Using Management Strategy Evaluation to address problems arising as a result of competing users of the South African horse mackerel resource

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Declaration

I hereby declare that this thesis has not been submitted to this or any other university for a degree, either in the same or different form.

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Abstract

The Cape horse mackerel (*Trachurus trachurus capensis*) has traditionally made an important contribution to the South African fishing industry and is a key component of the Benguela ecosystem. This thesis concerns the assessment and management of the South African horse mackerel resource. It starts with a brief review of the biology of the Cape horse mackerel and the history of the fishery, as well as of the Management Strategy Evaluation approach, which was applied in this work.

Assessments of the horse mackerel resource are currently undertaken through the combined efforts of the Demersal and Pelagic Scientific Working Groups (SWGs) of the South African Department of Agriculture, Forestry and Fisheries (DAFF). A joint effort is required because the resource is available to multiple fisheries: as directed catch to the midwater trawl fishery and as bycatch to the demersal trawl and pelagic purse-seine fisheries. Management of the resource is complicated by differences in the age-structures of the horse mackerel caught in each of these three fisheries.

The data available for the assessments are described, including the details of their collection and processing. Four age-structured production models (each reflecting different assumptions about the horse mackerel resource) are fitted to those data using the maximum-likelihood estimation method, and are used to provide assessments. Estimates of the current status of the stock indicate that it is healthy, putting it well above its Maximum Sustainable Yield (MSY) level. For the directed midwater fishery, MSY is estimated to be in the region of 50 000–100 000 tonnes per annum. However, the results of constant catch projections suggest that there is a pronounced yield-per-recruit effect, with even small bycatches of juvenile horse mackerel in the pelagic fishery having a pronounced negative effect on the level of a catches in the midwater fishery that can be sustained.

Since 2000, the annual bycatch of horse mackerel in the pelagic fishery had been regulated by a

Precautionary Upper Catch Limit (PUCL) of 5 000 tonnes. However, it became apparent that this fixed allocation was insufficient during years with high juvenile horse mackerel availability when, in 2011, the fishery was threatened with closure long before the quotas for its target species could be filled. A Management Procedure (MP) therefore needed to be developed that would provide flexibility in the annual PUCLs and thereby minimise operational constaints on the pelagic fishery. The performances of a number of Candidate Management Procedures (CMPs) are evaluated in this thesis through simulation testing and expressed in terms of expected catch and risk of resource depletion. Following discussions in the DAFF SWGs, a MP that limits the total bycatch over a three year period was selected for implementation and has been used to provide PUCL recommendations for the 2013 and following fishing seasons.

There are indications that estimates of horse mackerel abundance based on demersal swept-area surveys may be negatively biased, and hence that the resource is underutilised. Therefore, a MP is developed that experimentally increases the Total Allowable Catch (TAC) for the directed midwater fishery without increasing the risk of resource depletion. The CMPs considered use trends in abundance indices to monitor the response of the stock to management actions, and then adjust the annual allocation accordingly. Again, their performances are evaluated through simulation testing. The results show that the CMPs offer various trade-offs between improved catches and increased interannual TAC variability, and also in better short-term catch performance at the expense of lower long-term TACs. A MP that restricts the TAC to a 10% increase and 15% decrease per annum was chosen from the candidates considered and has subsequently been used by DAFF to recommend the midwater TACs for the 2013 and following fishing seasons.

Future suggestions for improving the assessment and management of the South African horse mackerel resource are discussed. These include adopting a Bayesian approach to refine the estimates of assessment precision and allow the incorporation of prior knowledge, investigating alternate approaches to modelling pelagic bycatches during projections and improving the model fits to catch-at-length data. Future research needs to provide estimates of the bias in the horse mackerel abundance estimates that are currently based on demersal swept-area surveys, possibly by performing concurrent trawl and hydro-acoustic surveys. Additionally, a wider range of CMPs than those considered in this thesis should be investigated for both the pelagic and midwater fisheries—the recommended CMP for the former did not seem to appreciably decrease the probability of disruptions to that fishery, while that for the latter did not consistently award larger increases in TAC when the resource was assumed to be more productive. Furthermore, robustness trials need to be conducted for all CMPs to ensure that they exhibit reasonable performance given a wider variety of plausible scenarios for the resource's dynamics.

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Contents

Chapter 1

Introduction

The South African horse mackerel resource is available to multiple fleets: as juveniles to the pelagic purse-seine fishery on the West Coast, and as adults which are concentrated mainly on the South Coast to the demersal and midwater trawl fisheries. Allocations for directed fishing are made to the midwater fishery only, for which yield-per-recruit is appreciably better than for the pelagic fishery (Horsten, 1999b). To allow for unavoidable bycatches of horse mackerel, relatively small allocations are made to the demersal fishery, which targets hakes, and to the pelagic fishery, which focuses on sardine and anchovy.

The pelagic fishery bycatch of horse mackerel had previously been regulated by a Precautionary Upper Catch Limit (PUCL) of 5 000 tonnes per annum. However, in 2011 large bycatches of juvenile horse mackerel in excess of allocations became impossible for that fishery to avoid, following good recruitment to the resource. The Department of Agriculture, Forestry and Fisheries (DAFF)* therefore agreed to an *ad-hoc* increase of 7 000 tonnes to the PUCL for that year only. However, concerns were raised that catches by the midwater fishery might be compromised should such upwards *ad-hoc* adjustments of the PUCL be required in the future. This phenomenon of "boom and bust" recruitment in the horse mackerel stock brought into question the appropriateness of the policy of a fixed allocation to the pelagic fishing sector.

A further management difficulty is that data for horse mackerel for assessment purposes are limited. The previous Total Allowable Catch (TAC) of 31 500 tonnes per annum for the midwater fishery is based on the assumption that swept-area abundance estimates from demersal research surveys provide unbiased estimates of abundance in absolute terms. In reality, these estimates

*Note that although the Fisheries Management branch of DAFF was previously Marine and Coastal Management, for simplicity that organisation will also be referred to as DAFF hereafter.

are certainly negatively biased because they ignore fish in the midwater, but to an unknown extent. The resources may therefore be substantially underutilised.

This work aims to address both of these issues using the Management Strategy Evaluation (MSE) framework. Suggestions have been made that annual pelagic hydro-acoustic surveys could serve as predictors of the strength of horse mackerel recruitment. If this hypothesis proves correct, then a Management Procedure (MP) could be developed that determines the extent to which the annual pelagic fishery allocation could be varied safely, based on those survey results. For the midwater fishery, this thesis seeks a MP that experimentally adjusts the TAC so as to secure improved utilisation without undue increase in the risk of unintended reduction of resource abundance. The performances of all Candidate Management Procedures (CMPs) considered in this thesis are evaluated through rigorous simulation testing.

1.1 Thesis overview

This thesis is comprised of seven chapters.

Chapter 2 provides a background to the work. It gives details on the management history and biology of South African horse mackerel. The MSE framework is also outlined, including its differences from the conventional "best assessment" approach, as well as its perceived benefits and drawbacks.

Chapter 3 lists the data available to assess the South African horse mackerel resource. It describes their collection, limitations and also how the raw data are analysed so as to be usable by this assessment. The chapter then concludes with an appendix on the Generalised Linear Model (GLM) standardisation procedure that was applied in order to develop a catch-per-unit-effort (CPUE) series from commercial midwater trawling data.

Chapter 4 reports on the horse mackerel age-structured production models (ASPMs) that form the Reference Set of operating models (OMs) for CMP testing. It also provides the results of corresponding sensitivity tests and constant catch projections. The full technical details of the models are given in an appendix. A second appendix pertains to the modelling of fishing selectivity for the midwater and demersal fleets.

Chapter 5 starts with an evaluation of the reliability of the pelagic hydro-acoustic survey results as predictors of horse mackerel recruitment. Those findings are then taken into account when constructing CMPs for the pelagic fishery. Finally, the performance of the CMPs are evaluated through simulation testing and reported in terms of performance statistics.

Chapter 6 is similar to the previous chapter in that it describes the process undertaken in developing and evaluating CMPs. However, these CMPs apply to the directed midwater fishery.

The thesis concludes with Chapter 7, which summarises the key findings from the preceding chapters. Additionally, recommendations towards improving the assessment and management of the horse mackerel resource are provided, along with general thoughts about the MSE approach.

Chapter 2

Technical Background

This chapters provide a context to this thesis. It starts with a discussion of the MSE approach to fisheries management that was adopted for this work. Next, details of the biology of Cape horse mackerel are given. The chapter concludes with an outline of the history of the South African horse mackerel fishery.

2.1 The Management Strategy Evaluation approach

Rademeyer *et al.* (2007) define a MP as “the combination of pre-defined data, together with an algorithm to which such data are input to provide a value for a TAC or effort control measure”. This may seem conceptually similar to a harvest control rule (HCR); however, the core difference is that a MP is fully specified and has been demonstrated, through simulation trials, to achieve reasonable management performance across a wide range of plausible assumptions about the resource (Butterworth, 2007). The MSE framework, which is synonymous with the MP approach, is a platform for developing and testing MPs (De Oliveira *et al.*, 2008).

The MSE approach was first developed during the late 1980s by the Scientific Committee of the International Whaling Commission (IWC) in response to the failure of the New Management Procedure (NMP), which had been used to provide recommendations on catch limits for over a decade. Butterworth (2007) gives two reasons as to why the NMP is generally perceived to have failed to facilitate scientific agreement. First, although a HCR had been specified, arguments still arose about the best estimates for its parameters (such as Maximum Sustainable Yield (MSY)). Second, once the parameter values had been decided, scientists would then debate how best to take account of uncertainty about those values. The

IWC Revised Management Procedure, which was accepted by the IWC in 1991, was designed to resolve these problems and is now taken to define an MP (IWC, 1992).

Many of these difficulties are not unique to the NMP, but rather are symptomatic of problems with the conventional approach to fisheries management. Typically, that involves using annual mathematical assessments of the resource based on all available data (e.g. survey biomass indices) to provide estimates of resource abundance and productivity, and then using those results to determine management actions. This annual choice of a single best assessment can prove problematic, because it introduces undesirable variability in TAC recommendations as the assessment methodology is (arguably) improved from year to year. It can also lead to much wasted time as scientists debate small modifications to data choices or methods used. Additionally, there is no formal basis for addressing uncertainty in model structure. Finally, the conventional approach of using short-term catch projections based on the current best assessment to provide management advice makes it difficult to maintain a long-term view of utilisation of the resource, because the risk of (unintended) stock depletion can be appreciably overestimated if feedback control is not taken into account (Butterworth, 2007).

Before discussing how the use of MPs resolves these issues, it is useful to outline the steps required to implement the MSE approach. First, mathematical models of the underlying dynamics for the stock must be constructed and conditioned on available data. These OMs are used during projections to simulate the response of the resource to different CMPs. Instead of selecting just one best assessment, a range of OMs—termed the Reference Set—is required to reflect the range of conflicting hypothesis about the stock. In doing so, uncertainties about model structure are implicitly taken into account. Rademeyer *et al.* (2007) recommend that only such alternative scenarios that are both highly plausible and have a large impact on simulation results should be included in the Reference Set. Other scenarios that are less plausible, or that seem to have little impact, should be identified as robustness tests, which will be described later.

The next stage in the process is to develop a wide variety of CMPs. These can either be model-based or empirical. The former depends on analyses of fishery data to obtain estimates of resource abundance and productivity, as would be done in a typical resource assessment. These results are then used in an HCR to provide management recommendations. In contrast, empirical MPs are not linked to a model of the resource, but provide recommendations directly from monitoring data. For example, if abundance indices show an increasing trend then the TAC may be increased, and *vice versa*. Model-based MPs tend to perform better, yet the empirical approach has the benefit of simplicity (Rademeyer *et al.*, 2007). Nevertheless, both types of MP will require that future monitoring data are generated during simulations. It is important that this is done with variances that reflect realistic levels of observation error, which are typically estimated from the fits of the OMs to historical data.

Finally, each CMP is evaluated through performance statistics that are calculated by projecting the resource forward for a period of (typically) 10–20 years. These statistics are designed to measure the success of an MP in achieving management objectives that are defined by stake-holders. The results are integrated over all OMs in the Reference Set. This process allows stake-holders to choose which CMP to implement, based on clear trade-offs in performance for what are often conflicting objectives. Robustness tests do not factor into these core reflections of performance, but rather serve as "tick tests" to ensure that the selected MP does not cause unacceptable drops in performance for any of those scenarios (Rademeyer *et al.*, 2007).

Butterworth (2007) lists several benefits that the MSE framework offers over the conventional approach to fisheries management. First, a MP must be pre-specified, meaning that the catch-control rule, the data to be used as input and the estimation method (if the MP is model-based) are decided beforehand. This pre-specification can save scientists considerable time, as it avoids the annual haggling over small modifications to the algorithm between reviews, which are usually planned only at wide time intervals of a few years. Second, the risk of unintended resource reduction is not overestimated, because an MP's response to future monitoring data (i.e. the feedback mechanism) is taken into account through long-term projections. Third, by including multiple alternate models of the resource in the Reference Set, as well in robustness tests, the MSE framework is consistent with the Food and Agriculture Organization's (of the United Nations) precautionary approach to fisheries management (Punt, 2006). Finally, the involvement of stake-holders in the process (particularly when defining management objectives) forces a long-term view of resource utilisation and promotes their support of the MP selected.

Nevertheless, there are some drawbacks to the MSE approach. Chief among them is the lengthy development time required compared to conventional assessments. Although initiating the process takes substantial resources, Butterworth and Punt (1999) argue that once an MP is in place, considerable savings can be achieved when producing subsequent TAC recommendations. Another difficulty is that an MP can be overly rigid and not allow for immediate action when a unforeseen problem with the resource becomes apparent. However, many implementations resolve this issue by making provisions for Exceptional Circumstances (ECs), which allow for unscheduled reviews of the MP. It is important that the conditions for these ECs are carefully defined in order to prevent their misuse. Finally, the MSE approach does not fully escape the problem of choosing a best assessment, as OMs comprising the Reference Set must still be selected. The difference is that an MP must necessarily have demonstrated its robustness to model uncertainty through its feedback mechanisms, which can somewhat compensate for a poor choice of a Reference Set.

Since their development by the IWC, MPs have been increasingly adopted worldwide. South Africa has been at the forefront of this movement, and provides an example of the formal incorporation of MPs in national fisheries legislation; the country's Marine Living Resources Act stipulates that the responsible minister should be advised on "the establishment and amendment of operational management procedures, including management plans" (Anon., 1998). They are now used to provide management advice for most major South African fisheries and are updated every three to five years (De Oliveira *et al.*, 2008). No other country has embraced the MSE framework as completely as South Africa; however, there are some other examples of its use. New Zealand began implementing an MP in the mid 1990s to manage a rock lobster fishery on the South Island. The approach has also been used in Australia, Europe and the USA to evaluate assessment techniques and HCRs, but has seldom lead to the formal adoption of what could accurately be termed an MP (Holland, 2010).

2.2 Horse mackerel biology

The Cape horse mackerel (*Trachurus trachurus capensis*) is a species in the family Carangidae. It is also known locally as maasbanker.

2.2.1 Stock-structure

Two stocks of Cape horse mackerel are recognised in southern African waters. A northern stock occurs off the coast of Namibia, and the southern stock extends from approximately the Orange River mouth off the West Coast of South Africa to as far as East London off its East Coast (Naish, 1990). The Namibian and South African stocks are believed to be separated by the intense upwelling cell offshore of Lüderitz in Namibia. This natural oceanographic barrier also separates the two countries' stocks of sardine and anchovy (Badenhorst, 2002).

Research on Cape horse mackerel in the late 1970s and early 1980s suggested that populations off the West Coast and South Coast of South Africa belonged to two distinct stocks. Hecht (1976) cited differences in the growth rates between coasts, while Crawford (1981) analysed the size distribution of catches. However, subsequent morphometric, biological and mitochondrial DNA studies almost a decade later indicated that the South Coast and West Coast stocks are in fact one and the same (Hecht, 1990; Naish, 1990).

2.2.2 Spawning and life cycle

The age-structure of the South African horse mackerel stock changes over its geographical range (Naish, 1990). Figure 2.1 illustrates this distribution. The adults are concentrated mostly along the South-East Coast and migrate westwards to the central and western Agulhas Bank during winter and spring to spawn. After spawning, they make the return migration eastwards (Naish *et al.*, 1991). Crawford (1981) identified the western end of the Agulhas Bank as the nursery area for larvae and juveniles. From there, they are transported northwards in the West Coast shelf-edge jet current in a process that has been similarly demonstrated for the reproductive products of sardine and anchovy (Badenhorst, 2002). Ultimately, they recruit to the pelagic fishery on the West Coast, and result in relatively large purse-seine landings of juvenile horse mackerel from January to March each year. As the fish mature, they migrate south and then east along the coast, eventually recruiting to the demersal and midwater fisheries on the South-East coast (Barange *et al.*, 1998).

Horse mackerel are serial spawners and appear to have two major spawning peaks, although the timing differs between the eastern and western Agulhas Bank. Hecht (1990) reported June and November as the main spawning periods on the eastern Bank, while Naish (1990) concluded that peak spawning on the western Bank occurs in August and February. Hecht (1990) found that fifty-percent maturity among Cape horse mackerel is reached at an age of approximately two years, and that all fish can be expected to be mature at age of three years. That paper also reports the sex ratio of a sample of the population to be 1:1, which agrees with a study by Uozumi *et al.* (1984). This result provides support for a sex-aggregated model.

2.2.3 Growth

The Cape horse mackerel has a maximum reported (fork) length of 60 cm and may live to more than ten years of age (Bianchi *et al.*, 1999). The length-at-age relationship used in the work presented in this thesis is taken from Kerstan (*pers. commn*) as quoted in Horsten (1999d).

This relationship takes the form of a von Bertalanffy growth curve:

$$l_a = l_\infty \left(1 - e^{-\kappa(a-t_0)}\right) \tag{2.1}$$

where

l_a is the expected total length of a fish of age a years in centimetres;

l_∞ is the asymptotic total length in centimetres;

κ , called the Brody growth coefficient, is a growth rate parameter; and

t_0 is the theoretical age at which length would be zero.

The mass-at-length relationship used for Cape horse mackerel is from Naish *et al.* (1991). It is provided by the power model:

$$w = \alpha \left(l\right)^{\beta} \tag{2.2}$$

where

w is the expected weight in grams of the fish;

l is the total length of the fish in centimetres; and

α and β are growth parameters.

Estimates for the parameters of these growth equations are reported in Table 2.1, while the growth curves themselves are illustrated in Figure 2.2. Hecht (1990) found no difference between the mean length-at-age of males and females. This provides further support for a sex-aggregated model.

Other length-at-age relationships have been reported; Kerstan and Leslie (1994) discuss differences among the growth coefficients given in the various studies available at that time. Those authors concluded that differences were the result of inadequate sampling efforts, long intervals between survey periods and differing ageing techniques.

2.2.4 Feeding, predation and vertical migrations

Cape horse mackerel are opportunistic feeders, generally eating whatever is available. They are zooplanktivorous and mainly feed on near-bottom aggregations of copepods and euphausids.

Larger horse mackerel also eat polychaete worms, chaetognaths, squid and other crustaceans (Badenhorst, 2002). Pillar and Barange (1998) reported that fish were infrequent in the diet, comprising approximately only 10%. However, it should be noted that the feeding behaviour of horse mackerel varies somewhat by locality and time. The same study by Pillar and Barange (1998) estimated a daily ration of 3.8% of body mass and a rapid gut evacuation rate of 0.22 hr^{-1} for the species. This appears to be in keeping with the general pattern noted for other relatively active fish with similar prey preferences. The high energetic requirements of continuous activity must be matched by high food consumption, which is thought to correspond to a high rate of gastric evacuation (Ruggerone, 1989).

Horse mackerel perform daily vertical migrations. At sunset they ascend as a population off the bottom where they become available to the midwater fishery, returning to the sea bed around dawn. Pillar and Barange (1998) argue that these migrations are not motivated by feeding; an analysis of stomach fullness and prey freshness showed that feeding occurs mainly during the day and drops to low levels after sunset. Instead, it has been suggested that migrating into warmer surface water at night may lead to increased digestion rates, allowing for more consumption during the next feeding period and hence higher growth rates (Pillar and Barange, 1998).

Horse mackerel are an important food source for many fish in South African waters. In particular, they account for up to 60% of the daytime diet of large hake on the South Coast (Badenhorst, 2002). Based on observations of hake from the West Coast, Pillar and Barange (1995) concluded that larger hake do not migrate extensively off the sea bed at night. This offers an additional explanation for the nocturnal vertical migrations performed by Cape horse mackerel. By moving closer to the surface, they may decrease the risk from demersal predators such as hake. However, the fish would then be more vulnerable to their pelagic predators such as snoek, cetaceans and the Cape fur seal. It is unlikely that any one factor can completely explain the vertical migrations of horse mackerel.

2.3 History of the horse mackerel fishery

The demersal trawl fishery targeting hake and sole began in the early 1900s. Horse mackerel would almost certainly have been taken as bycatch from the start; however, records of horse mackerel catches in that fishery are available from the 1950s only. Directed fishing of horse mackerel started only four decades later (in the early 1940s), when purse-seine trawlers caught them along with sardine on the West Coast. This pelagic fishery developed in order to satisfy the wartime demand for canned food (Badenhorst, 2002). However, the large surface schools of adult horse mackerel that this fishery originally targeted have long since disappeared. Pelagic catches peaked at 118 000 tonnes in 1954, but had declined to approximately 80 000 tonnes by the late 1950s; by the mid 1960s, these catches had dropped further to an average of 40 000 tonnes per annum (Figure 2.3). In 1968, the purse-seine fishery switched to targeting anchovy following the collapse of the South African sardine stock. Consequently, smaller-meshed nets were introduced and thereafter it seems reasonable to assume that the majority of horse mackerel caught in that fishery were juveniles (Johnston *et al.*, 2004).

In the mid 1960s, Japanese vessels started targeting horse mackerel using midwater trawl gear. The catches were originally in the region of 10 000 tonnes per annum; however, as more foreign trawlers entered the fishery, these catches increased, finally peaking at 93 000 tonnes in 1977 (Figure 2.3). This large harvest had an appreciable effect on the stock and caused a sudden drop in the catch rate for the fishery (Badenhorst, 2002). In that same year, South Africa declared an exclusive economic zone of 200 nautical miles and foreign fleets were consequently excluded except for limited access under license. The Japanese midwater trawl fleet received an allocation of between 8 000 and 25 000 tonnes per annum between 1977 and 1990.

In 1979, South African trawlers were first permitted to exploit horse mackerel in the midwater. Demersal vessels were allowed to carry both bottom and midwater gear, and would switch to the latter when large schools of horse mackerel were detected. Unfortunately, this arrangement has led to some difficultly in distinguishing between the two types of trawls in historical catch records, as it is believed that operators may have intentionally misreported the gear used in order to establish historical performance for the midwater fishery prior its formal introduction (Johnston *et al.*, 2004). Fishing was originally of an experimental nature, and only in 1983 were the first large landings made.

By 1990, a viable South African midwater horse mackerel fishery had been established. Its ability to make substantial catches which might exceed MSY meant that the horse mackerel resource now needed to be scientifically managed. This would preclude undue fishing pressure on the spawners on the South Coast or on the juveniles on the West Coast. In response, Punt (1989) developed a surplus production model based on the CPUE series from Japanese trawlers that was used to recommend a TAC of 35 000 tonnes for the fishery in 1990 and 1991. However, when the Japanese fleets were stopped from fishing in 1992 in order to make way for the local midwater trawl sector, the loss of the associated CPUE series meant that the surplus production model could no longer be applied (Durholtz, 2013). Additionally, acoustic survey results from 1991 suggested that the model may have severely underestimated horse mackerel abundance. Consequently, no assessment was made in 1992 and a combined Precautionary Maximum Catch Limit (PMCL) for the midwater and demersal fisheries of 40 000 tonnes was set.

Butterworth and Raubenheimer (1992) developed a Beverton-Holt yield-per-recruit type modelling approach that is appropriate for situations in which limited data are available. This determined the proportion of pre-exploitation biomass that could be harvested in order to achieve MSY. This approach was applied as a basis to set PMCLs of approximately 57 500 tonnes from 1993 to 1997. These catch limits were not reached. Additional management advice was that purse-seine catches of juveniles on the West Coast should be minimised so as not to reduce potential future catches in the midwater fishery on the South Coast.

In the mid 1990s there was an increasing trend in pelagic catches of horse mackerel, peaking at 26 000 tonnes in 1998 (Figure 2.3). It is believed that this was partly due to increased targeting of horse mackerel by purse-seine trawlers when the anchovy TAC was reduced (Johnston *et al.*, 2004). Although these catches may seem low compared to those of the 1950s, it should be noted that recent pelagic catches comprise mostly juveniles. Therefore, the number of fish per ton caught by purse-seine during the 1990s is much greater than was the case during the 1950s (Durholtz, 2013). In response to this worrying trend, the yield-per-recruit approach was refined by Butterworth and Clark (1996) which lead to a more conservative result, and consequently the PMCL for the period 1998-2000 was decreased to 34 000 tonnes.

During 1999, the first ASPM for the horse mackerel resource was developed, based on total annual landings of both trawl and pelagic fisheries (Horsten, 1999a). The results of corresponding biomass projections suggested that there was a pronounced yield-per-recruit effect, with even small pelagic catches having an appreciable negative effect on the level of trawl catch that could be maintained (Horsten, 1999b). Consequently, in 2000 the PMCL of 34 000 tonnes was maintained for the midwater and trawl fisheries, and a PUCL of 5 000 tonnes was introduced for the pelagic fishery. The annual purse-seine catch of juvenile horse mackerel has averaged 3 676 tonnes since then.

In order to accommodate new entrants into the midwater fishery, the ASPM was updated in 2001 and used to project the resource biomass forward under various management options (Johnston and Butterworth, 2001). Results showed that, given reasonable assumptions of stock-recruitment steepness and catchability, the PMCL could be experimentally increased to 44 000 tonnes for the next four years without any appreciable increase in risk to the resource. This limit comprised an allocation of 31 500 tonnes to the directed midwater trawl fishery, as well as a 12 500 tonne reserve to account for incidental bycatch in the demersal trawl fishery. An updated assessment in 2004 showed no negative response to the increased catches, and the PMCL therefore remained at 44 000 tonnes (Johnston and Butterworth, 2004). The results of the assessment by Johnston and Butterworth (2007) were very similar to those obtained previously, and the PMCL was thus unchanged.

A comprehensive update of the horse mackerel assessment that incorporated additional data, including a midwater CPUE series as well as commercial and survey length-frequency data, was completed in 2011 (Furman and Butterworth, 2011). The results were promising, indicating a 20% increase in abundance over the last five years as a result of good recruitment. However, long-term projections were similar to those from the 2007 assessment and therefore the PMCL was maintained at 44 000 tonnes for the 2012 fishing season.

In 2011, juvenile horse mackerel became impossible for the purse-seine trawlers to avoid following good recruitment to the resource. Consequently, a once-off *ad-hoc* increase of 7 000 tonnes was made to the PUCL in order to prevent the early closure of the pelagic fishery. However, this situation brought into question the appropriateness of a fixed catch limit for the fishery, and a joint Pelagic-Demersal Scientific Task Group was formed to investigate measures to introduce flexibility in the allocation. The Task Group was additionally asked to develop a MP to experimentally increase the allocation for the midwater fishery, as it was believed that the resource was underutilised. The chapters following provide details of the analyses developed for that Task Group and the Demersal Scientific Working Group (DSWG).

Table 2.1: Parameter values for the von Bertalanffy growth curve (Equation 2.1) and mass-at-length function (Equation 2.2) for Cape horse mackerel. Values reported are taken from Kerstan (*pers. commn*) as quoted in Horsten (1999d) and from Naish *et al.* (1991) respectively.

Parameter	Value
l_∞ (cm)	54.56
κ (yr^{-1})	0.183
t_0 (yr)	-0.654
α ($\mathrm{g/cm}^\beta$)	0.0078
β	3.011

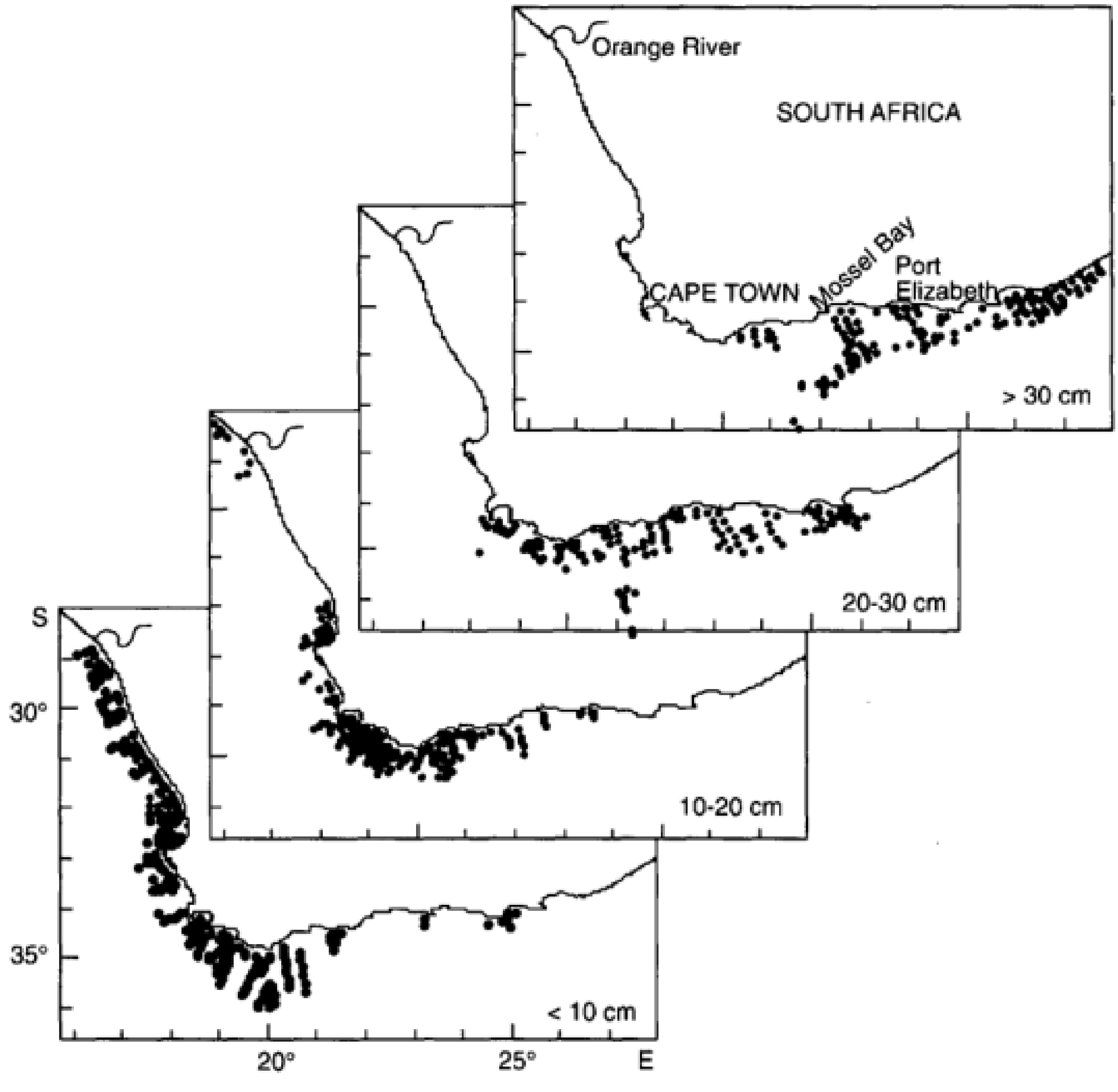

Figure 2.1: Spatial distribution of Cape horse mackerel by size as calculated from acoustic/midwater trawl surveys conducted in South African waters for the period 1984–1996 (from Barange *et al.*, 1998, with permission).

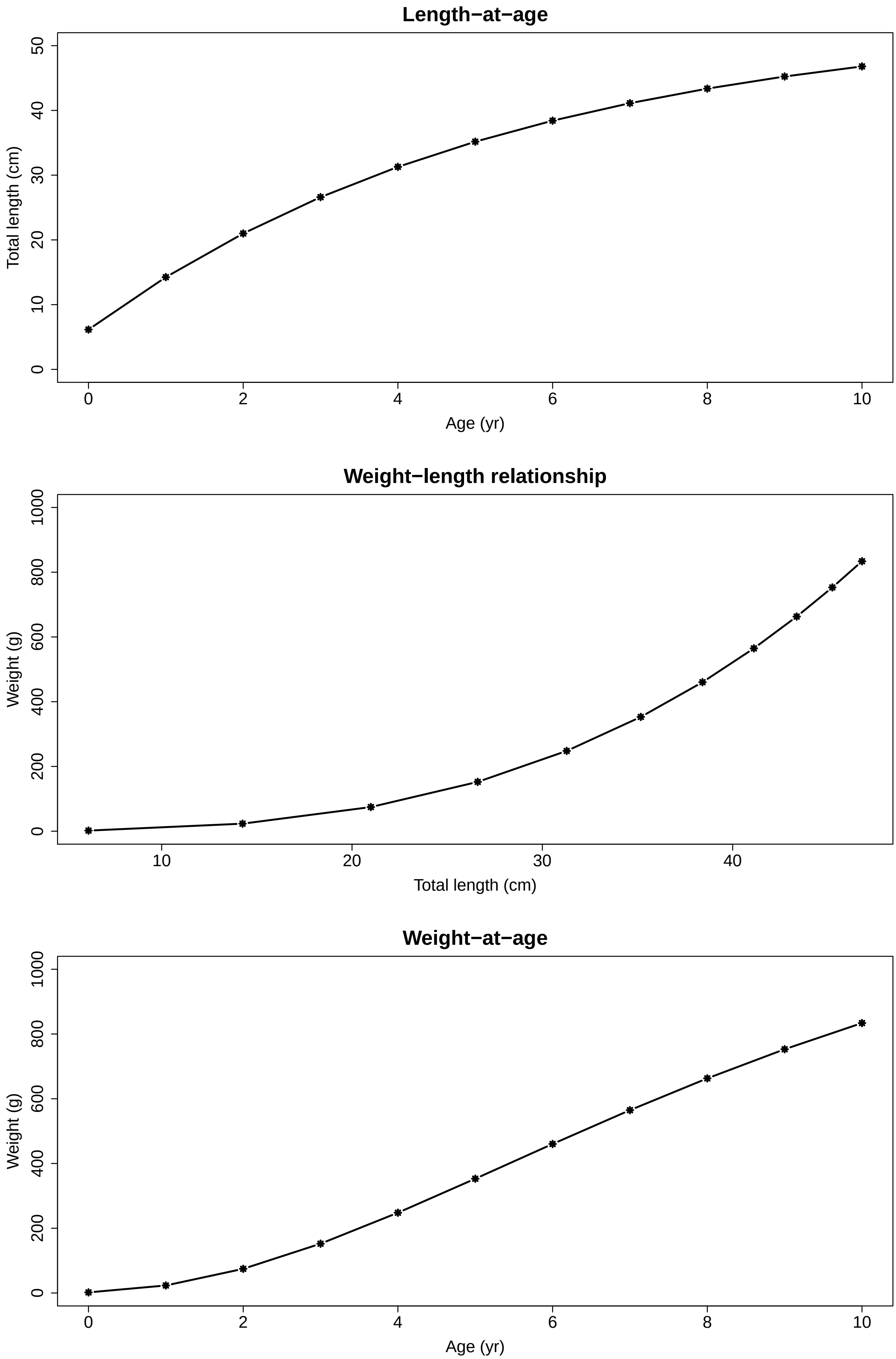

Figure 2.2: Length-at-age, weight-length and weight-at-age relationships for Cape horse mackerel as reported by Horsten (1999d) and Naish *et al.* (1991).

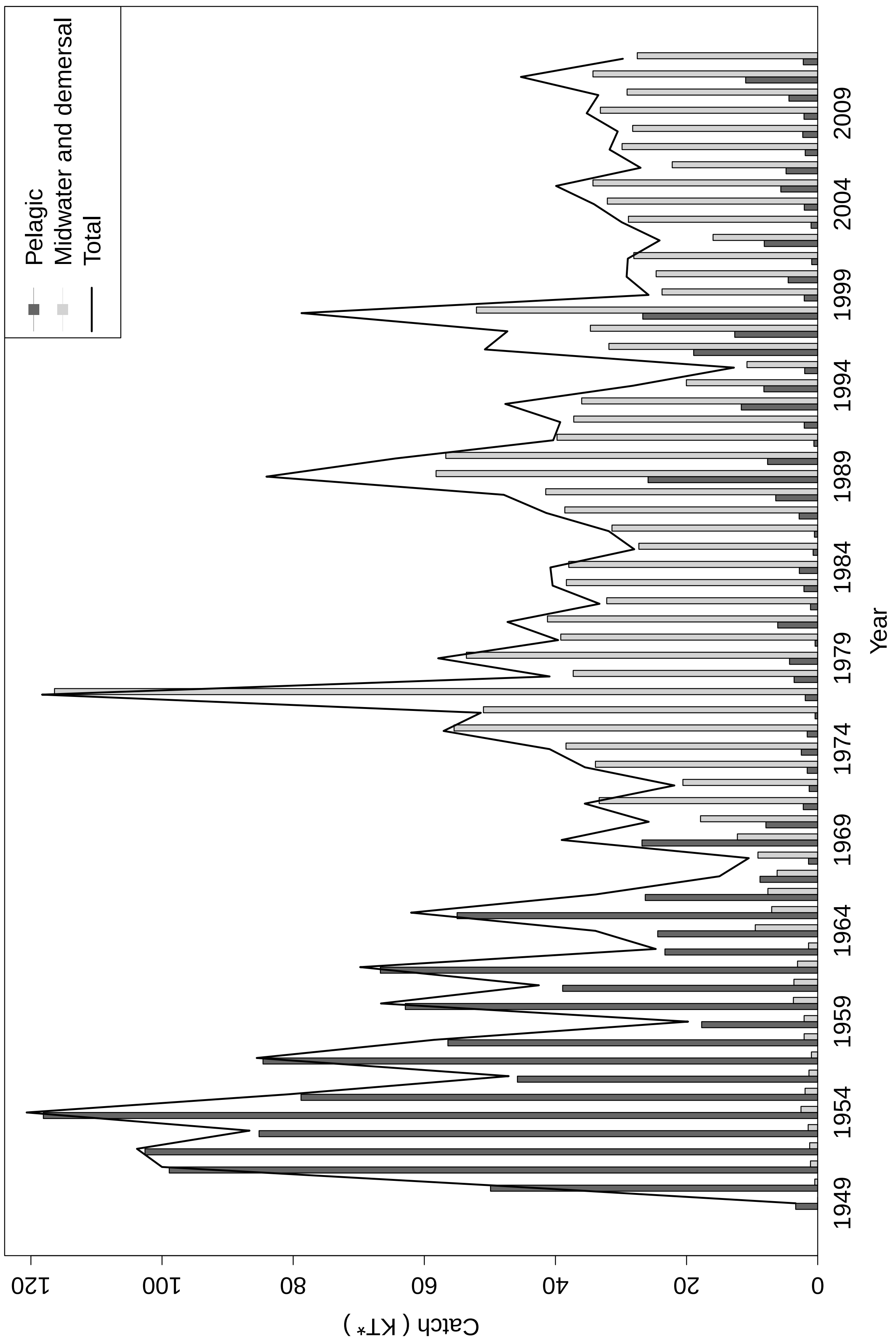

Figure 2.3: Catch history for the South African horse mackerel resource. Note that figures for the midwater and demersal trawl fleets are combined, because records do not distinguish between catches made by those fleets prior to 2000.

