005; Kell *et al.*, 2006).

Parameter uncertainty is often dealt with in MSE simulations by using Monte Carlo simulation methods to draw random values from probability distributions for different parameters or processes and evaluating the performance of management strategies for different combinations of parameter values (*e.g.* McAllister *et al.*, 1994; Kell *et al.*, 2006). Uncertainty in model structure can be assessed, for example, by comparing predictions generated using alternative operating models that have different underlying assumptions about dynamics of the fish stock (Butterworth & Punt, 1999; Dichmont *et al.*, 2006). Implementation error, which relates to the potential unknown consequences of management, *i.e.* because the effectiveness of management changes can be influenced by a large number of factors, including behavioural responses of fishers, has less often been explored using MSE (Butterworth & Punt, 1999). Examples of MSE evaluations that have evaluated the effects of implementation error are studies that have accounted for uncertainties associated with high-grading and discarding of fish (Punt *et al.*, 2005).

Past experience in the use of MSE

The MSE approach has been applied successfully to a number of fisheries worldwide (*e.g.* Punt *et al.*, 2002; Kell *et al.*, 2005). In Australia, for example, the Australian Fisheries Management Authority (AFMA) has used this approach when assessing fisheries for Southern Bluefin Tuna, eastern tuna and billfish, Eastern Gemfish, Orange Roughy, and sharks (Smith *et al.*, 1999). It has also been applied to managing other types of fisheries resources, such as prawns (*e.g.* Dichmont *et al.*, 2006) and abalone (Prince *et al.*, 2008). Furthermore, MSE has been increasingly applied in fisheries management to achieve ecosystem objectives (*e.g.* Sainsbury *et al.*, 2000).

Because MSE is relatively demanding of resources, in terms of the time and expertise required for model development as well as data requirements (Smith *et al.*, 1999), use of the approach to date has mainly been restricted to relatively large-scale commercial fisheries (*e.g.* Polacheck *et al.*, 1999; Punt *et al.*, 2005; Dichmont *et al.*, 2006). MSE models produced for such fisheries are typically not well suited to the types of data available for smaller scalefish fisheries around Australia and, in particular, for recreational fisheries. Due to the limited availability of, and difficulty of obtaining, reliable long-term CPUE data for such fisheries (Hall, 2005), stock assessments often rely heavily on equilibrium-based models such as catch curve analyses to estimate mortality from age composition data (*e.g.* Wise *et al.*, 2007). Although such stock assessment methods are relatively simple, evaluation of alternative management strategies for recreational fisheries is likely to be more complex because, unlike most commercial fisheries, they often employ a mixture of different input and output management controls for regulating exploitation, including bag and boat limits, size restrictions, temporal and spatial closures, and others.

An added dimension to the management of many Australian fisheries is that several state agencies use qualitative weight-of-evidence arguments in their advice to fisheries managers. In the recent case of the demersal scalefish fishery in Western Australia, for example, mortality estimates produced by catch curve analysis were combined with qualitative information regarding levels of risk associated with various identified threats to fish stocks in advice given to managers (Wise *et al.*, 2007). For MSE models to be more informative for such fisheries, it could be argued that they should be able to incorporate this type of weight-of-evidence information. Moreover, the traditional MSE framework designed to evaluate simulation-tested decision rules may not be particularly well suited to the procedures by which a number of Australia's fisheries are managed. For many Australian recreational fisheries, management decisions are made via review processes rather than according to pre-agreed modifications to catch or effort, as specified by a decision rule. The value of MSE for such fisheries for informing management (and engaging stakeholders) could thus be enhanced if models were designed to capture this reality, *i.e.* by allowing managers and stakeholders to explore the consequences of various combinations of management controls themselves over different time scales.

A factor that may be limiting more widespread use of MSE is that fisheries managers and other stakeholders with non-technical or non-science based backgrounds may be reluctant to adopt the approach because of its inherent complexity (Smith *et al.*, 1999; Rochet & Rice, 2009). Development of MSE models that are valuable to smaller-scale, data-poor fisheries presents a major challenge. Such models need to be able to provide robust and reliable stock assessment advice and effectively communicate this information to stakeholders with a wide range of backgrounds and levels of fisheries knowledge and experience.

Recruitment variability

This project was developed in response to a concern expressed by representatives of the Department of Fisheries, Western Australia, RecFishWest and WAFIC that recruitment variability poses a significant threat to the sustainability of fisheries for scalefish in WA. Recognition by the Western Australian Government that some fish stocks in the state are now over-exploited led the Minister in November 2007 to close the commercial fishery for several demersal scalefish species in waters between Lancelin (~ 130 km north of Perth) and Mandurah (~ 70 km south of Perth). More recently, a number of management changes were introduced to the recreational fishery for these species in waters between Kalbarri (~ 590 km north of Perth) and Augusta (~ 320 km south of Perth). These include a two month closure, tighter bag and boat limits, compulsory possession of release weights when fishing, and a "recreational fishing from boat licence" (Department of Fisheries, Western Australia, 2010).

The fact that people should be concerned about the implications of recruitment variability is well supported by the scientific literature. The research highlights that recruitment variability can indeed pose a significant threat to fish stocks, particularly if they are heavily exploited. For example, Koslow (1989), as cited by Koslow *et al.* (2000), noted that "highly autocorrelated recruitment variability increases the risk of stock collapse if fishing reduces the number of mature age classes below the interval between good recruitment events". Furthermore, if episodic recruitment occurs, Koslow *et al.* (2000) advise that "elimination of the older mature year classes may significantly impair the population's ability to withstand extended periods of poor recruitment". Similarly, Leaman and Beamish (1984) suggest that "age truncation will be most detrimental when reproductive success is highly variable, since stock maintenance may be dependent on the relative stability of reproductive output that results from a broad spectrum of age classes." Berkeley *et al.* (2004) advise that "age truncation, the removal of older age classes via fishing, occurs even at moderate levels of exploitation." An added concern is that recruitment variability is predicted to increase at low levels of population size (Myers, 2001).

Recruitment variability potentially has a number of important implications for fisheries management. For example, traditional management approaches, such as those based on the principal of a "spawn at least once policy" for protecting stocks from over-fishing (Myers & Mertz, 1998; Myers, 2001), may be inadequate for stocks which experience high levels of recruitment variability. It would thus be important for researchers and managers to have a good understanding of how the effectiveness of commonly used management controls is influenced by recruitment variability. Recruitment variability also has important implications for the reliability of stock assessments. For example, Koslow *et al.* (2000) point out that "assessment models assuming a mean annual recruitment to the population with random (*i.e.* non-autocorrelated) variability around that mean are inappropriate", *i.e.* as fishing poses a greater risk to stocks with highly-autocorrelated recruitment variability (Koslow, 1989). An understanding of which stock assessment approaches are compromised most by recruitment variability would be of benefit to both scientists and managers.

This project aimed to provide a generic MSE tool applicable to fisheries for which assessments rely heavily on mortality estimation using equilibrium-based methods such as catch curve analysis. Model development has focused on designing and testing the effectiveness of the program interface for communicating stock assessment information to people with potentially limited fisheries background and experience. The MSE model has been used to undertake Monte Carlo simulations to explore the implications of recruitment variability for different stock assessment and management approaches for two fish species that differ markedly in biology (West Australian Dhufish, *Glaucosoma hebraicum* and Tarwhine, *Rhabdosargus sarba*).

NEED

Although methods exist for predicting the likely outcomes of alternative management strategies when sufficient data exist to allow fitting of traditional fishery dynamics models, such data are lacking for many smaller-scale fisheries. Indeed, this situation is true of most recreational scalefish fisheries throughout Australia. For these fisheries, current stock assessments typically rely strongly on collection of age composition data and mortality-based estimation of stock status using simple, equilibrium-based models. The outputs of these assessments are sometimes combined with subjective information about factors that can place the fish stock at increased risk, and presented as advice to management in a "weight-of-evidence" framework.

The Department of Fisheries, Western Australia, RecFishWest, WAFIC, and the WA FRAB recognised an urgent need to assess the implications of variable recruitment for key demersal scalefish species in south-western Australia and to respond appropriately when developing management plans. Generic computer simulation tools and operating models (*i.e.* models that represent our best understanding of the fish stock and fishery) are required to assist in determining the most appropriate scientific responses (*i.e.* monitoring programs and analyses for producing reliable age-based assessments) and management responses (*i.e.* appropriate harvest control strategies). There is also an important need to develop tools to help facilitate effective communication of the outcomes of uncertain stock assessments (which are likely to be compromised by recruitment variability) to fishery stakeholders with varied backgrounds and experience.

OBJECTIVES

(1) Develop a generic management strategy evaluation framework capable of generating (through simulation) age composition data and results from catch curve analyses for species with a range of different life history characteristics (including different levels of recruitment variability), displaying these for decision-making in a weight-of-evidence framework (based on that currently being used in WA), accepting the resulting decisions and simulating the consequences of those decisions.

(2) Explore and advise on the effectiveness of alternative scientific, management and communication approaches to the types of data generated for species with a range of life history characteristics (including different levels of recruitment variability) considered by the management strategy evaluation framework, based on the results of scenario testing and use of Monte Carlo methods.

CHAPTER 2: Introduction to the MSE model

OVERVIEW OF MODEL

The MSE model developed in this study is designed to evaluate the outcomes of alternative options for scalefish stocks. The model is relevant to (the many) Australian fisheries for which assessments are based on mortality estimates derived through fitting catch curves to age composition data. It is a single-species and single-area, sex-, length- and age-structured model that can simulate the effects of a range of fisheries management controls, including those commonly applied to recreational fisheries (*i.e.* bag and boat limits, a minimum legal length for retention, and spatial and temporal closures). A proportional reduction in effort control and a quota control have also been added to the model, thus making it relevant to some other fisheries (including some commercial fisheries).

The model adopts an annual time step which is assumed to start midway through the main period of spawning of the fish species. The sequence of events undertaken by the model in each simulation run begins with an initialisation step to determine the initial state of the exploited fish stock under an existing management regime. Users of the program can then assess, from a range of stock status information provided by the model, the state of the fish stock at this initial state, after which changes to the existing management can be introduced. After a second assessment of the stock undertaken at the end of a specified simulation projection period, the outcomes of the management change can be evaluated.

In accordance with the general MSE framework, as summarised in Chapter 1, the model consists of two core components, namely the operating model and the management strategy. The operating model simulates the population dynamics of the fish stock and estimates the effects of management controls on fishing effort and catches taken by the fishery. The management strategy can be described as a combination of an observation model applied to sample the simulated population, an assessment model used to determine the state of the stock, and the decision-making model that simulates how management changes are chosen and implemented.

In a traditional MSE, the decision-making component of the management strategy consists of a decision rule, which specifies how management should be modified given the perceived state of the fish stock relative to a set of specified reference points. Such a specification allows the MSE to be run as a “closed loop” (Walters, 1998) as the decision-making process is internal to the MSE framework and can be automated. The application of such a fixed decision rule is based around a feedback loop in which the management of the fishery is automatically adjusted at set intervals, *e.g.* every five years over a total, specified projection period. This approach does not recognise, however, that the management of many fisheries, including many recreational fisheries in Australia, is often infrequent and undertaken at irregular intervals, and that decisions are usually based on the outcomes of a review process rather than according to a fixed decision rule. Our MSE model enables the user to decide on which combination of management controls is to be applied during a single, specified projection period, or alternatively, for fishing mortality to be adjusted according to a fixed decision rule.

The operating model constitutes a single-species model and thus assumes that, for the species being studied, no explicit interactions with other species influence the dynamics of the fishery. The simulated fish population is considered as a single, spatially homogenous entity and it is assumed that fishers are randomly distributed across the total area of the fishery. The model keeps track of the relative numbers of fish by age, length and sex, and describes the key dynamic biological processes characteristic of the simulated fish stock, such as recruitment, growth and mortality. It also explicitly accounts for the probability of sex change in functionally hermaphroditic fish species, thus making it applicable to both gonochoristic (separate sexes) and hermaphroditic fish species, including protogynous (female to male sex change) and protandrous hermaphrodites (male to female sex change).

The operating model simulates the combined effects of various input and output management controls on the exploited resource. These include (1) a boat limit, (2) a bag limit, (3) a minimum legal length for retention, (4) a temporal closure, (5) a spatial closure, (6) a proportional effort reduction control and (7) a catch quota. The model recognises the possibility that fish can experience post-release mortality (from hooking or barotrauma-related injuries caused when fish are rapidly brought to the surface from depth), or that they may die because of high-grading by fishers.

A detailed specification of the mathematical formulations underlying the MSE model is provided in Appendix 3.

INSTALLING AND USING THE MODEL

The MSE program (available as an .exe file) and all files required to run the software can be downloaded freely from http://www.cffrfisheriesmodelling.net/. The steps required to download the software are as follows. (1) Right-click on the website link for the MSE program and save it somewhere on the computer (*e.g.* the desktop).

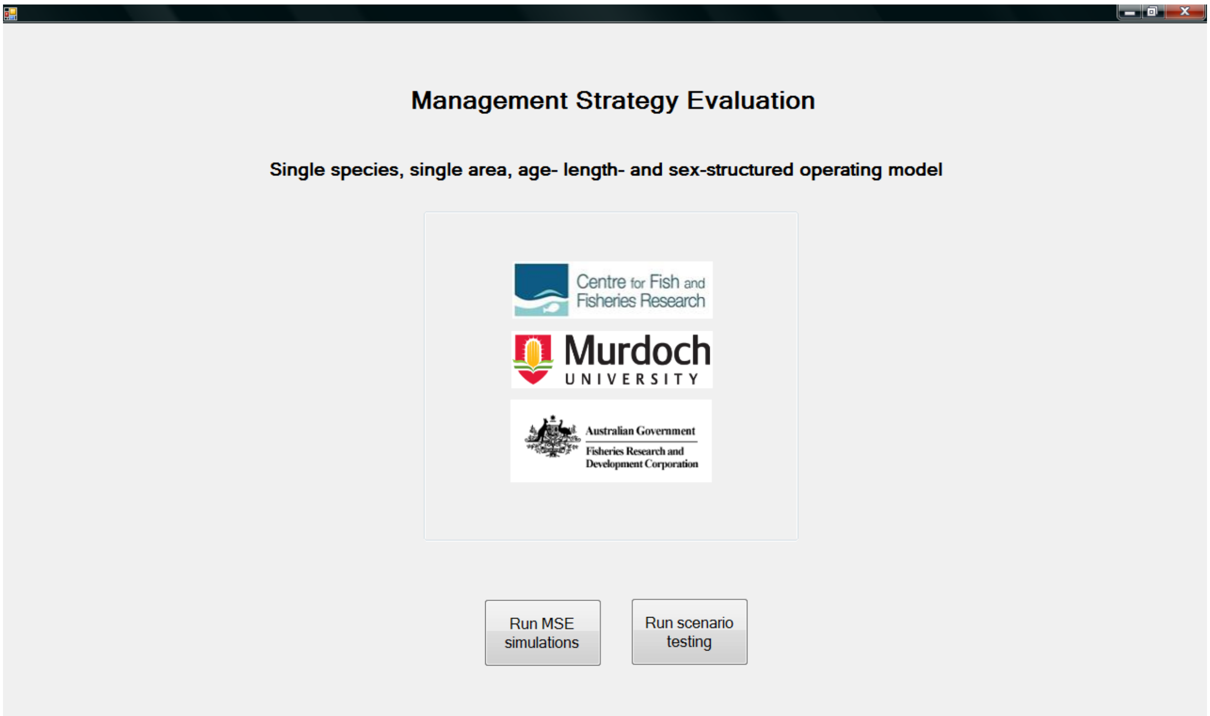
(2) Open the .zip file and copy the folders MSEDataFiles and MSEResultFiles directly to the C drive. (3) Copy the folders called Scenarios and Results onto the desktop. (Note that the above folders can be placed elsewhere on the computer, but the directory pathways will need to be specified each time the model is run). (4) Install the MSE program on the computer by clicking on the MSE.exe file.

The MSEDataFiles folder contains all of the data files required to run MSE simulations. Within MSEDataFiles are folders called DefaultSpecies, TestSpecies, Species1 and Species2, each of which contain three .txt files, namely SpeciesParameters.txt, SimulationParameters.txt and ManagementParameters.txt. The folder called DefaultSpecies (and Species1) contain parameter values to run simulations for West Australian Dhufish (*Glaucosoma hebraicum*), whilst TestSpecies and Species2 contain parameters for Yellowfin Whiting (*Sillago schomburgkii*) and Tarwhine (*Rhabdosargus sarba*), respectively. MSEDataFiles also contains folders named CatchCurveAnalyses and FinalCatchCurveAnalyses containing the data files required to run the different types of catch curve analysis available within the program. The MSEResultFiles folder is used to store all outputs from model simulations, including length and age composition data, levels of recruitment for different year classes, estimates of fishing mortality and the results of per recruit analyses. Note that this folder is initially empty (until simulations have been run).

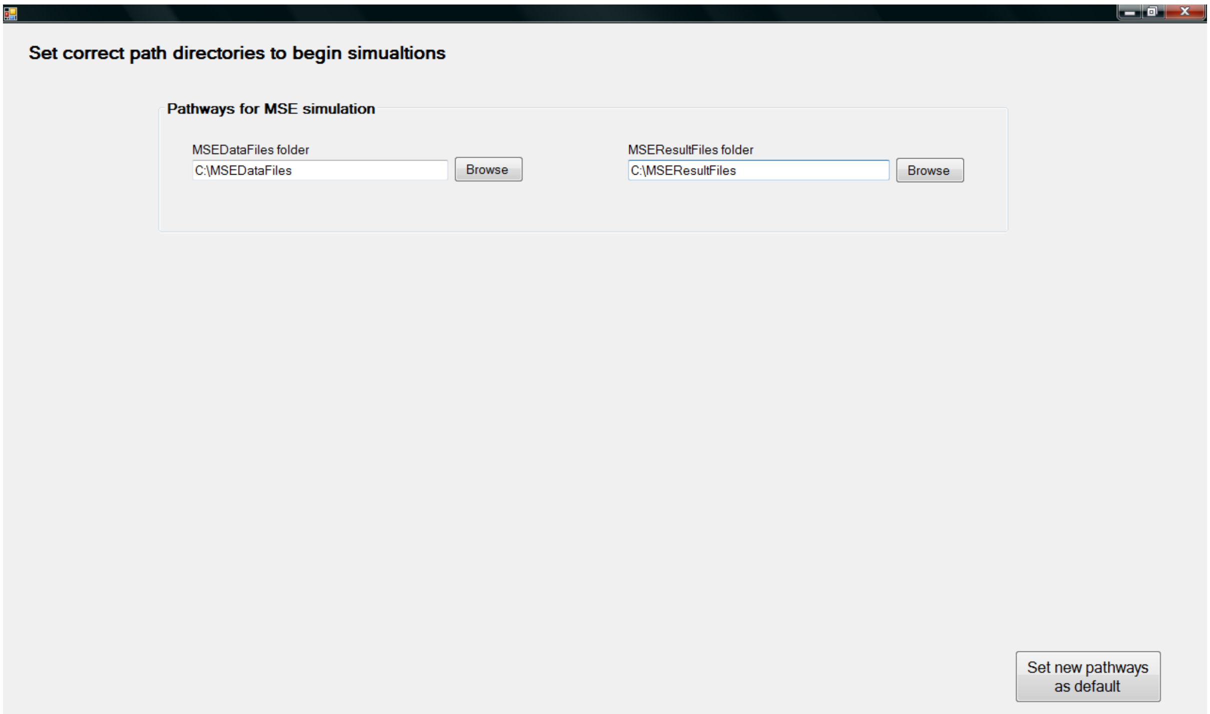
The MSE program can be run in two modes, a normal “MSE simulation mode” or a “scenario testing mode” (*i.e.* for workshop/student class situations). When running normal MSE simulations, the model inputs data from the files contained within the DefaultSpecies folder. The MSE model can be used to run simulations for essentially any species for which relevant data are available. The user can change parameter values using either the MSE program interface (after data have been read in), or by modifying the .txt files in the DefaultSpecies folder. Note that the program does not allow for changes in the layout of the .txt files (*i.e.* the order of parameters and spacing).

When in scenario testing mode, the user is presented with two options, to run either a "preliminary test trial" or a "scenario trial." The preliminary test trial represents a single scenario for a fishery (using data entered from the TestSpecies folder). It is intended that the user can run this scenario repeatedly to be able to familiarise themselves with the program and the effects of different management controls, before continuing to run other scenario trials. By default, the program contains 12 different fishery scenarios, *i.e.* it considers two fish species (using data files in the folders Species1 and Species2), three initial levels of exploitation and two levels of recruitment variability. The data files that specify aspects of the fishery scenarios are contained within the folder named Scenarios. The order in which the 12 scenarios are run is specified in the file called MSE.txt. Scenarios can be run multiple times by repeating the number for a particular scenario on several lines within MSE.txt. The file TrialNum.txt maintains a count of the scenarios that have been completed so that the next scenario can be read in correctly. To re-run all scenarios, delete the values in this file except the 0 on the first line, leaving the cursor at the start of the next line before saving. The scenarios (*i.e.* fish species, initial level of exploitation and level of recruitment variability) can be modified by altering the other three .txt files in the Scenarios folder.

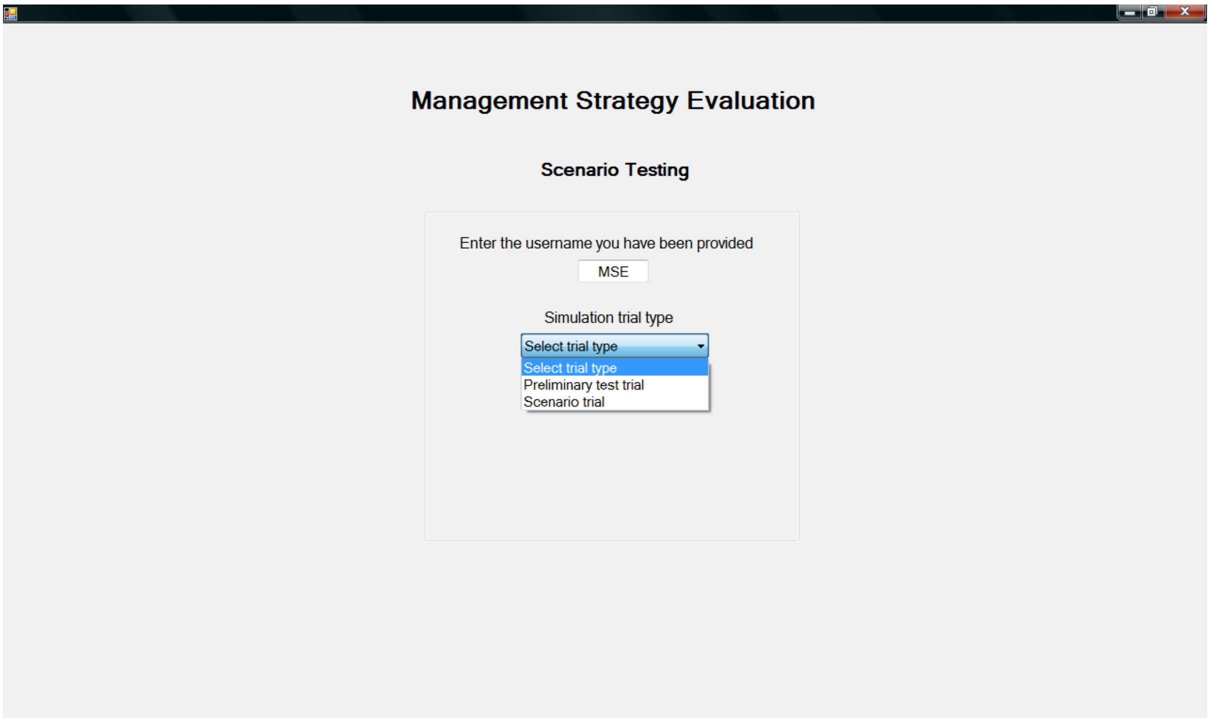
The steps required to run model simulations are described below, with examples of the various user screens.



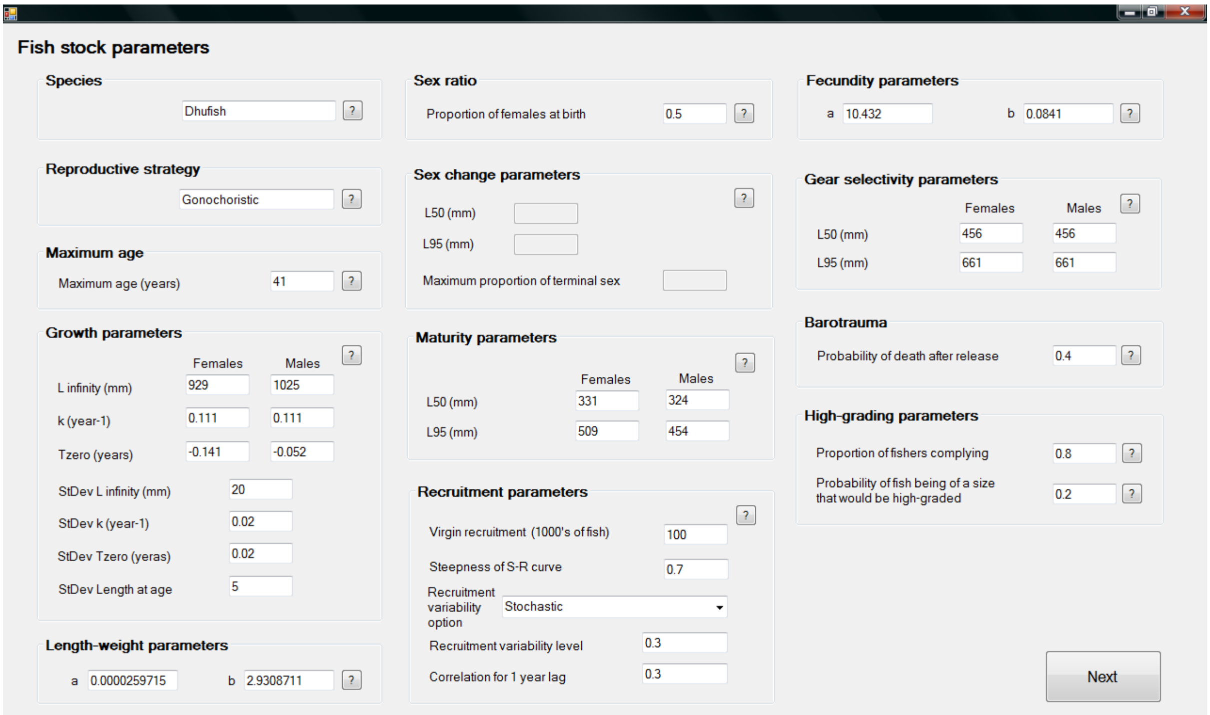
1. The above introductory screen appears when the MSE program is first opened. To use the program to undertake MSE simulations for research purposes, press "Run MSE simulations". To run the program in scenario testing mode, press "Run scenario testing".



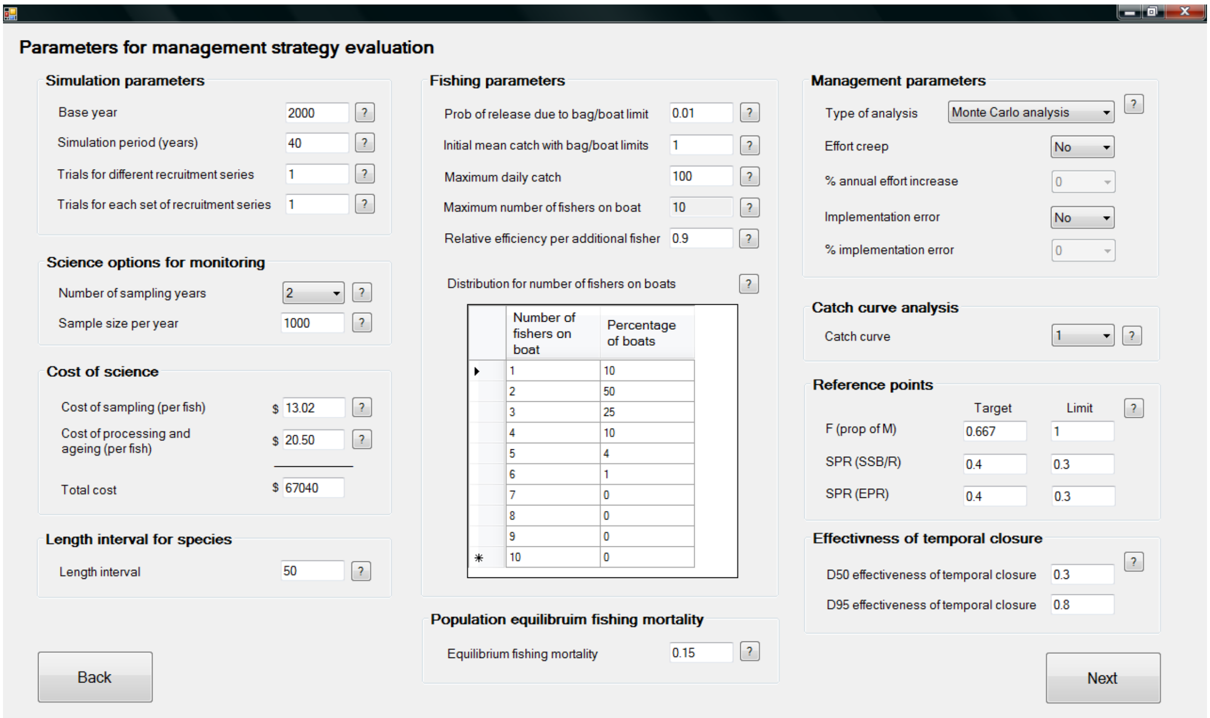
2. Prior to commencing any type of simulation, the MSE program completes a check to ensure that the directories to all folders with required data files can be found. If they cannot be located, the above form appears to enable the correct directories to the folders containing these files on the computer to be specified. Press "Set new pathways as default" to return to the introductory screen.



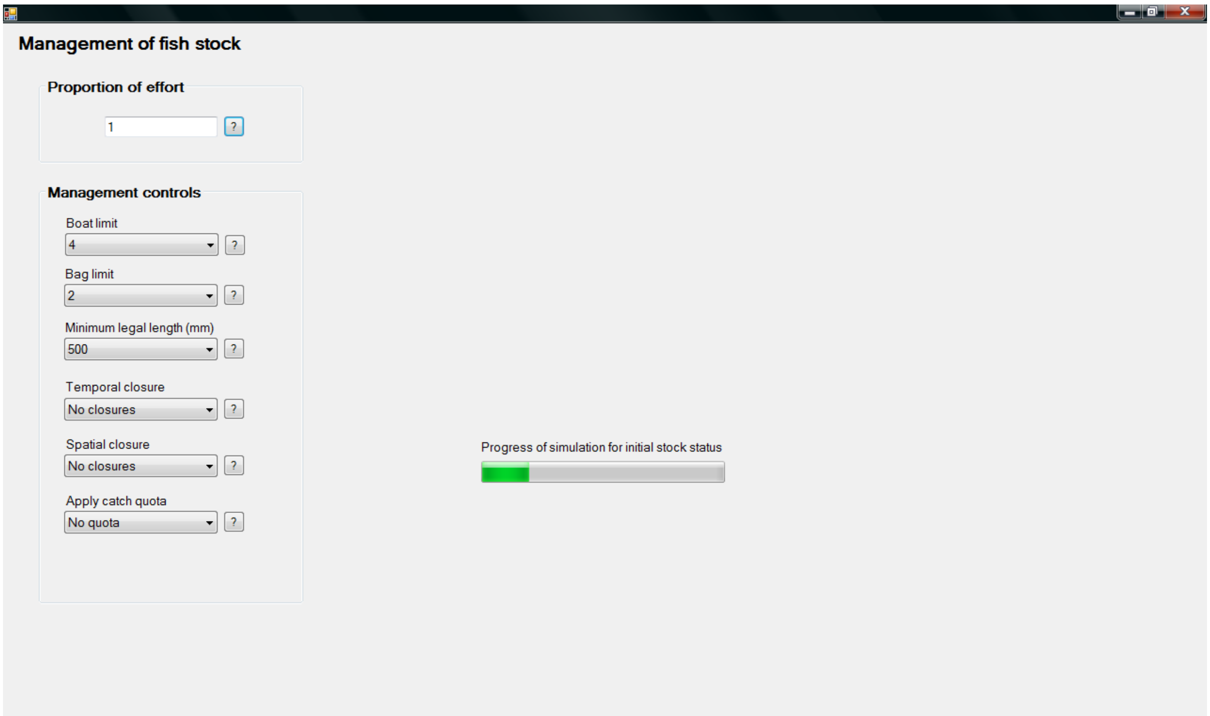
3. This screen appears only when the program is run in scenario testing mode (or the simulation proceeds directly to step 4). Enter the letters “MSE” as the username and click “Next”. Select either “Preliminary test trial” or “Scenario trial” before proceeding to read in data for the simulation.



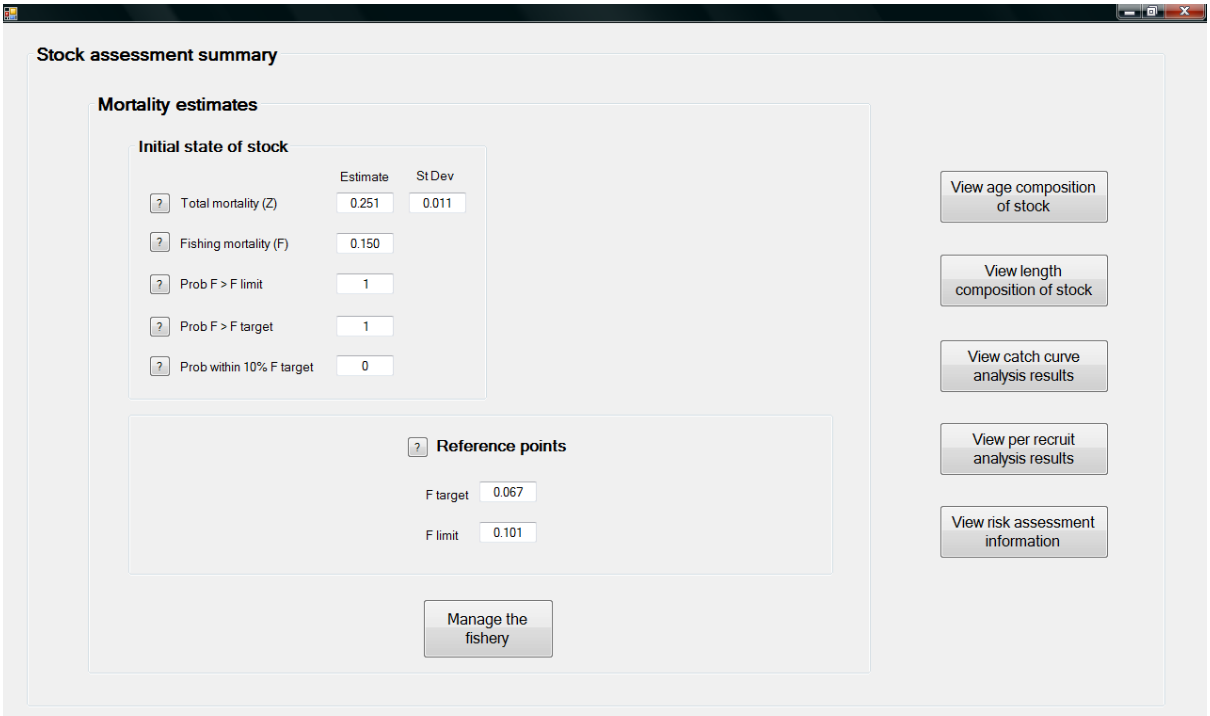
4. Read in the required biological parameter values for the species by pressing “Read parameters” and then continue by clicking the “Next” button (which appears when the data have been read in by the program).



