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2017/0824 June 2020



Stock Assessment for the Southern and Eastern Scalefish and Shark Fishery: 2018 and 2019



PART
1

2018



Principal investigator **G.N. Tuck**



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Cover photographs

Front cover, jackass morwong, orange roughy, blue grenadier, and flathead.

Report structure

Part 1 of this report describes the Tier 1 assessments of 2018. Part 2 describes the Tier 3 and Tier 4 assessments, catch rate standardisations and other work contributing to the assessment and management of SESSF stocks in 2018.



Stock Assessment for the Southern and Eastern Scalefish and Shark Fishery 2018 and 2019

Part 1: 2018

G.N. Tuck
June 2020
Report 2017/0824

Australian Fisheries Management Authority

Stock Assessment for the Southern and Eastern Scalefish and Shark Fishery: 2018

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7. Western Jackass Morwong (*Nemadactylus macropterus*) stock assessment based on data up to 2017 – development of a preliminary base case

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7.1 Executive Summary

This document presents a suggested base case for an updated quantitative Tier 1 assessment of jackass morwong (*Nemadactylus macropterus*) for presentation at the first SERAG meeting in 2018. The last full assessment was presented in Tuck et al. (2015). The preliminary base case has been updated by the inclusion of data up to the end of 2017, which entails an additional three years of catch, discard, CPUE, length-composition and conditional age-at length data and updates to the ageing error matrices since the 2015 assessment. One additional abundance index (2016) for the Fishery Independent Survey (FIS) was included. This document describes the process used to develop a preliminary base case for jackass morwong through the sequential updating of recent data to the stock assessment, using the stock assessment package Stock Synthesis (SS-V3.30.12).

Changes to the last stock assessment include: incorporation of discard estimates and estimation of retention for the trawl fleet; and using an updated tuning method.

Results show poor fits to the abundance data (catch rate and FIS), but acceptable fits to the length composition and conditional age-at-length data. This assessment estimates that the projected 2019 spawning stock biomass will be 69% of virgin stock biomass (projected assuming 2017 catches in 2018), identical to the depletion of 69% at the start of 2016 obtained from the last assessment (Tuck et al., 2015).

7.2 Introduction

7.2.1 Bridging from 2015 to 2018 assessments

The previous full quantitative assessment for western jackass morwong was conducted during 2015 (Tuck et al., 2015) using Stock Synthesis (version SS-V3.24U, Methot and Wetzel, 2013). The 2018 assessment uses the current version of Stock Synthesis (version SS-V3.30.12, Methot et. al, 2018), which includes some changes from SS-V3.24U.

As a first step in the process of bridging to a new model, the model was translated from version SS-V3.24U (Methot and Wetzel, 2013) to version SS-V3.30.12 (Methot et. al, 2018) using the same data and model structure used in the 2015 assessment. Once this translation was complete, improved features unavailable in SS-V3.24U were incorporated into the SS-V3.30.12 assessment. These included allowing smaller lower bounds on minimum sample sizes and estimating a parameter that tunes the standard deviation to abundance indices. Following this step, the model was re-tuned using the most recent tuning protocols, thus allowing the examination of changes to both assessment practices and the tuning procedure on the previous model structure. These changes to software and

tuning practices are likely to lead to changes to key model outputs, such as the estimates of depletion and the trajectory of spawning biomass. This initial bridging phase (Bridge 1) highlights changes that have occurred since 2015 simply through changes to software and assessment practices. The subsequent bridging exercise (Bridge 2) then sequentially updates the model with new data through to 2017.

The second part of the bridging analysis includes updating historical data (up to 2014), followed by including the data from 2015-2017 into the model. These additional data included new catch, discard, CPUE, FIS abundance indices, length composition data, conditional age-at-length data, an updated ageing error matrix and an additional CPUE index (trawl). The last year of recruitment estimation was extended to 2012 (2011 in the 2015 assessment). The use of updated software and the inclusion of additional data resulted in some differences in the fits to CPUE, conditional age-at-length data and length composition data. The usual process of bridging to a new model by adding new data piecewise and analysing which components of the data could be attributed to changes in the assessment outcome was conducted with the details outlined below.

7.2.2 Update to Stock Synthesis SS-V3.30.12 and updated catch history (Bridge 1)

The 2015 jackass morwong assessment (West2015_24U) was initially translated to the most recent version of the software, Stock Synthesis version SS-V3.30.12 (West2015_30_12). Figure 7.1 shows that the differences in the assessment results from this step were minimal.

New features available in the new version of Stock Synthesis, such as allowing smaller lower bounds on minimum sample sizes and estimating additional standard deviation to abundance indices were then incorporated (West2015_30_12New), followed by retuning using the latest tuning protocol (West2015_30_12Tuned). Details of the tuning procedure used are listed in Section 7.2.2.1. Revisions to the historical catches, up to 2014, and replacing the estimated 2015 catch with the actual 2015 catch were then added to this tuned version of the 2015 model (West2015_30_12ReviseCatch). This process demonstrates the outcomes that could theoretically have been achieved with the last assessment if we had the latest, software, tuning protocols and corrected data available in 2015. This initial bridging step, Bridge 1, does not incorporate any data after 2014 or any structural changes to the assessment.

When these time series are plotted together, there are virtually no changes in the translation to SS-V3.30.12, but considerable changes when the new features were added, and further changes when the model was retuned using current model tuning protocols. Revising the catch history to 2014 had very little effect (Figure 7.2 and Figure 7.3).

The results of Bridge 1 suggest that the stock was more depleted in 2016 than the 2015 assessment indicated. This is almost entirely due to changes in parameters that can be tuned, including variances that can be estimated internally and in the tuning procedure itself, rather than changes to the data or to the software.

Fits to the trawl CPUE (Figure 7.4) and the FIS (Figure 7.5) abundance index changed a little through this process, but both of these series have properties that make it difficult to achieve good fits. The estimated recruitment series show similar broad trends with a general revision downwards from using the new features in Stock Synthesis and using the new tuning procedure (Figure 7.6).

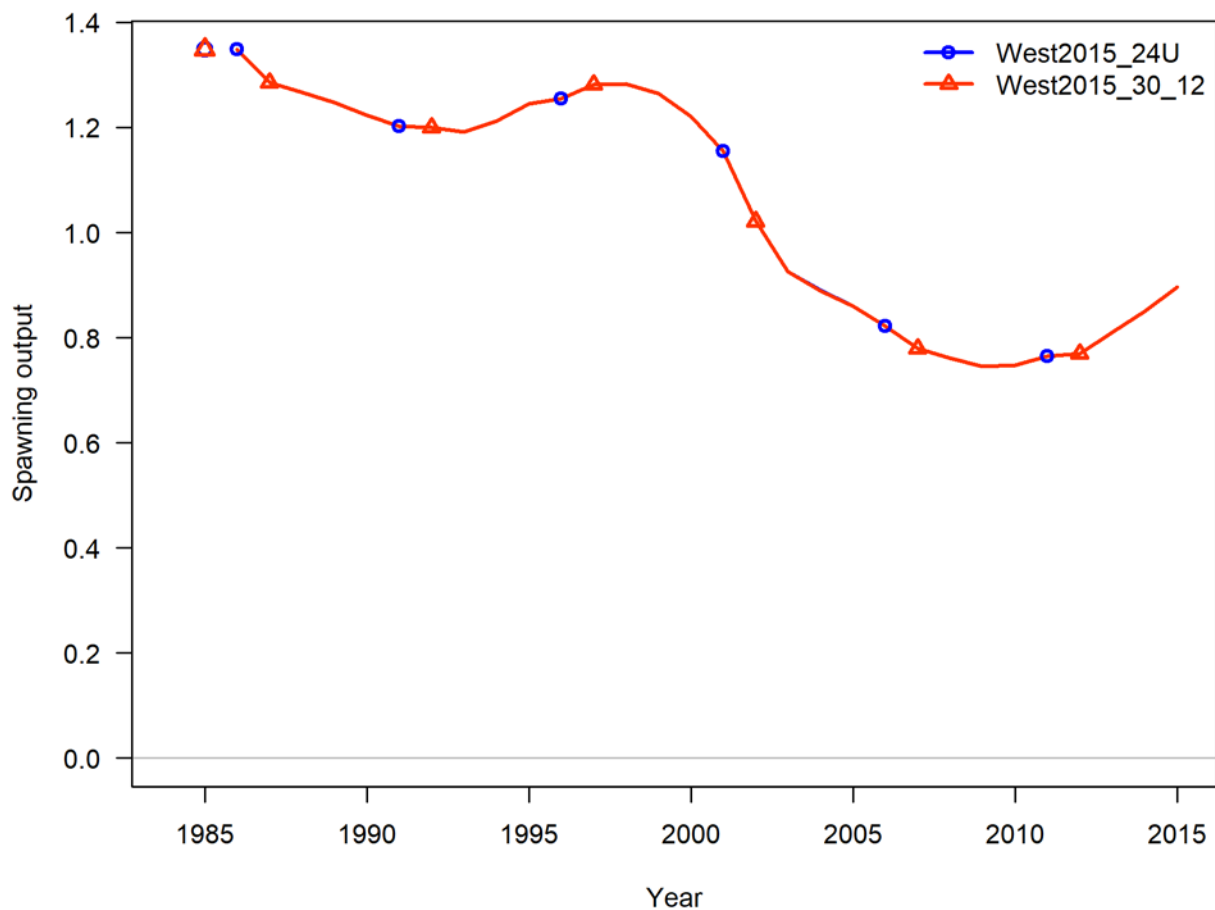


Figure 7.1. Comparison of the time-series of absolute spawning biomass from the 2015 assessment (West2015_24U – in blue), and a model with the same data converted to SS-V3.30 (West2015_30_12 – in red). This indicates very little change in the model output from updating the version of Stock Synthesis.

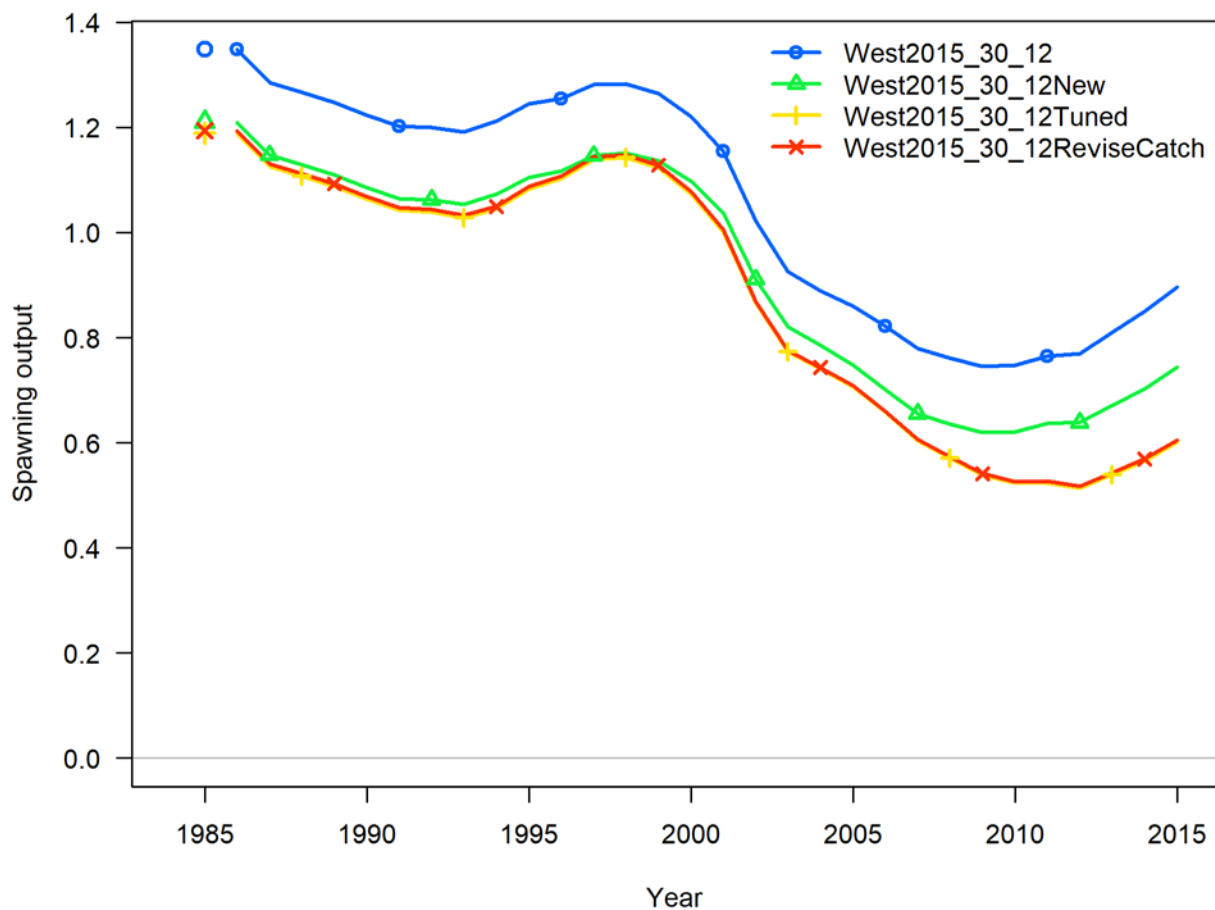


Figure 7.2. Comparison of the time-series of absolute spawning biomass from the 2015 assessment (West2015_30_12 – in blue), incorporating new features (West2015_30_12New – in green), retuning the model using the latest tuning protocols (West2015_30_12Tuned – in yellow) and revising the historical catch to 2014 and the projected catch in 2015 (West2015_30_12ReviseCatch – in red).

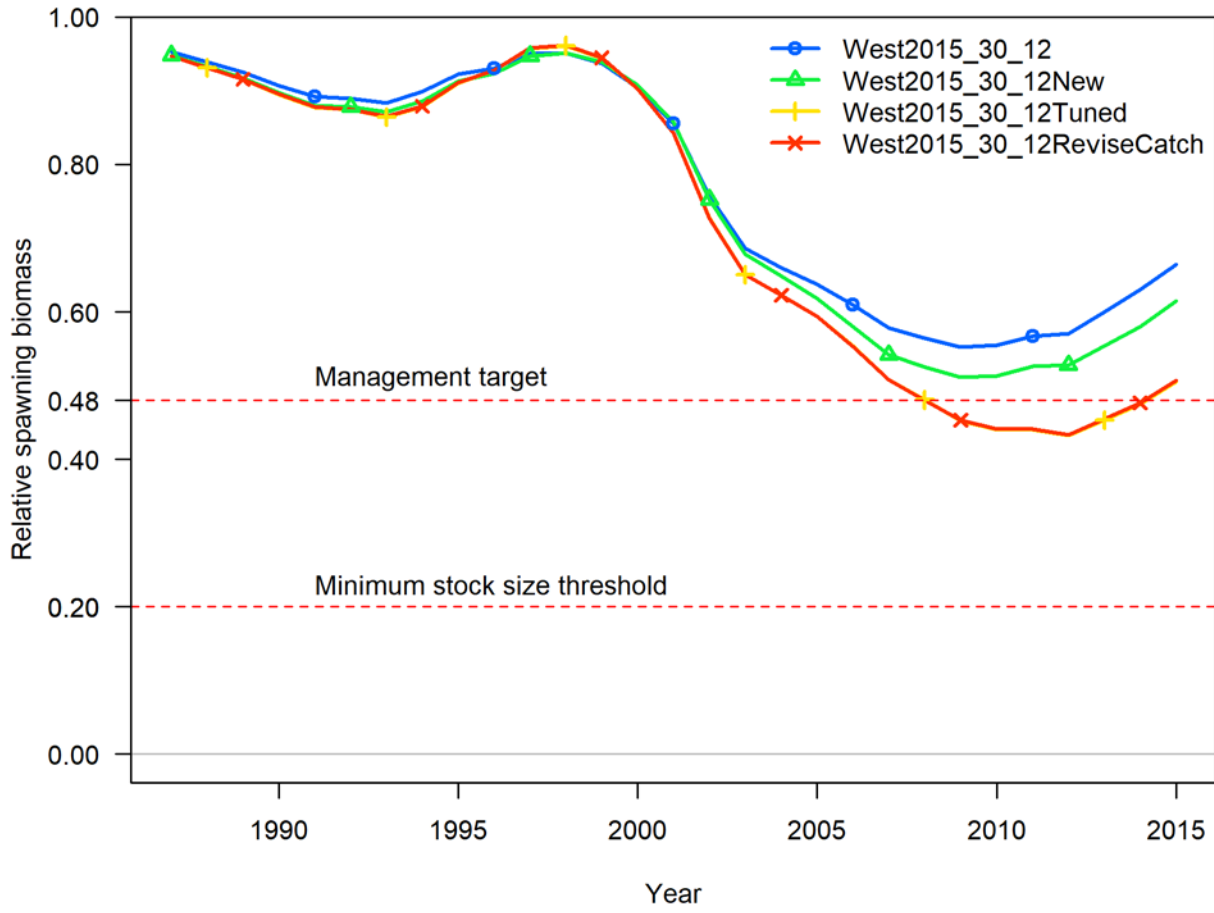


Figure 7.3. Comparison of the time-series of relative spawning biomass from the 2015 assessment (West2015_30_12 – in blue), incorporating new features (West2015_30_12New – in green), retuning the model using the latest tuning protocols (West2015_30_12Tuned – in yellow) and revising the historical catch to 2014 and the projected catch in 2015 (West2015_30_12ReviseCatch – in red).

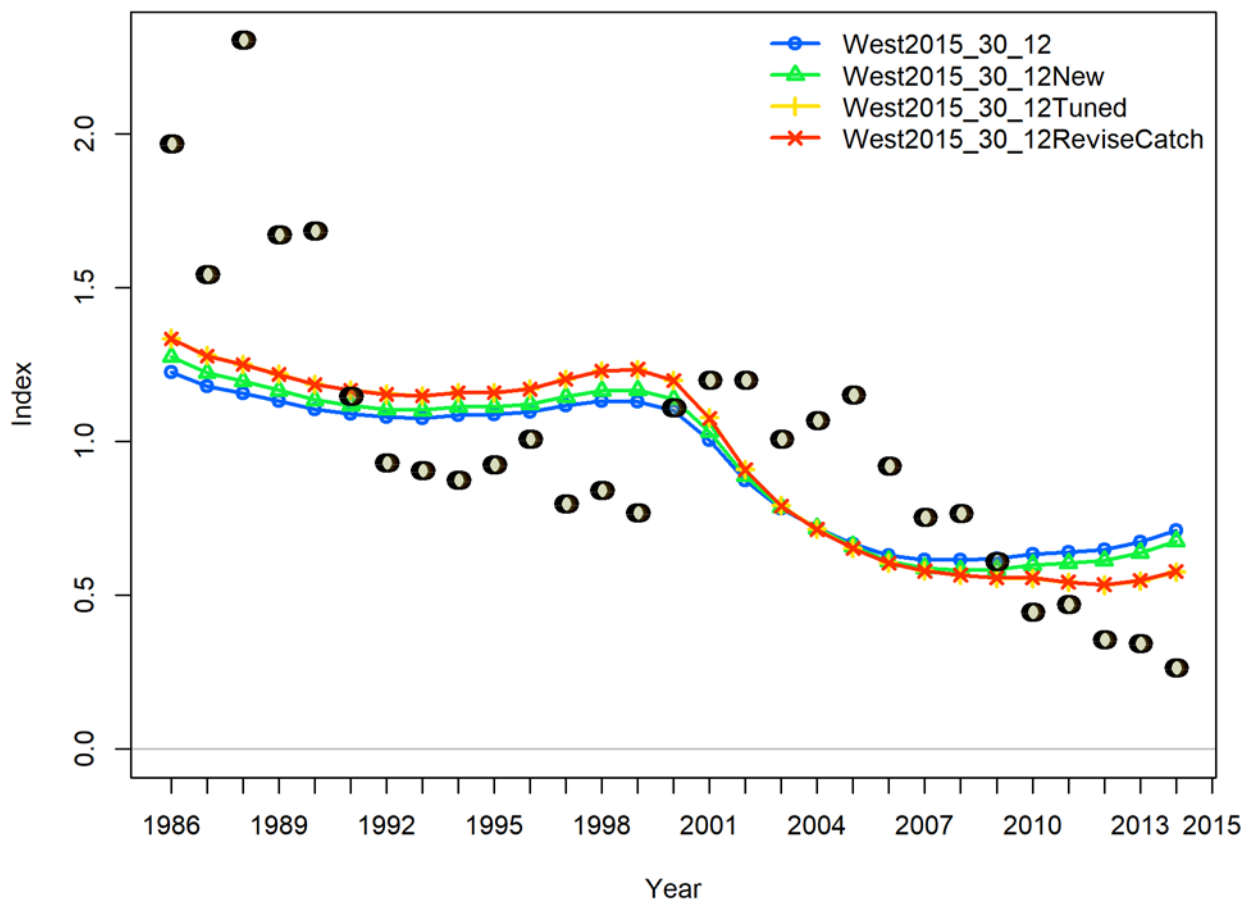


Figure 7.4. Comparison of the fit to the trawl CPUE index for the 2015 assessment (West2015_30_12 – in blue), incorporating new features (West2015_30_12New – in green), retuning the model using the latest tuning protocols (West2015_30_12Tuned – in yellow) and revising the historical catch to 2014 and the projected catch in 2015 (West2015_30_12ReviseCatch – in red).

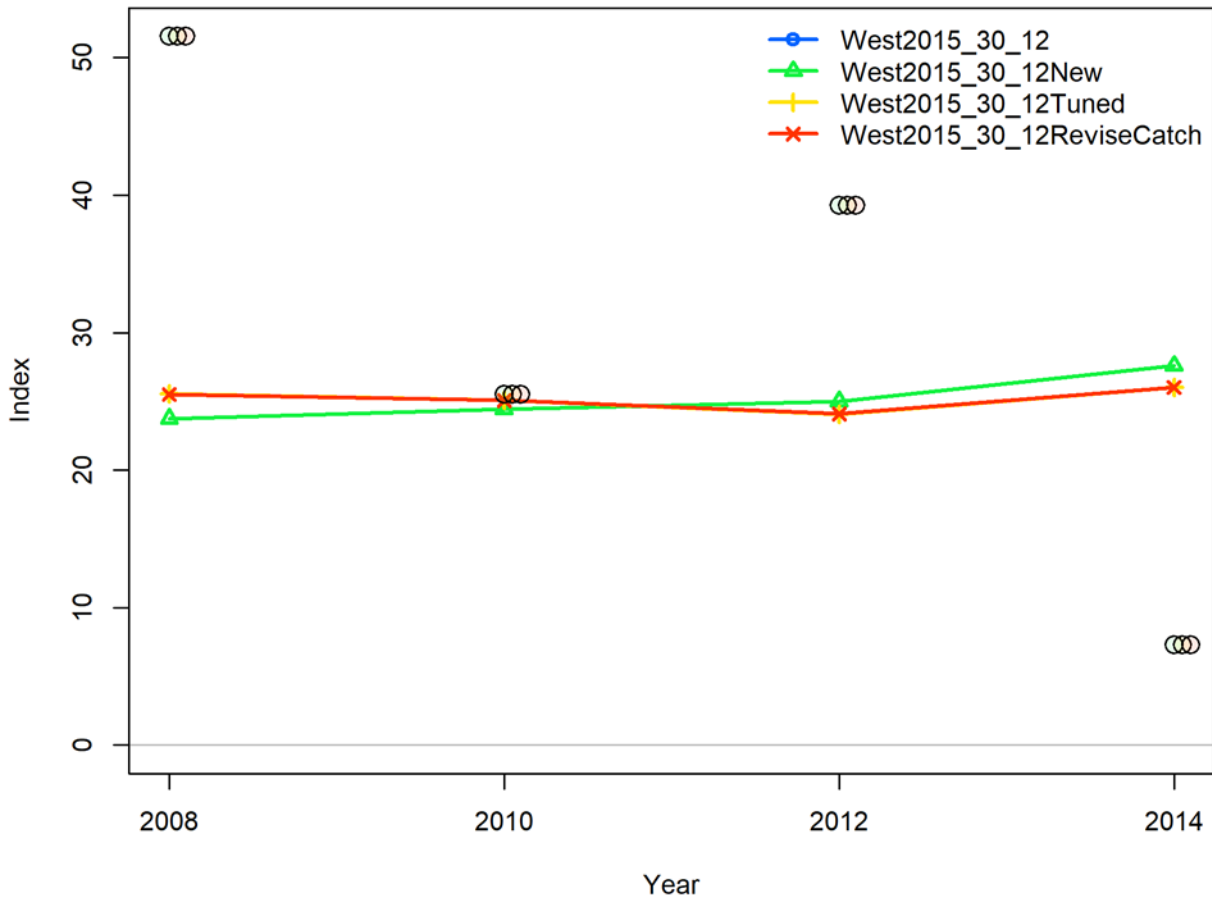


Figure 7.5. Comparison of the fit to the FIS for the 2015 assessment (West2015_30_12 – in blue), incorporating new features (West2015_30_12New – in green), retuning the model using the latest tuning protocols (West2015_30_12Tuned – in yellow) and revising the historical catch to 2014 and the projected catch in 2015 (West2015_30_12ReviseCatch – in red).

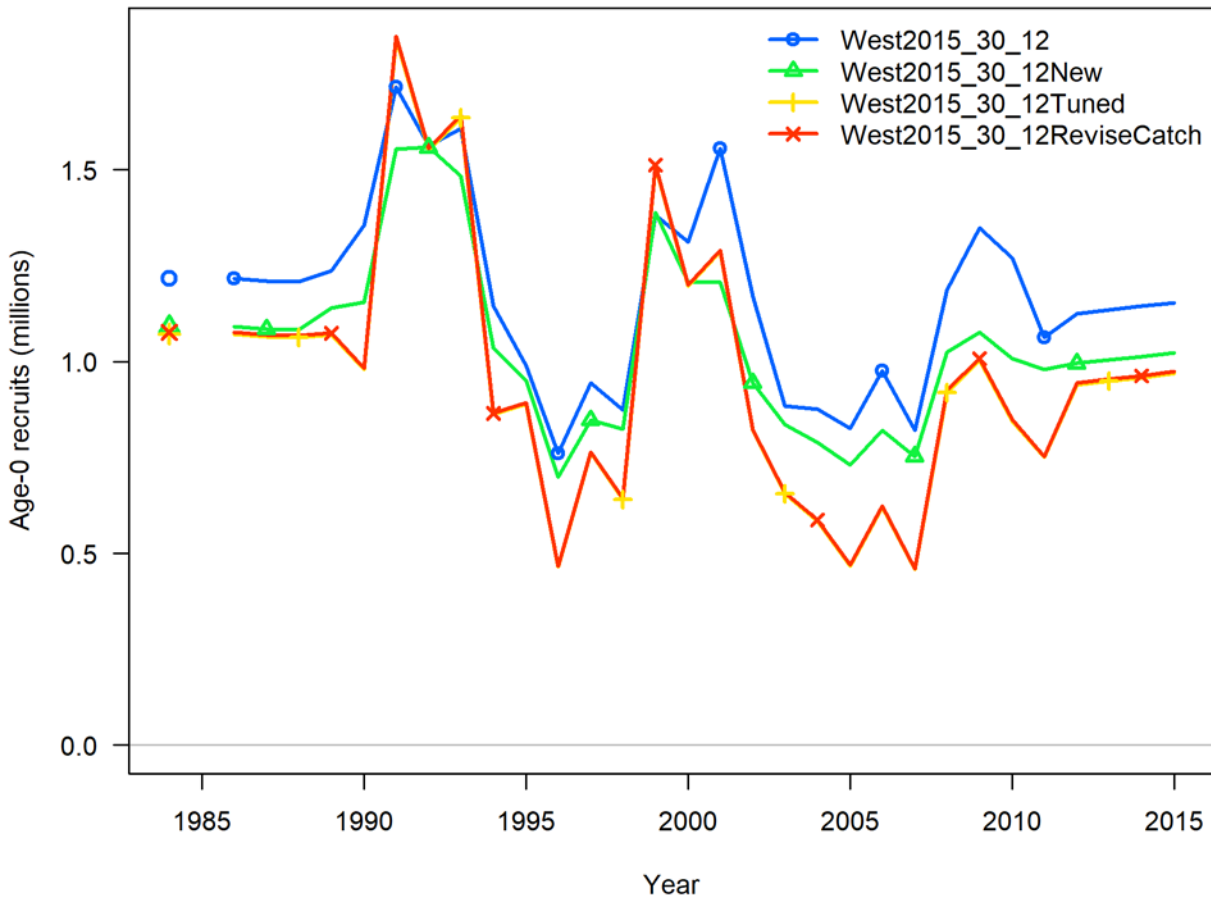


Figure 7.6. Comparison of the time series of recruitment from the 2015 assessment (West2015_30_12 – in blue), incorporating new features (West2015_30_12New – in green), retuning the model using the latest tuning protocols (West2015_30_12Tuned – in yellow) and revising the historical catch to 2014 and the projected catch in 2015 (West2015_30_12ReviseCatch – in red).

7.2.2.1 Tuning method

Iterative rescaling (reweighting) of input and output CVs or input and effective sample sizes is a repeatable method for ensuring that the expected variation of the different data streams is comparable to what is input (Pacific Fishery Management Council, 2018). Most of the indices (CPUE, surveys and composition data) used in fisheries underestimate their true variance by only reporting measurement or estimation error and not including process error.

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size is equal to the effective sample size calculated by the model. In SS-V3.30 it is possible to estimate an additional standard deviation parameter to add to the input CVs for the abundance indices (CPUE).

1. Set the standard error for the log of relative abundance indices (CPUE or FIS) to their estimated standard errors to the standard deviation of a loess curve fitted to the original data - which will provide a more realistic estimate to that obtained from the original statistical analysis. SS-V3.30 then allows an estimate to be made for an additional adjustment to the relative abundance variances appropriately.

An automated iterative tuning procedure was used for the remaining adjustments. For the recruitment bias adjustment ramps:

2. Adjust the maximum bias adjustment and the start and finish bias adjustment ramps as predicted by SS-V3.30 at each step.

For the age and length composition data:

3. Multiply the stage-1 (initial) sample sizes for the conditional age-at-length data by the sample size multipliers using the approach of Punt (2017).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length composition data using the ‘Francis method’ (Francis, 2011).
5. Repeat steps 2 - 4, until all are converged and stable (with proposed changes < 1 – 2%).

7.2.3 Inclusion of new data: 2015-2017

Starting from the translated, retuned 2015 base case model with updated data to 2014 (previously referred to as “West2015_30_12ReviseCatch”, but simplified to “West2015_30_12Updated” from here on), additional data from 2015-2017 were added sequentially to build a preliminary base case for the 2018 assessment:

1. Change final assessment year to 2017, add catch to 2017 (West2018_addCatch2017).
2. Add CPUE to 2017 (from Sporcic and Haddon (2018b)), and the FIS abundance index for 2016 (Knuckey et al 2017) (West2018_addCPUE2017).
3. Add new discard fraction estimates from 1994 to 2017 (West2018_addDiscards2017).
4. Add updated length frequency data to 2017 (West2018_addLength2017).
5. Add updated age error matrix and conditional age-at-length data to 2017 (West2018_addAge2017).
6. Change the final year for which recruitments are estimated from 2011 to 2012 (West2018_extendRec2012).
7. Retune using current tuning protocols, including Francis weighting on length-compositions and conditional age-at-length data (West2018_Tuned).

Inclusion of the new data resulted in a series of changes to the estimates of recruitment and the time-series of absolute and relative spawning biomass (Figure 7.7, Figure 7.8 and Figure 7.9), with gradual changes to these series as more data is added, with perhaps the largest change due to adding the new age data. Fits to the trawl CPUE index (Figure 7.10) and the FIS abundance index (Figure 7.11) both improve marginally as more data is added, but neither of these fits are particularly good. The trawl CPUE index jumps down suddenly in 1991 and then up in 2000, followed by a steady decline until 2014 and then a steady increase from 2015-2018. Given the longevity and catch history of jackass morwong in the west, it is impossible for the dynamics to respond quickly enough to these changes to fit the CPUE series very well. There are similar issues for the FIS abundance series, especially given the model is trying to fit both abundance series simultaneously, as well as fitting to age, length and discard data.