*Note that catches are given in kilotonnes (KT).

Chapter 3

Data

The fisheries branch of DAFF has provided the data necessary to assess the South African horse mackerel resource. Four main types are used: annual catches by mass, biomass estimates and catch-at-length distributions from biannual demersal swept-area surveys, catch rate estimates and catch-at-length distributions from the commercial midwater fleet, and juvenile biomass estimates from annual November pelagic hydro-acoustic surveys. These data sources are each discussed in their own sections below, while the specific data used in the assessment are given in tables at the end of the chapter. Figure 3.1 shows the periods encompassed by the various data sources. Appendix 3.A describes the GLM that is used to standardise the commercial midwater CPUE time-series.

In the assessment model, different selectivity-at-length functions are applied for the pelagic, midwater and demersal fleets. Consequently, with catch data available for each fleet, abundance and length-frequency data collected by a given fleet are assumed to reflect only the component of the horse mackerel population that is available to that fleet.

3.1 Historical catches

Annual catch records for horse mackerel are available from the beginning of the commercial fishery in 1949. Although they were undoubtedly also taken as a bycatch in the demersal trawl fishery that targeted hake and sole from the early 1900s, those catches were not reported.

The assessment model presented in this work distinguishes between catches made by the pelagic, midwater and demersal fleets; however, DAFF are unable to provide separate figures for catches in the midwater and demersal fisheries prior to 2000 (Singh, 2011). Therefore, the historical catches allocated to each fishery for these years are chosen to match the proportions of the combined midwater and demersal catches that were reported for each fishery according to the catch series used in the assessment by Johnston and Butterworth (2007), which were originally provided by Leslie (DAFF, *pers. commn*). Although those series are now known to be somewhat inaccurate, this catch split has little appreciable effect on the model's results. Variations on this assumption are included among the sensitivity tests conducted, which are listed in Section 4.4.

Singh (2011) found that there were two conflicting reports of combined demersal and midwater catches for the period 1988-1999. Table 3.1 compares the two datasets. As their differences are relatively minor (except for the last two years), the baseline assessment uses the means of the two datasets. Variations on this assumption are also included among the sensitivity tests performed, which are listed in Section 4.4.

Coetzee (DAFF, *pers. commn*) has provided catch data for the pelagic fishery. The total annual catches by mass for the pelagic fleet, together with the Johnston and Butterworth (2007) based split midwater and demersal catches that are used in the assessment model are listed in Table 3.2.

3.2 Demersal swept-area surveys

Demersal research surveys have been conducted by DAFF since 1983. They are used to calculate absolute biomass and length-frequency estimates. However, these surveys are not particularly suited to horse mackerel because the population has large pelagic and midwater components (Fairweather, 2009). Nevertheless, the survey estimates are included in the assessment because survey methods have remained consistent from year to year, and their results will therefore be generally less biased than catch rates from commercial data. On the other hand, research surveys are conducted over a relatively short period each year, whereas commercial data are collected from many vessels throughout the year. Consequently, abundance estimates based on survey data tend to show more variability.

The demersal surveys take place in Southern African waters up to four times a year: in summer and winter off the West Coast, and in spring and autumn off the South Coast. Data from the West Coast surveys are not used in this assessment, because the majority of horse mackerel on that coast are juveniles and therefore not demersal (Badenhorst, 2002). The standard

South Coast survey area stretches from the shore out to the 500 metre isobath, and from 20°E (Cape Agulhas) to 27°E (Port Alfred). Figure 3.2 shows the coverage of the surveys.

Stations are selected using a random stratified sampling design. This means that the survey area is divided into 100 metre depth strata and each depth stratum is further subdivided into substrata with widths of 1° of longitude . Sampling stations within each substratum are then selected at random, with the target number of stations per substratum proportional to the area of the substratum (Rademeyer *et al.*, 2010). An advantage of this design is that the variances for the abundance estimates can be calculated defensibly (Fairweather *et al.*, 2013).

Unfortunately there are limitations to these data. First, there were several years in which surveys were not conducted for various reasons. Second, many surveys had smaller ranges offshore, extending only from the coast out to the 200 metre isobath, instead of out to the 500 metre isobath. Finally, in 2000 the survey was conducted by the vessel *RV Dr Fridtjof Nansen*, instead of the usual *RV Africana*. Abundance estimates from these non-standard surveys are not used when fitting the assessment model. In September 2003 a new trawl net was introduced, but it is assumed here that this had little appreciable effect on survey results (Johnston and Butterworth, 2007). Consequently, no distinction is made in the model between data collected using the old and the new gear, whose use has roughly alternated since 2003.

Absolute survey biomass estimates are calculated by the swept-area method. They were provided by Fairweather (DAFF, *pers. commn*) and are reported in Table 3.3 along with their associated coefficients of variation (CVs). These data are shown graphically in Figure 3.3.

Survey catch-at-length estimates are derived by measuring a sample of the catch while still at sea. The recorded length of a fish is the total length (the distance between the tip of the snout and the tip of the tail fin), rounded down to the nearest centimetre (Cooper, 2013). The number of fish in each length class is then summed across all of the trawls in a survey. However, only the proportion of the total number of fish in each length class is used in the assessment model, because abundance information is already encapsulated in the aforementioned biomass estimates. Due to a combination of gear selectivity and natural mortality, relatively few horse mackerel are caught in the smallest and largest length classes. This can lead to large variability in the catch-at-length data for such lengths. To alleviate this effect, minus- and plus-groups are created by combining all length classes under a specified minimum length and over a specified maximum length respectively. For this assessment, the minus-group includes all fish less than 10 cm, while the plus-group includes all fish longer than 45 cm. The other length classes increase in graduations of 5 cm. Catch-at-length data from the spring and autumn demersal surveys were supplied by Fairweather (DAFF, *pers. commn*), and are listed in Table 3.4 and Table 3.5 respectively. These data are shown graphically in Figure 3.4.

3.2.1 Bias in demersal survey biomass estimates

It is strongly suspected that absolute biomass estimates from demersal swept-area surveys are negatively biased ("underestimated") to a large degree. Unfortunately, the assessment model does not have the power to estimate the extent of this bias. There are some good reasons to believe that this bias exists and might be substantial.

An assumption of the swept-area method is that all fish in the path of the net are caught. However, this assumption is unrealistic in the case of the horse mackerel research trawls owing to herding behaviour. Engas *et al.* (2001) reported such behaviour in hake during similar trawls, and Leslie (DAFF, *pers. commn*) believes that with horse mackerel being faster and more active than hake, the potential for herding is greater. Additionally, the long sweeps present on the demersal research gear could increase the impact of this behaviour (Leslie, DAFF, *pers. commn*). Nevertheless, it is reasonable to assume that the effects of herding will not vary much from year to year. Abundance estimates from the swept-area method should therefore be regarded as relative indices of abundance; however, the assessment model is not able to estimate this bias internally as there is insufficient content in the data available (Fairweather *et al.*, 2013; Rademeyer, 2003).

A second assumption of the swept-area method is that the proportion of horse mackerel in the water column that are catchable by the demersal trawl is consistent throughout the survey area; however, it is concerning that survey results do not reflect the pattern in commercial midwater trawling locations that is clear from the data (Figure 3.2). Conversations with trawler captains suggest that the regions offshore of Mossel Bay and Port Elizabeth are targeted as they are believed to have high horse mackerel densities, but demersal surveys do not indicate higher horse mackerel CPUEs in these heavily targeted regions or lower CPUEs off Tsitsikamma. The disparity between commercial and survey data may, in part, be due to the fact that the surveys are demersal, while commercial catches are taken from the midwater. This implies that the proportion of horse mackerel above the demersal trawl net may not be consistent. The absence of surveys in the heavily targeted region at about 200 metres offshore of Mossel Bay (as this area is not amenable to demersal trawls) suggests that the resultant negative bias may be large.

Finally, because of their distribution throughout the water column, the biomass of horse mackerel cannot be reliably estimated using demersal swept-area surveys in isolation. It is certain that a substantial amount of fish pass above the headline of the net and are consequently not sampled. It has been suggested by Durholtz (2013) that demersal surveys be conducted in tandem with the use of hydro-acoustic techniques to improve these abundance estimates.

3.3 Pelagic hydro-acoustic surveys

Although annual hydro-acoustic surveys of pelagic species were initiated by DAFF in 1984, sufficiently reliable estimates of horse mackerel biomass are available only from 1997. At that time, important changes were made to the survey method, including the adoption of species-specific target strength expressions (previously an estimate for Icelandic herring *Clupea harengus* was assumed to apply to all South African pelagic species) and the use of an echosounder that is not susceptible to receiver saturation (Coetzee *et al.*, 2008). Unfortunately, digital data from early horse mackerel surveys have been lost, so that it is impossible to revise older estimates so as to be consistent with post-1997 results (Coetzee, DAFF, *pers. commn*).

Recruit surveys are conducted in May and spawner biomass surveys in November. The typical May survey extends from the mouth of the Orange river on the West Coast to Cape Infanta on the South Coast, whereas the typical November survey extends from Hondeklip Bay on the West Coast to Port Alfred on the South Coast. In recent years the surveys have been extended farther eastwards with an increase in sampling effort on the Central and Eastern Agulhas Bank (de Moor *et al.*, 2008). Figure 3.5 shows the coverage of these surveys. The random stratified survey method of Jolly and Hampton (1990) has been used to estimate pelagic horse mackerel abundance. To determine species composition, trawls are performed concurrently with hydro-acoustic sampling.

One of the aims of this work is to assess the reliability of the pelagic survey biomass index as a predictor of horse mackerel recruitment for the following year. The May pelagic surveys are unsuitable for this purpose, because they provide biomass estimates for the year in which they are conducted. The November surveys—taking place closer to year-end—are therefore of primary interest. It is to be expected that data from the West Coast would prove the most useful, because the juvenile horse mackerel tend to congregate in that area.

Table 3.3 lists the biomass estimates as provided by Coetzee (DAFF, *pers. commn*), and Figure 3.3 plots the corresponding time-series.

3.4 Commercial midwater catches

Since 2003, an on-board observer programme has enabled DAFF to collect trawl-by-trawl data from commercial vessels with reasonable accuracy. These data are used in this assessment to estimate average annual catch rates, or CPUE, and length-frequency distributions for the midwater horse mackerel fishery. Only one midwater trawler, the *Desert Diamond*, is considered however, because it is the only vessel that is permitted to undertake purely horse mackerel directed fishing. If, for example, vessels from the bycatch fisheries were included in CPUE calculations, avoidance behaviour during periods of high horse mackerel abundance could introduce negative biases.

Two observers are deployed per vessel. Their job is to record the conditions of each trawl (including the time, location and weather) and to sample a portion of each catch to determine its size, length distribution and species composition (Cooper, 2013).

The data are standardised by means of a GLM in order to take into account co-variates that could bias the use of CPUE as an index proportional to abundance. The co-variates included in the GLM are *year*, *month* and *time*. This standardisation procedure is described in detail in Appendix 3.A.

The midwater CPUE index derived from the observer data according to the procedure in Appendix 3.A is reported in Table 3.3 and plotted in Figure 3.3, while the corresponding catch-at-length data are listed in Table 3.6 and shown graphically in Figure 3.4.

Table 3.1: Comparison between two equally plausible records of total annual catches of horse mackerel by the combined midwater and demersal fleets. Note that *Difference* refers to the result of subtracting the catches listed in *Dataset 1* from those listed in *Dataset 2* (Singh, 2011).

Year	Dataset 1 (KT)	Dataset 2 (KT)	Difference (KT)
1988	41.48	41.48	
1989	59.52	56.89	2.63
1990	56.72	56.72	
1991	37.86	41.66	−3.80
1992	34.52	39.89	−5.36
1993	36.00	36.00	
1994	20.03	20.03	
1995	10.79	10.79	
1996	32.00	31.70	0.30
1997	31.21	38.14	−6.93
1998	46.42	57.68	−11.26
1999	17.96	29.52	−11.56

Table 3.2: Fleet-disaggregated total annual landings of South African horse mackerel for the period 1949–2012. Note that records of horse mackerel catches by the midwater fleet are available from 1998 only (Singh, 2011; Coetzee, DAFF, *pers. commn*).

Year	Pelagic (KT)	Midwater (KT)	Demersal (KT)	Year	Pelagic (KT)	Midwater (KT)	Demersal (KT)
1949	3.36	0.00	0.00	1986	0.50	0.00	31.38
1950	49.90	0.00	0.45	1987	2.83	0.00	38.57
1951	98.90	0.00	1.11	1988	6.40	0.00	41.48
1952	102.60	0.00	1.23	1989	25.87	0.00	58.21
1953	85.20	0.00	1.46	1990	7.65	0.00	56.72
1954	118.10	0.00	2.55	1991	0.58	0.00	39.76
1955	78.80	0.00	1.93	1992	2.06	0.00	37.21
1956	45.80	0.00	1.33	1993	11.65	0.00	36.00
1957	84.60	0.00	0.96	1994	8.21	0.00	20.03
1958	56.40	0.00	2.07	1995	1.99	0.00	10.79
1959	17.70	0.00	2.08	1996	18.92	0.00	31.85
1960	62.90	0.00	3.71	1997	12.65	0.00	34.67
1961	38.90	0.00	3.63	1998	26.68	15.77	36.28
1962	66.70	0.00	3.08	1999	2.06	2.16	21.58
1963	23.30	0.00	1.40	2000	4.50	11.15	13.49
1964	24.40	0.00	9.52	2001	0.92	16.56	11.48
1965	55.00	0.00	7.02	2002	8.15	9.54	6.43
1966	26.30	0.00	7.60	2003	1.01	25.06	3.80
1967	8.80	0.00	6.19	2004	2.05	26.47	5.61
1968	1.40	0.00	9.12	2005	5.63	29.13	5.15
1969	26.80	0.00	12.25	2006	4.82	18.44	3.75
1970	7.90	0.00	17.87	2007	1.90	25.31	4.53
1971	2.20	0.00	33.33	2008	2.28	23.86	4.36
1972	1.30	0.00	20.56	2009	2.09	29.55	3.59
1973	1.60	0.00	33.90	2010	4.39	23.74	5.33
1974	2.50	0.00	38.39	2011	10.99	29.44	4.83
1975	1.60	0.00	55.46	2012	2.20	22.06	5.46
1976	0.40	0.00	50.98				
1977	1.90	0.00	116.40				
1978	3.60	0.00	37.29				
1979	4.30	0.00	53.58				
1980	0.40	0.00	39.19				
1981	6.10	0.00	41.22				
1982	1.10	0.00	32.18				
1983	2.10	0.00	38.33				
1984	2.80	0.00	37.97				
1985	0.70	0.00	27.28				
1986	0.50	0.00	31.38				

Table 3.3: GLM standardised CPUE and survey abundance data for South African horse mackerel for the period 1986–2012. Shaded values are excluded from the assessment because the corresponding surveys extended only from the coast out to the 200 metre isobath, instead of out to the 500 metre isobath. The underlined values are similarly excluded because the autumn survey in 2000 was conducted by the *RV Dr Fridtjof Nansen*, instead of the usual *RV Africana*. Data was provided by Coetzee, Fairweather and Singh (DAFF, *pers. commn*).

Year	CPUE	November pelagic survey		Autumn demersal survey		Spring demersal survey	
		West Coast biomass (KT)	Total biomass (KT)	Biomass (KT)	CV	Biomass (KT)	CV
1986						97.36	0.13
1987						332.97	0.14
1988				159.07	0.29		
1989				138.20	0.54		
1990				122.75	0.28	551.22	0.22
1991				352.19	0.23	575.01	0.17
1992				422.21	0.23	477.29	0.27
1993				435.28	0.20	307.17	0.16
1994				340.72	0.26	337.59	0.16
1995				195.13	0.24	276.37	0.23
1996				261.77	0.23		
1997		22.98	23.27	241.02	0.23		
1998		1.83	20.39				
1999		1.04	5.12	330.63	0.24		
2000		0.85	196.06	322.42	0.33		
2001		5.96	52.91			316.72	0.18
2002		4.26	15.29				
2003	0.62	10.32	21.44	146.72	0.24	231.36[*]	0.20[*]
2004	0.71	0.94	43.14	195.73[*]	0.32[*]	366.50[*]	0.19[*]
2005	0.90	8.14	12.45	175.04[*]	0.21[*]		
2006	1.00	11.96	49.80	386.57	0.20	350.28	0.19
2007	1.49	0.62	0.91	243.58[*]	0.40[*]	473.22[*]	0.19[*]
2008	1.02	1.51	11.51	279.86[*]	0.27[*]	300.00[*]	0.17[*]
2009	0.83	6.41	12.94	337.16[*]	0.24[*]		
2010	1.11	51.98	112.19	271.79	0.37		
2011	1.42	6.56	66.70	213.09[*]	0.22[*]		
2012	0.91	5.74	16.84				

[*] These values correspond to surveys that used the new trawl net, which was introduced in September 2003.

Table 3.4: Spring demersal survey catch-at-length for South African horse mackerel (shown as proportions of numbers) as used in the assessment model. Provided by Fairweather (DAFF, *pers. commn*).

Year	Total length (cm)								
	0–10	10–15	15–20	20–25	25–30	30–35	35–40	40–45	45+
1986	0.000	0.000	0.002	0.090	0.238	0.164	0.169	0.231	0.105
1987	0.000	0.000	0.116	0.223	0.160	0.206	0.124	0.129	0.043
2001	0.002	0.015	0.375	0.255	0.124	0.136	0.075	0.015	0.004
2003	0.000	0.050	0.068	0.376	0.367	0.091	0.040	0.008	0.001
2004	0.001	0.238	0.256	0.161	0.226	0.074	0.035	0.008	0.001
2006	0.008	0.267	0.243	0.288	0.144	0.041	0.008	0.001	0.000
2007	0.000	0.223	0.634	0.095	0.044	0.003	0.001	0.000	0.000
2008	0.001	0.027	0.458	0.429	0.068	0.010	0.005	0.002	0.000

Table 3.5: Autumn demersal survey catch-at-length for South African horse mackerel (shown as proportions of numbers) as used in the assessment model. Provided by Fairweather (DAFF, *pers. commn*).

Year	Total length (cm)								
	0–10	10–15	15–20	20–25	25–30	30–35	35–40	40–45	45+
1988	0.000	0.015	0.051	0.014	0.156	0.166	0.180	0.291	0.127
1992	0.000	0.072	0.046	0.105	0.374	0.273	0.056	0.043	0.030
1993	0.000	0.092	0.353	0.075	0.198	0.118	0.076	0.065	0.023
1994	0.000	0.027	0.157	0.220	0.298	0.254	0.029	0.010	0.004
1995	0.000	0.000	0.023	0.109	0.460	0.271	0.092	0.033	0.011
1996	0.000	0.000	0.001	0.023	0.542	0.308	0.111	0.013	0.002
1997	0.000	0.003	0.024	0.005	0.468	0.401	0.079	0.016	0.005
1999	0.000	0.010	0.169	0.063	0.082	0.522	0.114	0.033	0.006
2003	0.000	0.001	0.393	0.329	0.120	0.060	0.082	0.015	0.001
2004	0.022	0.142	0.432	0.055	0.186	0.100	0.053	0.008	0.001
2005	0.000	0.354	0.198	0.148	0.186	0.057	0.050	0.007	0.000
2006	0.001	0.033	0.239	0.345	0.282	0.063	0.030	0.006	0.000
2007	0.108	0.463	0.319	0.088	0.016	0.004	0.002	0.001	0.000
2008	0.001	0.071	0.382	0.384	0.150	0.009	0.001	0.002	0.000
2009	0.000	0.068	0.155	0.525	0.220	0.028	0.002	0.001	0.000
2010	0.000	0.056	0.068	0.527	0.294	0.044	0.003	0.006	0.001
2011	0.141	0.770	0.032	0.033	0.022	0.001	0.000	0.000	0.000

Table 3.6: Commercial midwater catch-at-length for South African horse mackerel from the *Desert Diamond* (shown as proportions of numbers) as used in the assessment model. Provided by Singh (DAFF, *pers. commn*).

Year	Total length (cm)								
	0–10	10–15	15–20	20–25	25–30	30–35	35–40	40–45	45+
2003	0.000	0.000	0.000	0.001	0.135	0.256	0.505	0.102	0.001
2004	0.000	0.000	0.000	0.012	0.241	0.382	0.328	0.036	0.001
2005	0.000	0.000	0.004	0.079	0.288	0.388	0.190	0.035	0.016
2006	0.000	0.000	0.006	0.113	0.339	0.403	0.126	0.010	0.003
2007	0.000	0.000	0.003	0.090	0.293	0.359	0.187	0.054	0.014
2008	0.000	0.001	0.043	0.256	0.328	0.246	0.111	0.014	0.001
2009	0.000	0.000	0.001	0.088	0.386	0.318	0.170	0.034	0.002
2010	0.000	0.000	0.018	0.220	0.378	0.255	0.100	0.026	0.003
2011	0.000	0.000	0.001	0.146	0.490	0.236	0.104	0.022	0.001

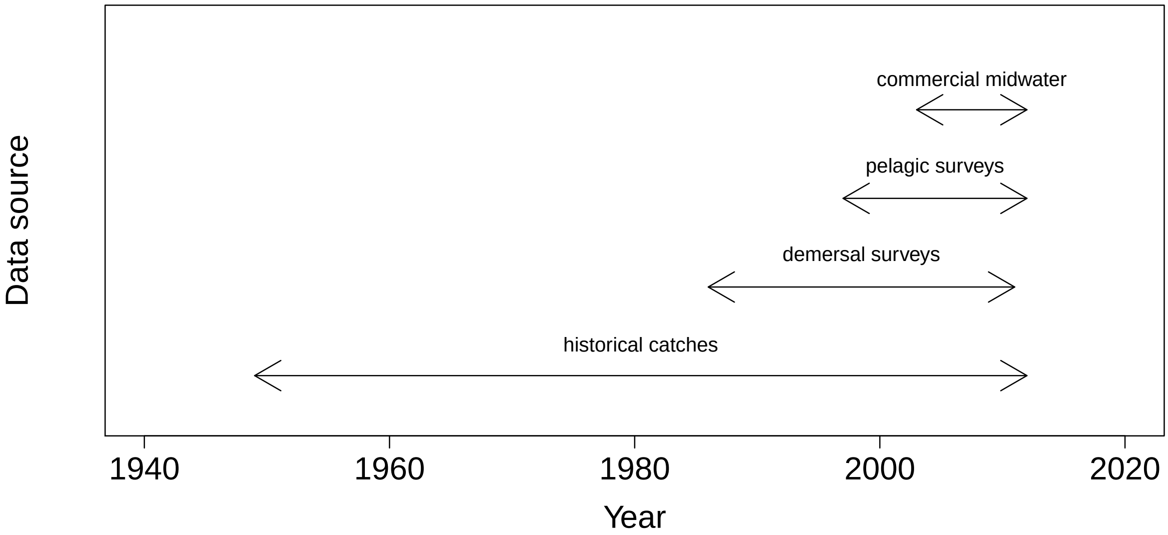

Figure 3.1: Comparison of the periods encompassed by the various data sources used in the assessment. Note that the arrows indicate when data collection started and concluded; however, intermediate years with missing data are not reflected.

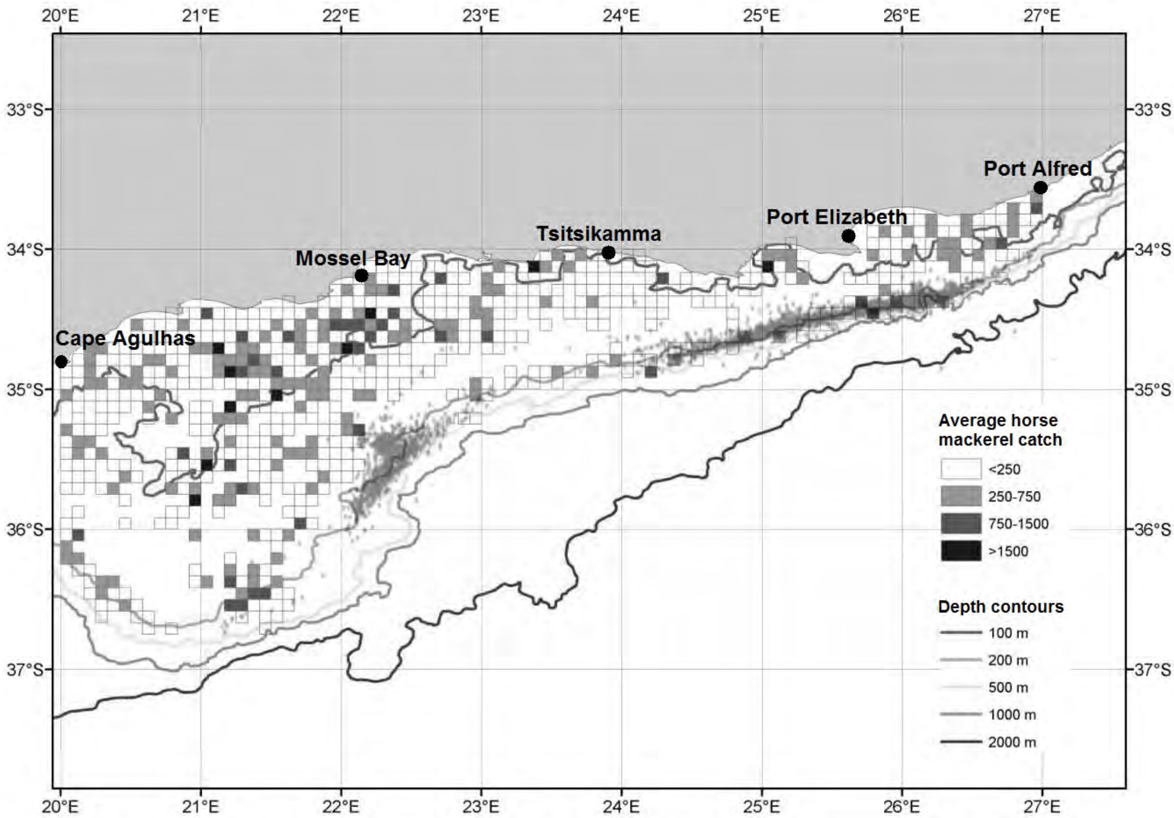

Figure 3.2: Comparison of the standard South Coast demersal survey area to commercial midwater trawling locations. The small shaded squares show survey coverage over the last decade, with darker squares reflecting higher average catches of horse mackerel. The small dots corresponds to the locations of commercial midwater trawls that targeted horse mackerel over the period 2003–2011. Adapted from a plot provided by Fairweather (DAFF, *pers. commn*).

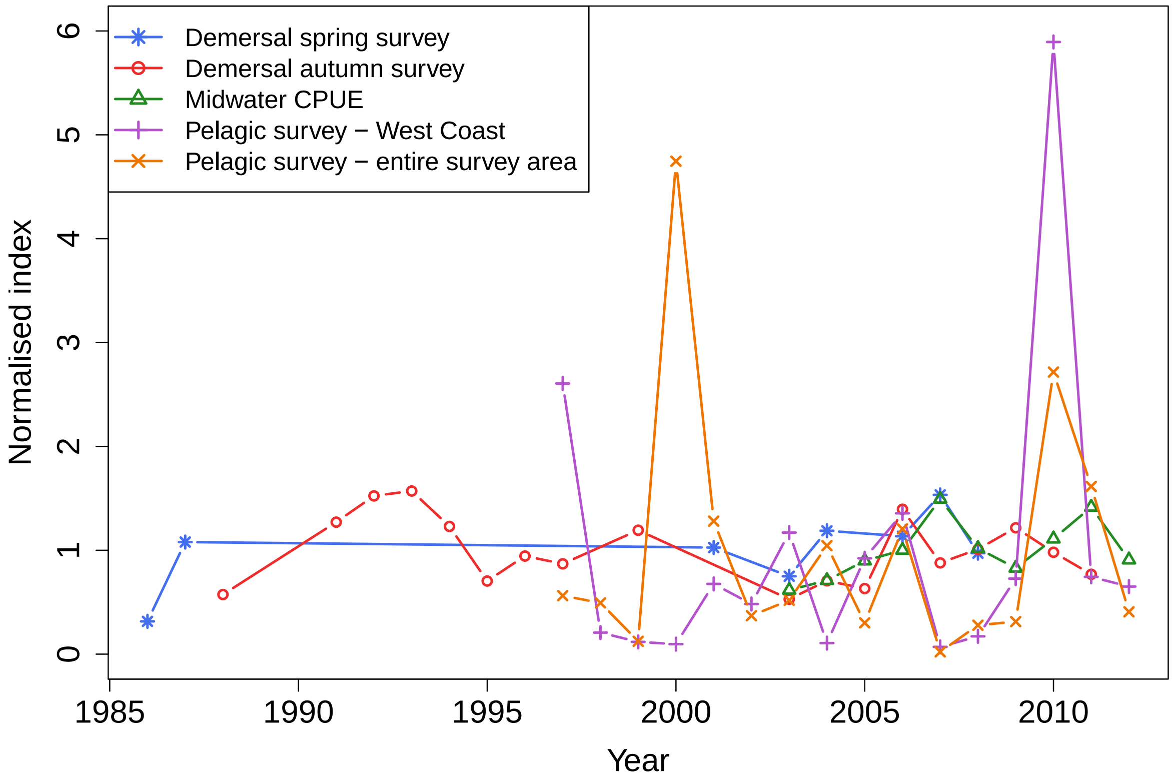

Figure 3.3: Normalised abundance indices for South African horse mackerel. Each index has been normalised by dividing it by its mean.

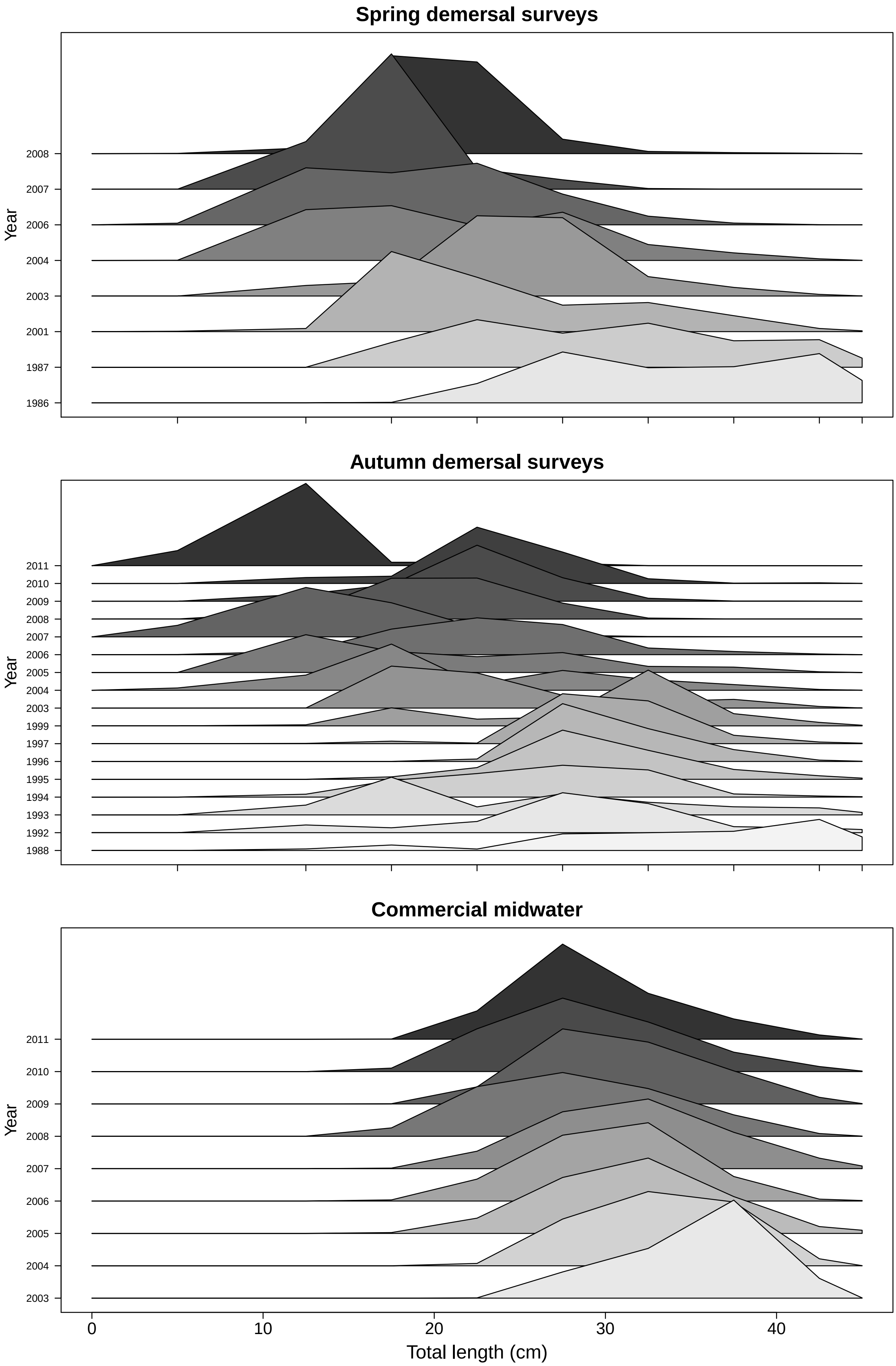

Figure 3.4: Catch-at-length data for South African horse mackerel as used to fit the assessment model.

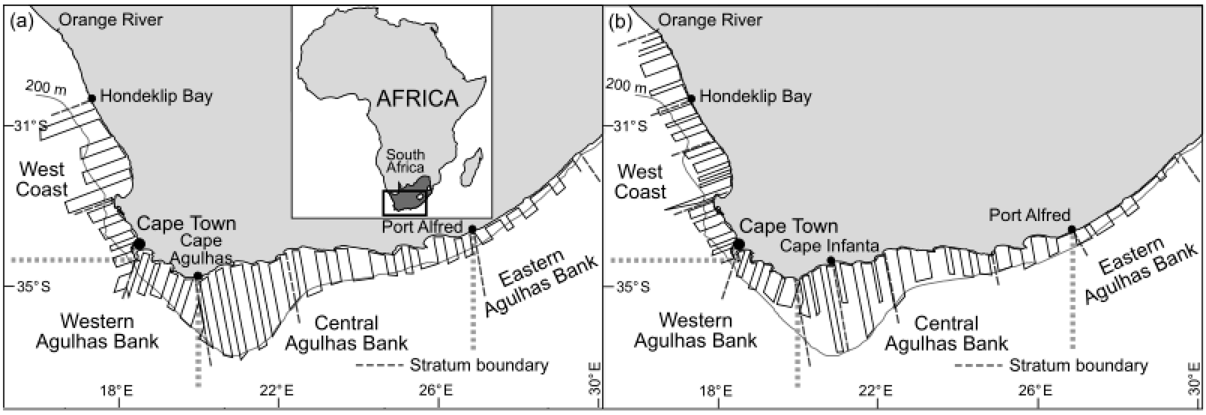

Figure 3.5: Track charts for (a) November and (b) May pelagic hydro-acoustic surveys (from de Moor *et al.*, 2008, with permission).

Appendix 3.A The GLM standardised midwater CPUE series

Detailed observer data for almost every midwater trawl off the South African coast since August 2003 were provided by van der Westhuizen and Cooper (DAFF, *pers. commn*). However, only data from the *Desert Diamond* were used to produce a CPUE series (Section 3.4), as it is the only vessel with directed horse mackerel fishing rights. This is a trawler of 7 765 gross registered tonnage that is owned by Oceana Group. It accounts for approximately 81% of horse mackerel caught (by mass) in the midwater. The remaining 19% of horse mackerel was taken by six other vessels.

The commercial CPUE needs to be standardised to in order to obtain a time-series for which the values are comparable across the years. Unlike in scientific surveys, a commercial trawler's behaviour does not necessarily remain consistent over time, and therefore biases can be introduced in CPUE as an index proportional to abundance. For example, if the *Desert Diamond* changed its routine and started targeting only an area with a naturally high density of horse mackerel, then the average CPUE would increase without there necessarily being a corresponding increase in abundance. A GLM takes several of these types of co-variates into account and applies a model to estimate their effects.

3.A.1 Method

The initial step was to clean the raw trawl data from the on-board observers. Entries with missing values or erroneous outliers were manually inspected and removed. Errors included unrealistically brief trawls (e.g. ≤ 5 minutes), fast trawl speeds (e.g. ≥ 40 knots) and large nets (e.g. with a vertical opening of $\geq 1\,000$ meters). In total, only 3 612 out of 4 334 recorded trawls are used to produce the CPUE time-series.

Co-variates initially considered for inclusion in the GLM were:

year, a categorical co-variate with eleven levels associated with the years: 2003–2013;

month, a categorical co-variate with six levels associated with the fishing month: grouped by pairs of months from January to December (i.e. Jan–Feb, Mar–Apr, etc.)

time, a categorical co-variate with eight levels associated with the time of day of the trawl: grouped by three hour periods from 00:00 to 24:00 (i.e. 00:00–03:00, 03:00–06:00, etc.)