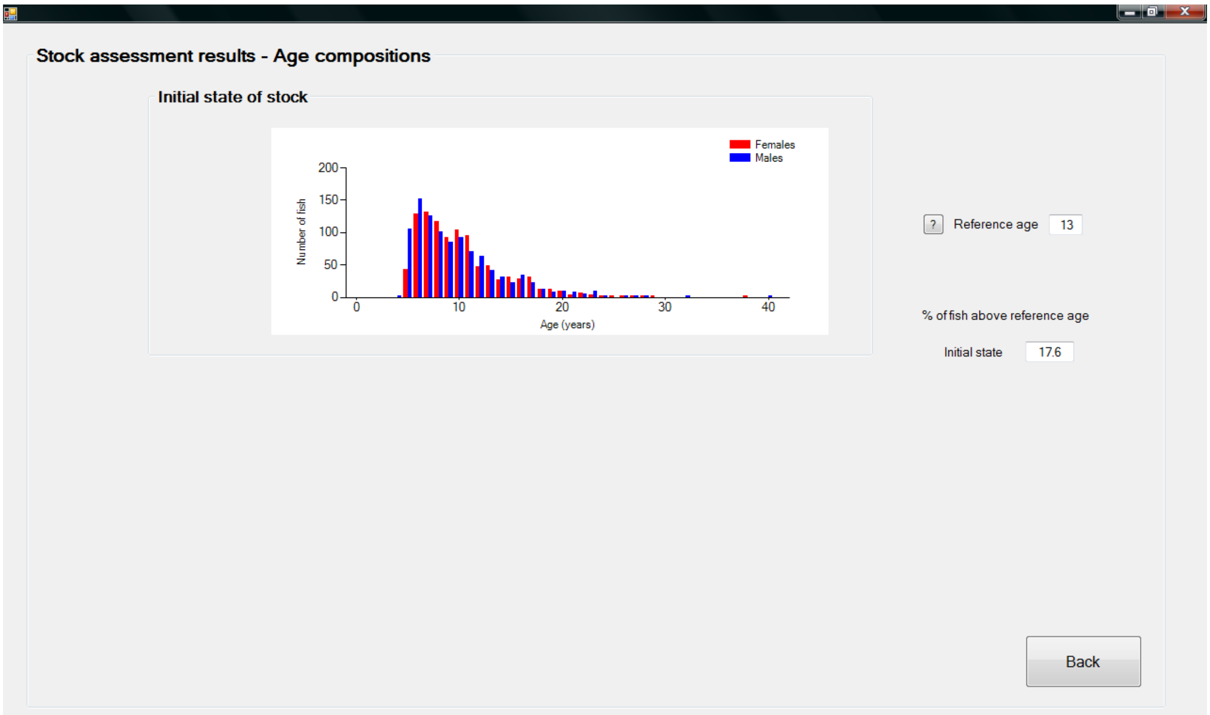
5. Read in the parameters which specify aspects of the simulation by pressing “Read parameters”. Click “Next” to proceed.



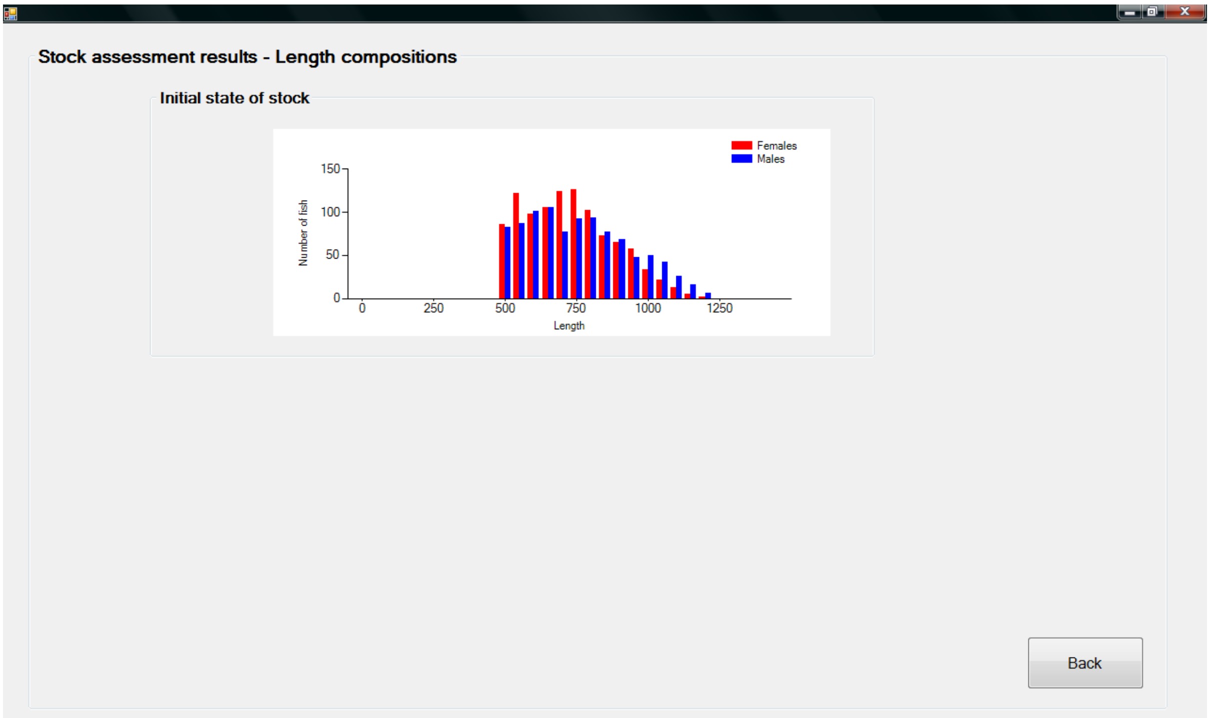
6. Press “Read parameters” to read in the values for the management controls being applied to the fish stock in its initial state. If in normal MSE simulation mode, these parameters can be changed by selecting desired values from the available lists for each control. To proceed, click “Run initial assessment” to produce information about state of the stock prior to the projection period.



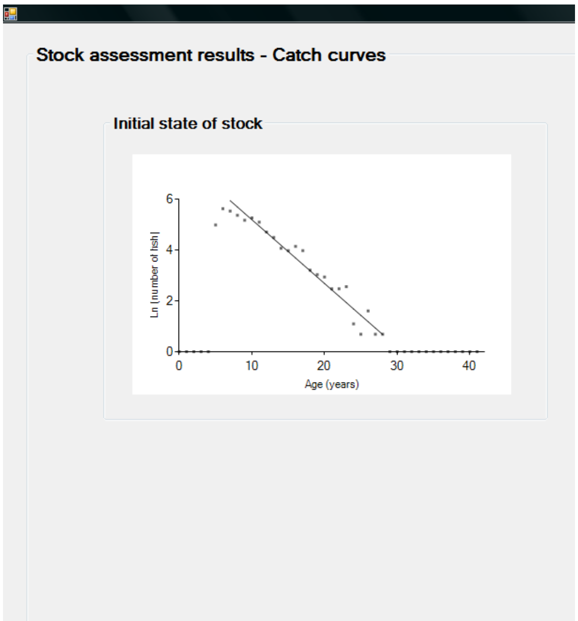
7. Once the initial state of the stock has been determined and an assessment undertaken, the results of catch curve analysis are presented, as shown in the above screen. The user can view additional information about the initial stock state by selecting from the buttons to the right of the form (see steps 8 to 12).



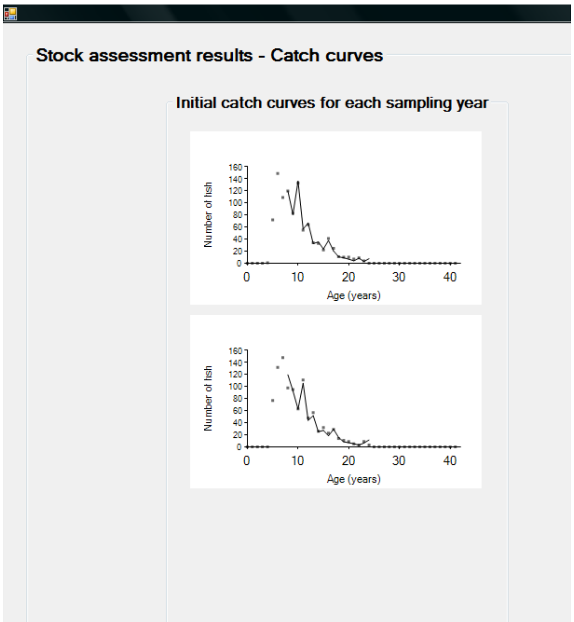
8. Example of age composition sample data. Note, if multiple years of samples are specified in the form containing the simulation parameters, data shown on this form are pooled for the different years. To return to the previous screen, select “Back”.



9. Example of length composition sample data. To return to the previous screen, select “Back”.

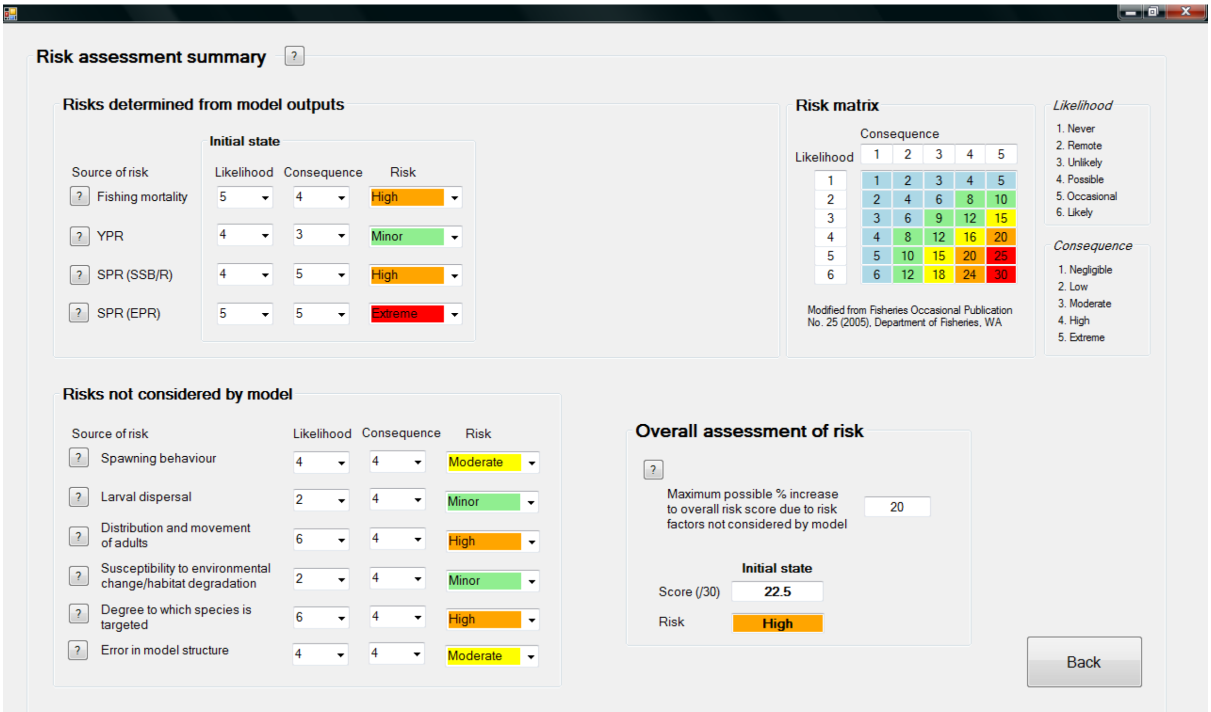


10a. Example of a catch curve analysis plot for a linear catch curve (Ricker, 1975) fitted to the natural logarithms of the frequencies of fish at age in sample data. Several other forms of catch curve analysis are available within the program.

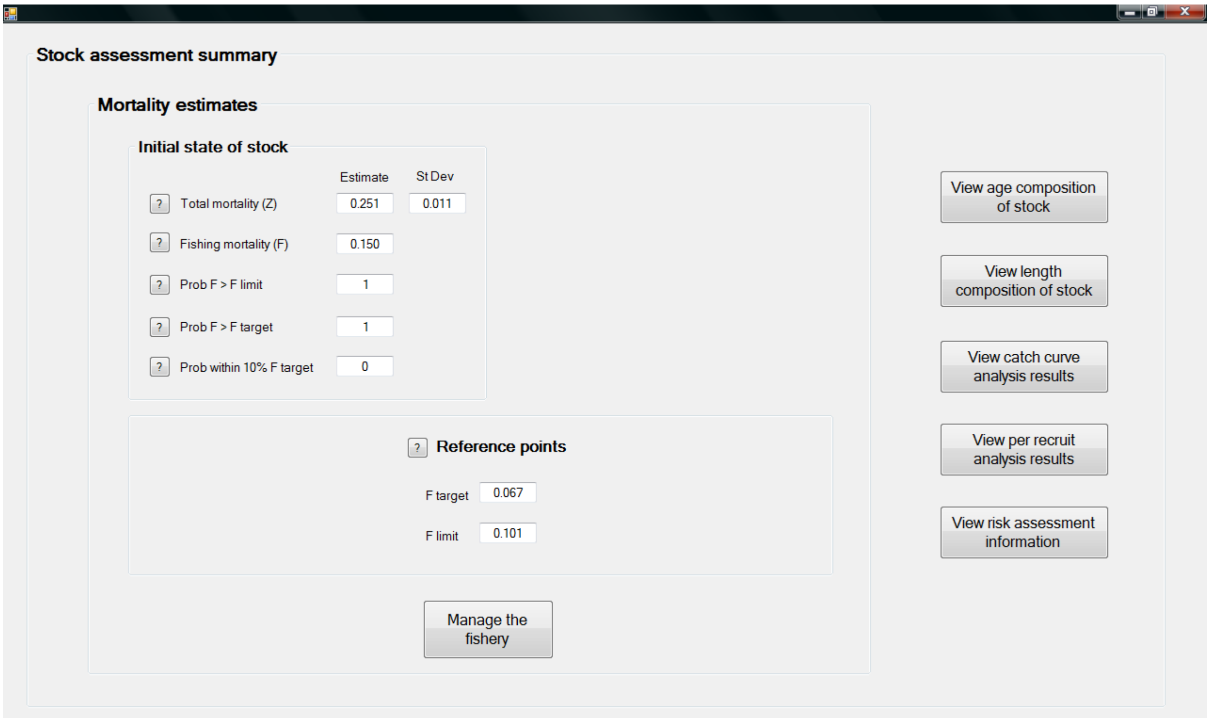


10b. Example of a plot showing the results of relative abundance analysis, an extended form of catch curve analysis (Deriso *et al*., 1985). This form of analysis requires at least two years of sample data.

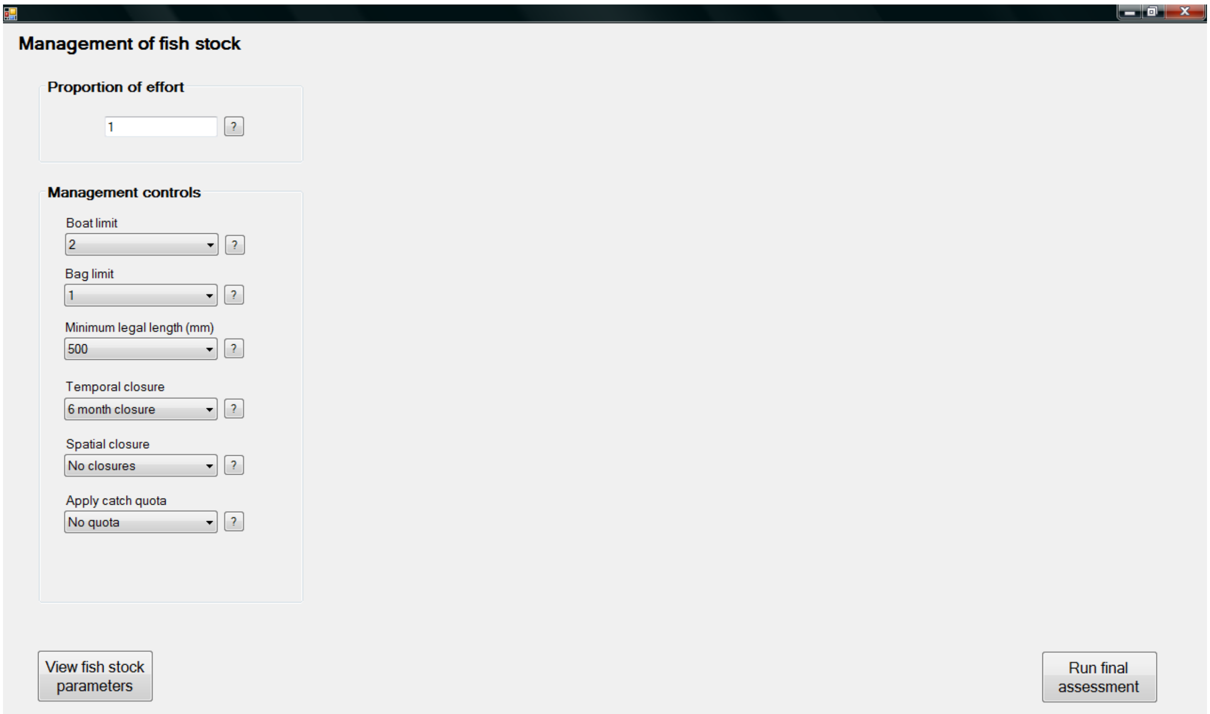
11. Example of results of per recruit analyses, including estimates of yield per recruit and spawning potential ratio (based on spawning biomass per recruit and egg per recruit).



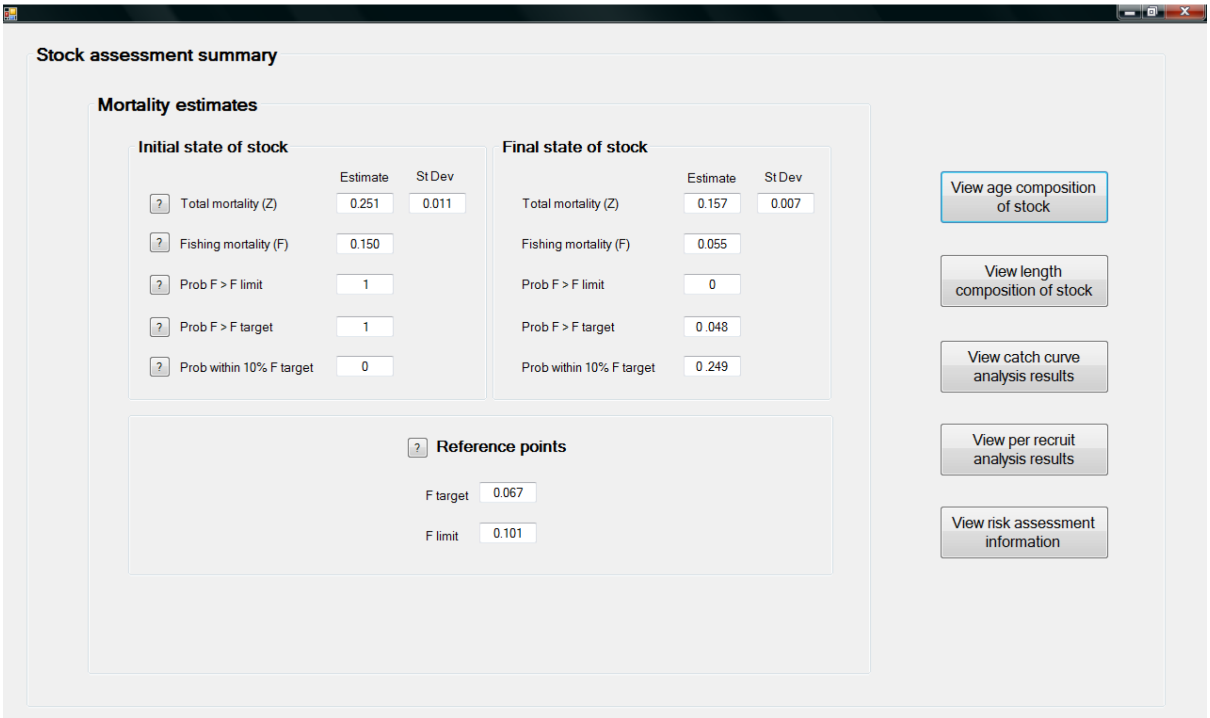
12. Example of results from a risk assessment for the stock in its initial state, based on mortality-based model outputs and other risks not considered directly by the model (assessed subjectively). Note that risks associated with subjective criteria are added to risks calculated from mortality estimates (see below for more details on risk calculations).



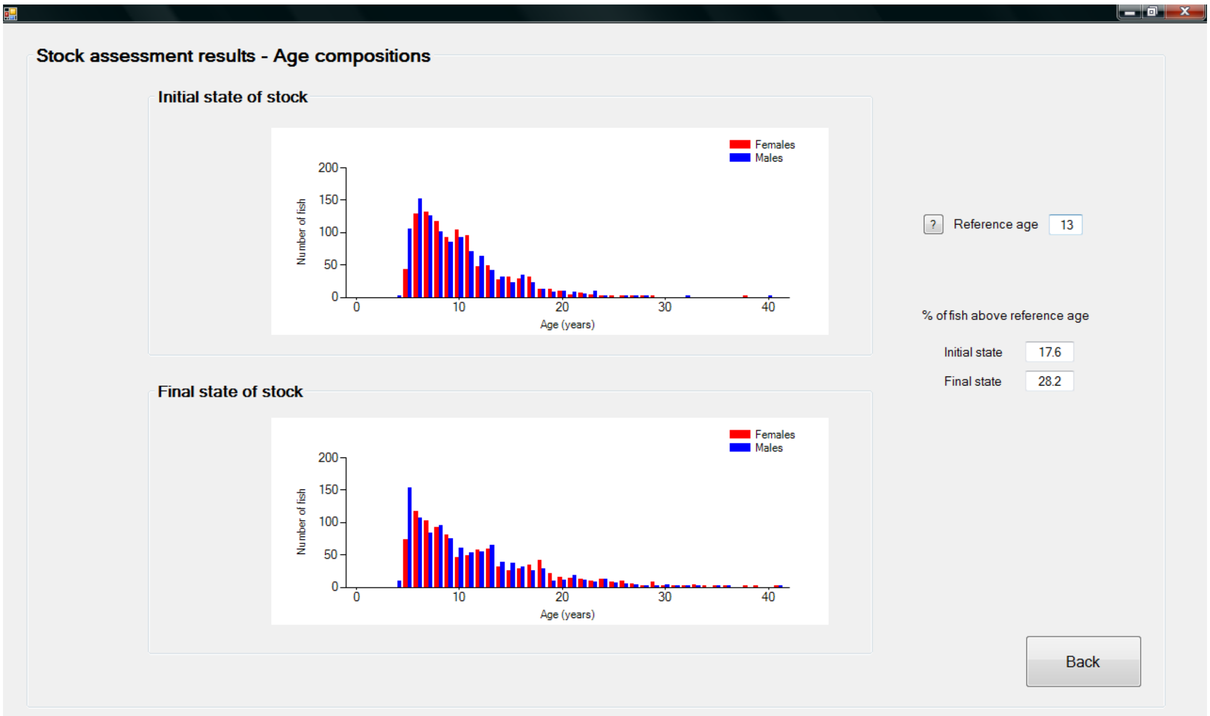
13. After viewing information about the initial state of the fish stock and returning to the “Stock assessment summary” screen, click on the “Manage the fishery” button.



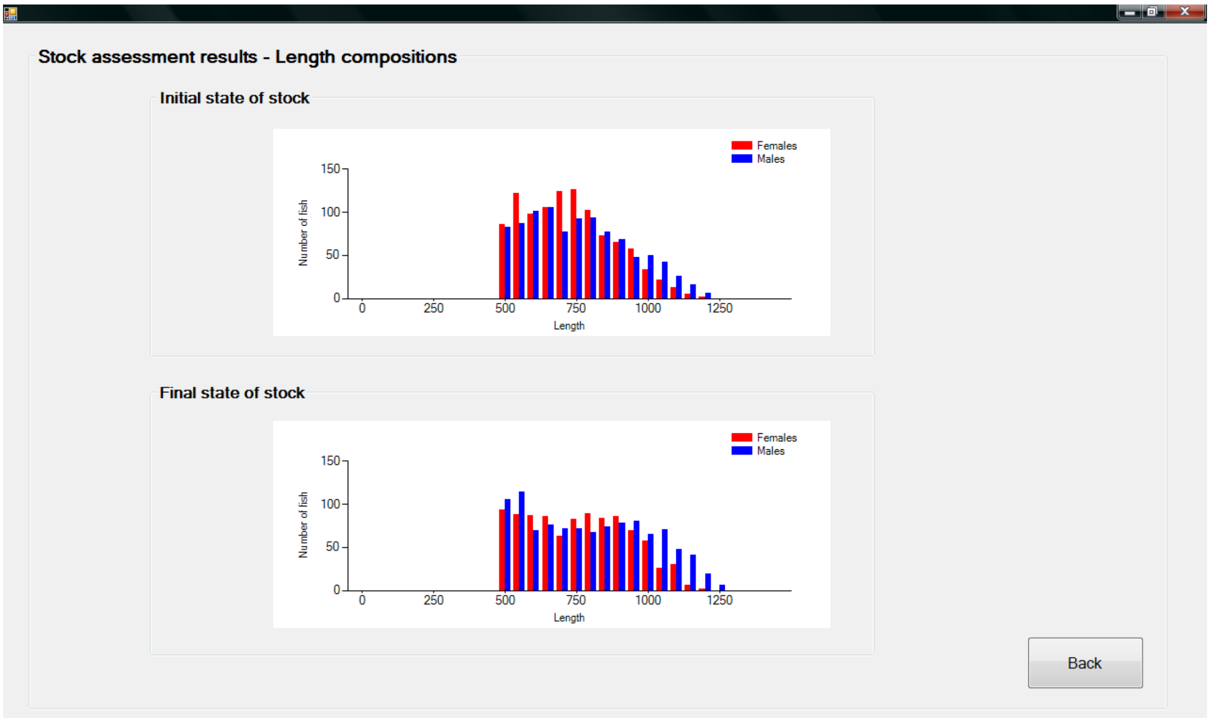
14. Change none, some or all of the values for management controls and click “Run final assessment” to run the simulation over the specified projection period and produce information about the stock in its final state.



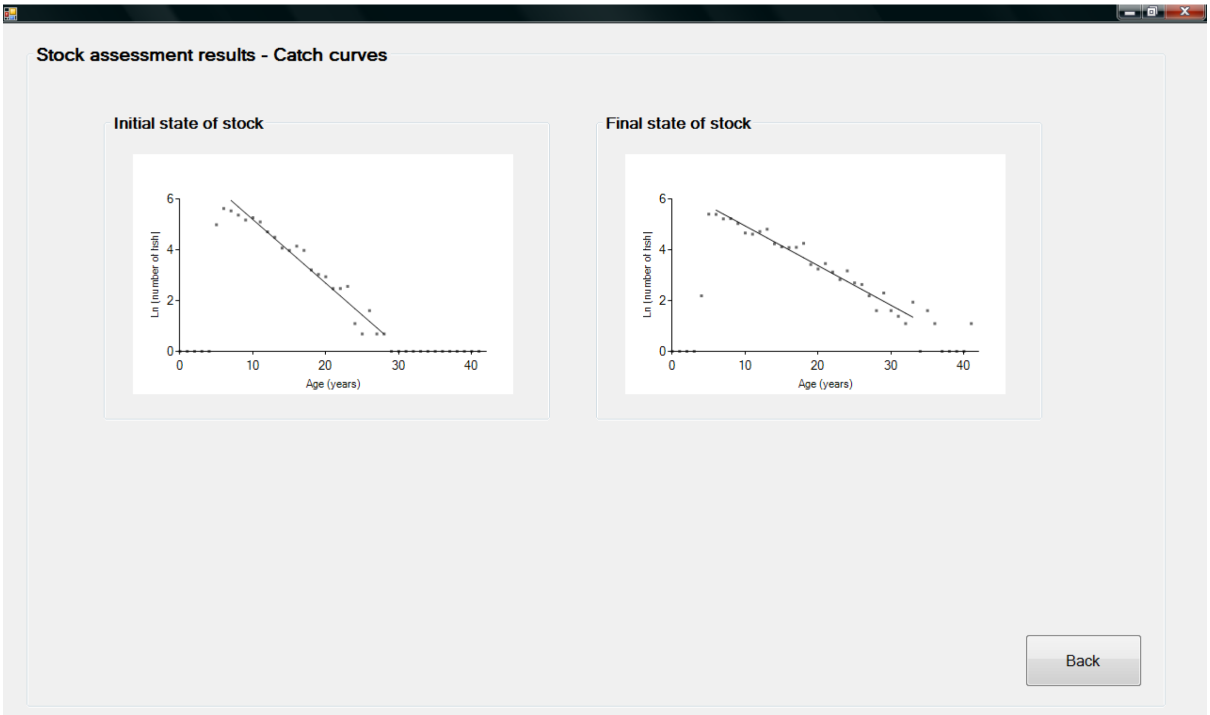
15. Example of information produced by catch curve analysis for the stock in its initial and final state. Additional information about the stock in its final state are presented in steps 16-20.



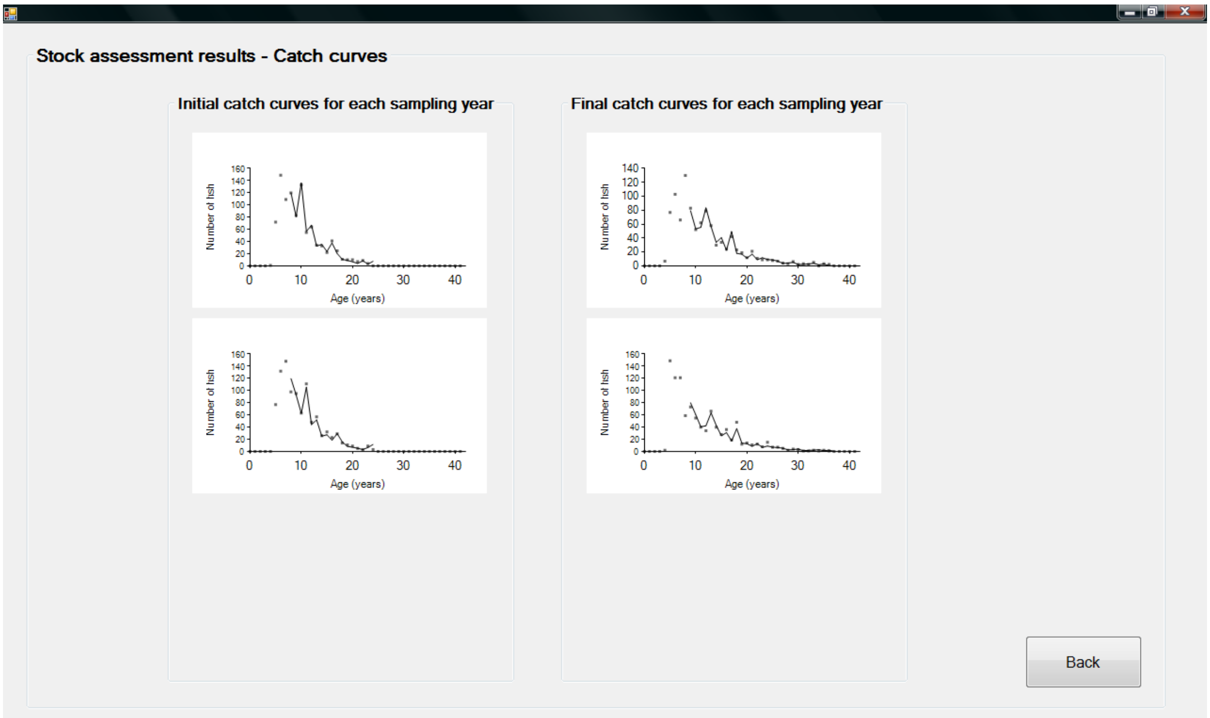
16. Example of age composition sample data for the stock in its initial and final state.



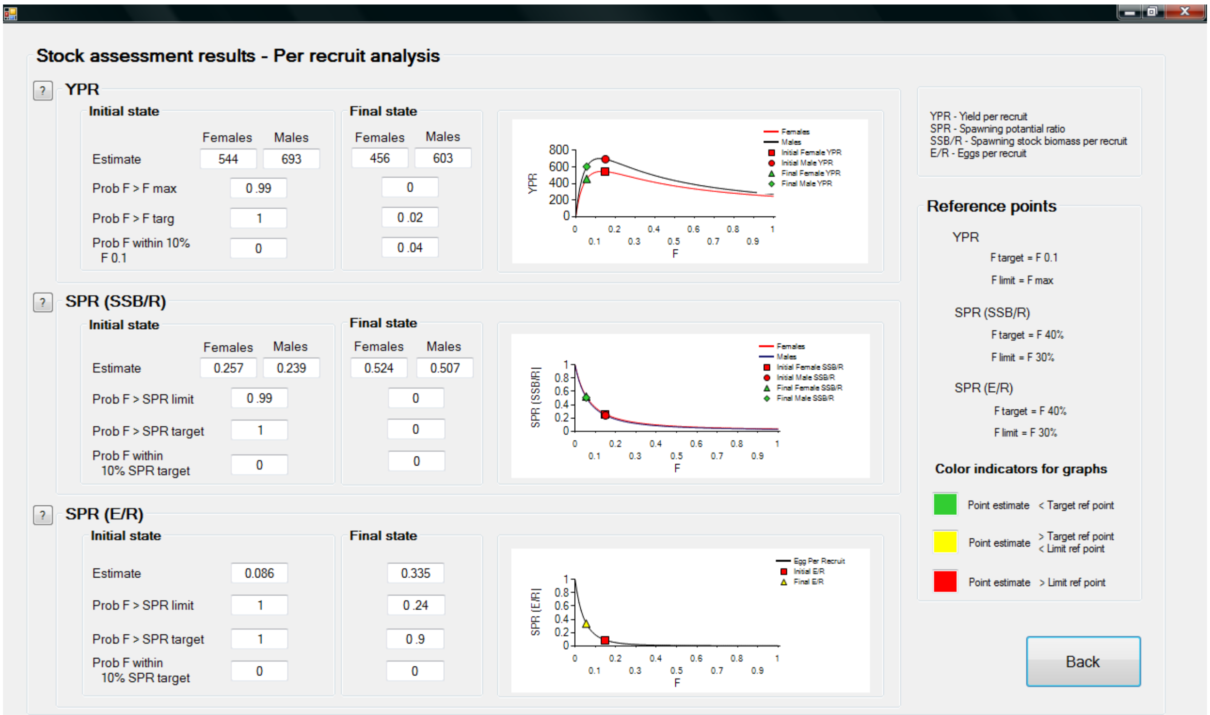
17. Example of length composition sample data for the stock in its initial and final state.