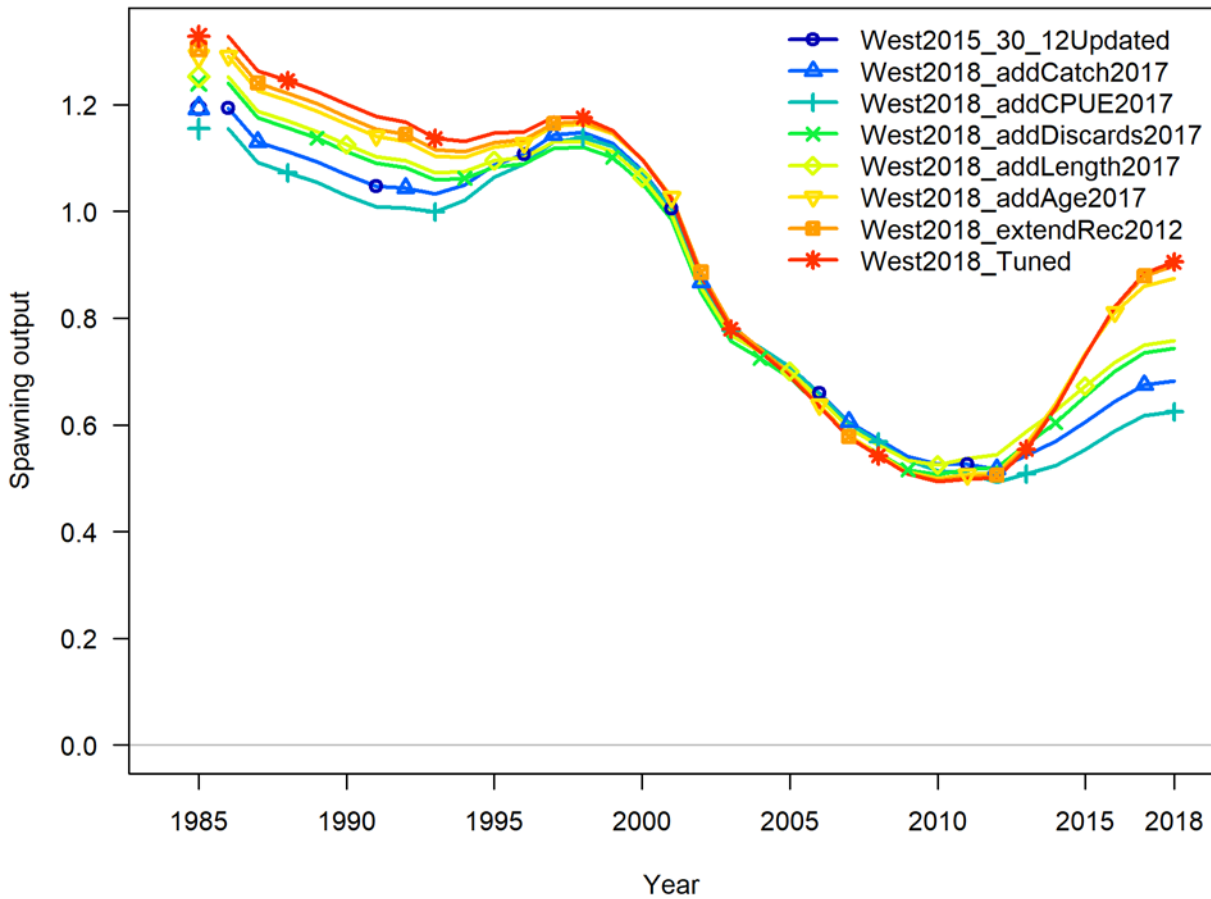


Figure 7.7. Comparison of the absolute spawning biomass time series for the updated 2015 assessment model converted to SS-V3.30.12 (West2015_30_12_updated- blue) with various bridging models leading to a proposed 2018 base case model (West2018_Tuned - red).

Since the 2015 assessment, standard changes to the procedures used in the Stock Synthesis assessments in the SESSF include:

1. Revised tuning procedures, still including use of Francis weighting for length-composition and conditional age-at-length data, but tuning the weight assigned to the CPUE series within Stock Synthesis, and
2. Improvements to how the recruitment bias ramp adjustment is calculated.

While it has been standard practice with most SESSF Stock Synthesis assessments to include discards for some years, due to data quality issues, the 2015 western jackass morwong assessment excluded discards. The assessment structure in 2018 has been changed by:

3. Including discard length compositions, and
4. Including discard estimates.

Inclusion of three years of new data resulted in relatively small changes to estimates of recruitment and the spawning biomass time series, although the time series of spawning biomass now appears to have been a little more variable, dipping below the target biomass from around 2006-2014, but then recovering strongly since 2014, with reduced fishing pressure and good recent recruitment.

Recruitment was only able to be estimated for one additional year, despite using three more years of additional data, with upward revisions to the recruitment estimates from 2010 and 2011 and above average recruitment estimated for 2012. These latest recruitment estimates may be further revised with the inclusion of additional data in future assessments, with new data that may help inform these recruitment estimates. The 2015 assessment estimated the depletion at the start of 2016 at 69%. The 2018 provisional base case has an estimate of depletion at the start of 2019 (projected assuming 2017 catches in 2018) of 69% of unexploited stock biomass, SSB_0 . The female equilibrium spawning biomass in 1986 is estimated to be 1,328 t (reduced from 1,501 t from the 2015 assessment) and in 2019 the female spawning biomass is projected to be 918t.

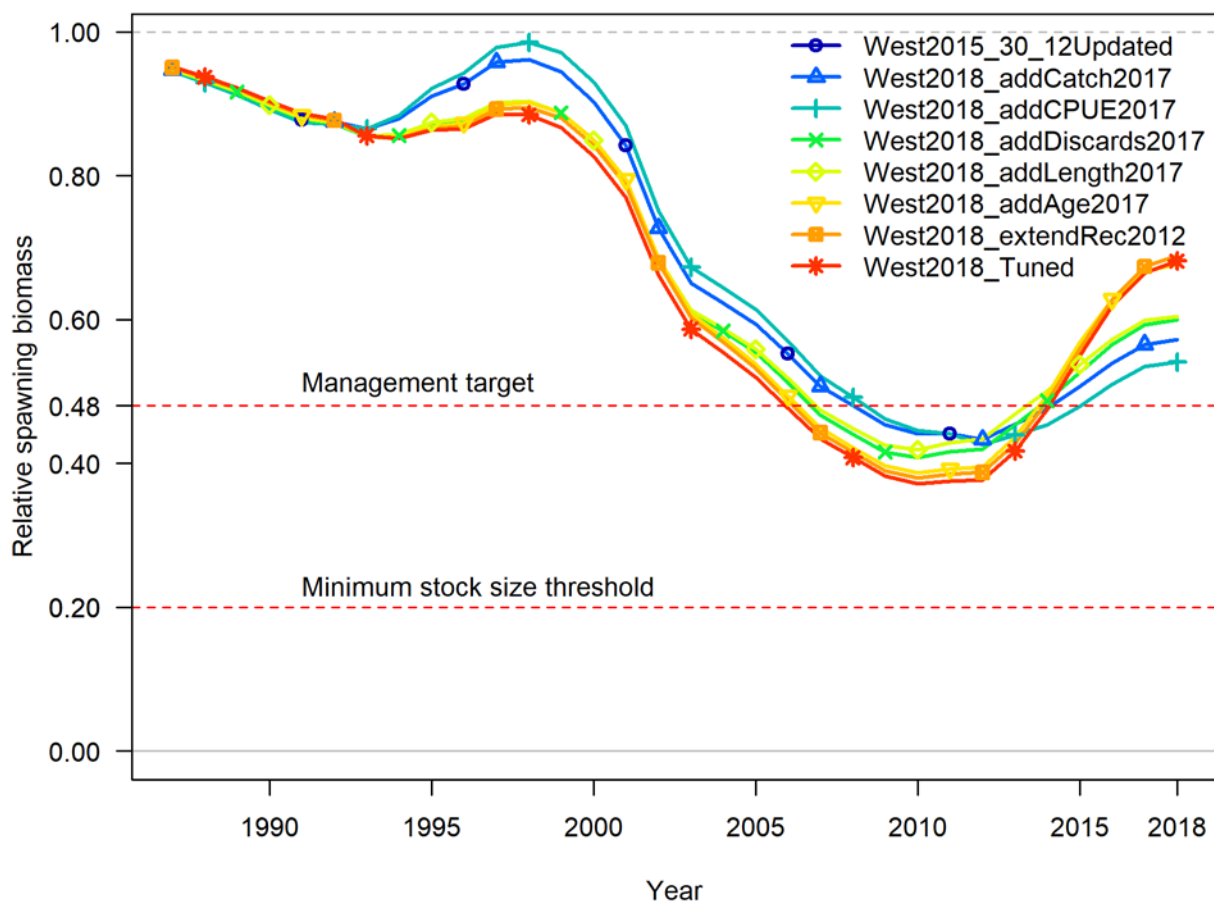


Figure 7.8. Comparison of the time series of relative spawning biomass for the updated 2015 assessment model converted to SS-V3.30.12 (West2015_30_12_updated - blue) with various bridging models leading to a proposed 2018 base case model (West2018_Tuned - red).

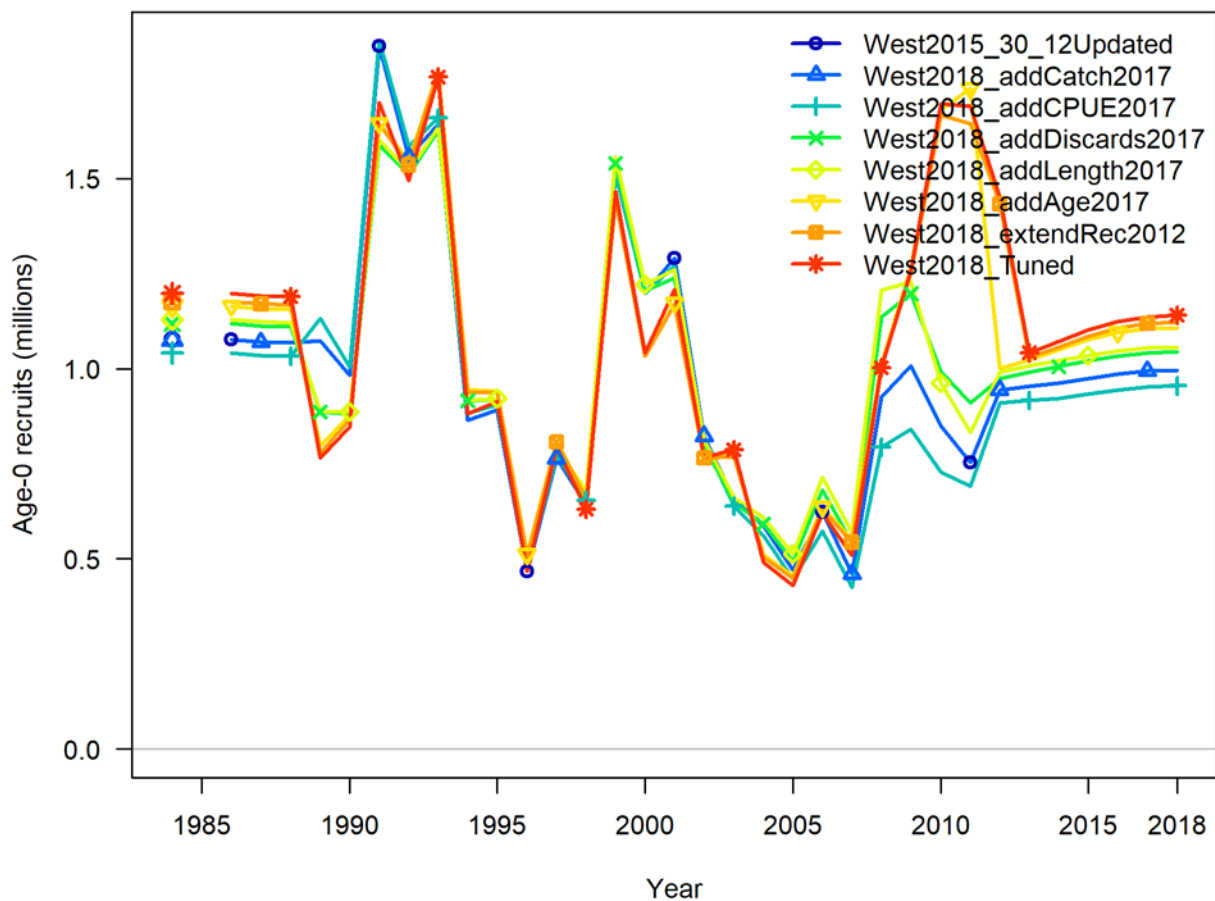


Figure 7.9. Comparison of the time series of recruitment from the updated 2015 assessment model converted to SS-V3.30.12 (West2015_30_12_updated- blue) with various bridging models leading to a proposed 2018 base case model (West2018_Tuned - red).

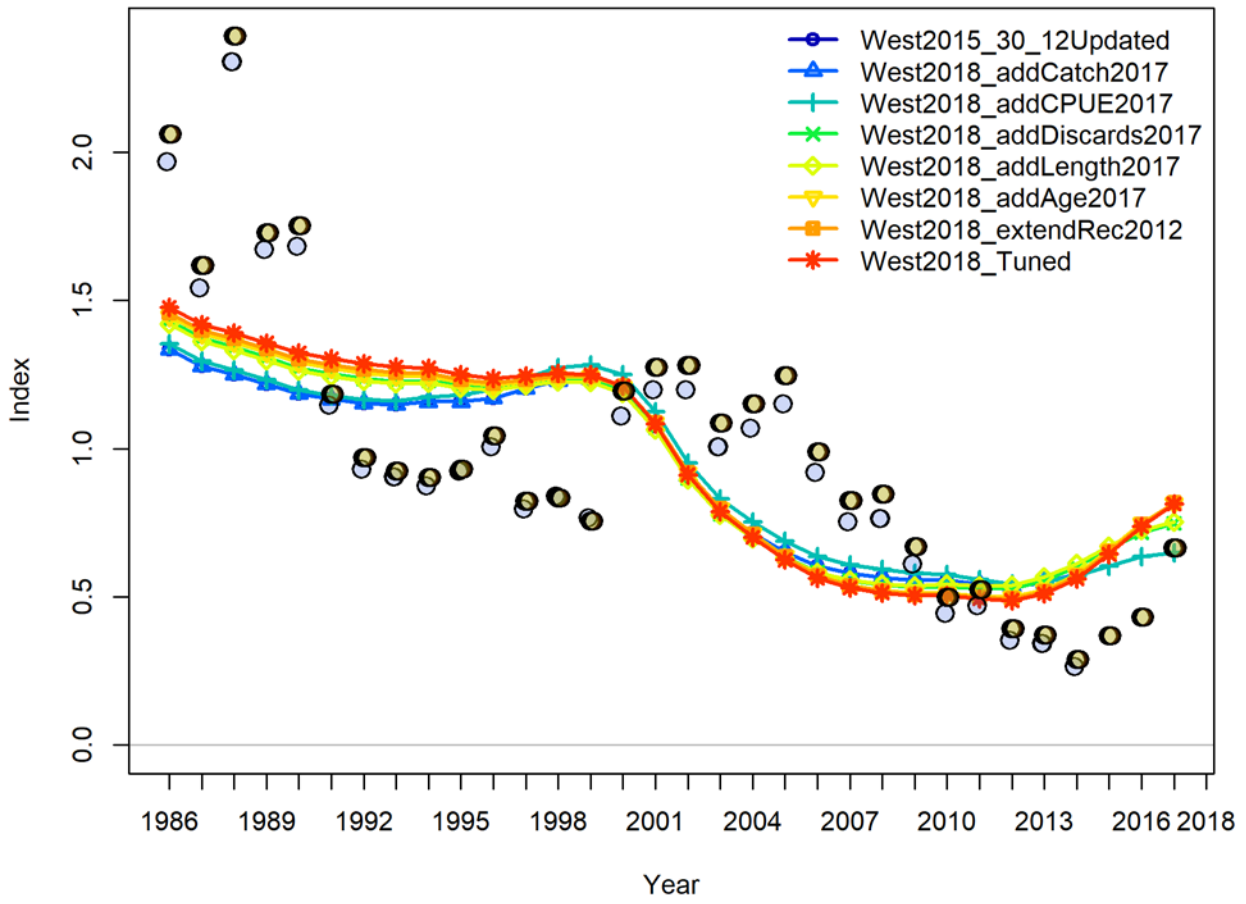


Figure 7.10. Comparison of the fit to the trawl CPUE index for the updated 2015 assessment model converted to SS-V3.30.12 (West2015_30_12_updated- blue) with various bridging models leading to a proposed 2018 base case model (West2018_Tuned - red).

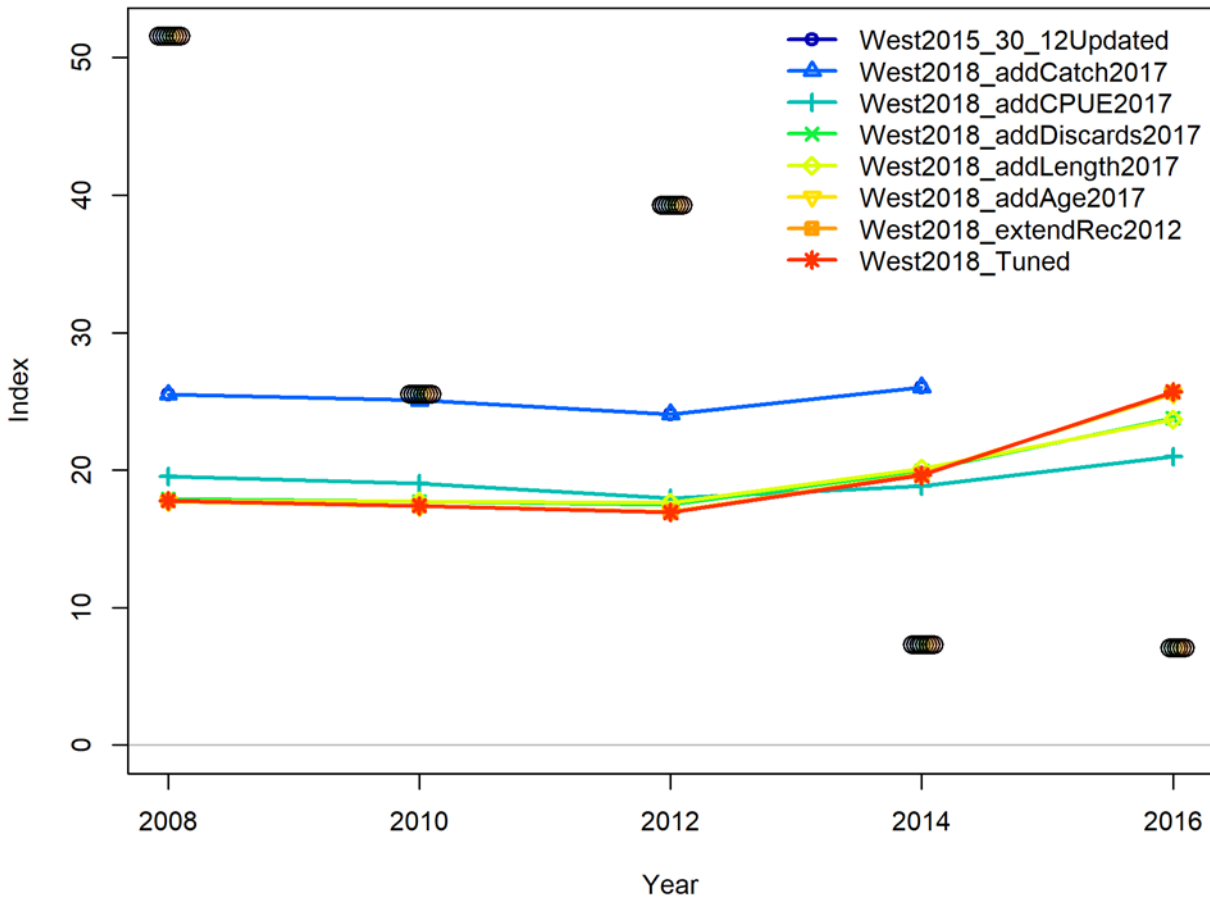


Figure 7.11. Comparison of the fit to the FIS for the updated 2015 assessment model converted to SS-V3.30.12 (West2015_30_12_updated- blue) with various bridging models leading to a proposed 2018 base case model (West2018_Tuned - red).

7.2.4 Likelihood profiles

As stated by Punt (2018), likelihood profiles are a standard component of the toolbox of applied statisticians. They are most often used to obtain a 95% confidence interval. Many stock assessments “fix” key parameters such as M and steepness based on *a priori* considerations. Likelihood profiles can be used to evaluate whether there is evidence in the data to support fixing a parameter at a chosen value. If the parameter is within the entire range of the 95% confidence interval, this provides no support in the data to change the fixed value. If the fixed value is outside the 95% confidence interval, it would be reasonable for a review panel to ask why the parameter was fixed and not estimated, and if the value is to be fixed, on what basis and why should what amounts to inconsistency with the data be ignored. Integrated stock assessments include multiple data sources (e.g., commonly catch-rates, length-compositions, and age-compositions) that may be in conflict, due for example to inconsistencies in sampling, but more commonly owing to incorrect assumptions (e.g., assuming that catch-rates are linearly related to abundance), i.e. model-misspecification. Likelihood profiles can be used as a diagnostic to identify these data conflicts (Punt, 2018).

Standard parameters to consider are natural mortality (M), steepness (h) and the logarithm of the unfished recruitment ($\ln R_0$).

For jackass morwong west, the likelihood profile for natural mortality, M , a parameter fixed in the model, is shown in Figure 7.12, with the total likelihood shown in black and components of the total likelihood from different data sources shown in a range of colours. This shows that the fixed value chosen for M (0.15yr^{-1}) is close to the minimum of the likelihood profile, 0.16 yr^{-1} , with the age, the discard data and the index data most influential in this likelihood profile. Note that the index data suggests a lower value for M (0.12) and the age data a higher value (0.22). While M is unlikely to be chosen based on results from a likelihood profile alone, the biology and the maximum age of this species suggest that the choice of $M = 0.15\text{ yr}^{-1}$ appears very reasonable, so there is no conflict between the likelihood profile and the biological considerations.

The likelihood profile for steepness, h , (Figure 7.13) suggests that there is little information in the model that can be used to inform this parameter (fixed at 0.7 in the model). The age data (higher steepness) and index data (lower steepness) are in conflict, and the 95% confidence interval is very broad. This likelihood profile is uninformative.

The likelihood profile for the logarithm of the unfished recruitment ($\ln R_0$, Figure 7.14) indicates a 95% confidence interval between values close to 6.8 to 7.3 , with the value estimated with the model at 7.088 . This range corresponds to an initial female spawning biomass range between around $1,000\text{t}$ and $1,650\text{t}$, indicating that virgin biomass is estimated with considerable uncertainty. There is little conflict from the different data sources in this likelihood profile.

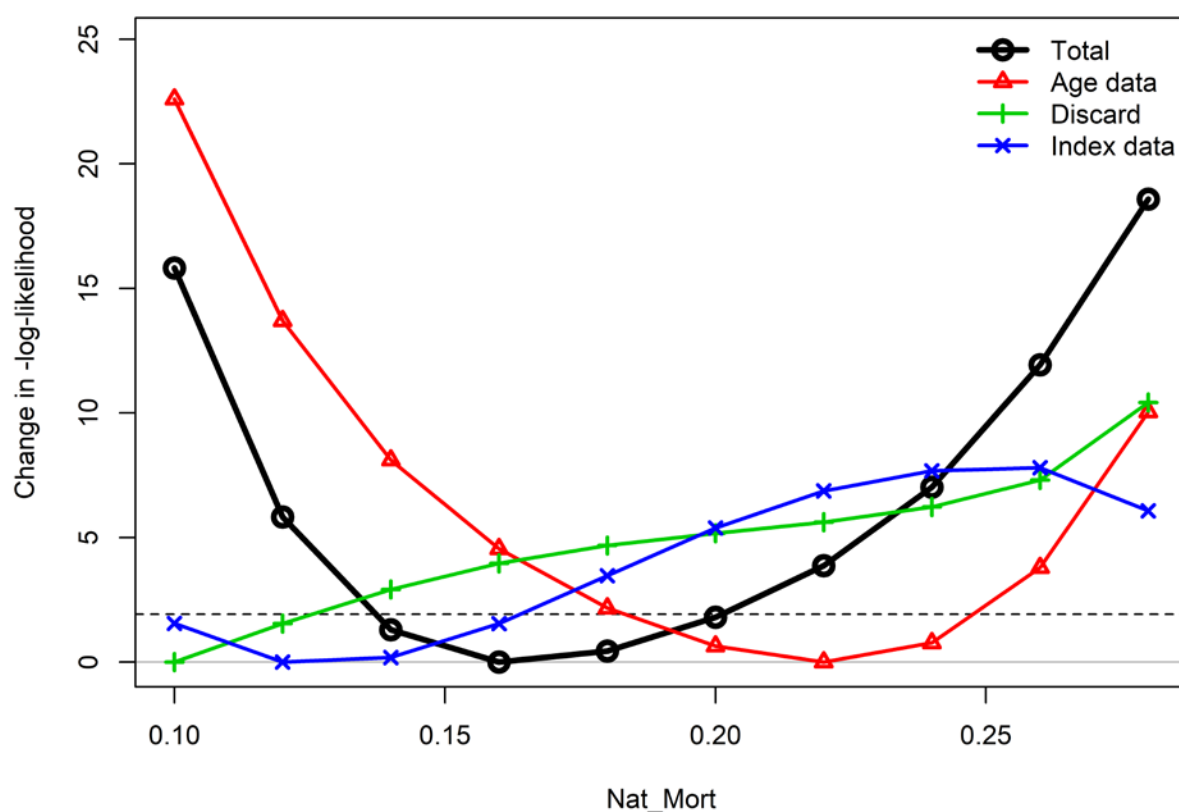


Figure 7.12. The likelihood profile for natural mortality. The fixed value for M is 0.15yr^{-1} .

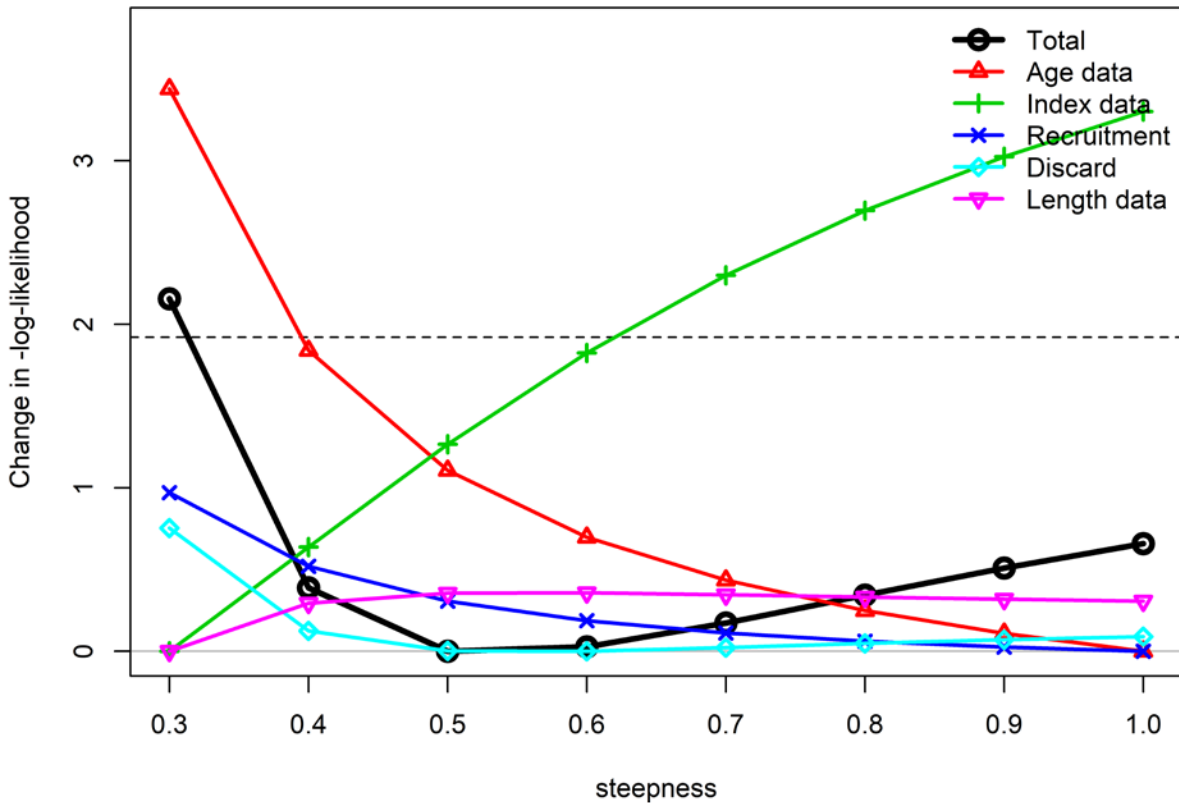


Figure 7.13. The likelihood profile for steepness. The fixed value for h is 0.7.

7.2.5 Retrospectives

A retrospective analysis was completed, starting from the most recent year of data, working backward in time and removing successive years of data from the assessment. This analysis can highlight potential problems and instability in an assessment, or some features that appear from the data.

A retrospective analysis for absolute spawning biomass is shown in Figure 7.15, with initially the data after 2017 removed (shown in blue), then successive years of data removed back to 2012 (shown in red). These time series shows a pattern with the minimum spawning biomass increasing as data is removed. The same pattern is observed when this is shown in terms of relative spawning biomass (Figure 7.16).

When this retrospective analysis is applied to the recruitment time series (Figure 7.17), the more recent data results in a revision downward to the recruitment estimates in the period 2008-2009, and a shift to a larger recruitment in the period 2010-2012. This analysis should probably have also included a change to the last year that recruitment is being estimated to prevent this pattern from occurring, and spurious recruitments being estimated at the end of the time series, with little data available to inform these estimates.

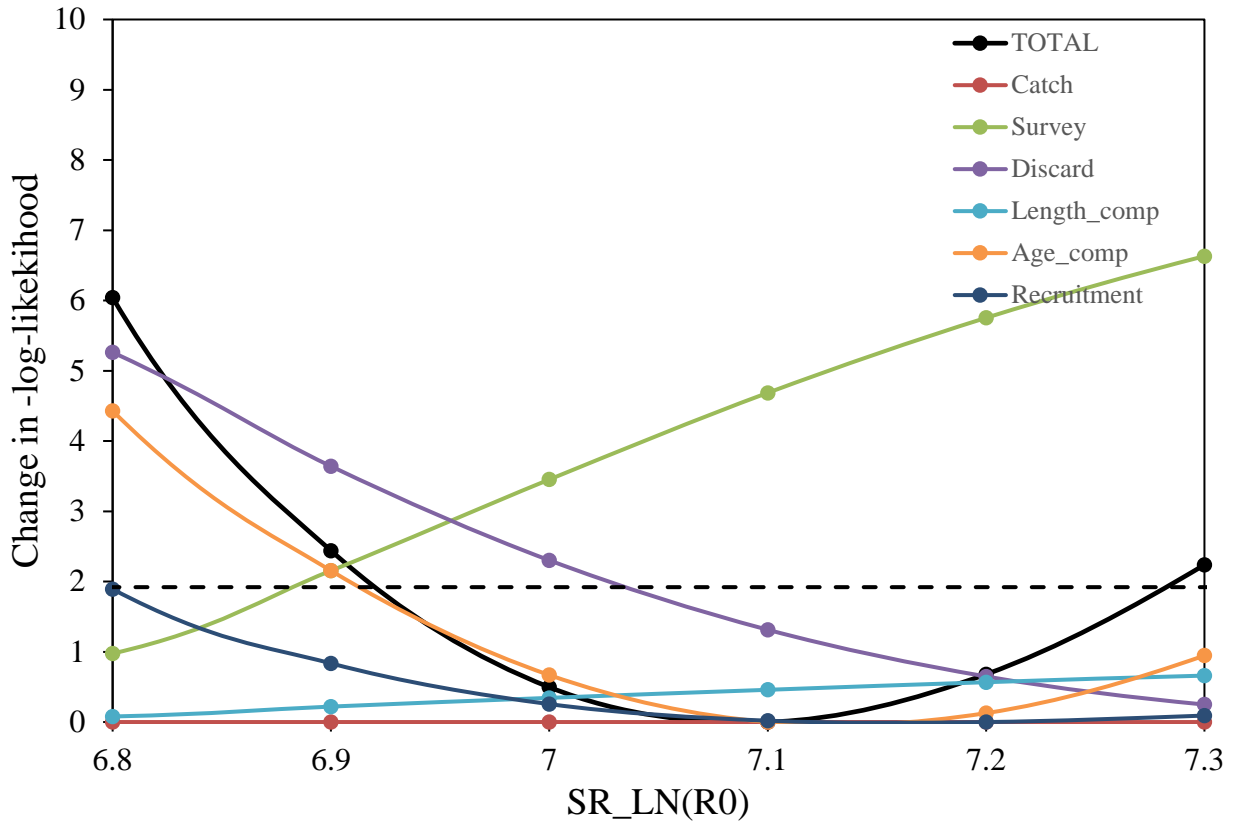


Figure 7.14. The likelihood profile for lnR0. This parameter is estimated in the model.

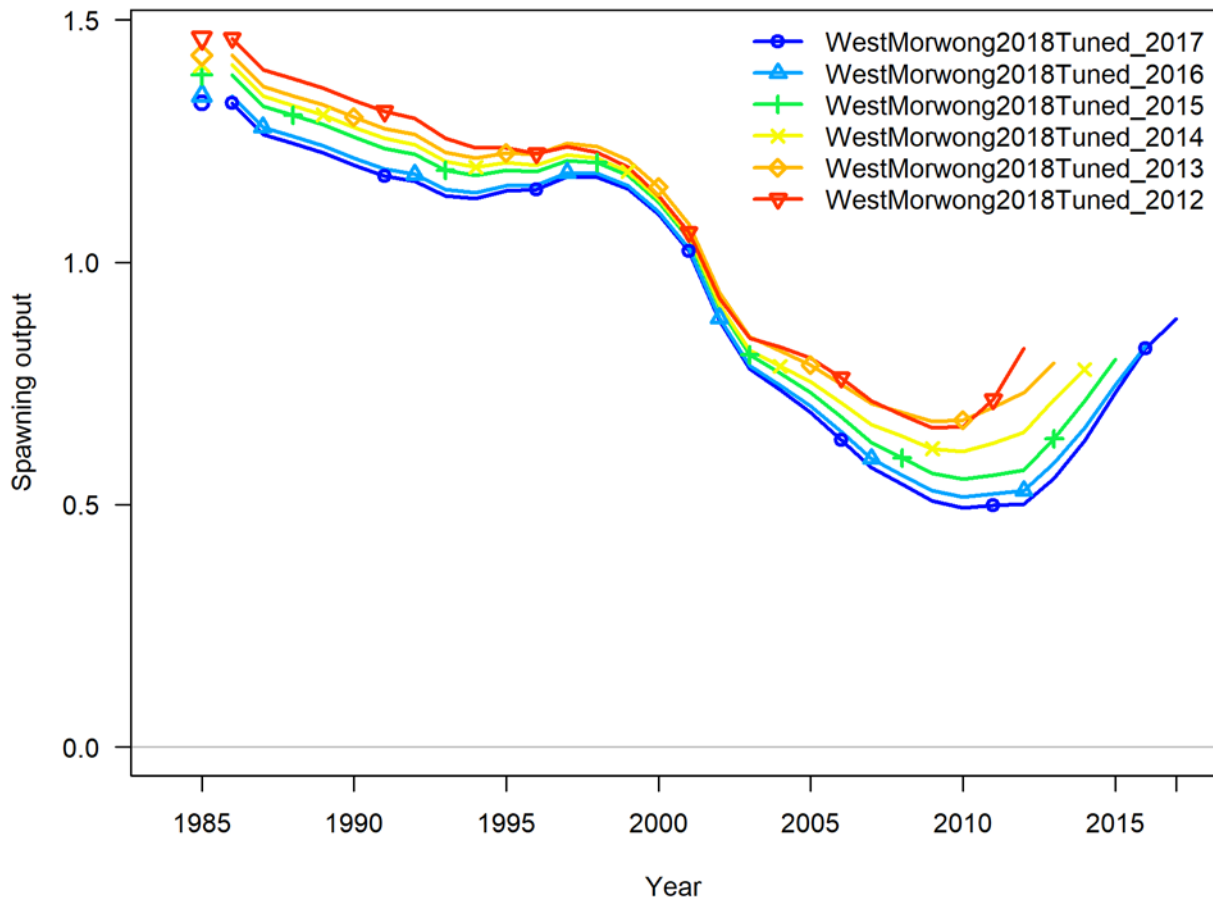


Figure 7.15. Retrospectives for absolute spawning biomass for western jackass morwong, with data removed back to 2017 (blue) and then successive years removed back to 2012 (red).