***longitude*,** a categorical co-variate with two levels associated with the longitude: grouped according to whether the trawl was conducted to west or east of 23.4°E (Plettenburg Bay);

***wind speed*,** a categorical co-variate with ten levels associated with wind speed on the Beaufort scale: 0–9;

***wind direction*,** a categorical co-variate with 8 levels associated with wind direction: grouped according to the cardinal (i.e. N, E, S, W) and ordinal (i.e. NE, SE, SW, NW) directions;

***depth*,** a continuous co-variate associated with the mean depth of the trawl net during the trawl; and

***lunar phase*,** a continuous co-variate associated with percentage of the moon illuminated during the trawl.

To better understand the effects of the candidate co-variates on CPUE, the mean CPUE was calculated at each level of each co-variate. Figure 3.A.1 shows the results. They suggest a possible linear relationship of CPUE with *depth* and with *lunar phase*; therefore these co-variates are treated as continuous explanatory variables. The other effects cannot be explained by similarly simple relationships, so they are treated as categorical co-variates. Latitude is not included as a co-variate, because it is clear from Figure 3.2 that there are two very narrow midwater fishing grounds on either side of 23.4°E, and thus *longitude* alone is sufficient to describe the trawling location. Reference levels for categorical co-variates were chosen to correspond to the levels with the most data points.

Using the forward selection method with various forms of GLM suggests that only *year*, *month* and *time* explain sufficient variance in the CPUE data to warrant inclusion in the GLM.

Two approaches to modelling CPUE have been explored: the log-normal and the Poisson models.

1) Log-normal model

The CPUE of a trawl is assumed to be log-normally distributed, and is given by

$$\ln(C/E + \delta) = \mu + \alpha_{year} + \beta_{month} + \gamma_{time} + \epsilon \ .$$

2) Poisson model

The weight of fish caught during a trawl is assumed to follow a Poisson distribution, and is given by

$$C = E \exp\left(\mu + \alpha_{year} + \beta_{month} + \gamma_{time} + \epsilon\right) \ .$$

For both of these models:

C is the mass of horse mackerel caught;

E is the fishing effort expended and is the product of *trawl duration*, *trawl speed* and *net height*; it is treated as an offset in the Poisson model and the denominator of the response variable in the log-normal model;

δ is a constant equal to a fixed percentage of the mean nominal CPUE, and is added to allow for the occurrence of zero CPUE values when taking logarithms;

μ is the intercept;

$\alpha_f/\beta_f/\gamma_f$ is the contribution from co-variate f; and

ϵ is the error term, which is assumed to follow a normal distribution.

An important difference between the log-normal and the Poisson models is the inclusion of the δ term in the log-normal model to allow for zero CPUE values. It is calculated as:

$$\delta = \theta\overline{CPUE}$$

However, this constant is somewhat arbitrary and—as will be shown shortly—it can introduce irregularities in the GLM. To explore this issue, results are compared for a variety of θ values. An additional reason that the Poisson model is considered is that it can accommodate zero catch numbers without the δ term.

The GLMs were implemented in GenStat® 15$^{\text{th}}$ Edition, which is produced by VSN International.

3.A.2 Results

Results are shown for four different GLMs: the Poisson model, and the log-normal model with $\theta = 0.01$, $\theta = 0.05$ and $\theta = 0.1$. Table 3.A.1 lists estimates of the GLM's parameters and their associated standard errors.

A standardised CPUE time-series is obtained from each GLM by setting *month* and *time* to their reference levels and varying only *year*. Figure 3.A.2 compares these series to the mean marginal CPUE series, which is calculated by averaging the raw CPUE values from the data for each year. To improve clarity—and because only the trends in CPUE are of interest—each series is normalised so that its mean value is equal to one. Note that although CPUE estimates for 2013 are given, they are not used in the assessment because trawl data from that year were incomplete at the time of writing.

Figure 3.A.3 shows diagnostic plots for the GLMs. These include histograms and Q-Q plots of the standardised residuals to check for normality, and scatter plots of standardised residuals versus fitted values to check for homoscedasticity.

3.A.3 Discussion

Upward trends in both the marginal CPUE and GLM standardised CPUE (Figure 3.A.2) are encouraging and consistent with abundance estimates from demersal surveys, which indicate a recent increase in exploitable biomass. The choice of a log-normal or Poisson model for the GLM has little appreciable effect on series.

Histograms and Q-Q plots (Figure 3.A.3) indicate that the standardised residuals are reasonably normal across all of the GLMs. However, it is clear that for the log-normal models, as θ decreases, the distribution becomes increasingly skewed to the left. For the log-normal model, normality is an important test for model adequacy, as the residuals are expected to be normally distributed. Unfortunately, this is not always appropriate for the Poisson model, as the residuals are expected to be normal only in the asymptotic limit (Müller, 2011).

The issue of homoscedasticity is also important. This means that if the GLMs are to provide accurate estimates of CPUE, then the variance of the residuals should be independent of the model-estimated value (CPUE for the log-normal model and mass caught for the Poisson model) and remain roughly constant. Scatter plots of standardised residuals versus fitted values (Figure 3.A.3) provide a way to check this requirement. The downward trend seen for the Poisson model is discouraging, as it shows that this GLM tends to underestimate small CPUE values and overestimate large CPUE values. Plots for the log-normal models are much more promising, showing reasonable homoscedasticity. However, there are clear patterns in the lower halves of the scatter plots, which are a consequence of the δ term. By adding a small positive constant to each observed CPUE, an artificial lower limit on their logarithms is imposed and, consequently, the variability of the residuals increases with fitted value. As expected, this effect becomes more noticeable as θ increases.

In light of these facts, the log-normal model with $\theta = 0.05$ seems to be preferable as it provides a middle ground in the trade-off between normality and homoscedasticity. Furthermore, given the minimal difference between the CPUE series produced by the various GLMs, it is reasonable to use that GLM as the baseline choice; nevertheless, the Poisson model is included in the assessment as a sensitivity test (Section 4.4). Table 3.3 gives the values of CPUE series that are used when fitting the assessment model.

Table 3.A.1: GLM results for *Desert Diamond* catch rate data for South African horse mackerel over the period 2003–2013. *s.e.* and *t pr.* refer to the standard error and t-probability of the estimate respectively. Note that the reference levels (*year: 2007*, *month: Nov–Dec* and *time: 3–6*) have an estimate of zero in the GLM.

Factor	Level	$\theta = 0.01$			$\theta = 0.05$			$\theta = 0.10$			Poisson		
		Estimate	s.e.	t pr.	Estimate	s.e.	t pr.	Estimate	s.e.	t pr.	Estimate	s.e.	t pr.
Constant		−0.09	0.08	0.261	0.01	0.07	0.882	0.12	0.06	0.051	−0.07	0.01	<.001
Year	2003	−0.91	0.11	<.001	−0.78	0.09	<.001	−0.69	0.08	<.001	−0.78	0.02	<.001
Year	2004	−0.74	0.08	<.001	−0.66	0.07	<.001	−0.60	0.06	<.001	−0.62	0.01	<.001
Year	2005	−0.51	0.08	<.001	−0.46	0.07	<.001	−0.42	0.06	<.001	−0.43	0.01	<.001
Year	2006	−0.43	0.09	<.001	−0.36	0.07	<.001	−0.32	0.07	<.001	−0.37	0.01	<.001
Year	2008	−0.42	0.08	<.001	−0.35	0.07	<.001	−0.31	0.06	<.001	−0.28	0.01	<.001
Year	2009	−0.65	0.08	<.001	−0.53	0.07	<.001	−0.46	0.06	<.001	−0.48	0.01	<.001
Year	2010	−0.32	0.08	<.001	−0.27	0.07	<.001	−0.24	0.06	<.001	−0.28	0.01	<.001
Year	2011	−0.10	0.09	0.273	−0.05	0.08	0.523	−0.03	0.07	0.683	0.08	0.01	<.001
Year	2012	−0.57	0.09	<.001	−0.45	0.08	<.001	−0.38	0.07	<.001	−0.25	0.01	<.001
Year	2013	0.70	0.21	<.001	0.66	0.17	<.001	0.62	0.15	<.001	0.72	0.03	<.001
Month	Jan–Feb	−0.11	0.07	0.118	−0.07	0.06	0.195	−0.06	0.05	0.234	−0.01	0.01	0.414
Month	Mar–Apr	−0.15	0.07	0.027	−0.14	0.06	0.012	−0.13	0.05	0.007	−0.16	0.01	<.001
Month	May–Jun	−0.52	0.07	<.001	−0.45	0.06	<.001	−0.40	0.05	<.001	−0.41	0.01	<.001
Month	Jul–Aug	−0.65	0.07	<.001	−0.56	0.06	<.001	−0.50	0.05	<.001	−0.56	0.01	<.001
Month	Sep–Oct	−0.24	0.07	<.001	−0.22	0.05	<.001	−0.20	0.05	<.001	−0.21	0.01	<.001
Time	0–3	0.45	0.06	<.001	0.40	0.05	<.001	0.36	0.05	<.001	0.31	0.01	<.001
Time	6–9	−0.01	0.09	0.91	−0.01	0.08	0.854	−0.02	0.07	0.819	−0.16	0.01	<.001
Time	9–12	−0.69	0.10	<.001	−0.53	0.08	<.001	−0.43	0.07	<.001	−1.20	0.01	<.001
Time	12–15	−0.14	0.12	0.235	−0.04	0.10	0.713	0.01	0.09	0.893	−0.77	0.02	<.001
Time	15–18	0.42	0.14	0.003	0.36	0.12	0.002	0.33	0.10	0.002	−0.04	0.02	0.05
Time	18–21	1.22	0.06	<.001	1.13	0.05	<.001	1.04	0.04	<.001	1.20	0.01	<.001
Time	21–24	0.78	0.06	<.001	0.69	0.05	<.001	0.62	0.04	<.001	0.68	0.01	<.001

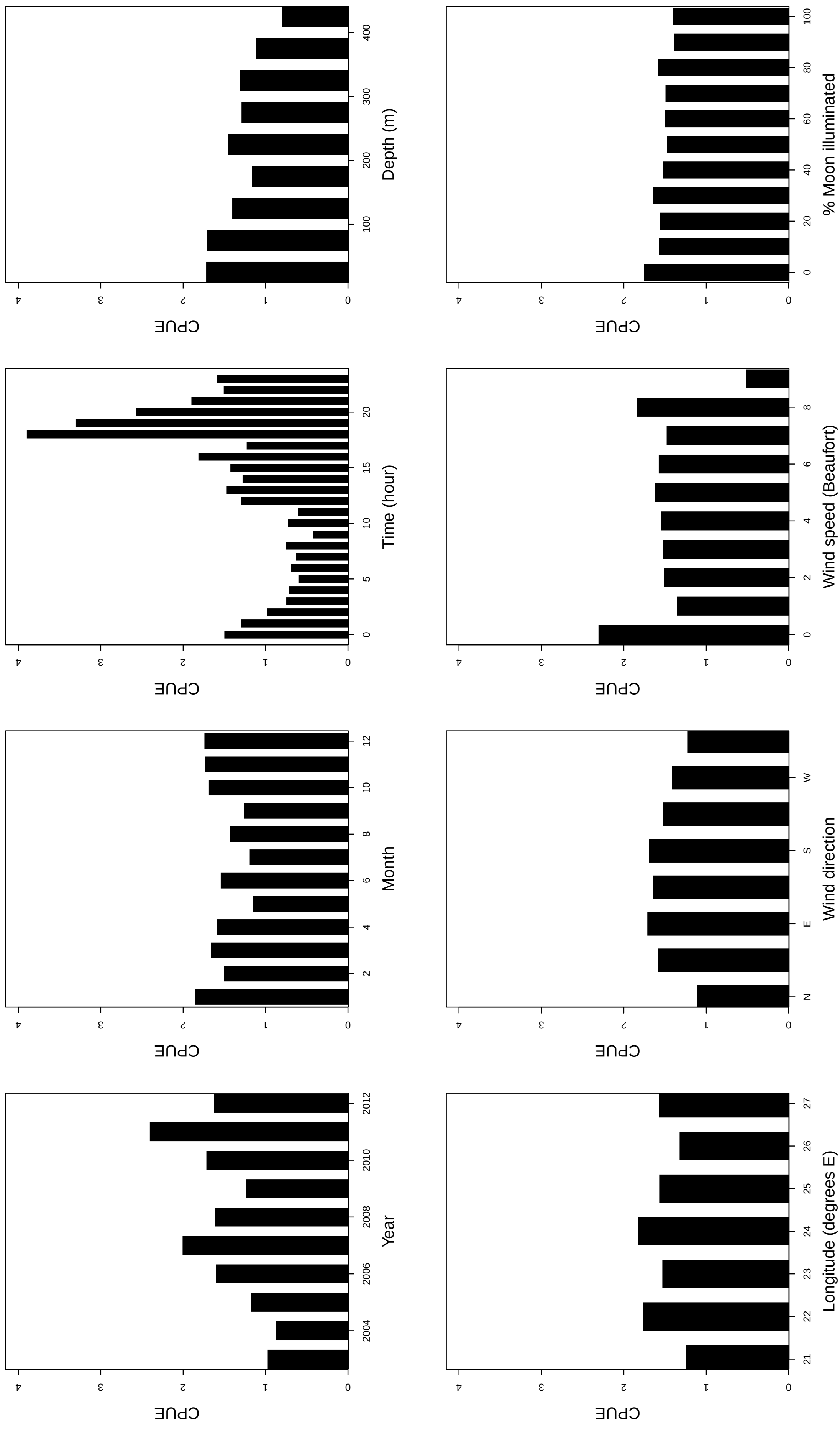

Figure 3.A.1: Relationship between marginal CPUE and candidate GLM co-variates. For categorical co-variates, mean CPUE is calculated directly from the data for each co-variate level. For continuous co-variates, ranges are split into discrete groups, and then mean CPUE is calculated as for categorical co-variates.

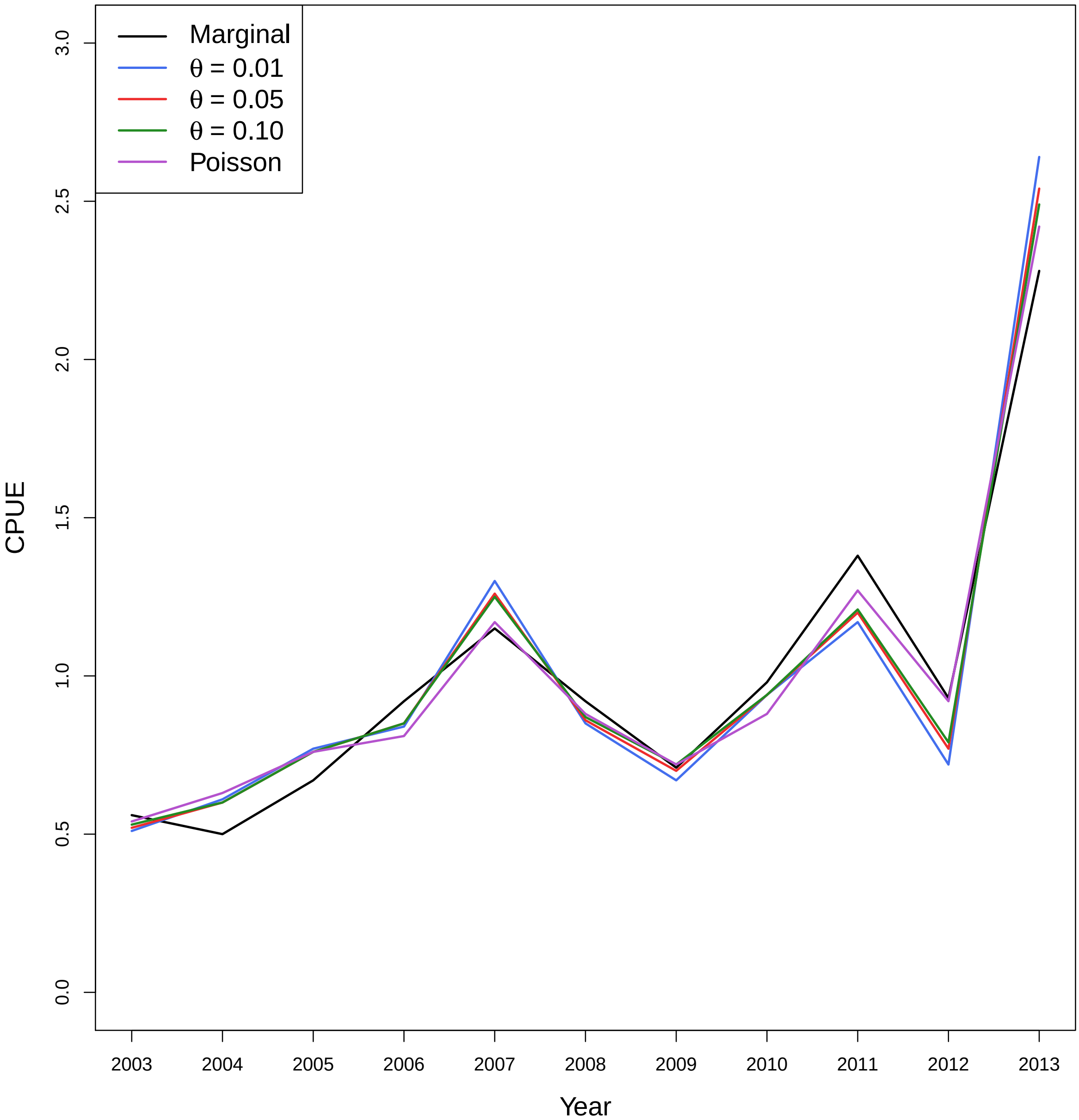

Figure 3.A.2: Comparison of the mean marginal CPUE series to the GLM standardised CPUE series produced by the log-normal models and the Poisson model. Note that although a CPUE value for 2013 is shown, that estimate is not used in the assessment model because at the time of writing data for 2013 were not all available.

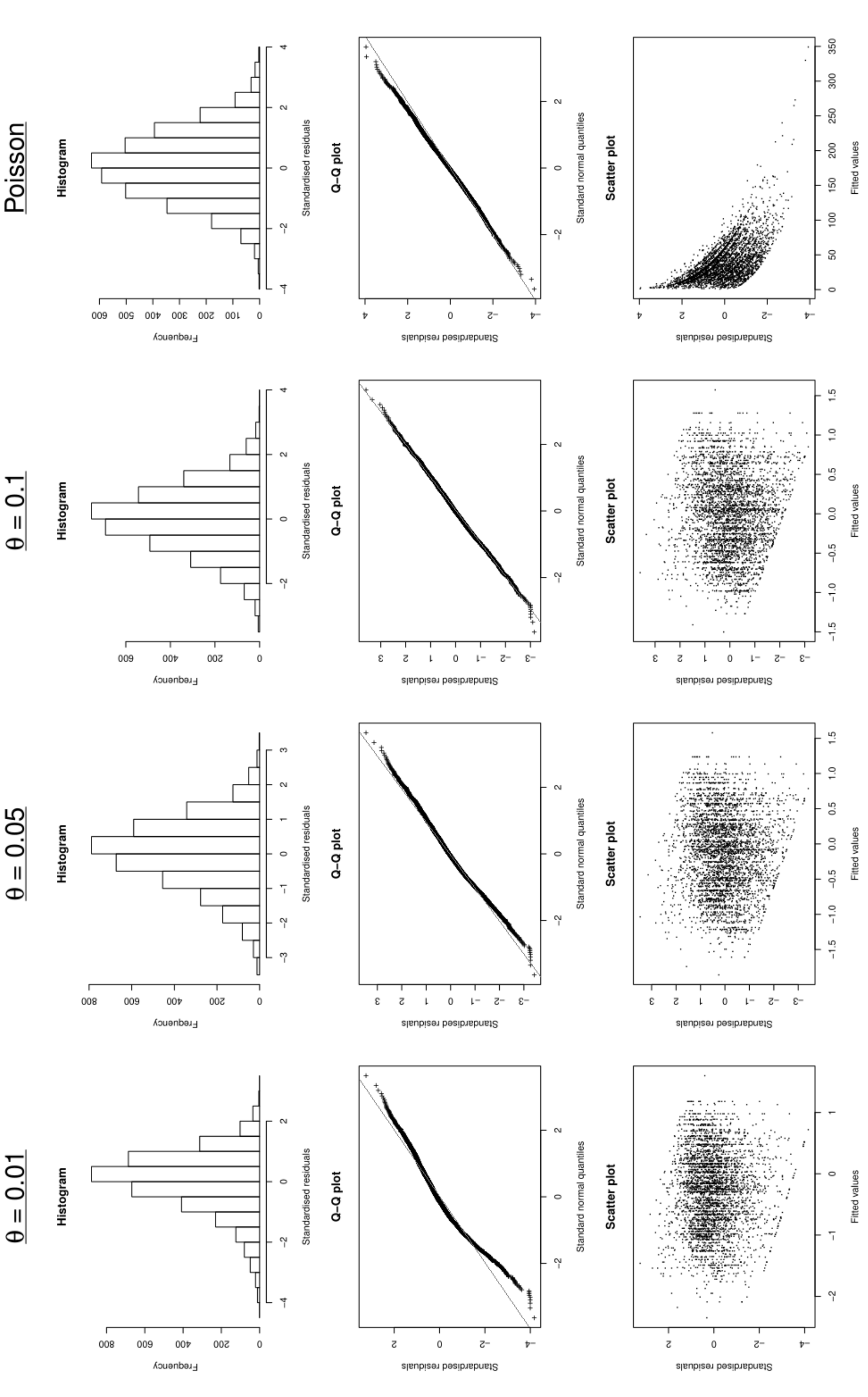

Figure 3.A.3: Diagnostic plots of the residuals from the four independent GLMs run on the commercial midwater data from the *Desert Diamond*. The rows of plots, working from top to bottom, present histograms, Q-Q plots and scatter plots of the standardised residuals against the fitted values.

Chapter 4

Assessment

This chapter introduces the Cape horse mackerel assessment, before MPs for the pelagic and midwater fisheries are considered. The first section outlines the assessment model, while Appendix 4.A gives the full model specifications. The following sections detail all of the model variants included in the Reference Set, the procedure for projecting the resource into the future, and the sensitivity tests. The chapter then concludes with a discussion of the assessment results. Appendix 4.B provides additional information on the development of time-varying fishing selectivity functions for the demersal and midwater fleets.

4.1 Population model

One of the goals of this work is to develop a MP for the pelagic fishery that is able to deal effectively with the unpredictable variations in recruitment that are characteristic of the horse mackerel resource. Catch-at-length data are therefore included in the assessment model to allow past recruitment fluctuations to be estimated. Virtual population analysis (VPA) or ASPM approaches seems to be preferable to biomass dynamics models because these data can be taken into account. ASPM is used over VPA, because it is more flexible, allowing extension to years for which catch-at-length data are not available.

The assessment process involves developing a model of the resource dynamics and conditioning its output to the available data by minimising a log-likelihood function. A single-stock model is used in this work. It is based largely on a previous assessment model by Johnston and Butterworth (2007), except that a midwater CPUE series, time-varying selectivity, and the aforementioned catch-at-length data and recruitment fluctuations are incorporated. The implementation is in

AD Model Builder$^{\text{TM}}$ by Otter Research, Ltd.

For the sake of brevity, only important features of the model are described in the following subsections. Full specifications are given in Appendix 4.A.

4.1.1 Dynamics

The ASPM reflects the dynamics of the resource over the period 1949–2013. In the past, the South African horse mackerel resource has been managed as two area-disaggregated stocks: West Coast and South Coast. This two-stock hypothesis was supported by differences in the size distributions of catches and growth rates between coasts (Crawford, 1981; Hecht, 1976). When investigating stock structure, Naish (1990) found similar differences; however, molecular genetic information strongly supported a single-stock hypothesis. Consequently, the resource has been managed as a single stock since 2001.

It is assumed that the South African horse mackerel population was in equilibrium at its carrying capacity in 1949. In reality, horse mackerel catches have been taken as bycatch in other fisheries since the 1900s but these catches were recorded only from 1949 shortly after substantial development of the pelagic fishery commenced. Nevertheless, the cumulative catch before then is unlikely to be high. The model's robustness to this assumption is evaluated using a sensitivity test in Section 4.4.

Pope's approximation to the Baranov equations is used to determine fishing mortality (Pope, 1972). It assumes that all catches are taken as a pulse in the middle of the fishing season, instead of continuously throughout. These simplified catch equations come at the cost of accuracy. However, it has been shown that provided mortality rates are not too high—as is the case for horse mackerel—the approximation error will be small (Branch, 2009).

The number of recruits at the start of a new year is related to the biomass of the mature component of the population (i.e. spawning biomass) of the previous year by a stock-recruitment relationship. A Beverton-Holt form is assumed. Additionally, stock-recruitment residuals that reflect natural fluctuations about expected recruitment are estimated for the years 1986–2011. This specific period is chosen because catch-at-length data, which can inform on the age-structure of the stock, are available for those years. Despite this variability in recruitment, the model assumes that at the start of the fishery in 1949, the population is stable at its unexploited equilibrium. The robustness to this assumption is explored in Section 4.4 using a sensitivity test.

Johnston and Butterworth (2007) set selectivity-at-age vectors externally for each fleet; these were developed after examining the length distributions of catches from the relevant fisheries. However, they result in poor fits to the newly incorporated midwater and demersal catch-at-length data. Selectivity functions for these fleets are therefore now estimated during the fitting procedure, and are assumed to have a Gaussian dependence on length. Demersal fishing selectivity is additionally assumed to vary over time, because the corresponding catch-at-length data show distinct patterns over the years. However, the confounding between time-varying selectivity and catchability introduces difficulties; this is addressed by normalising under the assumption that the demersal survey catchability remains constant over time. Appendix 4.B discusses these selectivity functions in greater detail.

4.1.2 Likelihoods

The assessment model is conditioned on survey abundance and catch-at-length data, and on commercial CPUE and catch-at-length data. Additional contributions to the negative of the (penalised) log-likelihood come from the stock-recruitment residuals and various penalty functions. They are discussed below.

The midwater CPUE and demersal survey time-series are both considered to be relative indices of abundance, each proportional to the biomass available to their respective fleets at midyear. However, without any estimates of biomass in absolute terms, the model is unable to estimate catchability coefficients for these indices reliably. The autumn survey is therefore treated as an absolute index by fixing its catchability to one of two values considered to be reasonable bounds. The value of this catchability parameter is a key uncertainty of the model. Likelihoods are calculated by assuming that observed indices are log-normally distributed about their expected values. Although estimates of sampling variability are given for each demersal survey (Section 3.2), the model estimates additional variance because there are likely to be other sources of variability; otherwise unrealistically high precision, and hence weight in the fitting procedure, would be accorded to these indices.

Because the assessment model is age-structured, catch-at-age estimates must be transformed into catch-at-length estimates before they can be compared to the observed catch-at-length data. This is done via an age-length matrix that is based on an input von Bertalanffy growth curve. The likelihood contributions are then calculated by comparing the model-predicted length distribution of horse mackerel catches with empirical data. Errors are assumed to be log-normally distributed.

The stock-recruitment residuals are also assumed to be log-normally distributed with no autocorrelation. Unfortunately, their variability cannot be estimated within the maximum likelihood framework used in this assessment, because the penalised likelihood function will always yield a minimum in the limit of the extent of this variability approaching zero. This issue is somewhat problematic, because recruitment fluctuations are of particular importance to the testing of pelagic MPs. While it could be dealt with by adopting a fully Bayesian methodology, it is simpler and adequate for present purposes to input the standard deviation for those residuals as a fixed value.

Finally, there are contributions to the negative log-likelihood from penalty functions. These do not correspond to any particular observed data or prior knowledge, but are instead included to discourage the optimisation routine from moving into unrealistic regions of the parameter space, such as those resulting in negative population counts or fishing mortality. All of the models presented in this work achieved convergence without triggering these penalty functions.

4.1.3 Parameters

Estimable parameters

A complete list of the forty parameters estimated by the model fitting procedure is given below:

K^{sp} is the pre-exploitation spawning biomass of horse mackerel;

q_{spr} is the catchability coefficient for the spring demersal survey abundance index;

ς_y is the fluctuation about the expected recruitment for year y, which is estimated for years 1986–2011;

μ^m is the centre of the Gaussian selectivity-at-length curve for the midwater fleet;

λ^m controls the width of the Gaussian selectivity-at-length curve for the midwater fleet;

$\mu^d_{y_1-y_2}$ is the centre of the Gaussian selectivity-at-length curve for the demersal fleet for years y_1–y_2, and is estimated for the periods 1949–1993, 1994–1997, 2004–2006 and 2007+;

$\lambda^d_{y_1-y_2}$ controls the width of the Gaussian selectivity-at-length curve for the demersal fleet for years y_1–y_2, and is estimated for the periods 1949–1993, 1994–1997, 2004–2006 and 2007+; and

σ^s_{add} is the square root of the additional variance for survey abundance index s (s is either aut for the autumn survey, or spr for the spring survey), and reflects variability not included in the corresponding survey CVs.

Input parameters

Some parameters cannot be estimated by the model, or are adequately specified by other studies and need not be estimated. They are therefore input with fixed values. The following is a list of these parameters:

q_{aut} is the catchability coefficient for the autumn demersal survey abundance index, and is assumed to be either 1, 0.75 or 0.5 (Section 4.2);

h is the "steepness" of the Beverton-Holt stock-recruitment function, and is assumed to be either 0.6, 0.75 or 0.9 (Section 4.2);

M is the natural mortality rate of horse mackerel, and is fixed at 0.3 yr^{-1}; although this choice is somewhat arbitrary (Johnston and Butterworth, 2007), Horsten (1999c) found key ASPM results to be fairly robust to alternative assumptions regarding this value (included as a sensitivity test in Section 4.4);

a_m is the age-at-maturity for South African horse mackerel, and is described by a knife-edge function of age with 100% of the population being sexually mature at 3 years (Butterworth and Clark, 1996; Hecht, 1990);

l_a is the expected length of a fish at age a in centimetres, and is based on the von Bertalanffy growth function given by Equation 2.1 and the growth parameters reported in Table 2.1

w_a is the weight in metric tonnes of a fish at age a, and is based on the length-at-age relationship described above, in combination with the mass-at-length function given by Equation 2.2 and the growth parameters reported in Table 2.1;

S^p_{a,y_1-y_2} is the fishing selectivity for the pelagic fleet for a fish at age a for years y_1–y_2, and is listed in Table 4.1 for the periods 1949–1962, 1963–1967 and 1968+;

σ_R is the standard deviation of the stock-recruitment log-residuals, and is assumed to be equal to 0.5, which is roughly typical for a species like horse mackerel;

γ is the CV of the length distribution of horse mackerel at any given age, and is assumed to be equal to 0.09 because this value provides good fits to catch-at-length data and lies within the expected range for a species like horse mackerel;

w_{cal} is the weighting of the catch-at-length likelihood contributions, and is fixed at 0.35 (a weighting of 1 is equivalent to being "unweighted").

4.2 Model variants

Given the limited data available at present, the assessment model is unable to reliably estimate the parameters q_{aut} (autumn survey catchability) and h (stock-recruitment steepness). Hence, they must be set externally. Note that q_{aut} can be thought of as a measure of the bias in the survey absolute biomass estimates (Section 3.2.1). For example, a value of 0.5 means that actual biomass is twice as large as the swept-area estimate from the surveys, whereas a value of 1 would mean that these surveys provide unbiased results. h determines the productivity of the resource, with a larger h corresponding to greater productivity. Appropriate values for these parameters are key uncertainties in the model.

Johnston and Butterworth (2007) identify the following combinations of q_{aut} and h as covering a realistic range:

- Model 1: $q_{aut} = 1.0$, $h = 0.6$ (most pessimistic)
- Model 2: $q_{aut} = 0.5$, $h = 0.6$
- Model 3: $q_{aut} = 1.0$, $h = 0.9$
- Model 4: $q_{aut} = 0.5$, $h = 0.9$ (most optimistic)

Additionally, these h values fall within the range expected for a demersal species according to the meta-analysis of stock-recruitment relationship steepness from Shertzer and Conn (2012).

These four variants are considered to be equally plausible and form the Reference Set of OMs. Model 1 and Model 4 are of particular importance, in that they are the most pessimistic and optimistic models respectively. When testing MPs, the pessimistic model will typically be the one that shows the greatest risk of resource depletion, while the optimistic model will give the best catch performance that can be expected.

Although results for all model variants in the Reference Set are often presented, it is sometimes impractical to do so. In these instances, only results corresponding to a Base Case model are shown. The model variant with $q_{aut} = 0.75$ and $h = 0.75$ is selected as the Base Case, because these parameters values fall in the middle of the bounds defined by the Reference Set. It is assumed that the Base Case model's results are reasonably representative of those of the Reference Set.

4.3 Projections

Projections are simulations of the future state of a fishery given present understanding of the resource dynamics as represented by an assessment model. By providing a basis to calculate fishery performance statistics, they give a means of testing CMPs and enable stake-holders to make informed decisions about trade-offs. In this section we look at projections under the assumption of constant future catches. Although in future such catches would of course vary, the results nevertheless provide a ready illustration of the effects of increasing catch allocations to the pelagic and midwater fisheries.

All permutations of the following constant catch scenarios are considered.

Future pelagic catch scenarios

- 0 tonnes annually
- 5 000 tonnes annually
- 10 000 tonnes annually
- 15 000 tonnes annually

Future midwater catch scenarios

- 35 000 tonnes annually
- 50 000 tonnes annually

Future annual demersal catches (which reflect unavoidable bycatch in the hake trawl fishery) are assumed to remain fixed at 5 463 tonnes per annum, which is the level reported for 2012 (the final year for which historical catch data were available at the time of this study).

The horse mackerel resource is projected 30 years into the future. Because there are stochastic elements in the model dynamics, 1 000 projections, each using different random numbers, are simulated for each future catch scenario, as explained below. This allows for realistic estimates of performance statistics. Additionally, the random number generator is seeded with the same value at the start of each set of 1 000 projections in order to eliminate the variability that would result from using different seeds; this allows for readier comparisons between scenarios.

To simplify projections, the time-varying fishing selectivities for the pelagic and demersal fleets are assumed to remain in the future at their 2012 values. Future stock-recruitment residuals are drawn randomly from a normal distribution with a standard deviation of σ_R. Additionally, they are assumed to be serially correlated, with a Pearson correlation coefficient of 0.47. This r value is taken from the serial correlation of the model-estimated residuals.

Future “observed” midwater CPUE, and autumn demersal and pelagic survey biomass estimates are generated during projections, because, as will be concluded from later chapters, these indices of abundance are potentially useful as inputs for many CMPs. Realistic observation errors are added to the expected values of these abundance indices by drawing them at random from the same log-normal distributions assumed in the assessment model (Equation 4.A.24,4.A.26, 4.A.28 and 4.A.29). The variance of the error distributions for the CPUE and pelagic survey indices are estimated in the assessment (Equation 4.A.30), while the variance for an autumn demersal survey abundance estimate is a combination of the estimated σ^{aut}_{add} (additional variance) and a CV (Equation 4.A.27). Future CVs are drawn randomly with replacement from historic autumn survey CVs (Table 3.3).

4.4 Sensitivity tests

When developing assessment models, it is often necessary to make intuitive assumptions for which there is little supporting evidence. The MSE approach requires that the robustness of the models to these assumptions is assessed with the help of sensitivity tests.

With twelve such tests in total considered here, it would be infeasible to show results for all four model variants in the Reference Set. Instead, the robustness of the Base Case model is reported in Section 4.5 by comparing its estimated spawning biomass trajectory and associated values of quantities pertinent to management to those of the various sensitivity tests (which are variants of this Base Case model).

Test A: Combined midwater and demersal catch series - "Alt. 88–99 catches"

As mentioned in Section 3.1, there are two conflicting reports of annual combined midwater and demersal catches for the period 1988–1999. Consequently, the means of the two datasets are used in the Base Case. This sensitivity test is performed by fitting the model when using Dataset 2 instead. This dataset is selected over Dataset 1, because it reports higher overall catches and thus is likely to lead to a more pessimistic assessment.

Test B: Midwater and demersal catch split - "Alt. catch split"

The assessment model requires separate historic catch series for both the midwater and demersal horse mackerel fisheries; however, for catches before 2000 only combined annual figures are available. In the Base Case these catches are allocated by assuming the same proportional split between fisheries as was reported in Johnston and Butterworth (2007). In this sensitivity test, it is assumed that all such combined figures reflect midwater catches only. The midwater fleet is chosen here in order to maximise contrast, because in the Base Case almost all of the catch over the period in question is allocated to the demersal fleet.

Test C: GLM-standardised CPUE index - "Poisson GLM"

In Appendix 3.A, four options were presented for the GLM standardised commercial midwater CPUE index. The log-normal GLM with $\theta = 0.05$ was selected because it is the most consistent with model assumptions. In this sensitivity test, the model is conditioned on the CPUE series produced by the Poisson GLM, because it differs most from that used in the Base Case model.

Test D: Variability in stock-recruitment residuals - "$\sigma_R = 0.7$"

The variability of stock-recruitment log-residuals, σ_R, cannot be estimated within a frequentist framework, hence in the Base Case model it is input with a value of 0.5. The robustness of the model to the value of this parameter is assessed by setting $\sigma_R = 0.7$. A higher value is chosen as greater variability will provide a more difficult test of an MP to achieve satisfactory risk-related performance.

Test E: Catch-at-length likelihood weighting - "$w_{cal} = 0.2$"

Catch-at-length contributions to the negative log-likelihood are down-weighted, because there are many more catch-at-length data—with ten length classes for each year—than there are abundance data, and contributions from the abundance indices should play a leading role. Furthermore, catch-at-length data are not independent but tend to be positively correlated, suggesting a value for $w_{cal} < 1$. In the Base Case model this weighting is fixed at 0.35. This sensitivity test explores the effects of instead using a lesser weighting of $w_{cal} = 0.2$.

Test F: Catch-at-length likelihood weighting - "$w_{cal} = 0.7$"

In the Base Case model $w_{cal} = 0.35$, but this sensitivity test uses a larger weighting of $w_{cal} = 0.7$.

Test G: Fitting to pelagic surveys - "Incl. pel. survey"

The model has little power to estimate fluctuations about expected recruitment, ς_y. It may therefore be beneficial to condition the model on West Coast pelagic hydro-acoustic survey biomass estimates (Section 3.3). Note that although these data are also available for the entire survey area, it is more likely that the West Coast estimates are correlated with horse mackerel recruitment. The associated likelihood component is described in full in Appendix 4.A

Test H: Variability in length-at-age - "Alt. LAA var."

Model catch-at-age estimates are converted into catch-at-length estimates with a fixed age-length matrix. In the Base Case it is assumed that at any given age, fish lengths are normally distributed with a standard deviation proportional to the expected length (and a constant of proportionality of 0.9). Here, the standard deviation of the length-at-age distribution is instead set to a constant value of 3 cm.

Test I: Functional form of selectivity-at-length - "Log-normal sel."

Fishing selectivity for the midwater and demersal fleets are assumed to have Gaussian dependence on length; however, fits to catch-at-length data suggest that other functional forms might be more appropriate. This sensitivity test investigates the effects of instead using log-normal selectivity-at-length, with both the location and scale parameters being estimated.