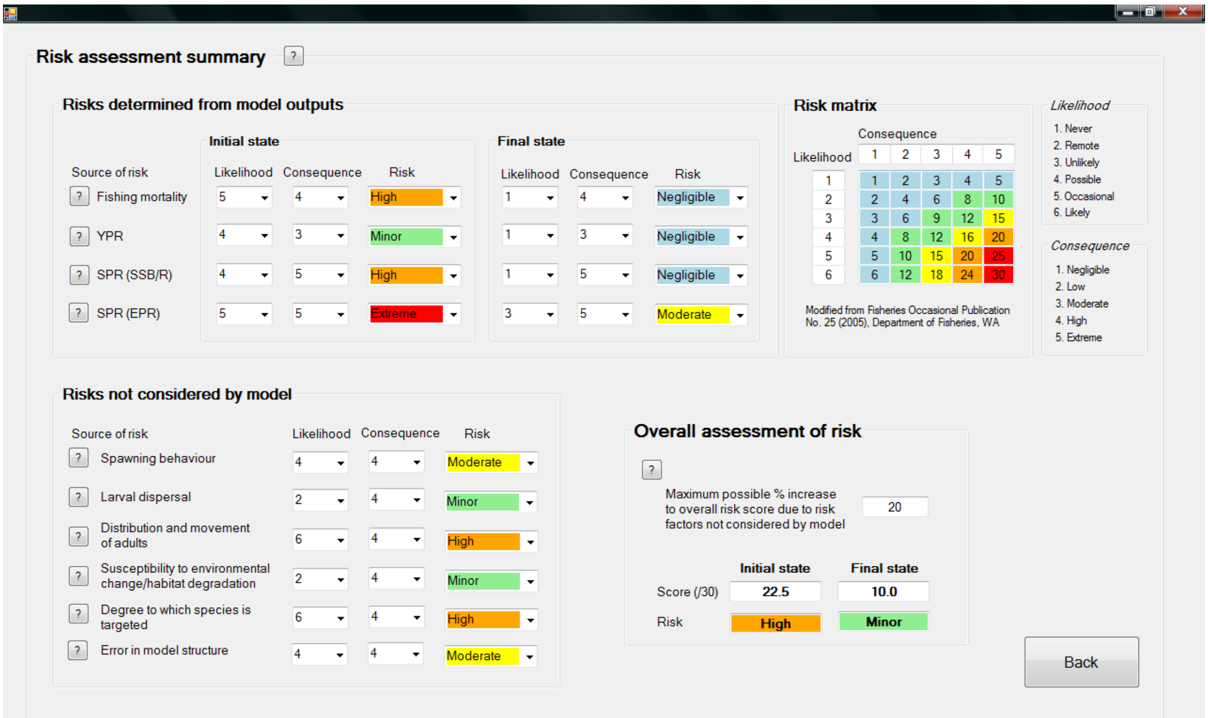
18a. Example of linear catch curve analysis plots for the stock in its initial and final state.



18b. Example of relative abundance analysis plots for the stock in its initial and final state.



19. Example of results of per recruit analyses for the stock in its initial and final state.



20. Example of risk assessment information for the stock in its initial and final state.

Notes on risk assessment approach

The methods used by the MSE program to provide risk assessment information are adapted from those described for an ecological risk assessment for Western Rock Lobster (Department of Fisheries, Western Australia, 2009). The levels of risk that various identified potential "hazardous events" pose to a fish stock are estimated according to the likelihoods of those events occurring and the levels of consequence of their occurrence. The likelihood of an event occurring is given a score ranging from 1 (=Never) to 6 (=Likely), and the level of consequence of an event is given a value ranging from 1 (=Negligible) to 5 (=Extreme). The product of the two values provides a risk score for each hazard (see risk matrix in figure above).

For hazardous events that relate to fishing mortality-based reference points (as assessed by the MSE assessment model using catch curve and per recruit analyses), the likelihood of an event occurring is calculated as follows:

$$
\begin{array}{ll}
\text{if } P(F > F_{\text{limit}}) = 1 & 6.\ \text{Likely} \\
\text{if } P(F > F_{\text{limit}}) > 0.5 & 5.\ \text{Occasional} \\
\text{if } P(F > F_{\text{limit}}) > 0 & 4.\ \text{Possible} \\
\text{if } P\left(F > F_{\text{target}}\right) = 1 & 3.\ \text{Unlikely} \\
\text{if } P\left(F > F_{\text{target}}\right) > 0.5 & 2.\ \text{Remote} \\
\text{if } P(F > F_{\text{target}}) = 0 & 1.\ \text{Never}
\end{array}
$$

where *P* refers to the probability of *F* exceeding the specified reference point. Likelihood scores for those hazardous events not calculated from catch curve or per recruit analyses are assessed subjectively. For all hazards, scores for the levels of consequence are subjectively assigned a value.

The risk scores for the hazardous events associated with fishing mortality are then averaged, as are also those for other identified hazards. An overall risk score is calculated by adding the average for the hazards associated with catch curve and per recruit analysis results to 20% of the average risk value for other identified hazards.

Note that this risk assessment approach was implemented very recently and that the results of the next chapter (from our scenario testing study) are based on an earlier version of the program without this analysis.

CHAPTER 3: Exploring the effectiveness of a fisheries modelling tool for communicating stock status information: a scenario testing study

INTRODUCTION

Fisheries management revolves around making choices (Hilborn & Walters, 1992). In theory, this involves managers or decision-makers making choices about which of a range of alternative possible management arrangements for a resource is most likely to achieve a set of specified management objectives (Punt & Hilborn, 1997; Lackey, 1998). To help ensure that appropriate choices are made, it is fundamentally important that decisions are based on the best available information about the current state of the resource being managed (Hilborn, 2003). Traditionally, the information on which fisheries management is based is that resulting from scientific stock assessments, provided to decision-makers in the form of fisheries management advice (Hilborn & Walters, 1992; Punt, 2008).

Conventional approaches to fisheries stock assessment are typically resource-intensive, largely because of the amount of data required for analyses (Hilborn & Liermann, 1998). To optimise use of available data, which for many fisheries come in a variety of forms, fisheries scientists frequently apply a range of statistical approaches in their analyses (Hilborn & Liermann, 1998; McAllister *et al.*, 2001). As a consequence, stock assessments commonly produce a wide range of complex outputs and results, each of which may be dependent on different assumptions. The main concern with this is that the complex and sometimes conflicting stock status information presented as advice to decision-makers is often used in an *ad hoc* manner, which can lead to poor management outcomes (Walters & Maguire, 1996).

Recognition of the limitations of traditional stock assessment approaches for dealing with uncertainties in fisheries led to the development of management strategy evaluation (MSE) (Smith, 1994). This simulation approach is used to predict the likely effectiveness of alternative management strategies before they are implemented, and thereby improve the likelihood of achieving desired management objectives (Sainsbury *et al.*, 2000; Kell *et al.*, 2007). MSE has been recognised for its potential value as a vehicle for involving not only scientists and managers, but also members of the fishing industry and other stakeholders in the management process (Smith *et al.*, 1999). Indeed, the adoption of such collaborative and participatory management systems has been argued by many as an important step forward for improving decision-making in fisheries (de la Mare, 1998; Kaplan & McCay, 2004; Johnson & van Densen, 2007).

One issue that may be acting to restrict the application of MSE to a broad range of fisheries throughout the world is that, because of its complexity, fishery stakeholders may struggle to fully understand the implications of MSE model outputs (Rochet & Rice, 2009). Lack of effective communication between scientists and other stakeholders has repeatedly been highlighted as one of the greatest challenges to successful fisheries management (de la Mare, 1998; Peterman, 2004). In a review of the role of MSE in the implementation of the Australian Fisheries Management Authority (AFMA) partnership approach, Smith *et al.* (1999) discuss the difficulties encountered with managers sometimes being reluctant to accept MSE as playing a major role in management because of its inherent technical complexity. The challenge is thus to develop robust models that are effective in communicating relatively complex stock assessment advice to people who may range broadly in background, fisheries knowledge and technical experience. If such tools can be developed, they should go a long way towards bridging the communication gap between science and management.

Although stock assessment advice ideally should be based on the results of many carefully designed simulations, there is considerable benefit from developing computer programs that allow the stakeholders to “pull the management levers” and thereby allow them to act as “fisheries managers” (Hilborn & Walters, 1992; Butterworth *et al*., 1997). Simulation gaming has been used widely as a tool for natural resource management (Barreteau *et al.*, 2007), particularly as a means for promoting discussion among stakeholders and for facilitating problem-solving. Simulation models can be excellent for providing simplified and easily understood representations of naturally complex systems and, as a result, the use of such models has also proven useful for teaching and training purposes (Martin *et al.*, 2007).

The two main objectives of this study were as follows:

(1) Design an effective and user-friendly program interface for our MSE simulation model so that it can potentially be used for a range of purposes, including education and for helping facilitate stakeholder involvement in fisheries management processes. As described in previous chapters, the MSE model is designed for fisheries that are relatively data-limited, but for which sufficient biological data are available to enable mortality-based assessments (using equilibrium approaches) and future predictions of stock status.

(2) Undertake a scenario testing study with university students using the MSE model to explore the effectiveness of the program for conveying stock status

information, and assess whether certain types of information have any influence on their decision-making.

MATERIALS AND METHODS

Overview of scenario testing procedures

The study involved running two computer workshops in which participants were asked to use the MSE program. Participants were presented with 12 fishery scenarios and were required to decide how to best manage the (simulated) fish stocks, given the management control options made available to them. The workshops were attended by 23 science students at Murdoch University, including nine undergraduate students in an initial workshop and 14 postgraduate students in a second workshop. At the start of both workshops, each participant was randomly assigned to a computer on which the MSE software had been installed. Prior to starting the exercise, the students were provided a brief (approximately 20 minute long) presentation about MSE and an outline of the computer workshop procedures.

The scenario testing exercise involved the participants viewing a range of information presented to them via the MSE program interface, from which they were required to assess the state of the simulated fishery, and then decide how to manage that fishery by pulling various “management levers”. The alternative management controls for regulating exploitation included a bag limit, a minimum legal length for retention, a temporal closure and a spatial closure. Once participants had entered their management decisions for a scenario (by changing none, some, or all of the different controls), a second assessment of the stock was then undertaken by the program at the end of the specified simulation projection period. Feedback was then provided to participants on the success of their management choices, including information about the stock at its final exploitation state and a score, calculated according to how well their management arrangements had met the specified management objective, *i.e.* how closely the final fishing mortality was to the target level of 2/3 of the natural mortality. All participants repeated the above-described procedure 12 times, *i.e.* with the scenario for the fishery differing on each occasion.

Prior to initiating the scenario testing exercise, students were asked to undertake a set of preliminary runs (for a single scenario not included in the experiment) to familiarise themselves with the program and its outputs. They were instructed to take as much time as they needed on these preliminary runs before commencing the scenario testing exercise (typically 30 minutes). Although the workshops were scheduled to run for three hours, all students completed the 12 scenarios within about 2 hours and thus time did not constrain the extent to which students could spend assessing information before making decisions.

Design of program interface for scenario testing

The different screens of the program interface (see Chapter 2) provided users with a wide range of information. This included details about the biological characteristics of the simulated fish species, parameters describing various aspects of the simualtion procedures (*e.g.* the projection period) and the initial values of the management controls specified for the simulated stock (Tables 3.1 and 3.2), as well as information about the state of the simulated stock. The stock assessment information provided to participants included age and length composition data from a specified sample of

Table 3.1. Biological parameters applied to simulations for the two fish species considered in the scenario testing study.

Species parameters	***Glaucosoma hebraicum***	***Rhabdosargus sarba***
Reproductive strategy	Gonochoristic	Gonochoristic
Maximum age (years)	41	11
von Bertalanffy growth parameters		
L_∞ female (mm)	929	290
k female (mm)	0.111	0.59
t_0 female (mm)	-0.141	0.12
L_∞ male (mm)	1025	290
k male (mm)	0.111	0.59
t_0 male (mm)	-0.052	0.12
Standard deviation of L_∞ (both sexes)	20	20
Standard deviation of k (both sexes)	0.02	0.05
Standard deviation of t_0 (both sexes)	0.02	0.05
Standard deviation of length at age (both sexes)	5	10
Length-weight parameters		
a	0.0000259715	0.000038822
b	2.9308711000	2.846243544
Sex ratio (proportion female at birth)	0.5	0.5
Maturity		
L_{50} female (mm)	331	177
L_{95} female (mm)	509	192
L_{50} male (mm)	324	170
L_{95} male (mm)	454	196
Recruitment		
Virgin recruitment (1000s of fish)	100	100
Steepness of stock-recruitment curve	0.7	0.7
Recruitment variability, $\sigma_{\ln R}$	*	*
Correlation for one year lag	0.3	0.3
Fecundity parameters		
a	10.432	5.0025
b	0.0841	17.557
Selectivity/vulnerability to capture		
L_{50} female (mm)	456	198
L_{95} female (mm)	661	235
L_{50} male (mm)	456	198
L_{95} male (mm)	661	235
Probability of post-release mortality	0.4	0.05
Probability of fishers complying	0.8	0.9
Probability of a captured fish being of a size that is high-graded	0.2	0.2

* dependent on the scenario, see table 3.3

Table 3.2. MSE simulation parameters and initial management parameters applied to simulations for the two fish species considered in the scenario testing study.

Simulation parameters	***Glaucosoma hebraicum***	***Rhabdosargus sarba***
Simulation projection period (years)	20	10
Number of simulation trials (recruitment series)	1	1
Number of trials per recruitment series	1	1
Sample size (number of fish)	1000	1000
Interval for length classes (mm)	50	20
Initial equilibrium fishing mortality ($years^{-1}$)	*	*
Catch curve analysis	Linear	Linear
F-based reference points		
F target (proportion of *M*)	0.667	0.667
F limit (proportion of *M*)	1	1
SPR (SSB/R) target	0.4	0.4
SPR (SSB/R) limit	0.3	0.3
SPR (E/R) target	0.4	0.4
SPR (E/R) limit	0.3	0.3
Probability of release due to bag/boat limit	0.01	0.01
Initial mean catch with bag/boat limit (number of fish)	1	2
Maximum daily catch of fishers (number of fish)	100	100
Maximum number of fishers per boat	10	10
50% effectiveness of temporal closure	0.3	0.3
95% effectiveness of temporal closure	0.8	0.8
Distribution for the numbers of fishers on boats	*# of fishers*	*% of boats*
	1	10%
	2	50%
	3	25%
	4	10%
	5	4%
	6	1%
	7-10	0%
Management parameters	***Glaucosoma hebraicum***	***Rhabdosargus sarba***
Bag limit (fish $trip^{-1}$)	4	8
Minimum legal length, MLL (mm)	500	230
Temporal closure (% of year closed to fishing)	0%	0%
Spatial closure (% of area closed to fishing)	0%	0%

* dependent on the scenario, see table 3.3

fish, estimates of mortality and plots from catch curve analysis, estimates of yield per recruit and spawning potential ratios (in terms of both spawning stock biomass per recruit and egg per recruit), and plots showing the relationship between these three outputs and fishing mortality. Note that three of the information screens presented in Chapter 2 have recently been modified to include information about the uncertainty around point estimates obtained from catch curve and per recruit analyses and to improve the way in which risk assessment information is presented by the program. The three original information screens presented to participants of the scenario testing workshops are shown in Fig. 3.1.

Several features have been built into the MSE program to make it sufficiently robust and user-friendly to be applied in workshop situations where participants may have no previous experience with the model and/or limited fisheries background knowledge. When the program is set to “scenario testing mode”, users are unable to change any of the input parameter values, and the alternative values for each management control are limited to a feasible range. Access to the various procedures and information screens is also tightly controlled so that analyses are undertaken in the correct order. To ensure that all information required for each program procedure is entered correctly, the buttons within the program interface that activate particular procedures or allow the user to navigate between program screens are only made visible when needed for the next step in the simulation procedure. To help users understand the information presented to them, help buttons have been added next to each of the boxes containing input and output parameters, explaining their meaning and relevance to the MSE procedure.

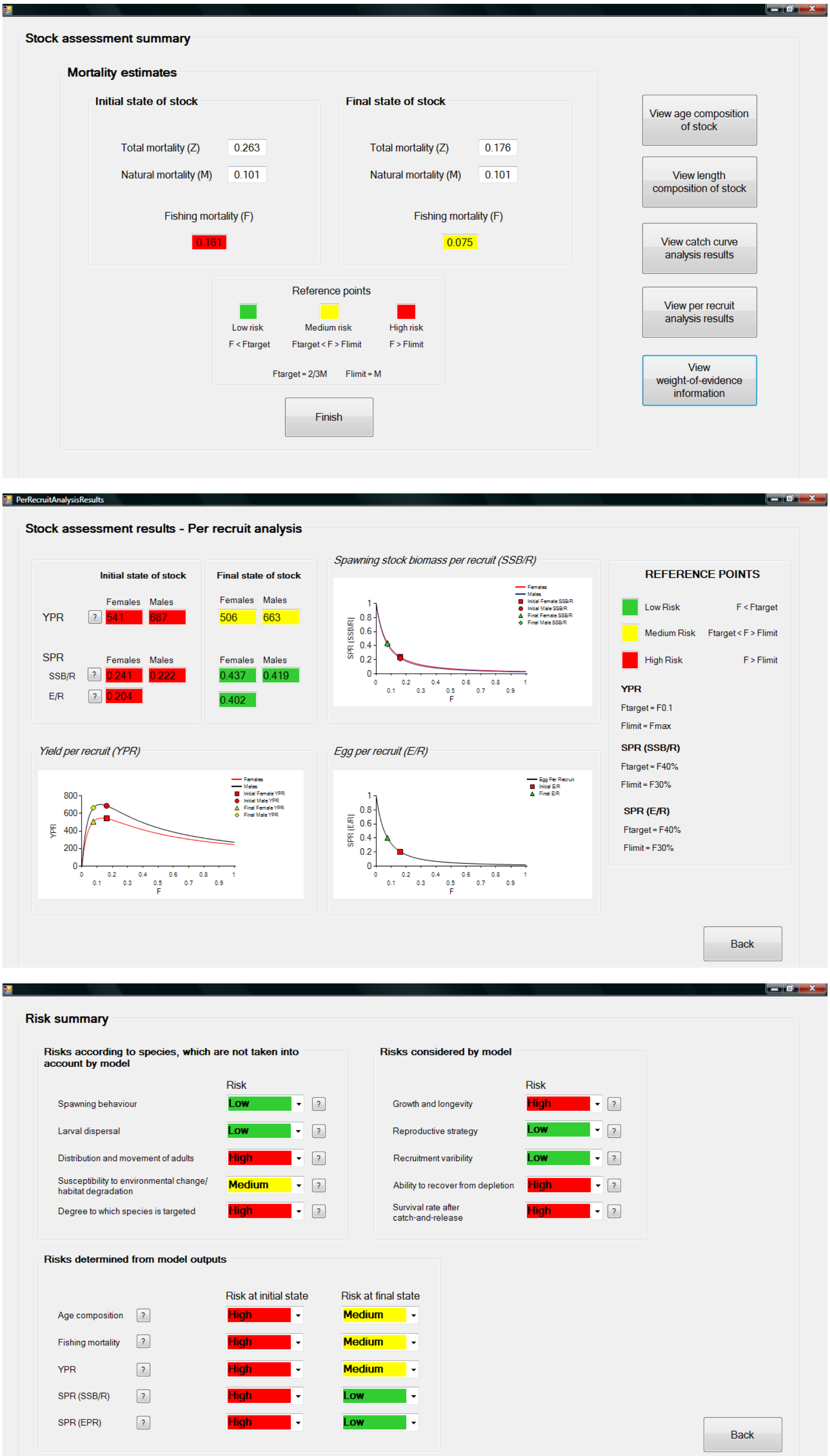


Fig. 3.1. Three of the stock status information screens that were presented to participants of the scenario testing workshops, in their original, pre-modified format.

Experimental design

To evaluate how different ways of communicating stock status information to participants influences decision-making, two alternative versions of the MSE program were used in the study. These were installed on computers located at opposite sides of the computer lab and differed only in the amount of information presented to users about the state of the fish stock. One version included a screen summarising various risks to the simulated stock, based on the outputs of the MSE program and on other risk information not considered directly by the model. These other risk categories, which were assessed subjectively (by us), included susceptibility of each species to fishing depending on their type of spawning behaviour, larval dispersal, distribution and movements of adults, susceptibility to environmental change/habitat degradation and the degree to which the fish species is targeted by fishers. This risk information is similar to that considered in the weight-of-evidence approach currently used by the Department of Fisheries, WA, for the assessment of scalefish species (Wise *et al.*, 2007).

Three different factors were considered within the 12 scenarios undertaken by each participant (Table 3.3). Each scenario represented (1) one of two fish species simulated using the operating model (see below), (2) one of three alternative initial levels of exploitation of the fish stock (low, moderate and high), and (3) one of two levels of variability in recruitment of the fish population (no recruitment variability and low recruitment variability, *i.e.* with the standard deviation of the natural logarithm of recruitment, $\boldsymbol{\sigma_{\ln R}}$, set to 0 and 0.3, respectively).

Table 3.3. Factorial design applied to scenario testing study, describing the 12 simulation scenarios completed by participants.

Scenario number	Fish species	Level of exploitation (F)	Recruitment variability ($\sigma_{\ln R}$)
Scenario 1	*Glaucosoma hebraicum*	Low (0.02)	None (0)
Scenario 2	*Glaucosoma hebraicum*	Low (0.02)	Low (0.3)
Scenario 3	*Glaucosoma hebraicum*	Moderate (0.08)	None (0)
Scenario 4	*Glaucosoma hebraicum*	Moderate (0.08)	Low (0.3)
Scenario 5	*Glaucosoma hebraicum*	High (0.16)	None (0)
Scenario 6	*Glaucosoma hebraicum*	High (0.16)	Low (0.3)
Scenario 7	*Rhabdosargus sarba*	Low (0.15)	None (0)
Scenario 8	*Rhabdosargus sarba*	Low (0.15)	Low (0.3)
Scenario 9	*Rhabdosargus sarba*	Moderate (0.35)	None (0)
Scenario 10	*Rhabdosargus sarba*	Moderate (0.35)	Low (0.3)
Scenario 11	*Rhabdosargus sarba*	High (0.55)	None (0)
Scenario 12	*Rhabdosargus sarba*	High (0.55)	Low (0.3)

The two fish species considered for the scenario testing exercise, namely West Australian Dhufish (*Glaucosoma hebraicum)* and Tarwhine (*Rhabdosargus sarba*), differ markedly in their biological characteristics. For example, the much larger-growing *G. hebraicum* can attain a maximum age of 41 years (Hesp *et al.*, 2002) whilst the shorter-lived *R. sarba* only reaches a maximum age of about 11 years (Hesp *et al.*, 2004). The order of the 12 scenarios completed by each workshop participant was randomised to ensure that they were unable to work together and that any potential effects of learning associated with continued use of the program on results of the analysis was minimised. The values of the management measures selected by the participants were logged by the program at the end of each scenario run and automatically saved to .txt files for later analysis.

Statistical analysis

Non-parametric multivariate analyses of data were undertaken using the PRIMER version 6 software package (Clarke & Gorley, 2006) with the PERMANOVA+ add-on module (Anderson *et al.*, 2008) to explore the null hypothesis of no significant differences between the suite of management decisions made by participants (1) during the two scenario testing workshops, (2) using the two alternative program versions, or (3) when undertaking the 12 scenarios. The data recorded for the 276 samples, *i.e.* values for the four management controls chosen by each participant in each scenario, were first subjected to normalisation to overcome differences in the units of measurements of the different variables of interest. A resemblance matrix, constructed between all pairs of samples and using Euclidean distance as the resemblance coefficient, was calculated from the normalised data and displayed using non-metric multi-dimensional scaling (nMDS) ordination.

Firstly, to obtain an overview of the main effects on the management decisions made by participants of the three key factors considered in the analysis (*i.e.* workshop, program version and scenario), and to detect any interactions between these, the above-mentioned Euclidean distance matrix was subjected to Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson, 2001; McArdle & Anderson, 2001). Following this preliminary analysis, a second PERMANOVA was undertaken to determine, in more detail, the influence on decision-making of the three factors considered within the scenarios undertaken by participants (*i.e.* fish species, initial level of exploitation, and level of recruitment variability of the simulated fish stock). Any potential interactions between these scenario factors were also assessed. All factors in the above analyses were considered to be fixed, and the null hypothesis of no significant differences among the various groups was rejected if

$p \leq 0.05$. The components of variation attributed to each factor considered in the PERMANOVA tests were used to determine their relative importance for explaining the overall variation in the data (Anderson *et al.*, 2008).

When PERMANOVA detected significant differences among either of the main effects, Analysis of Similarities tests (ANOSIM; Clarke & Green, 1988) were then used to examine those differences in more detail. Two-way crossed ANOSIM tests were applied when significant interactions between factors were detected, whereas one-way ANOSIM tests were used when no interactions were detected. For each ANOSIM test, the null hypothesis that there were no significant differences in the suite of management decisions among factor groupings was rejected when $p \leq 0.05$. The relative extent of any significant differences was assessed using the R-statistic, *i.e.* values close to 0 indicate little difference between groups, while those close to +1 indicate large differences between groups (Clarke & Green, 1988).

When ANOSIM pair-wise comparisons detected a significant difference between management decisions, Similarity Percentage analysis (SIMPER; Clarke, 1993) was used to determine which of the management controls (*i.e.* the bag limit, MLL, temporal closure or spatial closure) contributed most consistently to the observed effects, *i.e.* those which had relatively high dissimilarity to standard deviation ratios. Furthermore, Permutational tests of Multivariate Dispersions (PERMDISP; Anderson, 2006) were undertaken to identify any differences in the extent of dispersion within groups of samples representing the various levels of those factors for which significant differences had been detected. The null hypothesis of no difference in dispersion within different groups was rejected if the value of p was ≤ 0.05.

RESULTS

The preliminary PERMANOVA indicated that there was no significant difference between the management decisions made by participants during the two scenario testing workshops or between those using the two alternative versions of the MSE program ($p > 0.05$; Table 3.4). However, PERMANOVA did detect a significant difference between the management choices made for the different scenarios undertaken by participants ($p = 0.001$; Table 3.4). PERMANOVA detected no significant interactions between any of the three factors considered in this preliminary analysis (Table 3.4). As PERMANOVA did not detect a significant difference between the management choices of participants in the two workshops, for subsequent analyses, the data from the two workshops were pooled.

Table 3.4. Results of the preliminary PERMANOVA on the data for management decisions made by participants during the scenario testing workshops, including mean squares (MS), pseudo F-ratios, significance levels (*p*) and components of variation (COV). df = degrees of freedom. Significant results are highlighted in bold.

Factors	df	MS	Pseudo F	*p*	COV
Main effects					
Workshop (W)	1	4.347	2.575	0.060	0.020
Program version (P)	1	0.390	0.231	0.887	-0.010
Scenario (S)	11	**57.87**	**34.28**	**0.001**	**2.583**
Two-way interactions					
W × P	1	3.644	2.158	0.105	-0.030
W × S	11	1.408	0.834	0.726	-0.026
P × S	11	1.546	0.916	0.582	-0.013
Three-way interactions					
W × P × S	11	0.931	0.551	0.977	-0.139
Residual	228	1.688			1.688

The second PERMANOVA undertaken to further explore differences in decision-making among scenarios undertaken by participants showed significant differences between the management choices made for the two fish species ($p = 0.001$, Table 3.5) and for the different initial levels of exploitation for the simulated stock ($p = 0.001$, Table 3.5). No difference was detected between management decisions made for scenarios with different levels of recruitment variability ($p > 0.05$, Table 3.5). PERMANOVA demonstrated a significant interaction between species and level of exploitation ($p = 0.001$; Table 3.5). However, the relatively low components of variation for that interaction term indicates that it accounted for far less of the total variability in the data set than did differences in species and level of exploitation (Table 3.5).

Table 3.5. Results of PERMANOVA on the data for management decisions made by participants for the different scenarios undertaken, including mean squares (MS), pseudo F-ratios, significance levels (*p*) and components of variation (COV). df = degrees of freedom. Significant results are highlighted in bold.

Factors	df	MS	Pseudo F	*p*	COV
Main effects					
Species (S)	1	**362.5**	**218.4**	**0.001**	**2.615**
Level of exploitation (F)	2	**132.7**	**79.93**	**0.001**	**1.424**
Level of recruitment variability (R)	1	3.436	2.070	0.103	0.013
Two-way interactions					
S × F	2	**10.62**	**6.397**	**0.001**	**0.195**
S × R	1	0.961	0.579	0.602	-0.010
F × R	2	3.097	1.866	0.086	0.031
Three-way interactions					
S × F × R	2	1.039	0.626	0.687	-2.698
Residual	264	1.660			1.660

The nMDS plots provided good two-dimensional representations of the similarities and dissimilarities between the different factor groupings, *i.e.* all stress values were 0.09, indicating ordinations with minimal distortion of data in order to fit the required dimensions (Clarke & Warwick, 2001). As was also confirmed by the preliminary PERMANOVA (Table 3.4), there was no separation of the management decisions made by workshop participants using the two different model versions (Fig. 3.2). However, when management decisions were separated according to the different factors considered for the fishery scenarios, nMDS plots showed distinct clustering of data points for the two different fish species, and for the three different levels of initial exploitation of the fish stock (Fig. 3.3a,b). In contrast, when scenarios were separated according to the two different levels of recruitment variability specified for the simulated fish stocks, there was no tendency for data points to cluster (Fig. 3.3c).

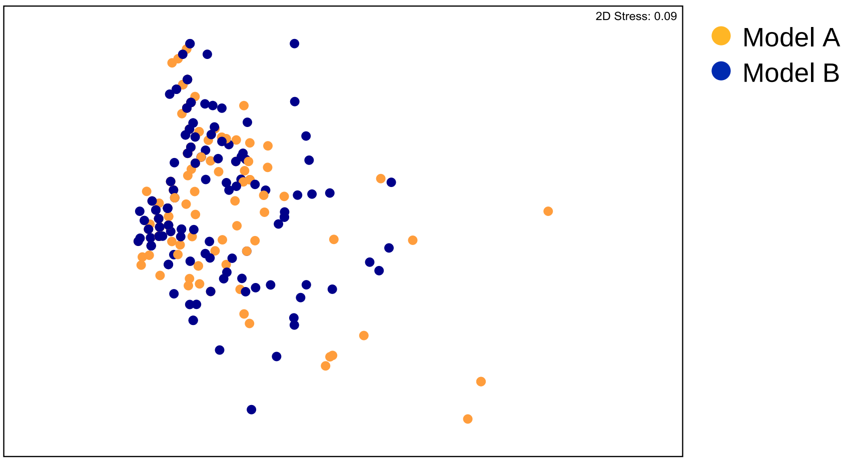
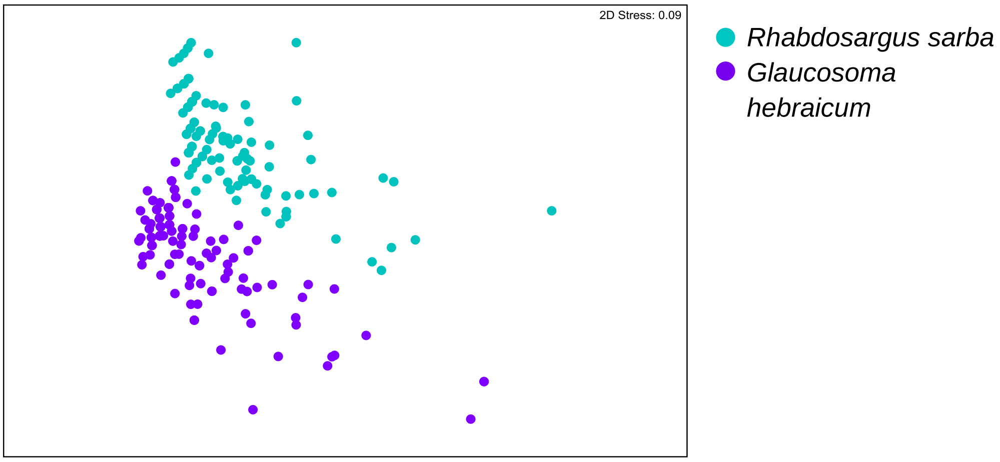
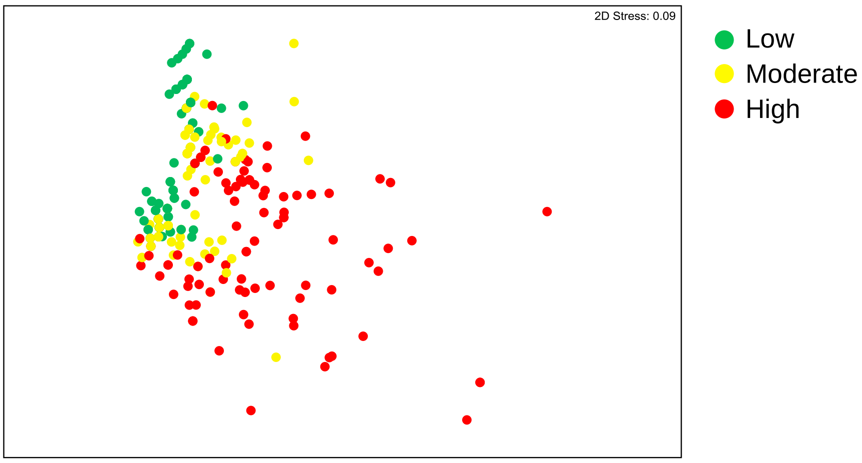


Fig. 3.2. nMDS ordination plot of management decisions made by users of the two alternative versions of the MSE program, with Model A presenting the additional weight-of-evidence summary screen, and Model B displaying outputs from the model only.

(a) Fish species



(b) Level of exploitation



(c) Recruitment variability

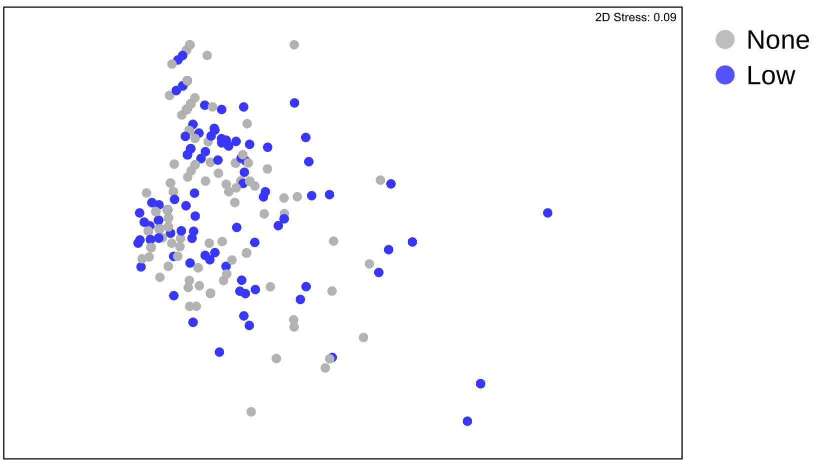


Fig. 3.3. nMDS ordination plots of management decisions made by participants for the different scenarios, when separated by (a) fish species, (b) level of exploitation, and (c) recruitment variability.