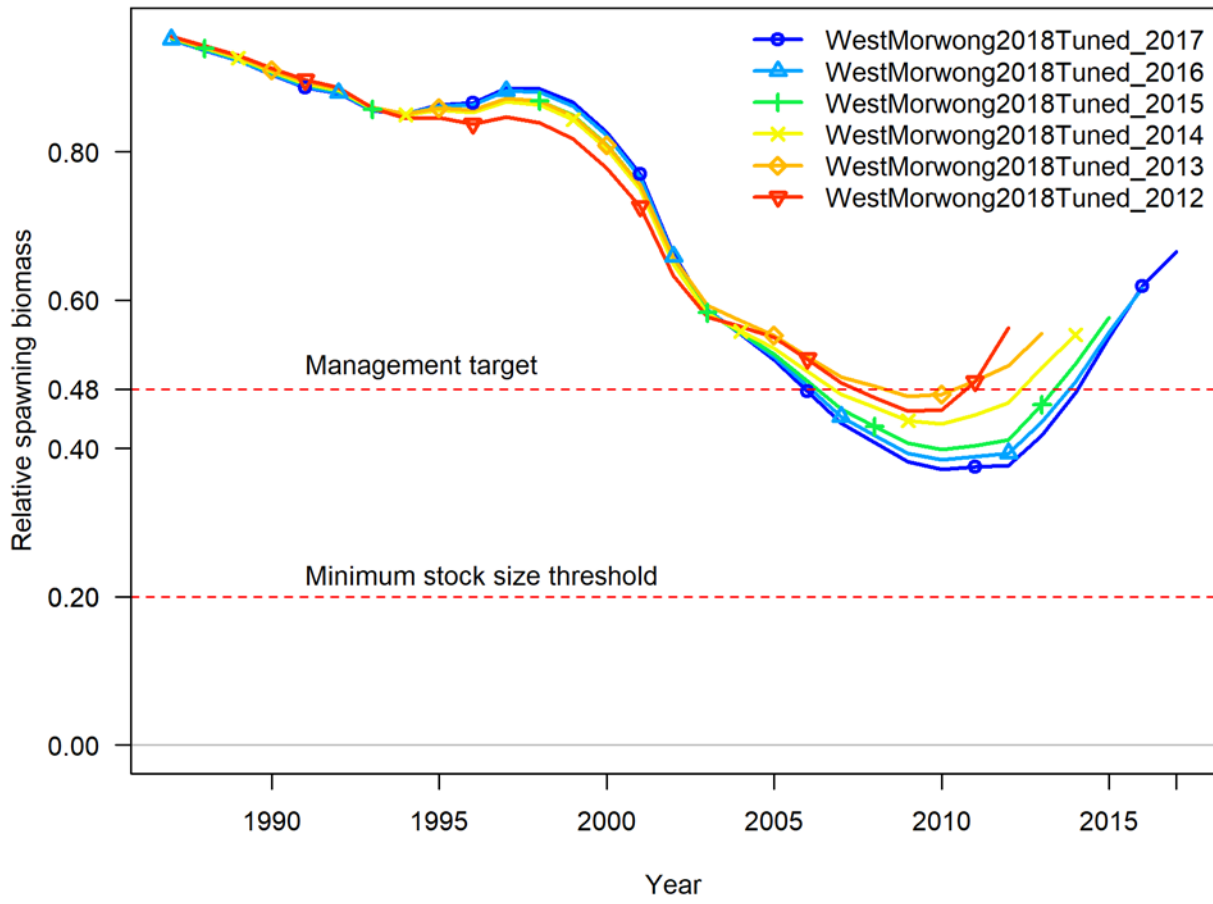


Figure 7.16. Retrospectives for relative spawning biomass for western jackass morwong, with data removed back to 2017 (blue) and then successive years removed back to 2012 (red).

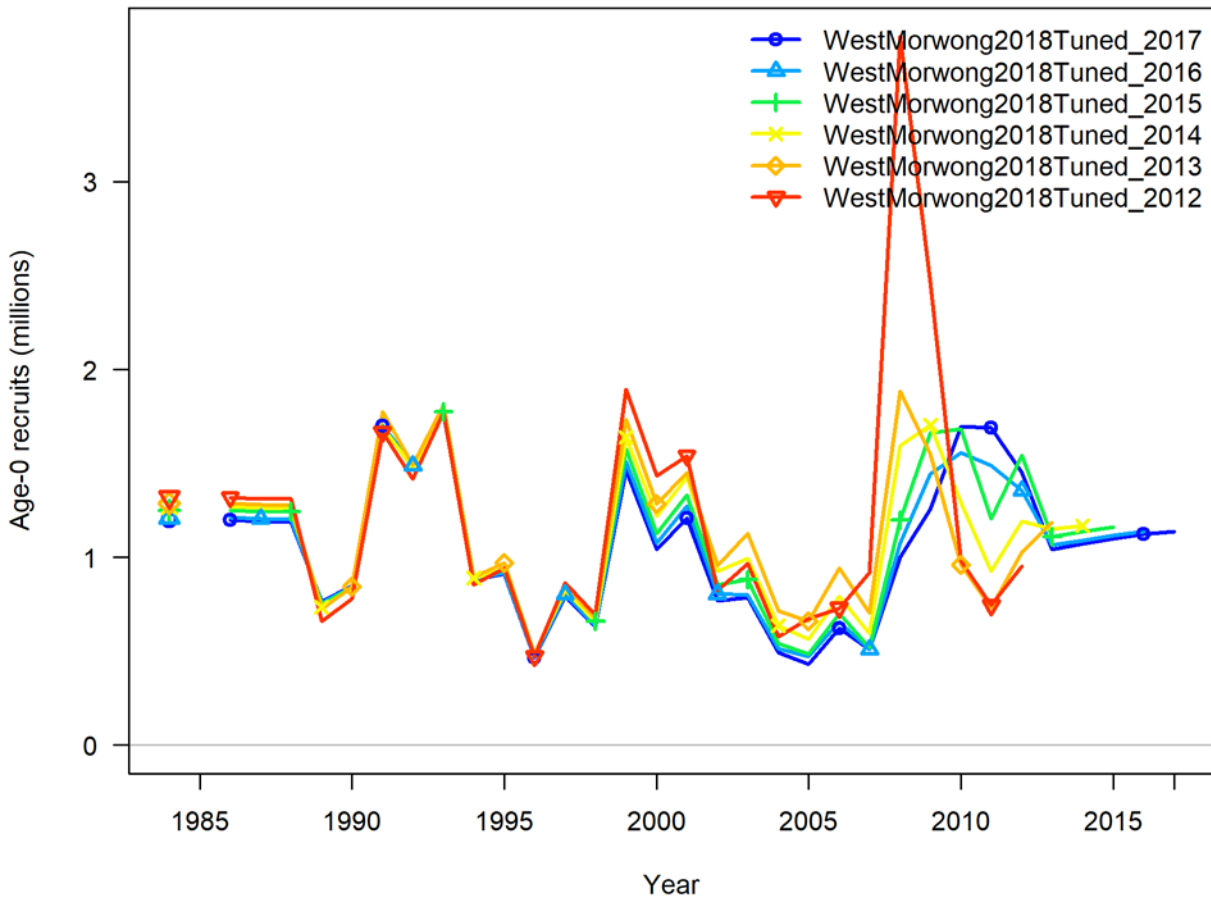


Figure 7.17. Retrospectives for recruitment for western jackass morwong, with data removed back to 2017 (blue) and then successive years removed back to 2012 (red).

7.2.6 Future work and unresolved issues

An alternative base case that could be considered is to remove the discard data and add the estimated discard amounts (in mass) into the catch fleet. The discard length-composition data are limited to two years and are quite variable. It is possible that discarding practices have changed considerably from year to year and between size-based and market-based discarding, which makes fitting to discard data difficult. The decision as to how to deal with discard data is somewhat subjective and could benefit from greater discussion at SERAG.

Two other sensitivities relating to the Fishery Independent Survey (FIS) would be useful.

1. Excluding all FIS data.
2. Including FIS length frequency data and estimating selectivity for the FIS fleet.

Any results from this assessment should be treated with considerable caution given the limited data quality and data quantity available for this assessment and the quality of the trawl CPUE data (Sporcic and Haddon 2018a). Given several sudden (step) changes in this time series, it may not be a very reliable index of abundance and instead may be reflecting other changes in the fishery that are not incorporated in this model. Sporcic and Haddon (2018a) indicate that the vessel factor changed its influence from 2001 onwards, suggesting a change in the fishery at that time.

Note that the preliminary base case model fit to the index of abundance is poor (Figure 7.10), as is the fit to the FIS abundance indices (Figure 7.11), with additional CVs on these abundance series estimated within the model at 0.18 and 0.6 respectively. It is possible that the data are neither sufficiently representative nor sufficiently rich to adequately assess this stock. Alternatively there may be other unknown issues with the fishery dynamics and the stock dynamics that have not been adequately represented in this model.

7.3 Acknowledgements

Age data were provided by Kyne Krusic-Golub (Fish Ageing Services), ISMP and AFMA logbook and CDR data were provided by John Garvey (AFMA). Franzis Althaus, Mike Fuller, Roy Deng, Claudio Castillo-Jordán, and Paul Burch (CSIRO) pre-processed the data. André Punt, Geoff Tuck, Sandra Curin-Osorio, Paul Burch, Malcolm Haddon, Robin Thomson and Miriana Sporcic are thanked for helpful discussions on this work. Ian Taylor, Richard Methot and Chantel Wetzel (NOAA Fisheries) are thanked for all the Stock Synthesis support and advice. Malcolm Haddon provided useful code for auto-tuning, Athol Whitten provided useful R code for organising plots.

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7.5 Appendix A

A.1 Preliminary base case diagnostics

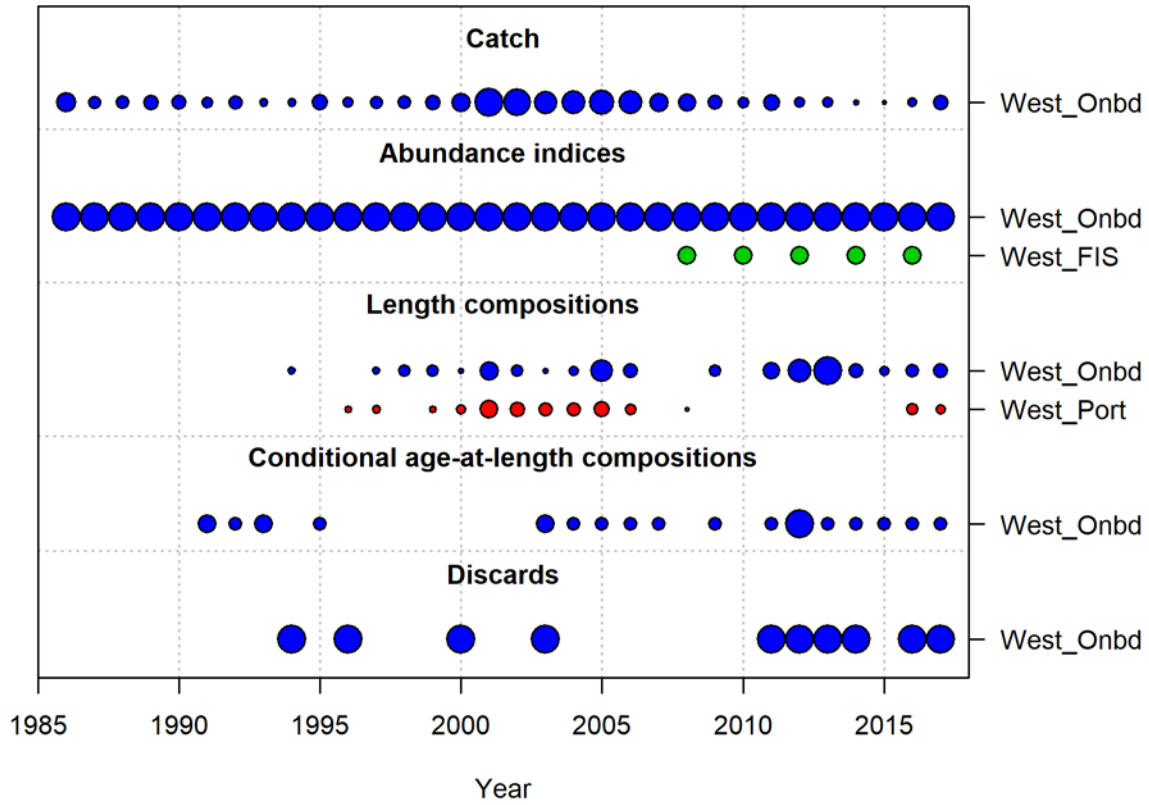


Figure A 7.1. Summary of data sources for western jackass morwong stock assessment.

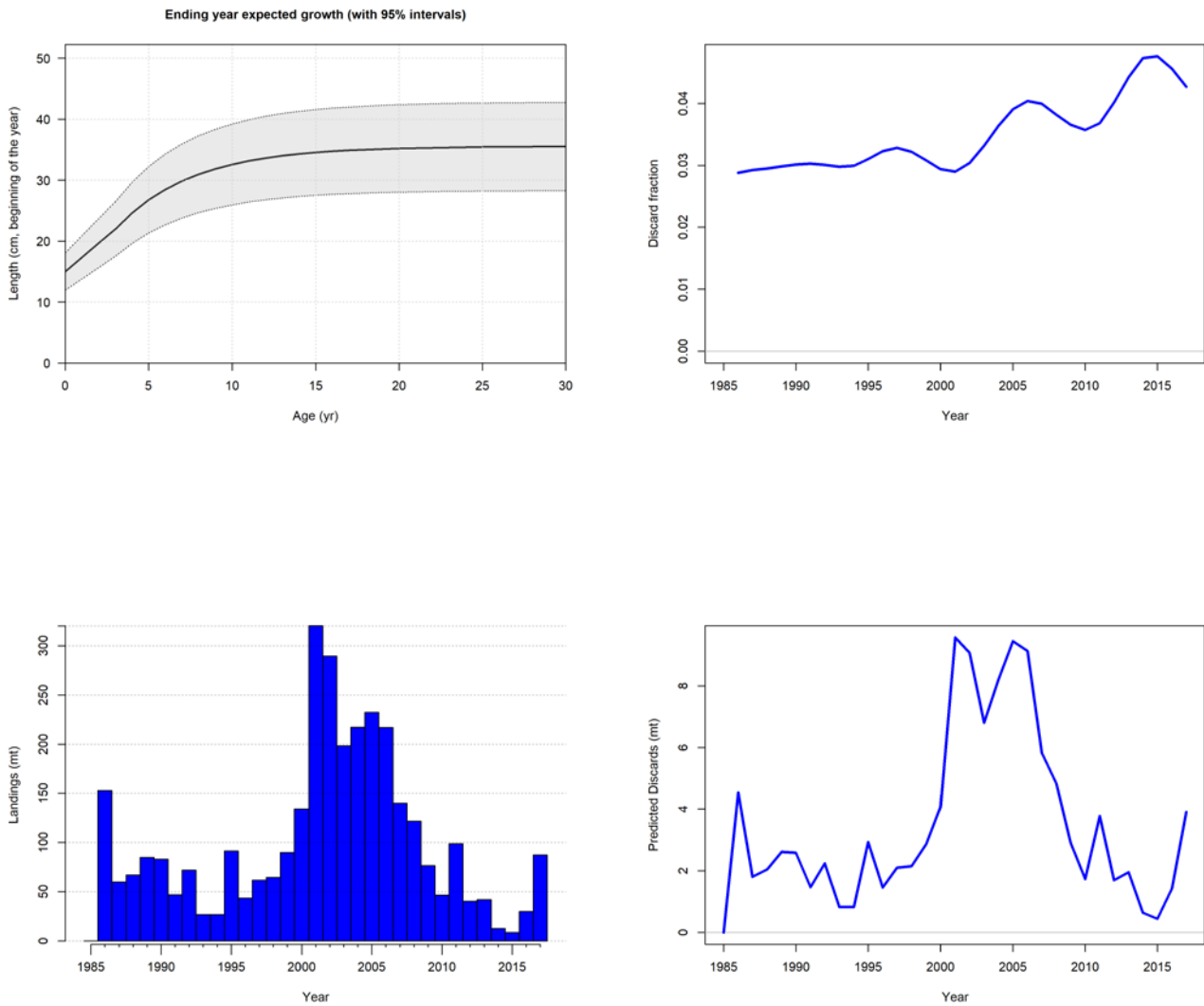


Figure A 7.2. Growth, discard fraction estimates, landings by fleet and predicted discards by fleet for western jackass morwong.

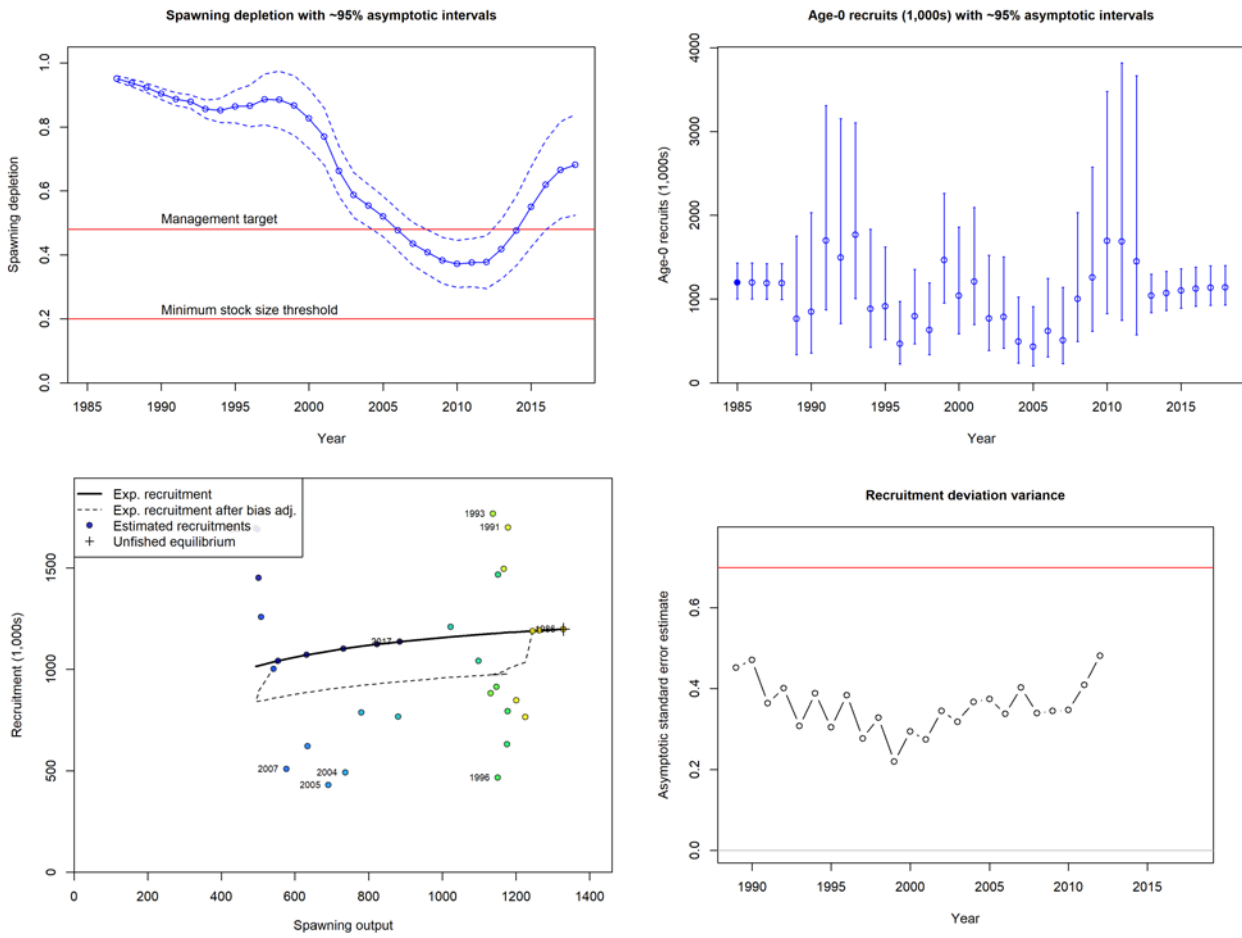


Figure A 7.3. Time series showing depletion of spawning biomass with confidence intervals, recruitment estimates with confidence intervals, stock recruitment curve and recruitment deviation variance check for western jackass morwong.

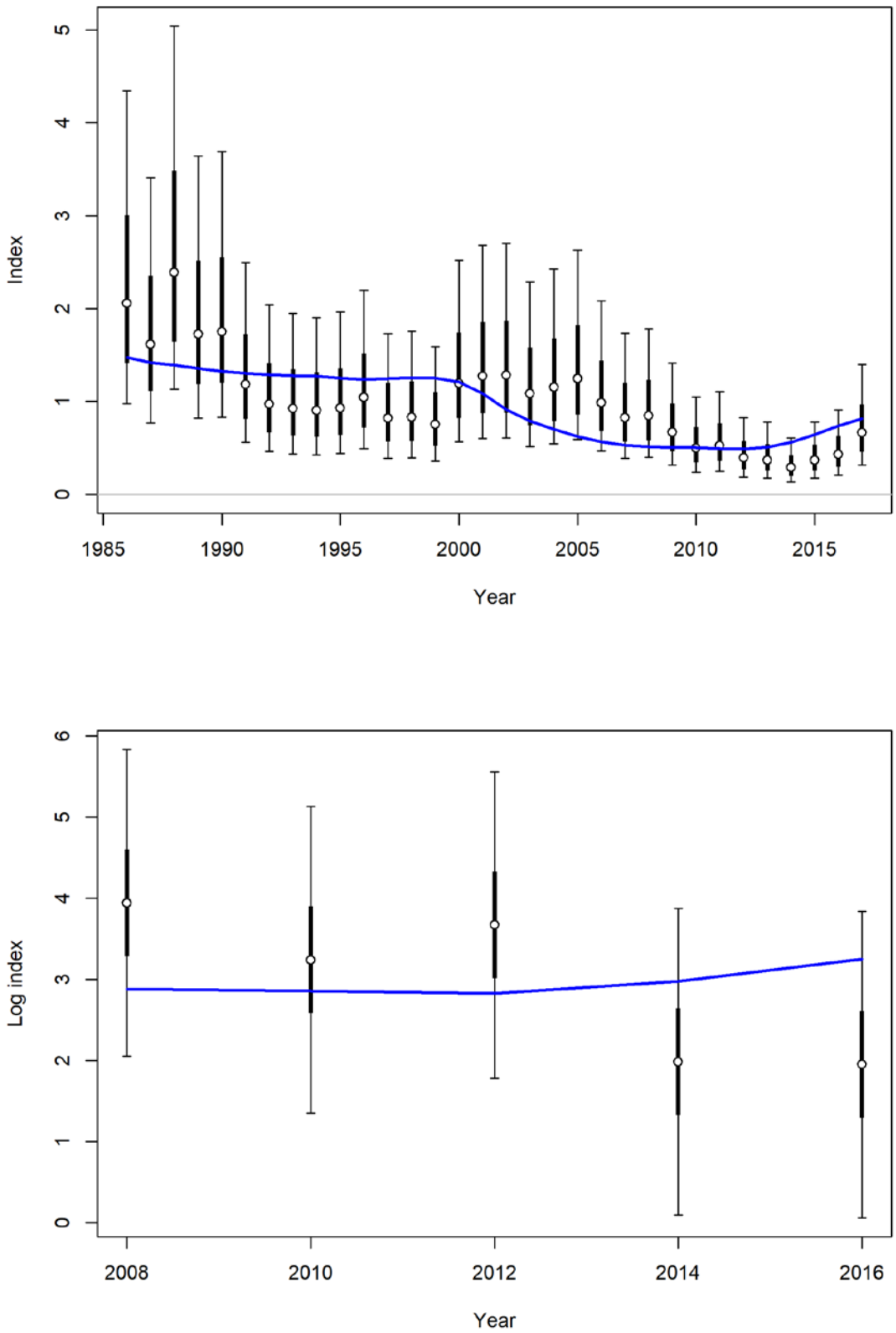


Figure A 7.4. Fits to CPUE by fleet for western jackass morwong: trawl and FIS.

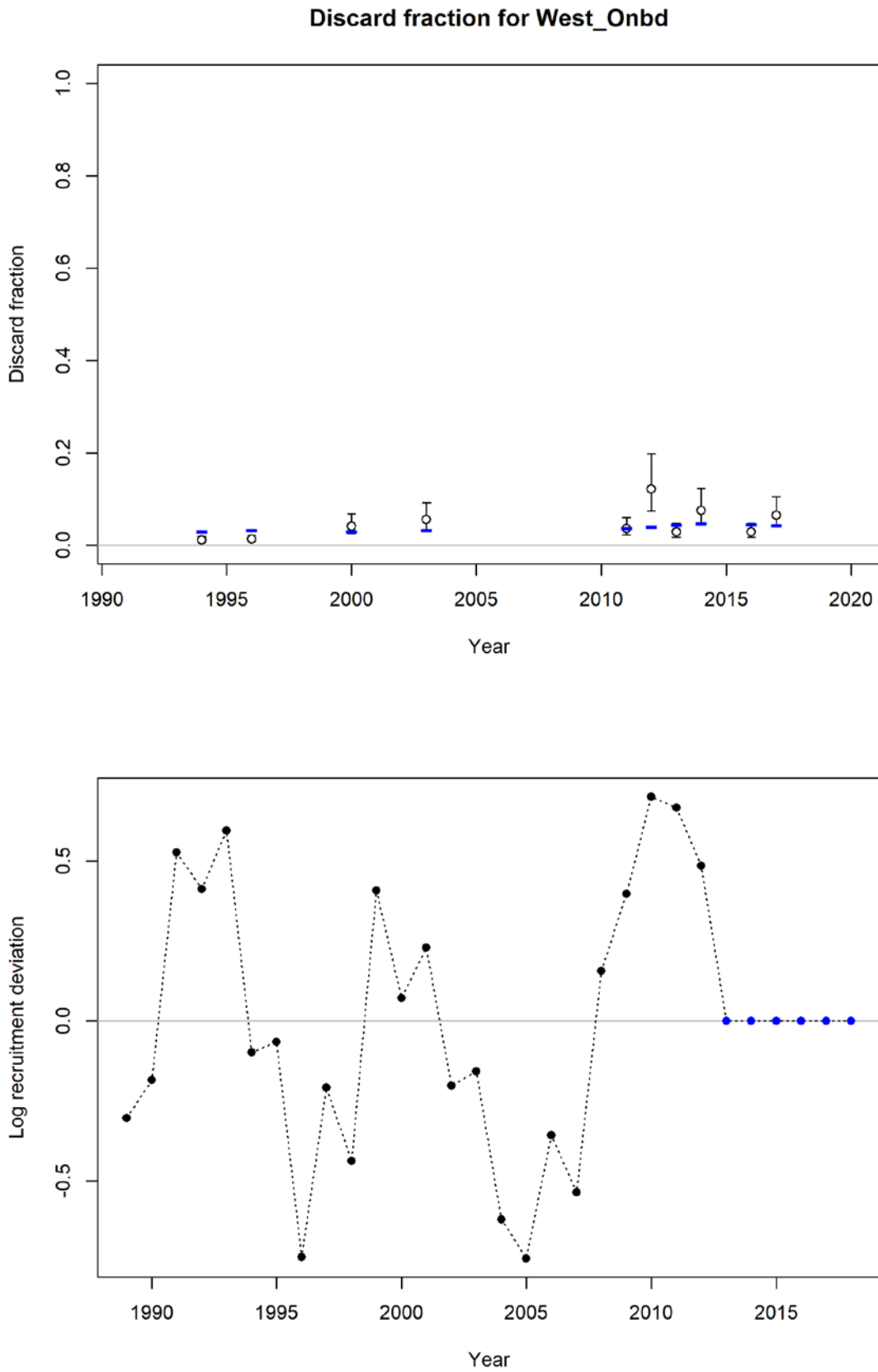


Figure A 7.5. Fits to discard rates (trawl) and recruitment deviations for western jackass morwong.

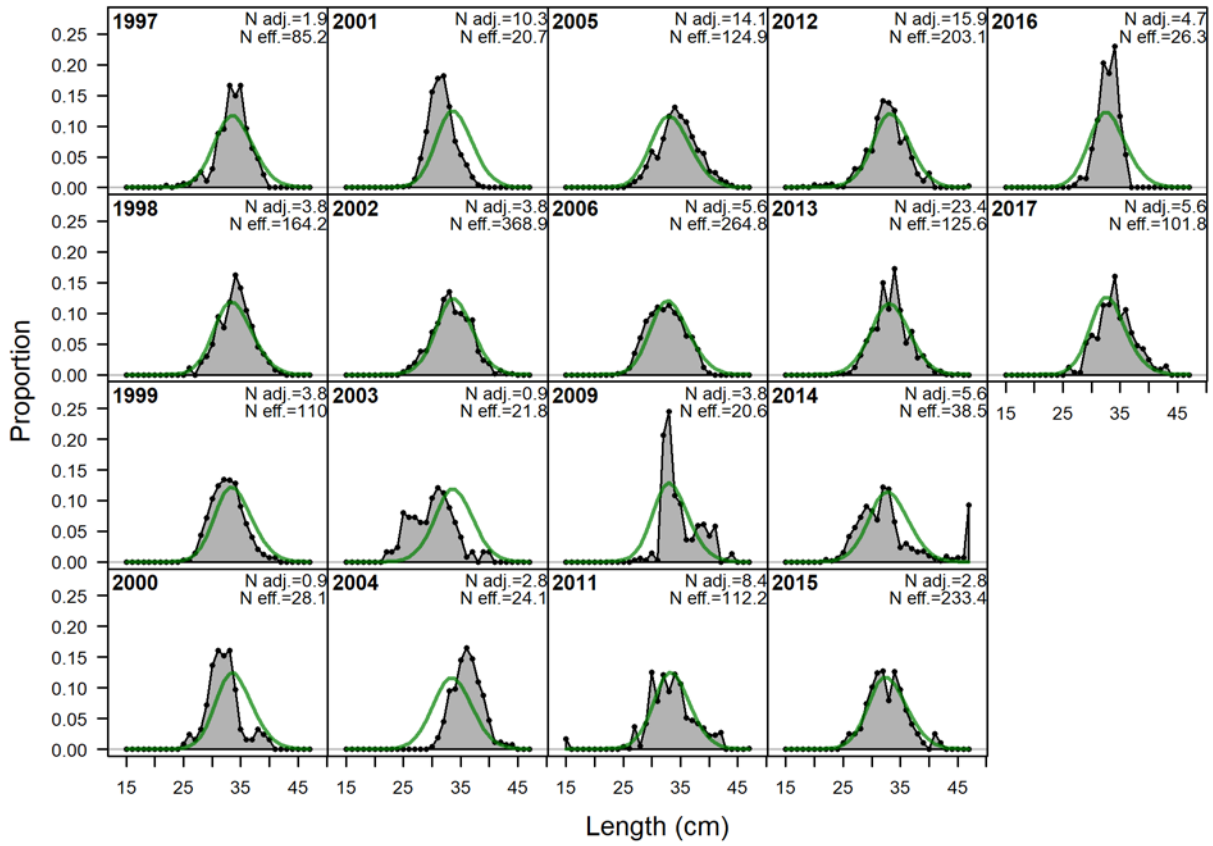


Figure A 7.6. Western jackass morwong length composition fits: trawl onboard retained.

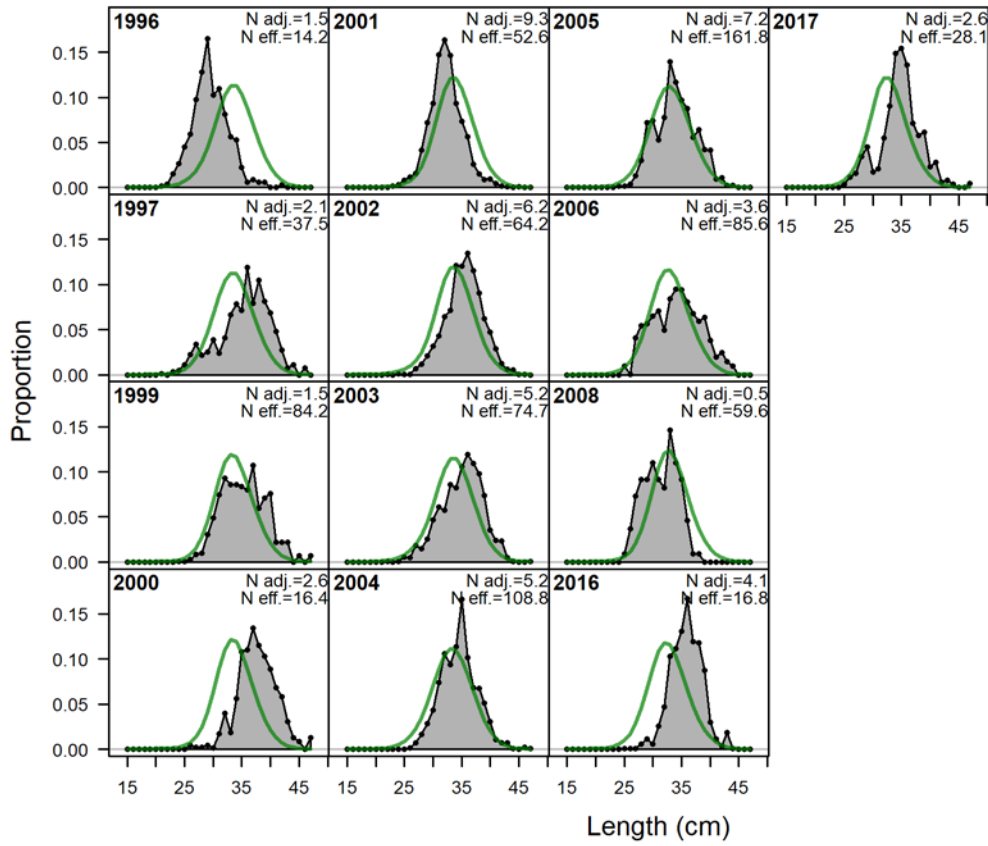


Figure A 7.7. Western jackass morwong length composition fits: trawl port retained.