Test J: Normalising time-varying selectivity - "Alt. sel. normalisation"

In the Base Case model, time-varying selectivity for the demersal fleet is normalised each year by dividing it by its mean over the 10–40 cm range. Here, demersal selectivity is instead normalised by dividing it by the largest selectivity in a year (i.e. the maximum demersal selectivity for each year is always equal to one).

Test K: Natural mortality - "$M = 0.5$"

In the Base Case model natural mortality, M, is constant for all ages and is input as 0.3. Following a suggestion in Johnston and Butterworth (2007), sensitivity analyses are reported for $M = 0.5$.

Test L: Initial spawning biomass - "$B_0 = 0.8K^{sp}$"

It is assumed in the Base Case model that spawning biomass was at its pristine level, K^{sp}, at the reported start of the horse mackerel fishery in 1949. However, that may not necessarily have been the case, especially given the large initial catches in the fishery. Robustness to this assumption is tested by setting initial spawning biomass to a value of $0.8K^{sp}$.

Test M: Recruitment residuals - "Recr. from 1976"

Recruitment residuals were estimated for 1986–2011 because this is the period for which catch-at-length data are available. However, there is likely some information in the data regarding recruitment for years prior to 1986. This sensitivity test explores the effect of estimating recruitment residuals for the period 1976–2011.

4.5 Results and Discussion

A summary of results for the models in the Reference Set—conditioned on the data shown in Figures 3.1, 3.3 and 3.4—is given in Table 4.2. It indicates that Model 2 and Model 4, both of which assume $q_{aut} = 0.5$, fit the data slightly better (by about two log-likelihood points) than the other model variants. The largest improvements are in the fits to the autumn demersal survey abundance index. This provides some further support for the notion that the demersal swept-area surveys are negatively biased. Best estimates of the current status of the resource are that it is healthy, from 37% to 80% of its pre-exploitation level according to the pessimistic model or the optimistic model respectively. MSY is predicted to be in the region of 50 000–100 000 tonnes per annum, occurring when spawning biomass is between 17% and 31% of its pre-exploitation level, again depending on the model variant. These MSY estimates are, however, not entirely appropriate, as they are calculated by assuming that all horse mackerel are caught by the midwater fleet. This fleet is chosen because in recent years it has caught the great majority of horse mackerel and will continue to do so for the foreseeable future. Additionally, because fishing selectivity and effective weight-at-age for the midwater fleet changes with time, for MSY calculations they are assumed to remian fixed at their 2012 values. If some juveniles are taken as bycatch in the pelagic fishery—as will likely always be the case—then the actual MSY will be lower than reported, because its yield-per-recruit is appreciably lower than that for the midwater fishery. Figure 4.1 shows stock-recruit pairs for the Base Case model.

Figure 4.2 shows that the Reference Set models fit the observed abundance indices reasonably well. Even so, there does seem to be some difficulty in matching the low biomass level in the latter half of the 1980s reflected by the surveys. The first two data points for the spring demersal survey index are particularly problematic and result in the poor fits to later values in that series. Poor fits to the demersal abundance indices from 2006 and onwards also warrant investigation. These issues can be partly explained by the large additional variance estimated for both surveys, ($\sigma_{add}^{aut} \approx 0.3$ and particularly $\sigma_{spr}^{add} \approx 0.7$), indicating that sampling variability alone is not sufficient to explain all of the variability in these indices; possibly the first two points for the spring survey are not comparable to the rest. Additionally, these demersal swept-area surveys are not an entirely reliable method for estimating horse mackerel abundance (Section 3.2.1).

Figure 4.3 compares observed catch-at-length data to the corresponding model estimates by averaging over the years for which such data are available. They fit the observations satisfactorily, with only slight differences between the model variants. However, there is some evidence of misspecification. For example, the model has trouble fitting to the demersal survey data for 1998–2003, which is likely related to the fact that the demersal selectivity function for this period is an average of the selectivity functions for the preceding period (1994–1997) and succeeding period (2004–2006), instead of being estimated freely like the others. It is also clear from Figure 4.3 that there has been a gradual change with time in the length distribution of demersal by-catches. Over the period 1986–1993 catches were widely spread across lengths, with a mode at 25–30 cm, while in recent years the distribution has narrowed and catches have been composed mostly of smaller fish in the 10–25 cm range. Figure 4.4 plots the standardised catch-at-length residuals to allow for the identification of systematic effects. It indicates that the model consistently overestimates the proportion of fish caught in the 30–35 cm range, most notably in the spring demersal surveys. This, along with the issues mentioned above, suggests that efforts to improve the quality of fits to catch-at-length data may be worthwhile.

Figure 4.5 and Figure 4.6 show model estimated spawning biomass and recruitment trajectories respectively. Figure 4.5 demonstrates that, as expected, the choice of input values for q_{aut} and h has a large impact on spawning biomass as a proportion of its pre-exploitation level (B^{sp}/K^{sp}). In contrast, the effect of h is less pronounced in terms of absolute spawning biomass, because it is q_{aut} that determines the absolute level of the autumn survey biomass estimates on which the models are conditioned. The large decrease in abundance in the 1950s reflects large landings of horse mackerel by the pelagic fleet. As these catches were reduced, and were replaced by demersal catches of adults, the resource slowly recovered.

However, several years of poor recruitment near the end of the 1980s (Figure 4.6), coupled with large pelagic catches (a high of 25 872 tonnes in 1989) brought the abundance to near historic lows. After this drop, the stock again recovered until the mid 2000s when a combination of strong recruitment (Figure 4.6) and the replacement of demersal catches with those by the midwater fleet (thereby targeting older fish) led to a substantial improvement in spawning biomass. Estimates of the current status of the resource encouragingly put the stock well above Maximum Sustainable Yield Level (MSYL).

Figure 4.7 plots the projected median spawning biomass trajectories for each combination of the future pelagic and midwater catch scenarios described in Section 4.3. Note that the colour of a line signifies which pelagic catch scenario is assumed in the corresponding projection, while a solid or dot-dashed line reflects the assumed future midwater catch scenario. From this figure it is again clear that the input value for q_{aut} has a larger effect on the projected future status of the resource in absolute terms than h. Of sixteen projections with $q_{aut} = 0.5$, only one shows the stock becoming completely depleted within 30 years, while of those with $q_{aut} = 1$ there are eight. Similarly, only three of sixteen projections with $h = 0.9$ predict resource depletion, while of those with $h = 0.6$ there are six. Thus the optimistic and pessimistic input values for q_{aut} lead to a wider range of optimistic to pessimistic projections, than those for h. Figure 4.7 gives a glimpse as to why the decision was made in the assessment by Johnston and Butterworth (2007) to restrict the pelagic PUCL to 5 000 tonnes and the midwater allocation to 31 500 tonnes. This situation closely corresponds to the future pelagic and midwater constant catch scenarios of 5 000 tonnes and 35 000 tonnes per annum respectively, which is the only one that allows a reasonable PUCL without appreciable risk of resource depletion for any model in the Reference Set. Finally, this figure demonstrates that the status of the horse mackerel resource is very sensitive to catches of juveniles, with an increase of 5 000 tonnes per annum in the pelagic fishery being roughly equivalent to an increase of 15 000 tonnes per annum in the midwater fishery. This is true across all model variants in the Reference Set, but may differ at levels of depletion not considered in these projections.

Table 4.3 and Table 4.4 give management quantity estimates, while Figure 4.8 illustrates spawning biomass trends, for all sensitivity tests listed in Section 4.4. The plots indicate that only Test J (alternate selectivity normalisation), Test K (larger natural mortality) and Test M (recruitment residuals estimated from 1976) are appreciably different from the Base Case model; however, Table 4.4 shows that Test K and Test M have negligible effects on the estimates of several important management quantities such as MSY, B^{sp}_{2013}/K^{sp} and $B^{sp}_{2013}/MSYL^{sp}$. Test M has a large impact on predicted spawning biomass between 1975 and 1995, but it does not strongly affect estimates of recent spawning biomass. Test J is therefore of primary concern. Figure 4.9 shows the selectivity-at-length curves for the midwater and demersal fleets for the Base Case model and its Test J variant. The larger abundance arises from the fact that the normalisation method used for Test J (scale by the inverse of the largest selectivity) leads to smaller curves than the method used for the Base Case (scale by the inverse of the average selectivity over the 10–40 cm range). Because of the confounding of catchability and selectivity, the smaller selectivity curves have the same effect as decreasing catchability. However, because the autumn survey catchability is fixed and not estimated in this model, total abundance must increase in order to provide reasonable fits to the autumn demersal survey biomass estimates. This issue demonstrates that the choice of scaling for the demersal selectivities has an appreciable impact on management of the resource and should be thoroughly investigated in future work. The midwater and demersal selectivity functions are discussed in greater detail in Appendix 4.B.

4.5.1 Retrospective analysis

A retrospective analysis was conducted by one year a time omitting the data that became available during the most recent year. Figure 4.10 gives these results. They show evidence of systematic effects in model-estimated historical spawning biomass (both in absolute and relative terms); however, there is little difference between estimates of recent spawning biomass.

Table 4.1: Selectivity-at-age vectors assumed for the pelagic fleet over three different periods (Johnston and Butterworth, 2007)

Age	Period		
(yr)	1949–1962	1963–1967	1968+
0	0.00	0.14	0.28
1	0.00	0.50	1.00
2	0.30	0.40	0.50
3	1.00	0.50	0.00
4	0.50	0.25	0.00
5	0.50	0.25	0.00
6	0.25	0.13	0.00
7	0.00	0.00	0.00
8	0.00	0.00	0.00
9	0.00	0.00	0.00
10+	0.00	0.00	0.00

Table 4.2: Summary of results for the Reference Set assessment models for the Cape horse mackerel resource. Although q_{aut} and h are fixed for all models, they are listed here to remind the reader which model variant corresponds to which combination of these parameter values. The first numbers shown are the best estimates, while the figures in parentheses are the Hessian-based CVs. "SR" and "CAL" refer to stock-recruitment and catch-at-length respectively. Biomass is reported in units of kilotonnes.

	Model 1	**Model 2**	**Model 3**	**Model 4**
q_{aut}	1.0	0.5	1.0	0.5
h	0.6	0.6	0.9	0.9
$-\ln L$: Total	−219.92	−221.36	−219.25	−221.53
$-\ln L$: Spr. survey	1.48	1.26	1.44	1.25
$-\ln L$: Aut. survey	−8.08	−8.94	−6.40	−8.83
$-\ln L$: CPUE	−12.44	−11.94	−12.01	−11.89
$-\ln L$: CAL spr. survey	−46.92	−47.00	−47.43	−47.26
$-\ln L$: CAL aut. survey	−86.12	−86.11	−86.36	−86.17
$-\ln L$: CAL commercial	−53.95	−54.32	−54.10	−54.25
$-\ln L$: SR residuals	−13.88	−14.31	−14.40	−14.38
K^{sp} (KT)	862 (0.04)	1 187 (0.08)	735 (0.07)	1 059 (0.09)
B^{sp}_{2013} (KT)	317 (0.21)	838 (0.16)	428 (0.10)	842 (0.16)
$MSYL^{sp}$ (KT)	264 (0.05)	362 (0.08)	129 (0.03)	184 (0.09)
MSY (KT)	52 (0.05)	70 (0.08)	70 (0.03)	100 (0.09)
B^{sp}_{2013}/K^{sp}	0.37 (0.18)	0.71 (0.10)	0.58 (0.10)	0.80 (0.10)
$B^{sp}_{2013}/MSYL^{sp}$	1.20 (0.18)	2.32 (0.10)	3.33 (0.10)	4.57 (0.10)
$MSYL^{sp}/K^{sp}$	0.31 (0.01)	0.31 (0.01)	0.18 (0.03)	0.17 (0.03)
q: Spr. survey	1.03 (0.28)	0.52 (0.27)	0.88 (0.27)	0.52 (0.27)
q: CPUE $(\times 10^{-6})$	2.49 (0.16)	1.14 (0.14)	1.99 (0.10)	1.13 (0.14)
σ: Additional spr. survey	0.71 (0.28)	0.69 (0.28)	0.71 (0.27)	0.69 (0.28)
σ: Additional aut. survey	0.29 (0.39)	0.27 (0.39)	0.34 (0.30)	0.27 (0.38)
σ: CPUE	0.18 (0.25)	0.18 (0.22)	0.18 (0.23)	0.19 (0.22)
σ: CAL spr. survey	0.09 (0.11)	0.09 (0.11)	0.09 (0.08)	0.09 (0.11)
σ: CAL aut. survey	0.12 (0.06)	0.12 (0.06)	0.12 (0.05)	0.12 (0.06)
σ: CAL commercial	0.09 (0.07)	0.09 (0.06)	0.09 (0.06)	0.09 (0.06)
μ^m	33.14 (0.02)	32.68 (0.02)	32.95 (0.02)	32.68 (0.02)
$\mu^d_{1949-1993}$	35.68 (0.08)	33.70 (0.06)	34.98 (0.05)	33.97 (0.06)
$\mu^d_{1994-1997}$	24.02 (0.07)	23.53 (0.06)	23.31 (0.06)	23.52 (0.06)
$\mu^d_{2004-2006}$	31.27 (0.04)	30.59 (0.04)	31.18 (0.04)	30.62 (0.04)
μ^d_{2007+}	20.72 (0.05)	20.54 (0.05)	20.23 (0.04)	20.51 (0.05)
λ^m	5.04 (0.09)	5.00 (0.08)	5.05 (0.08)	5.01 (0.08)
$\lambda^d_{1949-1993}$	9.51 (0.18)	9.20 (0.16)	9.66 (0.12)	9.26 (0.17)
$\lambda^d_{1994-1997}$	4.92 (0.18)	4.98 (0.17)	5.00 (0.17)	5.01 (0.17)
$\lambda^d_{2004-2006}$	8.27 (0.21)	8.07 (0.19)	7.84 (0.19)	8.05 (0.19)
λ^d_{2007+}	6.22 (0.16)	6.28 (0.16)	5.93 (0.15)	6.26 (0.16)

Table 4.3: Summary of results for the Base Case ($q_{aut} = 0.75, h = 0.75$) assessment model of the Cape horse mackerel resource, and sensitivity tests A–F, which are variants thereof. The first numbers shown are the best estimates, while the figures in parentheses are the Hessian-based CVs. Refer to Section 4.4 for a description of these tests, and to Table 4.2 for clarification on some of the units and notations.

	Base case	Test A	Test B	Test C	Test D	Test E	Test F
		Alt. 88–99 catches	Alt. catch split	Poisson GLM	$\sigma_R = 0.7$	$w_{cal} = 0.2$	$w_{cal} = 0.7$
$-\ln L$: Total	−220.95	−220.73	−221.45	−222.39	−214.88	−142.34	−413.23
$-\ln L$: Spr. survey	1.38	1.41	1.49	1.48	1.27	0.31	2.03
$-\ln L$: Aut. survey	−8.25	−8.14	−7.84	−8.45	−8.85	−11.56	−5.30
$-\ln L$: CPUE	−12.15	−12.17	−12.01	−13.87	−12.38	−14.89	−9.43
$-\ln L$: CAL spr. survey	−47.18	−47.18	−47.93	−46.84	−47.19	−24.29	−101.19
$-\ln L$: CAL aut. survey	−86.32	−86.31	−86.49	−85.71	−89.15	−46.94	−181.77
$-\ln L$: CAL commercial	−54.01	−53.97	−54.09	−54.44	−53.58	−30.35	−108.06
$-\ln L$: SR residuals	−14.41	−14.38	−14.57	−14.56	−5.00	−14.61	−9.51
K^{sp} (KT)	841 (0.08)	846 (0.07)	831 (0.07)	844 (0.07)	831 (0.08)	843 (0.08)	821 (0.07)
B^{sp}_{2013} (KT)	497 (0.18)	497 (0.18)	490 (0.19)	507 (0.18)	498 (0.19)	482 (0.19)	477 (0.20)
$MSYL^{sp}$ (KT)	207 (0.07)	208 (0.07)	204 (0.07)	208 (0.07)	204 (0.08)	205 (0.08)	202 (0.07)
MSY (KT)	64 (0.08)	65 (0.08)	63 (0.07)	65 (0.07)	64 (0.08)	64 (0.09)	63 (0.07)
B^{sp}_{2013}/K^{sp}	0.59 (0.13)	0.59 (0.13)	0.59 (0.13)	0.60 (0.12)	0.60 (0.13)	0.57 (0.13)	0.58 (0.14)
$B^{sp}_{2013}/MSYL^{sp}$	2.41 (0.13)	2.39 (0.13)	2.40 (0.13)	2.44 (0.12)	2.44 (0.14)	2.35 (0.13)	2.36 (0.14)
$MSYL^{sp}/K^{sp}$	0.25 (0.01)	0.25 (0.01)	0.25 (0.01)	0.25 (0.01)	0.25 (0.01)	0.24 (0.01)	0.25 (0.00)
q: Spr. survey	0.78 (0.28)	0.77 (0.28)	0.77 (0.28)	0.79 (0.28)	0.77 (0.28)	0.81 (0.25)	0.78 (0.30)
q: CPUE $(\times 10^{-6})$	1.76 (0.15)	1.76 (0.15)	1.78 (0.15)	1.77 (0.15)	1.74 (0.16)	1.85 (0.16)	1.82 (0.16)
σ: Additional spr. survey	0.71 (0.28)	0.71 (0.28)	0.72 (0.27)	0.71 (0.28)	0.70 (0.28)	0.61 (0.30)	0.77 (0.27)
σ: Additional aut. survey	0.29 (0.37)	0.29 (0.36)	0.30 (0.35)	0.28 (0.37)	0.27 (0.43)	0.20 (0.60)	0.37 (0.28)
σ: CPUE	0.18 (0.23)	0.18 (0.23)	0.18 (0.23)	0.15 (0.25)	0.18 (0.27)	0.14 (0.29)	0.24 (0.17)
σ: CAL spr. survey	0.09 (0.11)	0.09 (0.11)	0.09 (0.11)	0.10 (0.11)	0.09 (0.12)	0.11 (0.14)	0.08 (0.08)
σ: CAL aut. survey	0.12 (0.06)	0.12 (0.06)	0.12 (0.06)	0.12 (0.06)	0.12 (0.06)	0.13 (0.07)	0.11 (0.04)
σ: CAL commercial	0.09 (0.06)	0.09 (0.06)	0.09 (0.06)	0.09 (0.06)	0.09 (0.07)	0.09 (0.09)	0.09 (0.06)

Table 4.4: Summary of results for the Base Case ($q_{aut} = 0.75, h = 0.75$) assessment model of the Cape horse mackerel resource, and sensitivity tests G–M, which are variants thereof. The first numbers shown are the best estimates, while the figures in parentheses are the Hessian-based CVs. Refer to Section 4.4 for a description of these tests, and to Table 4.2 for clarification on some of the units and notations.

	Base case	Test G Incl. pel. surv.	Test H Alt. LAA var.	Test I Log-norm. sel.	Test J Alt. sel. norm.	Test K $M = 0.5$	Test L $B_0 = 0.8K^{sp}$	Test M Recr. from 1976
$-\ln L$: Total	−220.95	−212.50	−218.86	−219.85	−222.56	−223.71	−221.01	-231.06
$-\ln L$: Spr. survey	1.38	1.37	1.40	1.26	0.83	−0.04	1.34	-0.18
$-\ln L$: Aut. survey	−8.25	−8.36	−8.36	−8.81	−11.13	−11.52	−8.31	-10.80
$-\ln L$: CPUE	−12.15	−12.19	−12.09	−11.32	−11.04	−12.50	−12.16	-12.21
$-\ln L$: CAL spr. survey	−47.18	−46.47	−46.14	−47.63	−45.99	−46.13	−47.15	-49.04
$-\ln L$: CAL aut. survey	−86.32	−87.41	−84.07	−84.02	−85.73	−85.22	−86.32	-84.72
$-\ln L$: CAL commercial	−54.01	−53.97	−54.50	−54.79	−54.88	−53.98	−54.00	-54.65
$-\ln L$: SR residuals	−14.41	−12.80	−15.09	−14.54	−14.61	−14.32	−14.41	-19.47
K^{sp} (KT)	841 (0.08)	851 (0.07)	841 (0.08)	825 (0.08)	1 275 (0.11)	434 (0.06)	839 (0.08)	811 (0.07)
B^{sp}_{2013} (KT)	497 (0.18)	489 (0.18)	513 (0.18)	484 (0.19)	969 (0.19)	259 (0.17)	494 (0.19)	516 (0.17)
$MSYL^{sp}$ (KT)	207 (0.07)	209 (0.07)	205 (0.08)	206 (0.08)	313 (0.15)	106 (0.08)	206 (0.08)	199 (0.07)
MSY (KT)	64 (0.08)	65 (0.08)	66 (0.08)	63 (0.08)	96 (0.11)	67 (0.07)	64 (0.08)	61 (0.08)
B^{sp}_{2013}/K^{sp}	0.59 (0.13)	0.58 (0.12)	0.61 (0.13)	0.59 (0.13)	0.76 (0.11)	0.60 (0.13)	0.59 (0.13)	0.64 (0.13)
$B^{sp}_{2013}/MSYL^{sp}$	2.41 (0.13)	2.34 (0.12)	2.50 (0.13)	2.35 (0.13)	3.10 (0.16)	2.44 (0.14)	2.40 (0.13)	2.57 (0.13)
$MSYL^{sp}/K^{sp}$	0.25 (0.01)	0.25 (0.01)	0.24 (0.01)	0.25 (0.01)	0.25 (0.10)	0.25 (0.06)	0.25 (0.01)	0.25 (0.01)
q: Spr. survey	0.78 (0.28)	0.77 (0.28)	0.77 (0.28)	0.77 (0.28)	0.78 (0.26)	0.75 (0.26)	0.78 (0.28)	0.80 (0.24)
q: CPUE $(\times 10^{-6})$	1.76 (0.15)	1.80 (0.15)	1.72 (0.15)	1.76 (0.16)	2.62 (0.19)	2.20 (0.13)	1.77 (0.15)	1.63 (0.14)
σ: Additional spr. survey	0.71 (0.28)	0.70 (0.28)	0.71 (0.28)	0.69 (0.28)	0.66 (0.28)	0.67 (0.33)	0.70 (0.28)	0.57 (0.31)
σ: Additional aut. survey	0.29 (0.37)	0.28 (0.37)	0.28 (0.37)	0.27 (0.38)	0.20 (0.52)	0.24 (0.43)	0.28 (0.37)	0.23 (0.51)
σ: CPUE	0.18 (0.23)	0.18 (0.22)	0.18 (0.23)	0.20 (0.21)	0.20 (0.21)	0.17 (0.25)	0.18 (0.23)	0.18 (0.19)
σ: CAL spr. survey	0.09 (0.11)	0.10 (0.11)	0.10 (0.10)	0.09 (0.13)	0.10 (0.11)	0.10 (0.11)	0.09 (0.11)	0.09 (0.12)
σ: CAL aut. survey	0.12 (0.06)	0.12 (0.05)	0.13 (0.05)	0.13 (0.06)	0.12 (0.06)	0.12 (0.05)	0.12 (0.06)	0.13 (0.06)
σ: CAL commercial	0.09 (0.06)	0.09 (0.06)	0.09 (0.07)	0.09 (0.07)	0.09 (0.06)	0.09 (0.06)	0.09 (0.06)	0.09 (0.06)

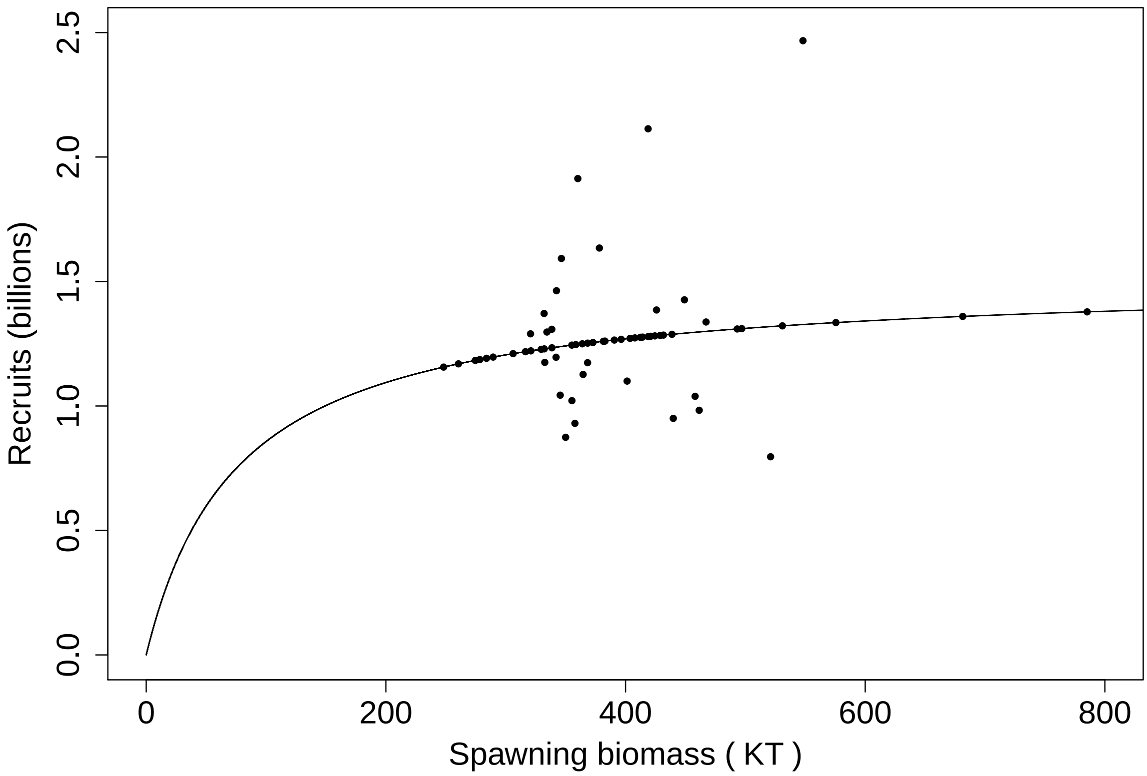

Figure 4.1: Stock-recruit pairs for the Base Case model with stock-recruit relationship. The points exactly on the curve are for the for which there are no data to estimate the stock-recruitment residuals so that these are set equal to zero.

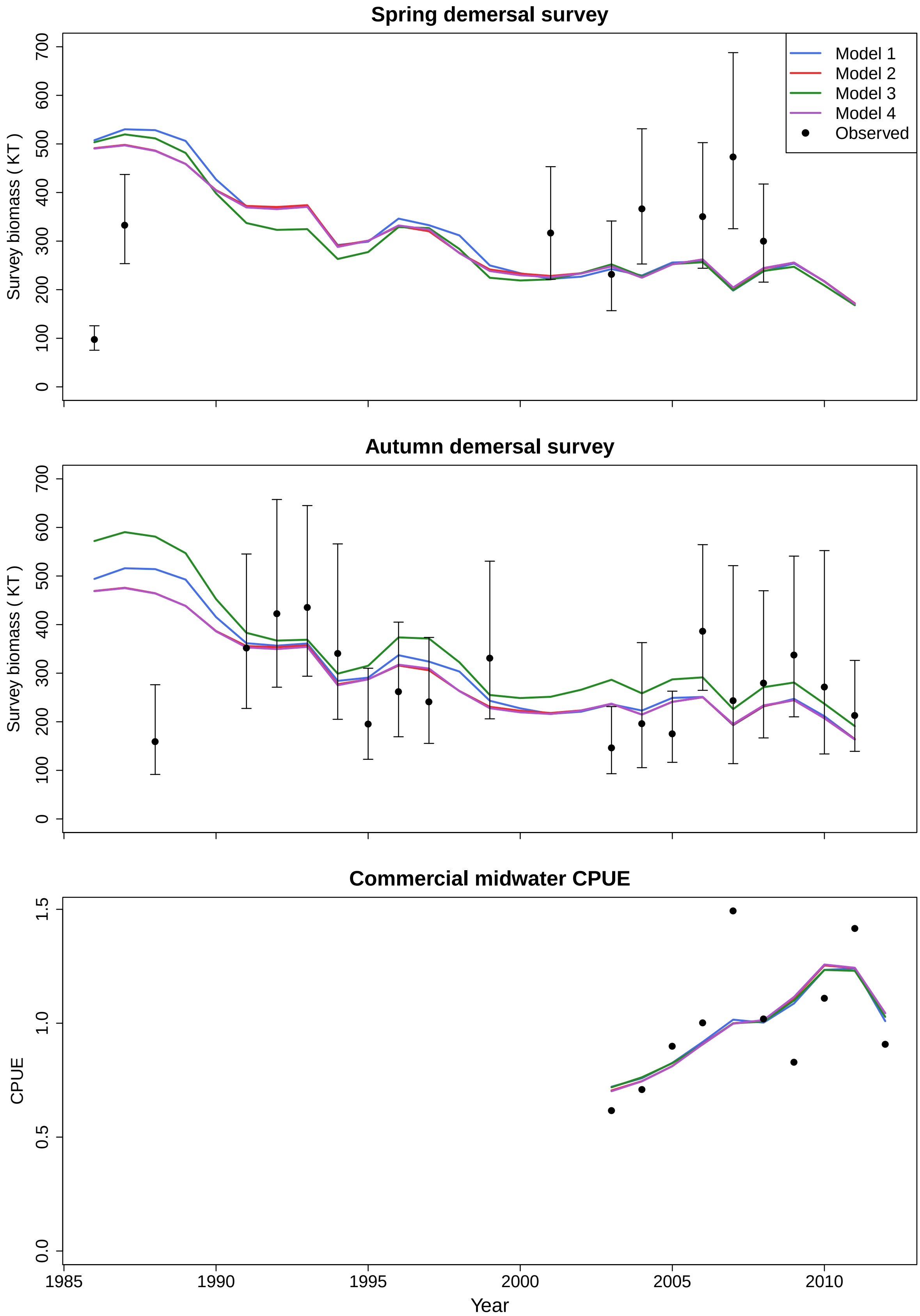

Figure 4.2: Reference Set assessment models fits to the abundance indices. In the top two plots it is difficult to distinguish between the trajectories of Model 2 and Model 4, because they are almost identical. This is true of all of the models in the bottom plot. Units for CPUE are not reported, because the index has been rescaled to have a mean observed value of one. Note that all plots share the same horizontal axis. Error bars for the top two plots show 95% confidence intervals under the assumption of log-normal distributions.

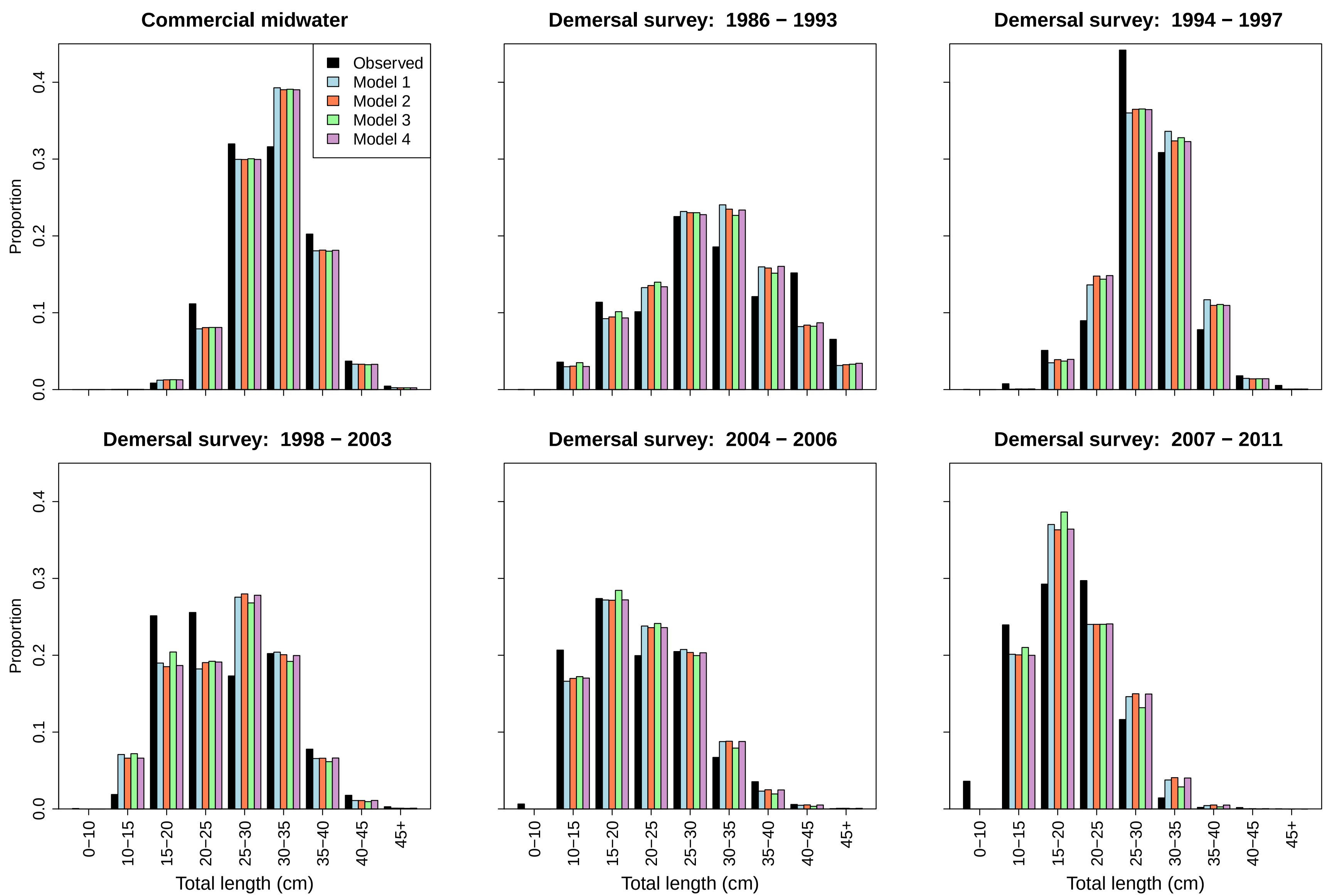

Figure 4.3: Reference Set assessment models fits to catch-at-length data, averaged over years. The demersal data have been separated into different periods according to the different selectivity functions that apply to each.

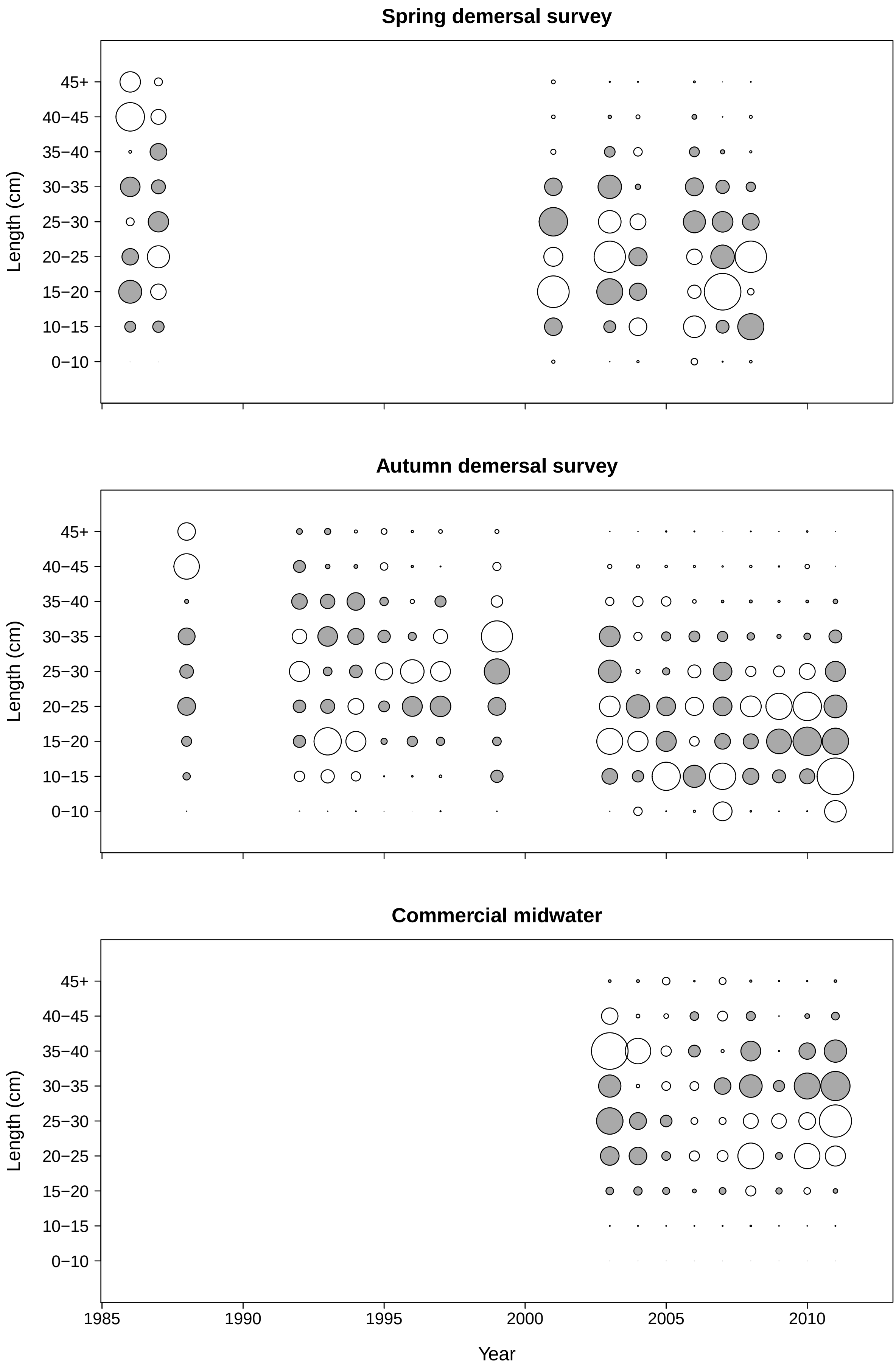

Figure 4.4: Bubble plots of catch-at-length standardised residuals for the Base Case assessment model. The area of a bubble is proportional to the magnitude of the corresponding residual. Positive residuals are represented by white bubbles, while those for negative residuals are grey. Note that all plots share the same horizontal axis.