Two-way crossed ANOSIM tests confirmed that, when comparing management decisions made for scenarios, significant differences were detected for the two fish species (p = 0.001, R = 0.77), and for the different levels of exploitation (p = 0.001, R = 0.37). Pair-wise comparisons of management choices made by participants for scenarios of the three different exploitation levels were all significant (p = 0.001), with values for R ranging from 0.24 (when comparing scenarios for low and moderate levels of exploitation) to 0.58 (when comparisons were made between scenarios for low and high levels of exploitation).

For the two significant factors, *i.e.* fish species and initial level of exploitation, two-way crossed SIMPER analysis employing these factors showed that the MLL control made the most consistent contribution to the average dissimilarity between decisions made for the two fish species, followed by that for the bag limit control. When scenarios for low *vs* moderate and low *vs* high initial exploitation were compared, SIMPER showed that the control consistently most responsible for the observed differences in management decisions was again the bag limit. For the comparison between scenarios for moderate and high exploitation, SIMPER showed that the four management controls made similar contributions to the observed differences in management decisions.

PERMDISP showed that the levels of dispersion among management decisions made by participants undertaking scenarios for the two fish species were similar (p > 0.05, t = 0.309). This was also the case between decisions for scenarios of low and moderate initial levels of exploitation (p > 0.05, t = 0.329). However, decisions were significantly more varied for scenarios for initially heavily exploited fish stocks compared with lightly exploited stocks (p = 0.001, t = 6.404) and moderately exploited stocks (p = 0.001, t = 5.903), as also shown by the nMDS plot (Fig. 3.3b).

DISCUSSION

Effectiveness of the program for communicating stock assessment information

The results of this study showed that the management decisions made by the workshop participants (science students at Murdoch University) exhibited a strong tendency to vary between scenarios. In particular, participants made different decisions between the scenarios for the two fish species and those for different initial levels of exploitation. The management choices made by participants during the workshops for the different scenarios were logical, indicating that the program interface was effective for communicating stock assessment information to the workshop participants. For example, the results showed that decisions varied far less when exploitation was low (*i.e.* when the stock was in a healthy state) whereas, as would be expected, decisions were far more variable in situations where there was a strong need for management changes to reduce exploitation. Given that the workshop participants received limited training in how to use the model before undertaking the scenario testing exercise, and as many would have had limited prior knowledge of fisheries stock assessment, this suggests that the program has potential for communicating stock status information to people of varying background. Focus is next turned to exploring which factors are likely to be important in developing software for conveying stock assessment information and how the design of the scenario testing experiment and program interface may have influenced the management decisions made by the workshop participants.

Important model design features for developing effective communication tools

As pointed out by Mathevet *et al.* (2007), when designing a computer simulation model for communicating information to stakeholders it is important to find an appropriate balance between simplification and realism. For complex systems, simplification is often required to facilitate stakeholder understanding of the various processes that influence the dynamics of the system. Realism (*i.e.* in terms of how well the model represents the system and the data used to inform the model) is equally important to the process as it allows stakeholders to project their newly acquired understanding back into reality (Mathevet *et al.*, 2007).

To simplify use of the MSE program and make it more robust for use in stakeholder and educational workshops, various features have been added to the program. A number of these features restrict certain user “freedoms” when exploring the program. Thus, we disallowed access to change input data such as values of the biological parameters of the fish species, the initial management measures to which stocks were subjected prior to simulations, and parameters that specify other aspects of the simulations. We also limited the values that users could select for the different management controls to feasible ranges (*e.g.* a spatial closure can only constitute between 0 and 100% of the total area). Finally, we restricted the route by which participants could navigate through the program interface to a single pathway, thereby ensuring that they activated each of the various program procedures in the correct order and viewed all of the available stock assessment information.

In this study, we used “traffic light” colour indicators to draw the attention of participants to key stock assessment results and help communicate the significance of these for the conservation status of the fish stock. The use of reference point-based traffic light indicators was originally developed by Caddy (1998; 1999) as a means of developing management approaches for those fisheries managers in developing countries with relatively limited expertise in population modelling (see also Caddy, 2002). As MSE is inherently complex, visual aids, such as the use of colour when presenting information, is likely to be particularly important for helping stakeholders without technical or scientific backgrounds interpret the risk implications of different stock assessment results.

Another feature that is likely to be important for facilitating understanding of the relevance of the various program parameters and outputs is the way in which "help" information is presented to users. In contrast to most other software, help buttons are distributed throughout the various screens of the interface adjacent to each parameter of interest. We elected to avoid making help information accessible only through a program menu search to eliminate the need for participants to check whether help was available for a particular aspect of the program through scrolling through a list and/or entering in a "search term" to find the relevant information.

Potential factors influencing decisions of participants

Although the results of the scenario testing experiment suggest that the program was effective in communicating stock assessment information, questions are raised as to the extent to which various factors may have influenced the decisions of the participants. For example, our efforts to simplify the use of the software by fixing the route by which participants can navigate through the program meant that stock assessment information was always accessed in a particular order, which potentially could have influenced the decisions made. It is thus possible, for example, that the additional weight-of-evidence information screen presented by one of the program versions had limited impact on decisions because the screen containing that information was presented last when participants may have already made up their minds based on previously displayed information.

A second potentially important factor influencing decisions was that the initial management arrangements displayed to participants prior to each simulation differed for the two fish species (*e.g.* the MLL of 500 mm for *G. hebraicum* was greater than that of 230 mm for *R. sarba)*. Although the different starting points for management of the two species undoubtedly influenced the decisions made, it would not have made sense to specify the same starting values for the MLL for the two species given that they attain very different maximum lengths. The alternative of not specifying any values for initial management controls would not have been realistic because most fish species are already subject to some particular management regime.

A third factor that would have influenced the decisions of the participants was the objective specified for the scenario testing exercise, *i.e.* for the stock to be in a “healthy” but productive state, as indicated by various reference points. Participants were continually reminded of this objective through the display of traffic light colour indicators for the various stock assessment outputs. The program further highlighted the objective by providing participants a score at the end of each scenario, calculated according to how close the final state of the exploited stock was to the target reference point for fishing mortality. Furthermore, if exploitation was well below the target reference point for mortality (F = 2/3M), participants were advised that the fishery was not productive, whereas if exploitation exceeded the limit reference point (F = M), feedback was provided that the stock was over-fished. As pointed out by Bertsche *et al.* (1996), an important element of computer simulation exercises is the need for participants to feel challenged, so that at the end of each simulation they will not want to stop but rather continue to test the consequences of different actions and improve on their performance. Thus, although the feedback provided to participants may have influenced their decisions, this element is important for capturing and maintaining interest. Future work aimed at exploring the various factors that most influence decision-making of program users could focus on further scenario testing experiments in which certain factors discussed in this section of the discussion are manipulated.

Implications of scenario testing results for fisheries management and MSE

The results of the scenario testing experiment indicated that of the four available management controls (bag limit, MLL, temporal closure and spatial closure), workshop participants most consistently made changes to existing bag limits. They also suggested that, when participants perceived the stock to be heavily fished, they tended to apply a wider range of management changes, *i.e* as indicated by the different levels of dispersion of points on the nMDS plots for scenarios of different levels of fishing pressure, and PERMDISP results. Given that the workshop participants had limited (or no) prior experience in fisheries management and, being university students, may have had views regarding conservation that differ from people within the fishing industry and/or the broader community, to what extent might their decisions reflect the types of decisions made in real life situations?

Taking the recent experience in Western Australia of the recreational fishery for demersal scalefish in the West Coast Bioregion, for example, management initially consisted of bag and boat limits and minimum legal lengths for retention. However, as exploitation of these stocks became increasingly heavy, existing bag and boat limits were tightened and a temporal closure and other measures, such as compulsory use of release weights, were introduced (Department of Fisheries, Western Australia, 2010). Although the decisions made by participants in the workshop resembled, to some extent, the pattern experienced by this particular fishery, the factors that influence management decisions in real life are complex and involve consideration of multiple and often conflicting objectives. Thus, in addition to the extent to which management changes may be needed (and recognised by all stakeholders), other factors such as the extent to which different stakeholders accept, or are opposed to, each of the different types of controls, will inevitably influence the types of management choices made by decision-makers. The influence of conservation and community groups on government policy regarding Marine Park planning, for example, is another factor influencing fisheries management.

The tendency for variability in management decisions among participants to increase as the initial stock state worsened is likely to reflect an increased difficultly for participants to predict the likely effects of management changes when a broader range of controls are used. Indeed, feedback from students who participated in the workshops indicated that they preferred to use controls that they thought they best understood and only tended to use the other (closure) controls when they considered that the situation demanded further action. It would appear likely that, to a certain extent, these decision-making behaviours do translate to real life situations. Obviously, as management arrangements become increasingly complex, the risk of unexpected consequences as a result of increased difficulty in being able to accurately predict their effects, will also increase. This point emphasises one of the great values of simulation in fisheries management, *i.e.* to explore the effectiveness of alternative combinations of management controls and learn from mistakes in a risk-free, simulated environment (Bertsche *et al.*, 1996).

Value of scenario testing for facilitating stakeholder discussion and education

Lack of effective communication between scientists and other stakeholders has repeatedly been highlighted as one of the greatest challenges to successful fisheries management (de la Mare, 1998; Peterman, 2004). In this regard, MSE could provide a valuable tool for facilitating increased stakeholder participation in fisheries decision-making (Smith *et al.*, 1999). This stems from the fact that simulation modelling has shown to be particularly valuable for problem solving in situations that involve many people or organisations whose actions need to be coordinated (Bertsche *et al.*, 1996). In particular, it is recognised that by providing a vehicle for engaging industry in decision-making, simulation models can also play an important role in facilitating greater stakeholder understanding and trust in the fisheries management process. The approach is likely to also be vital to fisheries co-management initiatives, where government administrators act more as arbiters among interests within the general public than as decision-makers in the public interest (Beierle & Cayford, 2002). Indeed, it has been widely accepted that inclusion of stakeholders in fisheries decision-making can lead to better management outcomes by increasing the efficiency of enforcing regulations through a higher level of compliance (Jentoft & McCay, 1995; Mikalsen & Jentoft, 2001).

Given the great potential of MSE for facilitating more informed, co-operative and robust fisheries management decision-making, it is a concern that its use is being limited by the fact that stakeholders and managers with non-science or non-technical backgrounds often struggle with its complexity (Rochet & Rice, 2009; Smith *et al.*, 1999). However, indications from our scenario testing results are that, if well designed, simulation models based on the concept of MSE can be effective for conveying stock assessment outputs to an audience with widely varying backgrounds. We suggest that the approach taken in this study to develop our model, with the intent purpose of testing its effectiveness for communicating relatively complex information from stock assessments to people with little or no experience in stock assessment, is of great value when developing software for stakeholder use. We also conclude that, although it is possible to maintain model sophistication and the level of complexity needed for robust analyses, modification of various aspects of the interface when applying the model to workshop situations allows users to more easily use the software and interpret the results in a manner more intuitive to non-scientists.

CHAPTER 4: Implications of sampling design and species biology on the effectiveness of mortality-based stock assessments and management: a Monte Carlo simulation study

INTRODUCTION

A critical component of fisheries stock assessment is estimation of the instantaneous rate of total mortality, *Z* (*e.g.* Smith, 1990; Smith *et al.*, 2009). For fisheries where extended time-series of catch and effort data or abundance indices are not available, such as for many recreational fisheries, stock assessments typically focus on catch-at-age analyses based on age composition data for samples of fish (Deriso *et al.*, 1985; Jensen, 1985; Carlile, 2005). One of the most common approaches for estimating *Z* from age composition data is catch curve analysis. A simple and very widely used form of catch curve involves fitting a linear regression line to the logarithms of the frequencies-at-age of fish in samples for all ages above that at which individuals are fully recruited into the fishery (Ricker, 1975). In this analysis, it is assumed that the negative of the slope of the regression line corresponds to the instantaneous rate of total mortality (Ricker, 1975; Smith, 1990).

There are a number of different approaches to catch curve analysis and views among scientists about the appropriateness of the different methods are varied. Simple catch curve methods like that of Ricker (1975) have frequently been criticised because of the significant assumptions required about the underlying data, which can be extremely difficult to satisfy (Deriso *et al.*, 1985; Hilborn & Walters, 1992; Dunn *et al.*, 2002). Particularly challenging are assumptions relating to the population being in a steady state, such as that recruitment and mortality are constant with respect to time and age (Ricker, 1975; Jensen, 1984; Murphy, 1997). However, because of their simplicity and data limitations for many fisheries (*e.g.* lack of abundance data), these approaches are still widely applied to many present day stock assessments for recreationally and commercially important fish species (Grandcourt *et al.*, 2008; Smith *et al.*, 2008; Griffiths, 2010). Researchers have attempted to deal with the issues of steady-state assumptions for catch curve analysis in various ways. For example, for species with high inter-annual recruitment variability, sample data collected over multiple years are often pooled to “smooth out” peaks and troughs in the age frequency distributions, thereby potentially reducing its impact on the analysis (Seber, 1973; Ricker, 1975). Others have explicitly accounted for recruitment variability by applying catch curve approaches that deal with individual year classes (*e.g.* Deriso *et al.*, 1985).

It has been frequently recognised that stock assessments based on age-structured catch data are highly dependent on the representativeness of these data (Sullivan *et al.*, 1994), yet this issue is often not well addressed. Although it is widely accepted that the precision of age data depends to a large extent on sample size (*e.g.* Campana, 2001), the high cost and significant resources required to sample exploited fish stocks will limit the amount of data available for stock assessment (Chen, 1996). To optimise the cost-effectiveness of age-based monitoring programmes, it is valuable to predict the likely effectiveness of different sampling strategies for yielding reliable mortality estimates (Aanes & Pennington, 2003). As the effectiveness of such monitoring programmes are likely to vary depending on the species and number of age classes present in samples (Brouwer & Griffiths, 2005), a tool enabling such analyses to be readily repeated for different situations would be of substantial value.

Recreational fisheries now account for a substantial component of the total retained catch of fish in many parts of the world (*e.g.* McPhee *et al.*, 2002; Cooke & Cowx, 2004). Unlike most commercial fisheries, recreational fisheries are generally open access (McPhee *et al.*, 2002) and this has important implications for management. To regulate catches, management of recreational fisheries usually relies on restricting the number of fish that are allowed to be retained by individual fishers (bag limits) and on boats (boat limits) during a fishing trip (Woodward & Griffin, 2003). Together with size limits such as minimum legal lengths for retention, and temporal and spatial closures, these management controls provide the most common forms of regulations for recreational fisheries (Woodward & Griffin, 2003).

As bag, boat and size limits can lead to substantial numbers of captured fish being released by fishers (because the maximum daily allowable limit for a species has been reached and/or because captured fish are too small to be legally retained), post-release mortality is likely to strongly impact on the effectiveness of these management controls (Woodward & Griffin, 2003). Although available evidence suggests that post-release mortality in some fish species can be substantial (*e.g.* Harley *et al.*, 2000; St John & Syers, 2005; Rudershausen *et al.*, 2007), few studies have evaluated the implications of this on the performance of management in recreational fisheries (Anderson, 1993; Homans & Ruliffson, 1999; Woodward & Griffin, 2003). To help ensure successful management outcomes, it would thus be of great benefit to understand its impact on the effectiveness of a range of commonly applied management controls for regulating exploitation.

The main aims of this study were to use Monte Carlo simulation methods to:

(1) Determine the influence of sample size and sampling frequency on the effectiveness of two alternative equilibrium-based approaches to catch curve analysis for producing reliable estimates of mortality. We have investigated the efficacy of these stock assessment approaches for two species of fish that differ markedly in biology, and explored the consequences of assuming two different levels of recruitment variability for these species.

(2) Explore the effectiveness of four different fisheries management controls common to recreational fisheries, *i.e.* a bag/boat limit, minimum legal lengths for retention (MLL), and temporal and spatial closures, for conserving the two fish species when low or high recruitment variability is assumed.

MATERIALS AND METHODS

Overview of simulation procedures

Monte Carlo simulation methods were employed to (1) explore the influence of sampling design on the effectiveness of two alternative forms of catch curve analysis for estimating mortality and (2) evaluate the effectiveness of four alternative management controls for regulating exploitation of fish stocks. All computer simulations were undertaken using the management strategy evaluation (MSE) model developed in this study (see Chapter 2; Appendix 3). Simulations were conducted for two fish species that differ markedly in biology, namely the West Australian Dhufish (*Glaucosoma hebraicum)* and Tarwhine (*Rhabdosargus sarba*). The former of these species is large (maximum length of ~1.2 m) and can live for over 40 years (Hesp *et al.*, 2002), whereas the latter is far smaller (maximum length of ~400 mm) and lives to only about 11 years of age (Hesp *et al.*, 2004). For each fish species, simulations were repeated for two levels of recruitment variability, *i.e.* a low level (by setting the standard deviation for the natural logarithm of recruitment, $\sigma_{\ln R}$, to 0.3), and a moderately high level ($\sigma_{\ln R}$ = 0.6), assuming the same level of autocorrelation in all simulations (correlation for 1 year lag = 0.3).

Influence of sampling design and species biology on the effectiveness of catch curve analysis

Analyses exploring the influence of sample size and sampling frequency on the effectiveness of two alternative forms of catch curve analysis for estimating mortality involved: (1) using the MSE operating model to simulate the dynamics of exploited populations of the two fish species, (2) employing the MSE sampling model to generate multiple random age composition samples for the two species, (3) fitting each of the two different catch curves to the data to yield estimates of total mortality, *Z*, (4) estimating the level of fishing mortality, *F*, by subtracting the point estimate for natural mortality, *M*, (derived using Hoenig's (1983) equation for estimating *M*) from the catch curve estimates for *Z*, and (5) comparing the estimated values for *F* to the specified (true) values of this parameter for each species (F = 0.1 year^{-1} for *G. hebraicum* and 0.4 year^{-1} for *R. sarba*).

Age composition samples for each of the two species were generated by sampling from the expected numbers of fish at age in the simulated population (assuming a multinomial distribution). The results of each analysis undertaken to determine the influence of the different factors considered (*i.e.* sample size, sampling frequency, fish species and recruitment variability level) on the effectiveness of catch curve analysis were based on 100 sets of age composition samples generated with different recruitment series. To ensure that random effects did not influence the results of any comparisons, the same 100 recruitment series were used for all factors by controlling the random number generator sequences applied to generate the age composition samples.

The effectiveness of two equilibrium-based catch curve methods for estimating mortality were explored. The first (CC1) was the linear catch curve of Ricker (1975), which assumes constant annual recruitment and constant mortality for all fish in the population above the age at full recruitment into the fishery (Fig. 4.1a). The second method (CC2) was relative abundance analysis (Deriso *et al.*, 1985, see also Hall *et al.*, 2004), which is an extended form of catch curve analysis that accounts for variable recruitment between years. As with CC1, constant mortality is assumed for all fully recruited fish (Fig 4.1b).

CC1 was fitted to 100 sets of simulated age composition samples of 500, 1000, 1500, 2000, 2500 and 3000 fish, collected during the year immediately prior to assessing the stock in its initial equilibrium state. The simulations were then repeated for all sample size categories, but fitting the catch curve to sample data collected during the two years, and then three years preceding the assessment (for the stock in its initial equilibrium state). For example, when a total sample of 1000 fish was drawn over two consecutive years, 500 samples were drawn from each of the ultimate and penultimate years leading up to the assessment. Note that when multiple years of data were used, the age samples for the different years were pooled before fitting the curve. CC2 can be fitted only if multiple years of age composition data are available and therefore this catch curve was fitted only to samples generated over two or three consecutive years. Although these multi-year data sets

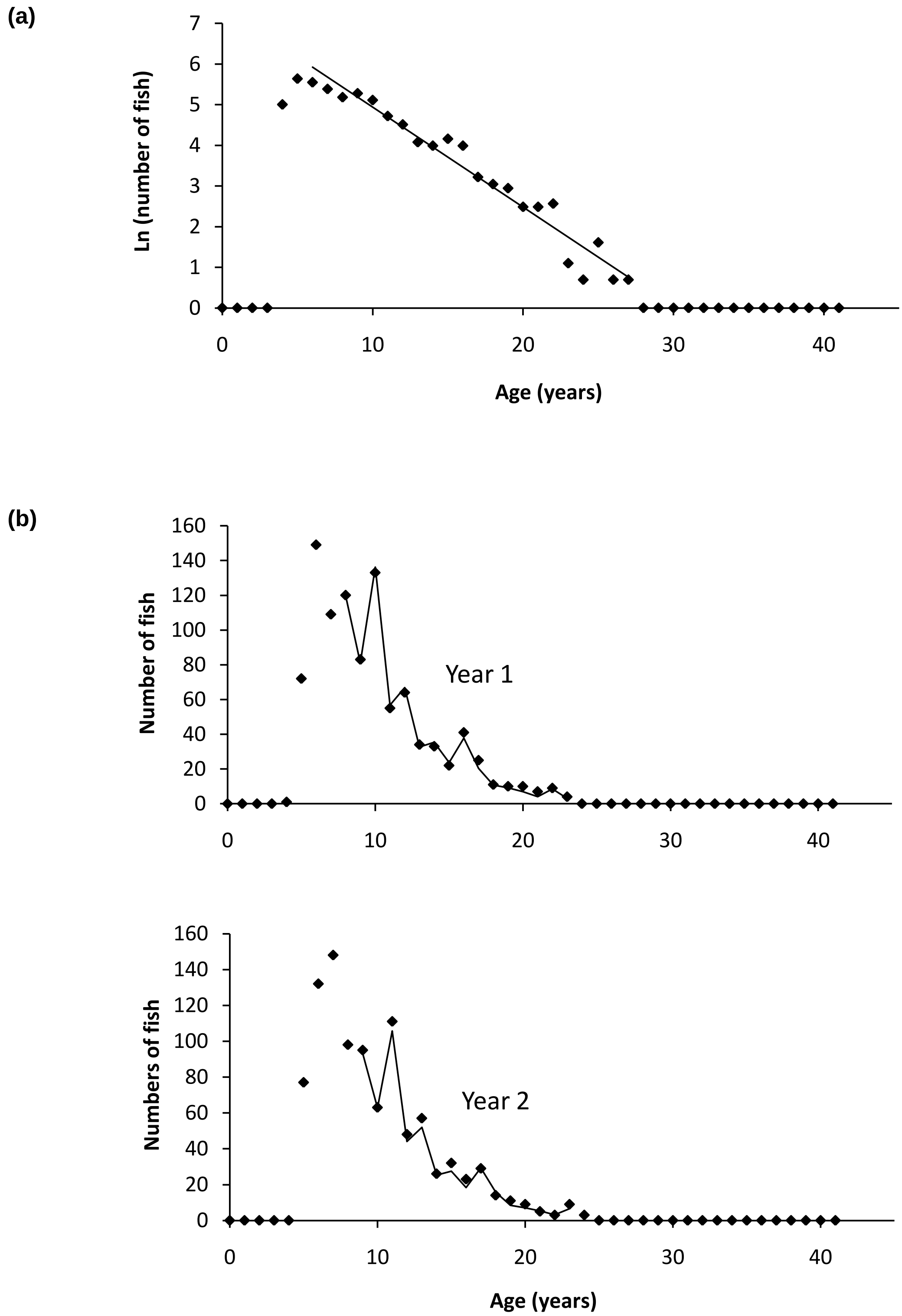


Fig. 4.1. The two catch curve approaches explored in this study; (a) the linear catch curve analysis (Ricker, 1975), *i.e.* CC1, and (b) the relative abundance analysis (Deriso *et al.*, 1985), *i.e.* CC2. Note that both catch curves have been fitted to the same age composition data except that, for CC1, data sampled in different years have been pooled whereas they have been kept separate for CC2.

were the same as those generated for simulations with CC1, CC2 was fitted simultaneously to the separate data for the different sampling years (rather than to the data after it had been pooled for the different years, as for CC1).

For every set of 100 simulations undertaken for the various factors being considered, the median values for the *F* estimates produced by the catch curve analyses were calculated and plotted against sample size, together with the lower and upper quartiles, and minimum and maximum values (*i.e.* box and whisker plots). The accuracy and precision of the various *F* estimates were assessed by calculating the relative bias, *B*, and the coefficient of variation, *CV*, as

$$B = \bar{x} - x'/\bar{x}$$

$$CV = \bar{x}/\sigma$$

where $\bar{x}$ is the mean of the 100 estimates for *F* produced by the catch curve analysis for each factor, x' is the corresponding true value for fishing mortality (as calculated by the operating model) and σ is the standard deviation for the estimated *F*.

Influence of species biology on the effectiveness of management controls

Simulations exploring the effectiveness of alternative management controls were undertaken for both fish species (*G. hebraicum* and *R. sarba*) and for the same two levels of recruitment variability considered above (low and moderately high). Four different management controls, all which are common to recreational fisheries, were evaluated; (1) a bag and boat limit, (2) a minimum legal length for retention, MLL, (3) a temporal closure, and (4) a spatial closure. See Appendix 3 for a detailed description of how each control has been implemented in the model. For each fish species and level of recruitment variability, a base case scenario was specified to enable the population dynamics of each fish stock at an initial level of exploitation and management regime to be simulated. These base case scenarios represented the fish stocks experiencing a high level of fishing pressure but with some existing protection through fisheries regulation (*i.e.* bag/boat limits and a MLL for retention). The parameters used for the simulations are specified in Table 4.1 (see also Table 3.1 for values of biological parameters for the two fish species).

To evaluate the effectiveness of each management control, 100 simulations were undertaken for each of a range of specified values for that control, keeping values for all other controls the same as specified for the stock in its initial state (*i.e.* as those specified for the base case scenario). The effect of each management change on the true rate of mortality being experienced by the stock was assessed 10 years after the change had been implemented. For each fish species and level of recruitment variability, the state of the stock during the initial period was determined using a single generated set of recruitment strengths for the different year classes present in the stock. The potential influence of recruitment variability on the effectiveness of the various controls for regulating fishing mortality during the projection period was considered by generating 100 different sets of recruitment series for fish "born" during that period.

For each set of 100 simulation runs, the median values (and other statistics) of the true level of fishing mortality for the exploited stock at its final state (*i.e.* at the end of the 10 year projection period) were calculated and graphed. For the MLL and spatial closure controls, the total numbers of fish killed (*i.e.* all fish above and below the age at full recruitment into the fishery) in the final year of the projection period were also

Table 4.1. Parameters used in Monte Carlo simulations to explore the effectiveness of different management controls for regulating exploitation of the two fish species.

Parameters	***Glaucosoma hebraicum***	***Rhabdosargus sarba***
Simulation period (years)	10	10
Number of simulation trials (recruitment series)	1	1
Number of trials per recruitment series	100	100
Initial equilibrium fishing mortality (year^{-1})	0.2	0.8
Initial mean catch with bag/boat limit (fish trip^{-1})	1	5
Initial management controls		
Boat/bag limit (fish trip^{-1})	5	20
Minimum legal length, MLL (mm)	500	240
Temporal closure (months of year closed to fishing)	0	0
Spatial closure (% of area closed to fishing)	0	0

calculated and plotted. Note that, for this second component of the study focusing on the effectiveness of the different management controls, catch curve analysis was not employed because only the true values of fishing mortality being experienced by the fish stocks (*i.e.* as calculated by the operating model) were of interest.

RESULTS

Influence of sampling design and species biology on the effectiveness of catch curve analysis

Linear catch curve – low recruitment variability

For *Glaucosoma hebraicum*, the median value for the 100 estimates for fishing mortality, F, calculated by fitting a linear catch curve (CC1) to an age composition sample of 500 fish, collected in the year immediately prior to the initial assessment, was similar (0.091 year^{-1}) to the true value of F for the population (0.1 year^{-1}) (Fig. 4.2a). Likewise, the middle half of the observations (*i.e.* those between the upper and lower quartiles) were always relatively close to the true value. The same situation was true when CC1 was fitted age composition samples of 1000, 1500, 2000, 2500

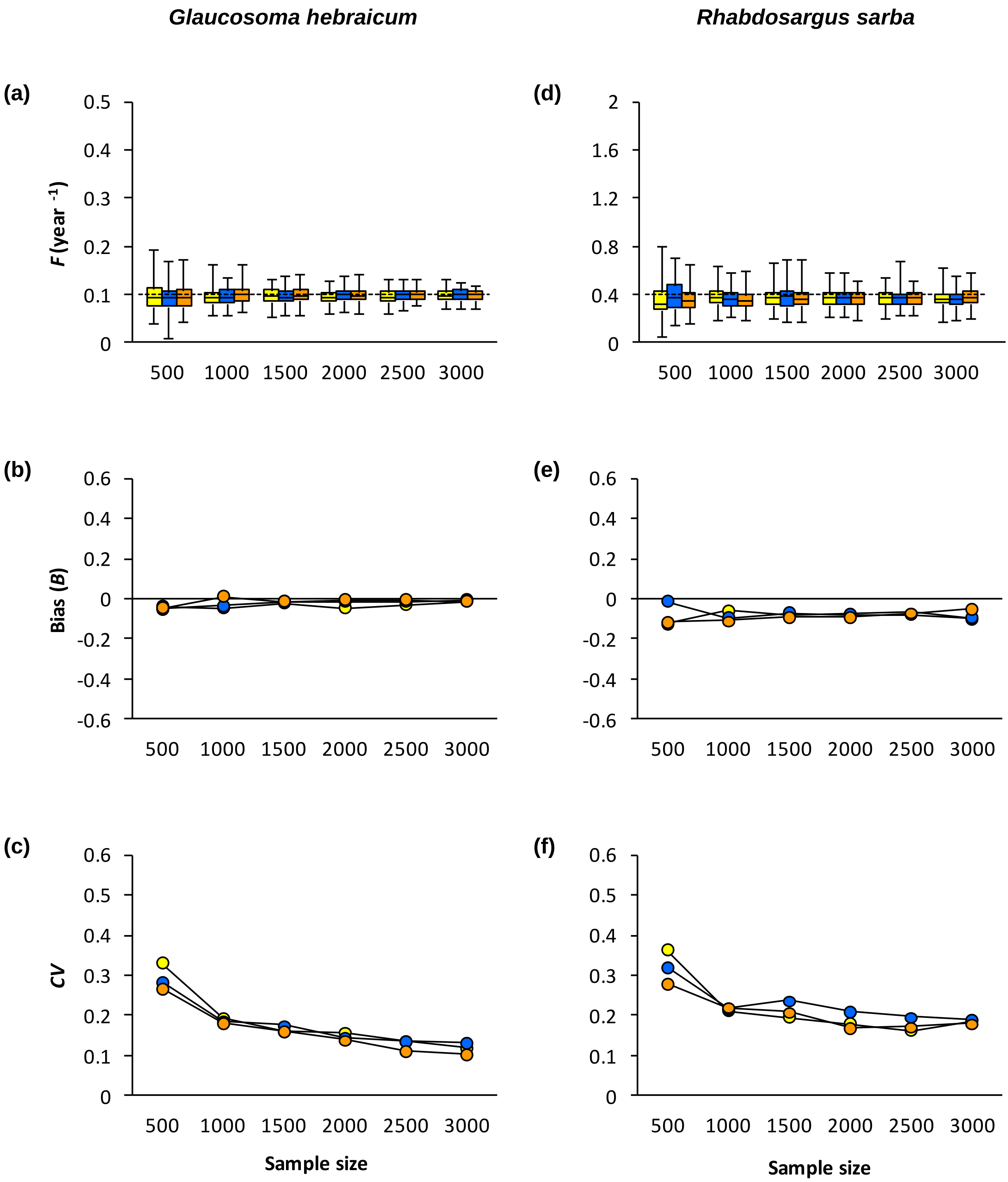


Fig. 4.2. Influence of sample size and sampling frequency (1 yr – yellow, 2 yrs – blue, 3 yrs – orange) on (a,d) estimates of fishing mortality, *F*, (b,e) relative bias, *B*, and (c,f) *CV* for the two fish species when recruitment variability is low, using linear catch curve (CC1). In a and d, the lower and upper bounds of each box represents the lower and upper quartiles of *F* estimates, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively.

and 3000 fish. As sample size increased, the range of values for the estimates of *F* tended to decline. For example, when CC1 was fitted to an age composition sample for 500 fish, estimates ranged between 0.042 and 0.194 year^{-1}, whereas when the sample size was increased to 1500 fish, they ranged between 0.556 and 0.134 year^{-1} (Fig. 4.2a). Spreading age composition samples over either two or three years did not lead to a conspicuously different trend to that observed when samples were taken from a single year (Fig. 4.2a). The similarity between the *F* estimates and the true value of *F* for *G. hebraicum* was reflected in low values for bias (ranging from -0.051 to 0.012) for all sample sizes and sampling frequencies (Fig. 4.2b). The *CV* values demonstrated that precision increased substantially as sample size increased from 500 to 1000 fish, and to a lesser extent with each successive increase in sample size thereafter (Fig. 1c). Sampling frequency had little effect on *CV* (Fig. 4.2c).

With *Rhabdosargus sarba*, CC1 always slightly underestimated *F*. For example, when fitted to samples of 500 and 3000 fish collected in one year, the respective median values for the *F* estimates (0.320 and 0.356 year^{-1}) were 20 and 11% less than the true value (0.4 year^{-1}) (Fig. 4.2d). As with *G. hebraicum*, the range of the estimated values for *F* (0.050 to 0.807 year^{-1}) was greatest when CC1 was fitted to the smallest age composition samples, *i.e.* for 500 fish (Fig. 4.2d). The tendency, for CC1 to yield underestimates of *F* for *R. sarba* regardless of sample size is reflected in the values for bias (based on the mean) always being negative (Fig. 4.2e). As with *G. hebraicum*, the *CV* declined conspicuously when sample size was increased from 500 to 1000 fish but showed little tendency to decline further with increasing sample size thereafter (Fig. 4.2f). Although, at each sample size, the estimates of *F* for *R. sarba* varied far more about the median value compared to *G. hebraicum*, (*cf.* Figs 4.2a,d), the *CV* values for the former species were only slightly higher (*cf.* Figs 4.2c,f). As with *G. hebraicum*, different sampling frequencies had little effect on the accuracy and precision of mortality estimates except, to some extent, on precision at the smallest sample size (Fig 4.2).