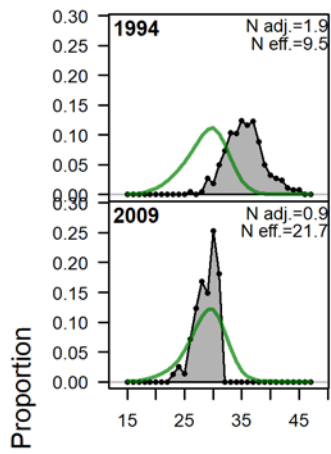


Figure A 7.8. Western jackass morwong length composition fits: trawl discarded.

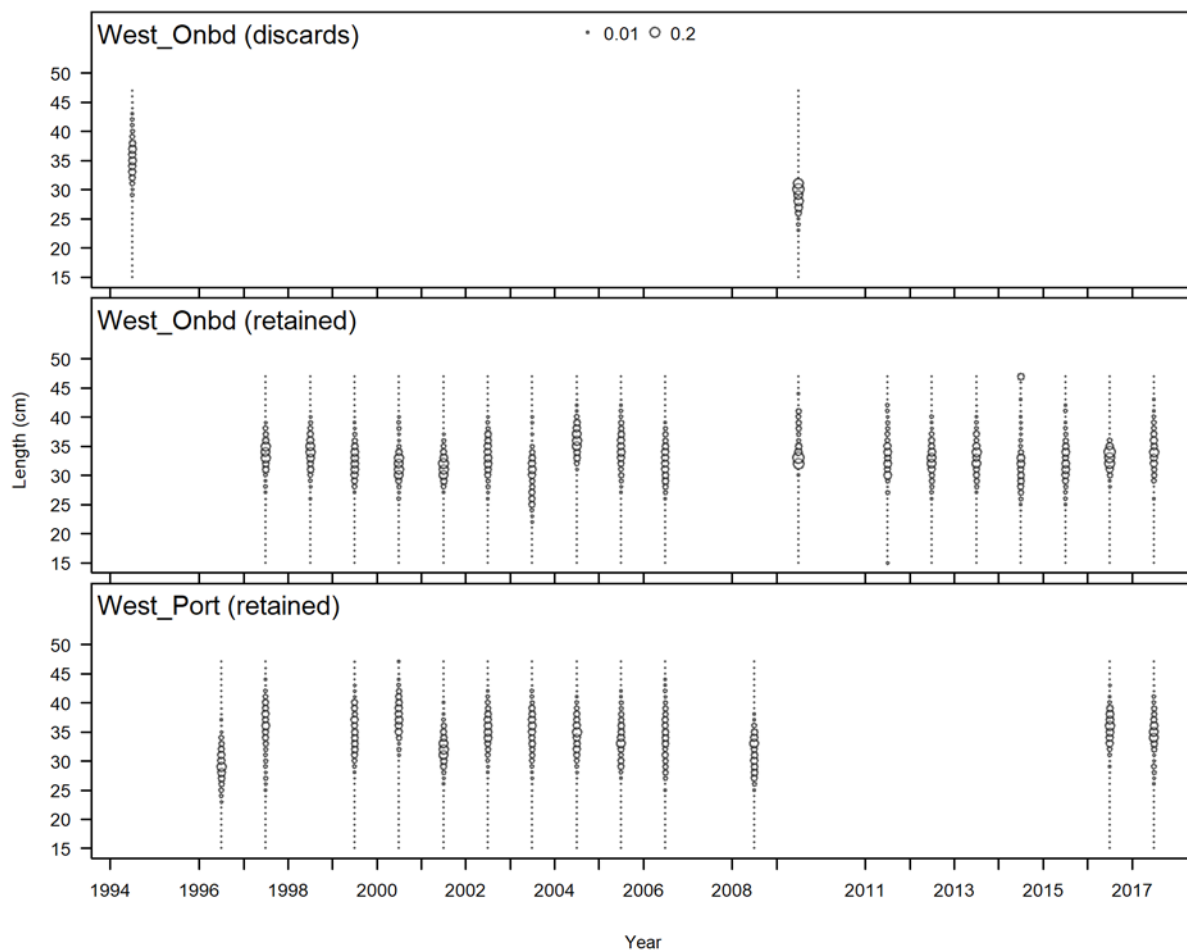


Figure A 7.9. Residuals from the annual length compositions (retained and discarded) for western jackass morwong displayed by year for trawl fleets.

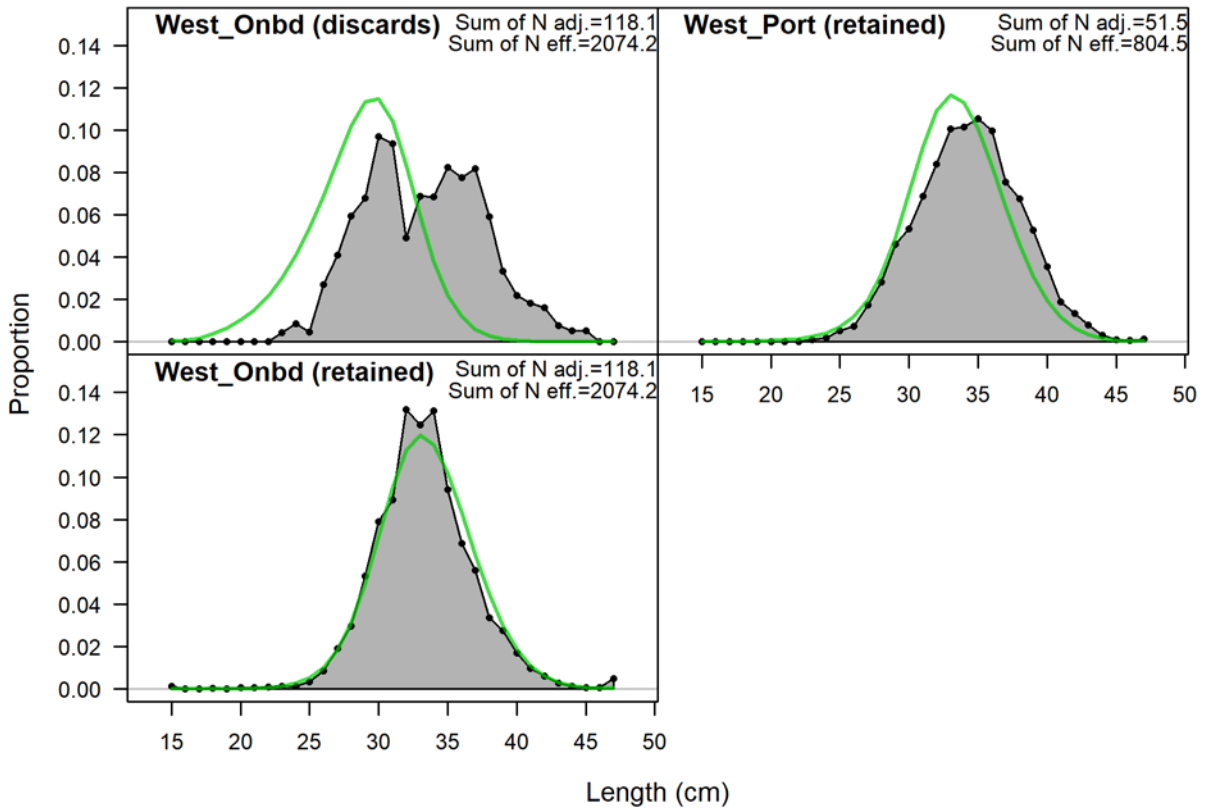


Figure A 7.10. Aggregated fits (over all years) to the length compositions for western jackass morwong displayed by fleet.

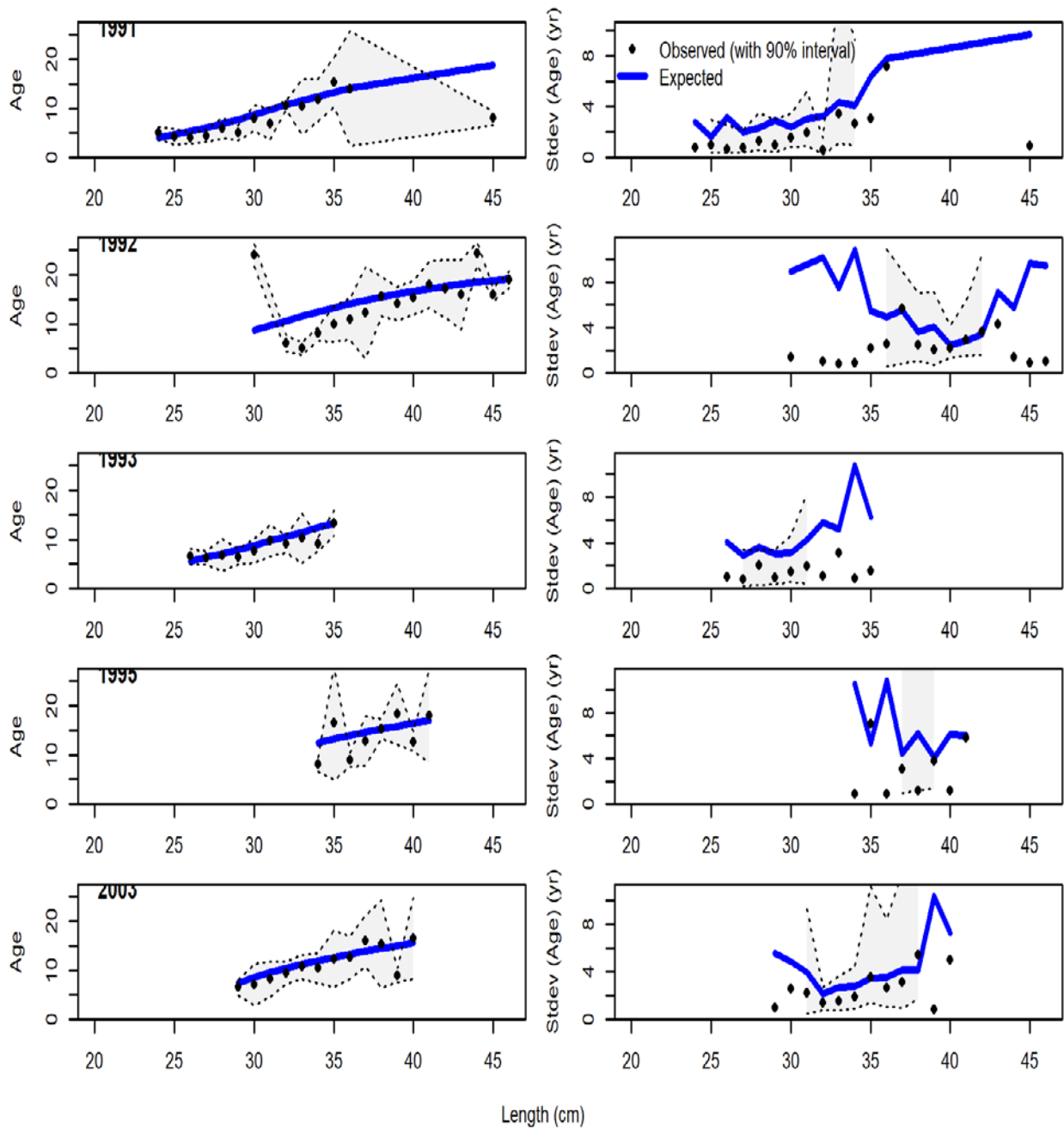


Figure A 7.11. Western jackass morwong conditional age-at-length fits: trawl part 1.

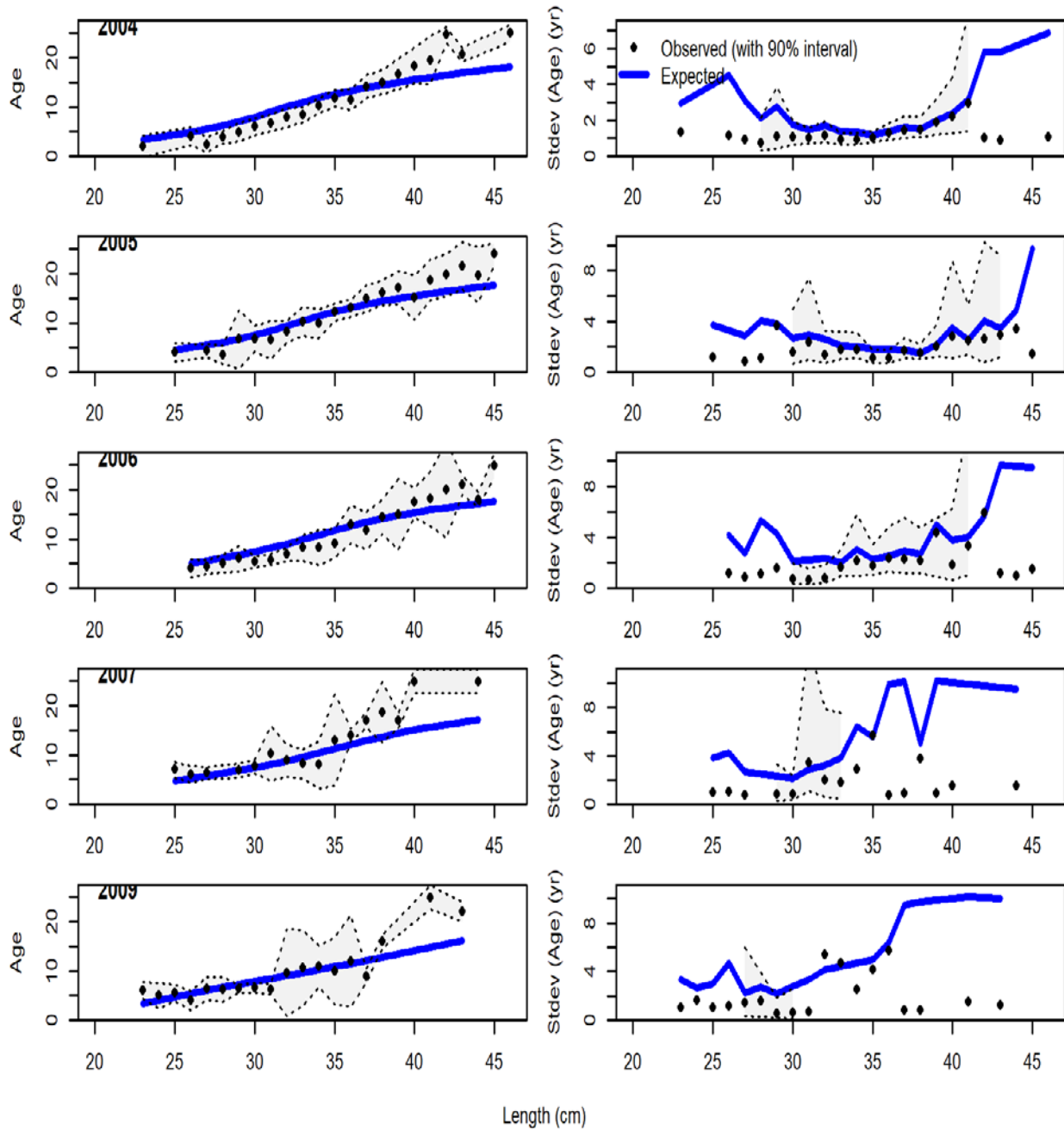


Figure A 7.12. Western jackass morwong conditional age-at-length fits: trawl part 2.

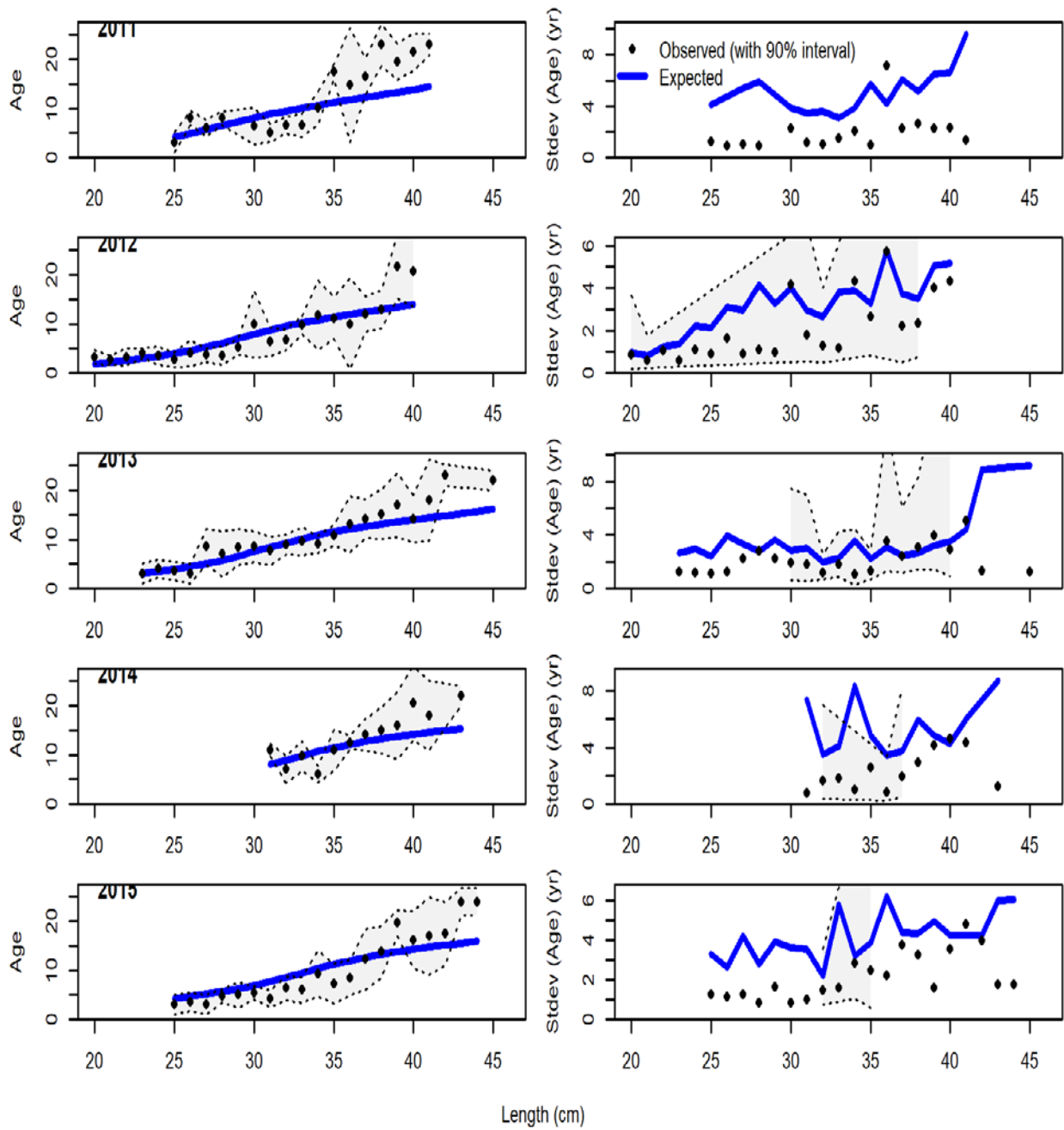


Figure A 7.13. Western jackass morwong conditional age-at-length fits: trawl part 3.

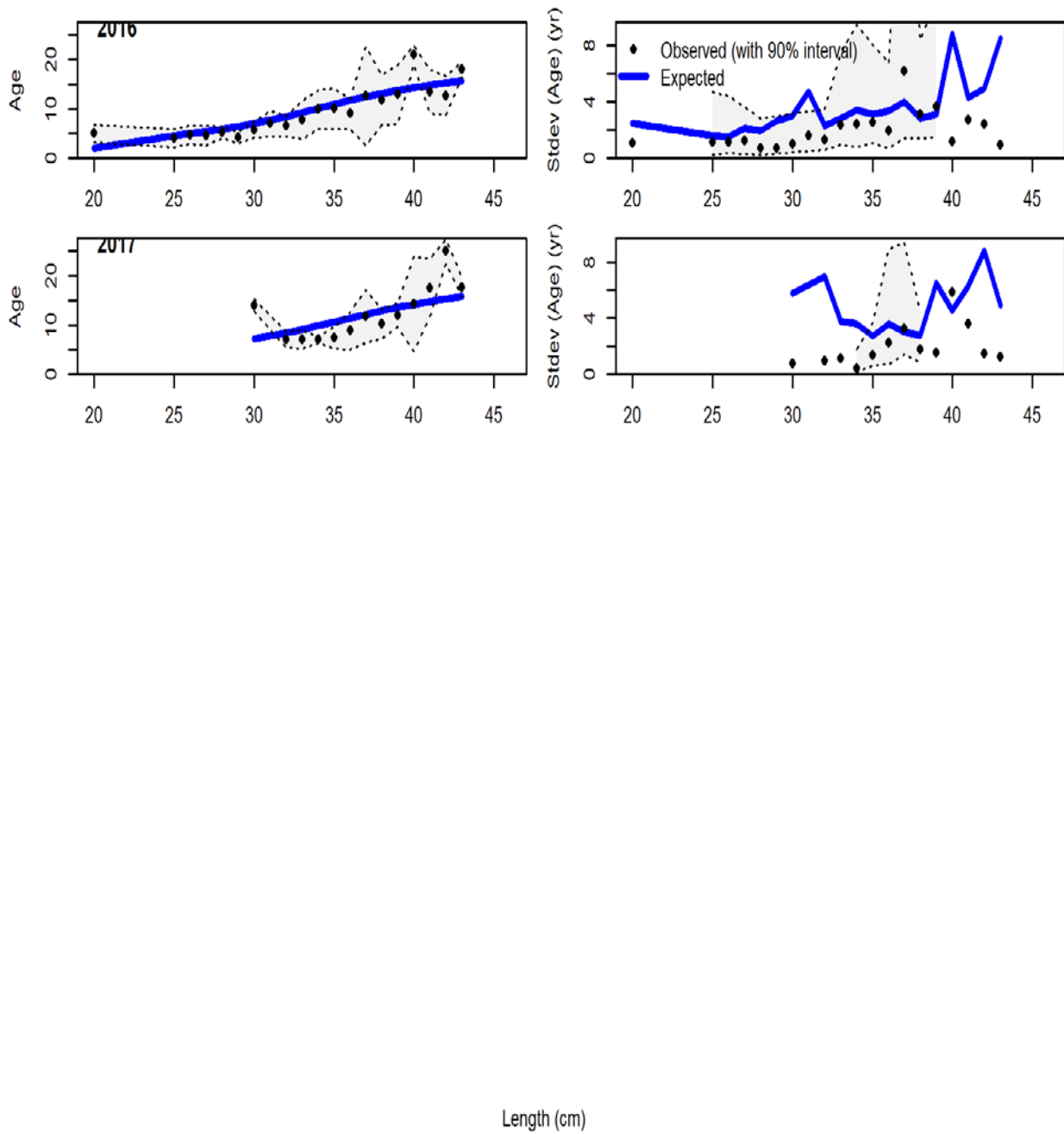


Figure A 7.14. Western jackass morwong conditional age-at-length fits: trawl part 4.

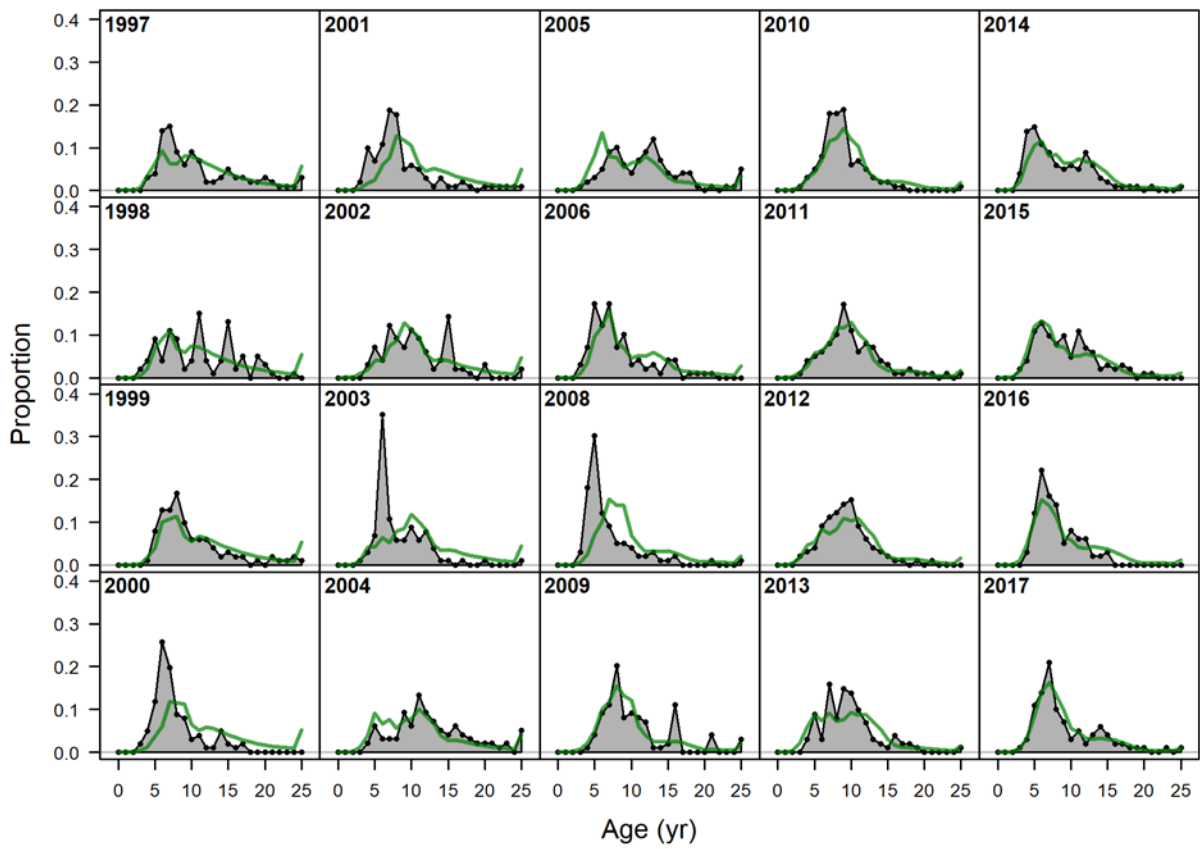


Figure A 7.15. Western jackass morwong implied fits to age: trawl onboard retained.

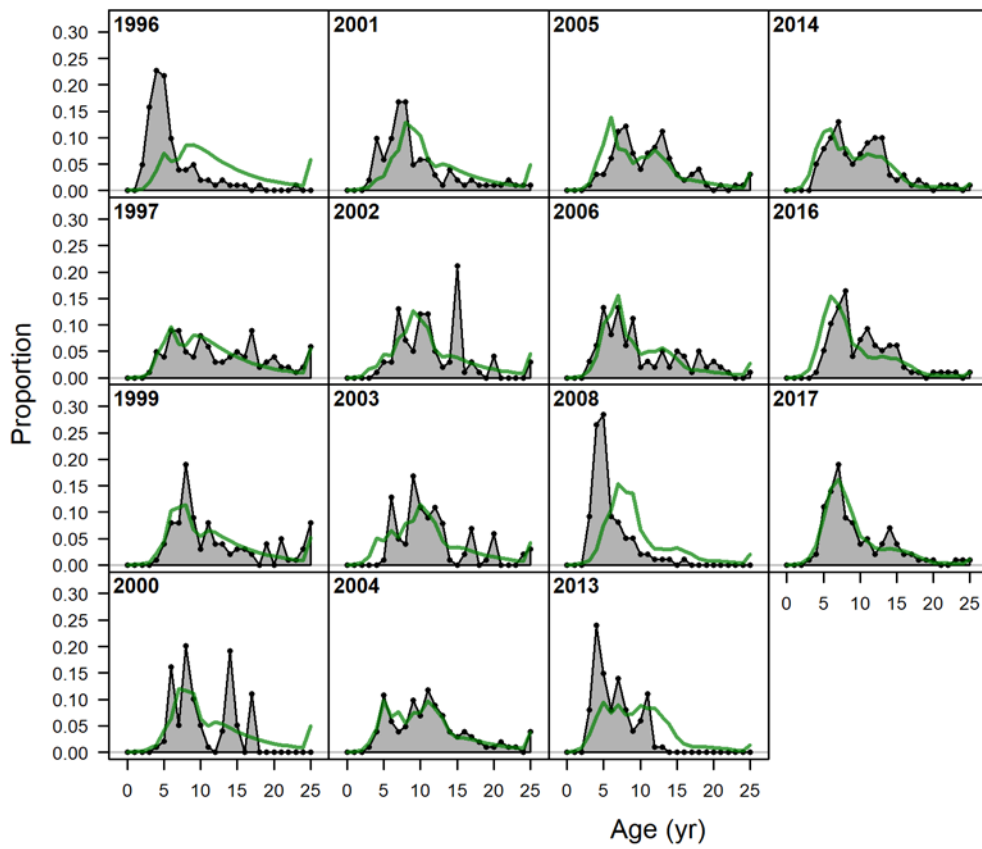


Figure A 7.16. Western jackass morwong implied fits to age: trawl port retained.

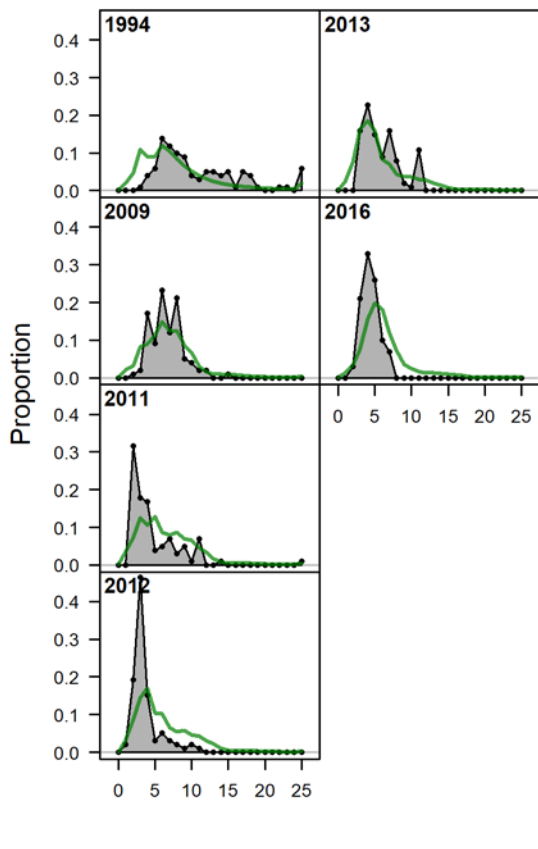


Figure A 7.17. Western jackass morwong implied fits to age: trawl onboard discarded.

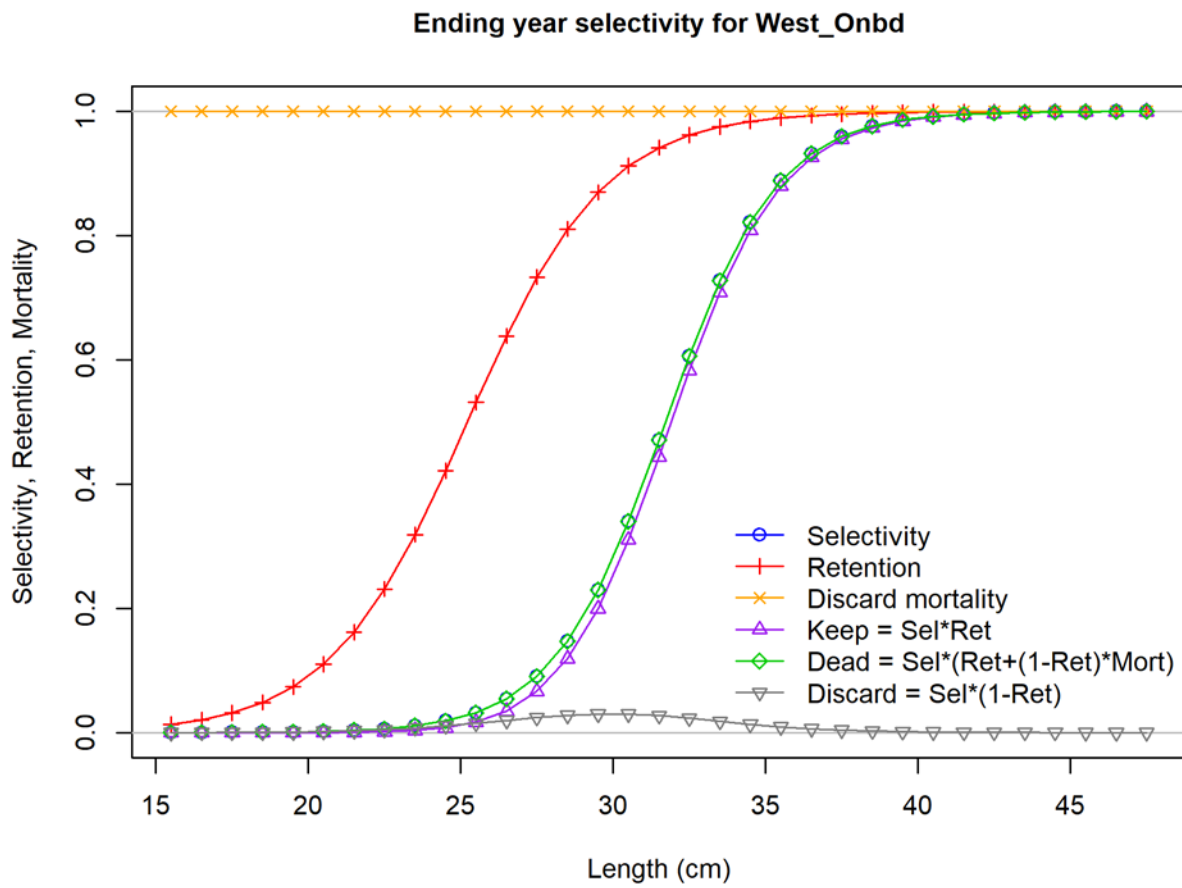


Figure A 7.18. Estimated selectivity and retention curves for western jackass morwong trawl fleet.