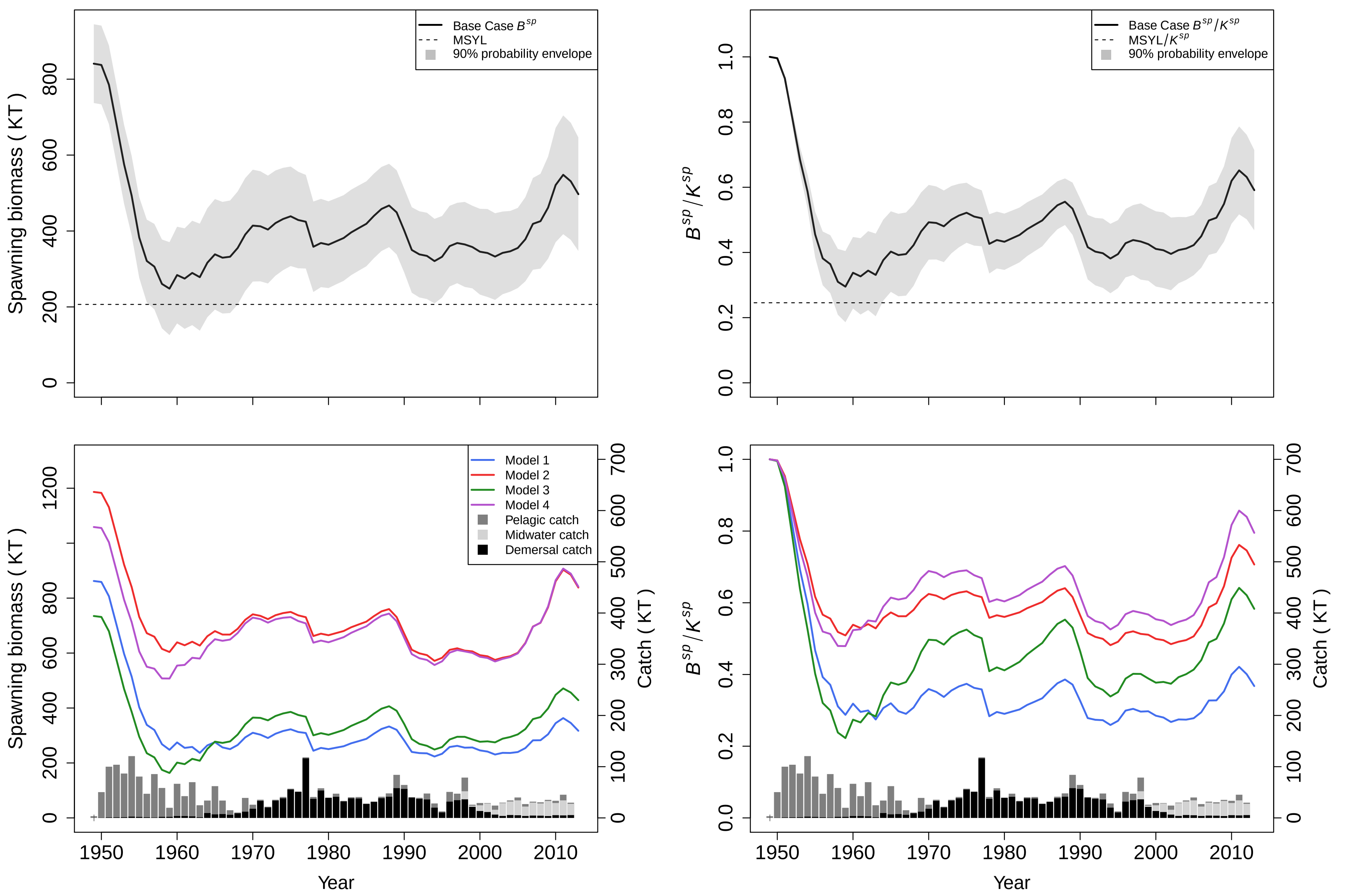

Figure 4.5: Estimated spawning biomass trajectories in absolute terms, and as proportions of their unexploited equilibrium levels. The top plots show the Base Case assessment model's estimated spawning biomass, along with its Hessian-based 90% probability envelopes and estimated MSYL. The bottom plots show the Reference Set assessment models' estimated spawning biomasses, together with observed catches for each fleet.

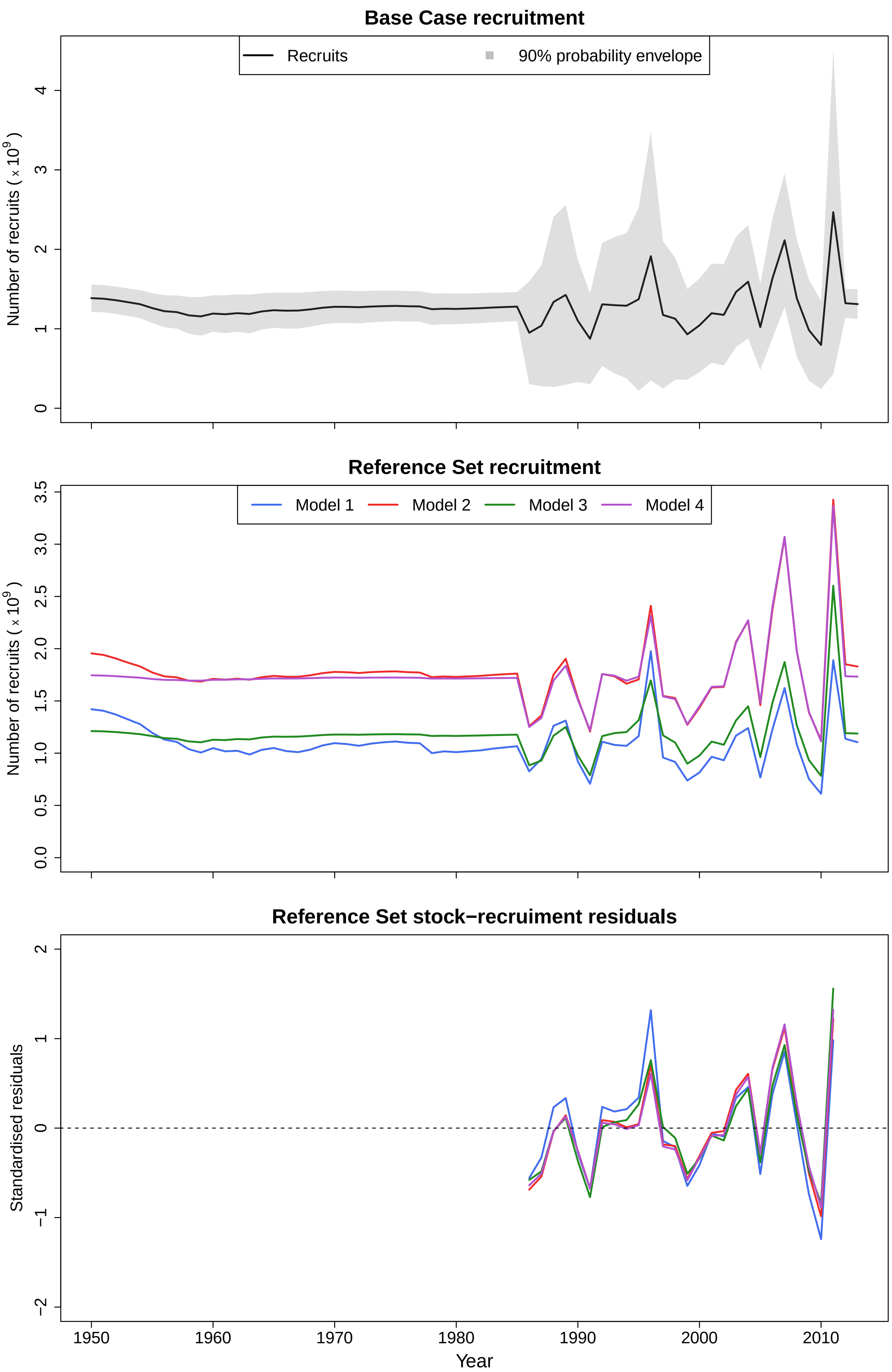

Figure 4.6: The top plot shows estimated recruitments for the Base Case assessment model, together with the Hessian-based 90% probability envelope. The bottom two plots show the Reference Set assessment models' estimated recruitments and stock-recruitment residuals.

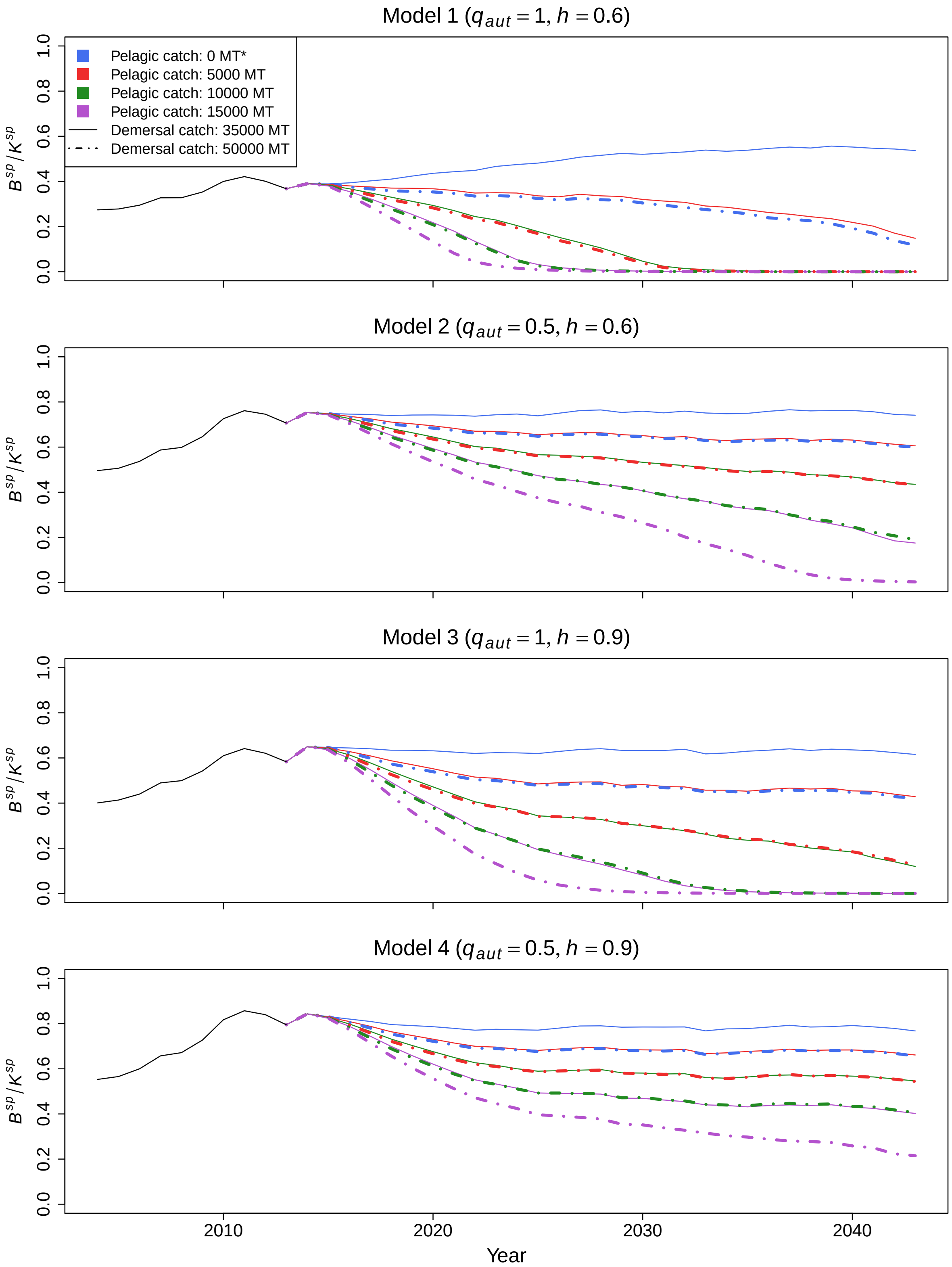

Figure 4.7: Median thirty year spawning biomass projections (shown as proportions of the unexploited equilibrium level) for the constant catch scenarios outlined in Section 4.3. Note that different colours indicate different future pelagic catch scenarios, while a solid or a dot-dashed line indicates different future midwater catch scenarios.

*Note that catches are given in metric tons (MT).

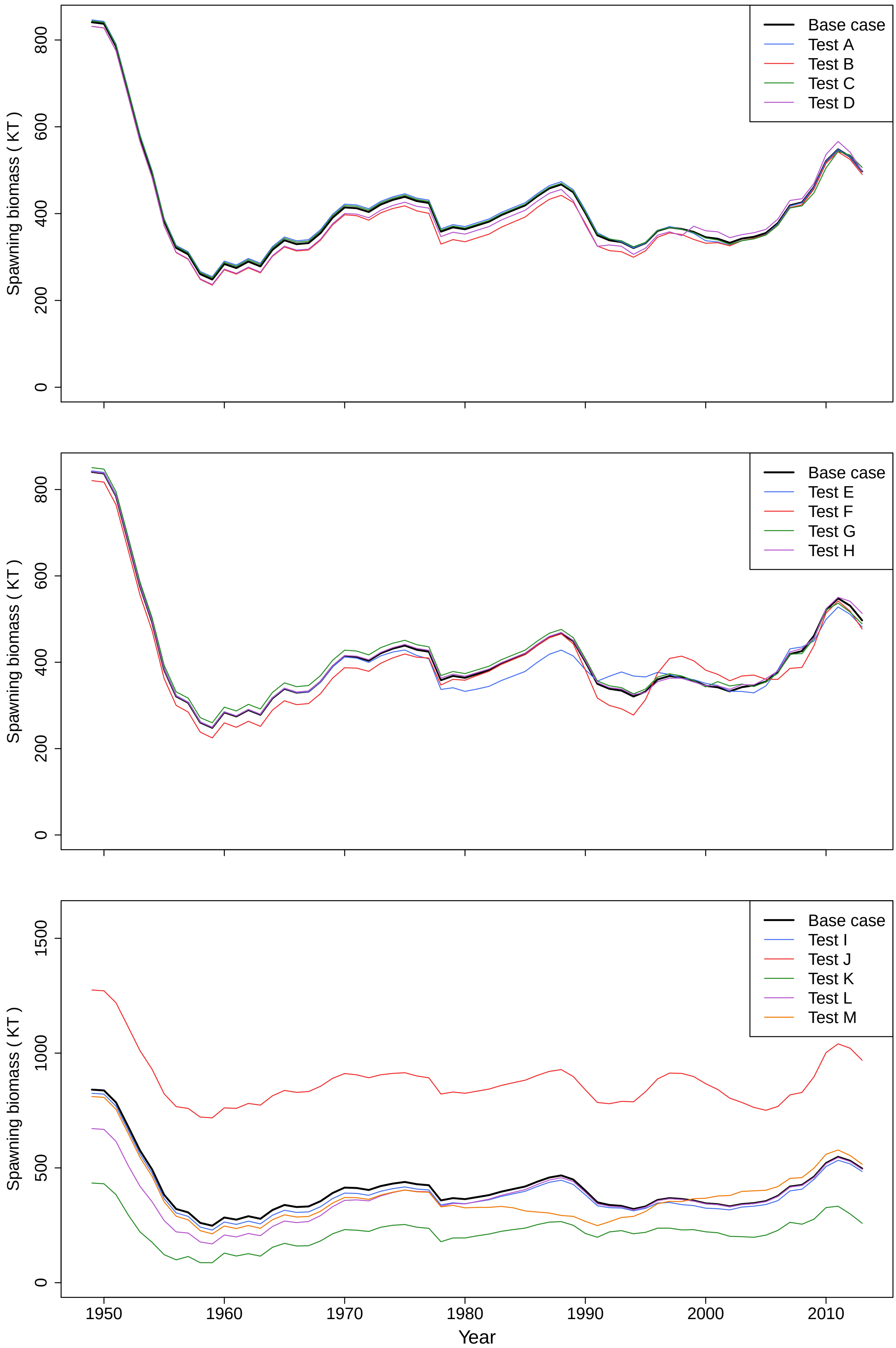

Figure 4.8: Time-series of spawning biomasses (in absolute terms) for the Base Case assessment model, and for all sensitivity tests which are variants of the Base Case.

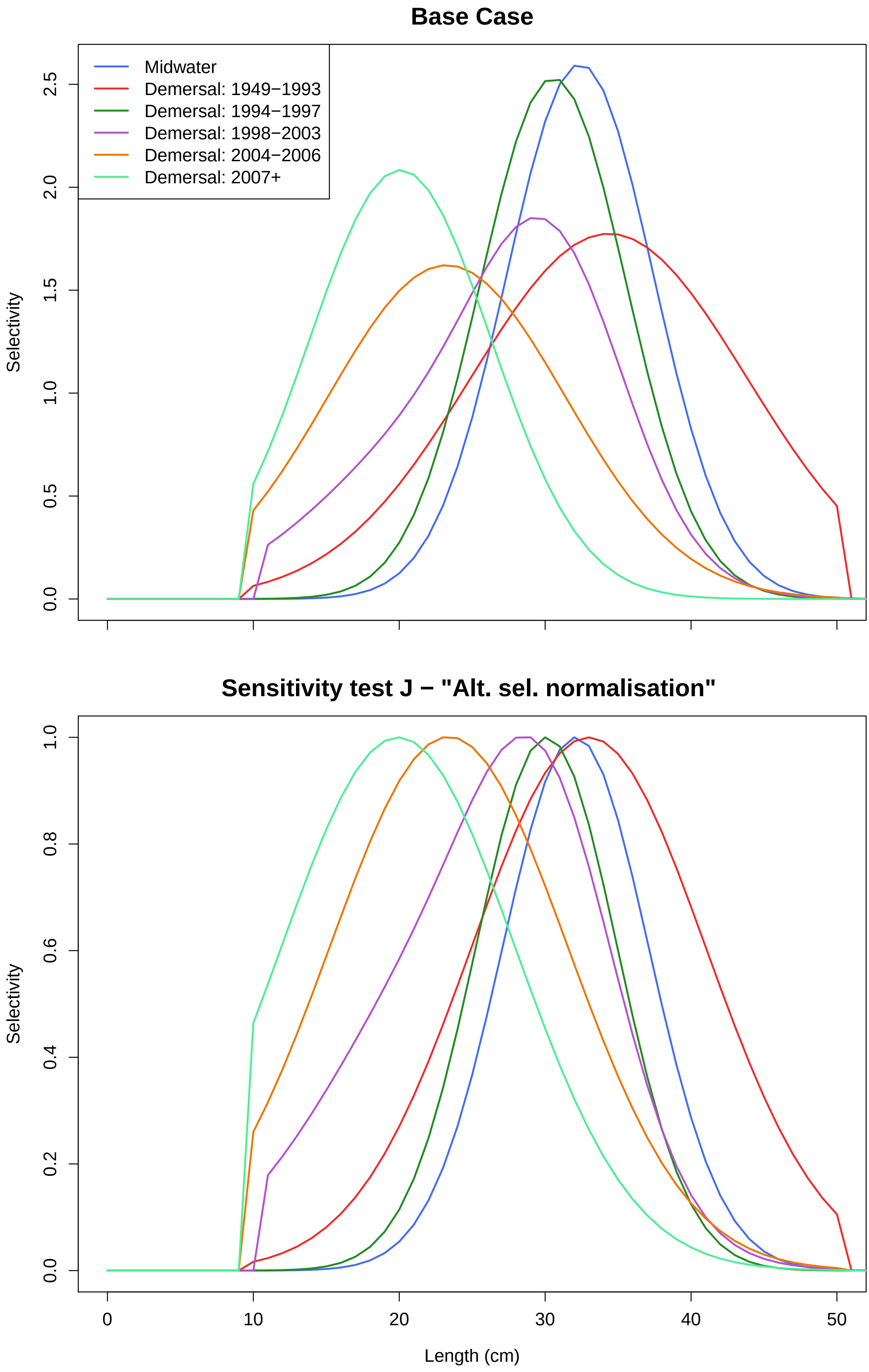

Figure 4.9: Length specific fishing selectivities for the midwater and demersal fleets for the Base Case model and sensitivity test J.

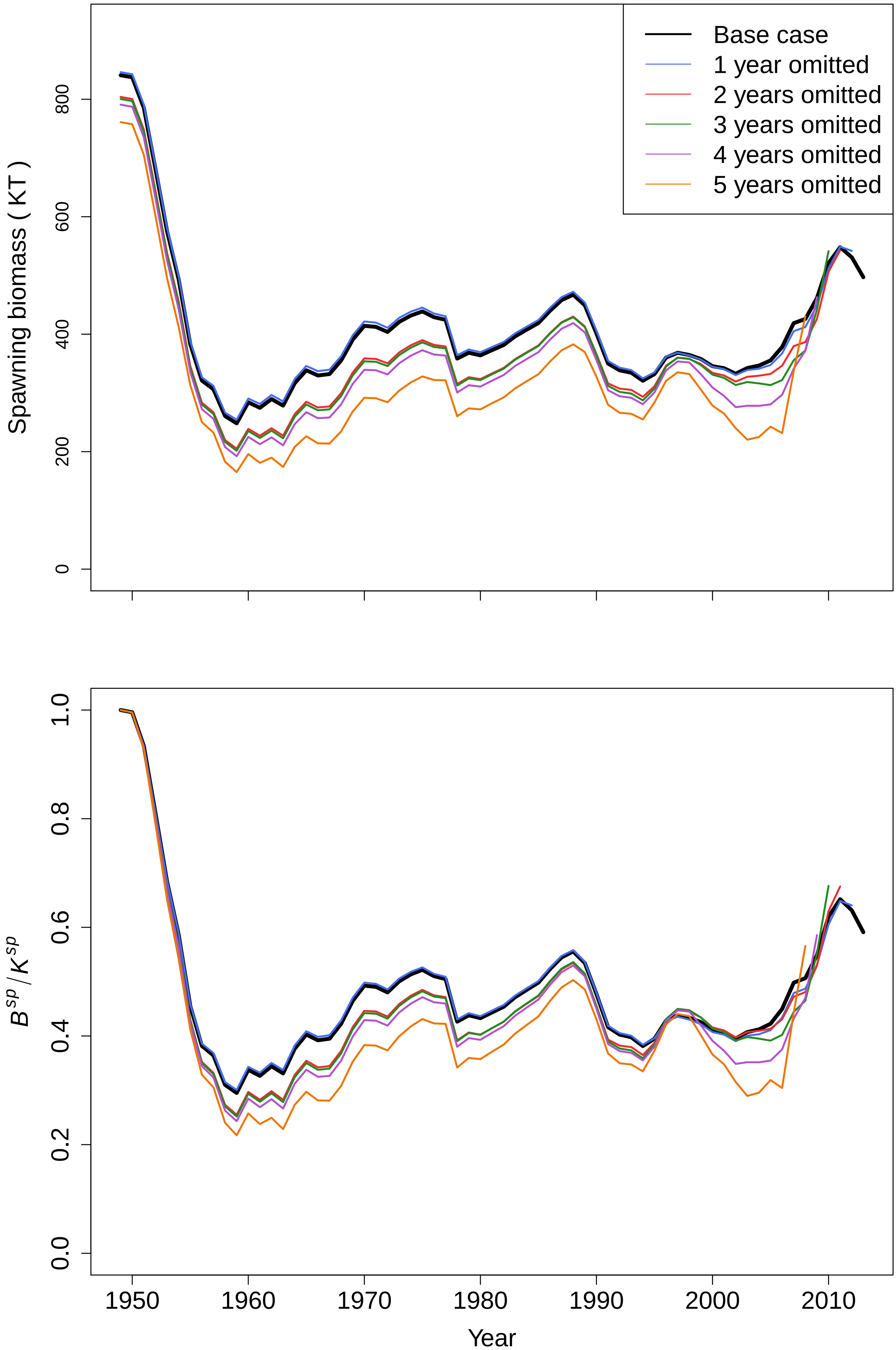

Figure 4.10: Results of retrospective analysis for spawning biomass in absolute and relative terms.

Appendix 4.A Mathematical details of the ASPM

4.A.1 Dynamics

The population dynamics are described by the following equations:

$$N_{y+1,0} = R_{y+1} \tag{4.A.1}$$

$$N_{y+1,a+1} = \left(N_{y,a}e^{-M/2} - C_{y,a}\right)e^{-M/2} \qquad 0 \leq a \leq m-2 \tag{4.A.2}$$

$$N_{y+1,m} = \left(N_{y,m}e^{-M/2} - C_{y,m}\right)e^{-M/2} + \left(N_{y,m-1}e^{-M/2} - C_{y,m-1}\right)e^{-M/2} \tag{4.A.3}$$

where

$N_{y,a}$ is the number of horse mackerel of age a at the start of year y;

$C_{y,a}$ is the total number of horse mackerel of age a taken in year y by the pelagic, midwater and demersal fleets combined;

R_y is the number of recruits (0-year olds) at the start of year y;

M is the natural mortality rate for horse mackerel; and

m is the minimum age within the plus-group and is set here to ten years old.

The approximation of the fishery as a pulse catch in the middle of the season is considered of sufficient accuracy for present purposes. Note that the model also assumes that recruitment to the population occurs at the start of the new year (Equation 4.A.1), even though in reality there are two spawning peaks roughly two months apart.

The total number of horse mackerel of age a caught each year is given by:

$$C_{y,a} = \sum_f C^f_{y,a} \tag{4.A.4}$$

where f indicates the fishery concerned and in this case is either p for pelagic, d for demersal or m for midwater.

The annual catch by mass for fleet f is given by:

$$\begin{aligned} C_y^f &= \sum_{a=0}^{m} w_{y,a}^f C_{y,a}^f \\ &= \sum_{a=0}^{m} w_{y,a}^f S_{y,a}^f F_y^f N_{y,a} e^{-M/2} \end{aligned} \tag{4.A.5}$$

where

$S_{y,a}^f$ is the fishing selectivity-at-age for fleet f for fish of age a in year y;

F_y^f is the fleet-specific fishing mortality for a fully selected age class in year y; and

$w_{y,a}^f$ is the effective weight of a horse mackerel of age a for fleet f in year y .

Fishing selectivity for the pelagic fleet is described by a selectivity-at-age function; therefore, that fleet's effective weight-at-age is simply given by a combination of the length-at-age and weight-at-length relationships discussed in Section 2.2.3:

$$l_a = 54.56 \left[1 - e^{-0.183(a+0.654)}\right] \tag{4.A.6}$$

$$w_a^p = 0.0078 \; l_{a+\frac{1}{2}}^{3.011} \times 10^{-6} \tag{4.A.7}$$

Because the fishing selectivities of the midwater and demersal fleets are modelled by selectivity-at-length functions, their effective weights-at-age must be calculated differently:

$$w_{y,a}^f = \frac{\sum_l w_l S_{y,l}^f A_{l,a}}{\sum_l S_{y,l}^f A_{l,a}} \tag{4.A.8}$$

where

w_l is the weight of a horse mackerel of length l (Equation 2.2);

$S_{y,l}^f$ is the fishing selectivity for fleet f for fish of length l in year y; and

$A_{l,a}$ is an age-length key, which gives the proportion of fish of age a that are of length l (detailed later in Equation 4.A.16).

Note that fishing selectivity for the midwater fleet is assumed to be time-invariant; therefore, the y subscript may be dropped when determining the effective weight-at-age for that fleet.

The fleet-specific exploitable component of abundance is taken to be given by exploitable biomass at midyear:

$$B_y^f = \sum_{a=0}^{m} w_{y,a}^f S_{y,a}^f N_{y,a} e^{-M/2} \tag{4.A.9}$$

or in terms of numbers of individuals:

$$N_y^f = \sum_{a=0}^{m} S_{y,a}^f N_{y,a} e^{-M/2} \tag{4.A.10}$$

The proportion of the resource harvested each year by fleet f is therefore given by:

$$F_y^f = C_y^f / B_y^f \tag{4.A.11}$$

and

$$C_{y,a}^f = S_{y,a}^f F_y^f N_{y,a} e^{-M/2} \tag{4.A.12}$$

Note that in terms of Equations 4.A.11 and 4.A.12 the model assumes the same fishing selectivity for the commercial demersal fleet and both demersal surveys. This simplifying assumption has been made because there are no catch-at-length data available to estimate selectivity functions for the commercial demersal fleet.

Fishing selectivities

Selectivity-at-age for the pelagic fleet is input and assumed to change with time. The same values are used as were used in the previous horse mackerel assessment model, which are reported in Table 4.1 (Johnston and Butterworth, 2007). Essentially, there is one selectivity function for the pre-1963 period and another for the post-1967 period, while for the period between (1963–1967) the average of those two selectivity functions is used.

In contrast, selectivity-at-length is estimated for both the midwater and demersal fleets. These are assumed to have a Gaussian form with length:

$$S_{y,l}^f = \begin{cases} \dfrac{e^{-\left(l-\mu_y^f\right)^2}}{2\left(\lambda_y^f\right)^2}, & \text{if } l_{\min}^f \le l \le l_{\max}^f \\ 0, & \text{otherwise} \end{cases} \tag{4.A.13}$$

where

μ_y^f is an estimated selectivity parameter that determines the centre of the Gaussian for fleet f in year y;

λ_y^f is an estimated selectivity parameter that determines the width of the Gaussian for fleet f in year y;

$l_{\min}^f$ is a fixed selectivity parameter that determines the smallest length class with non-zero selectivity for fleet f, and is set equal to 10 cm for both the demersal or the midwater fleets; and

$l_{\max}^f$ is a fixed selectivity parameter that determines the largest length class with non-zero selectivity for fleet f, and is set equal to 50 cm or 60 cm for the demersal or midwater fleets respectively.

Note again that the y subscript maybe be dropped when dealing with selectivity for the midwater fleet because it is time-invariant. Selectivity-at-length is then normalised according to:

$$S_{y,l}^f \to S_{y,l}^{*f} = S_{y,l}^f \;/ \sum_{l'=l_1}^{l_2} \frac{S_{y,l'}^f}{l_2 - l_1 + 1} \tag{4.A.14}$$

In other words, the selectivity function is scaled by the inverse of its average value over a certain length range. l_1 and l_2 are the same for both the midwater and the demersal fleets, and are set equal to 10 cm and 40 cm respectively.

Because the model is age-structured, selectivity-at-length must be transformed into selectivity-at-age using an age-length relationship:

$$S_{y,a}^f = \sum_l A_{l,a} S_{y,l}^f \tag{4.A.15}$$

It is assumed that the length distribution for horse mackerel of age a is described by a normal distribution with mean which is given by the von Bertalanffy growth curve input, and with a standard deviation that is proportional to this mean. Consequently, with length classes of 1 cm, $A_{l,a}$ is computed according to:

$$A_{l,a} = \tfrac{1}{2}\left[\mathrm{erf}\left(\frac{l + 0.5 - l_{a+0.5}}{\sqrt{2}\,(\gamma\; l_{a+0.5})}\right) - \mathrm{erf}\left(\frac{l - 0.5 - l_{a+0.5}}{\sqrt{2}\,(\gamma\; l_{a+0.5})}\right)\right] \tag{4.A.16}$$

where

erf is the error function;

$l_{a+0.5}$ is the expected midyear length for a horse mackerel of age a, which is calculated using the input von Bertalanffy growth curve given by Equation 2.1; and

γ is the CV of the length-at-age distribution, which is fixed at 0.9.

Stock-recruitment relationship

The spawning biomass in year y is given by:

$$B_y^{sp} = \sum_{a=a_m}^{m} w_a N_{y,a} \tag{4.A.17}$$

where

a_m is the age corresponding to 100% sexual maturity, which is assumed here to be described by a knife-edge function of age; and

w_a is the mass of a horse mackerel of age a at the start of the year.

The number of recruits at the start of fishing year y is related to the spawner stock size by a Beverton-Holt stock-recruitment relationship:

$$R\left(B_y^{sp}\right) = \frac{\alpha B_y^{sp}}{\beta + B_y^{sp}} e^{\varsigma_y} \tag{4.A.18}$$

where

α and β are stock-recruitment parameters; and

ς_y are stock-recruitment residuals reflecting fluctuations about expected recruitment in year y.

In order to work with estimable parameters that are more biologically meaningful than α and β, the stock-recruitment relationship is re-parameterised in terms of the pre-exploitation equilibrium spawning biomass, K^{sp}, and the steepness of the stock-recruitment relationship, h, where steepness is the fraction of pristine recruitment, R_0, that results when spawning biomass drops to 20% if its pristine level:

$$hR_0 = R\left(0.2K^{sp}\right) \tag{4.A.19}$$

from which it follows that:

$$h = \frac{0.2\left(\beta + K^{sp}\right)}{\beta + 0.2K^{sp}} \tag{4.A.20}$$

and hence:

$$\alpha = \frac{4hR_0}{5h-1} \tag{4.A.21}$$

and

$$\beta = \frac{K^{sp}(1-h)}{5h-1} \tag{4.A.22}$$

Given a value for the pre-exploitation spawning biomass K^{sp} of horse mackerel, together with the assumption of an initial equilibrium age-structure, pristine recruitment can be determined from:

$$R_0 = K^{sp} / \left[\sum_{a=a_m}^{m-1} w_a e^{-aM} + w_m e^{-mM} / \left(1 - e^{-M}\right) \right] \tag{4.A.23}$$

Numbers-at-age for subsequent years are then computed by means of Equations 4.A.1–4.A.18.

4.A.2 Likelihood functions

The model is fitted to three biomass indices and three sets of catch-at-length data. Stock recruitment residuals also contribute to the penalised negative log-likelihood that is minimised in the fitting process.

Abundance indices

The assessment model is ordinarily fitted to three abundance indices: spring and autumn demersal survey biomass estimates, and a commercial midwater CPUE series. Additionally, the model is fitted to pelagic hydro-acoustic survey abundance estimates as a sensitivity test. The associated likelihood contributions are calculated by assuming that the observed abundance index is log-normally distributed about its expected value:

$$I_y^s = \hat{I}_y^s e^{\epsilon_y^s} \qquad \text{or} \qquad \epsilon_y^s = \ln\left(I_y^s\right) - \ln\left(\hat{I}_y^s\right) \tag{4.A.24}$$

where

s indicates the abundance index concerned and is either *aut* for the autumn survey, *spr* for the spring survey, *cpue* for CPUE or *pel* for the pelagic index;

I_y^s is the observed value of index s in year y; and

$\hat{I}_y^s$ is the model predicted value of index s in year y.

The negative of the log-likelihood function (after removal of the constant) is then given by:

$$-\ln L = \sum_s \sum_y \left[\ln \sigma_y^s + \left(\epsilon_y^s\right)^2 / 2 \left(\sigma_y^s\right)^2\right] \tag{4.A.25}$$

The spring and autumn demersal survey biomass estimates are assumed to reflect demersal exploitable biomass:

$$\hat{I}_y^s = q_s B_y^d \tag{4.A.26}$$

where q_s is the catchability coefficient corresponding to index s. Note that the same demersal exploitable biomass B_y^d is used to fit both the autumn and spring demersal surveys even though they occur several months apart. Because a mid-year pulse catch assumption is made (Equation 4.A.9), this exploitable biomass does not account for fishing mortality that may occur between the two surveys. For these series, reliable estimates of sampling variability and additional variance are available; therefore, the standard deviations are calculated according to the following formula:

$$\sigma_y^s = \sqrt{\ln\left[1 + \left(CV_y^s\right)^2\right] + \left(\sigma_{add}^s\right)^2} \tag{4.A.27}$$

where

CV_y^s is the CV for survey s in year y, which is given in Table 3.3; and

σ_{add}^s is the model estimated additional variance for survey abundance index s.

The midwater CPUE index is assumed to reflect the midwater exploitable biomass:

$$\hat{I}_y^{cpue} = q_{cpue} B_y^m \tag{4.A.28}$$

and the pelagic hydro-acoustic survey index from November of year y is assumed to reflect recruitment in year $y+1$:

$$\hat{I}_y^{pel} = q_{pel} R_{y+1} \tag{4.A.29}$$

Reliable estimates of CVs and catchability are unavailable for the CPUE and pelagic abundance indices. Therefore, they are set to their maximum likelihood estimates:

$$\sigma^s = \sqrt{1/n \sum_y \left(\epsilon_y^s\right)^2} \tag{4.A.30}$$

$$\ln q_s = 1/n \sum_y \epsilon_y^s \tag{4.A.31}$$

Catch-at-length

Model estimated catch-at-length proportions are fitted to spring and autumn demersal survey length-frequency data, and commercial midwater length-frequency data.

Catch-at-age estimates (Equation 4.A.12) are transformed into catch-at-length estimates using age-length relationship $A_{l,a}$ (Equation 4.A.16):

$$C_{y,l}^f = \sum_{a=0}^{m} A_{l,a} C_{y,a}^f \tag{4.A.32}$$

where $C_{y,l}^f$ is the total number of horse mackerel of length l caught in year y.

The contribution of catch-at-length data to the negative of the log-likelihood function is then given by:

$$-\ln l = w_{cal} \sum_s \sum_y \sum_l \left[\ln \sigma_{cal}^s + \left(\sqrt{p_{y,l}^s} - \sqrt{\hat{p}_{y,l}^s}\right)^2 / 2 \left(\sigma_{cal}^s\right)^2\right] \tag{4.A.33}$$

where

w_{cal} is a weighting for this likelihood contribution, and is fixed at 0.35;

$p_{y,l}^s$ is the observed proportion of fish caught in year y that are of length l for dataset s;

$\hat{p}_{y,l}^s$ is equal to $C_{y,l}^f / \sum_l C_{y,l}^f$ and is the model predicted proportion of fish caught in year y that are of length l in dataset s, where f is the appropriate fleet; and

σ_{cal}^s is the standard deviation associated with catch-at-length dataset s, which is estimated in the fitting procedure by:

$$\sigma_{cal}^s = \sqrt{\sum_y \sum_l \left(\sqrt{p_{y,l}^s} - \sqrt{\hat{p}_{y,l}^s}\right)^2 / \sum_y \sum_l 1} \tag{4.A.34}$$

Note that allowance is made for a minus group (fish smaller than 10 cm) and a plus group (fish 46 cm and larger). Length classes are specified with intervals of 5 cm.

Stock-recruitment residuals

It is assumed that these residuals are log-normally distributed and are not serially correlated. Therefore, their contribution to the penalised negative log-likelihood is given by:

$$-\ln L = \sum_y \frac{\varsigma_y^2}{2\sigma_R^2} \tag{4.A.35}$$

where

ς_y is the estimated stock-recruitment residual for year y; and

σ_R is the input standard deviation of the log-residuals, which is assumed to be equal to 0.5.

Appendix 4.B Midwater and demersal selectivities

The previous Cape horse mackerel assessment model by Johnston and Butterworth (2007) did not incorporate length- or age-frequency data, whereas the model described in this work is conditioned on catch-at-length data from the directed midwater fishery (Section 3.4) and from bi-annual demersal surveys (Section 3.2). It is, therefore, not unexpected that the fixed selectivity-at-age curves used for both the midwater and demersal fleets in Johnston and Butterworth (2007) provide poor fits to the catch-at-length data. Therefore, these data are used instead to estimate separate selectivity functions for those fleets.