Linear catch curve – high recruitment variability

When recruitment variability of *G. hebraicum* was high and CC1 was fitted to age composition samples for 500 and 1000 fish collected in one year, the median values for the *F* estimates (0.078 and 0.087 year^{-1}, respectively) were considerably less than the true value of 0.1 year^{-1} (Fig. 4.3a). At these sample sizes, the *F* estimates became closer to the true value for *F* when samples were distributed over two years (0.087 and 0.093 year^{-1}, respectively) or three years (0.090 and 0.092 year^{-1}, respectively). Likewise, the bias values show that, when recruitment variability was high, CC1 estimated mortality more accurately when samples were distributed samples over multiple years (Fig. 4.3b). Thus, for example, at samples sizes of 500, 1500 and 3000 fish, the bias of the *F* estimates based on one year of data was always greater (0.268, 0.081 and 0.067, respectively) than when the samples were spread over two (0.100, 0.017 and 0.016) or three years (0.016, 0.033 and 0.008) (Fig. 4.3b). For each sample size category, doubling the level of recruitment variability in *G. hebraicum* led to the precision in mortality estimates produced by CC1 being approximately halved (*cf.* Figs 4.2c and 4.3c).

As when recruitment variability was low, CC1 also tended to underestimate *F* for *R. sarba* when the level variability was high (Fig. 4.3d). The extent of the negative bias was slightly greater when recruitment variability was doubled (*cf.* Figs 4.2e and 4.3e). As with *G. hebraicum*, when recruitment variability of *R. sarba* was high and

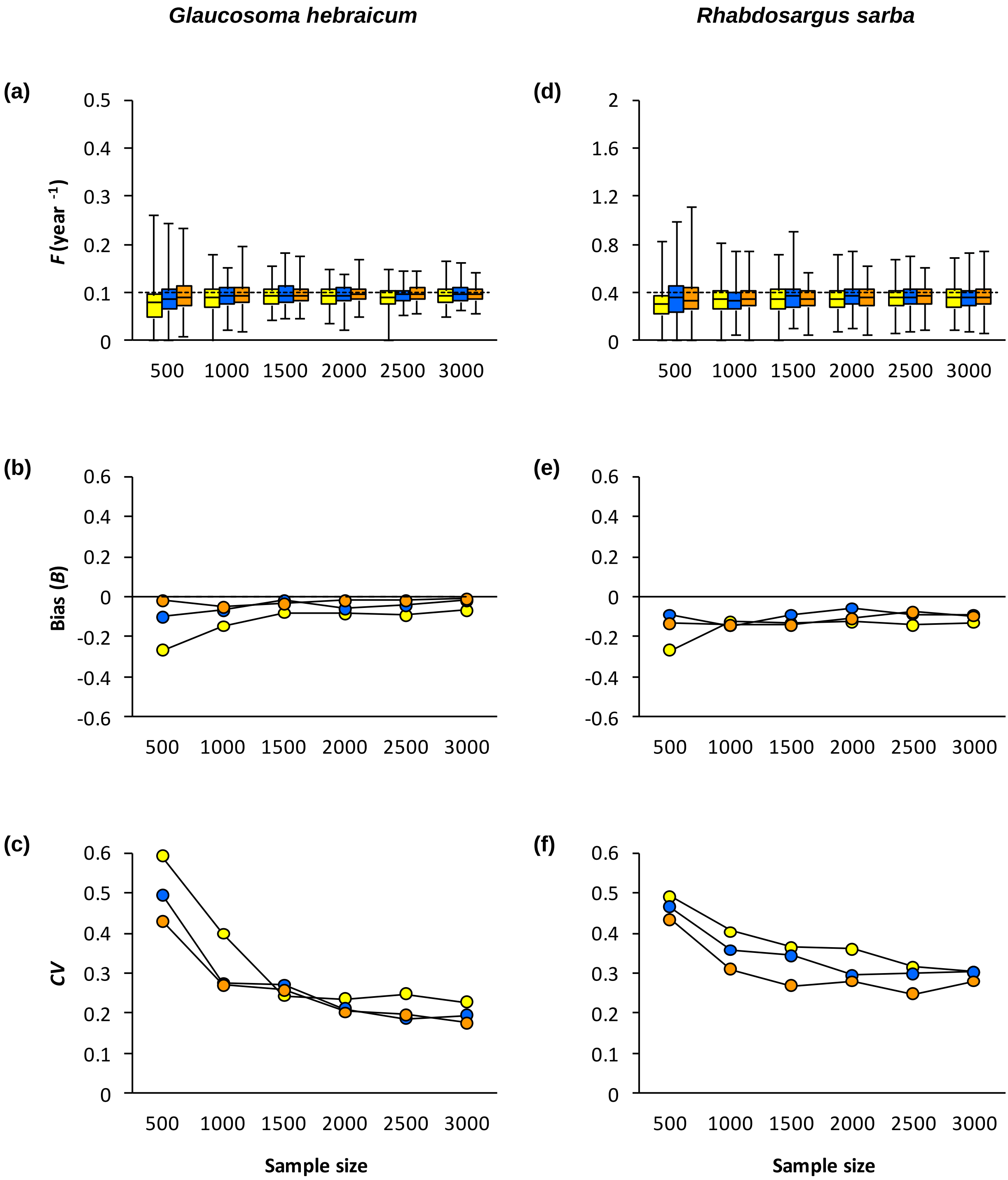


Fig. 4.3. Influence of sample size and sampling frequency (1 yr – yellow, 2 yrs – blue, 3 yrs – orange) on (a,d) estimates of fishing mortality, *F*, (b,e) relative bias, *B*, and (c,f) *CV* for the two fish species when recruitment variability is high, using linear catch curve (CC1). In a and d, the lower and upper bounds of each box represents the lower and upper quartiles of *F* estimates, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively.

CC1 was fitted to age composition data for 500 fish sampled in a single year, the estimates for *F* were highly biased (Fig. 4.3e). The fitting of CC1 to multiple years of data led to little improvement in the accuracy of mortality estimates, except when the sample size was 500 fish (Fig. 4.3e). As with *G. hebraicum*, doubling the level of recruitment variability of *R. sarba* led to reduced precision in the estimates of *F* (by about 1/3). Precision was typically greatest for *R. sarba* when CC1 was fitted to data sampled over three years and least when sampled in one year (Fig. 4.3f).

Relative abundance analysis – low recruitment variability

In comparison to CC1, which yielded mortality estimates for *G. hebraicum* that were almost always close to the true value regardless of sample size, relative abundance analysis (CC2) tended to overestimate *F*, particularly at low sample sizes. For example, when recruitment variability was low and 500 fish were sampled over two and three years (note that CC2 requires at least two years of data), the respective median estimates of *F* were as much as 27 and 36% greater than the true value for *F* (Fig. 4.4a). Consequently, the bias associated with the CC2 mortality estimates for *G. hebraicum* was always greater than for CC1 although, when sample sizes were high, these differences were close to negligible (*cf.* Figs 4.2b and 4.4b). When recruitment variability in *G. hebraicum* was low, the values of *CV* for the CC2 mortality estimates were similar to those of CC1 (*cf.* Figs 4.2c and 4.4c). When sample sizes for *R. sarba* were low, CC2 could not always be fitted to (all 100 sets of) the simulated age composition data. At larger sample sizes, the median values for the CC2 mortality estimates were always close to the true value (Fig. 4.4d), as reflected in the low levels of bias (-0.045 to 0.068) (Fig. 4.4e). As with *G. hebraicum*, the *CV*s for *R. sarba* with low recruitment variability were similar for CC1 and CC2 (*cf.* Figs 4.2f and 4.4f).

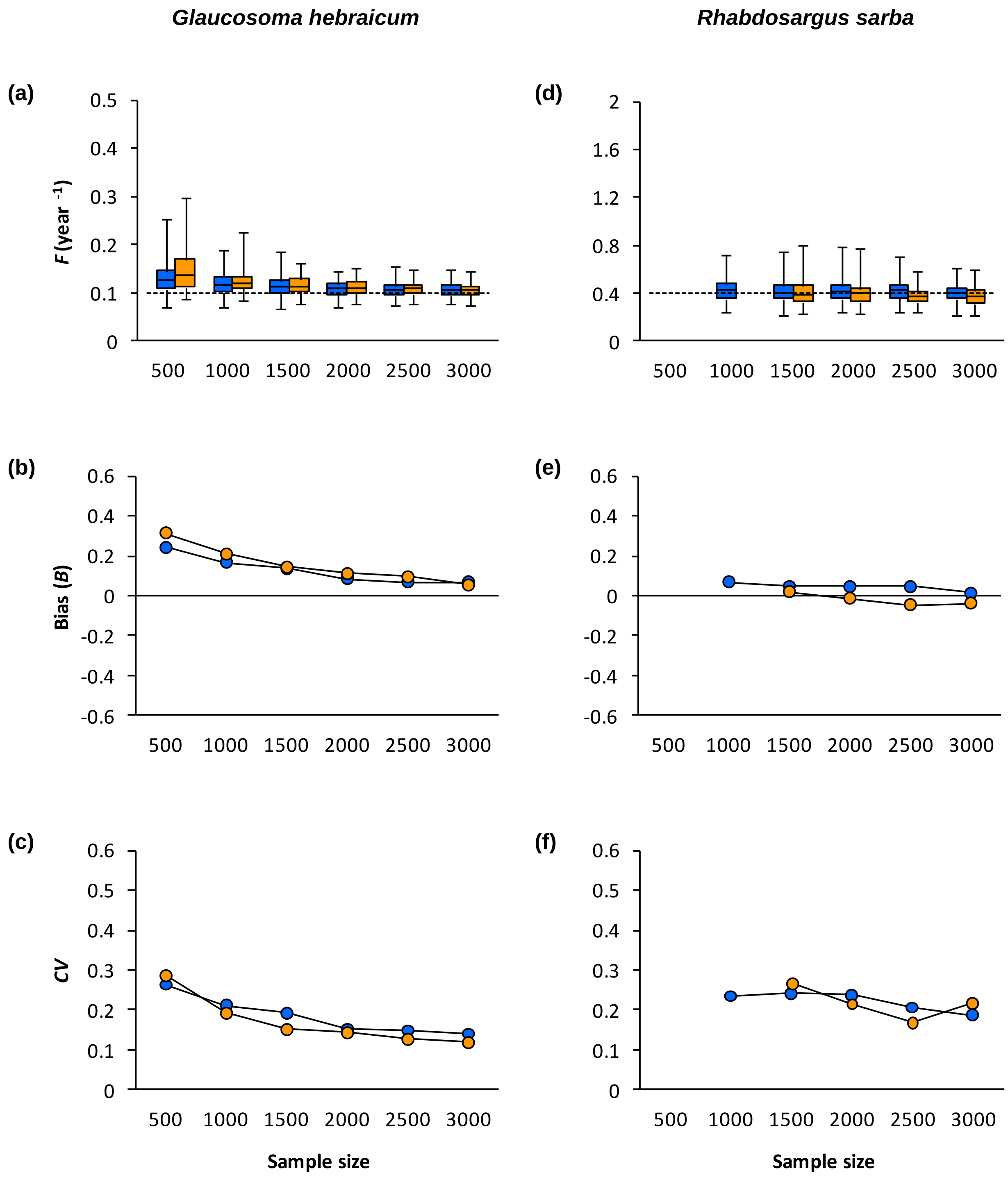


Fig. 4.4. Influence of sample size and sampling frequency (2 yrs – blue, 3 yrs – orange) on (a,d) estimates of fishing mortality, *F*, (b,e) relative bias, *B*, and (c,f) *CV* for the two fish species when recruitment variability is low, using relative abundance analysis (CC2). In a and d, the lower and upper bounds of each box represents the lower and upper quartiles of *F* estimates, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively. Note that for *R. sarba*, relative abundance analysis could not always be fitted to age composition data, particularly when sample sizes were low.

Relative abundance analysis – high recruitment variability

As when recruitment variability in *G. hebraicum* was low, CC2 produced positively biased mortality estimates when recruitment variability was high, with the estimates becoming closer to the true value only when sample sizes were high (Fig. 4.5a,b). The magnitude of bias at each sample size was similar for the two levels of recruitment variability (*cf.* Figs 4.4a,b and 4.5a,b). As with CC1, higher recruitment variability resulted in the mortality estimates produced by the relative abundance analysis being less precise (*cf.* Figs 4.4c and 4.5c). The precision of CC2 mortality estimates for *G. hebraicum* with high recruitment variability was substantially less when these were based on three rather than two years of samples (Fig. 4.5c). CC2 could only be fitted readily to age composition data generated for *R. sarba* (*i.e.* to all 100 simulated samples) when samples were distributed over two years and the sample size was relatively high. As with *G. hebraicum*, doubling recruitment variability in *R. sarba* had a limited impact on bias but led to substantially decreased precision (*cf.* Figs 4.4 and 4.5).

Influence of species biology on the effectiveness of management controls

Effectiveness of boat limit control for reducing mortality

When recruitment variability for *G. hebraicum* was low and the value of *F* for the stock at its initial equilibrium state was set to 0.2 year^{-1}, the specified initial management regulations (MLL = 500 mm, boat and bag limit = 5 fish trip^{-1}) had essentially no effect on fishing pressure (true *F*, after taking into account initial management controls = 0.197 year^{-1}). (Note that by setting the bag limit to equal the boat limit, this meant that catches were not constrained more by one control, and thus the results of the analysis would apply equally to either control when applied on

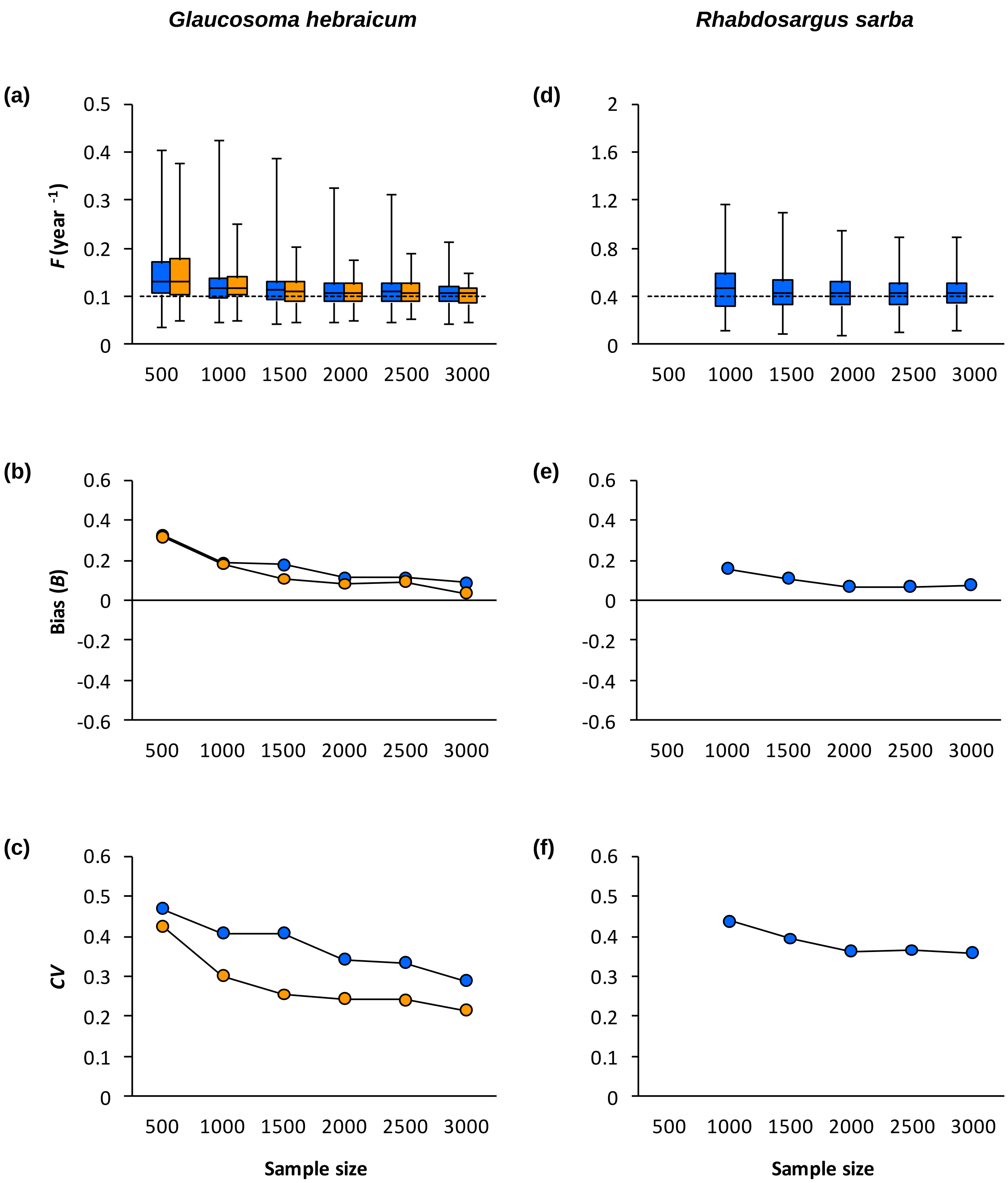


Fig. 4.5. Influence of sample size and sampling frequency (2 yrs – blue, 3 yrs – orange) on (a,d) estimates of fishing mortality, *F*, (b,e) relative bias, *B*, and (c,f) *CV* for the two fish species when recruitment variability is high, using relative abundance analysis (CC2). In a and d, the lower and upper bounds of each box represents the lower and upper quartiles of *F* estimates, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively. Note that for *R. sarba*, relative abundance analysis could not always be fitted to age composition data.

its own. Hereafter, we refer to the bag and boat limit controls, which were always kept the same, as just the boat limit.)

Reducing the boat limit for *G. hebraicum* from 5 to 4, 3 and 2 fish trip^{-1} for the projection period resulted in the median values for *F* declining by only 1.4, 5.5 and 15.7%, respectively (Fig. 4.6a). (Note that the values for *F* reported here and below represent median values for the true *F* after projection period, as calculated from the 100 simulations with different recruitment series). Further reducing the boat limit to 1 fish trip^{-1} resulted in *F* declining more substantially (by 34.5%, to 0.128 year^{-1}). Although mortality declined in a similar fashion when recruitment variabiity was high, the values for *F* were more variable, particularly when boat limit was 1 or 2 fish trip^{-1}. For example, when the boat limit was 2 fish trip^{-1}, *F* ranged between 0.147 and 0.186 year^{-1} when recruitment variability was high, compared with 0.155 to 0.173 year^{-1}, when recruitment variability was low (Fig. 4.6a).

For *R. sarba* with low recruitment variability and the equilibrium value of *F* prior to the projection set to 0.8 year^{-1}, initially setting the MLL to 240 mm and the boat limit to 20 fish trip^{-1} had essentially no effect on mortality (true *F*, after taking the boat limit into account = 0.790 year^{-1}). Although reducing the boat limit to 16 and 12 fish trip^{-1} had only a minor effect on mortality (*F* = 0.774 and 0.711 year^{-1}, respectively), for example, when the boat limit was changed to 8 fish trip^{-1}, the values for *F* lay between 0.296 and 0.702 year^{-1} (lower quartile = 0.415 year^{-1}, upper quartile = 0.545 year^{-1}). As with *G. hebraicum*, higher recruitment variability led to *F* being more variable, particularly when the boat limit was small (Fig. 4.6b).

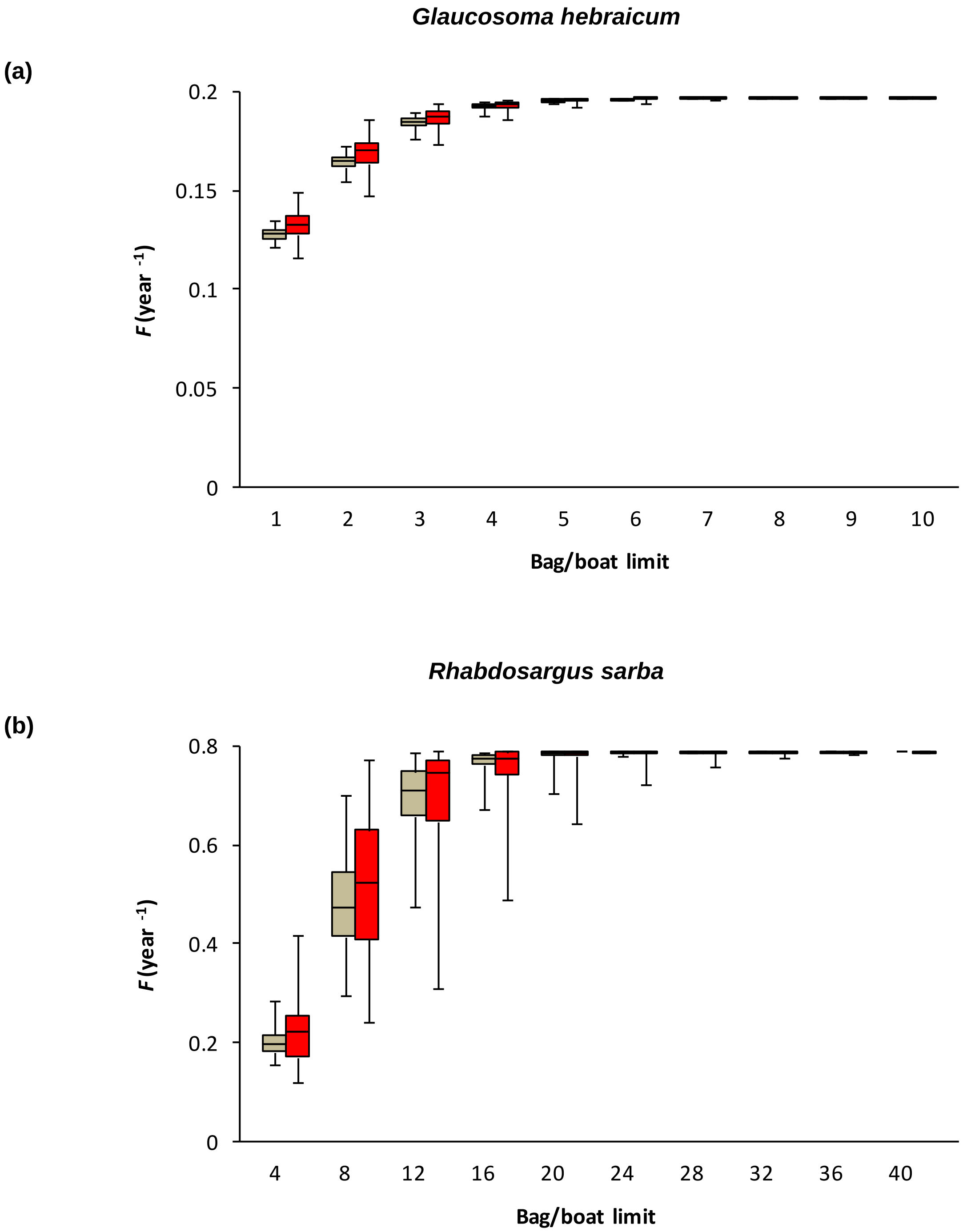


Fig. 4.6. Effect of the bag/boat limit control on the (true) level of fishing mortality, *F*, for fully-selected (a) *Glaucosoma hebraicum* and (b) *Rhabdosargus sarba* when recruitment variability is low (grey, left) and high (red, right). The lower and upper bounds of each box represents the lower and upper quartiles for values of *F*, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively.

Effectiveness of MLL control for reducing mortality

As would be expected, changing the MLL for *G. hebraicum* had essentially no effect on the level of mortality to which fully recruited fish were exposed (Fig. 4.7a). When recruitment variability was low, increasing the MLL from 300 to 750 mm reduced the total number of fish killed by fishing (*i.e.* fish of all sizes) in the final year of the projection period by only 18.6%, from 23,600 to 19,200 fish (Fig. 4.7b). Similarly, when recruitment variability was high, the median numbers of fish declined by only 16.8% as the MLL was increased from 300 to 750 mm. However, when recruitment variability was high, the numbers killed varied far more between different recruitment series. For example, the numbers killed when the MLL was set to 350 mm ranged by as much as 12,800 to 42,300 fish, compared with only 18,400 to 30,000 fish when recruitment variability was low (Fig. 4.7b).

As with *G. hebraicum*, changing the MLL for *R. sarba* from that specified for the stock in its initial state typically had no effect on the fishing mortality for fully recruited fish. In contrast to the situation with *G. hebraicum*, the extreme lower values for *F* were often much lower than the value of *F* for *R. sarba* when the stock was in its initial state (0.197 year^{-1}), particularly when recruitment variability was high and the MLL was low (Fig. 4.7c). Increasing the MLL for *R. sarba* had a greater effect on the numbers of fish killed than it did for *G. hebraicum* (*cf.* Fig. 4.7b,d). For example, when recruitment variability was set to a low level, increasing the MLL for *R. sarba* from 200 to 290 mm reduced the number of fish killed by as much as 41%, *i.e.* from 33,700 to 19,800 fish.

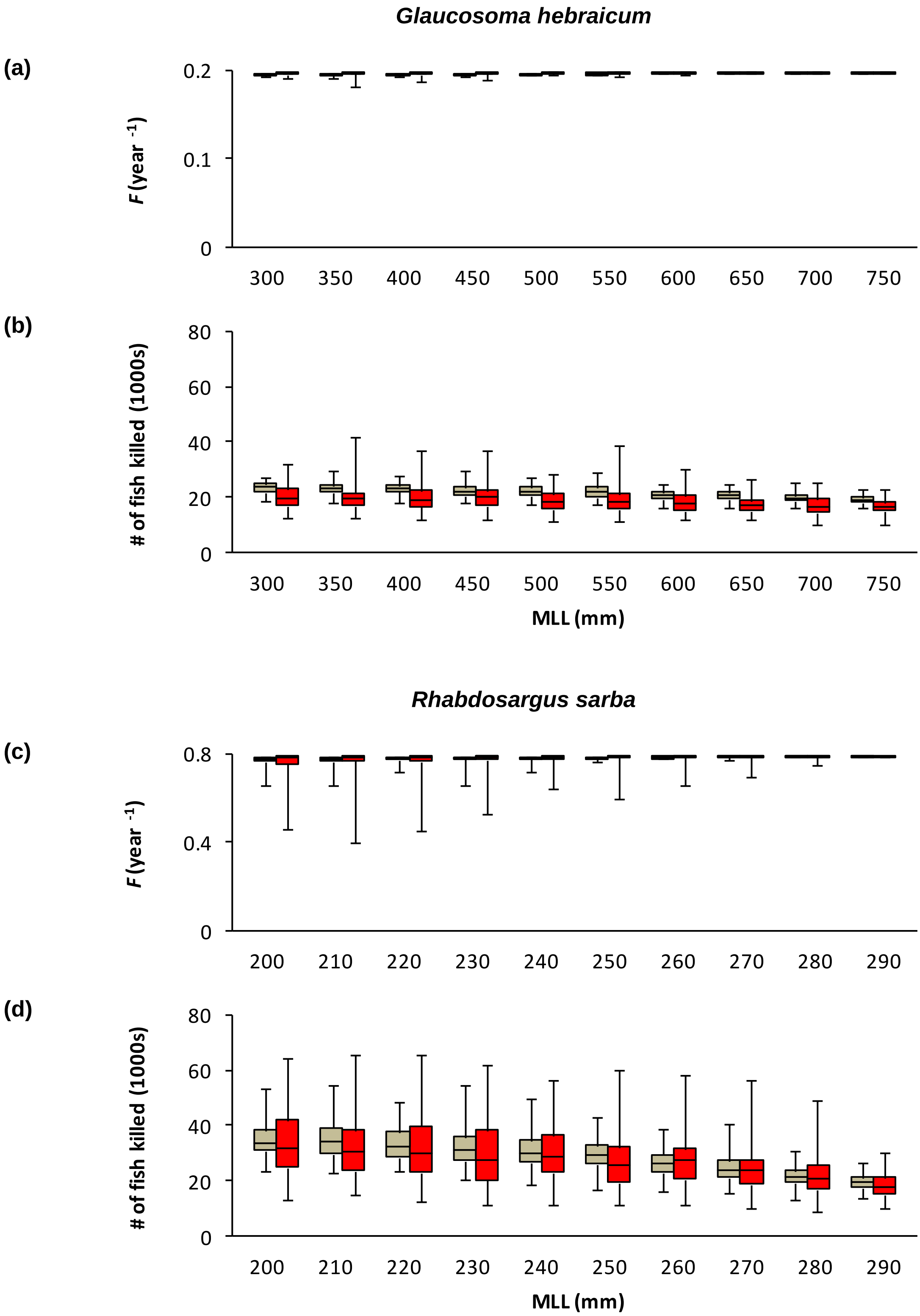


Fig. 4.7. Effect of the minimum legal length (MLL) control on (a,b) the (true) level of fishing mortality, *F*, for fully-selected fish and (c,d) the annual number of fish killed for the two fish species when recruitment variability is low (grey, left) and high (red, right). The lower and upper bounds of each box represents the lower and upper quartiles for values of *F* (a,c) or numbers of fish killed (b,d), respectively (from 100 simulations). The line in the middle of each box indicates the median value and the lower and upper bars show the minimum and maximum values, respectively.

Effectiveness of temporal closure control for reducing mortality

For *G. hebraicum* with low recruitment variability, introducing a two month temporal closure resulted in only a 5.5% reduction in *F*, from 0.197 to 0.186 year^{-1} (Fig. 4.8a). Extending the closure to 4 and 6 months resulted in *F* being reduced more substantially, *i.e.* by 18.8% (to 0.160 year^{-1}) and 39.1% (to 0.120 year^{-1}), respectively. Recruitment variability had essentially no impact on the effectiveness of temporal closures for *G. hebraicum* (Fig. 4.8a,b).

When recruitment variability in *R. sarba* was low, introducing temporal closures of 2, 4 and 6 months resulted in the median values of *F* being reduced by 5.6, 19.1 and 40.2%, respectively. The reductions for each closure duration were thus of a very similar magnitude for *R. sarba* as for *G. hebraicum* (*cf.* Fig. 4.8a,b). When recruitment variability was high, the minimum values for *F* for the different periods of closure were often substantially less than the median values for those periods (Fig. 4.8b).

Effectiveness of spatial closure control for reducing mortality

Introducing spatial closures impacted greatly on the mortality of fish in the area remaining open to fishing, particularly when the extent of the area closed to fishing was large (Figs 4.9a,c). For *G. hebraicum* with low recruitment variability, *F* for fully recruited fish within the open area increased from 0.197 year^{-1}, when there was no closure, to 0.247, 0.330 and 0.495 year^{-1}, when 20, 40 and 60% of the fishing grounds were closed to fishing, respectively. The same trend occured when recruitment variability was high (Fig. 4.9a). The total numbers of *G. hebraicum* killed as a result of fishing (*i.e.* mortality of fish of all sizes due to individuals being caught

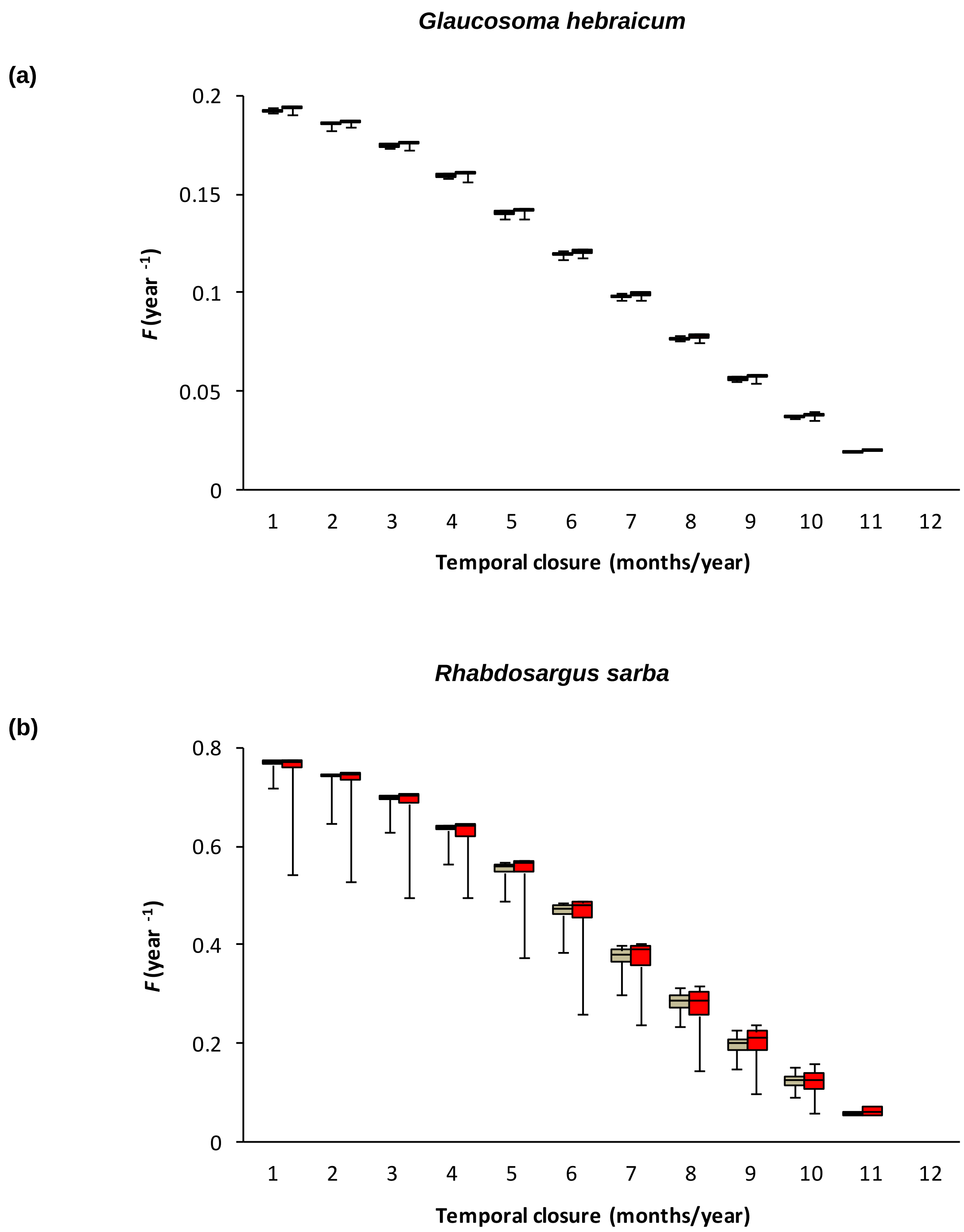


Fig. 4.8. Effect of the temporal closure control on the fishing mortality, *F*, of (a) *Glaucosoma hebraicum* and (b) *Rhabdosargus sarba* when recruitment variability is low (grey, left) and high (red, right). The lower and upper bounds of each box represents the lower and upper quartiles for values of *F*, respectively (from 100 simulations). The line in the middle of each box indicates the median value for *F* and the lower and upper bars show the minimum and maximum values for *F*, respectively.