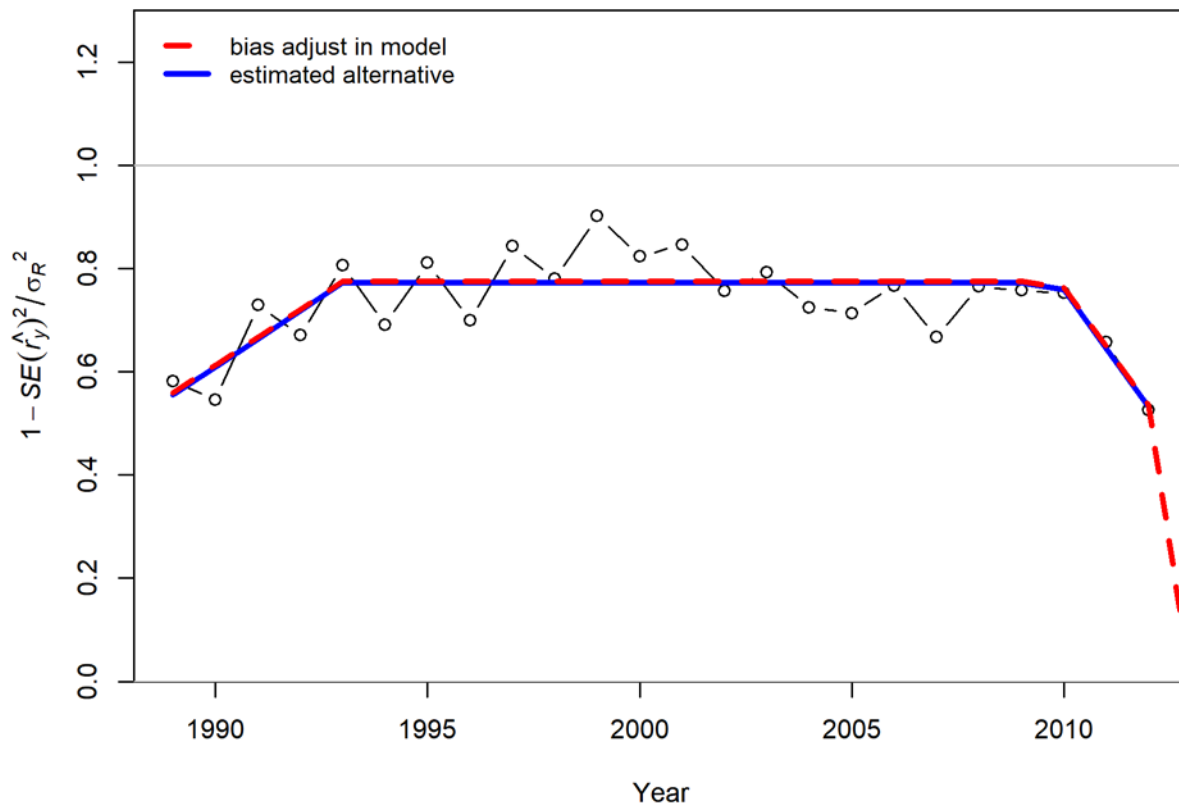


Figure A 7.19. Bias ramp adjustment for western jackass morwong.

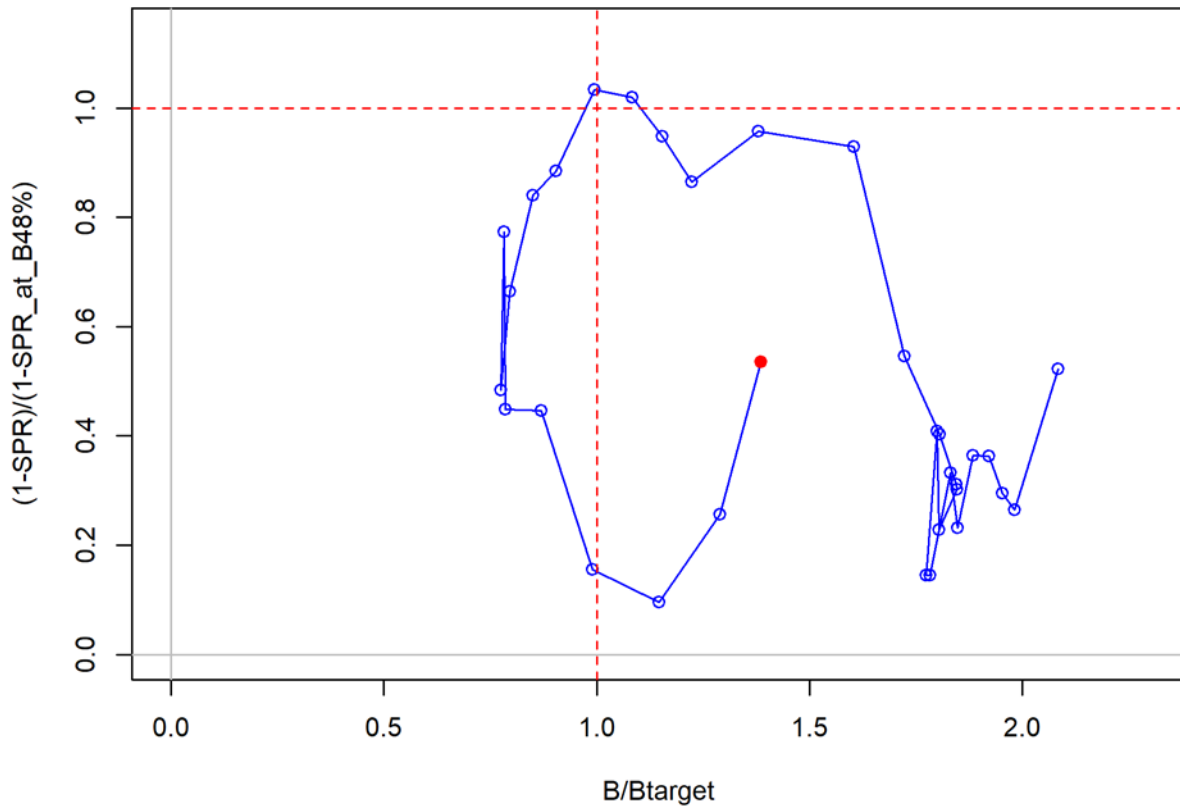


Figure A 7.20. Phase plot of biomass vs SPR ratio.

8. Western Jackass Morwong (*Nemadactylus macropterus*) stock assessment based on data up to 2017

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8.1 Executive Summary

This document updates the 2015 assessment Tier 1 assessment of western jackass morwong (*Nemadactylus macropterus*) to provide estimates of stock status in the SESSF at the start of 2019 and describes the base case assessment and some of the issues encountered during development. This assessment was performed using the stock assessment package Stock Synthesis (version V3.30.12.00). The 2015 stock assessment has been updated with the inclusion of data up to the end of 2017, comprising an additional three years of catch, discard, CPUE, length and age data and ageing error updates, including revisions to historical catch series, length frequencies and discard rates. A range of sensitivities were explored.

The base-case assessment estimates that current spawning stock biomass is 68% of unexploited stock biomass (SSB_0). Under the agreed 20:35:48 harvest control rule, the 2019 recommended biological catch (RBC) is 235 t, with the long term yield (assuming average recruitment in the future) of 158 t. The average RBC over the three year period 2019-2021 is 223 t and over the five year period 2019-2023, the average RBC is 212 t.

Exploration of model sensitivity showed variation in spawning biomass across all sensitivities ranging from 33% to 102% of SSB_0 with greatest sensitivity to natural mortality. Excluding this sensitivity to natural mortality, the other sensitivities showed a much narrower range, from 60% to 75% of SSB_0 .

Changes to the 2015 stock assessment include: estimating discards and retention rather than simply adding discards to landed catches; and using the latest agreed best practice tuning method. The updated assessment is consistent with the results from the 2015 assessment, despite an additional three years of data, improvements to data processing and modifications to Stock Synthesis. As in the 2015 assessment, results show poor fits to the abundance data (catch rate and Fishery Independent Survey (FIS)), but acceptable fits to the length composition and conditional age-at-length data.

8.2 Introduction

8.2.1 The fishery

Jackass morwong (*Nemadactylus macropterus*) have been landed in southern Australia since the inception of the steam trawl fishery off New South Wales in the early twentieth century (Fay 2004), with the initial fishery concentrating in the east (SESSF Zones 10, 20 and 30). Jackass morwong were not favoured during the initial years of this fishery, when the main target species was tiger flathead (*Neoplatycephalus richardsoni*). Declines in flathead catches and improved market acceptance led to increased targeting of jackass morwong during the 1930s and later years of the steam trawl fishery (Klaer, 2001). Annual estimates of landings of jackass morwong from the steam trawl fishery in the

east between 1915 and 1957 reached a peak of about 2,000 t during the late 1940s (Day and Castillo-Jordán, 2018b).

The fishery expanded greatly during the 1950s, with Danish seine vessels becoming the main vessels in the fishery. Landings of jackass morwong in NSW and eastern Victoria increased following WWII, and, at their peak in the 1960s, annual landings were of the order of 2,500 t. The fishery shifted southwards during this time, with the majority of the landed catches coming from eastern Victoria. Landings of morwong then dropped to around 1,000 t by the mid-1980s (Table 8.4), with landings in eastern Tasmania becoming an increasing proportion of catches. By the mid-1980s, the majority of jackass morwong was being landed by modern otter trawlers; with small landings by Danish seine vessels in eastern Victoria and eastern Bass Strait (Smith and Wayte, 2002). Catches were not recorded in the west (SESSF zones 40 and 50) until 1986.

Since the introduction of management measures into the South East Fishery in 1985, the recorded catch of jackass morwong has ranged between 111 t in 2015 (102 t in the east and 9 t in the west) to 1,652 t in 1989 (1567 t in the east and 85 t in the west). Annual landings of jackass morwong in the eastern zones declined to around 1,000 t during the 1990s and in 2017 are near their lowest recorded levels (Day and Castillo-Jordán, 2018b). The catches appear to have been constrained by the total allowable catch (TAC) in the periods 2002-2005 and 2008-2011. In 1992, an initial TAC was set at 1,500 t (Smith and Wayte, 2002), with this single TAC set to cover catches in both the east and the west. The agreed TAC was reduced to 1,200 t in 2000, to 960 t in 2003, briefly increased to 1,200 t in 2006, then further decreased to 878t in 2007. Since 2008 the TAC has varied between 450-600t. These changes to the TAC have been in response to stock assessments showing the stock to be at declining levels. The TAC was set at 450 t from 2009-2011 as a bycatch TAC i.e. the amount of unavoidable bycatch of morwong that could be expected from fishing for other species. Klaer and Smith (2008) calculated that in 2006, 59% of morwong trawl catch was caught as bycatch (mainly from flathead fishing). From the logbook data in 2006, the morwong trawl catch was 763 t. Thus 59% of this, or 450 t, is bycatch that is unavoidable if catches of species that have morwong as a bycatch stay the same as 2006 levels (Wayte, 2011).

Catches of jackass morwong in the west have been recorded since 1986 (153 t) with less than 100t caught annually in the west from 1987-1999, then catch totals exceeding 100t in the period 2000-2008 (with a peak of 320 t in 2001). All catches have been less than 100t since 2009, indeed less than 50 t in the period 2012-2016, with a 2017 western catch of 87 t. While the western catches were not included in stock assessments conducted before 2007, the TAC has always been set for the combined eastern and western stocks. Since 2007, the recommended biological catches (RBC) used to determine the TAC (for the combined stock) is simply the sum of the RBC for the eastern stock and the RBC for the western stock. The eastern and western stocks have been managed under a single TAC, so an RBC of zero for the eastern stock, (combined with a non-zero RBC from the western stock) still allowed a non-zero TAC to be set for the combined stock, and allowed some of that TAC to be taken in the eastern part of the stock.

Morwong is also caught in small quantities in state waters off NSW and Tasmania, and by the non-trawl sector of the fishery, although these landings are not large. This assessment does not consider landings from vessels in the non-trawl sector. The state catches have been added to the Commonwealth catches in the appropriate zone.

The assessment data for the western stock of jackass morwong comprises a single western trawl fleet. In the west, 50% recruitment to the fishery occurs at around 8 years old, compared to between three and seven years in the east.

8.2.2 Stock Structure

Genetic studies conducted by the CSIRO have found no evidence of separate stocks of jackass morwong in Australian waters. New Zealand and Australian stocks are however, distinct (Elliott et al., 1992). Analysis of otolith microstructure (Proctor et al., 1992) found differences between jackass morwong from southern Tasmania and those off NSW and Victoria, but it is unclear if such differences indicate separate stocks. Differences among jackass morwong in the western and eastern zones have been suggested (D.C. Smith, MAFRI, pers. comm. 2004; I. Knuckey, Fishwell, pers. comm. 2004), and it is assumed for the purposes of this assessment that there are separate stocks of jackass morwong in the eastern and western zones (Wayte, 2011).

8.2.3 Previous Assessments

Smith (1989) analysed catch and effort data for the Eden fishery (1971-72 to 1983-84), finding a significant decline in catch-per-unit-effort (CPUE) to 1980. Lyle (1989) analysed logbook data for Tasmania and western Bass Strait from 1976-84. No trends were apparent in these data.

The biomass of jackass morwong in the eastern zone was estimated to be about 10,000 t in the mid-1980s (Smith, 1989), using a combination of trawl surveys and VPA. Age-structured modelling of the NSW component of the fishery indicated that Maximum Sustainable Yield (MSY) is approached with a fishing mortality (F) between 0.2 and 0.3 yr⁻¹, and that the fishery was at optimum levels in the mid-1980s (Smith, 1989).

At the 1993 meeting of SEFSAG, the recent age data (from the Central Ageing Facility, CAF) and length data were presented together with new age and length data from southeastern Tasmania. Estimates of total mortality from catch curve analyses were similar to previous estimates in the early 1980s. Length and age data from southeastern Tasmania were characterised by a greater proportion of larger and older fish. Preliminary ageing data from sectioned otoliths were tabled at SEFAG in 1994 which suggested that morwong were longer lived (35 years) than previously thought (20 years).

In 1995, catch and unstandardised effort by major area in the fishery were derived from logbook records for the period 1986-94. Whereas the 1994 assessment stated that catch rates had remained relatively stable for the previous 4 years, GLM-standardized trawl catch rates exhibited a slow decline from 1987. Indeed, Smith and Wayte (2002) note that the mean unstandardised catch rate of jackass morwong has continued to decline, and, since 1996, has triggered AFMA's catch rate performance criterion.

An assessment in 1997 was based on the collation and analysis of catch and effort data, combined with new biological information on growth rates of jackass morwong. Information on length frequencies and the retained and discarded catch of jackass morwong was obtained from SMP data and the FRDC report by Liggins (1996). Further length-frequency data were available from NSW and Tasmanian state projects. Catch curve analysis on fish between 5 and 26 years old produced an estimate for total mortality of 0.18 yr⁻¹. This was considerably lower than previous estimates of 0.6 to 0.77 yr⁻¹ and was a direct result of the "new" maximum age. It is also lower than the values obtained by applying the 1993/94 age-length key (0.3 yr⁻¹) to length composition data. Using a value for M of 0.09 yr⁻¹, a fishing mortality (F) of 0.09 yr⁻¹ was estimated.

Klaer (2006) used a stock reduction analysis (SRA) method to model the population of jackass morwong off NSW using catch history data from 1915-61. This analysis led to a point estimate of

unexploited total recruited biomass of 29,400 tonnes, which is larger than spawning biomass, with a 1961 depletion level of 70%.

The first formal quantitative assessment of jackass morwong was conducted by Fay (2004) based on data to 2002, using Coleraine, a stock assessment software package. It used a generalised age-structured modelling approach to assess the status and trends of the jackass morwong trawl fishery in the eastern zones, using data from the period 1915-2002. The 2004 assessment indicated that the spawning biomass of jackass morwong was between 25-45% of the 1915 unexploited biomass. The base-case model estimated the current spawning biomass was 37% of the unexploited biomass. The model could not adequately reconcile changes in catch rates in the late 1980s with catches during this period.

The 2004 assessment was updated in 2006 using Coleraine with additional data that had become available since the previous assessment (Fay, 2006). Two recent (1986-2005) catch rate series were explored in the 2006 assessment. ShelfRAG originally chose to use a catch rate standardisation that was restricted to vessels which caught jackass morwong for at least 5 years and had a median annual catch of at least 5 t. Only shots in which at least 30 kg of jackass morwong were caught were included. The new standardized catch rate time series, which was chosen to be consistent with other SESSF species, also endeavoured to select targeted shots by selecting shots with ≥ 1 kg of morwong from vessels that had reported catches of morwong for three or more years and whose median annual catch was greater than 2 tonnes.

Base-case estimates of spawning depletion in 2006 when the model was fit to the ≥ 1 kg catch rate series indicated that the stock was at a low level, around 15% of the unexploited equilibrium state. This led to RBCs in 2007 of zero under all Tier 1 and Tier 2 harvest control rules (HCRs). If the model was fitted to the new age and length data but used the ≥ 30 kg catch rate index, estimates of current stock status were more optimistic, with spawning depletion in 2006 estimated to be 35% of the unexploited state. This assessment also recommended “accounting for the western areas of the SESSF” in future assessments.

The results of the 2006 assessment were clearly sensitive to the catch and effort data used to calculate a catch rate index that is representative of changes in biomass. As the estimated population trend is primarily driven by this catch rate index, the choice of data included is key to estimates of stock status for this population. For the 2004 assessment, it was considered that a ≥ 30 kg cut-off for catch and effort data was reasonable for morwong. However, the increasing trend in the number of shots catching small amounts of morwong from those vessels targeting the species (Day 2006) suggests that this might not be the case. The analysis by Day showed that the increase in small shots is not due to a change in reporting practices. In 2006 ShelfRAG decided to use the ≥ 1 kg catch rate as input to the base-case, as this was the more precautionary approach, no evidence against using this series was presented, and it is consistent with the approach used for other SESSF species.

The 2007 base-case assessment (Wayte and Fay, 2007) for the eastern stock estimated that the 2008 spawning stock biomass was 19% of unexploited stock biomass. This assessment was largely driven by the recent catch rate indices, which indicated a 70% decline in the stock over the last 20 years. The age and length data when fitted in the absence of the catch rate indices did not indicate the same magnitude of decline. In order to fit to the catch rate indices, the model estimated that recruitments were largely below average in the last 25 years, although there was some evidence for an above average recruitment in 2003. Depletion across all sensitivities varied between 11% and 28%.

A preliminary assessment for the western stock in 2007 indicated that the stock had declined in recent years as fishing pressure has increased, but spawning stock biomass was 63%, still considerably higher than the target level. The long-term RBCs estimated for the western stock were comparable with the 2007 catch levels. The single RBC calculated for jackass morwong (combining the east (0t) and west (297t) stocks) was 297t (using the 20:40:48 control rule), with this RBC coming entirely from the western part of the stock. The TAC was set allowing for unavoidable bycatch of jackass morwong in the east.

The 2008 base-case assessment for the eastern stock (Wayte and Fay, 2008) estimated that the 2009 spawning stock biomass was 19% of unexploited stock biomass. The 2007 assessment had estimated good recruitments for both 2003 and 2004. However, the limited amount of 2007 data used in the 2008 assessment did not support the high 2004 recruitment estimate. Several data types were not available for 2007, and, for the data that were available, sample sizes were lower than in previous years. The 2008 CPUE indices indicated that the stock abundance was unchanged from the previous year.

The 2008 base-case assessment for the western stock (Wayte and Fay, 2008), was still considered to be preliminary, due to limited data, and estimated that the 2009 spawning stock biomass was 68% of unexploited stock biomass. The single RBC calculated for jackass morwong (combining the east (0t) and west (381t) stocks) was 381t (using the 20:35:48 control rule), with this RBC coming entirely from the western part of the stock.

The 2009 assessment (Wayte, 2009) estimated recruitment deviations up to four years before the end of the data instead of two years as in previous assessments. This change was made because it was recognised that fish spawned two and three years before the end of the data will not be well-represented in the data, and this problem had been compounded in the years leading up to the 2009 assessment by poor data collection. The eastern trawl CPUE index showed a slight increase, and the 2003 recruitment continued to be estimated as above average – leading to a slight recovery in the current status of the stock to above the limit reference level (24%). Catch rates had declined in recent years, despite lower catches than in the past. To reconcile this information the 2009 base-case assessment estimated recruitments to have been consistently below average since the early 1980s. The 2009 assessment examined two other possible reasons for this decline: that recruitment is more closely related to stock size than previously assumed (i.e. steepness is lower); or that a regime shift has occurred. Both these models led to a better fit to the data than the base-case, but neither were accepted as a new base-case. The best estimate of lower steepness was considered to be unrealistically low for a Perciforme species such as morwong (Myers et al 1999). The regime shift model gave a more optimistic picture of current stock status than the other models, but the long term catch estimate was greatly reduced. It was considered that more evidence for the existence of a regime shift was required before this model was considered plausible.

The 2009 base-case assessment for the western stock (Wayte, 2009), was considered to be increasingly uncertain, with no recent length frequency data (for 2007 and 2008), and estimated that the 2010 spawning stock biomass was 70% of unexploited stock biomass. The single RBC calculated for jackass morwong (combining the east (143t) and west (367t) stocks) increased to 510t, with this RBC coming from both the eastern and western part of the stock.

The 2010 base-case assessment for the eastern stock (Wayte, 2010) estimated that current spawning stock biomass was 26% of unexploited stock biomass. Concern was expressed that catches in the east had continued to be above the eastern component of the (combined) RBC. The western stock assessment continued to be considered as increasingly uncertain, with no recent length frequency data

(for 2007-2009). Catches of morwong in the Great Australian Bight were found to be at a similar level to western morwong catches, but it is not known whether the GAB morwong form a separate stock.

In 2010 the RAG decided to include both port and onboard retained length frequency data (for both historic and current years) in future assessments, whereas previously only port data had been used. The 2010 assessment was run with this change in length frequency data (as well as any other changes to the data up to 2009), and very little change to the assessment result was seen. At the ShelfRAG meeting on October 3-4 2011, an alternative base-case assuming that eastern jackass morwong has undergone a shift to lower recruitment was presented and accepted and was used as the base-case for the eastern assessment (Wayte, 2011). The justification for this switch is well described in Wayte (2011), including MSE testing implications of assuming (or not) the recruitment shift. The western assessment uses the same assumptions as in previous years (no recruitment shift).

The 2010 base-case assessment for the western stock (Wayte, 2010), continued to be considered increasingly uncertain, with no recent length frequency data (for 2007-2009), and estimated that the 2010 spawning stock biomass was 70% of unexploited stock biomass. The single RBC calculated for jackass morwong (combining the east (228t) and west (329t) stocks) increased to 557t, with this RBC coming from both the eastern and western part of the stock.

The 2011 base-case assessment for the eastern stock (Wayte, 2011) accepted that there was a productivity shift for the eastern stock of jackass morwong and estimated that current spawning stock biomass was 35% of 1988 equilibrium stock biomass. The western stock assessment continued to be considered as increasingly uncertain, with no recent length frequency data (for 2007-2010).

The 2011 base-case assessment for the western stock (Wayte, 2011), continued to be considered increasingly uncertain, with no recent length frequency data (for 2007-2010), and estimated that the 2011 spawning stock biomass was 67% of unexploited stock biomass. The single RBC calculated for jackass morwong (combining the east (358t) and west (282t) stocks) increased to 640t, with this RBC coming from both the eastern and western part of the stock.

The 2015 base-case assessment for the eastern stock (Tuck et al., 2015a) estimated that current spawning stock biomass was 37% of 1988 equilibrium stock biomass. The western stock assessment (Tuck et al., 2015b) continued to be considered as increasingly uncertain, with no length frequency data for 2007-2010, limited age data, low samples size for length compositions, very low catches and conflict between the length and catch rate data. In this assessment, growth parameters were not estimated, and instead were fixed at the values estimated from the eastern assessment. The 2015 spawning stock biomass was estimated to be 69% of unexploited stock biomass. The single RBC calculated for jackass morwong (combining the east (314t) and west (249t) stocks) increased to 563t, with this RBC coming from both the eastern and western part of the stock.

8.2.4 Modifications to the previous assessments

The 2018 assessment uses Stock Synthesis version SS-V3.30.12.00, (Methot et al., 2018), updated from version SS-V3.24U (Methot and Wetzel, 2013) that was used in the 2015 assessment. New catch, discard, length and conditional age at-length data is available from the three year period from 2015-2017. In addition to these new and updated data, there is an updated standardised CPUE series for the western trawl fleets (Zones 40 and 50), each with three additional data points and updated estimates for the ageing error matrix.

8.2.4.1 Data-related issues

1. Length-frequency data are included separately for onboard and port data by fleet. Port and onboard fleets share a single selectivity pattern.
2. Length frequency data are weighted by shot or trip numbers rather than numbers of fish measured. A cap of 100 trips and 200 shots was used to set an upper limit on the sample size.
3. There is a single catch-rate time series dating back to 1986, western trawl (SESSF Zones 40 and 50).
4. No state catches have been included for the western assessment, with relevant state catches added into the appropriate fleets in the eastern assessment.
5. The ageing error matrix has been updated.
6. Catch, discard, length-composition, age-at-length, and catch rate data have been added for the period 2015-2017. The historical catch series (up until 2014) was also revised to incorporate changes in the catch database.

8.2.4.2 Model-related issues

1. Growth is assumed to follow a von Bertalanffy type length-at-age relationship, with all four growth parameters fixed at values obtained in the eastern assessment (Day and Castillo-Jordán, 2018b).
2. Natural mortality, M , is fixed (0.15) in the model.
3. Recruitment residuals are estimated from 1989-2012, with the last recruitment event estimated five years before the most recent available data, compared to 3 years before the most recent data in the 2015 assessment.
4. An updated tuning procedure has been used to balance the weighting of each of the data sources that contribute to the overall likelihood function, using Francis weighting for length data (Francis, 2011), Punt weighting for the conditional age-at-length data (Punt, 2017), balancing the CPUE series within Stock Synthesis, and improvements to the recruitment bias ramp adjustment.
5. Discards were estimated separately, using estimates of discard rates and retention estimated from discard length frequencies. The 2015 assessment ignored discarding in the west.
6. Discard rates for Tier 1 assessments are required by fishing fleet. This means that the discard estimates for TAC purposes used for Tier 3 and 4 assessments which are provided in the discard report (Burch et al., 2018) cannot be used in Tier 1 assessments. The discards from Burch et al. (2018) are produced using a set of rules to determine, for the entire quota fishery, whether sufficient data are available to make an annual fishery wide discard estimate. The discard rates calculated for and input to Tier 1 stock assessments are used to fit retention selectivity curves, so individual year values are not greatly influential on model estimated discard rates.
7. The Tier 1 discard estimates have been updated in 2018 to more closely match the discard calculations in Bergh et al. (2009). These estimates use ratios of total discards to (retained + discard) catch on a per shot basis, rather than aggregated across a whole strata, which are then weighted up according to CDR landings within zone and season (N. Klaer, pers. comm.).

The usual process of bridging to a new model by adding new data piecewise and analysing which components of the data could be contributing to changes in the assessment outcome was conducted (Day and Castillo-Jordán, 2018a).

8.3 Methods

8.3.1 The data and model inputs

8.3.1.1 Biological parameters

A single-sex model (i.e. both sexes combined) was used, as the length composition data for jackass morwong are not available by sex.

Age-at-length data was used as an input, with all four parameters of the von Bertalanffy growth equation fixed at the values obtained for the eastern stock (Day and Castillo-Jordán, 2018b). This follows the approach first adopted in the 2015 assessment (Tuck et al., 2015b), which was due to limited data and inconsistencies between different years of data leading to poor fits to the growth curve estimated for the west.

As in the 2015 assessment, M was fixed in the model at 0.15 and the base-case value for the steepness of the Beverton-Holt stock-recruitment relationship, h , is 0.7.

Jackass morwong become sexually mature at a length of about 24.5 cm, when the fish are around four years of age. Maturity is modelled as a logistic function, with 50% maturity at 24.5 cm fixed in the assessment. Fecundity-at-length is assumed to be proportional to weight-at-length. The parameters of the length-weight relationship are obtained from Smith and Robertson (1995) ($a=1.7 \times 10^{-5}$, $b=3.031$).

8.3.1.2 Fleets

The assessment data for the western stock of jackass morwong comprises a single fleet:

1. Western trawl – otter trawlers from SESSF Zones 40 and 50 (1986 – 2017).

8.3.1.3 Landed catches

The model uses a calendar year for all catch data. Landings data come from the Commonwealth logbook records for SESSF Zones 40 and 50, scaled up to the Catch Disposal Records (CDRs), in the same proportion as the ratio of the logbook totals for the same zones to the logbook total for all zones. Annual landed catches used in this assessment are shown in Figure 8.1, Figure 8.2 and listed in Table 8.1.

In order to calculate the RBC for 2019, it is necessary to estimate the calendar year catch for 2018. Without any other information, the 2018 catch was assumed to be the same as the 2017 catch. The recent TAC history, which only applies to the combined eastern and western stocks, is also listed in Table 8.1, alongside the catches of western stock of jackass morwong. The percentage of total catch taken in the west is quite variable, averaging around 20% since 1998, but ranging from 7% (in both 1998 and 2014) to 39% (2017).

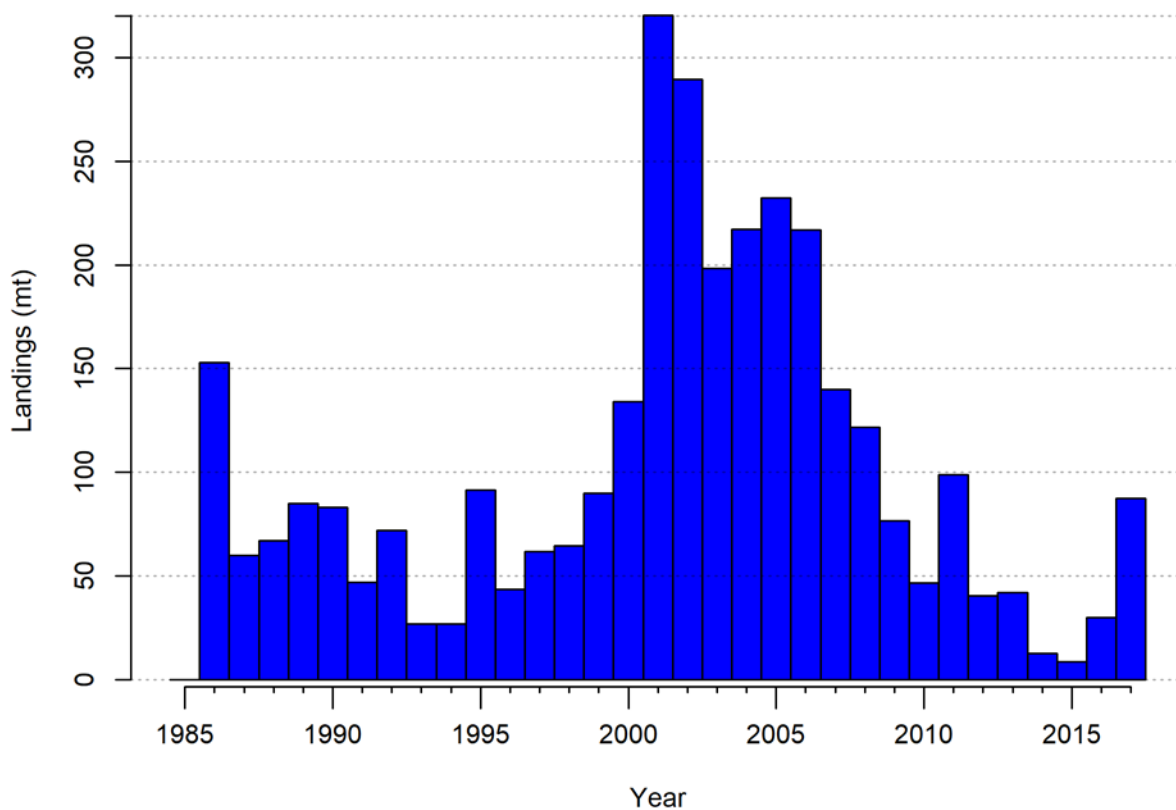


Figure 8.1. Total landed catch (tonnes) of western jackass morwong from 1986-2017 (stacked).