Because catch-at-length data are available, it is preferable for a number of reasons to estimate selectivity-at-length curves, instead of selectivity-at-age. First, fishing selectivity generally is a function of the size of a fish (e.g. the mesh size of a trawl net or the swimming speed of a fish on which its ability to escape the netting depends). Secondly, information is not lost by grouping the narrow length classes into broad age classes. Finally, it is simpler to visually inspect selectivity-at-length curves and understand how they impact predicted catch-at-length.

Many different functional forms were tested and all suggested that the selectivities should be dome-shaped. Gaussian functions with model estimated centres and widths were chosen because they fit the data well and reflect fairly standard usage for this purpose.

4.B.1 Time-varying selectivity

Figure 4.B.1 is a bubble plot of the catch-at-length standardised residuals for the Base Case model, but without time-varying demersal selectivity. The midwater residuals are not too problematic; however there is clear evidence of systematic effects in the residuals of the spring and autumn surveys, most notably for years before 1990 and for lengths greater than twenty-five centimetres in the most recent surveys.

The manner in which the observed demersal catches-at-length proportions (Figure 4.3), along with the systematic patterns in its residuals (Figure 4.B.1), vary with time suggests that there have been sudden changes in the availability of various sizes of horse mackerel. Therefore, in an attempt to remove the systematic errors, the demersal selectivity function was modified to vary with time. A different selectivity curve is estimated for each of four distinct periods in the history of the horse mackerel fishery. These periods are 1949–1993, 1994–1997, 2004–2006 and 2007+. For 1998–2003, the mean of the 1994–1997 and 2004–2006 selectivities is used, in order to reduce the number of parameters that are estimated and to force a relatively smooth change in selectivity over time. Comparison of Figure 4.B.1 to Figure 4.4, shows the improvement in the catch-at-length residuals that the introduction of time-varying selectivity provides.

4.B.2 Normalisation

Figure 4.B.2 compares predicted demersal exploitable biomass for the Base Case model without selectivity normalisation, to that of the Base Case model with selectivity normalisation. It shows that when selectivity is not normalised, a rapid drop in demersal exploitable biomass by more than 20% of its unexploited equilibrium level is predicted for 1994. This decrease is not due to an underlying reduction in abundance, rather it is caused by a change in demersal selectivity. This neatly illustrates the difficulties that time-varying selectivity can introduce. Since catchability and selectivity-at-length S_l are confounded, conventionally with time-independent S_l the largest of the S_l's is set to one to remove ambiguity. Thus interpretation problems arise if S_l changes over time, as it is not immediately clear how S_l should then be renormalised so that the catchability, which relates exploitable biomass to the abundance index, can be assumed to remain constant over time.

A fairly standard approach is adopted of scaling selectivity by the inverse of its mean over a pre-specified length range. 10–40 cm is chosen here, because it is clear from Figure 4.3 that the bulk of horse mackerel caught by demersal surveys fall within this range, even given the changes in length distribution with time. Figure 4.B.2 indicates that this normalisation method resolves the sudden drop in exploitable biomass. Additionally, a comparison of observed catch-at-length data to the values estimated by the Base Case model with normalisation and without (Figure 4.B.3), shows that normalising time-varying selectivity results in a somewhat better fit.

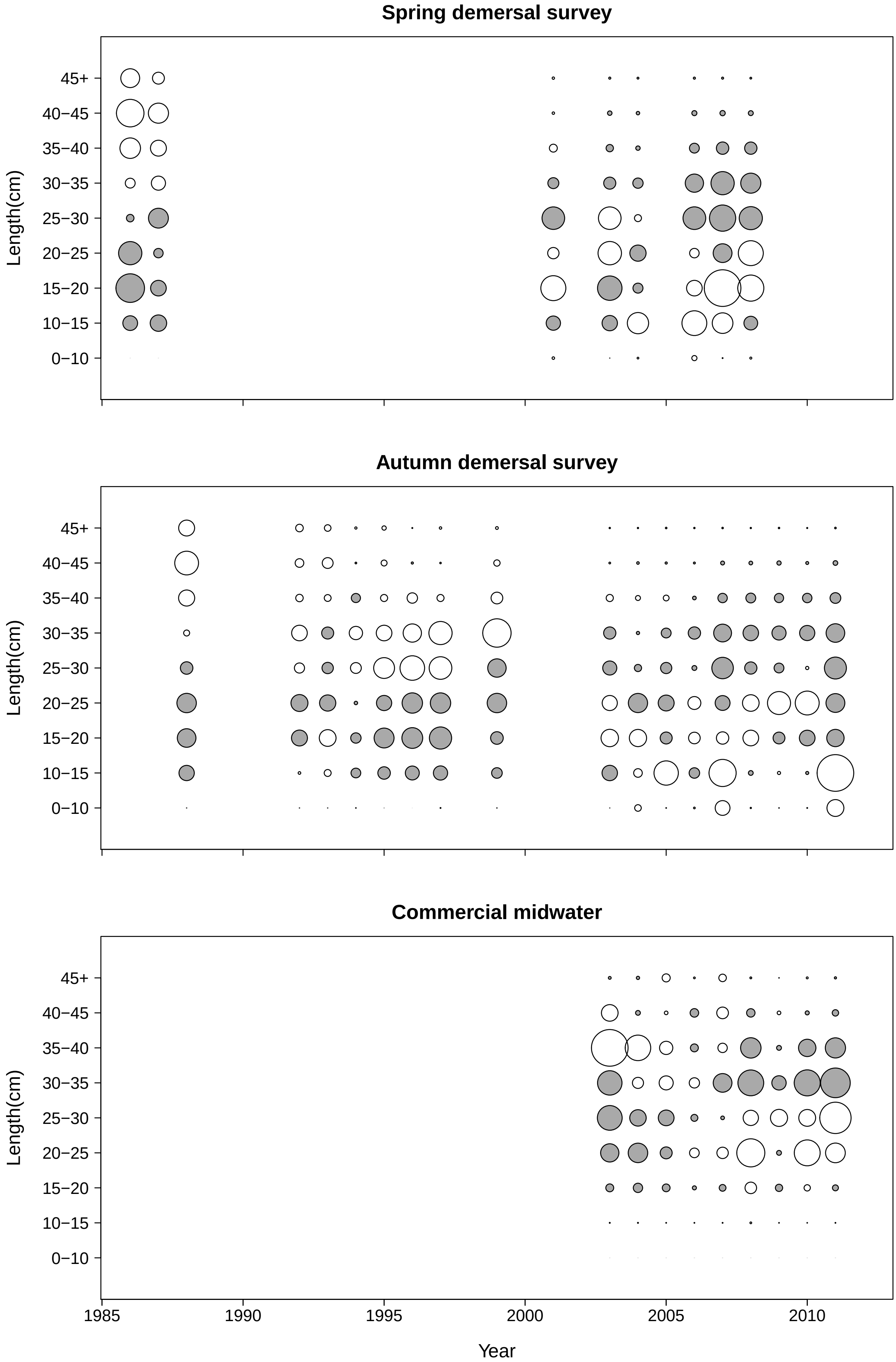

Figure 4.B.1: Catch-at-length residuals for the Base Case model, but without time-varying demersal selectivity. The area of a bubble is proportional to the magnitude of the residual. Positive residuals are represented by white bubbles, while those for negative residuals are grey. Compare to Figure 4.4.

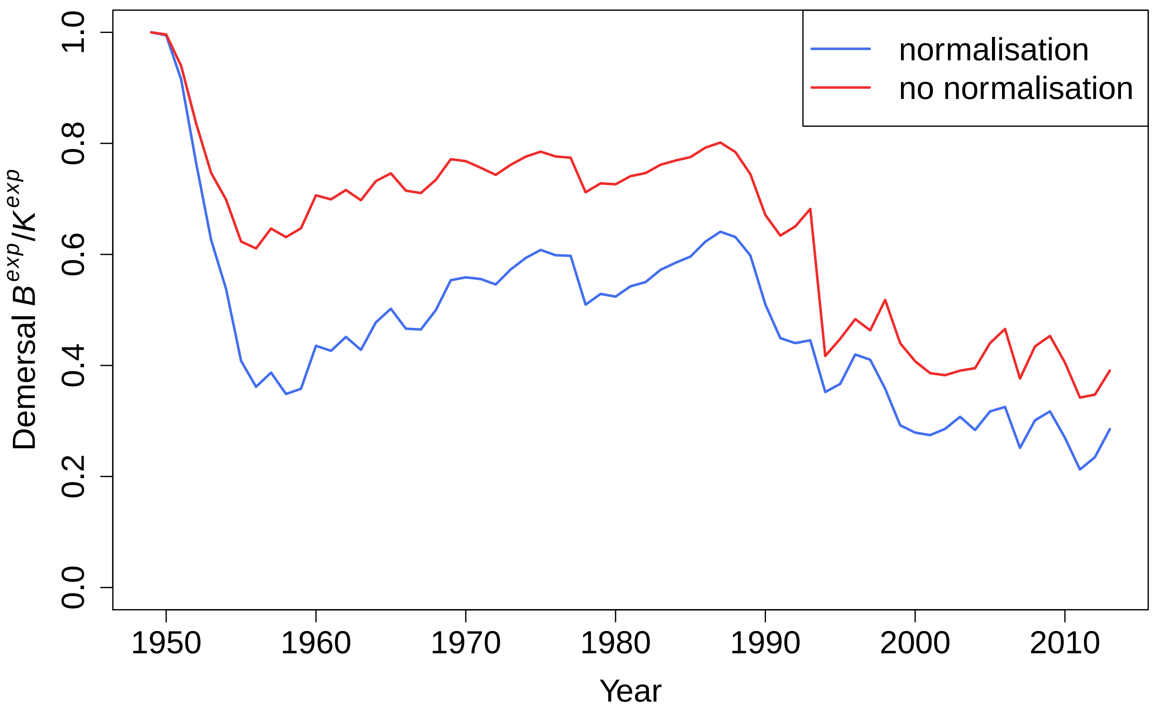

Figure 4.B.2: Comparison of estimated exploitable demersal biomass as a proportion of its unexploited equilibrium level for the Base Case model with demersal selectivity-at-length normalisation, and without demersal selectivity-at-length normalisation.

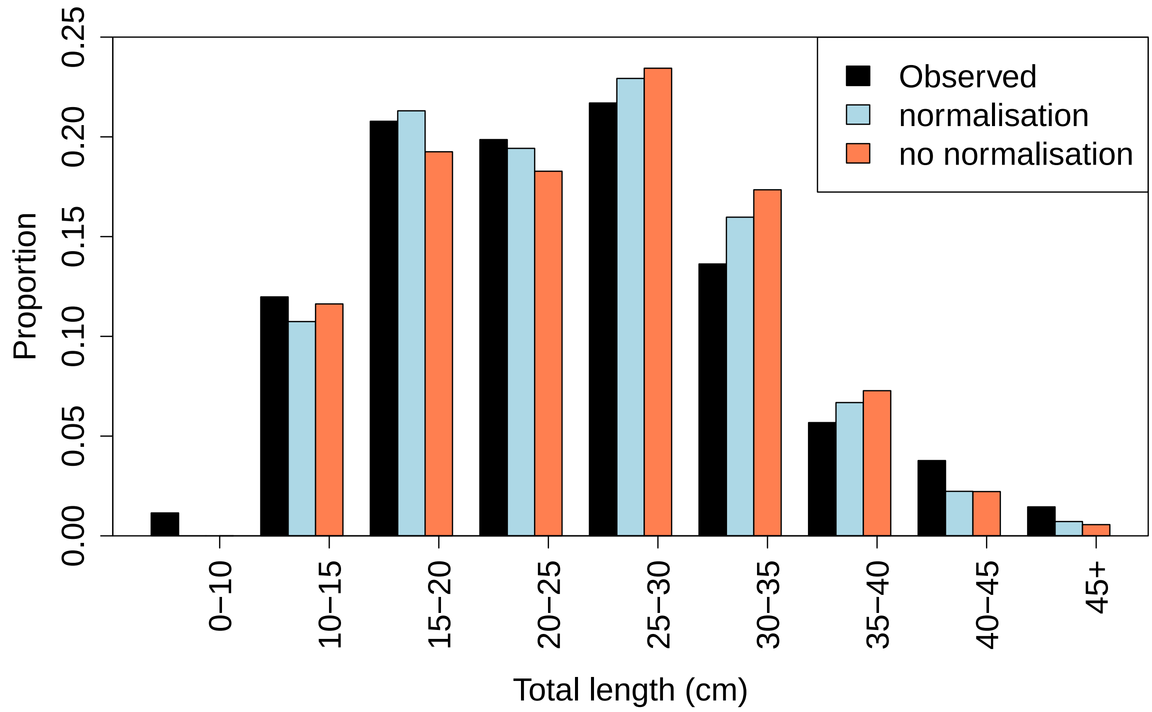

Figure 4.B.3: Comparison of observed catch-at-length data to estimates from the Base Case model with demersal selectivity-at-length normalisation, and without demersal selectivity-at-length normalisation. Values are averaged over the years for which such data are available.

Chapter 5

Precautionary Upper Catch Limit for the pelagic fishery

In Chapter 1 it was noted that juvenile horse mackerel are taken as bycatch in the pelagic purse-seine fishery, which targets sardine and anchovy. Since 2000, these bycatches have been limited by a PUCL of 5 000 tonnes per annum. However, this PUCL is problematic as occasionally juvenile horse mackerel are highly abundant or catchable and, as a result, the pelagic fishery is disrupted through having to avoid areas with high horse mackerel bycatch. This can lead to a substantial loss of sardine and anchovy catch. Therefore, the goal is to develop a dynamic PUCL rule that enables the pelagic fishery to continue unhindered during years of high juvenile horse mackerel abundance, without unduly decreasing directed midwater horse mackerel catches or increasing the risk of depletion of the horse mackerel resource.

The first section of this chapter describes the evaluation of pelagic hydro-acoustic surveys as potential predictors of horse mackerel recruitment. If they are found to be reliable, then these survey results could be useful in determining future PUCLs. Subsequent sections outline the set of candidate PUCL rules and the method used to evaluate their performance. Finally, the chapter closes with a discussion of the results and the associated recommendations.

5.1 Pelagic surveys as predictors of recruitment

As juveniles, horse mackerel tend to school with other pelagic species, and annual estimates of juvenile horse mackerel biomass are routinely calculated from pelagic hydro-acoustic surveys. Therefore, it has been suggested that these survey estimates of horse mackerel biomass could potentially serve as predictors of recruitment. If this is indeed the case, then the associated survey index could be used to develop an adaptive rule that adjusts the PUCL annually before an increased density of juvenile horse mackerel unnecessarily limits the pelagic fishery. At the very least, the index could provide a defensible basis for making *ad-hoc* increases like the one which was enacted in 2011. In that year, half of the PUCL had already been taken before the major fishing effort had started. An upwards adjustment of 5 000 tonnes was therefore made to the PUCL for that season only, and was followed by an additional increase of 2 000 tonnes later in the year (Coetzee, 2011).

Pelagic hydro-acoustic surveys are conducted biannually: in May and in November. However, the May surveys occur too early in the year to reliably predict the number of recruits for the following fishing season, which starts in January. Thus, the remainder of this section addresses an evaluation of the November pelagic hydro-acoustic survey biomass index as a predictor of horse mackerel recruitment. However, note that even the November surveys will miss recruits from the second spawning peak in late November on the eastern Agulhas Bank and in February on the western Bank (Section 2.2.2), which will affect the reliability of the survey as a predictor of recruitment.

The details of the history and methodology of the pelagic surveys have been given in Section 3.3, while the historical abundance estimates are reported in Table 3.3. Although it is expected that the index for the West Coast will prove to be a better predictor of juvenile abundance than the index for the entire assessment area (the West Coast has a high proportion of juvenile horse mackerel), both indices are evaluated. The 2010 biomass estimate for the West Coast (51 980 tonnes) is more than three standard deviations from that index's mean, while the same is true for the 2000 biomass estimate for the entire assessment area (196 060 tonnes). This suggests that both of those results should be treated as outliers. Consequently, because it is suspected that the pelagic abundance indices for the West Coast and the entire assessment area each have an outlier, the evaluations for these indices will be conducted both with and without their associated outlying data point.

To test the reliability of an index as a predictor of recruitment, the correlation between survey biomass estimates for November of year y and model-estimated recruitment for year $y + 1$ is measured via the Pearson correlation coefficient. This is done separately for each model variant in the Reference Set. Table 5.1 shows the resulting correlation coefficients. Figure 5.1 compares the normalised survey biomass estimates to estimated recruitment for all models in the

Reference Set. Figure 5.2 is a scatter-plot of the biomass estimates against recruitment, along with the corresponding best-fit linear regressions. Table 5.1 indicates that across all model variants there is a very strong correlation between model-estimated recruitment and pelagic survey results for the West Coast if the outlier is included; however, if the outlier is omitted, then the correlation is much weaker. This situation is reversed for the biomass estimates for the entire area, which show a reasonably strong correlation with recruitment only if the outlier is omitted. As was expected, the West Coast survey index generally shows a stronger correlation than that for the entire survey area. It is evident from Figure 5.1 that the pelagic biomass estimates show much greater variability than do the assessment results for recruitment. Some such damping effect is to be expected, as the length distribution data will tend to smooth out evidence for different cohort sizes in the assessments. Overall, the fairly strong correlations are promising and suggest that the pelagic survey biomass estimates for the West Coast could provide reliable predictors of juvenile horse mackerel abundance. Furthermore, it may be worthwhile to develop an adaptive PUCL rule that uses these survey results as inputs. However, the high sensitivity of the correlation coefficients to the inclusion of the outlying data points is a cause for concern and should be investigated further.

5.2 Candidate PUCL rules

An advantage of the MSE approach is that of a wide variety of MPs can be compared objectively. Several candidate PUCL rules for the pelagic bycatch fishery are therefore proposed. In the context of this evaluation, a desirable PUCL rule will have the following features.

Flexibility - The primary motivation for developing a new PUCL rule is that the old fixed PUCL of 5 000 tonnes proved insufficient during periods of high juvenile horse mackerel abundance. Therefore, the new PUCL rule should allow for larger bycatches of horse mackerel when justifiable, in order to prevent the early closure of the targeted pelagic fisheries.

Precautionary - The new PUCL rule should have a mechanism that provides security against the increased risk of stock depletion caused by the potentially larger bycatches.

Simplicity - It is preferable that the new PUCL rule is not overly-complicated, so that lay stake-holders can be involved in the formulation process and be in a position eventually to support whatever final decisions are made.

In Section 5.1 it was shown that the pelagic hydro-acoustic surveys can potentially predict the strength of horse mackerel recruitment, and thus determine when the PUCL can be increased without an unacceptable risk of resource depletion. However, the supposed reliability of this predictor depends on whether or not outliers are included in the survey index. Therefore, in the interests of better exploring the solution space, candidate PUCL rules that use these survey results as inputs are evaluated, as well as those that are not based on surveys.

5.2.1 Fixed PUCL

This PUCL rule limits the annual pelagic bycatch to a unchanging amount:

$$PUCL_{y+1} = PUCL_{fix} \tag{5.1}$$

where

$PUCL_{y+1}$ is the new PUCL for year $y + 1$; and

$PUCL_{fix}$ is the constant mass of horse mackerel that may be taken as bycatch in any single year.

The *fixed* PUCL rule is not viable for the fishery, because it does not allow for flexibility in bycatches during periods of high juvenile horse mackerel abundance. However, being the simplest possible rule, it is included here to provide a baseline against which the more sophisticated PUCL rules described below can be compared. This rule also represents the *status quo*—historically the fishery has been managed by a fixed annual PUCL.

5.2.2 Survey-based PUCL rules

These PUCL rules use a horse mackerel recruitment index based on pelagic survey results as input. This index is calculated according to:

$$I_y = \left(I_y^{pel} + I_{y-1}^{pel}\right)/2 \tag{5.2}$$

where

I_y is the recruitment index that is used as an input in all survey-based PUCL rules; and

I_y^{pel} is the observed abundance estimate for juvenile horse mackerel based on November pelagic hydro-acoustic surveys of the West Coast.

The average of the survey abundance estimates from the previous two years is used, in order that the calculated recruitment index is less sensitive to observation error in the abundance indices. The pelagic survey results are clearly somewhat erratic (Figure 5.1), so that if the recruitment index was instead defined to be the latest abundance estimates (i.e. $I_y = I_y^{pel}$) it would be more sensitive to such noise.

Next, two candidate PUCL rules that are both functions of I_y are considered.

1) Step

This rule allows for large fixed bycatches during years with strong predicted recruitment, and smaller fixed bycatches otherwise:

$$PUCL_{y+1} = \begin{cases} PUCL_{min} & \text{if } I_y \leq I_{step} \\ PUCL_{max} & \text{if } I_y > I_{step} \end{cases} \tag{5.3}$$

2) Piecewise linear

This rule increases the PUCL linearly from a minimum to a maximum value depending on the recruitment index:

$$PUCL_{y+1} = \begin{cases} PUCL_{min} & \text{if } I_y \leq I_{min} \\ PUCL_{min} + \frac{PUCL_{max} - PUCL_{min}}{I_{max} - I_{min}} \left(I_y - I_{min}\right) & \text{if } I_{min} < I_y \leq I_{max} \\ PUCL_{max} & \text{if } I_y > I_{max} \end{cases} \tag{5.4}$$

where

$PUCL_{max}$ = 15 000 tonnes is the largest possible PUCL, and was determined through consultation with stake-holders during Scientific Working Group (SWG) meetings;

$PUCL_{min}$ is a parameter that determines the smallest PUCL that may be set;

I_{step} is a parameter of the step function that determines the value of I_y at which the PUCL changes;

I_{min} is a parameter of the piecewise linear function that determine the value of I_y at which the PUCL reaches its minimum; and

I_{max} is a parameter of the piecewise linear function that determine the value of I_y at which the PUCL reaches its maximum.

The *piecewise linear* rule will be more flexible than the *step* rule, because it is able to provide variable increases to the bycatch limit depending on the survey results. However, the *step* rule is simpler (though has the potential problem of arguments developing on which option to take if I_y turns out to be very close to I_{step}).

5.2.3 PUCL$_3$ rules

The $PUCL_3$ rules limit the bycatch over any three year period to a fixed amount, in other words:

$$PUCL_{y+1} = PUCL_3 - bycatch_y - bycatch_{y-1} \tag{5.5}$$

where

$PUCL_3$ is the fixed, total amount that may be caught over a three year period; and

$bycatch_y$ refers to the actual horse mackerel bycatch that was taken by the pelagic fishery in year y.

These simple PUCL rules should allow for flexibility in annual allocations, without relying on indices of horse mackerel recruitment.

However, there is a potential, although unlikely, drawback. If an unusually high bycatch follows two years of low bycatch, then the PUCL for the subsequent two years would also be low. This issue can be resolved by additionally limiting the PUCL in any single year. A rule that restricts the PUCL to one half of $PUCL_3$ is therefore also evaluated:

$$PUCL_{y+1} = \min\left[PUCL_3 - bycatch_y - bycatch_{y-1}\ ;\ \frac{1}{2}PUCL_3\right] \tag{5.6}$$

The $PUCL_3$ rule with this additional condition is referred to as $\boldsymbol{PUCL_3}$ ***reserve***, while the rule without this condition is called $\boldsymbol{PUCL_3}$ ***no reserve***.

5.3 Method

In terms of the MSE approach, the performance of a CMP needs to be evaluated by simulation. Section 4.3 describes the general projection methodology for the assessment model; however, a few modifications must be made in order to test the candidate PUCL rules. First, the resource is projected only 10 years into the future. This relatively brief period was chosen because it can be argued that research into the bias in the demersal survey abundance estimates—possibly allowing for less conservative management of the resource—will certainly be completed in the near future. Second, instead of assuming that future pelagic PUCLs remain constant at either 0, 5 000, 10 000 or 15 000 tonnes per annum, they are automatically determined by the candidate rule being assessed. Third, it is no longer assumed that the pelagic bycatch in a given year will be equal to the PUCL. After all, a PUCL differs from a TAC in that it does not specify how much is expected to be taken from a targeted stock, but rather defines the upper catch limit for a resource component that would ideally not be exploited at all. Section 5.3.1 details how the pelagic bycatches are modelled. Finally, future catches by the midwater fleet are assumed to remain constant at 38 115 tonnes per annum, which is the allocation to this fleet for the 2014 fishing season.

Once the projections have been computed, each candidate PUCL rule is assessed in terms of performance statistics that have been developed in collaboration with stake-holders. Section 5.3.2 describes these statistics. Varying the parameters of a PUCL rule generally involves changing the trade-offs between the performance statistics for conflicting objectives. For example, an increase in the projected mean annual catch will result in a decrease in the projected resource biomass. Therefore, in order to provide a consistent basis for comparison, the parameters of all of the PUCL rules are tuned to give the same risk-related performance for the pessimistic model (Model 1). This model is selected because it the most demanding in terms of achieving acceptable risk. The target risk is defined to be the same that was deemed acceptable in relation to the 2007 horse mackerel assessment (Johnston and Butterworth, 2007). Specifically, the target *risk of depletion* is defined as no more than a 5% chance that spawning biomass is depleted to lower than 5.6% of its pristine level during the next 10 years. This corresponds to maintaining the same lower fifth percentile for spawning biomass after 10 years as results under pelagic bycatches of 5 000 tonnes per annum for the most pessimistic OM. The parameters of the PUCL rules are automatically tuned with the help of an optimisation routine (Section 5.3.3).

5.3.1 Modelling undercatch

Figure 5.3 compares the historical pelagic bycatches to PUCLs from 2000 until 2012. It is clear that annual allocations have often not been taken fully by industry, and have even been exceeded on occasion (though it is assumed that in the future real-time monitoring of the fishery will prevent these overcatches). Therefore, it is important to model the extent of these future undercatches when evaluating MPs, otherwise the calculations may assume that unrealistically large bycatches are taken. The assumption is made that in the absence of a PUCL, future bycatches would vary as they did during the period 1968–1999, which is prior to the implementation of the PUCL in 2000. The year 1968 is chosen as a starting point, because by that year "anchovy" nets with a smaller mesh size had been fully introduced in the pelagic purse-seine industry due to the collapse of the sardine stock. Similar nets are still used to this day. Given the size of juvenile horse mackerel, this would likely have had an appreciable effect on the extent of horse mackerel bycatches in the pelagic fishery (Johnston *et al.*, 2004). By analysing data for this period, it was hoped that a relationship could be found between bycatches and some quantity, which would enable the model to predict future bycatches.

However, relatively poor correlations with the biomass assumed to be available to the pelagic fishery ($r \approx 0.12$) and to the assessed recruitment ($r \approx 0.3$) resulted. While it might seem that the relationship between recruitment and bycatch is sufficiently strong for this to be used, Figure 5.4 indicates that the strength of their correlation is largely due to the outlying recruitment estimate for 1996. If this data-point is ignored, then their correlation coefficient drops to approximately 0.15.

Instead therefore, future bycatches are generated by randomly drawing them with replacement from the series of bycatches from the period 1968–1999. However, it is difficult to determine how confident one can be that future bycatches will reflect the past distribution. Therefore, for simulation purposes, all future pelagic bycatches are multiplied by a scaling factor:

$$draw_y = k\ draw_y^* \tag{5.7}$$

where

$draw_y^*$ is the bycatch for year y drawn from the past distribution;

$draw_y$ is the scaled bycatch for year y; and

k is the constant bycatch scaling factor.

The parameter k provides a means of specifying the extent to which the future bycatches might, on average, be larger than the past bycatches. However, bycatches must never exceed the PUCL. In other words:

$$bycatch_y = \begin{cases} draw_y, & \text{if } draw_y \leq PUCL_y \\ PUCL_y, & \text{if } draw_y > PUCL_y \end{cases} \tag{5.8}$$

where

bycatch_y is the bycatch for year y that is actually used in the simulation; and

PUCL_y is the PUCL for year y.

5.3.2 Performance statistics

Performance statistics provide stake-holders with a means of selecting a PUCL rule that achieves a balance in the trade-off between the conflicting management objectives of lower disruption to the pelagic (mainly anchovy and sardine) industry, higher average bycatches and lower risk of resource depletion. The statistics which were chosen to quantify these objectives are as follows.

Median spawning biomass

This statistic gives an indication of the expected health of the resource after 10 years, and for any one simulation it is given by:

$$B^{sp}_{2023}/K^{sp} \tag{5.9}$$

However, each 10 year projection actually consists of 1 000 separate simulations with alternate realisations of stochastic effects. Therefore, this quantity is in fact a distribution, and the median is used. Additionally, because this statistic is calculated for each of the four OMs in the Reference Set, the mean value across those variants is reported.

Risk of depletion

This statistic reflects the risk that a PUCL rules poses to the resource. It is given by the minimum value during the projection period of the lower fifth percentile of the distribution of spawning biomass relative its pristine level:

$$\min_{y=2014,\ldots,2023}\left(\text{lower } 5^{th} \text{ percentile of } B^{sp}/K^{sp}\right) \tag{5.10}$$

Because the pessimistic model will always show the worst risk-related performance, this quantity is calculated for that model only. Further, recall that this statistic will not vary in the results reported because all PUCL rules are tuned to maintain the same target value of 5.6% for this statistic.

Mean annual catch

This statistic indicates the expected annual pelagic bycatch and for any one simulation is given by:

$$\tfrac{1}{8}\sum_{y=2015}^{2022} C_y^p \tag{5.11}$$

For each OM in the Reference Set the median of the distribution of this statistic is calculated. The mean of those medians is reported. Note that only years after 2014 are considered, because earlier PUCLs have already been decided by DAFF and hence they are not affected by the candidate PUCL rule being evaluated.

Probability of disruption

This statistic reflects the probability for any given year that the pelagic fishery will be closed early due to a PUCL being reached. For any one simulation, it is calculated by:

$$\tfrac{1}{8}\sum_{y=2015}^{2022}\begin{cases}1 & \text{if } PUCL_y = bycatch_y\\ 0 & \text{otherwise}\end{cases} \tag{5.12}$$

Again, the median of the distribution of this statistic is calculated for each OM in Reference Set, and the mean of those medians is reported.

Bootstrapping to estimate precision

Bootstrapping is used to calculate estimates of the precision of the performance statistics reported. Each projection for a model actually consists of 1 000 independent model runs with different realisations of stochastic effects. Results for this first set will not be exact because of

Monte Carlo error. Bootstrapping to estimate the precision of such results involves selecting a new set of 1 000 runs from the original set of 1 000 runs using random sampling with replacement. A new estimate for each performance statistic is then calculated from the new set of runs. This process is then repeated 10 000 times to produce a distribution for each performance statistic from which estimates of precision can be determined.

5.3.3 Tuning and optimisation

It would be difficult and time-consuming to search manually for the optimal parameters of each PUCL rule, while simultaneously ensuring that all give the target *risk of depletion*. Instead, a constrained non-linear optimisation routine was used to maximise the objective function:

$$f(\vec{x}) = C(\vec{x}) \tag{5.13}$$

subject to the non-linear constraint

$$r(\vec{x}) = r_{tar} \tag{5.14}$$

where

$\vec{x}$ is a vector of parameters for the PUCL rule being considered;

C is the performance statistic, *mean annual* (pelagic) *catch*;

r is the performance statistic, *risk of depletion*; and

r_{tar} is the target *risk of depletion*, which has a value of 5.6%.

It may seem counter-intuitive to seek to maximise *mean annual catch* in a bycatch fishery. After all, the adaptive PUCL is intended to lessen disruptions to the pelagic industry during periods of high juvenile horse mackerel abundance. The problem is that *probability of disruption* is not a continuous function of the PUCL rules' parameters; thus it is difficult for the optimisation routine to minimise that performance statistic. Instead, *mean annual catch* (which is continuous) is maximised as a proxy for *probability of disruption*. For two simulations given the same random series of pelagic bycatches, *mean annual catch* could increase only if a previously insufficient PUCL was increased. Therefore, that performance statistic in some sense measures the amount of undisrupted fishing that will take place.

To avoid settling in local maxima, a grid search was conducted over starting parameter values for each PUCL rule in the optimisation routine. Effectively, this is an exhaustive search through a manually specified subset of reasonable initial parameter values. Unfortunately, due to the long computational time required for each function evaluation, the size of these subsets was somewhat limited.

5.4 Results and discussion

Table 5.2a reports the optimal parameter values for each PUCL rule at various levels of the bycatch scaling factor, k. It is evident that as k increases, smaller changes in the parameter values are required in order to maintain the same target *risk of depletion*. This phenomenon is more apparent in Figures 5.5, 5.6 and 5.7 which illustrate these changes in the parameter values for the *fixed*, $PUCL_3$, and *step* and *piecewise linear* rules respectively. For k values greater than two, the optimal parameter values do not change appreciably. All recommendations will consequently correspond to the optimal parameter values for $k = 2$, so that if k has been underestimated it is unlikely that there will be a sudden negative response of the resource.

A summary of the performance of the PUCL rules is given in Table 5.3a, while those results are presented graphically in Figure 5.8. The *piecewise linear* rule performed the best, with the highest mean annual catches and the lowest probabilities of disruption for all values of k considered. The *step* rule did only slightly worse, with the same probabilities of disruption and mean catches that differed by less than 100 tonnes. However, Table 5.2a and Figure 5.7 indicate that for $k >= 2$ the optimal value of $PUCL_{min}$ (the lowest possible PUCL) for both of those PUCL rules is zero. This is potentially problematic, because it means that for some years the pelagic industry would not be permitted to fish at all. Tables 5.2b and 5.3b report the results for the *step* and *piecewise linear* PUCL rules when $PUCL_{min}$ was constrained to be greater than a more reasonable 2 000 tonnes. In that case, the *piecewise linear* and *step* rules performed no better than the $PUCL_3$ rules, with similar mean annual catches and increased probabilities of disruption.

The $PUCL_3$ rules performed slightly better than the baseline *fixed* rule in terms of mean catch. There is little appreciable difference between the *reserve* and *no reserve* variants of this rule. Therefore, deciding between them should primarily be a choice left to industry preference.

In summary, if industry is not deterred by the possibility of there being years with zero PUCL, then the *piecewise linear* PUCL rule with $I_{min} = 3\,064$, $I_{max} = 4\,667$ and $PUCL_{min} = 0$ would be recommended for implementation. This should result in the highest mean catches and the least disruptions to industry compared to the other PUCL rules. Otherwise, $PUCL_3$ *no reserve* with $PUCL_3 = 15\,589$ and $PUCL_3$ *reserve* with $PUCL_3 = 16\,814$, appear to be the next best options. By not depending on survey results, these rules have the added benefits of simplicity and predictability. Furthermore, because the $PUCL_3$ rules are functions only of past bycatches, they afford the pelagic industry some amount of influence regarding future PUCLs, i.e. if they can manage greater restraint in one year, their flexibility in the next will be greater

Table 5.1: Pearson's correlation coefficient between the pelagic hydro-acoustic survey biomass indices and recruitment as estimated by the models in the Reference Set.

Survey dataset	Correlation coefficient			
	Model 1	Model 2	Model 3	Model 4
West Coast - outlier included	0.71	0.68	0.81	0.67
West Coast - outlier omitted	0.30	0.25	0.32	0.24
Entire survey area - outlier included	0.34	0.31	0.37	0.30
Entire survey area - outlier omitted	0.64	0.60	0.69	0.58

Table 5.2: Optimal parameter values for the PUCL rules for various values of k for all models in the Reference Set.

(a) Results for the PUCL rules as described in Section 5.2 (i.e. without additional constraints).

PUCL rule	Parameter	Bycatch scaling factor		
		$k = 1$	$k = 2$	$k = 3$
Fixed	$PUCL_{fix}$	8 775	4 826	4 300
Step	I_{step}	2 112	3 816	4 154
Step	$PUCL_{min}$	4228	0	0
Piecewise linear	I_{min}	1 858	3 064	3 580
Piecewise linear	I_{max}	2 323	4 667	5 519
Piecewise linear	$PUCL_{min}$	4 211	0	0
$PUCL_3$ no reserve	$PUCL_3$	18 919	15 589	13 816
$PUCL_3$ reserve	$PUCL_3$	21 272	16 814	13 871

(b) Results for the *step* and *piecewise linear* PUCL rules subject to the constraint $PUCL_{min} > 2\,000$.

PUCL rule	Parameter	Bycatch scaling factor		
		$k = 1$	$k = 2$	$k = 3$
Step ($PUCL_{min} > 2\,000$)	I_{step}	2 112	5 631	10 492
Step ($PUCL_{min} > 2\,000$)	$PUCL_{min}$	4 228	2 112	3 249
Piecewise linear ($PUCL_{min} > 2\,000$)	I_{min}	1 858	3 097	4 203
Piecewise linear ($PUCL_{min} > 2\,000$)	I_{max}	2 323	8 939	12 457
Piecewise linear ($PUCL_{min} > 2\,000$)	$PUCL_{min}$	4 211	2 215	2 310

Table 5.3: Optimal performance statistics for the PUCL rules for various values of k for all models in the Reference Set (combined by taking means). Note that *risk of depletion* is the same for all results, because the PUCL rules were tuned to give the same value. Section 5.3.2 describes how these performance statistics are calculated. Bootstrap CVs are reported in brackets.