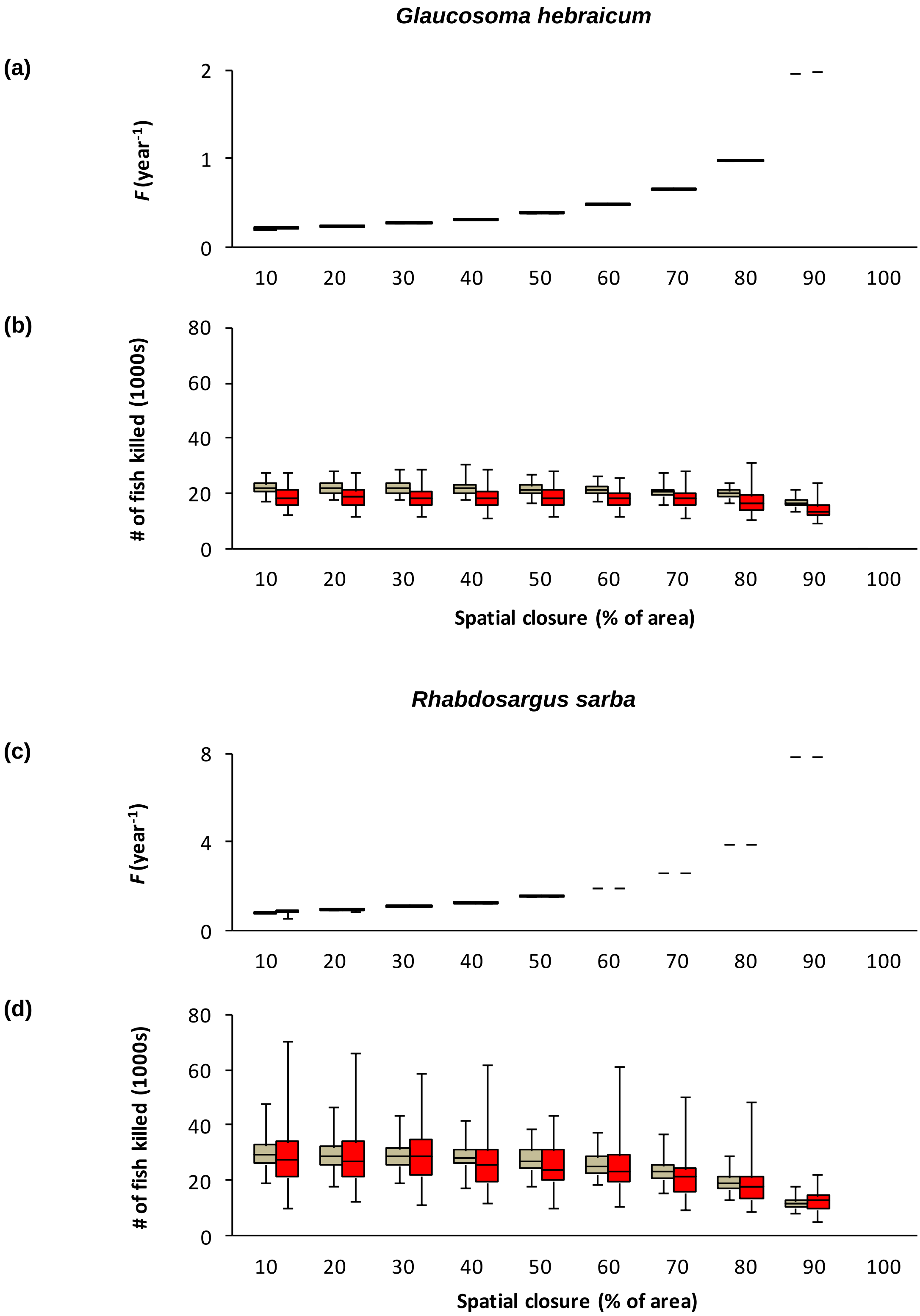


Fig. 4.9. Effect of the spatial closure control on (a,c) the (true) level of fishing mortality, *F*, for fish in the area open to fishing and (b,d) the annual number of fish killed for the two fish species when recruitment variability is low (grey, left) and high (red, right). The lower and upper bounds of each box represents the lower and upper quartiles for values of *F* (a,c) or numbers of fish killed (b,d), respectively (from 100 simulations). The line in the middle of each box indicates the median value and the lower and upper bars show the minimum and maximum values, respectively.

and retained or from injury on release) remained at a similar level (20,600 to 22,000 fish) when between 10 and 70% of the total area was closed (Fig. 4.9b). The numbers of fish killed declined only slightly to 19,900 and 16,700 fish, respectively, when the closure was extended to 80 and 90% of the total fishing area. When recruitment variability was high, the numbers of fish killed were consistently slightly less, *e.g.* between 18,000 and 18,800 fish, for area closures of 10 to 70% (Fig. 4.9b).

A similar situation occurred with *R. sarba*, with spatial closures exceeding 50% of the total area resulting in large increases in mortality within the area remaining open to fishing (Fig. 4.9c). The total numbers of *R. sarba* killed only declined substantially when most of the area was closed to fishing (Fig. 4.9d). For corresponding levels of closure, the median numbers of fish killed were only slightly affected by different levels of recruitment variability. There was more variability in the numbers of fish killed (for different recruitment series) when recruitment variability was high (Fig. 4.9d).

DISCUSSION

Influence of sampling design and species biology on the effectiveness of catch curve analysis

The simulations undertaken in this study indicate that, for a short-lived species such as *Rhabdosargus sarba* (maximum age of ~11 years; Hesp *et al.*, 2004), linear catch curve analysis (CC1) (Ricker, 1975) is likely to underestimate mortality. In contrast, relative abundance analysis (CC2) (Deriso *et al.*, 1985) is indicated to often provide overestimates of mortality for longer-lived species, such as *Glaucosoma hebraicum* (maximum age of 41 years; Hesp *et al.*, 2002), unless sample sizes are relatively large (several thousand fish). High recruitment variability is predicted to increase bias of mortality estimates produced by CC1, particularly when this type of analysis is based on samples collected during a single year. In contrast, recruitment variability does not appear to impact strongly on the accuracy of mortality estimates produced by CC2. The simulations also indicated that distributing samples (equally) over three rather than two years results in little improvement in accuracy of mortality estimates produced by both types of catch curve. Both sample size and recruitment variability were shown to impact strongly on the precision of mortality estimates. This is particularly true for CC1, when samples are drawn from a single year.

The simulations undertaken provide some indication of the sample sizes likely to be required for both short-lived and long-lived fish species with different levels of recruitment variability for estimating mortality using catch curve analysis. As shown by the analyses, the sample size required is likely to vary considerably depending on the age structure of the fish population and on the type of catch curve analysis employed. It also needs to be recognised that the sample size necessary for reliably estimating mortality will reflect the degree to which samples are representative of the fish population. In this study, individual age composition samples were drawn randomly from the simulated fish stock, thereby providing samples that were fully representative of the underlying population. However, in some fisheries, *e.g.* commercial trawl fisheries (Pennington *et al.*, 2002), fish may be caught in large "clusters" during individual fishing trips. In such cases, because fish within each cluster tend to be more similar to each other than to other fish (Aanes & Pennington, 2003), the numbers required to provide a representative sample of the population is likely to be greater than indicated by this study. With recreational fisheries for species such as *G. hebraicum*, this is likely to be less of a problem as bag and boat limits will prevent large numbers of fish from being retained during a single fishing trip. It is concluded that, if samples for such species are collected from many fishers who collectively operate throughout the fishery, the simulation results provide a reasonable indication of sample sizes required for estimating mortality using the two types of catch curve analysis explored in this study. A further word of caution when interpreting the results of these simulations is that the data have been simulated assuming that, on average, all fish in the stock were subject to the same (probability of) mortality throughout life, which may well not be the case with many fisheries. Deviation from this equilibrium assumption will impact on the reliability of mortality estimates produced by catch curve analysis.

The simulations indicate that, even when sample sizes are substantial, both types of catch curve analysis (*i.e.* CC1 and CC2) can produce biased mortality estimates. They also suggest that the level of bias produced by these two forms of catch curves can differ between species. An important question is thus whether these differences are due to an artefact of some aspect of the analysis, or whether such differences are likely to be real. One possible explanation is that that the differences were due to random differences in sample data used in the analyses. However, this could be disproved. Because the random number generator sequence used by the MSE model was controlled, this meant that that the sample data (*i.e.* age compositions) to which CC1 and CC2 were fitted were identical, confirming that the differences in bias were due to the analyses themselves.

The question thus remains as to which factors contributed to the biases in mortality estimates produced by the two catch curve methods. As shown in this study, other workers have also found that linear catch curve analyses tend to underestimate mortality (Murphy, 1997; Dunn *et al.,* 2002). Murphy (1997) suggests this is because this type of analysis violates the assumption of linear regression analysis that the variance is equal throughout the range of values for the independent variable, *i.e.* age groups. The variances of the logged abundances are positively correlated with age until the distribution is truncated when zero frequencies appear in samples for older age groups (Murphy, 1997). Ultimately, this can result in underestimation of mortality. The effects of violating the linear regression assumption of constant variance can be reduced by truncating age frequency distributions and employing a minimum threshold abundance rule for fitting the linear regression, such as five individuals per age class (Chapman & Robson, 1960). However, Murphy (1997) shows that even this does not fully eliminate the negative bias. Our finding that CC1 tended to perform better for the long-lived *G. hebraicum* than the shorter-lived *R. sarba* indicates that the effects of violating the assumption of constant variance when using simple, linear catch curve analysis are less for longer-lived species. This is likely to be because outliers in the age distributions for older fish will have less influence on analyses for populations for which there are many age groups.

The next question is why CC2 (relative abundance analysis) tended to require larger sample sizes for *G. hebraicum* than *R. sarba* to produce accurate mortality estimates (*cf.* Figs 4.4b,e and 4.5b,e). This may be because CC2 involves estimating not only mortality but, at the same time, also the relative strengths of the different year classes. Fitting CC2 thus required estimation of 42 parameters for *G. hebraicum*, compared with only 12 for *R. sarba*. To maximise program speed, we employed an algorithm (using AD Model Builder) which rapidly estimates recruitment parameters for all year classes at once. However, for a long-lived species such as *G. hebraicum*, it may be advantageous to use a stepwise approach to introduce parameters into the analysis one at a time, *i.e.* to employ a forward stepwise forward selection algorithm (Sokal & Rohlf, 1995). Likelihood ratio tests (Cerrato, 1990) can be used to determine, at each step, whether introducing an additional year class as a recruitment parameter will significantly improve the fit of the model, with the process being terminated when this does not occur. Use of this approach to fitting relative abundance analysis for Western Blue Groper (*Achoerodus gouldii*), which can live to 70 years, resulted in only 11 year classes being introduced into the analysis (Coulson *et al.*, 2009). With fewer parameters, this approach to fitting relative abundance analysis is likely to be more reliable when sample sizes are small.

As also concluded by Allen (1997) and Dunn *et al.* (2002), this study has shown that mortality estimation is sensitive to recruitment variability. The fact that high recruitment variability negatively impacted on the precision of mortality estimates, regardless of whether linear catch curve or relative abundance analysis was used, has important implications for management. For fish species or stocks with high recruitment variability, it is important for managers to be aware that stock assessment advice is likely to be less precise than if recruitment variability is low, even when assessments are based on large sample sizes and on analyses allowing for variable recruitment.

Influence of species biology on the effectiveness of management controls

The Monte Carlo simulations undertaken to evaluate the effectiveness of alternative management controls for reducing fishing mortality indicate that some are likely to be far more effective than others, and that the value of applying these controls will vary for different species. The simulations also suggest that, although recruitment variability, on average, has limited impact on the effectiveness of each control, the outcomes of changes to the management controls are less predictable when recruitment variability is high.

The simulations exploring the effectiveness of boat limits suggested that, if the two fish species were initially heavily exploited at approximately twice their rates of natural mortality, an 80% boat limit reduction (from 5 to 1 fish trip^{-1} for *G. hebraicum* and 20 to 4 fish trip^{-1} for *R. sarba*) would reduce *F* far more for *R. sarba*. The calculations for determining the effectiveness of boat (and bag) limit controls take into account a number of factors, including the distribution for the number of fishers on boats, mortality rate, catch rate, level of fisheries compliance (*i.e.* probability of fishers high-grading) and post-release mortality rate.

As the distribution specified for the number of fishers on boats was kept the same for the two species, this would not have influenced the result. Furthermore, since both fish species were initially exploited at similar levels (relative to their rate of natural mortality), the observed differences in the effectiveness of the control are not likely to be linked to this factor. Although the boat limits specified for *G. hebraicum* (initially 5 fish trip^{-1}) and *R. sarba* (initially 20 fish trip^{-1}) differed, they were of a similar magnitude relative to the catch rates specified for the two species (1 and 5 fish trip^{-1}, respectively). Thus, differences in values specified for the boat limit controls would not have markedly influenced the results. The levels of compliance by fishers when targeting *G. hebraicum* (0.8) and *R. sarba* (0.9) were also similar, suggesting that high-grading did not strongly affect the result (this was confirmed by further exploratory simulations – data not shown). In contrast to all of the above factors, the much greater probability of post-release mortality specified for *G. hebraicum* (0.4) compared to *R. sarba* (0.05) is likely to have significantly impacted on the effectiveness of the boat limit control for regulating mortality of the two species.

As the outcomes of applying changes to the boat (and bag) limit can be influenced by a variety of factors, the effectiveness of this management control for different fisheries is likely to differ markedly. As shown in this study, model simulations by Woodward & Griffin (2003) also highlight the importance of accounting for post-release mortality (and high-grading) when estimating the effectiveness of boat limits. Although the model takes into account a range of factors that can influence the value of a boat limit control for regulating fishing mortality, it does not take into account the influence of fisher behaviours on the effectiveness of this control. As fishers may respond to a reduced boat limit by fishing more often, the effectiveness of this control may be less than that indicated by the simulation results.

An important consideration regarding bag and boat limits is that these controls are likely to have the greatest effect on the most successful fishers (Woodward & Griffin, 2003). Thus, if relatively few fishers take a large proportion of the total catch, imposing tighter bag and boat limits can be an option in recreational fisheries for reducing the impacts of those fishers, whilst having minimal effect on the fishing experience of the majority of anglers. However, the finding that F for *G. hebraicum* was still too high (*i.e.* $F > M$) after the boat limit had been reduced to only 1 fish $trip^{-1}$ does highlight the point made by Cox *et al.* (2002) that, on their own, bag and boat limit controls are unlikely to protect a stock from heavy exploitation. This is particularly true in situations where fishing effort is likely to continue to increase due to an expanding human population.

The simulations showed that changing the minimum legal length for retention (MLL) of a species does not influence mortality, as estimated by catch curve analysis for fish above the age at full recruitment into the fishery. This result would be expected assuming, as in the analysis, that changing the MLL does not influence catch rates (although this could occur as a result of fishers changing behaviours in response to the management change), and that fish samples have been collected from catches taken by fishers subject to the management restrictions (rather than by researchers with exemption permits). This is because changing the MLL is not predicted to affect the exploitation rate of fully recruited fish, but rather only act to restrict the range of ages in the sample data used for catch curve analysis.

Given that MLL restrictions are typically intended for protecting small and young fish, measures of their effectiveness need to take into account the extent to which under-sized fish suffer post-release mortality (Kirchner *et al.*, 2001). The results indicated that raising the MLL for *G. hebraicum*, which is a species that is considered to experience relatively high post-release mortality (St John & Syers, 2005), will have little effect on overall mortality (as indicated by estimates of the numbers of fish killed in the final year of the projection period). However, for *R. sarba*, which is likely to experience far less post-release mortality, the analysis suggests that increasing the MLL will reduce overall mortality substantially. As also demonstrated in simulation studies of the recreational fishery for Red Snapper (*Lutjanus campechanus*) in the Gulf of Mexico by Woodward & Griffin (2003), the results of this study show that post-release mortality can have a major impact on the effectiveness of MLL controls.

In discussing minimum length restrictions, Kirchner *et al.* (2001) state that their value is “questionable”, highlighting the example of the Silver Kob (*Argyrosomus inodorus*) stocks in South Africa, which have collapsed despite a MLL being in place since 1940 (citing Griffiths, 1997). MLL restrictions have often been applied with the intention of ensuring that individuals “spawn at least once” before being harvested. However, this policy is likely to be inadequate for many species. For example, it fails to recognise that, for species with high recruitment variability, spawning by individuals over several years is likely to be important as a bet-hedging strategy for helping ensure long-term reproductive success despite long periods of unfavourable environmental conditions for larval survival (Leaman & Beamish, 1984). It also does not recognise that egg production increases disproportionately with size of fish, and that egg quality/larval survival may increase with maternal age (Berkeley *et al.*, 2004; Bobko & Berkeley, 2004). For these and other reasons, some fisheries are managed using maximum size limits and “slot limits”, both which aim to protect larger/older fish in the population. It should be noted that MLL controls have also been used in some fisheries as a means for increasing yields. On the basis of analyses of creel survey data, Paukert and Willis (2002) and Isermann *et al.* (2007) conclude that a MLL will be more effective for improving yields (and increasing the number of larger and older fish in a population) when individuals grow rapidly and when exploitation is high but natural mortality is low. In conclusion, the effectiveness of an MLL will depend on the level of post-release mortality and on the proportion of the life span during which fish are protected. The latter will depend to some extent on the pattern of growth of the species during the earlier part of life.

The simulation results, for both *G. hebraicum* and *R. sarba*, indicate that the effectiveness of short temporal closures (< 2-3 months) for reducing *F* may be limited. Temporal closures are predicted to become increasingly effective (relative to duration) as closure length expands up to about 6 months. The results indicate that, beyond this point, the effectiveness of temporal closures, relative to duration, tends to “level off” and then begin declining. This trend is due largely to assumptions made about the ways fishers are likely to respond to temporal closures. Our model calculations take into account that temporal closures will induce a pulse of fishing effort when the fishery re-opens, as fishers aim to compensate for their losses during the closed period (*e.g.* Guénette *et al.*, 1998; Coleman *et al.*, 2004). The calculations also assume that there is likely to be an upper limit on the amount of fishing activity that fishers will undertake during any period of time (*e.g.* Watson *et al.*, 1993), although, as highlighted by Cox *et al.* (2002), this may be untrue in the long-term for recreational fisheries. As has been pointed out in other studies, understanding the dynamics of fishing fleets and how fishers respond to new regulations is crucial for reliably predicting the effects of management (Sluczanowski, 1984; Branch *et al.*, 2006; Haapasaari *et al.*, 2007).

Although the above-discussed assumptions are likely to apply to both species, a range of other factors may be important in influencing the effectiveness of temporal closure controls for different types of fisheries. For example, factors such as market demands (for commercial fishers) and temporal differences in the abundance of target species can all effect behavioural responses of fishers to management changes (Allen & McGlade, 1986; Somers & Wang, 1997; Pradhan & Leung, 2004). For recreational fisheries, the timing of temporal closures is also likely to be important, *e.g.* depending on when many people are likely to fish. For multi-species fisheries, fishing effort is likely to be re-allocated towards species not affected by the closure (Holland & Sutinen, 1999; Little *et al.*, 2008) or to other, unprotected, fishing grounds (Somers & Wang, 1997). Furthermore, temporal closures may be used for various reasons other than to simply reduce overall mortality, *e.g.* they are often applied as a means to protect spawning aggregations (Cox *et al.*, 2002). With commercial fisheries for short-lived species with highly variable recruitment (*e.g.* prawns), temporal closures may be used to optimise economic profits by limiting catches of small animals, *i.e.* preventing growth over-fishing (Watson *et al.*, 1993).

Although outside the scope of this study, the MSE model is readily able to be extended to incorporate additional factors that may influence the effectiveness of temporal and other closures. For recreational fisheries, information on seasonal differences in fishing effort for different fisheries, such as from recent creel-survey data, are likely to be of particular importance for improving model predictions of the effects of temporal closures.

For spatial closures, the simulation results for both species suggest that, as the proportion of the total fishery area closed to fishing increases, mortality in the area remaining open to fishing will increase markedly. The simulations also indicated that, for both *G. hebraicum* and *R. sarba*, a spatial closure of even as much as 50% is likely to have very limited impact on overall fishing mortality for the population. This highlights the point that when applying area closures, fishing effort in areas remaining open to fishing needs to be controlled using other measures (Jones, 2001; Kaiser, 2005; Greenstreet *et al.*, 2009; Halpern *et al.*, 2010).

The indications from the simulation results for spatial closures that, at extreme levels of fishing pressure, mortality in the open area will continue to rise exponentially as the proportion of area closed increases, may not be true. As pointed out by Smith *et al.* (2010), the increased fishing pressure in the areas remaining open to fishing will lead to depletion of the portion of stock in those areas over time, thereby reducing the incentive for people to continue fishing. Future work on this aspect could focus on relaxing the assumption made by this model and many others (Guénette *et al.*, 1998; Lynch, 2006) that fishing effort and the amount of area closed are directly correlated. Further development could also focus on alternative assumptions about the spatial distribution of fishing effort and rates of migration of fish between open and closed areas.

In discussing spatial closures, it needs to be pointed out that they may be of value for a variety of purposes other than for managing single target species (see *e.g.* Jones, 2001). As we have produced a single-species model, it is not suited for exploring broader questions about the effectiveness of this type of control for social and ecosystem objectives. Furthermore, we point out that some other types of models, *e.g.* agent-based models, are ideally suited for exploring questions about the consequences of behavioural responses of fishers to fisheries management controls, and complex interactions between fleet dynamics, fish stocks and management processes (Soulié & Thébaud, 2006; Hesp *et al.*, 2010). In the context of management controls, we consider the MSE model produced in this study is valuable for helping facilitate understanding of the broad implications of applying commonly used controls for conserving target species. The modular nature of the program developed in this study readily allows extension of the model to incorporate and explore different factors and alternative assumptions about how different management controls are likely to impact on fish stocks and fishers.

BENEFITS

As specified in the application, this project benefits researchers at the Department of Fisheries, WA, by providing a generic Management Strategy Evaluation (MSE) tool for simulating the effectiveness of alternative science and management options for fish species with different life cycle characteristics (including recruitment variability).

Emphasis was placed on developing an effective program interface for communicating the results of relatively complex stock assessment analyses. The results of scenario testing experiments with science students and feedback from stakeholder workshops, during which participants were required to interpret stock assessment outputs produced by the program and make decisions as to how fish stocks should be managed in different (hypothetical) circumstances, demonstrated the effectiveness of the software in this regard. The program has good potential to be used by researchers and managers as an educational tool for helping to communicate stock assessment information to stakeholders with varying backgrounds and fishery knowledge. Stock assessment information is displayed by the program in a weight-of-evidence framework, thereby aligning with approaches being adopted by the Department of Fisheries in WA, and several other fisheries agencies in Australia, for assessing fish stocks using age-based monitoring programs.

The results of Monte Carlo simulation studies provided insight as to how recruitment variability and other biological characteristics of fish species influence the effectiveness of strategies for collecting age composition samples, stock assessment analyses and different management options. This information will benefit researchers for developing and reviewing age-based monitoring programmes, as well as for understanding the reliability of stock assessment approaches for estimating mortality. It will benefit managers for understanding the implications of recruitment variability for the effectiveness of management and the likely effects of different levels of fisheries compliance on stock status. Industry will benefit from the project through better understanding of implications of recruitment variability of fish for conserving fish stocks and through well informed management.

As recommended by Dr James Scandol when reviewing our original application for this project, the MSE model has been developed in Visual Basic.net (with Windows forms). To as great an extent as possible, the various program procedures have also been partitioned into classes. This modular structure of the program code will facilitate ease of reusing the model algorithms in the future. As also recommended by Dr Scandol, the program is not linked to Microsoft Office extensions such as Excel, thereby avoiding problems associated with people not being able to use the software due to version control issues. The program is deployed in a manner which is simple to download and install, and does not require any additional software (other than the standard Windows operating environment). We have chosen to use .txt files for storing the input and output data, making it easy for people to use the software on their own computers. To enhance program speed, the vb.net program is linked (via shell commands) to procedures in AD Model Builder (for fast optimisation) for fitting catch curves. Note that this feature of the model does not require users of the MSE software (other than the programmers) to download AD Model Builder.

The project has been of great benefit to early-career scientists at Murdoch University, through introduction of concepts related to fisheries modelling and simulation.

Ultimately, such training will be of benefit to the Department of Fisheries, WA, and to the fishing industry. Use of the MSE program has also been incorporated into undergraduate teaching courses at Murdoch University. The software is able to be freely accessed via our website http://www.cffrfisheriesmodelling.net.

FURTHER DEVELOPMENT

The management strategy evaluation (MSE) model produced in this study will continue to be developed over the next two and a half years, at least. In part, this will involve extension of the model to allow for size-related movements between habitats for inshore species such as Silver Trevally (*Psuedocaranx georgianus*) and King George Whiting (*Sillaginodes punctata*) for FRDC project 2010/001. This will involve adding a spatial component to the model to allow estimation of mortality for species which are exploited in different habitats during different life stages. We will continue to use scenario testing as a means of exploring and improving the effectiveness of the program interface for communicating stock assessment information.

The major focus of this project has been on developing a robust and user-friendly MSE simulation model that can be applied to fisheries assessed using age-based monitoring data. Although the majority of parameters used by the model for simulations undertaken during this study were obtained from available literature on the biology of different fish species, a few of the data inputs, such as for the distribution for numbers of fishers on boats and estimates of mean catch rates for the two fish species, were not well estimated. Improving inputs for these parameters will be important in the future for use of the program by researchers and managers at the

Department of Fisheries, WA, as a stock assessment tool. Scientists at the Department have indicated that some data exist within their organisation that could potentially be used to provide better estimates of the parameters indicated.

Aspects of the analyses undertaken for this report, *i.e.* scenario testing experiments and Monte Carlo simulations, will be extended over the coming year for completion of Emily Fisher's PhD thesis. A new focus of Emily's work is likely to involve extending the Monte Carlo simulation studies to explore the effectiveness of different types of catch curve analysis for detecting responses of fish stocks to management changes. As traditional catch curve methods are subject to the assumption of constant mortality, they are not well suited for this purpose. Development of catch curve approaches that can account for changes in mortality over time are very much needed to determine whether management aimed at rebuilding fish stocks has been effective.

PLANNED OUTCOMES

The planned outcomes for this project, as stated in the original application, and descriptions as to how they have been met, are as follows.

(1) A method of exploring the effectiveness of a harvest strategy based on an assessment employing the types of data available for the scalefish fisheries of south-western Australia, and taking recruitment variability into account, will be available to the Department of Fisheries, WA. By employing this approach, researchers at the Department will be able to explore the effectiveness of

management responses to adverse levels of inter-annual recruitment and thereby improve the robustness of the management strategies that they employ. The benefit for commercial and recreational fishers and the community is that the risk posed to the sustainability of fish stocks by recruitment variability will be reduced.

The project has led to the development of a management strategy evaluation (MSE) modelling tool. The model is ideally suited to the types of data available for scalefish species in south-western Australia, *i.e.* for which biological data such as life cycle parameters and size and age composition are typically available. The model generates mortality-based stock assessment advice, in a weight-of-evidence framework (determined using risk assessment analysis), aligning well with the types of assessment procedures currently being adopted by researchers at the Department of Fisheries, WA. The model enables exploration of the effectiveness of a range of management responses for species with different levels of recruitment variability, including episodic recruitment (and levels of autocorrelation of recruitment between years). The Monte Carlo simulation analyses undertaken for this project provide researchers and managers insight into how recruitment variability influences the effectiveness of different sampling strategies, mortality analyses and management approaches, thereby facilitating informed decision-making. Ultimately, this will help reduce the risk posed by recruitment variability to the sustainability of scalefish stocks.

(2) Operating models that represent several alternative hypotheses relating to the behaviour and life history of a fish stock, that are readily modifiable to represent a range of alternative species, will be available for use by fisheries scientists, allowing exploration of the appropriateness of alternative harvest strategies.

Improved fishery management should result, improving the sustainability of the resource for the benefit of commercial and recreational fishers and the community.

The operating model developed as part of the MSE simulation program is widely applicable to many fish species. Thus, it can be applied to gonochoristic and hermaphroditic species, to short-, medium- and long-lived fish species, and to fish with low, medium or high inter-annual variability in recruitment, or episodic recruitment. It also enables the science and management implications of different assumptions, such as for recruitment variability, to be explored. All parameters of the operating model are easily modifiable, enabling fisheries scientists to explore the appropriateness of alternative harvest strategies for a wide range of circumstances.

(3) A generic harvest strategy framework, that employs a modular structure and thereby facilitates the development and use of alternative modules representing operating models, monitoring and assessment methods and decision rules, will be available to the Department of Fisheries, WA. The availability of this tool will improve the quality of future assessments of the appropriateness and robustness of alternative harvest strategies. Commercial and recreational fishers and the community will benefit from the fact that the fish stocks are more likely to be sustained.

The MSE program employs a modular structure, using classes in vb.net as a means of structuring the various model routines. The program has been designed in a manner that readily allows for development and use of alternative modules within the existing model framework, which can represent alternative operating models, monitoring methods and decision rules. A range of alternative assessment methods and decision rules have already been built into the model. More can added. On the very sound advice provided by Dr James Scandol, we have focused on developing a user-friendly program with an effective interface. By making management strategy evaluation methods within reach of those researchers and managers without extensive modelling and stock assessment experience, this will help facilitate better uptake of the model and ultimately better management of fish stocks.

CONCLUSIONS

We have developed a generic fisheries management strategy evaluation (MSE) program that is widely applicable to scalefish fisheries for species relying on catch curve analysis-based mortality estimates for stock assessment. As well as providing a useful simulation tool for exploring questions related to fisheries monitoring, assessment and management, the model has potential as an education tool, and for facilitating stakeholder participation in management discussions.

In developing computer software for use by people with potentially limited technical experience and/or fisheries stock assessment knowledge, it is important to reach an appropriate balance between simplicity and realism. For such software to be valuable to fisheries scientists, this process cannot compromise on the level of sophistication of the analyses or on program functionality. Results and feedback from scenario testing workshops, in which people with different backgrounds used the program, provided strong evidence that the MSE program is an effective means for communicating stock assessment information. We thus conclude that, if sufficient focus is placed on designing (and testing) the program interface, it is possible to develop fisheries simulation tools for both scientific and communication/educational purposes. When developing a program interface for use in workshop situations, the ability to “switch off” some aspects of program functionality, such as to only allow users to change certain parameters (*i.e.* those of interest for the workshop), and ensure that users can only take one route when navigating their way through the program, is important.

Monte Carlo simulations demonstrated that the effectiveness of equilibrium-based stock assessment approaches (catch curves) for estimating mortality can be influenced by a range of factors, including the life history characteristics of the fish species, fish sampling design and type of analysis.

Regardless of sample size, linear catch curves can produce mortality estimates with substantial negative bias, which was demonstrated for the short-lived species, *Rhabdosargus sarba*. Such bias is likely to reflect violation of an assumption of linear regression when using this type of catch curve. As pointed out by other workers, the bias is potentially reduced by truncating age frequency distributions to include only age categories for which there are several fish.

Relative abundance analysis, an extended form of catch curve analysis that takes into account recruitment variability, was shown to produce mortality estimates for the long-lived species *Glaucosoma hebraicum* that are positively biased, unless sample size is large. It is likely that such bias can be reduced by modifying the approach used in this study to fit the analysis, *i.e.* only estimate the relative recruitment strengths for those year classes for which recruitment is shown to deviate statistically from the mean level of recruitment for all year classes. As it has been shown that effectiveness of a certain type of catch curve analysis may vary for different fish species (depending on their age structures), it is recommended that, for any situation, the results of multiple approaches to catch curve analysis be compared. Simulation testing of the effectiveness of catch curve analyses for each fish species for which mortality is estimated is also recommended.

High annual recruitment variability is shown to lead to reduced precision of mortality estimates produced by catch curve analysis. The analysis also showed that, when recruitment variability is high, mortality estimates are more reliable when based on data collected over multiple consecutive years. When recruitment variability is high, stock assessment advice is likely to be less reliable (precise).

As also shown in some other simulation studies, a range of factors can influence the effectiveness of management controls. If the key objective of management is to reduce fishing mortality, bag and boat limits and minimum size restrictions are most effective for species for which post-release mortality is low. When exploitation pressure is high, these controls, on their own, are not likely to be sufficiently effective for species with substantial post-release mortality. If suitably long, temporal closures can be an effective means for reducing exploitation pressure. Spatial closures are indicated to have limited effectiveness for reducing overall fishing mortality, *i.e.* because fishers can redirect their attention to areas that remain open. The results of simulation studies indicate that, although, on average, the effectiveness of the different management controls considered in this study is similar at different levels of recruitment variability, management outcomes are likely to be more variable when recruitment variability is high. This is likely to be the case because, at any given time, the effectiveness of a particular control for reducing fishing mortality will depend, to a certain extent, on the current abundance of fish of those ages that are vulnerable to effects of fishing, which is influenced by recruitment variability.

The MSE model can provide valuable insights as to how various factors influence the effectiveness of several commonly-used management controls. However, the effects of several factors, such as the behavioural responses of fishers, and temporal and spatial trends in fishing effort and fish abundance, are either not considered or not well accounted for by the model. With further model development, several of these factors could be better considered by the MSE model. Indeed, MSE provides an ideal platform for exploring which variables are likely to influence most the outcomes of management, thereby enabling those areas in most need of research/investment to be identified.

As with any single-species model, the MSE model produced in this project does not account for multi-species interactions. The MSE model should be considered as one of several valuable tools for exploring and communicating the effectiveness of various fisheries assessment and management approaches, for fisheries managed using age-based assessments. Comparisons of outputs by the MSE program with those of other types of models, *e.g.* multi-species models, are likely to be required to fully understand the range of effects that result from applying different types of management controls.

The project has provided very valuable training to students and early career fisheries researchers at Murdoch University.

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APPENDIX 1: Intellectual property

The information produced in the study is not suited to commercialisation.

APPENDIX 2: Project staff

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APPENDIX 3: Model description

This appendix provides the reader a mathematical description of the MSE model. An overview of the MSE framework is provided in Chapter 1 and a general introduction to the model and program interface is provided in Chapter 2.

Because this appendix is of a highly-mathematical nature, it was developed using LaTeX, a software package which is extensively used by mathematicians. LaTeX provides numerous features that assist in writing complex mathematical equations and produces a publication quality document that presents the equations in a well laid-out form, such as might appear in a mathematical textbook. This appendix thus differs somewhat in layout and style from that of the main body of the report.