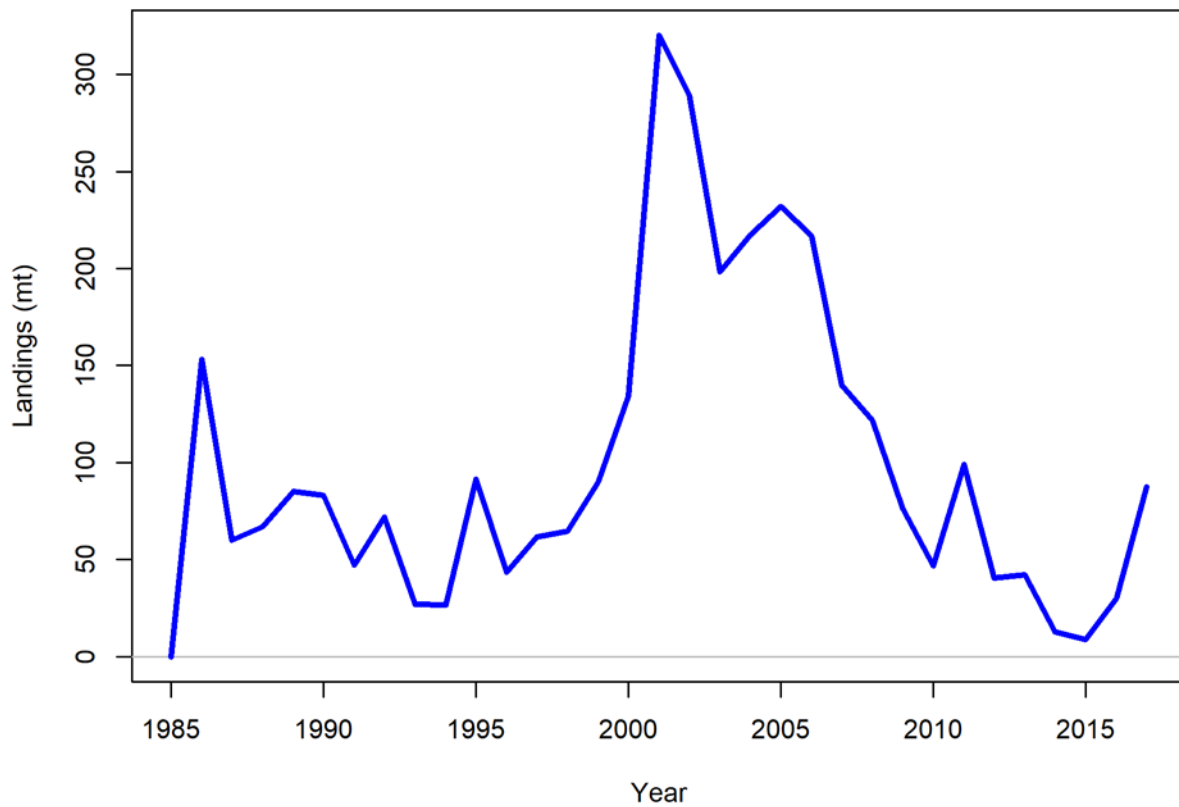


Figure 8.2. Total landed catch (tonnes) of western jackass morwong from 1986-2017 (lines).

Table 8.1. Total retained catches (tonnes) of western jackass morwong for calendar years from 1986-2017 and TAC (combined eastern and western stocks) for 1992-2018.

| Year | Catch | TAC |
|------|-------|------|
| 1986 | 153 | |
| 1987 | 60 | |
| 1988 | 67 | |
| 1989 | 85 | |
| 1990 | 83 | |
| 1991 | 47 | |
| 1992 | 72 | 1500 |
| 1993 | 27 | 1500 |
| 1994 | 27 | 1500 |
| 1995 | 91 | 1500 |
| 1996 | 44 | 1500 |
| 1997 | 62 | 1500 |
| 1998 | 65 | 1500 |
| 1999 | 90 | 1500 |
| 2000 | 134 | 1200 |
| 2001 | 320 | 1185 |
| 2002 | 289 | 950 |
| 2003 | 198 | 960 |
| 2004 | 217 | 960 |
| 2005 | 232 | 960 |
| 2006 | 217 | 1200 |
| 2007 | 140 | 878 |
| 2008 | 122 | 560 |
| 2009 | 77 | 450 |
| 2010 | 47 | 450 |
| 2011 | 99 | 450 |
| 2012 | 41 | 568 |
| 2013 | 42 | 568 |
| 2014 | 13 | 568 |
| 2015 | 9 | 598 |
| 2016 | 30 | 474 |
| 2017 | 87 | 513 |
| 2018 | | 505 |

8.3.1.4 Discard rates

Information on the discard proportions of jackass morwong by fleet is available from the ISMP for 1994-2016. This program was run by PIRVic from 1992-2006 and by AFMA from 2007. These data are summarised in Table 8.2. Discard rates were estimated from on-board data which gives the weight of the retained and discarded component of those shots that were monitored (Burch et al., 2018). Discard proportions vary amongst years and have been as high as 12% (in 2012).

Table 8.2. Discard proportions from 1994 to 2017 with sample sizes for each data point. Entries in grey indicate data that are not used either due to small sample size (less than 10 samples – although there are none of these here) or because the value is too close to zero (less than 0.01).

| Year | discard proportion | n |
|------|--------------------|-----|
| 1994 | 0.0111 | 46 |
| 1995 | | |
| 1996 | 0.0129 | 53 |
| 1997 | | |
| 1998 | 0.0042 | 29 |
| 1999 | | |
| 2000 | 0.0413 | 17 |
| 2001 | 0.0011 | 45 |
| 2002 | 0.0013 | 38 |
| 2003 | 0.0560 | 17 |
| 2004 | 0.0003 | 49 |
| 2005 | 0.0058 | 111 |
| 2006 | 0.0063 | 29 |
| 2007 | | |
| 2008 | 0.0072 | 23 |
| 2009 | 0.0087 | 12 |
| 2010 | | |
| 2011 | 0.0362 | 32 |
| 2012 | 0.1210 | 18 |
| 2013 | 0.0286 | 33 |
| 2014 | 0.0748 | 16 |
| 2015 | | |
| 2016 | 0.0290 | 21 |
| 2017 | 0.0644 | 22 |

Discard practices can be variable between years for reasons that are difficult to model, such as changes in market demands or issues with quota availability, with some years having very low discard rates and others having considerable discard rates. Without a mechanism to explain these years of very low discarding, discarding practices are assumed to be constant through time. Including those years with very low discard rates forces the model to fit very low discard rates to all years, due to the low absolute variation associated with low discard rates, even those years when discarding is known to be higher, and underestimates discarding over all years. As a result, years with very low discard proportions (less than 1%) are excluded as inputs to stock synthesis (the greyed figures in the proportion columns in Table 8.2) giving more believable estimates of discarding in general. Note that any discard estimate coming from a sample size of less than 10 would also be excluded as it is unlikely to be representative of typical discarding practices.

Observations were then used to estimate discard rates (Figure 8.3) and hence discarded catches for each fleet (Figure 8.4), with estimated discard rates of between 3% and 5% for the trawl fleet, and less than 10 t in all years.

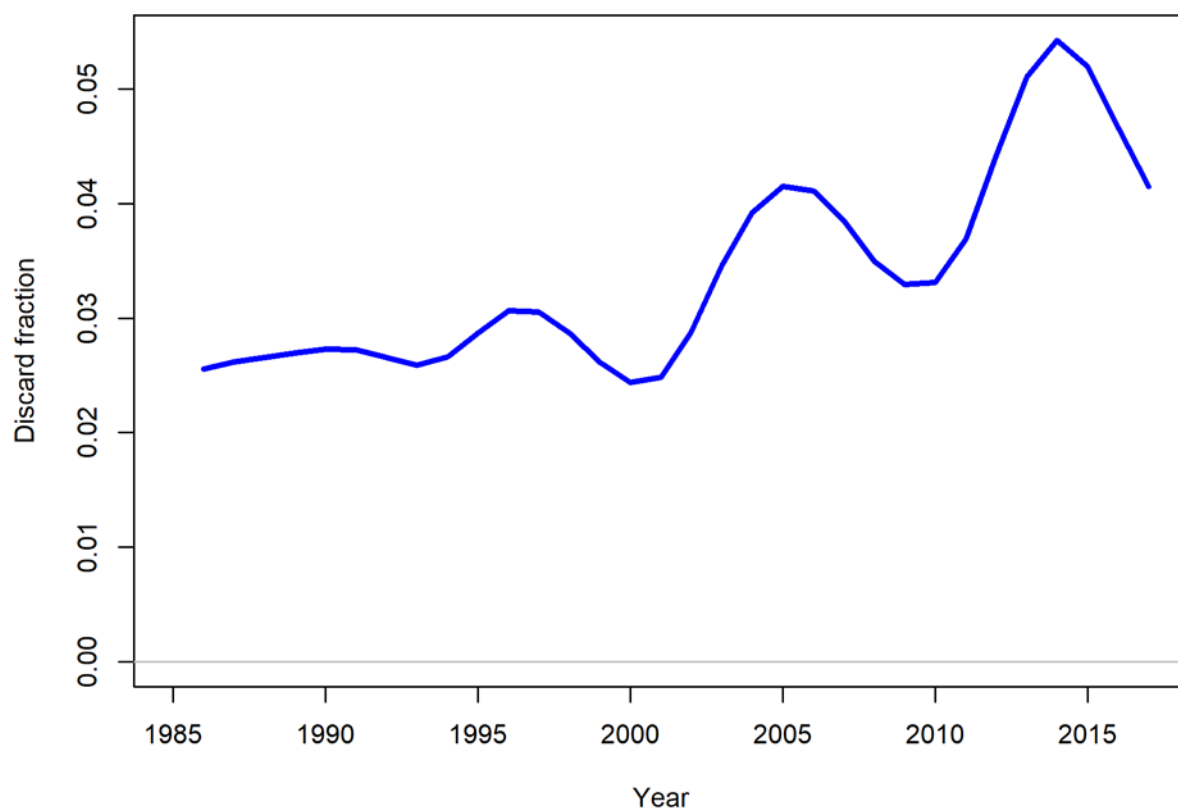


Figure 8.3. Model estimates of discard fractions by fleet, western trawl (blue).

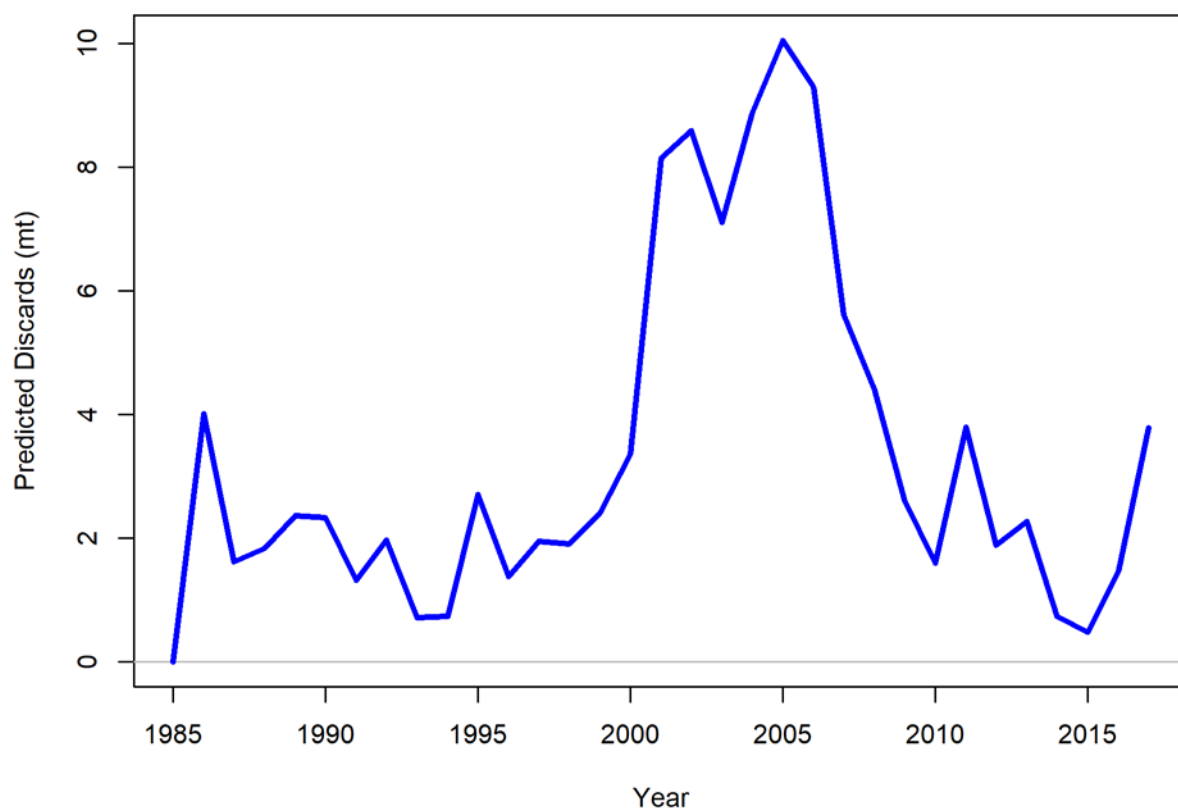


Figure 8.4. Estimated discards (tonnes) of jackass morwong (Zones 40 and 50) in the SESSF from 1986-2017, otter trawl (blue).

8.3.1.5 Catch rate and FIS abundance indices

Catch and effort data from the SEF1 logbook database were standardised using GLMs to obtain indices of relative abundance (Sporcic and Haddon (2018b); Table 8.3) from the period 1986-2017 for the western trawl fleet. In the stock synthesis assessment, the coefficient of variation is initially set at a value equal to the root mean squared deviation from a loess fit (Sporcic and Haddon, 2018a) and additional variance is estimated for this CPUE index to tune the input and output variances.

Table 8.3. Standardised catch rate indices and coefficient of variation (Sporcic and Haddon, 2018b) for the western trawl fleet for western jackass morwong and the FIS abundance indices. The coefficient of variation is initially set at a value equal to the root mean squared deviation from a loess fit (Sporcic and Haddon, 2018a).

| Year | Catch rate | cv | FIS | cv |
|------|------------|-------|--------|-------|
| 1986 | 2.060 | 0.192 | | |
| 1987 | 1.617 | 0.192 | | |
| 1988 | 2.392 | 0.192 | | |
| 1989 | 1.728 | 0.192 | | |
| 1990 | 1.751 | 0.192 | | |
| 1991 | 1.183 | 0.192 | | |
| 1992 | 0.969 | 0.192 | | |
| 1993 | 0.924 | 0.192 | | |
| 1994 | 0.902 | 0.192 | | |
| 1995 | 0.931 | 0.192 | | |
| 1996 | 1.043 | 0.192 | | |
| 1997 | 0.822 | 0.192 | | |
| 1998 | 0.833 | 0.192 | | |
| 1999 | 0.755 | 0.192 | | |
| 2000 | 1.195 | 0.192 | | |
| 2001 | 1.273 | 0.192 | | |
| 2002 | 1.281 | 0.192 | | |
| 2003 | 1.085 | 0.192 | | |
| 2004 | 1.151 | 0.192 | | |
| 2005 | 1.247 | 0.192 | | |
| 2006 | 0.988 | 0.192 | | |
| 2007 | 0.824 | 0.192 | | |
| 2008 | 0.845 | 0.192 | 51.564 | 0.335 |
| 2009 | 0.669 | 0.192 | | |
| 2010 | 0.497 | 0.192 | 25.525 | 0.335 |
| 2011 | 0.525 | 0.192 | | |
| 2012 | 0.392 | 0.192 | 39.263 | 0.335 |
| 2013 | 0.369 | 0.192 | | |
| 2014 | 0.288 | 0.192 | 7.269 | 0.335 |
| 2015 | 0.369 | 0.192 | | |
| 2016 | 0.432 | 0.192 | 7.031 | 0.335 |
| 2017 | 0.664 | 0.192 | | |

8.3.2 Stock assessment method

The restrictions used in selecting data for analysis for Danish seine fleet were: (a) vessels had to have been in the fishery for three or more years, (b) the catch rate had to be larger than zero and (c) catches in zone 40 and 50 only.

Abundance indices for western jackass morwong for the FIS surveys conducted between 2008 and 2016 are provided in Table 8.3. FIS abundance values are reported for all years for jackass morwong for the whole fishery (east and west, Knuckey et al., 2015, Knuckey et al., 2017), but only separated into zones reflecting the fleets used in Tier 1 assessments in 2016 in this report. The 2016 value for

western jackass morwong (Knuckey et al., 2017) is listed in Table 8.3, along with values calculated previously for the earlier FIS years and first reported here. As with the CPUE indices, the coefficient of variation is initially set at a value equal to the root mean squared deviation from a loess fit (Sporcic and Haddon, 2018a) and additional variance is estimated for this abundance index to tune the input and output variances.

8.3.2.1 Length composition data

Port and onboard length composition data are both used separately, with the gear selectivity estimated jointly from both port and onboard data, as is the standard practice in the SESSF stock assessments. For onboard data, the number of shots, is considered to be more representative of the information content in the length frequencies than the number of fish measured. For port data, the number of shots is not available, but the number of trips can be used instead. In the 2018 assessment, the initial sample size associated with each length frequency in the assessment is the number of shots or trips.

Length data were excluded for years with less than 100 individual fish measured, as this was considered to be unrepresentative (with excluded data listed in grey in Table 8.4 and Table 8.5). Sample sizes for retained length frequencies, including both the number of individuals measured and number of trips are listed in Table 8.5 for each fleet and year for the period 1996-2017 and for discarded length frequencies in Table 8.4 for the period 1994-2016.

Length composition information for the retained component of the catch by the western Commonwealth trawl fleet is available from port sampling for the period 1996-2017 and from onboard sampling from 1997-2017. Onboard data collected by the ISMP were used to calculate the length frequency of the discarded component of the catch for six years only from 1994-2016.

Table 8.4. Number of onboard discarded lengths and number of shots for length frequencies included in the base case assessment by fleet 1994-2016. Entries in grey indicate data that are not used due to small sample size (less than 100 fish measured) or, in the case of the 1994 sample, because SERAG decided that the sample looked unrepresentative.

| year | fleet western trawl # fish | (discard) western trawl # shots |
|------|-------------------------------------|--|
| 1994 | 233 | 2 |
| 2009 | 112 | 1 |
| 2011 | 9 | 2 |
| 2012 | 59 | 10 |
| 2013 | 23 | 8 |
| 2016 | 86 | 4 |

Table 8.5. Number of port and onboard retained lengths and number of shots or trips for length frequencies included in the base case assessment by fleet 1996-2017. Entries in grey indicate data that are not used due to small sample size (less than 100 fish measured).

| year | fleet (retained) | | FIS # fish | trawl onboard # shots | trawl port # trips | FIS # shots |
|------|----------------------------|----------------------|---------------|-----------------------------|--------------------------|----------------|
| | trawl onboard # fish | trawl port # fish | | | | |
| 1996 | | 364 | | | 3 | |
| 1997 | 245 | 505 | | 2 | 4 | |
| 1998 | 373 | 2 | | 4 | 1 | |
| 1999 | 412 | 341 | | 4 | 3 | |
| 2000 | 124 | 572 | | 1 | 5 | |
| 2001 | 1434 | 2232 | | 11 | 18 | |
| 2002 | 859 | 1918 | | 4 | 12 | |
| 2003 | 124 | 1680 | | 1 | 10 | |
| 2004 | 397 | 873 | | 3 | 10 | |
| 2005 | 2116 | 1426 | | 15 | 14 | |
| 2006 | 820 | 690 | | 6 | 7 | |
| 2008 | 47 | 109 | 512 | 2 | 1 | 15 |
| 2009 | 140 | | | 4 | | |
| 2010 | 72 | | 300 | 2 | | 16 |
| 2011 | 208 | | | 9 | | |
| 2012 | 318 | | 362 | 17 | | 14 |
| 2013 | 723 | 53 | | 25 | 1 | |
| 2014 | 241 | 61 | 434 | 6 | 1 | 19 |
| 2015 | 151 | | | 3 | | |
| 2016 | 284 | 359 | 366 | 5 | 8 | 15 |
| 2017 | 324 | 210 | | 6 | 5 | |

8.3.2.2 Age composition data

An estimate of the standard deviation of age-reading error was calculated by André Punt (pers. comm., 2018) using data supplied by Kyne Krusic-Golub and a variant of the method of Richards *et al.* (1992) (Table 8.6). Age-at-length measurements provided by Kyne Krusic-Golub of Fish Ageing Services Pty Ltd, are available from 1991-2017 for the western trawl fleet (Table 8.7).

Table 8.6. Standard deviation of age reading error (A Punt pers. comm. 2018).

| Age | sd |
|------------|-----------|
| 0.5 | 0.255696 |
| 1.5 | 0.255696 |
| 2.5 | 0.27765 |
| 3.5 | 0.300684 |
| 4.5 | 0.324851 |
| 5.5 | 0.350208 |
| 6.5 | 0.376813 |
| 7.5 | 0.404727 |
| 8.5 | 0.434015 |
| 9.5 | 0.464744 |
| 10.5 | 0.496985 |
| 11.5 | 0.530813 |
| 12.5 | 0.566306 |
| 13.5 | 0.603546 |
| 14.5 | 0.642618 |
| 15.5 | 0.683613 |
| 16.5 | 0.726626 |
| 17.5 | 0.771756 |
| 18.5 | 0.819106 |
| 19.5 | 0.868787 |
| 20.5 | 0.920913 |
| 21.5 | 0.975604 |
| 22.5 | 1.03299 |
| 23.5 | 1.09319 |
| 24.5 | 1.15636 |
| 25.5 | 1.22264 |
| 26.5 | 1.29218 |
| 27.5 | 1.36514 |
| 28.5 | 1.4417 |
| 29.5 | 1.52202 |
| 30.5 | 1.60629 |

Table 8.7. Number of age-length otolith samples included in the base case assessment for the western trawl fleet 1991-2017.

| Year | otoliths |
|------|----------|
| 1991 | 94 |
| 1992 | 83 |
| 1993 | 42 |
| 1995 | 28 |
| 2003 | 83 |
| 2004 | 474 |
| 2005 | 282 |
| 2006 | 156 |
| 2007 | 51 |
| 2009 | 49 |
| 2011 | 41 |
| 2012 | 87 |
| 2013 | 118 |
| 2014 | 37 |
| 2015 | 71 |
| 2016 | 103 |
| 2017 | 59 |

Implied age distributions for retained and discarded fish are obtained by transforming length frequency data to age data by using the information contained in the conditional age-at-length data from each year and the age-length relationship. Implied age distributions can be calculated separately for both onboard and port fleets and for the retained and discarded length frequencies, and can be calculated from 1997-2017 for the western trawl fleet.

8.3.2.3 Input data summary

The data used in this assessment is summarised in Figure 8.5, indicating which years the various data types were available.

Data by type and year, circle area is relative to precision within data type

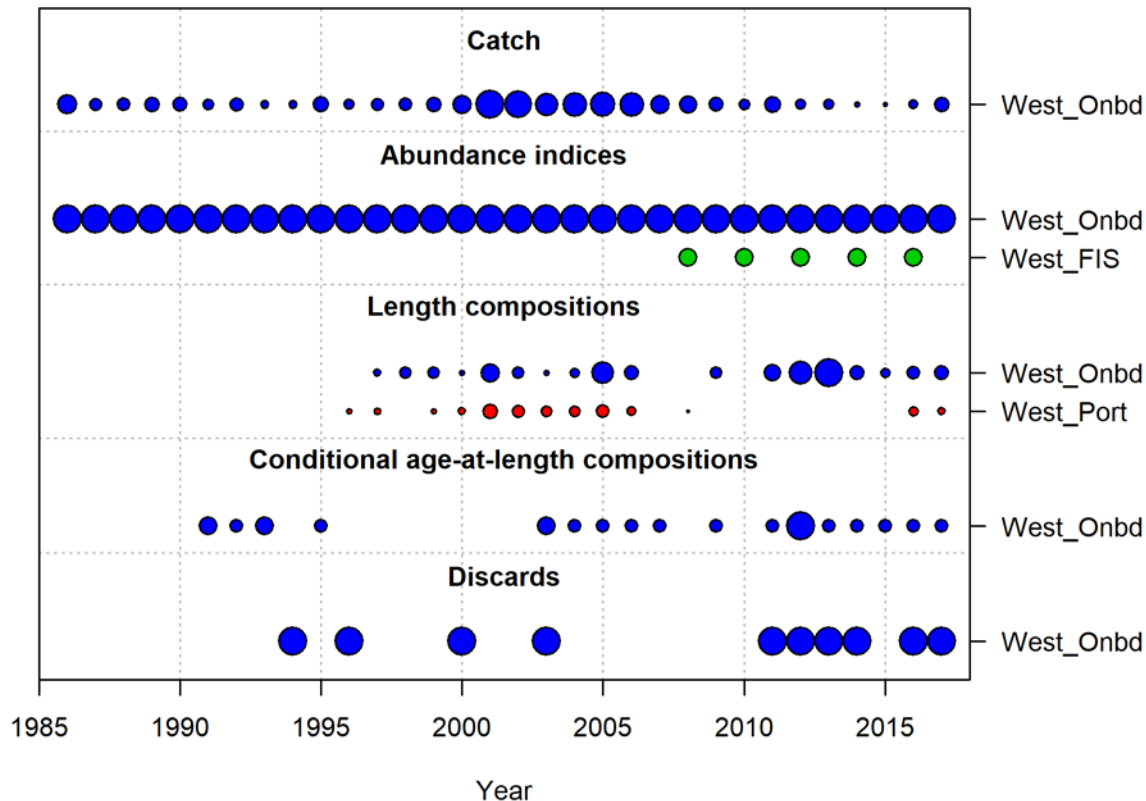


Figure 8.5. Summary of input data used for the western jackass morwong assessment.

8.3.3 Stock assessment method

8.3.3.1 Population dynamics model and parameter estimation

A single-sex stock assessment for western jackass morwong was conducted using the software package Stock Synthesis (version SS-V3.30.12.00, Methot *et al.* 2018). Stock Synthesis is a statistical age- and length-structured model which can allow for multiple fishing fleets, and can be fitted simultaneously to the types of information available for jackass morwong. The population dynamics model, and the statistical approach used in the fitting of the model to the various types of data, are described in the SS technical documentation (Methot, 2005) and are not reproduced here. Some key features of the base-case model are:

- Jackass morwong constitute a single stock within the area of the fishery (SESSF Zones 40 and 50).
- The population was at its unfished biomass with the corresponding equilibrium (unfished) age-structure at the start of 1986.
- The CVs of the CPUE indices for the western trawl fleets were initially set to the root mean squared deviation from a loess fit to the fleet specific indices (Sporcic and Haddon, 2018a) and then tuned to match the model-estimated standard errors by estimating an additional variance parameter within Stock Synthesis.

- d) One fishing fleet is modelled.
- e) The selectivity pattern for the western trawl fleet was modelled as length-specific, logistic and time-invariant. The two parameters of the selectivity function for this fleet were estimated within the assessment.
- f) Retention was also defined as a logistic function of length, and the inflection and slope of this function were estimated.
- g) The rate of natural mortality, M , is assumed to be constant with age, and also time-invariant. The value for M was fixed (0.15) within the model in this assessment.
- h) Recruitment to the stock is assumed to follow a Beverton-Holt type stock-recruitment relationship, parameterised by the average recruitment at unexploited spawning biomass, R_0 , and the steepness parameter, h . Steepness for the base-case analysis is set to 0.7. Deviations from the average recruitment at a given spawning biomass (recruitment residuals) are estimated for 1989 to 2012. Deviations are not estimated prior to 1989 or after 2012 because there are insufficient data to permit reliable estimation of recruitment residuals outside of this time period.
- i) The value of the parameter determining the magnitude of the process error in annual recruitment, σ_R , is set equal to 0.7 in the base case. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Methot and Taylor (2011).
- j) A plus-group is modelled at age thirty years.
- k) Growth of jackass morwong is assumed to be time-invariant, meaning there is no change over time in mean size-at-age, with the distribution of size-at-age being estimated along with the remaining growth parameters within the assessment. No differences in growth related to gender are modelled, because the stock is modelled as a single-sex.
- l) The sample sizes for length and age frequencies were tuned for each fleet so that the input sample size was approximately equal to the effective sample size calculated by the model. Before this retuning of length frequency data was performed by fleet, any sample sizes with a sample size greater than 100 trips or 200 shots were individually down-weighted to a maximum sample size of 100 and 200 respectively.

8.3.3.2 Relative data weighting

Iterative reweighting of input and output CVs or input and effective sample sizes is an imperfect but objective method for ensuring that the expected variation is comparable to the input (Pacific Fishery Management Council, 2018). This makes the model internally consistent, although some argue against this approach, particularly if it is believed that the input variance is well measured and potentially accurate. It is not necessarily good to down weight a data series just because the model does not fit it, if in fact, that series is reliably measured. On the other hand, most of the indices we deal with in fisheries underestimate the true variance by only reporting measurement and not process error.

Data series with a large number of individual measurements such as length or weight frequencies tend to overwhelm the combined likelihood value with poor fits to noisy data when fitting is highly partitioned by area, time or fishing method. These misfits to small samples mean that apparently simple series such as a single CPUE might be almost completely ignored in the fitting process. This model behaviour is not optimal, because we know, for example, that the CPUE values are in fact derived from a very large number of observations.

Length compositions were initially weighted using trip and shot numbers, where available, instead of numbers of fish measured and by adopting the Francis weighting method (Francis 2011) for age and length composition data.

Shot or trip number is not available for all data, especially for some of the early length frequency data. In these cases, the number of trips was inferred from the number of fish measured using the average number of fish per trip for the relevant gear type for years where both data sources were available. The number of trips were also capped at 100 and the number of shots capped at 200. Samples with less than 100 fish measured per year were excluded.

These initial sample sizes, based on shots and trips, are then iteratively reweighted so that the input sample size is equal to the effective sample size calculated by the model using the Francis weighting method for length data and the Punt weighting method for conditional age-at-length data.

8.3.3.3 *Tuning procedure*

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size is equal to the effective sample size calculated by the model. In SS-V3.30 there is an automatic adjustment made to survey CVs (CPUE).

1. Set the standard error for the relative abundance indices (CPUE, acoustic abundance survey, or FIS) to their estimated standard errors for each survey or for CPUE (and FIS values) to the root mean squared deviation of a loess curve fitted to the original data (which will provide a more realistic estimate to that obtained from the original statistical analysis). SS-V3.30 then re-balances the relative abundance variances appropriately.
2. The initial value of the parameter determining the magnitude of the process error in annual recruitment, σ_R , is set to 0.7, reflecting the variation in recruitment for jackass morwong. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Methot and Taylor (2011).

An automated tuning procedure was used for the remaining adjustments. For the conditional age-at-length and length composition data:

3. Multiply the initial sample sizes for the conditional age-at-length data by the sample size multipliers using the approach of Punt (2017).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length composition data using the 'Francis method' (Francis, 2011).
5. Repeat steps 3 and 4, until all are converged and stable (proposed changes are < 1%).

This procedure may change in the future after further investigations but constitutes current best practice.

8.3.3.4 *Calculating the RBC*

The SESSF Harvest Strategy Framework (HSF) was developed during 2005 (Smith *et al.* 2008) and has been used as a basis for providing advice on TACs in the SESSF quota management system for fishing years 2006-2016. The HSF uses harvest control rules to determine a recommended biological

catch (RBC) for each stock in the SESSF quota management system. Each stock is assigned to one of four Tier levels depending on the basis used for assessing stock status or exploitation level for that stock. Jackass morwong is classified as a Tier 1 stock as it has an agreed quantitative stock assessment.

The Tier 1 harvest control rule specifies a target and a limit biomass reference point, as well as a target fishing mortality rate. Since 2005 various values have been used for the target and the breakpoint in the rule. In 2009, AFMA directed that the 20:40:40 (B_{lim} : B_{MSY} : F_{targ}) form of the rule is used up to where fishing mortality reaches F_{48} . Once this point is reached, the fishing mortality is set at F_{48} . Day (2008) determined that for most SESSF stocks where the proxy values of B_{40} and B_{48} are used for B_{MSY} and B_{MEY} respectively, this form of the rule is equivalent to a 20:35:48 (B_{lim} : Inflection point: F_{targ}) strategy.

This document reports RBCs calculated under the 20:35:48 strategy.

8.3.3.5 Sensitivity tests and alternative models

A number of tests were used to examine the sensitivity of the results of the model to some of the assumptions and data inputs:

1. $M = 0.1 \text{ yr}^{-1}$.
2. $M = 0.2 \text{ yr}^{-1}$.
3. $h = 0.6$.
4. $h = 0.8$.
5. 50% maturity at 22 cm.
6. σ_R set to 0.65.
7. σ_R set to 0.75.
8. Estimate growth.
9. Double the weighting on the length composition data.
10. Halve the weighting on the length composition data.
11. Double the weighting on the age-at-length data.
12. Reduce the weighting on the age-at-length data.
13. Double the weighting on the survey (CPUE) data.
14. Halve the weighting on the survey (CPUE) data.
15. Exclude the Fishery Independent Survey abundance indices.
16. Include the Fishery Independent Survey length frequency data and estimate selectivity for the FIS.

The results of the sensitivity tests are summarized by the following quantities (Table 8.11):

1. SSB_0 : the average unexploited female spawning biomass.
2. SSB_{2019} : the female spawning biomass at the start of 2019.
3. SSB_{2019}/SSB_0 : the female spawning biomass depletion level at the start of 2019.

4. Mortality: the model estimated value for mortality.
5. RBC_{2019} : the recommended biological catch (RBC) for 2019.
6. $RBC_{2019-21}$: the mean RBC over the three years from 2019-2021.
7. $RBC_{2019-23}$: the mean RBC over the five years from 2019-2023.
8. $RBC_{longterm}$: the longterm RBC.

The RBC values were calculated for the agreed base case only.

8.4 Results and Discussion

8.4.1 The base-case analysis

8.4.1.1 Transition from the 2015 base case to the 2017 base case

Development of a preliminary base case and a bridging analysis from the 2015 assessment (Tuck et al., 2015b), was presented at the September 2017 SERAG meeting (Day and Castillo-Jordán, 2018a), including updating the version of Stock Synthesis and sequentially updating data. This bridging analysis is not repeated in this report.

8.4.1.2 Parameter estimates

Figure 8.6 shows the estimated growth curve for jackass morwong. All growth parameters are fixed in the model, based on values estimated in the 2018 eastern jackass morwong assessment (Day and Castillo-Jordán, 2018b). The parameter values are listed in Table 8.8.