(a) Results for the PUCL rules as described in Section 5.2 (i.e. without additional constraints).

k	PUCL rule	Performance statistics			
		Risk of depletion	B^{sp}_{2023}/K^{sp}	Mean catch (KT)	Probability of disruption
1	Fixed	5.6% (0.13)	54.6% (0.02)	4.0 (0.01)	12.5% (0.10)
1	Step	5.6% (0.11)	53.4% (0.02)	4.5 (0.01)	12.5% (< 0.01)
1	Piecewise linear	5.6% (0.12)	53.3% (0.02)	4.5 (0.02)	12.5% (< 0.01)
1	$PUCL_3$ no reserve	5.6% (0.11)	54.6% (0.02)	4.1 (0.01)	25.0% (< 0.01)
1	$PUCL_3$ reserve	5.6% (0.12)	54.6% (0.02)	4.1 (0.01)	25.0% (0.10)
2	Fixed	5.6% (0.16)	54.7% (0.02)	3.8 (0.01)	50.0% (< 0.01)
2	Step	5.6% (0.10)	52.1% (0.01)	4.8 (0.02)	50.0% (< 0.01)
2	Piecewise linear	5.6% (0.09)	52.0% (0.01)	4.8 (0.02)	50.0% (< 0.01)
2	$PUCL_3$ no reserve	5.6% (0.12)	53.9% (0.02)	4.2 (0.01)	50.0% (0.01)
2	$PUCL_3$ reserve	5.6% (0.11)	53.7% (0.02)	4.4 (0.01)	50.0% (< 0.01)
3	Fixed	5.6% (0.14)	54.4% (0.02)	3.9 (< 0.01)	75.0% (< 0.01)
3	Step	5.6% (0.11)	50.8% (0.02)	5.1 (0.02)	62.5% (< 0.01)
3	Piecewise linear	5.6% (0.12)	50.8% (0.01)	5.1 (0.02)	62.5% (< 0.01)
3	$PUCL_3$ no reserve	5.6% (0.13)	54.2% (0.02)	4.0 (0.02)	75.0% (0.06)
3	$PUCL_3$ reserve	5.6% (0.14)	54.3% (0.02)	3.9 (0.01)	75.0% (0.05)

(b) Results for the *step* and *piecewise linear* PUCL rules subject to the constraint $PUCL_{min} > 2\,000$.

k	PUCL rule	Performance statistics			
		Risk of depletion	B^{sp}_{2023}/K^{sp}	Mean catch (KT)	Probability of disruption
1	Step ($PUCL_{min} > 2\,000$)	5.6% (0.11)	53.4% (0.02)	4.5 (0.01)	12.5% (< 0.01)
1	Piecewise linear ($PUCL_{min} > 2\,000$)	5.6% (0.12)	53.3% (0.02)	4.5 (0.02)	12.5% (< 0.01)
2	Step ($PUCL_{min} > 2\,000$)	5.6% (0.10)	52.8% (0.02)	4.3 (0.01)	59.4% (0.04)
2	Piecewise linear ($PUCL_{min} > 2\,000$)	5.6% (0.11)	52.4% (0.01)	4.4 (0.01)	62.5% (0.01)
3	Step ($PUCL_{min} > 2\,000$)	5.6% (0.16)	53.3% (0.02)	4.1 (0.01)	75.0% (0.01)
3	Piecewise linear ($PUCL_{min} > 2\,000$)	5.6% (0.11)	52.5% (0.02)	4.1 (0.02)	75.0% (< 0.01)

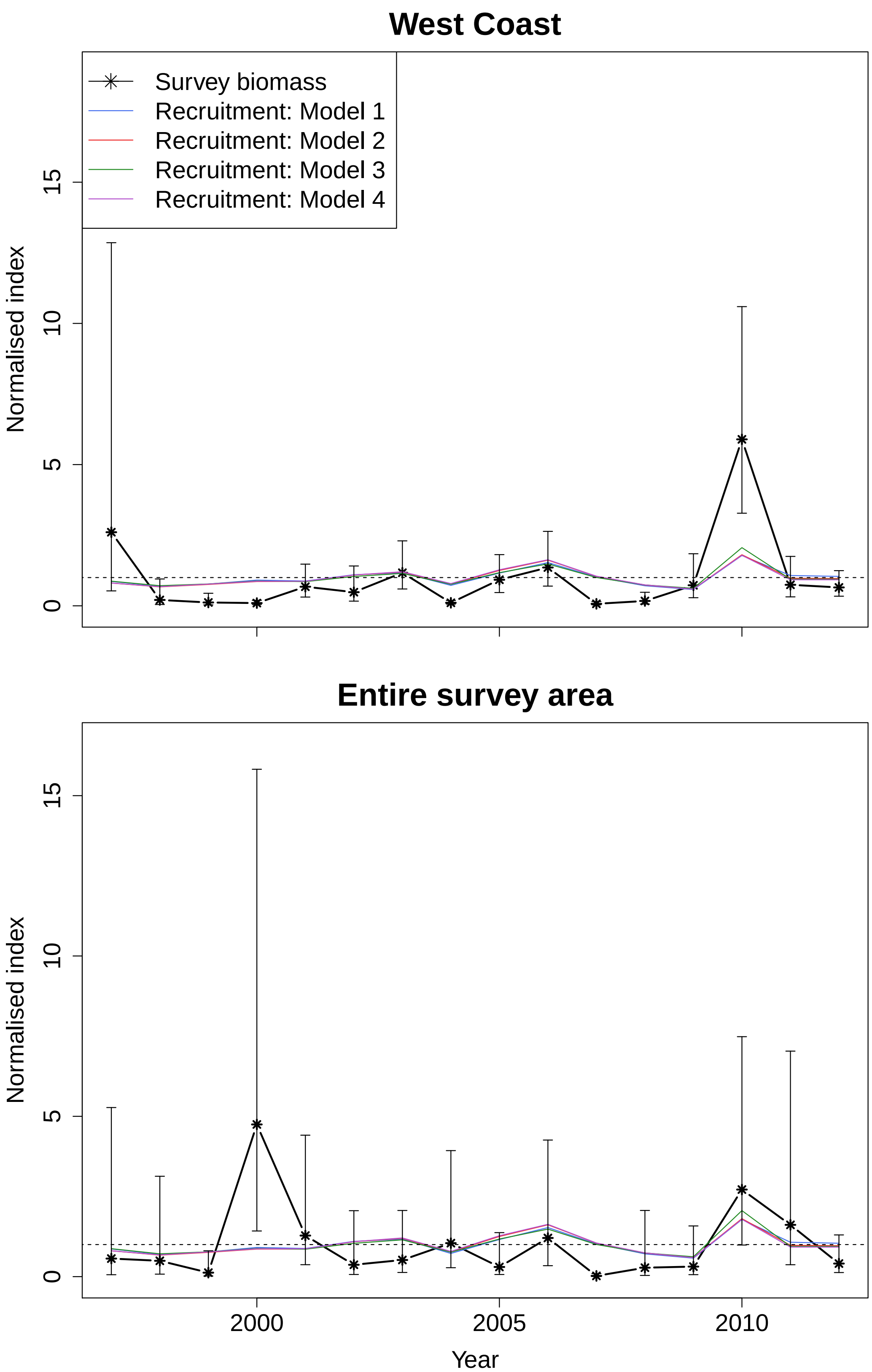

Figure 5.1: Comparison between the November pelagic hydro-acoustic survey estimated biomass indices for juvenile horse mackerel and recruitment as estimated by the models in the Reference Set. The series have been normalised by dividing each by its mean. Error bars show 95% confidence intervals.

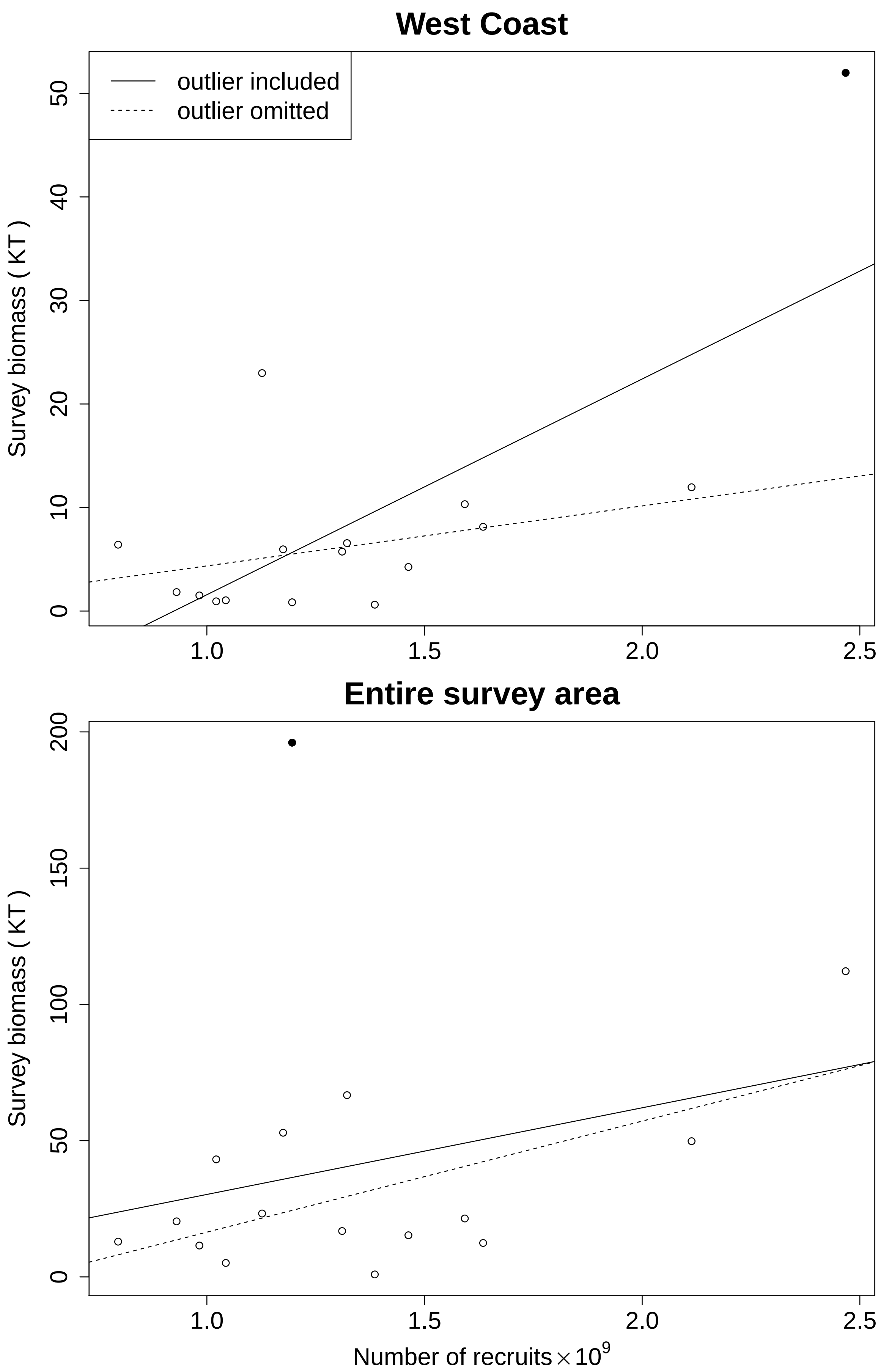

Figure 5.2: Linear regressions between the November pelagic hydro-acoustic survey biomass indices for juvenile horse mackerel and recruitment as estimated by the *base case* model in the Reference Set, with outliers from the surveys both included and omitted. The data points represented by solid circles have been identified as outliers.

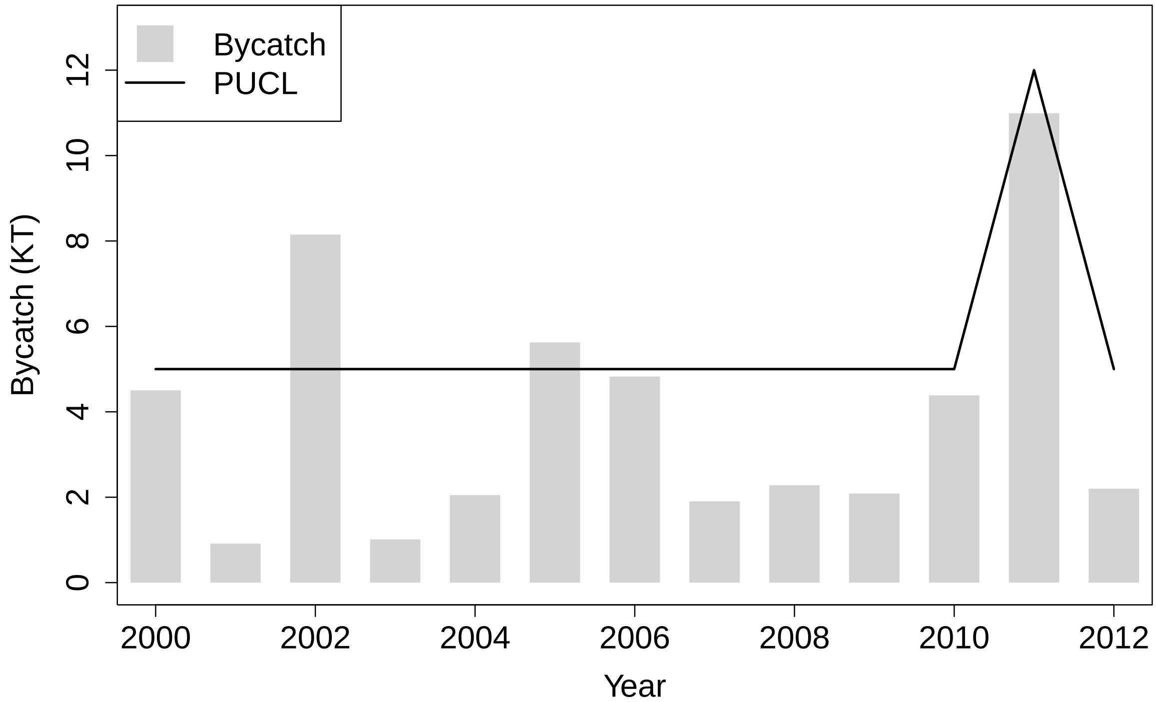

Figure 5.3: Comparison between actual pelagic horse mackerel bycatches and annual PUCLs.

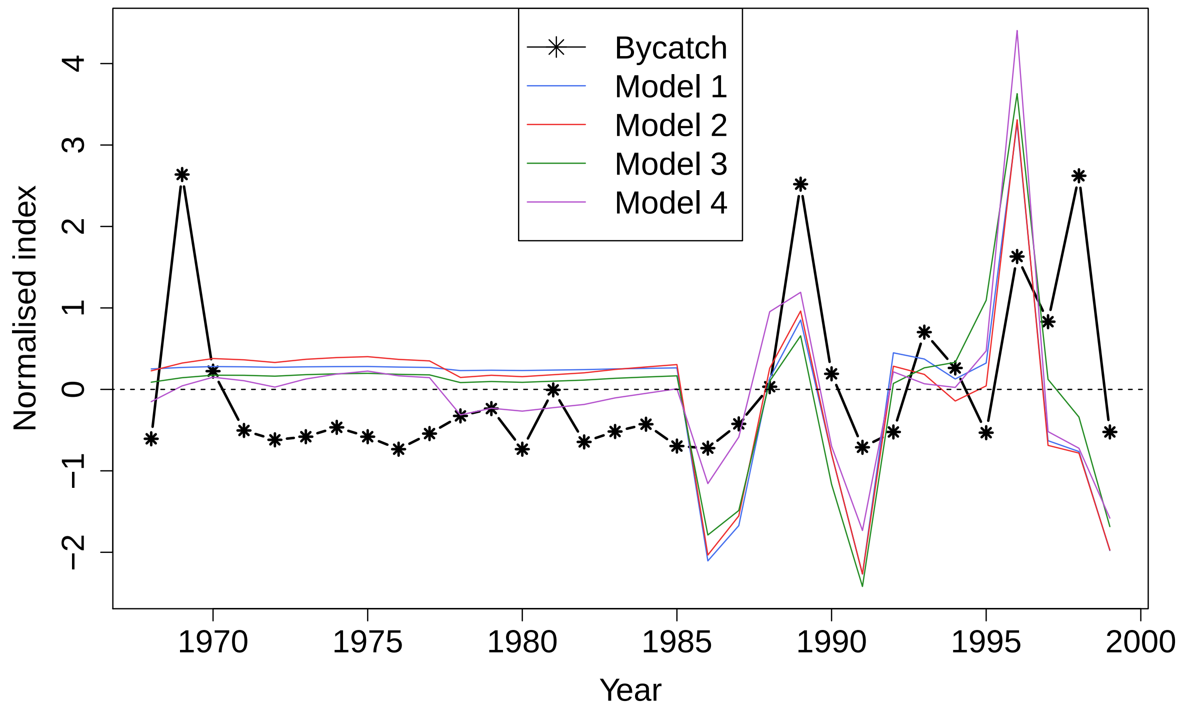

Figure 5.4: Comparison between pelagic horse mackerel bycatches and annual recruitment as estimated by the models in the Reference Set. Each series has been normalised by subtracting its mean and dividing by its standard deviation.

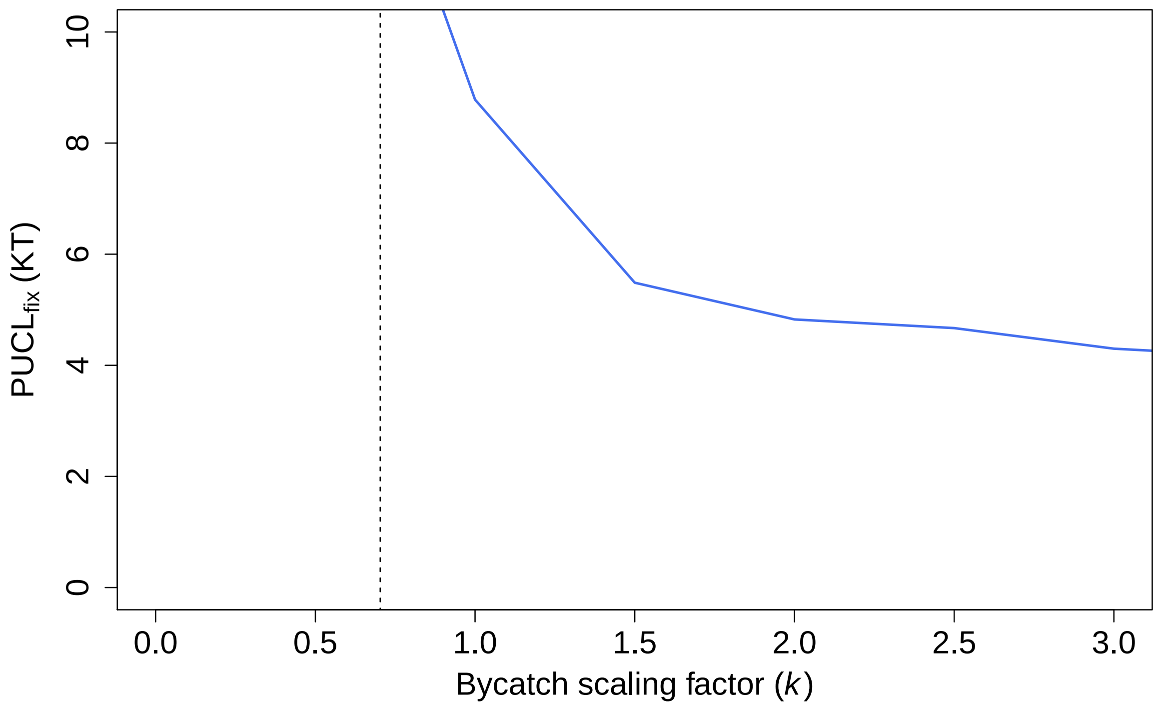

Figure 5.5: PUCL$_{fix}$ values required at various levels of k in order to maintain the target *risk of depletion* for the *fixed* PUCL rule for all models in the Reference Set.

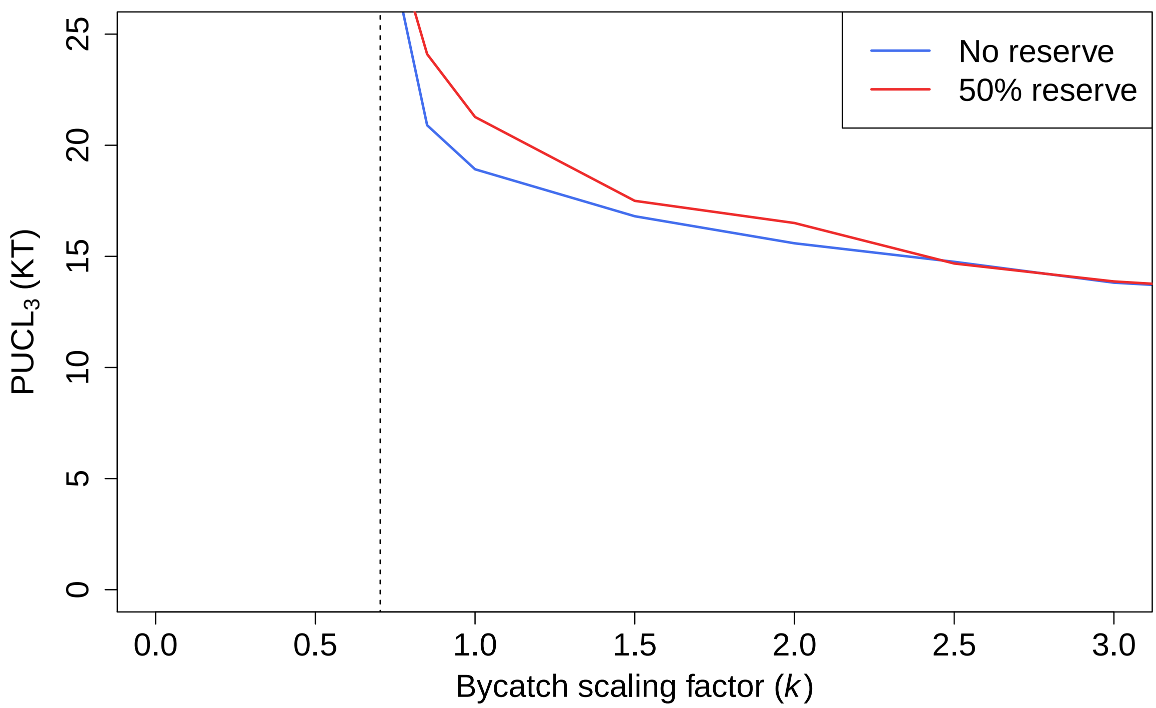

Figure 5.6: PUCL$_3$ values required at various levels of k in order to maintain the target *risk of depletion* for the $PUCL_3$ rules for all models in the Reference Set.

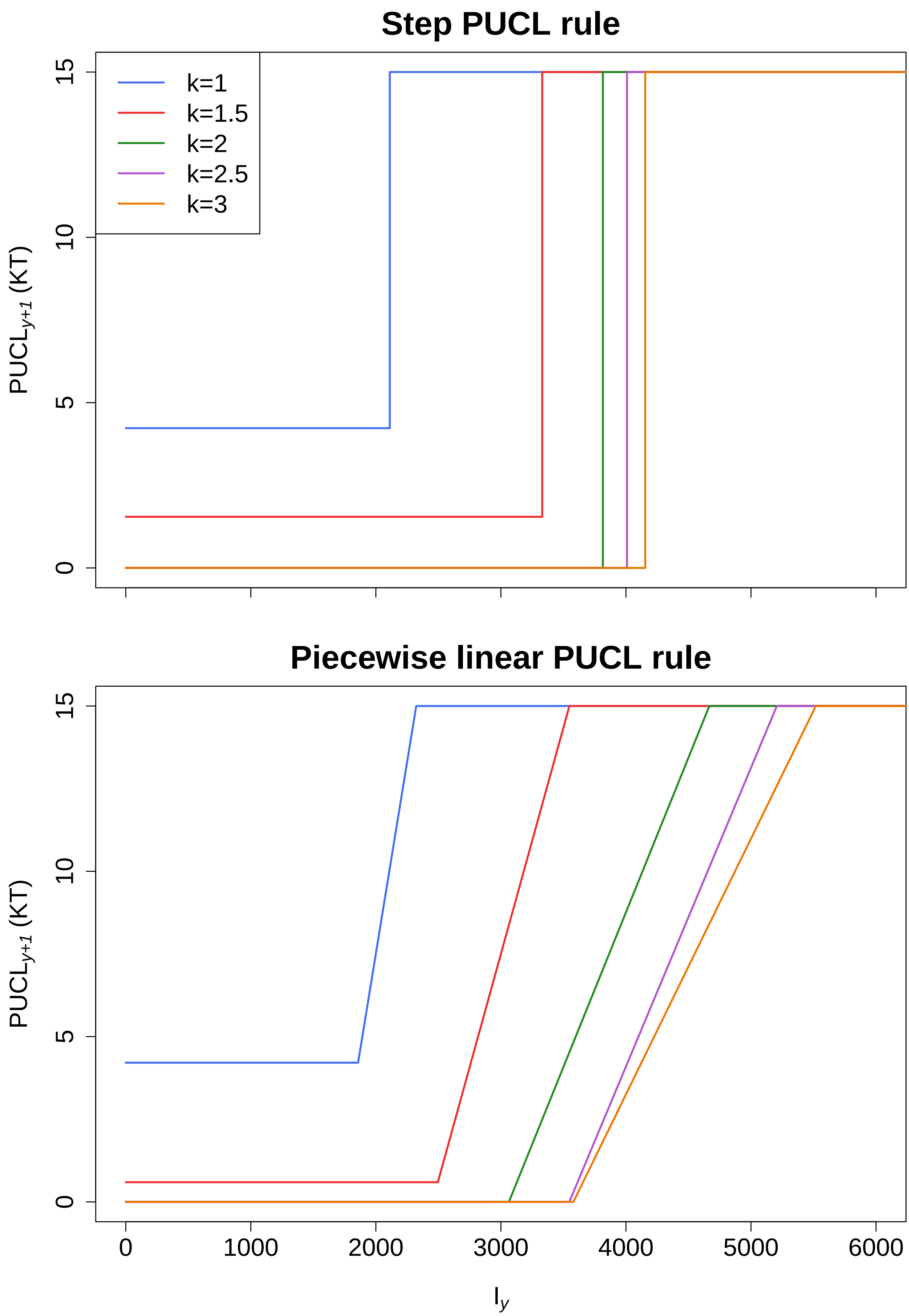

Figure 5.7: Step and piecewise linear functions required at various levels of k in order to maintain the target *risk of depletion* for the survey-based PUCL rules for all models in the Reference Set.

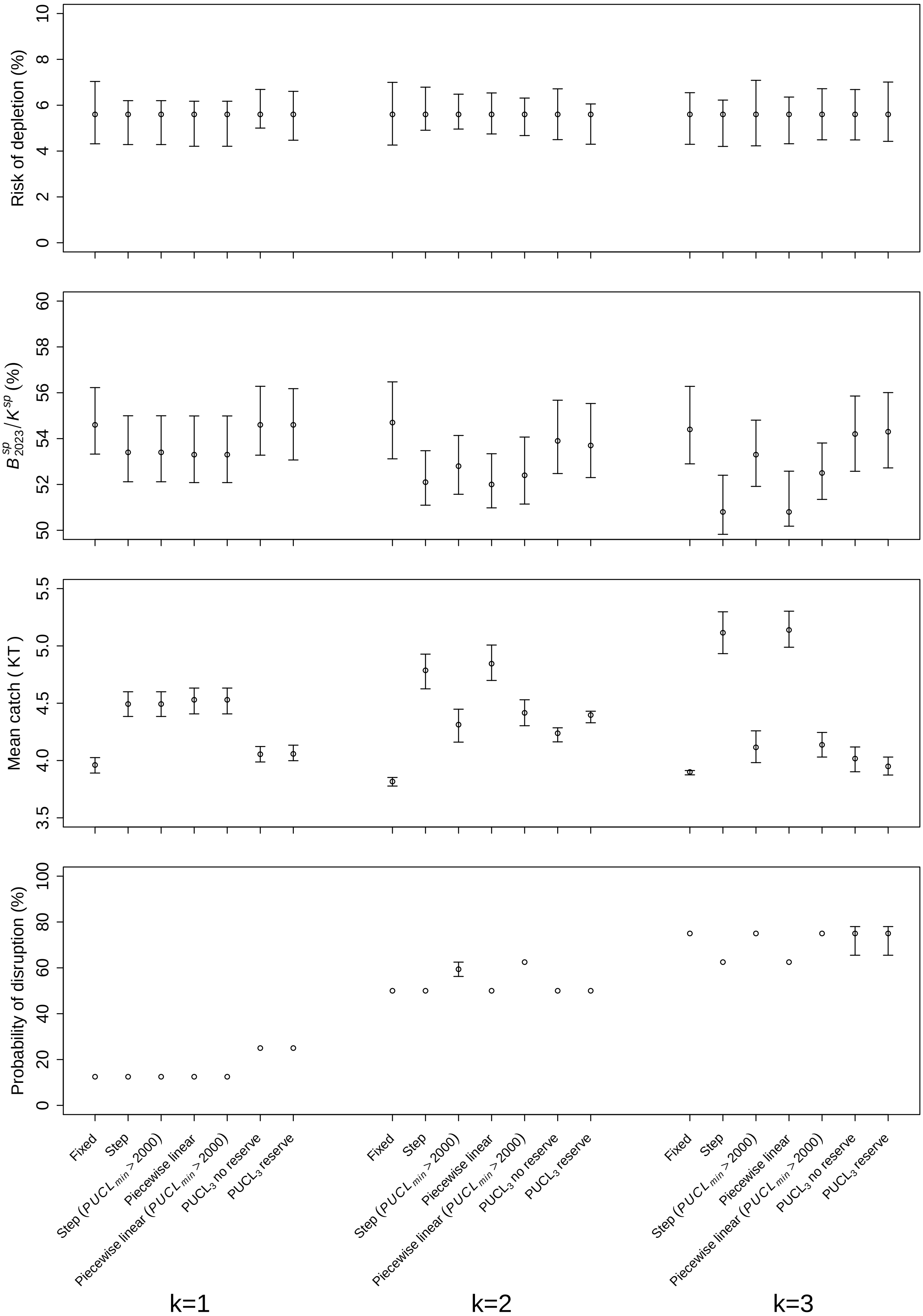

Figure 5.8: Performance statistics for the PUCL rules at various values of k for all models in the Reference Set (combined by taking means). Bootstrap 90% probability envelopes are indicated with error bars. Note that they are not shown for most results in the bottom plot, because *probability of disruption* is a step function and therefore most of the associated 90% probability envelopes include only the median values. Also note that the vertical axes for these plots have been expanded for clearer differentiation of the results for the different PUCL rules.

Chapter 6

Total Allowable Catch for the midwater fishery

It is strongly suspected that absolute biomass estimates from demersal swept-area surveys are negatively biased ("underestimated"), and that therefore the resource is underutilised (Section 3.2.1). This chapter describes the use of the MSE approach to develop and evaluate an adaptive rule that experimentally increases the midwater TAC, while maintaining an acceptably low risk of resource depletion. It is hoped that the candidate rules will be able to distinguish between the different OMs in the Reference Set, providing larger TAC increases for the more productive model variants.

The first section provides the details of the various TAC rules considered, including their technical specifications and the motivation behind their designs. The following section describes the methodology used to evaluate the candidate rules. Although it is largely similar to that used when testing PUCL rules in the previous chapter, there are several important distinctions. The chapter then concludes with a presentation and discussion of the key results.

6.1 Candidate TAC rules

A constant catch rule which maintains the TAC at its 2014 level of 38 115 tonnes per annum is included as a baseline against which the effectiveness of the other candidate rules can be compared. These other rules take the form:

$$TAC_{y+1} = \Delta_y TAC_y \tag{6.1}$$

where

TAC_y is the TAC for the midwater fishery for year y; and

Δ_y is an output of the TAC rule that reflects the percentage change in TAC from year y to year $y+1$.

Δ_y is given by the piecewise linear function:

$$\Delta_y = \begin{cases} 1 - X_{decr} & \text{if } I_y < I_{decr} \\ 1 - X_{decr} + \frac{X_{incr}+X_{decr}}{I_{incr}-I_{decr}}\left(I_y - I_{decr}\right) & \text{if } I_{decr} \leq I_y < I_{incr} \\ 1 + X_{incr} & \text{if } I_y \geq I_{incr} \end{cases} \tag{6.2}$$

where

X_{decr} is a fixed parameter that determines the largest possible percentage decrease in TAC from one year to the next;

X_{incr} is a fixed parameter that determines the largest possible percentage increase in TAC from one year to the next;

I_y is a variable which gives the ratio of recent abundance to the mean abundance over the period 2003–2009 for year y (described in detail in Section 6.1.1 below);

I_{decr} is a fixed parameter that determines the value of I_y below which the TAC is decreased by the maximum percentage (X_{decr}); and

I_{incr} is a fixed parameter that determines the value of I_y above which the TAC is increased by the maximum percentage (X_{incr}).

Figure 6.1 illustrates the form of these candidate TAC rules. Effectively, if the recent abundance index is high compared to the 2003–2009 average (i.e. $I_y \gg 1$), then the TAC is increased; conversely, if the recent abundance index is low (i.e. $I_y \ll 1$), then the TAC is decreased. This rule will consequently tend to adjust the TAC until horse mackerel biomass is in equilibrium at the level which gives an I_y corresponding to $\Delta_y = 1$ (i.e. TAC is unchanged from year to year). For example, if the parameters of the TAC rule are chosen such that $\Delta_y = 1$ at $I_y = 0.9$, then we are in fact specifying a target biomass that is approximately 90% of the average 2003–2009 biomass. It is hoped that this form of rule will lead to larger TAC increases for the more optimistic OMs with higher K^{sp} and associated sustainable yield estimates, because the same proportional changes in biomass are equivalent to larger changes in absolute terms for these models. This should result in larger equilibrium midwater catches for these more optimistic OMs.

Stake-holders were asked at a meeting of the DAFF Horse Mackerel Task Team to identify reasonable parameter values for the TAC rules. It was decided there that X_{decr} and X_{incr} would be fixed, while I_{decr} and I_{incr} would be free to take whatever values necessary to achieve optimal results. The following variants for the TAC rule are evaluated:

- *Base* case with $X_{decr} = 15\%$, $X_{incr} = 10\%$
- *Smaller* X_{incr} case with $X_{decr} = 15\%$, $X_{incr} = 5\%$
- *Larger* X_{decr} case with $X_{decr} = 20\%$, $X_{incr} = 10\%$

6.1.1 Combined index of abundance

The index of abundance I_y, which is used as an input for the TAC rules, quantifies the recent abundance of horse mackerel relative to its average level over the period 2003–2009. It is given by a weighted average of the recent relative levels of the midwater CPUE and autumn demersal survey abundance indices:

$$I_y = wI_y^{cpue} + (1 - w)\, I_y^{aut} \tag{6.3}$$

where

I_y^s is the recent abundance for year y relative to its average over the period 2003–2009 according to index s ($s = cpue$ for the CPUE index and $s = aut$ for the autumn demersal survey index); and

w specifies the weighting of each term in the aggregated value and is set equal to 0.85 (see below).

The terms in Equations 6.3 are given by:

$$I_y^{cpue} = \frac{1/3 \sum_{j=y-3}^{y-1} CPUE_j}{1/7 \sum_{j=2003}^{2009} CPUE_j} \quad \text{and} \quad I_y^{aut} = \frac{1/3 \sum_{j=y-2}^{y} aut_j}{1/7 \sum_{j=2003}^{2009} aut_j} \tag{6.4}$$

where $CPUE_y$ and aut_y are the actual CPUE and autumn demersal survey biomass estimates for year y respectively. The numerators in the expressions for I_y^{cpue} and I_y^{aut} use the mean value of the indices over three years, instead of just the most recent year, in order that they better reflect the levels of abundance (and their trends) rather than the noise in their associated indices. Note that according to Equations 6.1–6.4, the TAC for year $y+1$ depends on the values of the autumn demersal survey index for years y, $y-1$ and $y-2$, and on the values of the CPUE index for years $y-1$, $y-2$ and $y-3$. It does not depend on the CPUE for year y, because it is unlikely that that value would be finalised in time to be able to calculate the following year's TAC in advance.

The value of $w = 0.85$ (Equation 6.3) is determined through the inverse-variance weighting method. This approach minimises the variance of the aggregated index I_y by weighting each random variable in proportion to the inverse of its variance. The variance of each series is calculated using its model-estimated standard deviation. However, the variances of the demersal survey biomass estimates vary from year to year (Section 3.2); therefore, the median of that index's CVs was used in combination with its model-estimated additional variance σ_{add}^{aut} to estimate its average total variance. It is not surprising that the CPUE index receives a much higher weighting than the survey index—CPUE data are collected throughout the entire fishing season, while surveys are conducted over only a comparatively brief period each year.

The spring demersal survey abundance estimates are not included in the combined index I_y for two reasons. First, the last such survey was conducted in 2008; therefore, it would be unwise to assume that they will take place regularly in the future. Second, the model-estimated additional variance for this survey is approximately 0.7. Given its magnitude in comparison to the variance of the CPUE and autumn demersal survey indices (Table 4.2), it would have little impact on the inverse-variance weighted average I_y.

6.1.2 Missing demersal surveys

Autumn demersal surveys were not conducted in 2012 and 2013 because the research vessel *RV Africana* was unavailable. The 2014 autumn demersal survey was conducted by the commercial vessel *FV Andromeda*; however, the resulting horse mackerel biomass estimate first needs to be calibrated to the existing *RV Africana* series before it can be used. Unfortunately, it is unlikely that this work will be accomplished owing to a lack of resources at DAFF. These are not the only missing survey data—over the period 1988–2011 there are three other years without horse mackerel biomass estimates from autumn surveys (Table 3.3). However, this problem is particularly problematic in calculating the projected TACs for 2015, 2016 and 2017, because the control rule for these years depends on the missing 2012, 2013 and 2014 survey biomass estimates.

At a meeting of the DAFF DSWG on 17 June 2014 where various options were debated, it was eventually decided that given these missing autumn survey data, the projected TACs for 2015, 2016 and 2017 will be calculated using the CPUE index only. This is equivalent to setting $w = 1$ in Equation 6.3 for those years. It is unlikely that this choice will have an appreciable impact on the projection results, as the CPUE index is much more heavily weighted than the survey index in the aggregated index I_y.

6.1.3 Revised CPUE series

In late 2014 it was discovered that there were several omissions in the *Desert Diamond* midwater trawling data that had been provided by DAFF and were used to produce a CPUE series (Section 3.A). Singh *et al.* (2014) subsequently provided a revised CPUE series that incorporated these omitted data. Table 6.1 compares the original CPUE series to the revised CPUE series.

Because of the lateness of this discovery and a fast approaching deadline for the 2015 TAC recommendation, it was not possible to revise the entire assessment and PUCL rule evaluation process. Instead, only the OMs used to simulation test the midwater TAC rules were conditioned on the revised CPUE series. Also, the revised CPUE series is used as input for the TAC rules (Equation 6.4).

6.2 Method

Candidate midwater TAC rules are evaluated by projecting the dynamics of the resource ten years into the future. The methodology is similar to that adopted in testing PUCL rules, outlined in Section 5.3, but there are a few differences. First, the future midwater catch for any given year is set equal to the TAC for that year, which is determined by the candidate TAC rule being assessed. Furthermore, for these projections the pelagic bycatch scaling factor k is fixed at a value of 2 in line with the precautionary approach that was adopted when determining the PUCL recommendations in Chapter 5, and midwater TAC recommendations in Furman and Butterworth (2012). Finally, future PUCLs are determined by the $PUCL_3$ *no reserve* rule with

$PUCL_3 = 15\,589$ tonnes. This rule is used for projections, because at a meeting of the DAFF Pelagic Scientific Working Group in June 2014 it was decided to implement that MP.