THE OPERATING MODEL

Age-, length- and sex-structure

The population may be represented as an array $\boldsymbol{N}_t$ containing the numbers of fish of each sex lying within each age and length class at the start of year t. Thus,

$$\boldsymbol{N}_t = \begin{pmatrix} n_{1,1,f,t} & n_{1,2,f,t)} & \cdots & n_{1,J,f,t} \\ n_{2,1,f,t)} & n_{2,2,f,t)} & \cdots & n_{2,J,f,t} \\ \vdots & \vdots & \ddots & \vdots \\ n_{K,1,f,t} & n_{K,2,f,t)} & \cdots & n_{K,J,f,t} \\ n_{1,1,m,t} & n_{1,2,m,t} & \cdots & n_{1,J,m,t} \\ \vdots & \vdots & \ddots & \vdots \\ n_{K,1,m,t} & n_{K,2,m,t} & \cdots & n_{K,J,m,t} \end{pmatrix} \tag{1}$$

where $n_{k,j,s,t}$ represents the number of fish of age class j $(1 \leq j \leq J)$, length class k $(1 \leq k \leq K)$ and sex s $(f = \text{females}, m = \text{males})$ at the start of year t. The columns of this matrix are vectors that contain the numbers of fish within each of the age classes, *i.e.*

$$\boldsymbol{N}_t = \begin{pmatrix} \boldsymbol{N}_{1,t} & \boldsymbol{N}_{2,t} & \cdots & \boldsymbol{N}_{J,t} \end{pmatrix} \tag{2}$$

and where the number of fish of integer age a_j at the start of time step t, *i.e.* the number of fish in age class j, is written as the column vector:

$$\boldsymbol{N}_{j,t} = \begin{pmatrix} n_{1,j,f,t} \\ n_{2,j,f,t} \\ \vdots \\ n_{K,j,f,t} \\ n_{1,j,m,t} \\ n_{2,j,m,t} \\ \vdots \\ n_{K,j,m,t} \end{pmatrix} \tag{3}$$

The maximum age (years) of the simulated fish species considered within the model is denoted by A. Individual fish are classified into J age classes, where

$$J = \lfloor A \rfloor \tag{4}$$

$\lfloor A \rfloor$, the floor function of A, is the greatest integer age less than or equal to A, *i.e.* the greatest integer age explicitly considered in the model. The fish of integer age $a_j = j$ years are assigned to age class j. That is, the lower boundary of ages within age class j is a_j. The model considers only those fish within the population with integer ages greater than or equal to 1. Thus, recruitment is considered to represent the number of fish that recruit to age class 1, *i.e.* survive to an integer age of 1 year. Fish with integer ages greater than or equal to J are assigned to age class J (*i.e.* this age class is treated as a plus-group).

The model follows size cohorts of fish through time by classifying individuals into length classes according to their length at age. Individual length classes are identified using the subscript k $(1 \leq k \leq K)$, where K is the length class containing the largest fish considered by the model (*i.e.* this length class acts as a plus-group). The lower and upper bounds of length class k are denoted by L_k^{lower} and L_k^{upper}. Fish with lengths smaller than the lower bound of the first length class are assigned to length class 1. Although this specification allows for the use of length classes with different class intervals, the model that has been implemented for this study assumes that all length classes have a common length class interval, L_{int}. To ensure that the whole size range of lengths of individuals in the simulated fish species was covered, the number of length classes used in the model, which was implemented for this study, was set to:

$$K = 1.5 \max \left(L_{\infty,f}, L_{\infty,m} \right) / L_{\text{int}} \tag{5}$$

where $L_{\infty,s}$ is the asymptotic length of individuals of sex s. The lower and upper bounds of each length class, L_k^{lower} and L_k^{upper} , were determined as

$$L_k^{\text{lower}} = (k-1)L_{\text{int}} \tag{6}$$

$$L_k^{\text{upper}} = kL_{\text{int}} \tag{7}$$

L_k refers to fish of a length that is equal to the length of fish at the midpoint of length class k, and is determined as

$$L_k = (L_k^{\text{lower}} + L_k^{\text{upper}})/2 \tag{8}$$

Basic population dynamics

In broad terms, the population dynamics of the exploited fish stock is modelled as:

$$\boldsymbol{N}_{j,t} = \begin{cases} \boldsymbol{R}_t & \text{if } j = 1 \\ \boldsymbol{G}\boldsymbol{X}\boldsymbol{S}_{t-1}\boldsymbol{N}_{j-1,t-1} & \text{if } 1 < j < J \\ \boldsymbol{G}\boldsymbol{X}\boldsymbol{S}_{t-1}\boldsymbol{N}_{j-1,t-1} + \boldsymbol{G}\boldsymbol{X}\boldsymbol{S}_{t-1}\boldsymbol{N}_{j,t-1} & \text{if } j = J \end{cases} \tag{9}$$

where $\boldsymbol{R}_t$ is a vector containing the numbers of fish of age 1 of each sex within each length class, which recruit to the population in year t (see below), $\boldsymbol{S}_t$ is a diagonal matrix, the diagonal elements of which contain the probability that individuals of each length class and sex will survive till the end of year t, $\boldsymbol{X}$ is a matrix containing the probabilities that fish of each length class and sex will either change sex or remain of the same sex at the end of the time step, and $\boldsymbol{G}$ is a growth matrix containing the probabilities that fish will move to a new length class or remain within the same length class at the end of year t. That is, recruitment, survival, sex change and growth are considered as discrete events in the annual time step. Thus, the number of fish at the start of the time step is multiplied by the survival matrix to estimate the number of individuals surviving to the end of the year. The resulting vector is then multiplied by the sex change matrix to allow for sex change if the species is hermaphroditic. Finally, the vector is multiplied by the growth transition matrix to allow for the change in size composition that results from growth.

The number of fish within each length class and of each sex that recruit to age 1 in year t is $\boldsymbol{R}_t$, where

$$\boldsymbol{R}_t = \begin{pmatrix} R_{1,f,t} \\ R_{2,f,t} \\ \vdots \\ R_{K,f,t} \\ R_{1,m,t} \\ \vdots \\ R_{K,m,t} \end{pmatrix} \tag{10}$$

where $R_{k,s,t}$ is the number of recruits of sex s in length class k in year t. For each sex, the proportion of age 1 fish recruiting into length class k, $p_{k,s} = R_{k,s,t} / \sum_{k=1}^{K} R_{k,s,t}$, is calculated using the sex-specific von Bertalanffy growth curves, assuming a normal distribution of lengths around each age and normal distributions of the parameters around their point estimates and specified values of the respective standard deviations (see Equation 12). The expected number of age 1 fish in length class k and sex s in year t is given by

$$n_{k,1,s,t} = p_{k,s} \phi_s R_t \tag{11}$$

where ϕ_s is the proportion of recruits that are of sex s and the scalar variable R_t is the total number of recruits (over all length classes and both sexes) in year t. The values for $p_{k,s}$ were estimated using the NORMP routine of Allen Miller (latest revision - 30, March, 1986), based upon algorithm 5666 from Hart *et al.* (1968), "Computer approximations".

Growth

The patterns of growth of fish of each sex, or if the species is hermaphroditic and the growth of the two sexes is not conspicuously different, of the sexes combined, are described using the von Bertalanffy growth equation. $L(a, s)$, the length (mm) at age a of a fish of sex s, is determined as

$$L(a, s) = L_{\infty,s} \left\{1 - \exp\left[-k_s \left(a - t_{0,s}\right)\right]\right\} \tag{12}$$

where $L_{\infty,s}$ is the asymptotic length (mm) of individuals of this species and sex, k_s is the growth coefficient determining the rate (year^{-1}) at which the lengths of individuals of this sex approach the asymptotic length, and $t_{0,s}$ is the theoretical age (years) at which the expected length would be zero. Because the start of each model time step (*i.e.* biological year) corresponds to an assigned approximate birth date for the species, the lengths of fish determined for integer values of age, *e.g.* at a_j, the integer age of fish in the jth age class, using the above equation represent the lengths of fish of this integer age at the beginning of the time step.

An estimate of length at age for a fish within the jth age class midway through the annual time step, $L(a_j + 0.5, s)$, subsequently referred to as the length at mid-age, is given by

$$L(a_j + 0.5, s) = L_{\infty,s} \left\{1 - \exp\left[-k_s \left(a_j + 0.5 - t_{0,s}\right)\right]\right\} \tag{13}$$

During each biological year, individuals in a length class will grow and may move to a larger length class, or, if growth is insufficient, will remain in their current length class. A transition matrix, $\boldsymbol{G} = \{g_{j,k,s}\}$ is used to represent the probability that a fish of length class k will move to length class j due to the growth that occurs within each annual time step. Transition between size classes is assumed to be a discrete event that occurs at the end of the biological year. The growth transition matrix may be written as:

$$\boldsymbol{G} = \begin{pmatrix} g_{1,1,f} & g_{1,2,f} & \cdots & g_{1,K,f} & 0 & 0 & \cdots & 0 \\ g_{2,1,f} & g_{2,2,f} & \cdots & g_{2,K,f} & 0 & 0 & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots & \vdots & \vdots & \ddots & \vdots \\ g_{K,1,f} & g_{K,2,f} & \cdots & 1 & 0 & 0 & \cdots & 0 \\ 0 & 0 & \cdots & 0 & g_{1,1,m} & g_{1,2,m} & \cdots & g_{1,K,m} \\ 0 & 0 & \cdots & 0 & g_{2,1,m} & g_{2,2,m} & \cdots & g_{2,K,m} \\ \vdots & \vdots & \ddots & \vdots & \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & 0 & g_{K,1,m} & g_{K,2,m} & \cdots & 1 \end{pmatrix} \tag{14}$$

where $g_{j,k,s}$ is the proportion of fish of length class k and sex s that grow to length class j during a time step. Note that the columns of $\boldsymbol{G}$ sum to one. The assumption that there is no negative growth requires that $g_{j,k,s} = 0$ for all values of $j < k$.

Growth of fish in each time step is accounted for in the model after the relative numbers of fish that survive mortality in the current year have been determined, and if the species is hermaphroditic, sex change has taken place. The number of fish in each age-, length- and sex class is calculated by multiplying the vector containing the current numbers of fish in the stock of that age class within each length class for each sex, *i.e.* $\boldsymbol{X}\boldsymbol{S}_{t-1}\boldsymbol{N}_{j,t}$ by the growth transition matrix $\boldsymbol{G}$ to produce a vector containing the numbers of individuals of each sex within each length class for the age class after surviving natural and fishing mortality, and undergoing sex change (if any) and growth. For fish in age classes $j < J$, the resulting vector represents the number of fish within the next age class at the start of the next year, *i.e.*

$$\boldsymbol{N}_{j,t} = \boldsymbol{G}\boldsymbol{X}\boldsymbol{S}_{t-1}\boldsymbol{N}_{j-1,t-1} \tag{15}$$

Weight at length and age

The weight W (kg) of a fish of length L (mm) is calculated using the weight-length relationship:

$$W(L) = a_{\mathrm{WL}} L^{b_{\mathrm{WL}}} \tag{16}$$

where a_{WL} and b_{WL} are the parameters of this power-function. An estimate for the weight of fish of sex s in age class j at mid-age, $W(L(a_j + 0.5, s))$, is determined from their length at mid-age, $L(a_j + 0.5, s)$, as

$$W(L(a_j + 0.5, s)) = a_{\mathrm{WL}} L(a_j + 0.5, s)^{b_{\mathrm{WL}}} \tag{17}$$

Sex ratio

In gonochoristic species, the proportion of age 1 recruits that are of sex s (f= females, m= males), ϕ_s, is

$$\phi_f + \phi_m = 1 \tag{18}$$

Following recruitment, the proportion of fish within length class k that, for a gonochoristic species, are of a given sex is determined in the model by the growth of the individuals of each sex, and by the effects of fishing mortality and gear selectivity.

For hermaphroditic species, it as assumed that, in the absence of fishing mortality and gear selection, the probability that fish of length L are of the terminal sex s_{term}, $P_L^{s_{\mathrm{term}}}$, is a generalised logistic function of the length. Thus,

$$P^{s_{\mathrm{term}}}(L) = \phi_{s_{\mathrm{term}}} + \frac{\phi_{s_{\mathrm{term}}}^{\max} - \phi_{s_{\mathrm{term}}}}{1 + \exp\left[-\log_e(19)\frac{L - L_{50}^{s_{\mathrm{term}}}}{L_{95}^{s_{\mathrm{term}}} - L_{50}^{s_{\mathrm{term}}}}\right]} \tag{19}$$

where ϕ_{s_term} is the proportion of age 1 fish that are of the terminal sex, $\phi_{s_\mathrm{term}}^{\max}$ is the maximum proportion of individuals in an unexploited stock that will ultimately be of the terminal sex, and $L_{50}^{s_\mathrm{term}}$ and $L_{95}^{s_\mathrm{term}}$ are the lengths at which 50% and 95% of fish are of the terminal sex. This slightly generalised form of the logistic curve allows for diandric hermaphroditism, *i.e.* where some (but not all) individuals of the terminal sex have changed sex before having first become mature as the initial sex, and also for the possibility that not all individuals will ultimately change sex.

The probability that a fish in length class k is of the terminal sex, $P_k^{s_\mathrm{term}}$, is calculated as

$$P^{s_\mathrm{term}}\left(L_k\right) = \phi_{s_\mathrm{term}} + \frac{\phi_{s_\mathrm{term}}^{\max} - \phi_{s_\mathrm{term}}}{1 + \exp\left[-\log_e(19)\frac{L_k - L_{50}^{s_\mathrm{term}}}{L_{95}^{s_\mathrm{term}} - L_{50}^{s_\mathrm{term}}}\right]} \tag{20}$$

Sex change in hermaphroditic species

The probability that fish in length class k undergo sex change at the annual time step, X_k, may be calculated from the change in length that is expected to occur. From the von Bertalanffy growth curve, fish of length L_k and sex s would be expected to grow to $L_k + \left(L_{\infty,s} - L_k\right)\left(1 - \exp\left[-k_s\right]\right)$. Thus,

$$X_k = \frac{\left(P^{s_\mathrm{term}}\left(L_k + \left(L_{\infty,s} - L_k\right)\left(1 - \exp\left[-k_s\right]\right)\right) - P^{s_\mathrm{term}}\left(L_k\right)\right)}{\left(1 - P^{s_\mathrm{term}}\left(L_k\right)\right)} \tag{21}$$

The resulting probability X_k thus represents the fraction of individuals of hermaphroditic species within length class k that will change from the initial to the terminal sex over the annual time step. This may be written as $X_{k,f\to m}$ for a protogynous species, and as $X_{k,m\to f}$ for a protandrous species. For a gonochoristic or protogynous species, $X_{k,m\to f} = 0$, while for a gonochoristic or protandrous species, $X_{k,m\to f} = 0$. Thus, the sex transition matrix, $\boldsymbol{X}$, may be written as

$$\boldsymbol{X} = \begin{pmatrix} \boldsymbol{X}_{f\to f} & \boldsymbol{X}_{m\to f} \\ \boldsymbol{X}_{f\to m} & \boldsymbol{X}_{m\to m} \end{pmatrix} \tag{22}$$

where the sub-matrices $\boldsymbol{X}_{f\to f}$, $\boldsymbol{X}_{m\to f}$, $\boldsymbol{X}_{f\to m}$ and $\boldsymbol{X}_{m\to m}$ represent the proportions of females remaining as females, males changing to females, females changing to males and males remaining as males, respectively. These sub-matrices may be written as

$$\boldsymbol{X}_{f\to f} = \begin{pmatrix} 1 - X_{1,f\to m} & 0 & \cdots & 0 \\ 0 & 1 - X_{2,f\to m} & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & 1 - X_{K,f\to m} \end{pmatrix} \tag{23}$$

$$\boldsymbol{X}_{m\to f} = \begin{pmatrix} X_{1,m\to f} & 0 & \cdots & 0 \\ 0 & X_{2,m\to f} & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & X_{K,m\to f} \end{pmatrix} \tag{24}$$

$$\boldsymbol{X}_{f\to m} = \begin{pmatrix} X_{1,f\to m} & 0 & \cdots & 0 \\ 0 & X_{2,f\to m} & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & X_{K,f\to m} \end{pmatrix} \tag{25}$$

$$\boldsymbol{X}_{m\to m} = \begin{pmatrix} 1 - X_{1,m\to f} & 0 & \cdots & 0 \\ 0 & 1 - X_{2,m\to f} & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & 1 - X_{K,m\to f} \end{pmatrix} \tag{26}$$

where $X_{k,f\to m}$ is the proportion of fish of length class k that change sex from female to male at the end of the time step if the species is protogynous (zero otherwise) and $X_{k,m\to f}$ is the proportion of fish of length class k that change sex from male to female if the species is protandrous (zero otherwise).

Maturity

The proportion of fish in length class k that are mature is determined for each sex of gonochoristic species, and for the initial sex of hermaphroditic species, as

$$\psi_{k,s}^{\mathrm{mat}} = \left\{ 1 + \exp(-\log_e(19) \frac{L_k - L_{50,s}^{\mathrm{mat}}}{L_{95,s}^{\mathrm{mat}} - L_{50,s}^{\mathrm{mat}}} \right\}^{-1} \tag{27}$$

where $L_{50,s}^{\mathrm{mat}}$ and $L_{95,s}^{\mathrm{mat}}$ are the lengths at which 50% and 95% of individuals of that sex are mature, and L_k, which is the length of fish at the midpoint of the length class, is assumed to represent the average length of fish in this length class. Note that, for hermaphroditic species, it is assumed that all fish of the terminal sex are mature.

Fecundity

Two alternative approaches are used in the operating model to describe the relationship between fish length and fecundity. The first method estimates fecundity (batch or annual fecundity, depending on the input parameters) from a linear relationship between the natural logarithms of length and fecundity, whilst the other employs a cubic polynomial function, as used by Wise *et al.* (2007) for the fecundity of West Australian Dhufish (*Glaucosoma hebraicum*). Thus, the fecundity of females in length class k is denoted by $\mathrm{BF}_{k,f}$, and is determined from the length at the midpoint of the length class, L_k, as

$$\mathrm{BF}_{k,f} = \exp\left[a_{\mathrm{fec}} \log_e\left(L_k\right) - b_{\mathrm{fec}}\right] \tag{28}$$

or, in the case of *G. hebraicum* (Wise *et al.*, 2007)

$$\mathrm{BF}_{k,f} = \left(b_{\mathrm{fec}} L_{a_j+0.5,f} - a_{\mathrm{fec}}\right)^3 \tag{29}$$

where a_{fec} and b_{fec} are the parameters of these fecundity functions and which are specific for the fish species simulated. For immature fish where length $L_k \leq L^{\text{mat}}_{50,f}$, it is assumed that $\text{BF}_{a_j+0.5,f}$ equals zero.

Spawning biomass

The contribution to the mature biomass (kg) of a fish of sex s in length class k and sex s at the beginning of each biological year (*i.e.* at the time of spawning), $S_{k,s}$, is $\psi^{\text{mat}}_{k,s} W\left(L_k\right)$, where $\psi^{\text{mat}}_{k,s}$ is the proportion of fish of that length class and sex that are mature and $W(L_k)$ denotes the individual body mass of those fish. The total spawning biomass of each sex in the stock at each time step, $S_{s,t}$, is calculated as

$$B^{\text{sp}}_{s,t} = \sum_{j=1}^{J} \sum_{k=1}^{K} \psi^{\text{mat}}_{k,s} n_{k,j,s,t} W\left(L_k\right) \tag{30}$$

where $n_{k,j,s,t}$ is the number of fish of age class j, length class k and sex s in the stock.

Stock-recruitment

$\hat{R}_t$, the expected recruitment of age 1 fish (thousands of fish) in year t is assumed to follow the Beverton and Holt (1957) stock-recruitment relationship, and is calculated from the female spawning biomass in the preceding spawning season, $S_{f,t-1}$, as

$$\hat{R}_t = \frac{B^{\text{sp}}_{f,t-1}}{a_{\text{SRR}} + b_{\text{SRR}} B^{\text{sp}}_{f,t-1}} \tag{31}$$

where a_{SRR} and b_{SRR} are parameters of this function. Equilibrium recruitment in the absence of fishing, R^0, referred to as virgin recruitment, is calculated as

$$R^0 = (\text{SBR}^0_f - a_{\text{SRR}})/(\text{SBR}^0_f b_{\text{SRR}}) \tag{32}$$

where SBR_f^0 is the spawning biomass per recruit for females in an unexploited stock at equilibrium (Mace, 1994). The virgin spawning biomass (kg) of this unexploited stock at equilibrium, $S^{\mathrm{sp},0}$, is calculated as

$$S^{\mathrm{sp},0} = \mathrm{SBR}_f^0 R^0 \tag{33}$$

A re-parameterised form of the Beverton and Holt stock-recruitment relationship has been used in this study. This employs a steepness parameter, z, which is defined as the proportion of the virgin recruitment that is produced when the spawning biomass has been reduced to 20% of the virgin spawning biomass (Francis, 1992). Using the steepness parameter, the stock-recruitment parameters a_{SRR} and b_{SRR} can be calculated as functions of z, R_0 and S_0, where

$$a_{\mathrm{SRR}} = S^{\mathrm{sp},0}(1-z)/(4zR_0) \tag{34}$$

$$b_{\mathrm{SRR}} = (5z-1)/(4zR_0) \tag{35}$$

Recruitment variability

The operating model introduces inter-annual variability in recruitment of age 1 fish to the simulated fish stock by drawing for each year a random variate, ϵ_t, from a selected statistical distribution and calculating the annual recruitment, R_t, as

$$R_t = \hat{R}_t \exp\left[\epsilon_t - \frac{\sigma_R^2}{2}\right] \tag{36}$$

ϵ_t is the natural logarithm of the annual deviation in recruitment from its expected value, $\hat{R}_t$, which is calculated from the preceding year's spawning biomass using the Beverton and Holt stock-recruitment relationship, and σ_R^2 is the standard deviation of the normal distribution of the log-transformed values of recruitment. The term $-(\sigma_R^2)/2$ provides an adjustment to the value of the annual recruitment deviation that corrects for the bias in the mean value for recruitment, which arises as a result of the logarithmic transformation. Prior to commencing the MSE analysis, the distribution of ϵ_t may be selected from one of three alternative statistical distributions. The alternative distributions assume that

1. recruitment deviations are log-normally distributed.

$$\epsilon_t \sim N(0, \sigma_R^2) \tag{37}$$

2. recruitment deviations are log-normally distributed and auto-correlated between successive years.

$$\epsilon_t = \rho\eta_{t-1} + \left(1-\rho^2\right)^{0.5}\eta_t \text{ where } \eta_t \sim N(0, \sigma_R^2) \tag{38}$$

3. recruitment is episodic and auto-correlated. e_t is thus determined as

$$\epsilon_t = \exp\left[\rho\eta_{t-1} + \left(1-\rho^2\right)^{0.5}\eta_t - 1\right] \text{ where } \eta_t \sim N(0, \sigma_R^2) \tag{39}$$

Selectivity

The selectivity of the fish in length class k, *i.e.* the vulnerability of individuals of the length of fish at the midpoint of the length class to being caught by the fishing gear, is denoted by V_k. This selectivity is calculated as

$$V_k = \left\{1 + \exp\left[-\log_e(19)\frac{L_k - L_{50}^{\text{Vuln}}}{L_{95}^{\text{Vuln}} - L_{50}^{\text{Vuln}}}\right]\right\}^{-1} \tag{40}$$

where L_{50}^{Vuln} and L_{95}^{Vuln} are the lengths at which fish have vulnerabilities of 50% and 95% of fish that are fully vulnerable to the fishery.

Mortality

The survival matrix that results from the combined effects of natural and fishing mortality may be written as

$$\boldsymbol{S}_t = \begin{pmatrix} s_{1,f,t} & 0 & \cdots & 0 & 0 & \cdots & 0 \\ 0 & s_{2,f,t} & \cdots & 0 & 0 & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & s_{K,f,t} & 0 & \cdots & 0 \\ 0 & 0 & \cdots & 0 & s_{1,m,t} & \cdots & 0 \\ 0 & 0 & \cdots & 0 & 0 & \cdots & 0 \\ \vdots & \vdots & \ddots & \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & \cdots & 0 & 0 & \cdots & s_{K,m,t} \end{pmatrix} \tag{41}$$

where $s_{k,s,t}$ is the proportion of fish of length class k and sex s that survive both natural and fishing mortality, *i.e.* M and $F_{k,s,t}$, respectively, over the annual time step t. Note that the proportion surviving over the annual time step may be calculated as the product of the proportions surviving over each of the number of shorter time periods into which the year may be divided, thereby allowing the calculation of the effects of bag and boat limits with greater accuracy.

In the absence of fishing mortality, the total mortality of fish of sex s in length class k in year t is equal to the instantaneous rate of natural mortality. Thus, in this case, the proportion of fish that survive from the start to the end of the annual time step is calculated as

$$s_{k,s,t} = \exp(-M) \tag{42}$$

Natural mortality is estimated from the maximum age, A, of the fish species using Hoenig's (1983) mortality equation for fish, *i.e.*

$$M = \exp\left\{1.46 - 1.01 \log_e A\right\} \tag{43}$$

If the fish in the stock are subjected to fishing, and it is assumed that $F_{k,s,t}$ is the effective instantaneous rate of mortality of fish of sex s in length class k due to fishing in year t after allowing for all input and output controls, the instantaneous rate of total mortality (year^{-1}) for fish of that sex and length class in that year, $Z_{k,s,t}$, is assumed to equal the sum of the instantaneous rates of natural mortality (year^{-1}), M, and the length-, and sex-specific fishing mortality, $F_{k,s,t}$. That is,

$$Z_{k,s,t} = M + F_{k,s,t} \tag{44}$$

The fraction of fish that survive from the beginning to the end of the annual time step then becomes

$$s_{k,s,t} = \exp\left(-Z_{k,s,t}\right) \tag{45}$$

The value of the "initial equilibrium fishing mortality", F_{init}, which is input to the MSE prior to the start of the simulation runs represents the instantaneous rate of capture of fish in the absence of input or output controls. This, in combination with the input and output controls on fishing mortality, determines the initial state of the stock, *i.e.* the state when the stock is at an exploited equilibrium. After initialising the system state to values that represent this equilibrium state, the initial management strategy that is to be imposed must be specified or determined, and the instantaneous rate of capture and input and output management controls associated with this strategy must be applied by the operating model of the MSE over the requisite/specified projection period. After application of the input controls, the resulting instantaneous rate of capture will determine the probability that fish of different lengths and sexes are caught within year t, while the various output controls associated with the management strategy will determine whether the caught fish are landed or released. The level of discard mortality will determine whether released fish die as a result of capture and release, or survive. Note that the extent to which fishers comply with the input and output controls is a factor that also needs to be taken into account.

If $F_{k,s,t}$ is now considered to represent the instantaneous rate of capture after adjusting for the effects of all input controls, where this rate is dependent on length class k, sex s and time step t, an approximation to the proportion of fish that survive from the start to the end of the time step may be calculated as

$$s_{k,s,t} = \exp\left[-\left(Z_{k,s,t}\right)\right] + \frac{F_{k,s,t}}{Z_{k,s,t}}\left[1 - \exp\left(-Z_{k,s,t}\right)\right] p_{k,s,t}^{\mathrm{rel}}\left(1 - P^{\mathrm{rmort}}\right) \tag{46}$$

where $Z_{k,s,t} = M + F_{k,t,s}$ represents the total mortality if all captured fish were to be retained by fishers, $p_{k,s,t}^{\mathrm{rel}}$ is the fraction of the fish that, as a consequence of output controls, are released after capture rather than landed, and D is the proportion of fish that die after release. The fraction of fish that die as a result of fishing, either through capture and landing or through death following release as a consequence of barotrauma or hook-related injury is

$$\frac{F_{k,s,t}}{Z_{k,s,t}}\left[1 - \exp\left(-Z_{k,s,t}\right)\right]\left(1 - p_{k,s,t}^{\mathrm{rel}} + p_{k,s,t}^{\mathrm{rel}} P^{\mathrm{rmort}}\right) \tag{47}$$

while the fraction of fish that are caught and landed is

$$\frac{F_{k,s,t}}{Z_{k,s,t}}\left[1 - \exp\left(-Z_{k,s,t}\right)\right]\left(1 - p_{k,s,t}^{\mathrm{rel}}\right) \tag{48}$$

The above equations represent only an approximation to the true proportion surviving, however, as the release of fish may be considered to be a continuous process, which reduces the instantaneous rate at which the abundance of fish in the population declines. In the case of a MLL output control, the proportion of fish that are released depends on size, while in the case of a bag or boat limit, the proportion released is density-dependent.

The fishing mortality to be applied throughout the projection period, taking into account the various input controls, is determined by the exploitation component of the operating model. To calculate this fishing mortality, the routine adjusts the current level of fishing mortality, which, following the initialisation step, is F_{init}, by the extent to which fishing effort is modified, and by estimating the extent to which fishing mortality is influenced by the various input controls applied by the user to regulate fishing mortality. The effect of the various input and output controls is discussed in greater detail below.

Exploitation

The exploitation component of the operating model simulates the combined effects of various input and output management controls on the fishery, and thus on the resource. The fishing intensity exerted by the recreational fishers may be calculated as $E/(AT)$, where E is the fishing effort, A is the area over which the effort is applied, and T is the period over which the effort is applied (Gulland, 1969).

It is assumed in this study that effort is measured as the number of fishing trips by boats with a single recreational fisher on board. That is, the units of effort are ”fishing trips by boats with one fisher”, referred to subsequently in this document as ”fishing trips”. When calculating fishing effort, fishing trips by boats with more than one fisher on board must be converted to the equivalent number of fishing trips by boats with a single fisher.

The instantaneous rate of capture of fish, which are of sizes that are fully-vulnerable to the fishing gear, is denoted by F, and is proportional to fishing intensity. Thus,

$$F = \frac{qE}{AT} \tag{49}$$

where the constant q is referred to as the catchability coefficient. A is the area occupied by the fish stock and, for convenience, is typically assigned the value 1, in which case fishing intensity may be considered to be the average fishing effort per unit of time, where time is measured in years. In the absence of temporal closures, the period over which the recreational fishery operates is usually the full year, and thus T may also be considered to have the value 1. Temporal closures, which reduce the period over which recreational fishers may operate, are considered in more detail below. E is then the number of fishing trips undertaken by recreational fishers over the full year, standardised to the units in which fishing effort is measured, *i.e.* fishing trips by boats with a single fisher. In many other fishery models, F is considered to be the instantaneous rate of fishing mortality. However, as the model in this study considers the effect of output controls, which allow the release and possible survival of some of the fish that are caught, it is more appropriate in this study to refer to F as the instantaneous rate of capture of fish that are fully-vulnerable to the fishing gear.

Standardising the measure of fishing effort

The unit of fishing effort used in this study is a fishing trip by a boat with a single fisher on board. As noted above, fishing trips by boats with more than one fisher on board must be converted into the equivalent number of fishing trips by boats with a single fisher. The latter value is the number of fishing trips by a boat with a single fisher that would retain the same number of fish as would be retained by the boats with more than one fisher on board.

Effects of input management controls

Three alternative input management controls for regulating the level of fishing mortality being experienced by the simulated stock are considered by the model. These include (1) a proportional effort reduction control, (2) a temporal closure control, and (3) a spatial closure control.

Proportional effort reduction

The level of fishing mortality impacting the stock after a proportional reduction in fishing effort (control 1) has been applied, and which is applied during the projection period, F_t^{applied}, is calculated as

$$F_t^{\text{applied}} = F_t^{\text{current}} \left(1 - \psi_{E_{\text{reduced}}}\right) \tag{50}$$

where F_t^{current} is the instantaneous rate of capture prior to application of the input controls associated with the management strategy that is to be applied to the fish stock, and $\psi_{E_{\text{reduced}}}$ is the proportion by which fishing effort is required to be reduced (possibly zero). At the start of the simulation, F_t^{current} is set to the value of F_{init}, *i.e.* the user-specified equilibrium value for fishing mortality for the stock in its initial state. The new level of fishing mortality, F_t^{applied}, will be used as F_t^{current} when determining the fishing mortality to be applied following the next assessment.

The instantaneous rate of capture applied by fishers, F_t^{applied}, will be moderated by any temporal or spatial closures imposed by the management strategy. We discuss below the effect of imposition of a temporal closure, followed by the imposition of a spatial closure.

Temporal closure

The instantaneous rate of capture of fish in the stock, taking account of any temporal closure, F_t^{TC}, is determined as

$$F_t^{\mathrm{TC}} = F_t^{\mathrm{applied}} \left[1 - \psi_{t_{\mathrm{closed}}} \mathrm{TC}^{\mathrm{effect}}\right] \tag{51}$$

where $\psi_{t_{\mathrm{closed}}}$ is the proportion of the year that the fishery is closed to fishing (possibly zero) and $\mathrm{TC}^{\mathrm{effect}}$ is the effectiveness of the temporal closure. If the closure is 100% effective, the above equation reflects the application of the instantaneous rate of capture over a period that is less than or equal to the full year. $\mathrm{TC}^{\mathrm{effect}}$, which is assumed to be related to the duration of the temporal closure, is described as

$$\mathrm{TC}^{\mathrm{effect}} = \left\{1 + \exp\left[-\log_e(19)\frac{\psi_{t_{\mathrm{closed}}} - D_{50}^{\mathrm{effect}}}{D_{95}^{\mathrm{effect}} - D_{50}^{\mathrm{effect}}}\right]\right\}^{-1} \tag{52}$$

where D_{50}^{effect} and D_{95}^{effect} are the durations for which a temporal closure is 50% and 95% effective. The latter term is introduced to compensate for fishers' behaviour, as this typically results in additional effort being applied both before and after the closed season. The extent to which fishers can compensate for the loss of time is reduced as the period of closure is increased.

Spatial closure

It is assumed that, if a spatial closure is implemented, fishing effort will be displaced from the area closed to fishing to the area that remains open. It is also assumed that there is no movement of fish between the open and closed areas. Thus, while it is currently assumed in the model that there is no interchange of fish between closed and open areas, and therefore that the fish within the area closed to fishing are exposed only to natural mortality, those fish in the open area will experience an increase in instantaneous rate of capture. The value of the instantaneous rate of capture after further accounting for any spatial closure, F_t^{AC}, is therefore given by

$$F_t^{\mathrm{AC}} = \frac{F_t^{\mathrm{TC}}}{1 - \psi_{A_{\mathrm{closed}}}} \tag{53}$$

where $\psi_{A_{\mathrm{closed}}}$ is the proportion of the fished area that is closed to fishing. It is thus this level of fishing mortality that will determine the fraction of fish that are caught within the area open to fishing, during the period open to fishing, as a result of the fishing mortality applied by fishers. Note again that, in the absence of output controls, F_t^{AC}, which by convention is termed the ”fishing mortality”, represents both the instantaneous rates of capture and fishing mortality, but when output controls are introduced, only determines the instantaneous rate of capture.