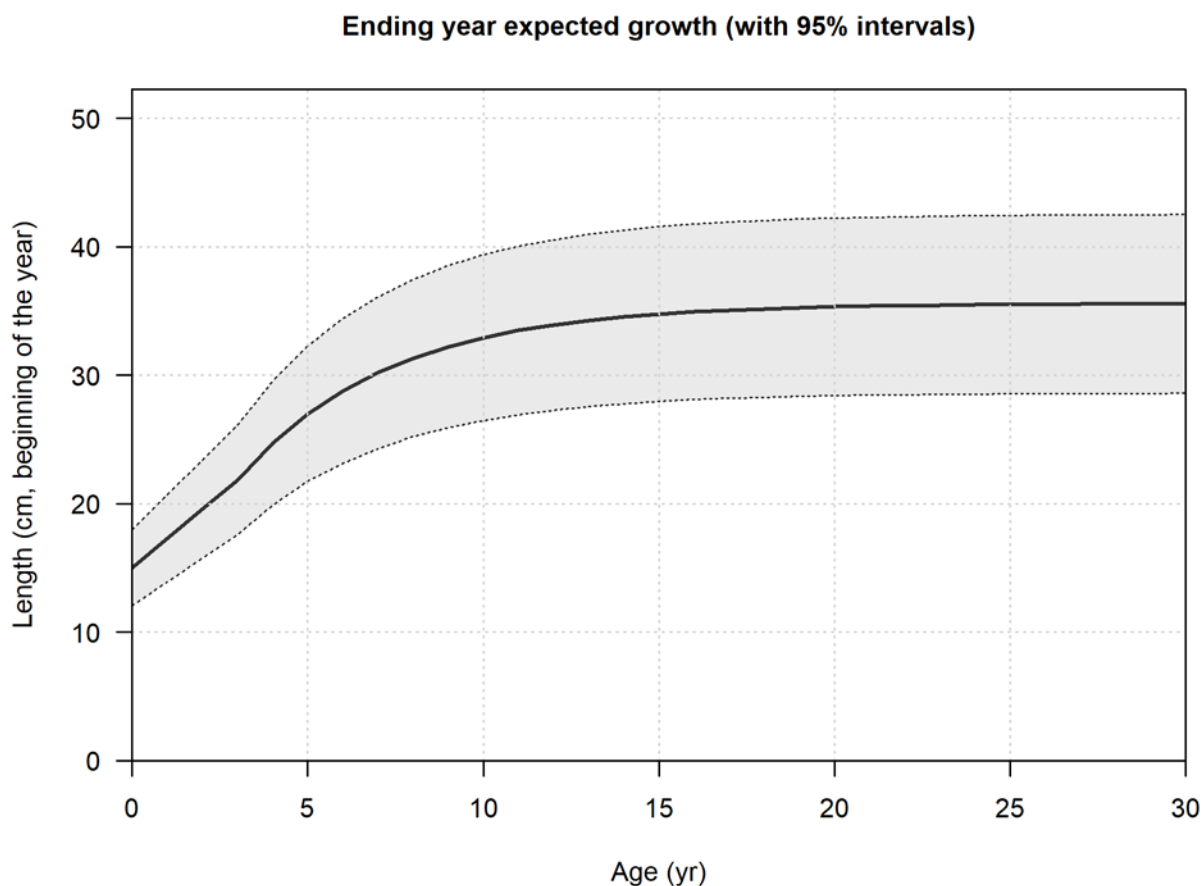


Figure 8.6. Fixed growth curve for western jackass morwong, using parameters estimated from the eastern morwong stock assessment.

Table 8.8. Summary of parameters of the base case model.

| Feature | Details | |
|-----------------------|-----------|--|
| Natural mortality | fixed | 0.15 |
| Steepness h | fixed | 0.7 |
| σ_R in | fixed | 0.7 |
| Recruitment devs | estimated | 1989-2012, bias adjustment ramps 1981-91 and 2017-19 |
| CV growth | fixed | 0.104 |
| Growth K | fixed | 0.217 |
| Growth l_{min} (cm) | fixed | 22 |
| Growth l_{max} (cm) | fixed | 35.2 |

Selectivity is assumed to be logistic for the western trawl fleet. The parameters that define the selectivity function are the length at 50% selection and the spread (the difference between length at 50% and length at 95% selection). The estimates of these parameters for the western trawl fleet are 31.8cm and 6.34cm, slightly larger than the selectivity estimated in the 2015 assessment. Figure 8.7 shows the selectivity and retention functions for each of the commercial fleets. The estimate of the parameter that defines the initial numbers (and biomass), $\ln(R_0)$, is 7.09 for the base case.

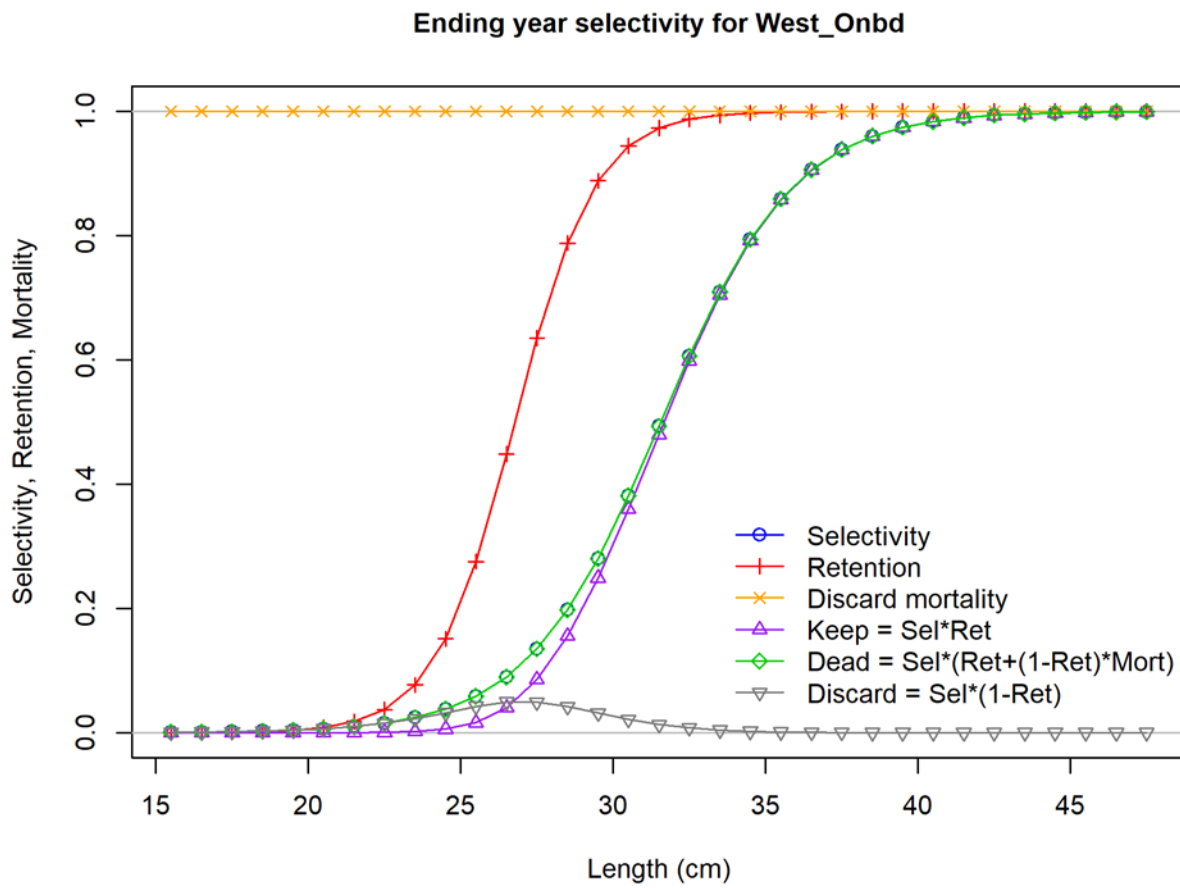


Figure 8.7. Selectivity (blue/green) and retention (red) functions for the western trawl fleet.

8.4.1.3 Fits to the data

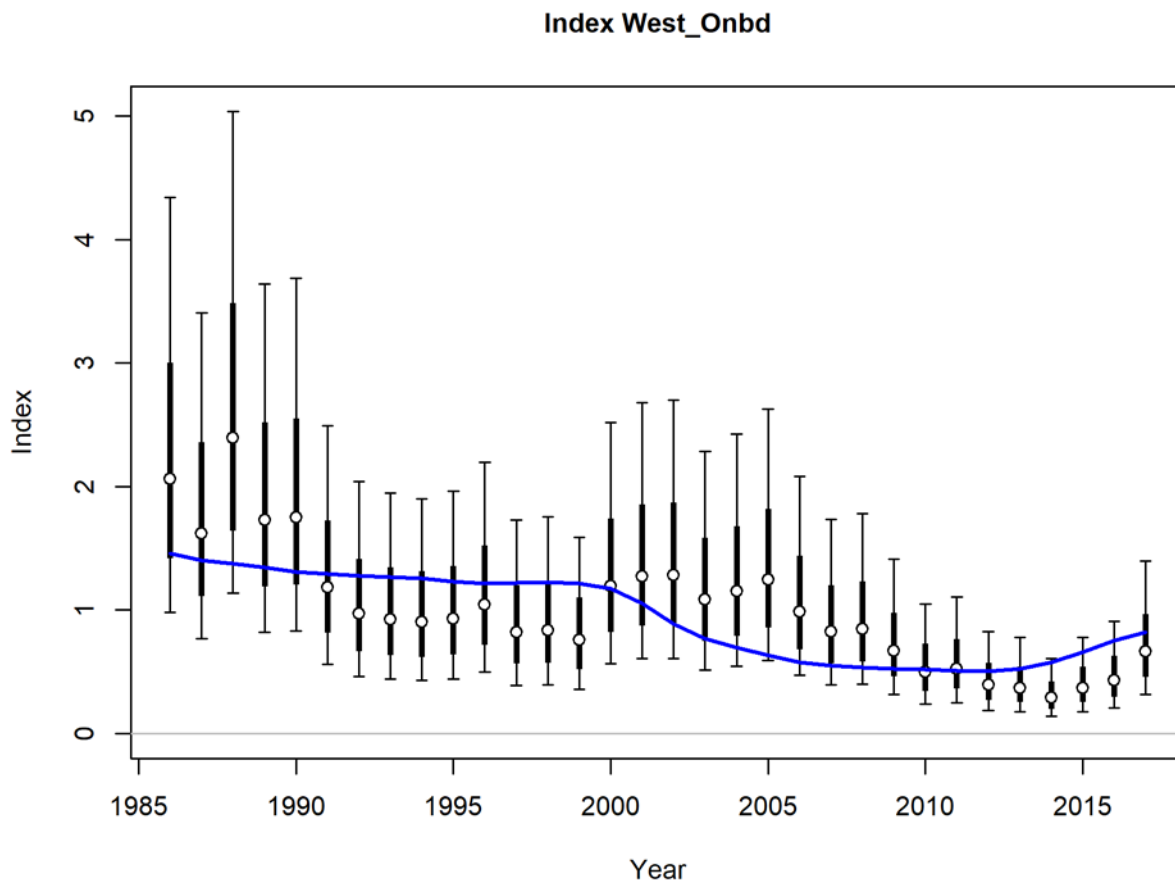


Figure 8.8. Observed (circles) and model-estimated (blue line) catch rates vs year, with approx 95% asymptotic intervals for the western trawl fleet. The thin lines with capped ends should match the thick lines for a balanced model. This index is balanced by estimating an additional variance parameter within Stock Synthesis which in this case is positive, suggesting the model requires more variance than the initial values from the loess fit to achieve a good fit.

The fits to the catch rate indices are poor for the western trawl fleet, with the fitted values all too low from 1986-1990, switching to all too high from 1992-1999, switching to all too low from 2001-2009 and then too high from 2012-2017. Further the fitted values do not really reflect the trends and the changes in the catch rate data, missing the step down seen in the data around 1991 and the step up around 2000, missing the gradual decline from 2005-2014. The only trend that seems reasonably well captured is the short term increase at the end of the series (2014-2017) and the overall decline in the complete time series. The fit in the 2015 assessment was of similar poor quality, as noted by Tuck et al, (2015b), perhaps indicating some conflict between data sources, insufficient quality or quantity of data to enable a quality assessment to be produced or possibly a CPUE series that is not tracking abundance.

It is notable that the standardised catch rate series shows an increase in the recent years (2014-2017), which breaks the pattern seen in recent assessments where the catch rate index continued to decline as new data points were added. From 2008-2014, the FIS abundance series shows a steeper decline than the CPUE series, and the assessment also fails to fit the FIS abundance series well (Figure 8.8). The

last two points in the FIS abundance series (2014 and 2016) do not show the short term increase seen both in the CPUE series (2014-2017) and in the abundance predicted by the model.

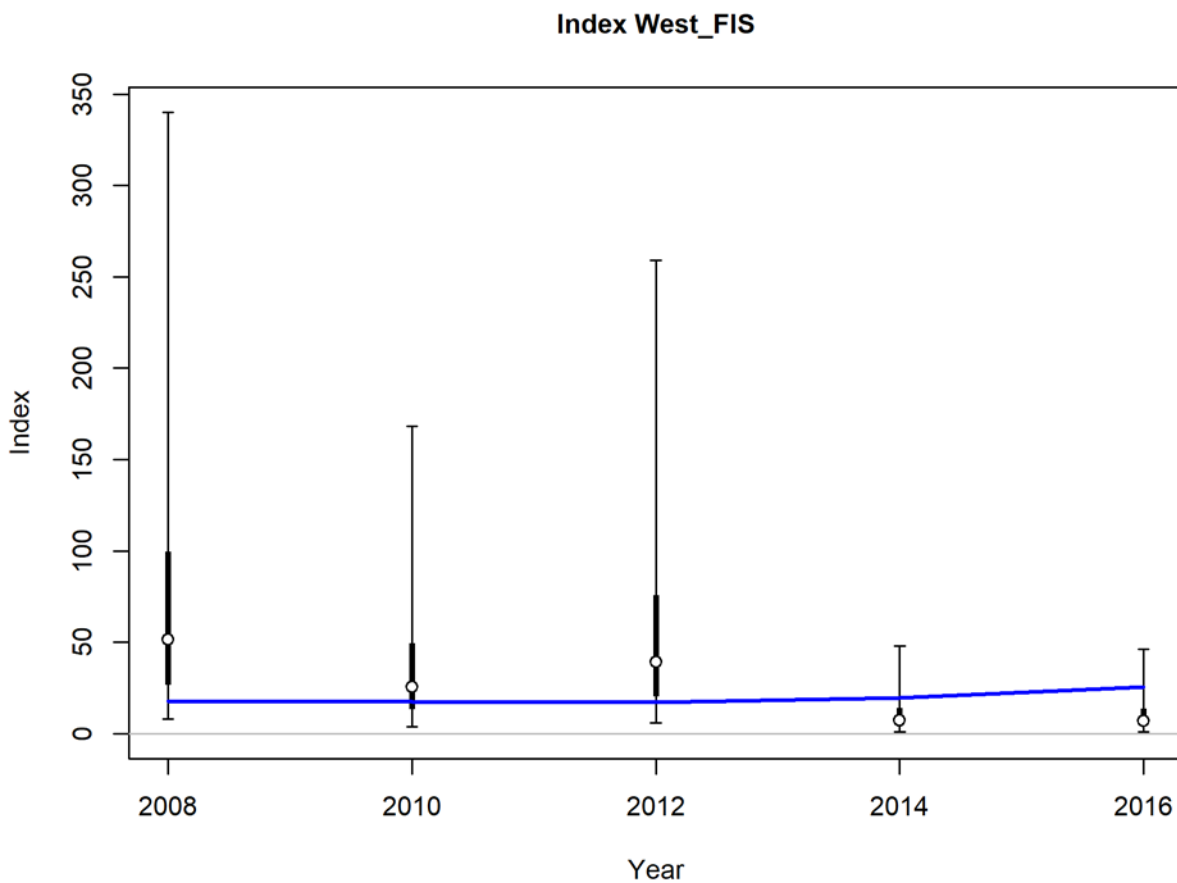


Figure 8.9. Observed (circles) and model-estimated (blue line) catch rates vs year, with approx 95% asymptotic intervals for western FIS. The thin lines with capped ends should match the thick lines for a balanced model. This index is balanced by estimating an additional variance parameter within Stock Synthesis, which in this case is positive and large, suggesting the model requires much more variance than the initial values from the loess fit to achieve a good fit.

The fits to the discard rate data (Figure 8.10) are reasonable, given the variability in the data. The discarding rate and the fits suggest that discarding is generally low (around 5% maximum). Fits to the age and length composition data for discarded catches are shown in Appendix A. Fits to the length composition and conditional age-at-length data seem reasonable, and it appears there is some conflict between fits to the abundance indices and these other data sources.

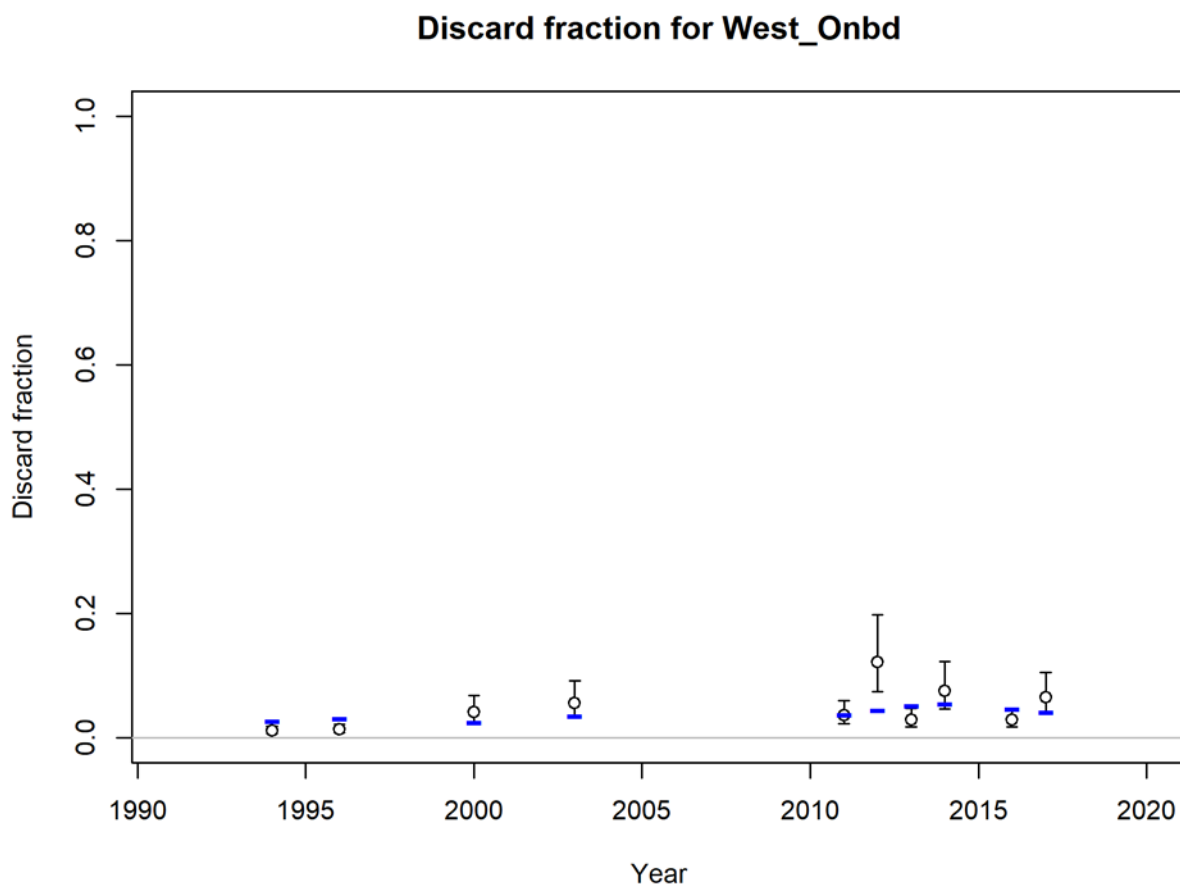


Figure 8.10. Observed (circles) and model-estimated (blue line) catch rates vs year, with approx 95% asymptotic intervals for western FIS. The thin lines with capped ends should match the thick lines for a balanced model. This index is balanced by estimating an additional variance parameter within Stock Synthesis, which in this case is positive and large, suggesting the model requires much more variance than the initial values from the loess fit to achieve a good fit.

The base-case model is able to fit the aggregated (across years) retained length-frequency distributions quite well (Figure 8.11 and Appendix A). The fits to the discard length frequencies come from a single year, and do not fit as well as the retained length data.

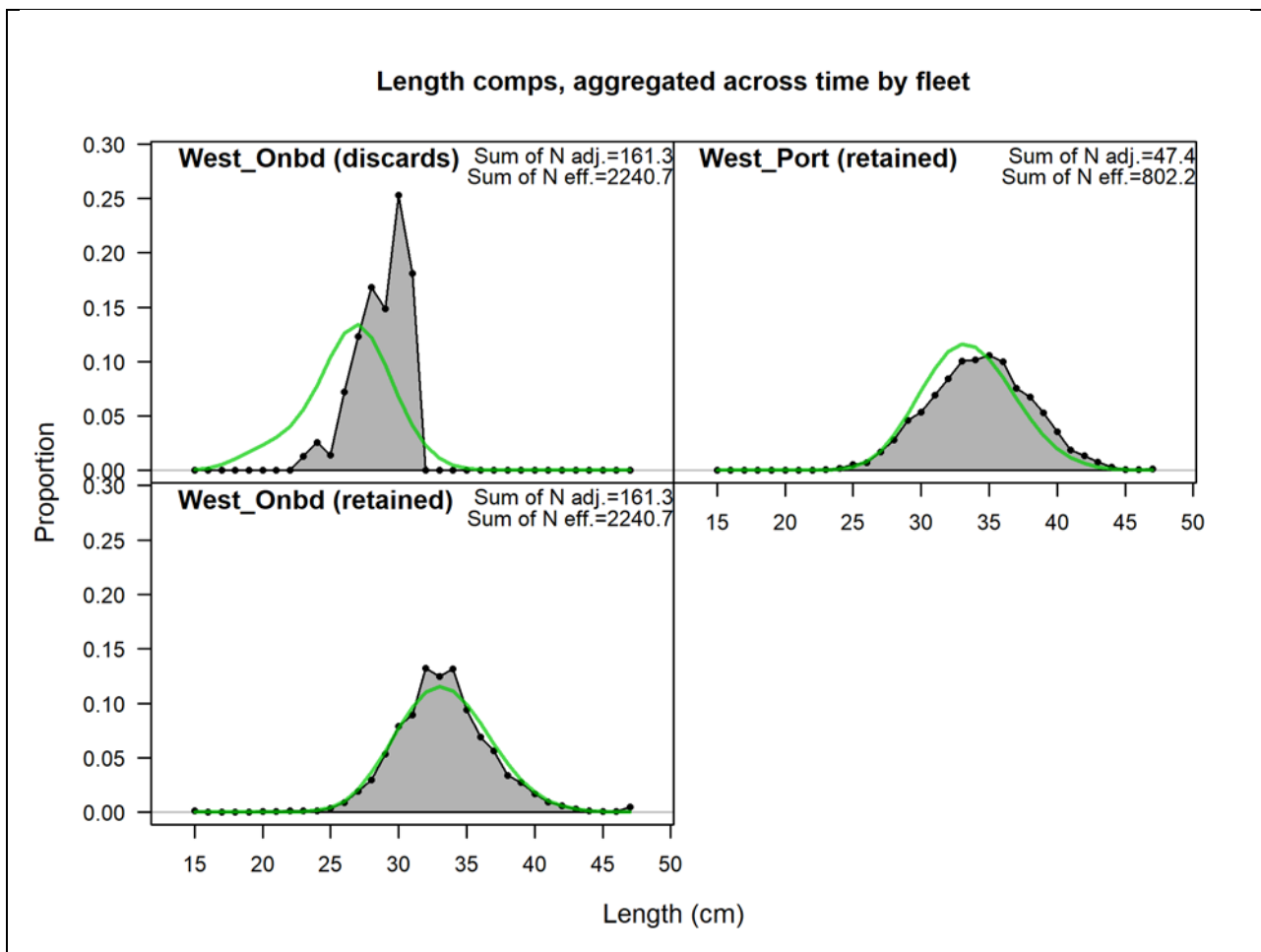


Figure 8.11. Fits to retained and discarded length compositions, separated by port and onboard samples, aggregated across all years. Observed data are grey and the fitted value is the green line.

The implied fits to the age composition data are shown in Appendix A. The age compositions were not fitted to directly, as age-at-length data were used. However, the model is capable of producing implied fits to these data for years where length frequency data are also available, even though they are not fitted directly in the assessment. The model fits the observed age data reasonably well for both retained and discarded age data.

Note that there are separate implied fits to age for the port and onboard data. There is only one set of age data, but this needs to be scaled up to length data (using an age-length key) to get implied fits to age, as the age data is not representative of the stock as a whole. This scaling up to length data can be done using either the onboard length data or the port length data – so it appears that there are two sets of age data.

The conditional age-at-length data is quite noisy between years, with occasionally quite large changes in mean age between adjacent years, in some instances larger changes than would be expected through biology and fishing mortality. The mean age varies between 8 and 15 years for western trawl. This variability in the age-at-length data is likely to be due to spatial or temporal variation in collection of age samples. The fits to conditional age-at-length are as good as can be expected, considering the noise in the data. Residuals for these fits and mean age for each year, aggregated across length bins, are shown in Appendix A.

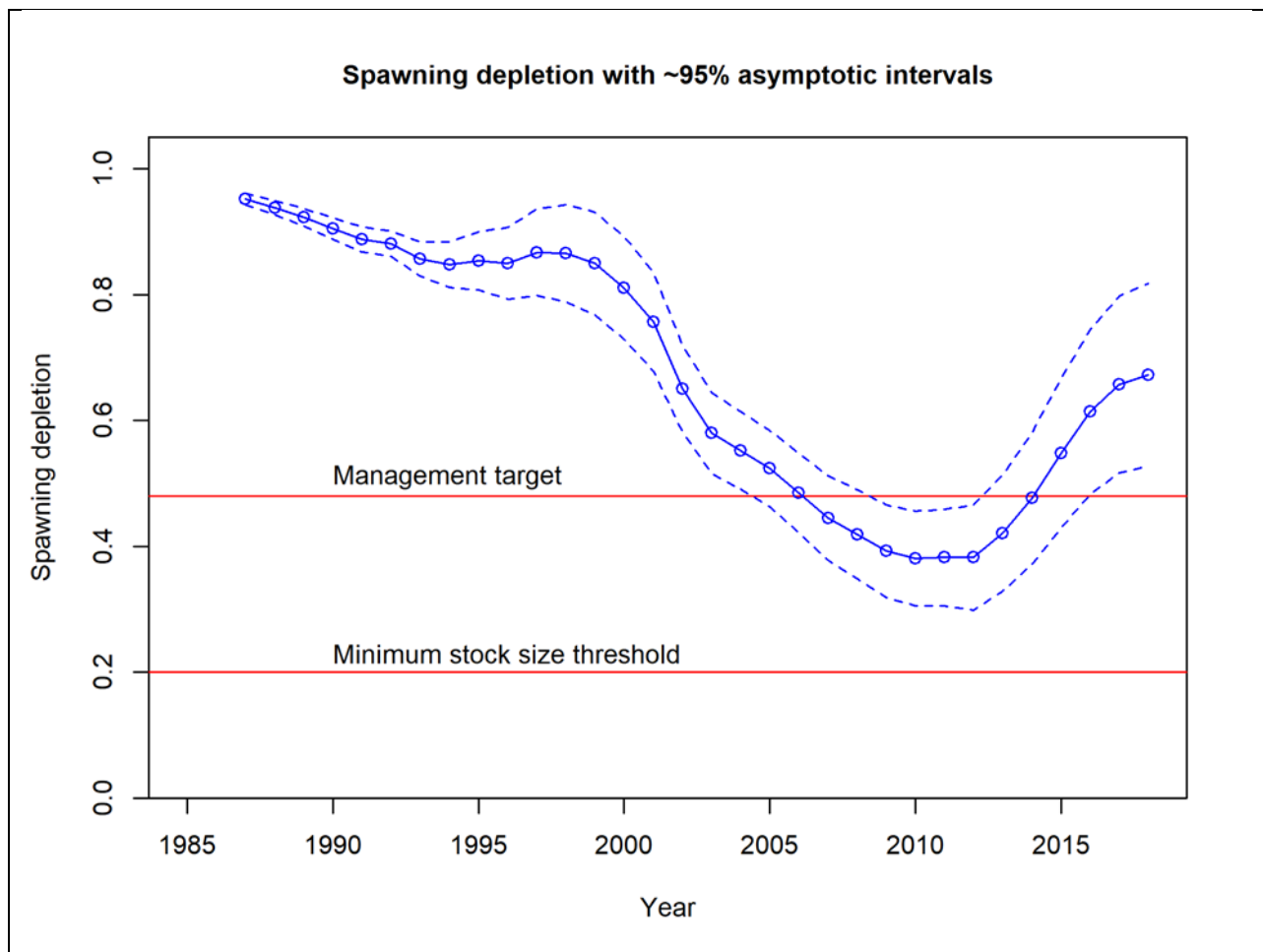


Figure 8.12. Time-trajectory of spawning biomass depletion (with approximate 95% asymptotic intervals) corresponding to the MPD estimates for the 2018 base-case analysis for jackass morwong.

8.4.1.4 Assessment outcomes

The current spawning stock biomass (Figure 8.12) is estimated to be 68% of unfished stock biomass (i.e. 2019 spawning biomass relative to unfished spawning biomass), albeit with considerable uncertainty (with 95% asymptotic intervals from around 55% to 80%). This compares to an estimate of 69% at the start of 2016 obtained from the last assessment (Tuck et al., 2015). The stock declines slowly from the beginning of the fishery in 1986, before a sharp decline beginning in the early 2000s corresponding to an increase in catch. The stock is estimated to decline to below 40% SSB_0 in 2012, before increasing to over 60% SSB_0 since 2015 and gradually increasing since then. These changes in estimated spawning biomass occur during a period of rapid rises in catches, increasing by a factor of 6 in a 5 year period to 2001 (from less than 50 t to over 300 t), and then a tenfold decline in catches to 2015 (less than 30 t). Catches have increased in 2016 and 2017, notably while catch rates were also increasing.

Recruitment has been variable, but the most recent 5 estimated recruitment events have all been above average, with the estimate of the 2011 recruitment revised upwards from the value obtained in the 2015 assessment.

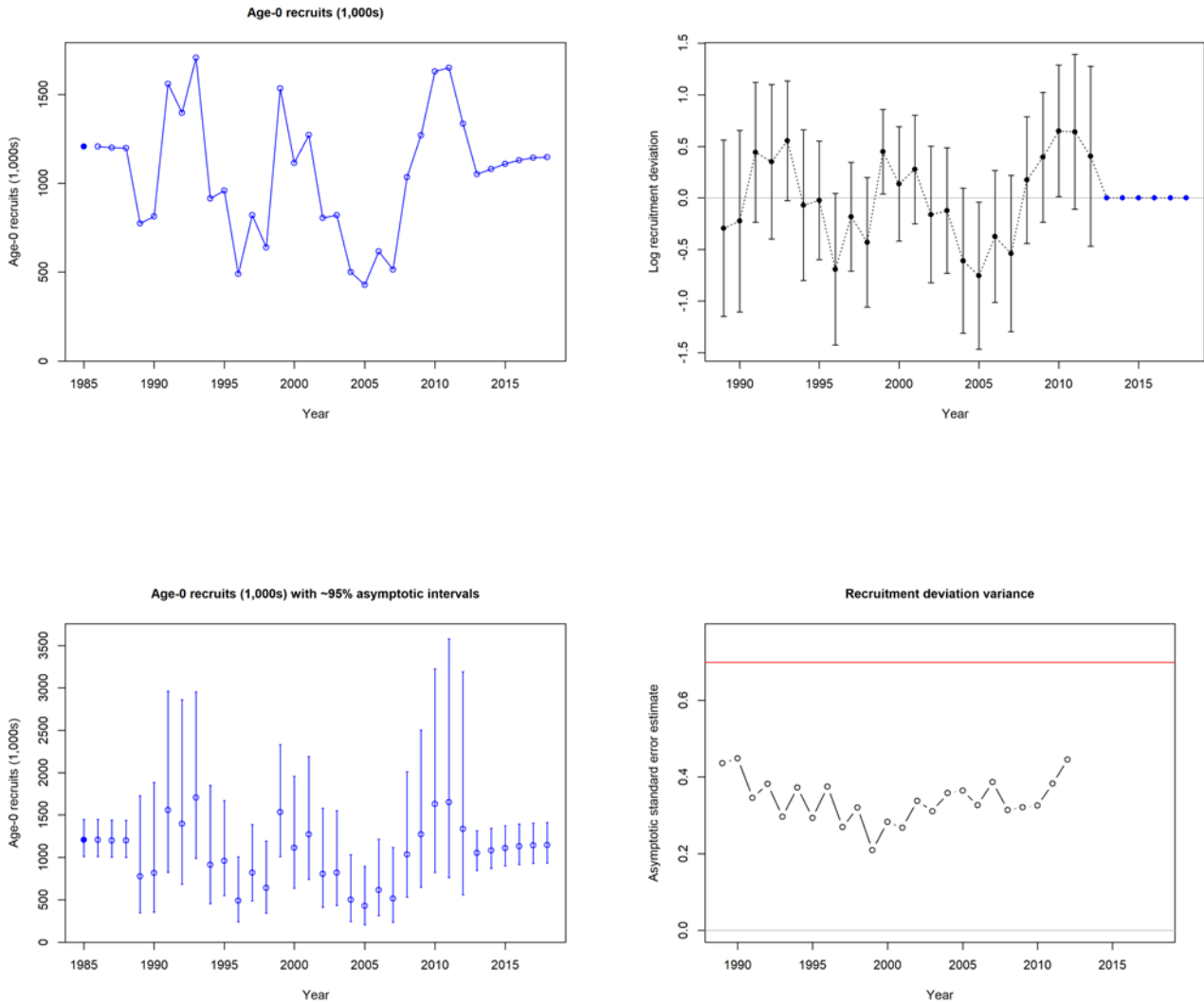


Figure 8.13. Recruitment estimation for the base case analysis. Top left : Time-trajectories of estimated recruitment numbers; top right : time trajectory of estimated recruitment deviations; bottom left : time-trajectories of estimated recruitment numbers with approximate 95% asymptotic intervals; bottom right: the standard errors of recruitment deviation estimates.