6.2.1 Performance statistics

The following statistics are used to assess the performance of each TAC rule:

Median spawning biomass

This statistic reflects the expected status of the resource at the end of the projection period. It is defined here in the same manner as for the PUCL projections (Equation 5.9).

Risk of depletion

This statistic quantifies the risk of resource depletion. It is given by the minimum value over the projection period of the lower fifth percentile of spawning biomass relative to its pristine level for the most pessimistic OM. Equation 5.10 describes the statistic as it was used for the evaluation of candidate PUCL rules, and applies here as well.

Mean annual catch

This statistic gives the expected annual directed midwater horse mackerel catch over the projection period. For a single simulation it is defined as:

$$\tfrac{1}{8} \sum_{y=2015}^{2022} C_y^m \tag{6.5}$$

Catches before 2015 are not included in the average, because the TACs for those years have already been decided by DAFF, and thus they should not factor into the performance evaluation of the candidate TAC rules.

Catch range

This statistic refers to difference in average midwater catch for the optimistic model ($q_{aut} = 0.5$, $h = 0.9$) and the pessimistic model ($q_{aut} = 1$, $h = 0.6$) over the projection period. In other words:

$$\frac{1}{8}\left(\sum_{y=2015}^{2022} C_y^{m,optim} - \sum_{y=2015}^{2022} C_y^{m,pessim}\right) \tag{6.6}$$

It is included as a performance statistic, because it is hoped that the candidate TAC rules will show an ability to differentiate between the two OMs and provide greater increases for the higher productivity optimistic scenario.

Average Annual Variation (AAV)

This statistic is expressed as a percentage and indicates the average proportional variation in TAC from one year to the next. Lower values are associated with increased industrial stability. For a single simulation it is given by:

$$\frac{1}{8}\sum_{y=2015}^{2022} \frac{\left|C_y^m - C_{y-1}^m\right|}{C_{y-1}^m} \tag{6.7}$$

Recall that when evaluating an MP, once stochastic effects are taken into account, the result for each performance statistic is a distribution. Consequently, the medians of these distributions are calculated, except for *risk of depletion* which reflects the lower fifth percentile of projected spawning biomass. Additionally, because the tests are run for each of the four OMs in the Reference Set, the means of those medians are reported unless otherwise stated (i.e. *risk of depletion* and *catch range*).

6.2.2 Tuning and optimisation

As was the case for candidate PUCL rules, the parameters of each TAC rule are tuned to give the same risk-related performance as was considered acceptable in relation to the results in Johnston and Butterworth (2007). Specifically, this risk is no more than a 5% chance that spawning biomass will be depleted to lower than 5.6% of its pristine level over the projection period. The same risk target is selected as was used for the PUCL rules, because it is hoped that the TAC rules will be able to secure improved utilisation without undue increase in the risk of unintended reduction of resource abundance. Again, MATLAB's constrained non-linear optimisation routine is used to maximise the objective function:

$$f(\vec{x}) = C(\vec{x}) \tag{6.8}$$

subject to the non-linear constraint

$$r(\vec{x}) = r_{tar} \tag{6.9}$$

where

$\vec{x}$ is a vector of the parameters being optimised for the TAC rule being assessed (i.e. I_{decr} and I_{incr});

C is the performance statistic, *mean annual* (midwater) *catch*;

r is the performance statistic, *risk of depletion*; and

r_{tar} is the target *risk of depletion*, which has a value of 5.6%.

A grid search of initial values for the parameters being optimised was conducted in order to find global maxima. As the optimisation need only be performed for one value of the bycatch scaling factor ($k = 2$) and there are, on average, fewer free parameters, this grid search is more exhaustive than that used for the candidate PUCL rules.

6.3 Results and discussion

Table 6.2 contrasts the results of the various candidate TAC rules under the idealised situation where there are no random errors about the expected values of future CPUE and demersal survey abundance indices. It shows that the adaptive control rules secure increases in average annual catches that range between 8 000 and 15 600 tonnes, for the same *risk of depletion*, in comparison to the constant allocation case. The average annual proportional change in TAC varies from 5.1% to 10.8%. Note that *risk of depletion* has the highest bootstrap CVs of all the performance statistics. This is particularly important, because this statistic was used to tune the parameters of each rule; therefore, small variations in risk would have appreciable effects on the values of the control parameters and also on the other statistics. The *smaller* X_{incr} rule performed the worst in terms of *mean annual catch*, but the best in terms of *AAV*.

Table 6.3 reports the results of the TAC rule evaluations once account is taken of realistic noise in future abundance indices, while Figure 6.2 presents those results graphically. The improvements in mean annual catch over the constant catch case drop to approximately 4 400 tonnes. The *base* rule suffers the greatest deterioration in performance with the introduction of these observation errors. Figure 6.3 illustrates the projected catches for each TAC rule. Figure

6.4 indicates that there is little difference in projected median spawning biomass among the TAC rules. It also illustrates how the rules achieve their results: by decreasing the range of the 90% probability interval of spawning biomass compared to the constant catch case, they allow the median spawning biomass to decrease while maintaining the same *risk of depletion*, which results in increased utilisation. When selecting which TAC rule to implement, stake-holders have to consider the trade-offs between improved catches and increased interannual TAC variability Catch increases are effectively guaranteed until 2016 and it is unlikely that they will decrease below the current level of 38 115 tonnes per annum until at least 2022 (Figure 6.3). By that time, further research on the extent of bias in the swept-area abundance estimates will allow an almost certain increase in the current survey abundance estimates to be accepted; this will in turn enable a refinement of the approach adopted here, towards another which would yield higher catches for the same perceived risk as at present.

Table 6.1: Comparison of the original CPUE series used in the assessment and described in Section 3.A, to the revised CPUE series from Singh *et al.* (2014).

Year	Original CPUE	Revised CPUE
2003	0.62	0.80
2004	0.71	0.69
2005	0.90	0.82
2006	1.00	1.00
2007	1.49	1.25
2008	1.02	0.91
2009	0.83	0.86
2010	1.11	1.13
2011	1.42	1.42
2012	0.91	0.68

Table 6.2: Projected performance of the candidate TAC rules (see text for definitions) under the idealised situation where random noise **is not added** to future observed abundance indices. Note that the parameters of each rule have been tuned to maintain a *risk of depletion* statistic (Equation 5.10) of 5.6% and to maximise *mean annual catch*. Bootstrap CVs are reported in brackets.

	TAC rule			
	Constant	*Base*	*Smaller* X_{incr}	*Larger* X_{decr}
X_{decr}	0%	15%	15%	20%
X_{incr}	0%	10%	5%	10%
I_{decr}	-	0.89	0.83	0.77
I_{incr}	-	0.92	0.87	0.97
Risk of depletion	5.6% (0.12)	5.6% (0.11)	5.6% (0.13)	5.6% (0.13)
Median B^{sp}_{2023}/K^{sp}	53.9% (0.02)	42.5% (0.01)	47.8% (0.02)	44.7% (0.01)
Mean annual catch (KT)	38.1 (0.00)	53.7 (0.02)	46.1 (0.01)	49.7 (0.01)
Catch range (KT)	0	8.0 (0.18)	4.7 (0.06)	5.1 (0.08)
AAV	0.0%	10.8% (0.02)	5.1% (0.03)	8.9% (< 0.01)

Table 6.3: Projected performance of the candidate TAC rules when random noise **is added** to future observed abundance indices. Note that the parameters of each rule have been tuned to maintain a *risk of depletion* statistic (Equation 5.10) of 5.6% and to maximise *mean annual catch*. Bootstrap CVs are reported in brackets.

	TAC rule			
	Constant	*Base*	*Smaller* X_{incr}	*Larger* X_{decr}
X_{decr}	0%	15%	15%	20%
X_{incr}	0%	10%	5%	10%
I_{decr}	-	0.84	0.77	0.79
I_{incr}	-	1.01	0.96	1.01
Risk of depletion	5.6% (0.12)	5.6% (0.14)	5.6% (0.17)	5.6% (0.14)
Median B^{sp}_{2023}/K^{sp}	53.9% (0.02)	47.5% (0.02)	49.6% (0.01)	47.5% (0.02)
Mean annual catch (KT)	38.1 (0.00)	43.4 (0.01)	41.1 (0.01)	43.1 (0.01)
Catch range (KT)	0	4.0(0.06)	2.9 (0.04)	4.7 (0.05)
AAV	0.0%	9.2% (< 0.01)	5.4% (0.01)	9.3% (0.01)

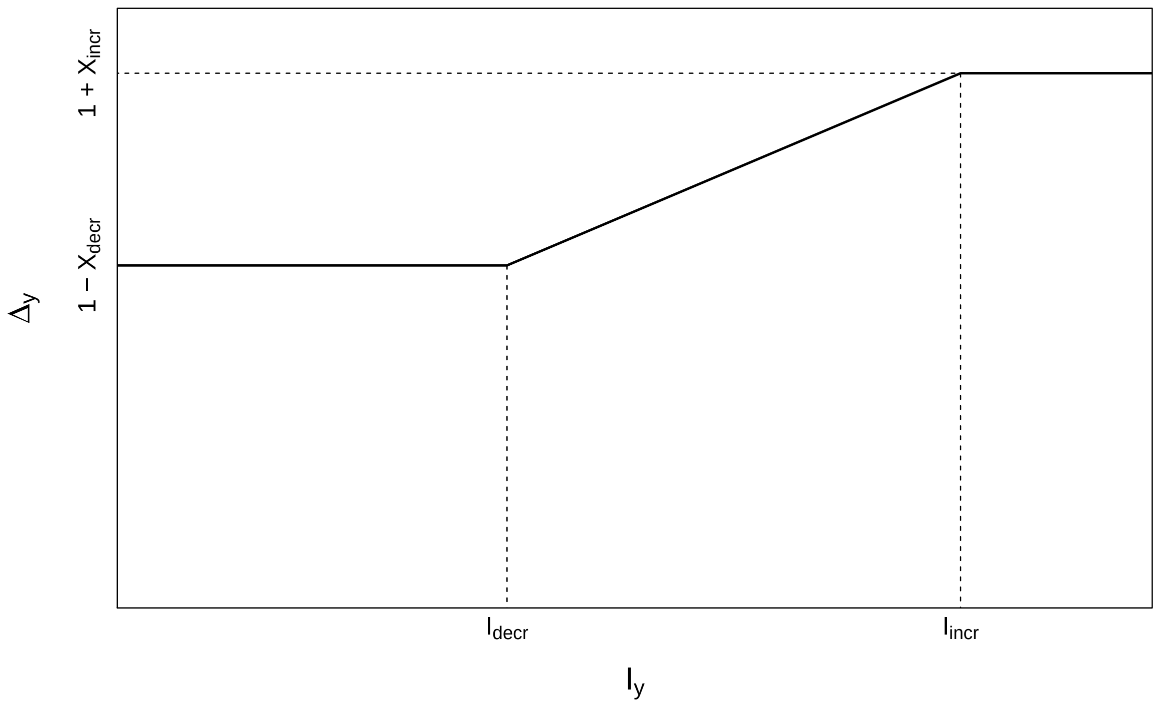

Figure 6.1: Illustration of the shape of Δ_y as a function of I_y for all candidate TAC rules, and how it is affected by the control parameters X_{decr}, X_{incr}, I_{decr} and I_{incr}.

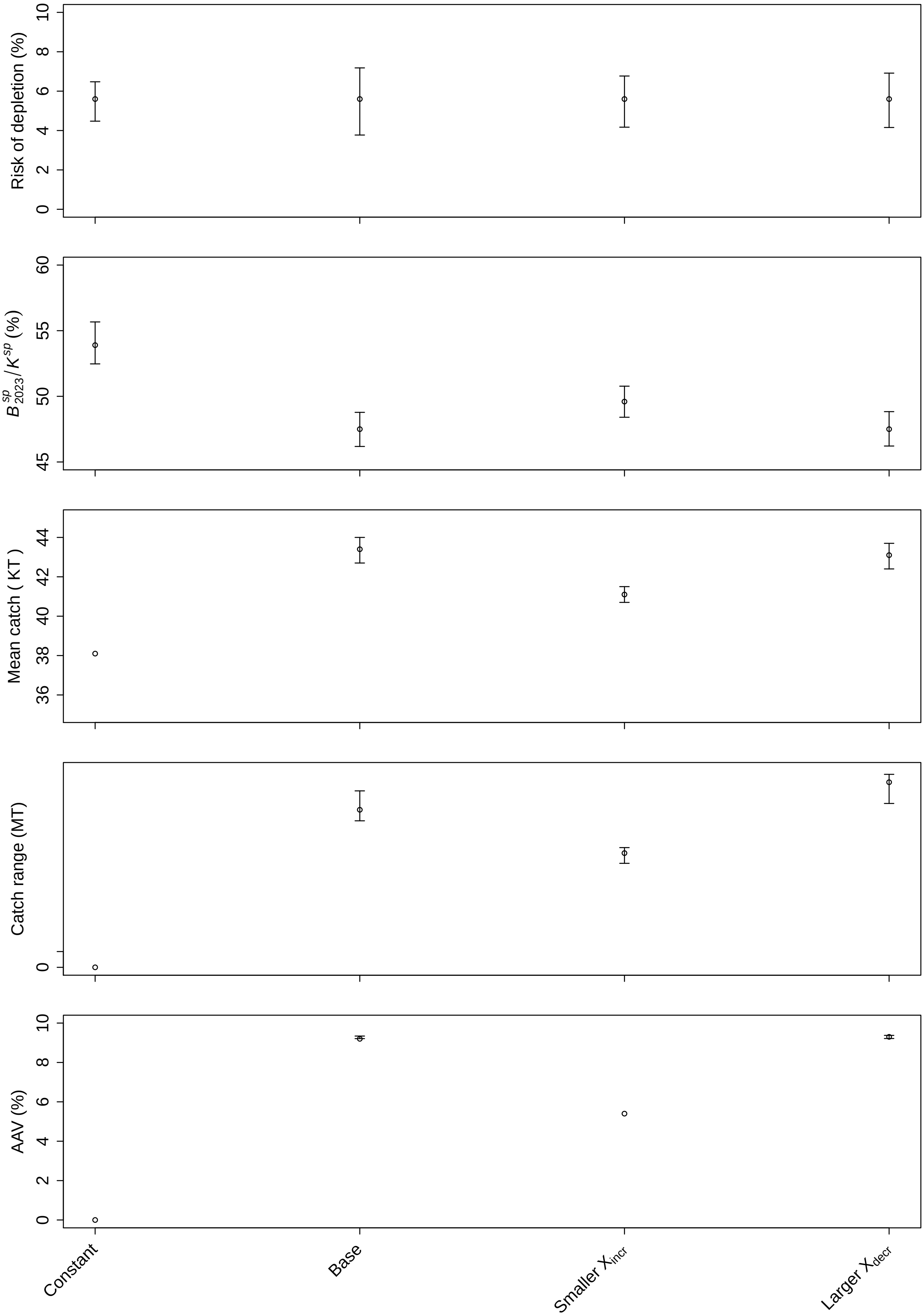

Figure 6.2: Projected performance of the candidate TAC rules when random noise **is added** to future observed abundance indices. Bootstrap 90% probability envelopes are indicated with error bars. Note that the vertical axes for these plots have been expanded for clearer differentiation of the results for the different TAC rules.

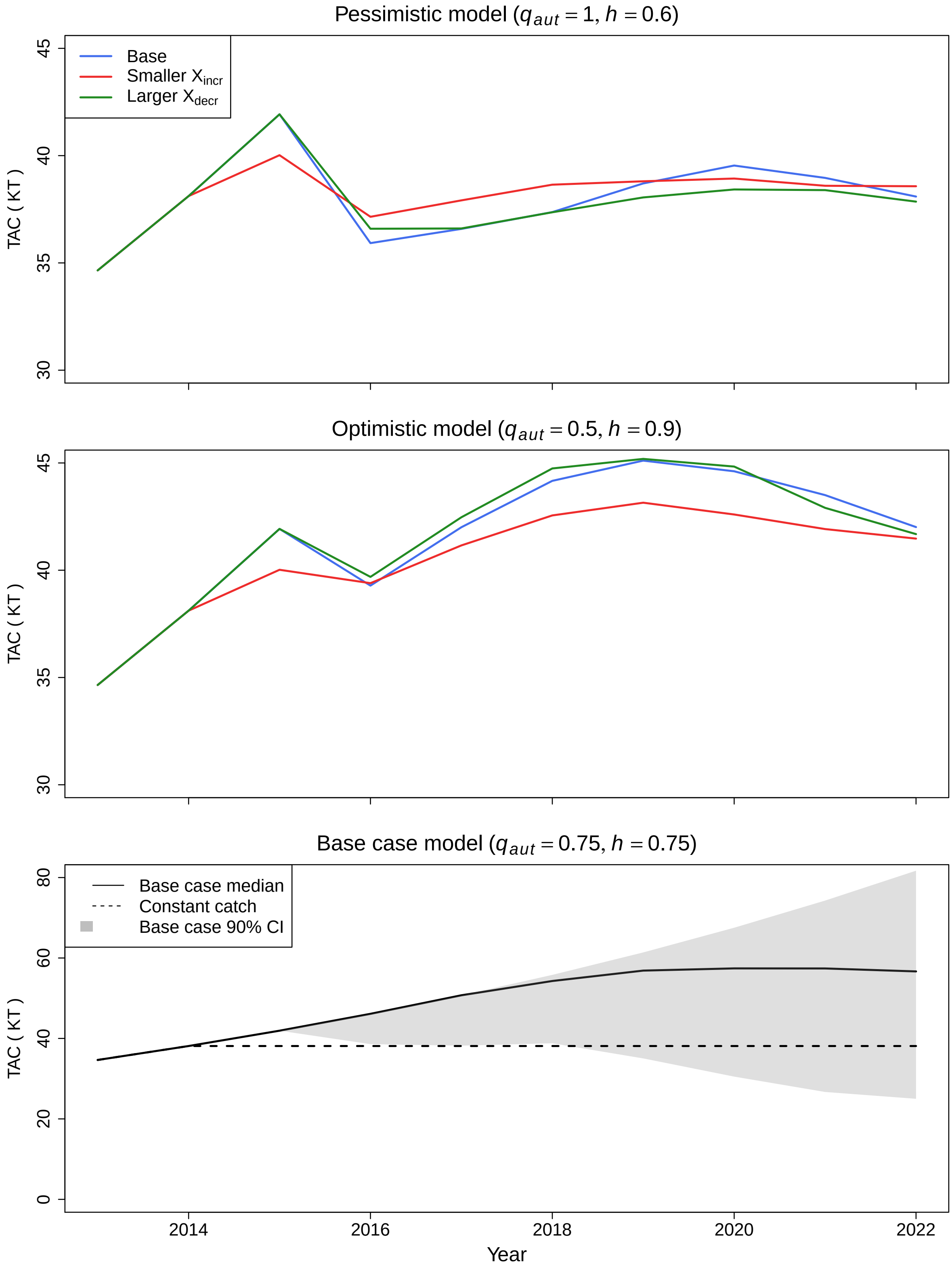

Figure 6.3: The top two plots show projected median annual midwater TACs under the pessimistic and optimistic scenarios for all candidate TAC rules. Note that the vertical axis for these two plots has been expanded for clearer differentiation of the results for the different rules. The bottom plot compares medians and 90% probability envelopes of projected annual midwater TACs for the *base* and *constant catch* TAC rules for the *base case* OM. Random noise was added to future observed abundance indices during projections.

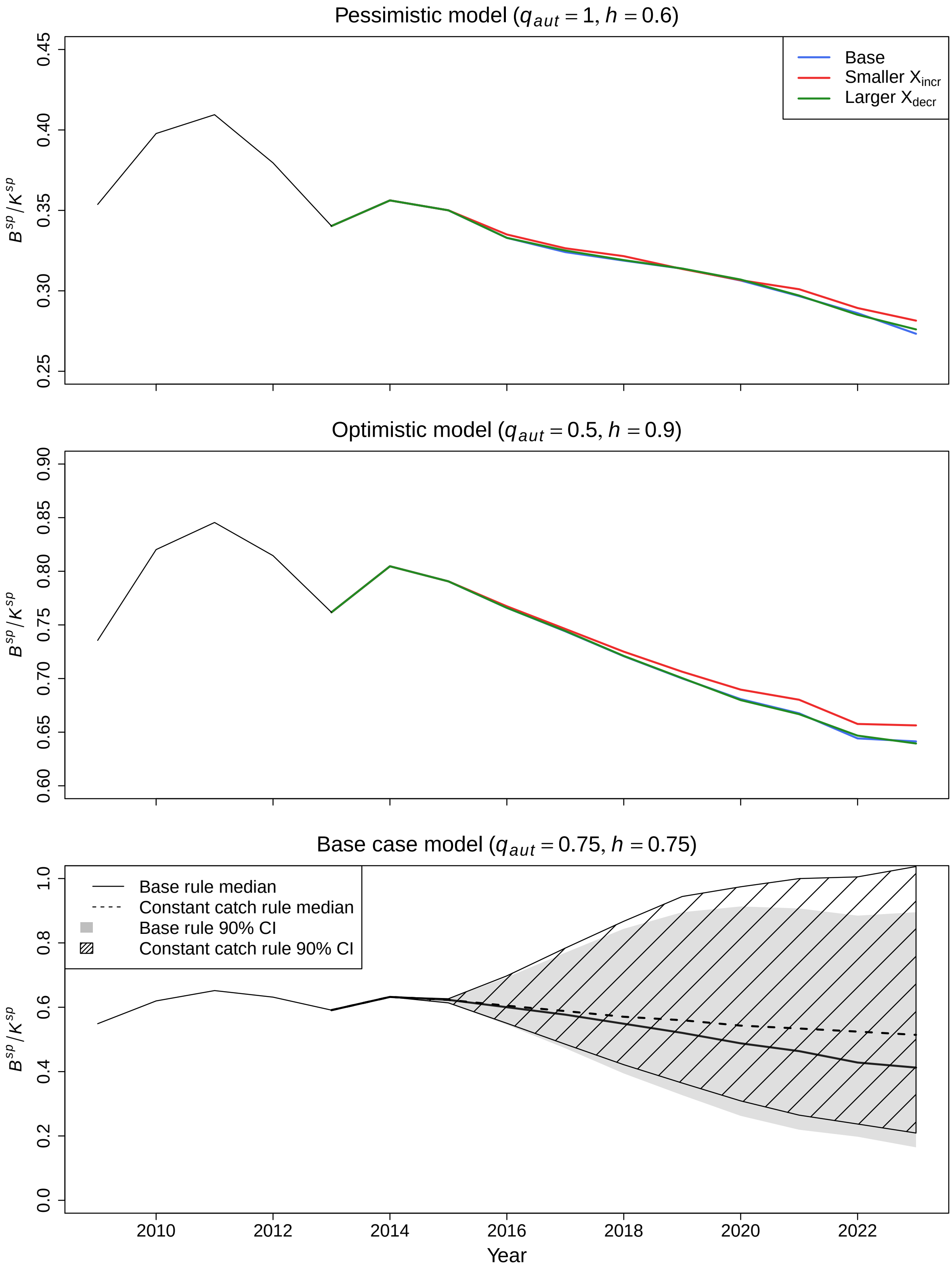

Figure 6.4: The top two plots show projected median spawning biomass under the pessimistic and optimistic scenarios for all candidate TAC rules. It is difficult to distinguish between the trajectories for the different rules because they are almost identical, even though the vertical axis for these two plots only has been expanded for clearer differentiation of the results for the different rules. The bottom plot compares medians and 90% probability envelopes of projected spawning biomass for the *base* and *constant catch* TAC rules for the *base case* OM. Random noise was added to future observed abundance indices during projections.

Chapter 7

Conclusion

This thesis has described how the MSE approach was used to develop MPs for the pelagic fishery bycatch of, and the directed midwater fishery for, horse mackerel. A summary is given below of the key findings of the individual chapters, as well as the management decisions that have been based upon them.

The four assessment models that form the Reference Set of OMs for MP testing were detailed in Chapter 4. Results demonstrated that they fit the observed horse mackerel data satisfactorily, but there is some evidence of systematic errors in the fits to the catch-at-length data. To partly alleviate this, length-specific fishing selectivity for the demersal fleet was allowed to vary with time; however, model-estimated abundance was shown to be very sensitive to the method chosen to normalise this time-varying selectivity. Nevertheless, all indications are that the horse mackerel resource is healthy, with even the most pessimistic model in the Reference Set estimating that spawning biomass is currently above the MSYL. Given the same assumptions regarding future pelagic bycatches, projections closely matched those of Johnston and Butterworth (2007). They also confirmed the analysis of Horsten (1999b), which showed that even small pelagic bycatches of juveniles have a pronounced negative effect on the level of sustainable adult catch that is possible.

Chapter 5 described the evaluation of CMPs to determine the annual PUCL for horse mackerel in the small pelagics fishery. The aim was to provide sufficient flexibility in annual allocations so as to prevent early closures of the fishery in years with high levels of horse mackerel bycatch, without placing undue stress on the resource. Analyses showed that horse mackerel biomass estimates based on the November pelagic surveys on the West Coast may have some value as predictors of recruitment. Furthermore, during simulation testing, MPs that used those survey results as inputs performed better than those that did not; however, these MPs were deemed infeasible because they did not permit any juvenile horse mackerel bycatch if surveys indicated poor recruitment. When a constraint was placed on those MPs to limit the minimum PUCL, their performances degraded so as to be similar to those of the other CMPs. The next best MPs were the so-called $PUCL_3$ rules, which limit the bycatch over any consecutive three year period. They performed better than a fixed annual PUCL in terms of allowing larger average annual bycatches; however, they were not able to adequately reduce the probability of early closures disrupting the pelagic fishery (though see also comments in Section 7.1.5 below).

Chapter 6 explained the process undertaken to evaluate the CMPs that set the annual TAC for the directed midwater fishery for horse mackerel. The extent of the negative bias in absolute horse mackerel abundance estimates based on demersal surveys is unknown. Therefore, it was hoped that an MP could be found that experimentally increases the allocation to the directed midwater fishery without increasing the risk of resource depletion. All MPs considered incorporated an index of abundance that is related to a weighted average of the last three years of autumn demersal survey and CPUE data. In simulation testing, all provided appreciable increases in average catch over the current TAC with no deterioration in terms of risk-related performance. In many cases they operated as anticipated, according larger increases in TAC for the OMs in the Reference Set that reflect more optimistic outlooks for the resource. The CMPs that were presented to stake-holders for selection offered various trade-offs in terms of the mean annual catch against the extent of interannual variation in TAC.

In 2012, preliminary versions of the analyses described in this thesis were presented to DAFF DSWG (Furman and Butterworth, 2012). Based on the initial results, it was decided to implement the $PUCL_3$ *no reserve* rule to manage bycatches of horse mackerel in the pelagic fishery, and the *base* TAC rule (i.e. $X_{decr} = 15\%$ and $X_{incr} = 10\%$) to manage the directed midwater fishery. Parameter values selected for the MPs were $PUCL_3 = 18\,000$ tonnes, $I_{decr} = 0.55$ and $I_{incr} = 1.26$. Subsequently, two years' further data have been incorporated into the assessment and improvements have been made to the OMs, as reported in the preceding chapters. Therefore this study currently presents the most up-to-date view of the resource. In 2014, DAFF altered the control parameters of the horse mackerel MPs to be in line with the management recommendations set out above; $PUCL_3$ was reduced to 15 589 tonnes, and I_{decr} and I_{incr} were adjusted to 0.84 and 1.01 respectively.

Nevertheless, many key uncertainties remain. The following sections list these problems and suggest possible solutions to ensure better assessment and management of the resource.

7.1 Future work

7.1.1 Measure bias in demersal swept-area surveys

It is evident throughout this thesis that the value assumed for the autumn demersal survey catchability coefficient q_{aut} (which is a measure of the assumed bias in the corresponding abundance estimates) has a large effect on results. For example, if the OM with $q_{aut} = 1$ were removed from the Reference Set and the MPs were instead tuned to give the same risk-related performance for what would then be the most pessimistic model (Model 3 with $q_{aut} = 0.5$ and $h = 0.6$), simulation tests show that no PUCL would be necessary and the *base* TAC rule for the midwater fishery could allow for an average annual catch of approximately 120 000 tonnes rather than the currently projected average annual catch of 43 400 tonnes. This highlights the potential value of research on the bias in these survey estimates of abundance. Smith *et al.* (2011) suggest that analyses of concurrent trawl and acoustic surveys could be used to estimate the proportion of the stock not available to the demersal trawls. Plans by DAFF to further such studies have yet to be implemented due to budgetary constraints and repairs needed to DAFF's main research vessel (Durholtz, 2013).

7.1.2 Adopt a Bayesian approach

The maximum likelihood estimation method on which this horse mackerel assessment is based is usually attempted before a Bayesian method because it is relatively simple to implement; however, it has some drawbacks. Estimates of precision in model parameter values presented in this thesis are Hessian-based and therefore only approximate, while a Bayesian approach would facilitate taking fuller account of uncertainty related to model structure and parameter values (Punt and Hilborn, 2001). A major benefit of the Bayesian approach is that it provides a scientifically defensible basis for incorporating prior information into the OMs. Input values for some parameters were admittedly somewhat arbitrary, and the inclusion of priors would allow existing knowledge from a variety of sources to be taken into account. For example, data from the RAM II database and results from assessments of other similar stocks could be used to inform steepness h, natural mortality M (to which the model results showed some sensitivity) and recruitment variability σ_r (Smith *et al.*, 2011).

7.1.3 Improve the model fits to the catch-at-length data

At the moment there is some evidence of misspecification in the model fits to the commercial midwater and demersal survey catch-at-length data (Figure 4.3). Furthermore, the choices of periods for which different demersal selectivity functions are assumed are somewhat arbitrary. Given the sensitivity that the model results showed to the method used to normalise time-varying selectivity, it would be worthwhile to explore other forms of selectivity functions (e.g. piecewise-linear functions) and approaches to modelling time-varying selectivity. Additionally, the possibility of asymptotic selectivity for the demersal and midwater fleets should be further investigated as these fleets target large fish.

7.1.4 Improve modelling of future bycatches

In an effort to model future horse mackerel bycatches in the pelagic fishery, relationships between past bycatches, and model-estimated recruitment and biomass were investigated. However, the correlations were found to be weak. Therefore, bycatches are currently modelled by sampling them with replacement from the series of historical bycatches. Alternative approaches to modelling these bycatches should be examined. For example, Smith *et al.* (2011) suggested that the magnitude of bycatches may be related to the juvenile abundance estimates for horse mackerel derived from the November pelagic surveys.

7.1.5 Evaluate alternate CMPs

Many CMPs did not behave satisfactorily in all situations, and further alternatives should therefore be considered. It was hoped that the candidate PUCL rules would show an ability to decrease the amount of disruptions to the small pelagics fishery caused by closures as a result of reaching horse mackerel limits; however, they seemed to offer little improvement in this regard (Table 5.3). It is currently unclear whether this is due to an inappropriate definition of the associated performance statistic *probability of disruption*—it is a discontinuous function and small changes may not be reflected—or rather inadequate design of the CMPs. That is not to say that the $PUCL_3$ *no reserve* rule that has been implemented is ineffective. Flexibility has been introduced in the annual pelagic fishery allocation, and basing choices on a bycatch scaling factor set equal to two (i.e. $k = 2$) is taking a pessimistic view of what is likely to happen in the future. Nevertheless, further simulation trials to investigate the performance of other rules are necessary: for example, rules that incorporate survey results from May pelagic recruit surveys or November pelagic surveys of the entire South African coast, instead of results from November pelagic surveys of the West Coast only (Smith *et al.*, 2011). Also, an approach similar to that used in the management of South African sardine could be considered, in which the expected bycatch of juvenile sardine is related to the anchovy TAC (De Oliveira and Butterworth, 2004). However, joint management of the horse mackerel and sardine resources would likely introduce considerable complications given that sardine assessments are already combined with those for anchovy for joint management of those two species.

The performance of the CMPs for the midwater fishery were also not entirely satisfactory. They would often "overshoot" the ideal TAC and self-correct by decreasing catches towards the end of the projection period (Figure 6.3). Other combinations of the control parameters X_{decr} and X_{incr} should be considered, as well as different types of MPs including model-based approaches.

7.1.6 Conduct robustness trials for CMPs

The sensitivity of the underlying population models to various assumptions was investigated in Chapter 4. However, owing to time constraints and the urgency with which DAFF needed control rules adopted, no robustness trials were conducted. Ideally, they should play the role of "tick tests", ensuring that the performances of CMPs do not deteriorate appreciably for any of these trials (Rademeyer *et al.*, 2007). The sensitivity tests in the aforementioned chapter provide an indication of some uncertainties that should be included in the set of robustness trials. Additionally, they should explore the consequences of the anticipated autumn demersal survey data not becoming available in time to be used as inputs in the candidate TAC rules.

7.1.7 Miscellaneous

- South African horse mackerel were taken as bycatch from the 1900s; however, catches were recorded only from 1949. Therefore, the missing historical catches should be reconstructed to the extent possible along with an assessment of the uncertainty around these catches.
- Data from several non-standard demersal surveys were not incorporated in the assessment model. In the future, they could perhaps be included if GLM standardisation is used to take changes in survey vessel, depth and gear into account.

- All catch-at-length data were compiled into bins of 5 cm; however, this may be too coarse, resulting in the loss of useful information. Although it is unlikely to make an appreciable difference to the core results, the use of smaller bins should be explored.

- A small constant was added to all CPUE values in the commercial midwater dataset to allow the log-normal GLM to run despite zero CPUE values; however, this solution is somewhat arbitrary and can lead to highly non-normal residual plots. It may be preferable to avoid these problems by fitting two compound models to these data, i.e. different GLMs for zero and for non-zero catches.

- Currently horse mackerel catch-at-length data from the pelagic fishery are not available. If possible, it would be worthwhile to start collecting these data to help models better reflect the bycatches in that fishery.

- The periods chosen for time-varying selectivity for the demersal surveys are somewhat arbitrary (Section 4.B.1). A possible alternative to investigate would involve estimating different selectivity functions for each gear type that was used in those surveys.

- Currently, effective weights-at-age for the demersal and midwater fleets are affected by their fishing selectivities (Equation 4.A.8). However, effective weights-at-age for the pelagic fleet does not take fishing selectivity into account; this should be rectified in the interests of consistency.

7.2 Lessons learnt from these applications of the Management Strategy Evaluation approach

The work presented in this thesis used the MSE approach. Instead of management recommendations being based on the results of a single best assessment, four OMs encapsulating what are considered to be the key uncertainties for the horse mackerel resource comprised the Reference Set. Sensitivity tests were then conducted in order to investigate the impact of other uncertainties. Next, a variety of CMPs for the pelagic and midwater fisheries were simulation tested on this Reference Set. The resulting performance statistics were then presented to stake-holders to allow them to make informed decisions on which CMPs to implement. Several important lessons that were learnt during this process are discussed below.

Acquiring the necessary data to complete a thorough assessment of a resource is seldom a simple task, and sufficient time should always be allowed for their compilation. This is made especially difficult when the data in question spans a long period and therefore falls under the jurisdiction of changing management organisations. Even then, other problems can arise. If data are not properly archived and backed-up, then there is always the danger that they may be lost (as was the case with the pre-1997 pelagic hydro-acoustic survey data for horse mackerel). Also, there can be conflicting reports of historical observations, as was the case for the combined demersal and midwater catch data for the period 1988–1999, which are almost impossible to resolve and add an additional element of uncertainty to model results. Finally, it cannot plausibly be assumed for the purposes of simulation testing that anticipated future data will be available indefinitely. For example, demersal surveys anticipated in 2012 and 2013 could not be conducted because of mechanical and other problems with the research vessel.

The decision of which OMs to include in the Reference Set is critically important as this can affect the values of performance statistics profoundly. In Chapters 5 and 6 a "worst case scenario" method was adopted regarding *risk of depletion* by defining this statistic to reflect the risk-related performance for the most pessimistic model only. While this conservative approach was taken to ensure the safety of the resource, it could result in large losses in potential catches. Butterworth *et al.* (1996) argue that performance statistics should be weighted averages over the different scenarios hypothesised, where weights are proportional to the relative plausibilities of each scenario. However, in practice it is difficult to determine these weightings. Robustness trials provide a means of checking that CMPs perform satisfactorily in all foreseeable scenarios. Unfortunately, unavoidable time constraints did not allow for these tests to be conducted for this work.

Fisheries science is a highly practical field and the MSE process will necessarily be scrutinised by managers and industry stake-holders who are non-specialists. It is important to achieve buy-in from these decision makers so as to get continued support even if recommendations are detrimental to industrial interests in the short term, for example when TAC reductions are required. One way to do this is to involve the stake-holders as much as possible (Punt and Donovan, 2007). For example, they could be asked to help define performance statistics that are of interest to them. Additionally, these indicators of performance should be presented in such a way as to render them easily understood by those who are not well-versed in statistics. It was particularly difficult to report estimates of uncertainty and risk-related performance clearly. Finally, it is critical that scientists do not undermine the scientific process and inadvertently assume management responsibilities by defining objectives and performance statistics without consulting stake-holders.

An unexpected by-product of the MSE process is that it highlighted the value of further research into the bias in swept-area demersal survey abundance estimates. Furthermore, it was even possible to give an approximate estimate of the potential gain in annual catches from this research. It is conceivable that if a robustness test for a particular scenario necessitated the adoption of a conservative MP, then if further research could disprove the hypothesis underlying that scenario, a less conservative MP could be implemented instead. In this way, the MSE approach can help to rank the importance of different research projects. Butterworth and Punt (1999) describe how the Australian Fisheries Management Agency agreed to base research prioritisation on the results from MP development processes.

Butterworth *et al.* (1997) argue that the MSE approach is more cost-effective than the conventional "best assessment" approach, because time is saved debating the typically annual assessment exercises. This was evident in the management of the horse mackerel resource. A great deal of time was indeed spent constructing OMs and evaluating CMPs, all the while consulting with decision makers during frequent SWGs. However, once a consensus had been reached on which MPs to implement, it was a relatively simple task to compile inputs required for those MPs and present the resulting management recommendations, which were endorsed with little debate. Unless considerable changes are made to the assessment model or MPs in the near future, these time-savings should continue. Of course, sometimes the situation does not allow for a thorough MSE process to be completed, as was case when *ad-hoc* increases to the PUCL were requested by industry in 2011: the pelagic fishery was facing early closures and swift measures had to be taken in order to prevent the loss of potential profits. Therefore, it is important to find a balance between thoroughness and urgency. This difficultly is exacerbated in South Africa where resources (both human and financial) are limited, and in practice fisheries must be prioritised according to their socio-economic importance.

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