The fraction of the fish in the area closed to fishing that experience only natural mortality, is $\psi_{A_{\mathrm{closed}}}$, while the fraction that experience the increased mortality associated with fishing is $1 - \psi_{A_{\mathrm{closed}}}$. In the absence of output controls, the proportion of the fully-vulnerable fish, which are alive at the start of the annual time step and which survive to the end of that time step, is

$$\psi_{A_{\mathrm{closed}}} \exp\left[-M\right] + \left(1 - \psi_{A_{\mathrm{closed}}}\right) \exp\left[-\left(M + F_t^{\mathrm{AC}}\right)\right] \tag{54}$$

Thus, in the absence of output controls, the overall fishing mortality experienced by the fish stock is

$$F_t = -\log_e\left[\psi_{A_{\mathrm{closed}}} + \left(1 - \psi_{A_{\mathrm{closed}}}\right) \exp\left(-F_t^{\mathrm{AC}}\right)\right] \tag{55}$$

The probability that a fish, which is fully vulnerable to the fishing gear, is caught is the product of the fraction of the population that is exposed to capture and the fraction of the population in the area open to fishing that is expected to be caught, *i.e.*

$$P^{\mathrm{cap}} = (1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_t^{\mathrm{AC}}}{M + F_t^{\mathrm{AC}}} \left[1 - \exp\left[-\left(M + F_t^{\mathrm{AC}}\right)\right]\right] \right\} \tag{56}$$

For fish that are not fully vulnerable to the fishing gear, the instantaneous probability of capture is reduced from F_t^{AC} by the relative vulnerability of those fish. Thus, if F_t^{AC} is the instantaneous probability of capture of fully vulnerable fish in year t, fish in length class k will experience a probability of capture in that year of

$$F_{k,t}^{\mathrm{AC}} = V_k F_t^{\mathrm{AC}} \tag{57}$$

Thus, the probability of capture of a fish in length class k in year t is

$$P_{k,t}^{\mathrm{cap}} = (1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_{k,t}^{\mathrm{AC}}}{M + F_{k,t}^{\mathrm{AC}}} \left[1 - \exp\left[-\left(M + F_{k,t}^{\mathrm{AC}}\right)\right]\right] \right\} \tag{58}$$

The probability of capture of a fish in length class k in a time step of duration τ in year t is

$$P_{k,t}^{\mathrm{cap}_\tau} = (1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_{k,t}^{\mathrm{AC}}}{M + F_{k,t}^{\mathrm{AC}}} \left[1 - \exp\left[-\left(M + F_{k,t}^{\mathrm{AC}}\right)\tau\right]\right] \right\} \tag{59}$$

Effects of output management controls

The output management controls considered by the model, *i.e.* those implemented to regulate the catches retained by fishers, include (1) a control on the minimum size at which fish are legally able to be retained by fishers, MLL, (2) daily bag and boat limits, and (3) a catch quota. The model also accounts for the effect of post-release mortality of fish of different sizes and ages, resulting from these controls.

MLL

Assuming that the lengths of fish within each length class are uniformly distributed, the probability that a fish in length class k is of a size greater than the MLL specified for that fish species, $P_k^{L\geq\mathrm{MLL}}$, is

$$P_k^{L\geq\mathrm{MLL}} = \begin{cases} 0 & \text{if } L_k^{\mathrm{upper}} < \mathrm{MLL} \\ 1 & \text{if } L_k^{\mathrm{lower}} \geq \mathrm{MLL} \\ \frac{L_k^{\mathrm{upper}} - \mathrm{MLL}}{L_k^{\mathrm{upper}} - L_k^{\mathrm{lower}}} & \text{otherwise} \end{cases} \tag{60}$$

where L_k^{lower} and L_k^{upper} are the lower and upper bounds of length class k.

The probability that a fish in length class k is captured in year t and has a length $\geq$ MLL , $P_{k,t}^{L\geq\mathrm{MLL,cap}}$, is

$$P_{k,t}^{L\geq\mathrm{MLL,cap}} = P_{k,t}^{\mathrm{cap}} P_k^{L\geq\mathrm{MLL}} \tag{61}$$

while the probability that a fish in length class k is captured in year t and has a length that is $<$ MLL , $P_{k,t}^{L<\mathrm{MLL,cap}}$, is

$$P_{k,t}^{L<\mathrm{MLL,cap}} = P_{k,t}^{\mathrm{cap}} (1 - P_k^{L\geq\mathrm{MLL}}) \tag{62}$$

The total expected catch of legal-sized fish over all age and size classes and both sexes, in the absence of bag limits and/or catch quota, C_t, is therefore

$$C_t = \sum_{s=f}^{m} \sum_{j=1}^{J} \sum_{k=1}^{K} P_{k,t}^{L\geq\mathrm{MLL,cap}} n_{k,j,s,t} \tag{63}$$

The probability that a fish in length class k is caught, has a length that is $<$ MLL and dies, $P_{k,t}^{L<\mathrm{MLL,dies}}$, is calculated as

$$P_{k,t}^{L<\mathrm{MLL,dies}} = P_{k,t}^{L<\mathrm{MLL,cap}} D \tag{64}$$

D is the probability that an undersized fish is illegally retained or is released and dies from injuries associated with being caught and released, and is determined as

$$D = (\psi^{\mathrm{comply}} P^{\mathrm{rmort}}) + (1 - \psi^{\mathrm{comply}}) \tag{65}$$

where ψ^{comply} is the proportion of fishers complying with the fishing regulations and P^{rmort} is the probability that fish will die if caught and then released.

The probability that a fish in length class k has a length that is $<$ MLL and survives after capture and release, $P_{k,t}^{L<\mathrm{MLL,surv}}$, is

$$P_{k,t}^{L<\mathrm{MLL,surv}} = P_{k,t}^{L<\mathrm{MLL,cap}} (1 - D) \tag{66}$$

The probability that a fish in length class k has a length that is $\geq$ MLL and is retained following capture because the fisher has not exceeded the bag and/or boat limit for that species, $P_{k,t}^{L\geq\mathrm{MLL ret}}$, is described by

$$P_{k,t}^{L\geq\mathrm{MLL,ret}} = P_{k,t}^{L\geq\mathrm{MLL,cap}} (1 - P_{\mathrm{BL}}^{\mathrm{rel}}) \tag{67}$$

where $P_{\mathrm{BL}}^{\mathrm{rel}}$ is the probability that a fish is released because of the bag and boat limit restrictions, an input parameter specified prior to model simulations.

Initially in this study, the effect of minimum legal length was modelled by calculating the catch of undersized fish over a period of fishing $\leq$ one year. However, it was recognised that release of fish modifies the instantaneous rate of mortality of released fish. Accordingly, the approach was later changed to one which modified the instantaneous rate of capture and thus the rate at which the number of fish in the stock declined. Thus, after allowing for the effect of the minimum legal length,

$$F_{k,t}^{\mathrm{MLL}} = F_{k,t}^{\mathrm{AC}} \left\{ P_k^{L \geq \mathrm{MLL}} + P_k^{L < \mathrm{MLL}} D \right\} \tag{68}$$

where the proportion of fish of legal size that are caught and retained in the annual time step is

$$(1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_{k,t}^{\mathrm{AC}} P_k^{L \geq \mathrm{MLL}}}{M + F_{k,t}^{\mathrm{MLL}}} \left[1 - \exp \left[- \left(M + F_{k,t}^{\mathrm{MLL}} \right) \right] \right] \right\} \tag{69}$$

and the proportion that survive to the end of the annual time step is

$$\psi_{A_{\mathrm{closed}}} \exp \left[-M \right] + (1 - \psi_{A_{\mathrm{closed}}}) \exp \left[- \left(M + F_{k,t}^{\mathrm{MLL}} \right) \right] \tag{70}$$

For a shorter time step of duration τ, the proportion of fish of legal size that are caught and retained in the time step is

$$(1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_{k,t}^{\mathrm{AC}} P_k^{L \geq \mathrm{MLL}}}{M + F_{k,t}^{\mathrm{MLL}}} \left[1 - \exp \left[- \left(M + F_{k,t}^{\mathrm{MLL}} \right) \tau \right] \right] \right\} \tag{71}$$

and the proportion that survive to the end of the time step is

$$\psi_{A_{\mathrm{closed}}} \exp \left[-M\tau \right] + (1 - \psi_{A_{\mathrm{closed}}}) \exp \left[- \left(M + F_{k,t}^{\mathrm{MLL}} \right) \tau \right] \tag{72}$$

Bag and boat limits

This section describes calculations in the model which estimate the proportional reduction in catch resulting from application of the output management controls, after having accounted for the input controls. The broad steps that are undertaken are:

1. Determine whether, when both bag and boat limits are applied, the bag or the boat limit constrains retained catches for trips by boats with different numbers of fishers.

2. Calculate the expected numbers of fish caught (*i.e.* fish retained and released) when no bag and/or boat limit is applied.

3. Calculate the fishing effort, *i.e.* the number of boat trips, based on an estimate of catchability derived from the data supplied to the MSE prior to commencing the simulations.

4. Calculate the mean CPUE for fishers when retained catches are, and are not, constrained by bag and/or boat limits.

5. Calculate the mean CPUE for retained and released fish for boats with different numbers of fishers.

6. Calculate the relative frequency distributions for total catches (*i.e.* unconstrained) and retained catches (*i.e.* constrained) for boats with different numbers of fishers.

7. Calculate the expected numbers of fish caught and retained by fishers when a bag and/or boat limit is applied.

8. Calculate the mean total and retained catch per boat with a given number of fishers.

9. Calculate the proportion of catch retained and released for boats with a given number of fishers.

10. Calculate the overall proportion of catch retained and released.

11. Estimate the expected catch (*i.e.* as the number of fish caught and retained).

12. Determine the proportion of the fish of each sex s, age class j, and length class k that survive capture and either retention or release and post-release mortality associated with their capture.

Determining if catches are constrained by the bag or boat limit, when both are applied

The effect of bag and boat limits is dependent on the number of fish that are caught within each fishing trip. The output control has an effect only if this catch exceeds the bag or boat limit. The boat limit, however, is typically determined by the number of fishers on board the boat during the fishing trip. The combined bag limit for the x fishers in the fishing party is $\mathrm{BagL}_x^{\mathrm{comb}}$, which may be calculated as

$$\mathrm{BagL}_x^{\mathrm{comb}} = x\mathrm{BagL} \tag{73}$$

where BagL is the individual bag limit for a fisher. If x is such that the combined bag limit for those fishers, $\mathrm{BagL}_x^{\mathrm{comb}}$, does not exceed the boat limit, BoatL, the total catch for the trip by the fishers in the boat is constrained by the combined bag limit, otherwise the total catch is constrained by the boat limit. The control that acts to constrain catches, BL_x, is thus determined as

$$\mathrm{BL}_x = \begin{cases} \mathrm{BoatL} & \text{if } \mathrm{BoatL} < \mathrm{BagL}^{\mathrm{comb}} \\ \mathrm{BagL}_x^{\mathrm{comb}} & \text{if } \mathrm{BoatL} \geq \mathrm{BagL}_x^{\mathrm{comb}} \end{cases} \tag{74}$$

Estimating total catch, effort and mean catch per unit of effort with no bag/boat limits

The equations that are presented below assume that, in the absence of bag and boat limits, the catch for a fishing trip by a boat with a single fisher is known and constant. However, as it is assumed that recruitment to the stock occurs only at the beginning of each annual time step, the instantaneous rate of capture is constant during the time step, and growth occurs at the end of the time step, the abundance of legal-sized fish in the stock will decline through the time step. Moreover, when bag and boat limits are introduced, some released fish will survive, and thus the decline in abundance will not be as great as that which would occur if there was no bag or boat limit.

When calculating the effects of the bag and boat limits in the model, however, it is assumed that the value of catch per unit of standard effort that would be obtained in the absence of the bag and boat limits, U_t, is the average value obtained by calculating the ratio of the theoretical catch that would be obtained in the absence of bag and boat limits and the estimate of standard fishing effort, F/q. In calculating the estimate of the theoretical catch (of retained and released fish) in the absence of a bag/boat limit, it is necessary to take the effects of the various input controls, of selectivity of the fishing gear and of the MLL regulation into account, and the resulting catch is the total over sex s, length class k and age class j.

In the initial formulation of this model, the average catch per standard boat trip over the full year was calculated using a single time step, and thus fails to take the change in catch rate within the year into account when estimating the effects of bag and boat limits. By dividing the year into n_τ smaller time steps of duration τ, a more accurate assessment of the effect of the bag and boat limit may be obtained. To facilitate the presentation of the methods that are used for this calculation, the number of fish of each sex s, within age class j and length class k, at the start of time step i (for $0 \leq i \leq I$) within year t, is denoted by $n_{k,j,s,t,i}$.

The equation used to calculate the probability of capture of a fish in length class k in year t, $P^{\mathrm{cap}}_{k,t}$, may be modified to represent the probability of capture of these fish in a time step of duration τ. Thus, this probability, $P_{k,t}^{\mathrm{cap}_\tau}$, may be written as

$$P_{k,t}^{\mathrm{cap}_\tau} = (1 - \psi_{A_{\mathrm{closed}}}) \left\{ \frac{F_{k,t}^{\mathrm{AC}}}{M + F_{k,t}^{\mathrm{AC}}} \left[1 - \exp\left[-\left(M + F_{k,t}^{\mathrm{AC}} \right) \tau \right] \right] \right\} \tag{75}$$

The probability that a fish in length class k is caught within time step i of year t and has a length $\geq$ MLL, $P_{k,t}^{L \geq \mathrm{MLL},\mathrm{cap}_\tau}$, is therefore

$$P_{k,t}^{L \geq \mathrm{MLL},\mathrm{cap}_\tau} = P_{k,t}^{\mathrm{cap}_\tau} P_k^{L \geq \mathrm{MLL}} \tag{76}$$

The probability that a fish in length class k is caught within time step i of year t and has a length $<$ MLL, $P_{k,t}^{L < \mathrm{MLL},\mathrm{cap}_\tau}$, is therefore

$$P_{k,t}^{L < \mathrm{MLL},\mathrm{cap}_\tau} = P_{k,t}^{\mathrm{cap}_\tau} \left(1 - P_k^{L \geq \mathrm{MLL}} \right) \tag{77}$$

The probability that a fish in length class k has a length that is $<$ MLL and survives after capture and release within time step i, $P_{k,t}^{L < \mathrm{MLL},\mathrm{surv}_\tau}$, is

$$P_{k,t}^{L < \mathrm{MLL},\mathrm{surv}_\tau} = P_{k,t}^{L < \mathrm{MLL},\mathrm{cap}_\tau} (1 - D) \tag{78}$$

The number of fish of sex s, age class j, and length class k that are caught in time step i in year t is therefore

$$P_{k,t}^{L \geq \mathrm{MLL},\mathrm{cap}_\tau} n_{k,j,s,t,i} \tag{79}$$

and thus, during time step i, the expected catch per standard boat trip in the absence of bag or boat limits is

$$U_t = \frac{\sum_{s=f}^{m} \sum_{j=1}^{J} \sum_{k=1}^{K} P_{k,t}^{L \geq \mathrm{MLL},\mathrm{cap}_\tau} n_{k,j,s,t,i}}{(F\tau)/q} \tag{80}$$

CPUE for boats with different numbers of fishers

The catch that is made by a boat during a fishing trip depends on the number of fishers on board the boat. Simplistically, the catch per boat trip might be expected to be proportional to the number of fishers, x. The number of fish in the immediate area under the boat is limited, however, and it is therefore likely that the catches made by individual fishers during the fishing trip will decline as x increases. That is, the relative efficiency of each fisher within a fishing party of x fishers is likely to decrease as the number of fishers increases. Thus, if U_t is the mean catch per unit of standard effort of legal-sized fish (before discard due to bag or boat limit, or catch quota) for an individual unit of fishing effort, *i.e.* a boat trip with a single recreational fisher on board, the total catch of legal-sized fish that is expected (before discard due to a bag or boat limit, or catch quota) for a boat trip when x fishers are on board, $U_{x,t}$, is assumed to be

$$U_{x,t} = xU_t \mathrm{re}^{(x-1)} \tag{81}$$

where $\mathrm{re}^{(x-1)}$ is the assumed relative efficiency of each individual in the fishing party when there are x recreational fishers in the fishing party. The value of an estimate of re is supplied as input to the MSE prior to the start of the simulation runs. Using this, estimates of the expected catch per trip by a boat with x recreational fishers on board, $U_{x,t}$, may be calculated for the time step.

Distributions of total and retained catches for trips by boats with different numbers of fishers

The catch on a single fishing trip may be considered to be a random variate drawn from the statistical distribution of such catches, where the mean of that distribution is related to the abundance of fish in the population within the area open to fishing at the time of the fishing trip, the vulnerability of those fish to the fishing gear, and the number of fishers on board the boat during the fishing trip. It is assumed in this study that the total number of legal-sized fish that is caught (before bag and boat limits or a catch quota) during a boat trip with x fishers on board is a random variate from a Poisson distribution with a mean catch per boat trip equal to $U_{x,t}$. The probability of capturing exactly y legal-sized fish, $P_{x,t}^{\text{cap}}\left(y|\mu = U_{x,t}\right)$, is therefore

$$P_{x,t}^{\text{cap}}\left(y|\mu = U_{x,t}\right) = \frac{U_{x,t}^{y}\exp\left[-U_{x,t}\right]}{y!} \tag{82}$$

The probability of capturing Y or less legal-sized fish may be determined from the cumulative distribution function for the Poisson distribution and is thus

$$P_{x,t}^{\text{cap}}\left(y \leq Y|\mu = U_{x,t}\right) = \frac{\Gamma\left(Y+1, U_{x,t}\right)}{Y!} \tag{83}$$

It follows that the probability of capturing Y or more legal-sized fish is

$$P_{x,t}^{\text{cap}}\left(y \geq Y|\mu = U_{x,t}\right) = 1 - \frac{\Gamma\left(Y, U_{x,t}\right)}{(Y-1)!} \tag{84}$$

If the combination of bag and boat limits constrains the total catch for the fishing trip by a boat with x recreational fishers on board to a maximum catch of BL_x fish, then the probability of capturing y fish, where $0 \leq y \leq \mathrm{BL}_x$ is

$$P_{x,t}^{\text{cap}}\left(y|\mu = U_{x,t}\right) = \begin{cases} \frac{U_{x,t}^{y}\exp\left[-U_{x,t}\right]}{y!} & \text{if } 0 \leq y < \mathrm{BL}_x \\ 1 - \frac{\Gamma\left(\mathrm{BL}_x, U_{x,t}\right)}{\left(\mathrm{BL}_x - 1\right)!} & \text{if } y = \mathrm{BL}_x \end{cases} \tag{85}$$

From this, the proportion of fish that are retained following capture, ψ^{ret}, and the proportion that die as a result of capture, $\left(1 - p^{\text{rmort}}\right)\psi^{\text{ret}} + p^{\text{rmort}}$, may be calculated. An estimate of retained catch may then be determined as

$$\sum_{s=f}^{m}\sum_{j=1}^{J}\sum_{k=1}^{K} P_{k,t}^{L \geq MLL,\text{cap}_\tau} n_{k,j,s,t,i}\psi^{\text{ret}} \tag{86}$$

and the number of fish of sex s, length class k, and age class j, that survive to the beginning of time step i, after allowing for the mortality associated with capture and release is

$$\begin{aligned} n_{k,j,s,t,i} &= n_{k,j,s,t,i-1}\left\{ \psi_{A_{\mathrm{closed}}} \exp\left[-M\right] + \left(1-\psi_{A_{\mathrm{closed}}}\right) \exp\left[-\left(M+F_{k,t}^{\mathrm{AC}}\right)\right] \right. \\ &\quad \left. + P_{k,t}^{L<\mathrm{MLL},\mathrm{surv}_\tau} + P_{k,t}^{L\geq MLL,\mathrm{cap}_\tau} \psi^{\mathrm{rel}}\left(1-P^{\mathrm{rmort}}\right) \right\} \end{aligned} \tag{87}$$

Proportion of catch that is released when bag and boat limits are applied

Details of the distribution of the relative numbers of fishing trips by boats in which the fishing party contains x $(1 \leq x \leq X)$ fishers are specified as input to the MSE prior to commencing the simulation trials. The maximum number of fishers within any fishing party is denoted by X, and the proportion of trips in which there are x fishers is denoted by P_x. When bag and boat limits are applied, the average catch for a fishing trip by a boat with x fishers on board is reduced from $U_{x,t}$ to

$$\sum_{y=1}^{\mathrm{BL}_x} y P_{x,t}^{\mathrm{cap}}\left(y|\mu = U_{x,t}\right) \tag{88}$$

The number of boat trips by a vessel with x fishers on board may be converted to the equivalent standard effort, *i.e.* the number of boat trips with a single fisher on board that would retain the same catch for the trip, by multiplying by the factor

$$\frac{\sum_{y=1}^{\mathrm{BL}_x} y P_{x,t}^{\mathrm{cap}}\left(y|\mu = U_{x,t}\right)}{\sum_{y=1}^{\mathrm{BL}_1} y P_{1,t}^{\mathrm{cap}}\left(y|\mu = U_{1,t}\right)}, \tag{89}$$

Note that this factor takes the combined effect of the bag and boat limits on boat trips with different numbers of fishers into account.

If the mean catch per unit of standard effort of legal-sized fish is U_t, the average catch (retained and released fish) per trip over all fishing boats, regardless of the number of fishers on board each boat, is

$$\sum_{x=1}^{X} U_{x,t} P_x \tag{90}$$

and the average number of fish that are caught and retained per trip by each boat, regardless of the number of fishers on board, is

$$\sum_{x=1}^{X}\sum_{y=1}^{\mathrm{BL}_x} y P_{x,t}^{\mathrm{cap}}\left(y|\mu = U_{x,t}\right) P_x \tag{91}$$

The proportion of fish that are retained in the average trip when bag and boat limits are applied, ψ^{ret}, is therefore

$$\psi^{\mathrm{ret}} = \frac{\sum_{x=1}^{X}\sum_{y=1}^{\mathrm{BL}_x} y P_{x,t}^{\mathrm{cap}}\left(y|\mu = U_{x,t}\right) P_x}{\sum_{x=1}^{X} U_{x,t} P_x} \tag{92}$$

while the proportion of fish that are required to be released because of the bag and boat limits, ψ^{rel} is

$$\psi^{\mathrm{rel}} = 1 - \psi^{\mathrm{ret}} \tag{93}$$

Of the fish that are released, a proportion P^{rmort} are expected to die as a result of barotrauma, hooking injury, or other mortality associated with capture and release. Thus, by implementing the bag and boat limit controls, the proportion of fish that are expected to die is the sum of the proportion retained and the proportion that are released but die as a result of capture and release, *i.e.*

$$\psi^{\mathrm{ret}} + \left\{1 - \psi^{\mathrm{ret}}\right\} P^{\mathrm{rmort}} \tag{94}$$

which may be simplified to

$$\left(1 - P^{\mathrm{rmort}}\right) \psi^{\mathrm{ret}} + P^{\mathrm{rmort}} \tag{95}$$

It should be noted that the effectiveness of the bag and boat limit regulation will be reduced if fishers fail to comply with the regulations, or if high-grading occurs.

Catch quota

If either a catch quota is not applied, or it is applied and the size of the quota, Q, is equal to or greater than the expected retained catch, N^{ret}, the number of fish of each age, length class and sex, $N_{a_j,l_k,s}$, at the end of the time step is calculated after removing the fish that die. If $Q < N^{\mathrm{ret}}$, then $N_{a_j,l_k,s}$ is calculated by removing the fish that die and reducing probability of capture, P^{cap}, to account for the effect of the quota. The level to which P^{cap} should be reduced because of the quota is estimated iteratively, calculating the expected number of retained fish after resetting P^{cap} as

$$P^{\mathrm{cap}} = P^{\mathrm{cap}} Q / N^{\mathrm{ret}} \tag{96}$$

This process terminates when $(N^{\mathrm{ret}} - Q)/Q$ is zero.

The unexploited stock

Determination of the values of the parameters of the stock-recruitment relationship requires an estimate of the unexploited spawning biomass to be calculated. The fishery is assumed to be at equilibrium with annual recruitment equal to the average level of recruitment. Since the virgin stock is assumed to be unexploited, input and output controls have no impact and the only source of mortality is natural mortality.

The calculation commences by calculating the proportion of the fish that will lie within each age and length class when survival is determined from natural mortality. Thus, the operating model is run with recruitment set to the level of annual recruitment, R^0, entered as input to the MSE, repeating the sequence of addition of recruitment, and calculating the effects of survival, sex change (if hermaphroditic), and growth over sufficient years for the system state to achieve an approximate equilibrium state. The spawning biomass of the unexploited stock, $S^{\mathrm{sp},0}$, is then calculated and the values of the parameters a_{SRR} and b_{SRR} of the stock-recruitment relationship determined from $S^{\mathrm{sp},0}$, R^0, and z.

The initial exploited equilibrium state of the stock

This section gives a brief description of the sequence of calculations undertaken by the model to establish the initial state of the simulated fish stock prior to the projection period. This initial state is set to reflect that which would be expected if the population was at an exploited equilibrium and the fishery was subject to a suite of management controls. In broad terms, when there are no bag or boar limits, the operating model determines the initial state of the stock by:

1. Calculating the expected number of fish per recruit (assuming the stock is at equilibrium) within each length class, of each sex and age, allowing for total mortality in the stock and taking into account growth and, in hermaphroditic species, sex change. This is accomplished by setting the annual recruitment to one fish, and running the model with the initial level of the instantaneous rate of capture, F_{init}, for sufficient years to ensure that the model's representation of the system state has reached equilibrium. The model takes any initial spatial and temporal closures, and minimum legal length regulations into account when these calculations are undertaken.

2. Calculating the spawning biomass per recruit and the expected level of annual recruitment to the stock when at equilibrium. The expected level of average annual recruitment is calculated by considering the equilibrium spawning biomass as the product of spawning biomass per recruit and the equilibrium level of recruitment, then solving the stock-recruitment relationship to determine the average annual level of recruitment.

3. Multiplying the expected number of fish per recruit that is caught and retained by the

average recruitment to estimate the average catch.

4. Dividing the average catch by the mean catch per fishing trip for the initial equilibrium state of the fishery, which was specified to the MSE prior to commencing the simulations, to produce an estimate of the fishing effort (number of standard boat trips, *i.e.* equivalent number of trips by boats with a single fisher on board.

5. Dividing the initial instantaneous rate of capture F_{init} by the fishing effort to obtain an estimate of the catchability coefficient,q.

6. Calculating the initial age- and length-compositions of the fish stock for a time series of annual recruitment levels, where the latter are determined by randomly selecting variates from the statistical distributions that describe the deviations of annual recruitment from the average level. These calculations are thus determined by undertaking, within each year throughout the historical period, the following events:

   (a) calculate the annual recruitment R to the stock, taking into account the fact that, in accordance with the specifications provided to the MSE before commencing the simulation, recruitment may be variable or episodic, and that it may also be auto-correlated between successive years;

   (b) add the new recruits produced each year to the stock at the beginning of the annual time steps;

   (c) allow for the instantaneous rate of capture for the historical year (and the effect of input and output controls in that year), possible sex change (in hermaphroditic fish) and growth of fish in that year; and

   (d) update the age- and length-compositions of the stock at the end of the time step, after which the process is repeated for the next year of historical data, and so on, until the system state at the end of the last year of historical data has been determined.

THE OBSERVATION MODEL

The observation model simulates the collection of age- and length-composition data to be used by the assessment model to obtain information about the state of the exploited stock. These types of data are those most commonly applied to stock assessments for recreational fisheries in Western Australia (Wise *et al.*, 2007). The model estimates the age- and length-compositions of the stock from the number of fish present in the population at the beginning of the time step in which sampling is undertaken. As the samples represent catches taken by recreational fishers, samples are only drawn for fish with lengths $\geq$ MLL. The vulnerability of fish in length class k to being caught by fishers, *i.e.* $V_{k,s}^{\mathrm{MLL}}$, which depends on the selectivity of the fishing gear for fish in that length class V_k and the length of the fish in the length class L_k relative to the MLL, $P_k^{L\geq\mathrm{MLL}}$, is

$$V_{k,s}^{\mathrm{MLL}} = V_k P_k^{L\geq\mathrm{MLL}} \tag{97}$$

The expected frequency of fish within age class j for each sex, $N'_{j,s}$, is proportional to the sum over all length classes of the product of this vulnerability and the number of fish within the length classes, *i.e.*

$$N'_{j,s} = \sum_{k=1}^{K} n_{k,j,s,t} V_{k,s}^{\mathrm{MLL}} \tag{98}$$

where $n_{k,j,s,t}$ is the number of fish of sex s in length class k and age class j. The expected frequency of fish of length class k and sex s, over all ages, $\hat{N}_{k,s}$, is calculated as

$$\hat{N}_{k,s} = \sum_{j=1}^{J} n_{k,j,s,t} V_{k,s}^{\mathrm{MLL}} \tag{99}$$

Assuming that the age- and length-compositions of the stock are multinomially distributed, samples are generated by drawing random observations, ix, from the expected distributions for age, $N'_{j,s}$, and length, $\hat{N}_{k,s}$, for the simulated population, employing the algorithm described by Devroye (1986). The arguments of the algorithm are n, the number of observations (fish), P_i, the expected proportion of fish in the age or length category i, n_{cat}, the number of age or length categories and ix_i, the i^{th} random observation.

THE ASSESSMENT MODEL

The model assesses the state of the simulated fish stock employing catch curve and per recruit analyses, the details of which are provided below.

Catch curve analyses

The model enables a range of alternative types of catch curve analyses to be employed for estimating total mortality from age composition sample data.

Catch curve 1

Linear regression of natural logarithms of frequencies at age (Ricker, 1975)

Model assumptions: (1) annual recruitment is constant, (2) the total mortality of fish above the age at which fish are fully recruited into the fishery is constant, and (3) the frequencies of fish at age in the age composition samples are log-normally distributed about their expected values. The integer age at which fish are fully recruited into the fishery, t_c, was determined as one year above the modal age in the age-frequency sample.

From these assumptions, the number of fish in age class j, *i.e.* of integer age a_j, in the age composition sample, N'_{a_j}, may be written as

$$N'_{a_j} = N'_{t_c} \exp\left[-Z\left(a_j - t_c\right)\right] \exp\left[\epsilon_j\right] \tag{100}$$

where $\epsilon_j \sim N(0, \sigma^2)$ and $t_c \leq a_j \leq a_{\max}$, and where $a_{\max}$ is selected as the integer age of the last age class for which there is a non-zero frequency. By taking the natural logarithms of both sides of this equation, the relationship may be expressed as the linear regression model

$$\log_e N'_{a_j} = \log_e N'_{t_c} - Z(a_j - t_c) \tag{101}$$

where total mortality, Z, represents the negative of the slope of the regression equation.

Catch curve 2

Frequencies at age represent a sample from a multinomial distribution

Model assumptions: (1) annual recruitment is constant, (2) the total mortality of fish above the age at which fish are fully recruited into the fishery is constant, and (3) the age composition sample represents a sample drawn from a multinomial distribution where the expected proportions at age are determined from the expected number of fish surviving to each integer age.

The estimated proportions of fish at integer age a_j, where $t_c \leq a_j \leq A$, in year t, denoted $\hat{P}_{a_j,t}$, is calculated as

$$\hat{P}_{a_j,t} = \frac{N'_{a_j,t} \exp\left[-Z(a_j - a_r)\right]}{\sum_{a_j=t_c}^{A} N'_{a_j,t} \exp\left[-Z(a_j - t_c)\right]} \tag{102}$$

where $N'_{t_c,t}$ is constant, *i.e.* $N'_{t_c,t} = N'_{t_c}$(Hall *et al.*, 2004; Coulson *et al.*, 2009). Note that the model may be fitted to the data for one year or for pooled data from samples drawn from several consecutive years. The log-likelihood, λ, of the observed age frequencies can be calculated as

$$\lambda = \sum_t \sum_{a_j=a_r}^{A} N'_{a_j,t} \log\left(\hat{P}_{a_j,t}\right) \tag{103}$$

Catch curve 3

Relative abundance analysis (Deriso et al., 1985)

Model assumptions: As in catch curve 2, except that annual recruitment is assumed to be variable. The relative levels of recruitment of the different year classes, R_y, are estimated as parameters in the model.

Per recruit analyses

The estimated yield per recruit for fish of sex s at the estimated level of fishing mortality F, $\mathrm{YPR}_{F,s}$, is calculated as

$$\mathrm{YPR}_{F,s} = \sum_{j=1}^{J}\sum_{k=1}^{K} \frac{V_k F}{M + V_k F}\left[1 - \exp\left(-(M + V_k F)\right)\right] W(L_k)\psi_k^s \exp\left(-(M + V_k F)\right) \quad (104)$$

where $W_{L_k,s}$, is the estimated weight of fish and $\psi_{k)}^s$ is the proportion of fish of sex s in length class k. The estimate of fishing mortality used in this analysis is the value derived from the catch curve analysis.

The spawning stock biomass per recruit for sex s at fishing mortality F, $\mathrm{SBR}_{F,s}$, is determined as

$$\mathrm{SBR}_{F,s} = \sum_{j=1}^{J}\sum_{k=1}^{K} W(L_k)\psi_k^s \psi_{k,s}^{\mathrm{mat}} \exp\left[-(M + V_k F)\right] \quad (105)$$

where $\psi_{k,s}^{\mathrm{mat}}$ is the expected proportion of fish of sex s and length class k that are mature. The estimated number of eggs per recruit at F, EPR_F, is

$$\mathrm{EPR}_F = \sum_{j=1}^{J}\sum_{k=1}^{K} \mathrm{BF}_{k,f} W(L_k)\psi_k^s \psi_{k,s}^{\mathrm{mat}} \exp\left[-(M + V_k F)\right] \quad (106)$$

where $\mathrm{BF}_{k,f}$ is the fecundity of females in length class k.

The spawning potential ratio, SPR, in terms of spawning stock biomass per recruit and eggs per recruit, is calculated as

$$\mathrm{SPR(SBR)} = \mathrm{SBR}_{F,s}/\mathrm{SBR}_{0,s} \tag{107}$$

$$\mathrm{SPR(EPR)} = \mathrm{EPR}_{F}/\mathrm{EPR}_{0} \tag{108}$$

where $\mathrm{SBR}_{0,s}$ and $\mathrm{EPR}_{0,s}$ are the estimated levels of spawning stock biomass per recruit and eggs per recruit, respectively, for the stock at its virgin state.

THE DECISION-MAKING PROCESS

Depending on the type of analysis to be undertaken, management decisions during model simulation runs can be determined according to either a fixed decision rule or through directly specifying the management controls to be applied. In accordance with traditional MSE models, the fixed decision rule is used to automatically adjust the management, given the current state of an exploited stock relative to user-specified reference points. The application of fixed decision rules enables prediction of the likely effectiveness of alternative sets of pre-defined management strategies over an extended time frame.

Two alternative pre-determined decision rules, both which are F-based, have been implemented in the model. These act to proportionally reduce the level of fishing mortality depending on the estimated level of F for the stock at its initial state (or at the end of the previous projection period), relative to the specified reference points. The first decision rule calculates the proportional reduction in F, $\psi_{F_{\mathrm{reduced}}}$, as

$$\psi_{F_{\mathrm{reduced}}} = \begin{cases} 0 & \text{if } F < F_{\mathrm{thr}} \\ 0.5 & \text{if } F_{\mathrm{thr}} \leq F < F_{\mathrm{lim}} \\ 0.75 & \text{if } F \geq F_{\mathrm{lim}} \end{cases} \tag{109}$$

where F_{thr} and F_{lim} are the threshold and limit reference point values for F, as specified by the user.

The second decision rule rather determines $\psi_{F_{\mathrm{reduced}}}$ as

$$\psi_{E_{reduced}} = \begin{cases} 0 & \text{if } F < F_{\mathrm{thr}} \\ 1 - (1/(F/F_{\mathrm{tar}})) & \text{if } F \geq F_{\mathrm{thr}} \end{cases} \tag{110}$$

where F_{tar} is the target reference point values for F.

SOFTWARE IMPLEMENTATION

Microsoft Visual Basic.NET (version 3.5 SP1) in Visual Studio 2008 Express Edition (version 9.0.21022.8 RTM; Microsoft, 2007) was used as the primary platform for model development, with AD Model Builder (Otter Research Ltd.) being employed to undertake the catch curve analyses, some of which are computationally intensive.

The model is freely available for download from the website http://cffrfisheriesmodelling.net.

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