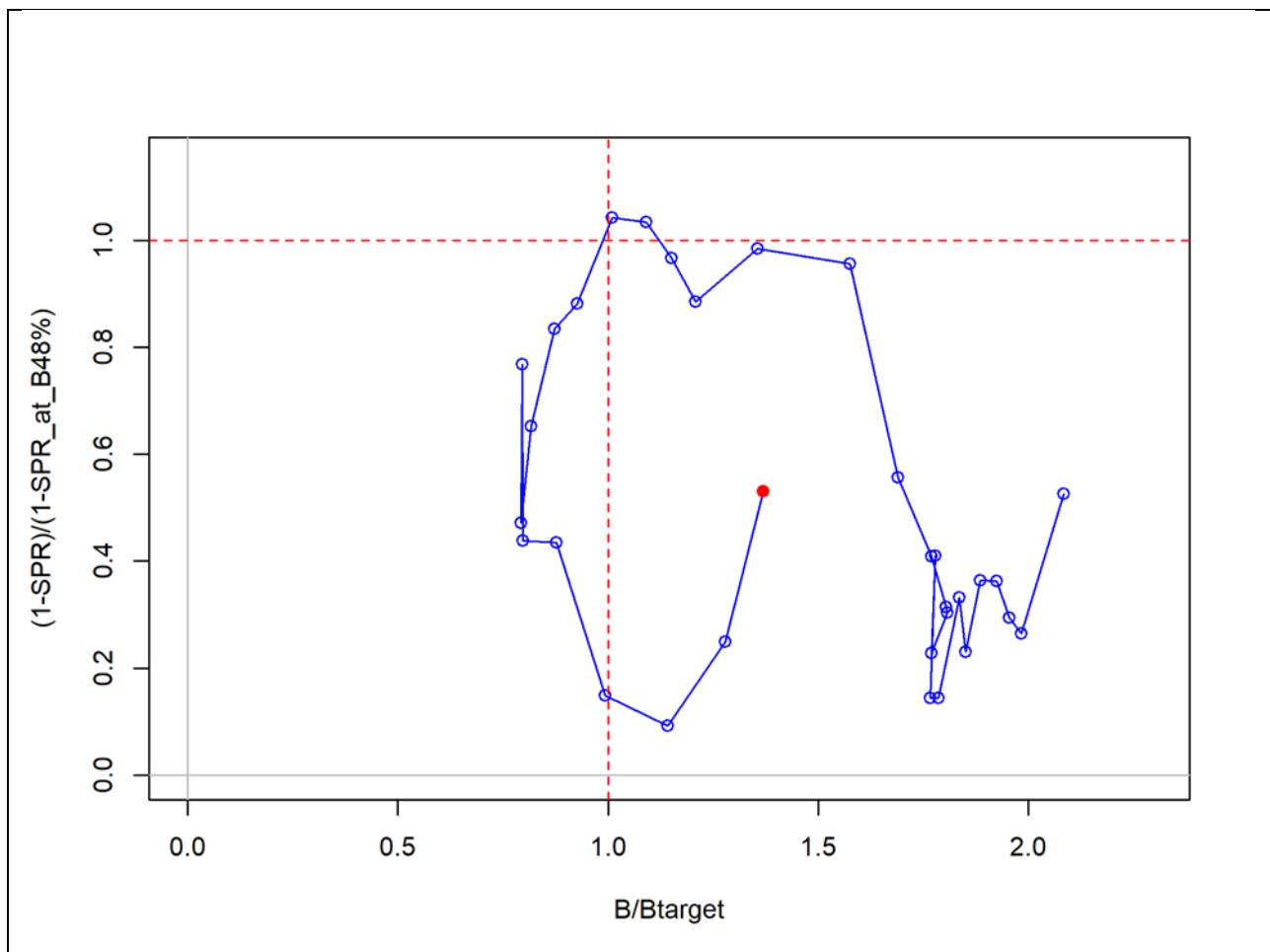


Figure 8.14. Kobe plot base case, showing the trajectory of spawning biomass (relative to B_0) plotted against 1-SPR, which is a proxy for fishing mortality, essentially integrating fishing mortality across fleets in the fishery.

Figure 8.14 shows a Kobe plot for the base case. This plot shows a time series of spawning biomass plotted against spawning potential ratio, which provides a measure of overall fishing mortality, and shows the stepwise movement in this space from the start of the fishery, in the right, when there was low fishing mortality and high biomass to 2018 (the red dot) where the biomass is above the target (to the right of the vertical red dashed line) and the fishing mortality is below the target fishing level (below the horizontal red dashed line). This trajectory shows an increase in overall fishing mortality as the fishery developed from 1986, with movement from the bottom right corner towards the top left corner, when the biomass is below the target and the fishing mortality is above the target rate. The fishing mortality was gradually reduced from around 2005 and had been below the “overfishing limit” for the last 11 years, with the spawning biomass stabilising and then increasing over this same period.

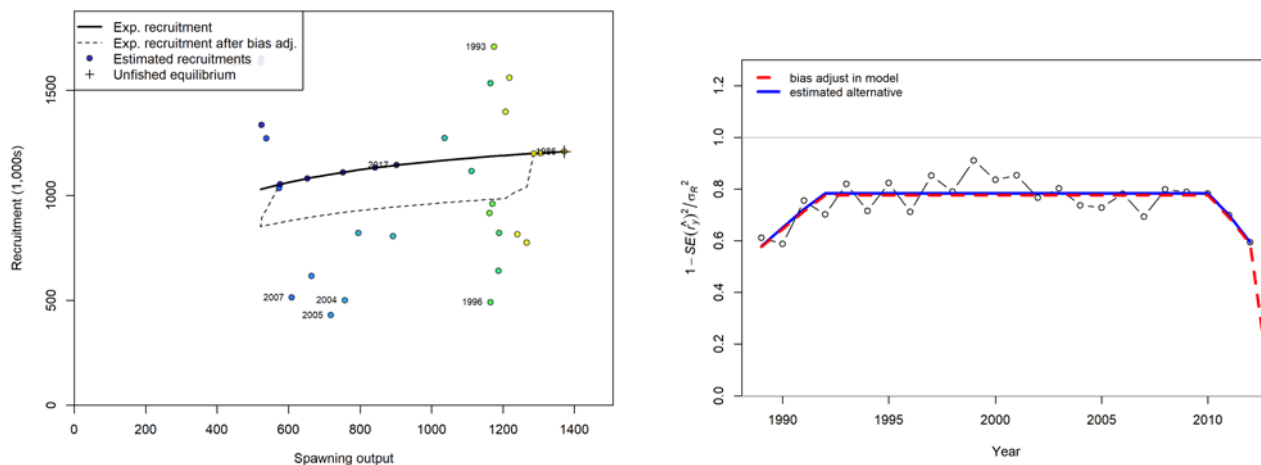


Figure 8.15. Recruitment estimation for the base case analysis. Left: the stock-recruit curve and estimated recruitments; right: bias adjustment.

The time-trajectories of recruitment and recruitment deviation are shown in Figure 8.13. Estimates of recruitments since 1989 are variable. They feature three periods with above average recruitment for at least three consecutive years, around 1993, 1999 and 2010 with other periods with several years of consecutive below average recruitment.

The base-case assessment estimates that current spawning stock biomass is 68% of unexploited stock biomass (SSB_0). The 2019 recommended biological catch (RBC) under the 20:35:48 harvest control rule is 235 t (Table 8.9) and the long term yield (assuming average recruitment in the future) is 158 t (Table 8.11). Averaging the RBC over the three year period 2019-2021, the average RBC is 223 t and over the five year period 2019-2023, the average RBC is 212 t (Table 8.11). The RBCs for each individual year from 2019-2023 are listed in Table 8.9 for the base case.

Table 8.9. Yearly projected RBCs (tonnes) across all fleets under the 20:35:48 harvest control rules all assuming average recruitment from 2014 for the agreed base case with January spawning and improved fits to growth (sensitivity 17).

| Year | RBC |
|------|-----|
| 2019 | 235 |
| 2020 | 223 |
| 2021 | 211 |
| 2022 | 201 |
| 2023 | 192 |

8.4.1.5 Discard estimates

Model estimates for discards for the period 2019-23 with the 20:35:48 harvest control rule are listed in Table 8.10 for the for the base case, with a range of 8 to 9 t.

Table 8.10. Yearly projected discards (tonnes) across all fleets under the 20:35:48 harvest control rules with catches set to the calculated RBC for each year from 2019 to 2023 for the base case.

| Year | Discards |
|------|----------|
| 2019 | 8.6 |
| 2020 | 8.3 |
| 2021 | 8.1 |
| 2022 | 8.0 |
| 2023 | 7.9 |

8.4.2 Sensitivity tests and alternative models

Results of the sensitivity tests are shown in Table 8.11. As with the 2015 assessment, biomass depletion is not overly sensitive to changes in parameters, except for natural mortality. Estimating the growth parameters improves the fit to age data, but at the expense of producing a growth curve that does not seem biologically reasonable.

This assessment is also not very sensitive to the weighting placed on the length compositions. However it is more sensitive to changing weightings on age and CPUE data, with the increased weight on the CPUE leading to lower spawning biomass values (depletion 60%) and increased weight on the age data suggesting higher spawning biomass values (depletion 75%), suggesting that these data sources are in conflict. Despite these changes in biomass depletion, the changes in likelihood values with changes to the weighting of different data sources, are relatively small (Table 8.12). This likelihood table also suggests that there is often conflict between the discard likelihood and other components, with the likelihood change to the discard component being relatively large (in absolute terms) but in the opposite direction to changes in weighting in either the length, age or survey data.

The base case includes FIS abundance indices. Two sensitivities to inclusion of FIS data include removing all FIS data, and including FIS length frequencies and FIS abundance indices, and then estimating selectivity for the FIS. The changes to the biomass depletion are minimal in each case. This may be due to the relatively short FIS abundance time series, with only 5 data points, compared to 32 data points for the standardised CPUE index and 21 years of length frequency data and 17 years of conditional age-at-length data.

8.4.3 Future work and potential issues with this assessment and data

8.4.3.1 Quality and quantity of input data

Any results from this assessment should be treated with considerable caution given the limited data quality and data quantity available for this assessment and the quality of the trawl CPUE data (Sporcic and Haddon 2018a). Given several sudden (step) changes in this time series, it may not be a very reliable index of abundance and instead may be reflecting other changes in the fishery that are not incorporated in this model. Sporcic and Haddon (2018a) indicate that the vessel factor changed its influence from 2001 onwards, suggesting a change in the fishery at that time.

Note that the base case model fit to the index of abundance is poor (Figure 8.8), as is the fit to the FIS abundance indices (Figure 8.9), with additional CVs on these abundance series estimated within the model at 0.19 and 0.63 respectively. It is possible that the data are neither sufficiently representative

nor sufficiently rich to adequately assess this stock, or that one or both are not good indices of abundance. Alternatively, there may be other unknown issues with the fishery dynamics and the stock dynamics that have not been adequately represented in this model.

8.4.3.2 *Non-representative length data*

Some length frequencies still appear to have a small amount of suspicious data, which may require further checking and quality control. The onboard length frequency in 2011 has a small outlying spike of fish of length 15cm or less, which may have been measured in cm but recorded in mm. At the other end of the spectrum, there is a spike of large fish in the 2014 length frequency, which was not present in the data in the 2015 assessment. This may represent additional large fish erroneously added to the database since 2015. Neither anomaly is having a large impact on the assessment but improving the data quality where there are potential recording errors would be preferable and would improve the overall fits to the data.

8.4.3.3 *Likelihood profiles*

Likelihood profiles were conducted on natural mortality, steepness and R_0 for the preliminary base case (Day and Castillo-Jordán, 2018a), and have not been repeated for the final base case. These likelihood profiles suggested that the fixed value for natural mortality was supported by the data, there was little information about steepness and the initial biomass is quite uncertain. The base case virgin spawning biomass is estimated at 2,743 t with the likelihood profile on R_0 suggesting 95% confidence intervals at around 2,000 t and 3,300 t.

8.4.3.4 *Retrospectives*

Preliminary retrospective analyses were also conducted on the preliminary base case (Day and Castillo-Jordán, 2018a). This analysis showed some patterns suggesting revisions to both the timing and the value of the lowest point in depletion, as additional recent data was removed, and revisions to the timing for when the spawning biomass begins to recover. Further analysis of these patterns would be useful in future.

Table 8.11. Summary of results for the base-case and sensitivity tests. Recommended biological catches (RBCs) are only shown for agreed base case model models (Case 17).

| Case | | SSB ₀ | SSB ₂₀₁₉ | SSB ₂₀₁₉ /SSB ₀ | RBC ₂₀₁₉ | RBC ₂₀₁₉₋₂₁ | RBC ₂₀₁₉₋₂₃ | RBC _{longterm} |
|------|--|------------------|---------------------|---------------------------------------|---------------------|------------------------|------------------------|-------------------------|
| 0 | base case (M 0.15, h 0.7, 50% mat 24.5cm) | 2,743 | 1,868 | 0.68 | 235 | 223 | 212 | 158 |
| 1 | M 0.1 | 2,128 | 707 | 0.33 | | | | |
| 2 | M 0.2 | 5,688 | 5,814 | 1.02 | | | | |
| 3 | h 0.6 | 2,761 | 1,740 | 0.63 | | | | |
| 4 | h 0.8 | 2,734 | 1,969 | 0.72 | | | | |
| 5 | 50% maturity at 22cm | 2,922 | 2,053 | 0.70 | | | | |
| 6 | $\sigma_R = 0.65$ | 2,728 | 1,909 | 0.70 | | | | |
| 7 | $\sigma_R = 0.75$ | 2,762 | 1,821 | 0.66 | | | | |
| 8 | estimate growth | 3,012 | 1,957 | 0.65 | | | | |
| 9 | wt x 2 length comp | 2,744 | 1,840 | 0.67 | | | | |
| 10 | wt x 0.5 length comp | 2,730 | 1,854 | 0.68 | | | | |
| 11 | wt x 2 age comp | 2,763 | 2,074 | 0.75 | | | | |
| 12 | wt x 0.5 age comp | 2,673 | 1,672 | 0.63 | | | | |
| 13 | wt x 2 CPUE | 2,501 | 1,505 | 0.60 | | | | |
| 14 | wt x 0.5 CPUE | 2,879 | 2,101 | 0.73 | | | | |
| 15 | no FIS | 2,754 | 1,895 | 0.69 | | | | |
| 16 | include FIS length frequencies | 2,741 | 1,889 | 0.69 | | | | |

Table 8.12. Summary of likelihood components for the base-case and sensitivity tests. Likelihood components are unweighted, and cases 1-17 are shown as differences from the base case. A negative value indicates a better fit, a positive value a worse fit.

| Case | | Likelihood | | | | | |
|------|--|------------|--------|---------|-------------|----------|-------------|
| | | TOTAL | Survey | Discard | Length comp | Age comp | Recruitment |
| 0 | base case (M 0.15, h 0.7, 50% mat 24.5cm) | 475.19 | -12.66 | 17.34 | 37.28 | 434.97 | -1.98 |
| 1 | M 0.1 | 14.91 | -0.11 | -3.13 | -0.09 | 16.31 | 2.07 |
| 2 | M 0.2 | 2.41 | 5.63 | 0.64 | 0.16 | -4.88 | 0.20 |
| 3 | h 0.6 | -0.29 | -0.55 | -0.10 | 0.02 | 0.26 | 0.07 |
| 4 | h 0.8 | 0.28 | 0.45 | 0.08 | -0.02 | -0.19 | -0.05 |
| 5 | 50% maturity at 22cm | 0.03 | 0.05 | -0.01 | 0.00 | -0.02 | 0.00 |
| 6 | $\sigma_R = 0.65$ | -0.23 | 0.36 | 0.10 | -0.05 | 0.30 | -0.95 |
| 7 | $\sigma_R = 0.75$ | 0.28 | -0.36 | -0.12 | 0.05 | -0.22 | 0.93 |
| 8 | estimate growth | -23.30 | 0.05 | -1.16 | 1.18 | -22.73 | -0.64 |
| 9 | wt x 2 length comp | 0.34 | -0.34 | 1.04 | -0.79 | 0.59 | -0.16 |
| 10 | wt x 0.5 length comp | 0.37 | 0.14 | -0.86 | 1.20 | -0.11 | 0.01 |
| 11 | wt x 2 age comp | 1.52 | 1.28 | 1.50 | 0.46 | -3.93 | 2.19 |
| 12 | wt x 0.5 age comp | 2.43 | -1.52 | -2.37 | -0.14 | 7.94 | -1.43 |
| 13 | wt x 2 CPUE | 1.81 | -3.75 | 2.53 | -0.24 | 2.90 | 0.41 |
| 14 | wt x 0.5 CPUE | 0.52 | 2.08 | -1.22 | 0.20 | -0.72 | 0.17 |
| 15 | no FIS | -2.32 | -2.07 | -0.18 | 0.01 | -0.15 | 0.07 |
| 16 | include FIS length frequencies | 12.99 | -0.15 | 0.25 | 4.99 | 7.63 | 0.27 |

8.5 Acknowledgements

Age data were provided by Kyne Krusic-Golub (Fish Ageing Services), ISMP and AFMA logbook and CDR data were provided by John Garvey (AFMA). Franzis Althaus, Mike Fuller, Roy Deng, Claudio Castillo-Jordán, and Paul Burch (CSIRO) pre-processed the data. André Punt, Geoff Tuck, Sandra Curin-Osorio, Paul Burch, Malcolm Haddon, Robin Thomson and Miriana Sporcic are thanked for helpful discussions on this work. Ian Taylor, Richard Methot and Chantel Wetzel (NOAA Fisheries) are thanked for all the Stock Synthesis support and advice. Malcolm Haddon provided useful code for auto-tuning, Athol Whitten provided useful R code for organising plots.

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8.7 Appendix A

A.1 Fits to length composition, implied fits to age composition, and diagnostics for fits to conditional age-at-length data.

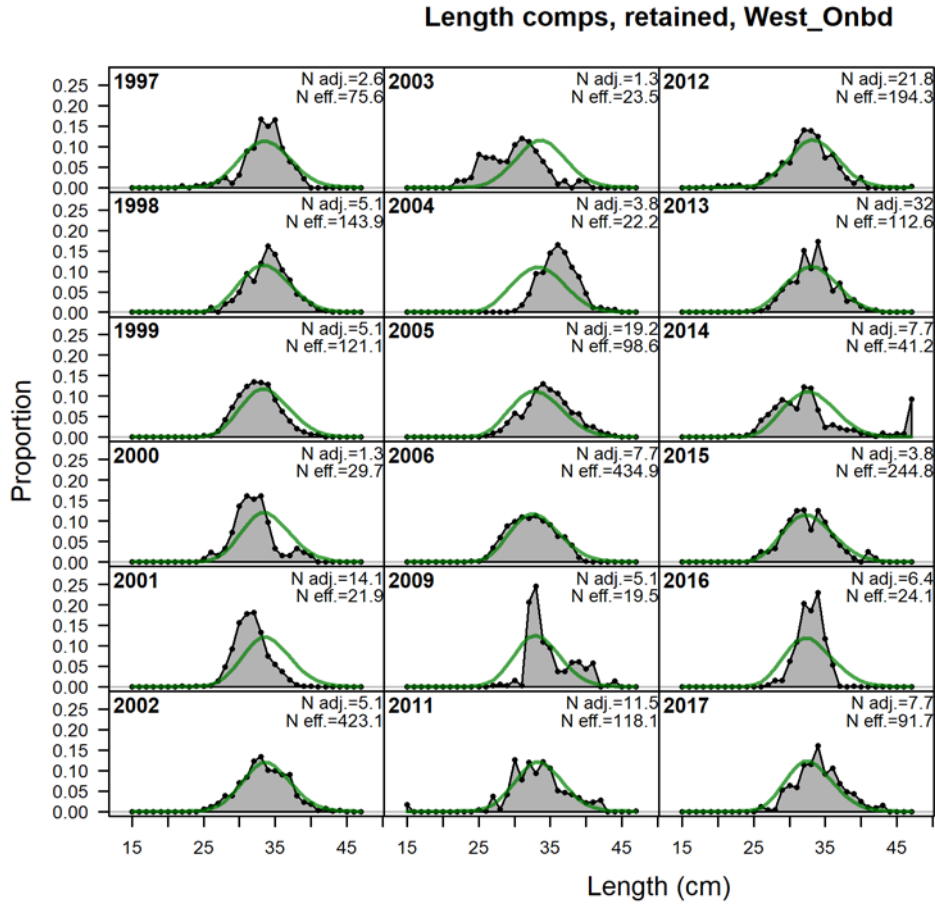


Figure A 8.1. Jackass morwong length composition fits: western trawl onboard retained.

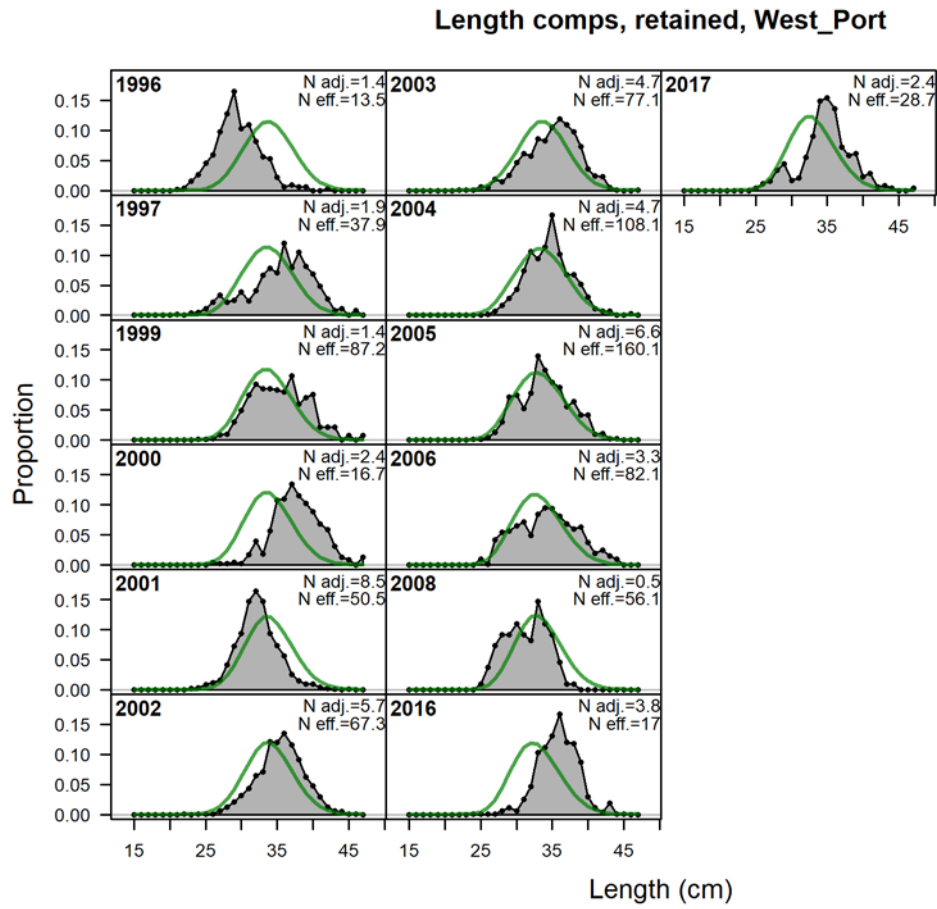


Figure A 8.2. Jackass morwong length composition fits: western trawl port retained.

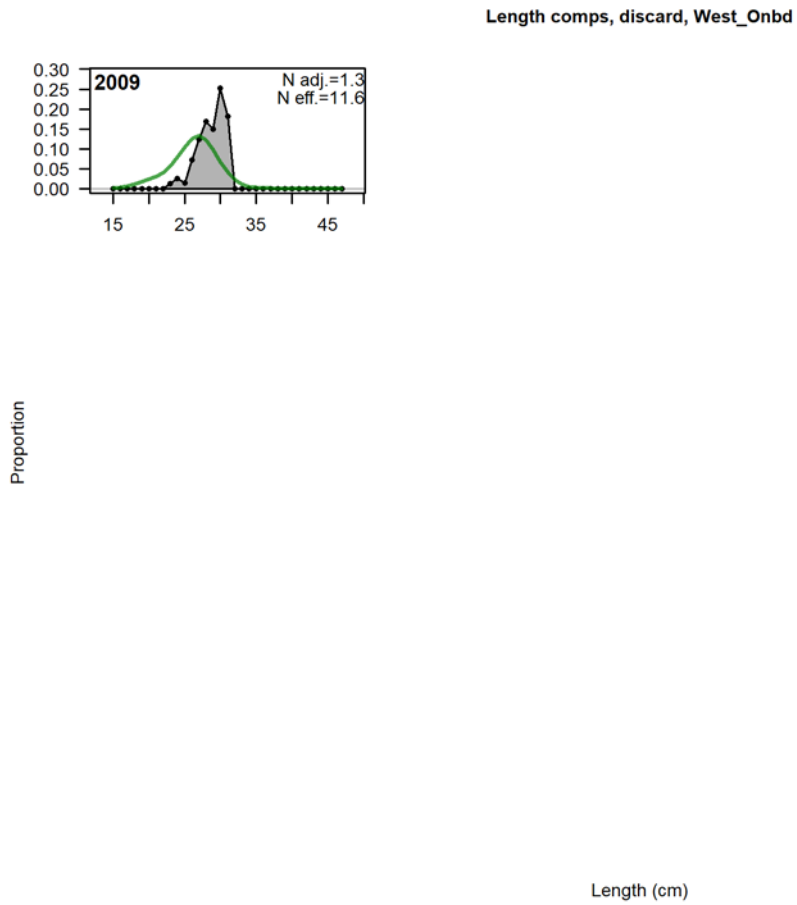


Figure A 8.3. Jackass morwong length composition fits: western trawl discarded.

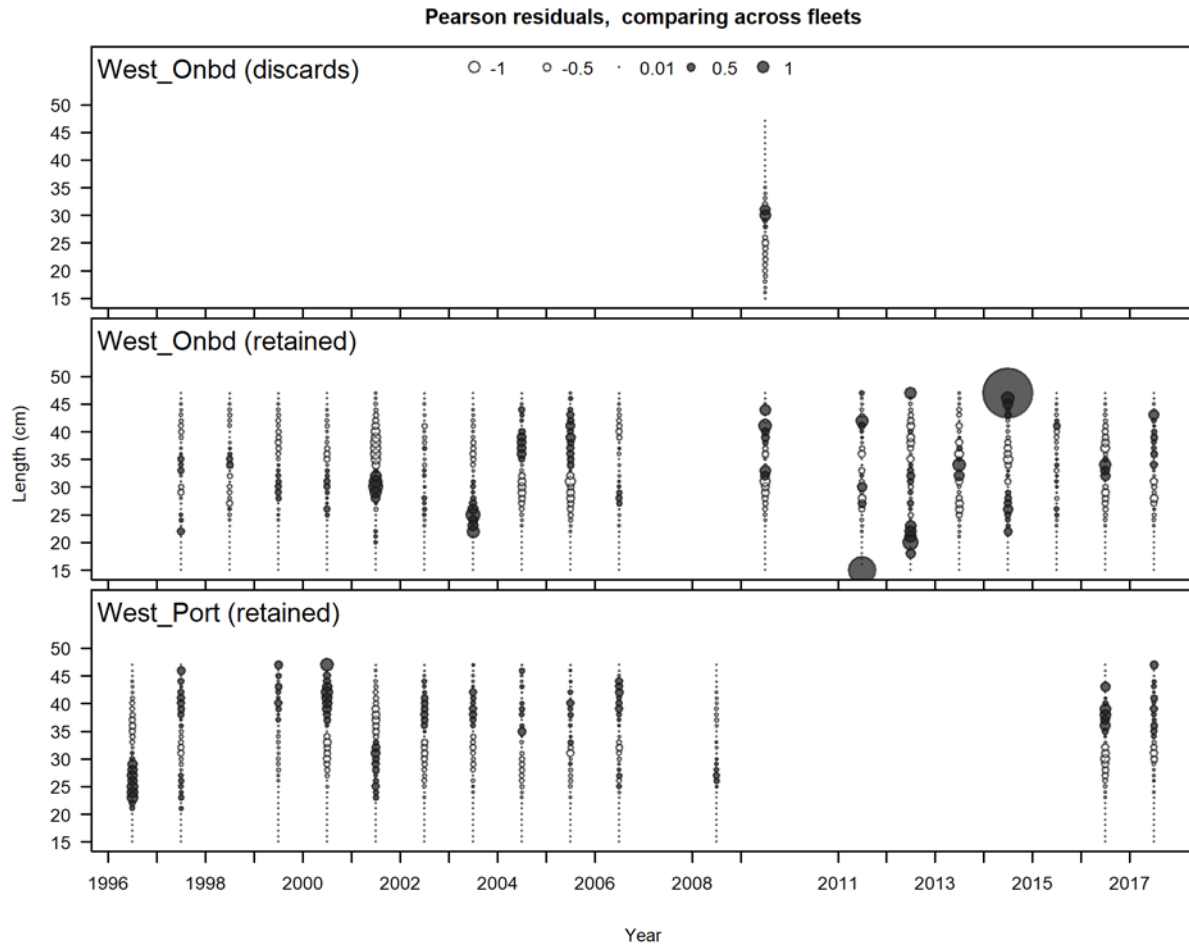


Figure A 8.4. Residuals from the annual length composition data for jackass morwong displayed by year and fleet for western trawl fleets (retained and discarded).

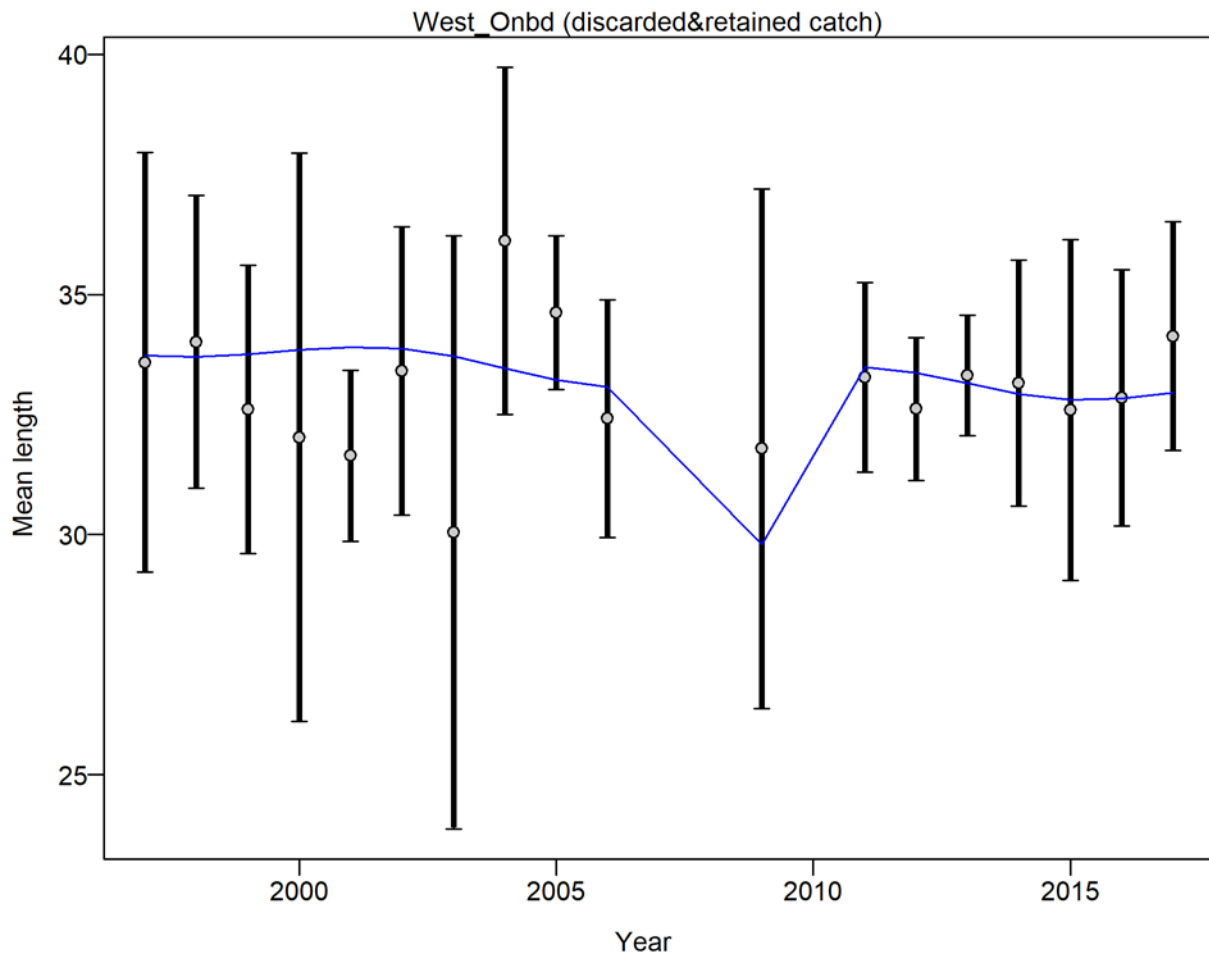


Figure A 8.5. Mean length for jackass morwong from western trawl onboard with 95% confidence intervals based on current samples sizes. Francis data weighting method TA1.8: Thin capped lines matching thick lines indicate this is well balanced.

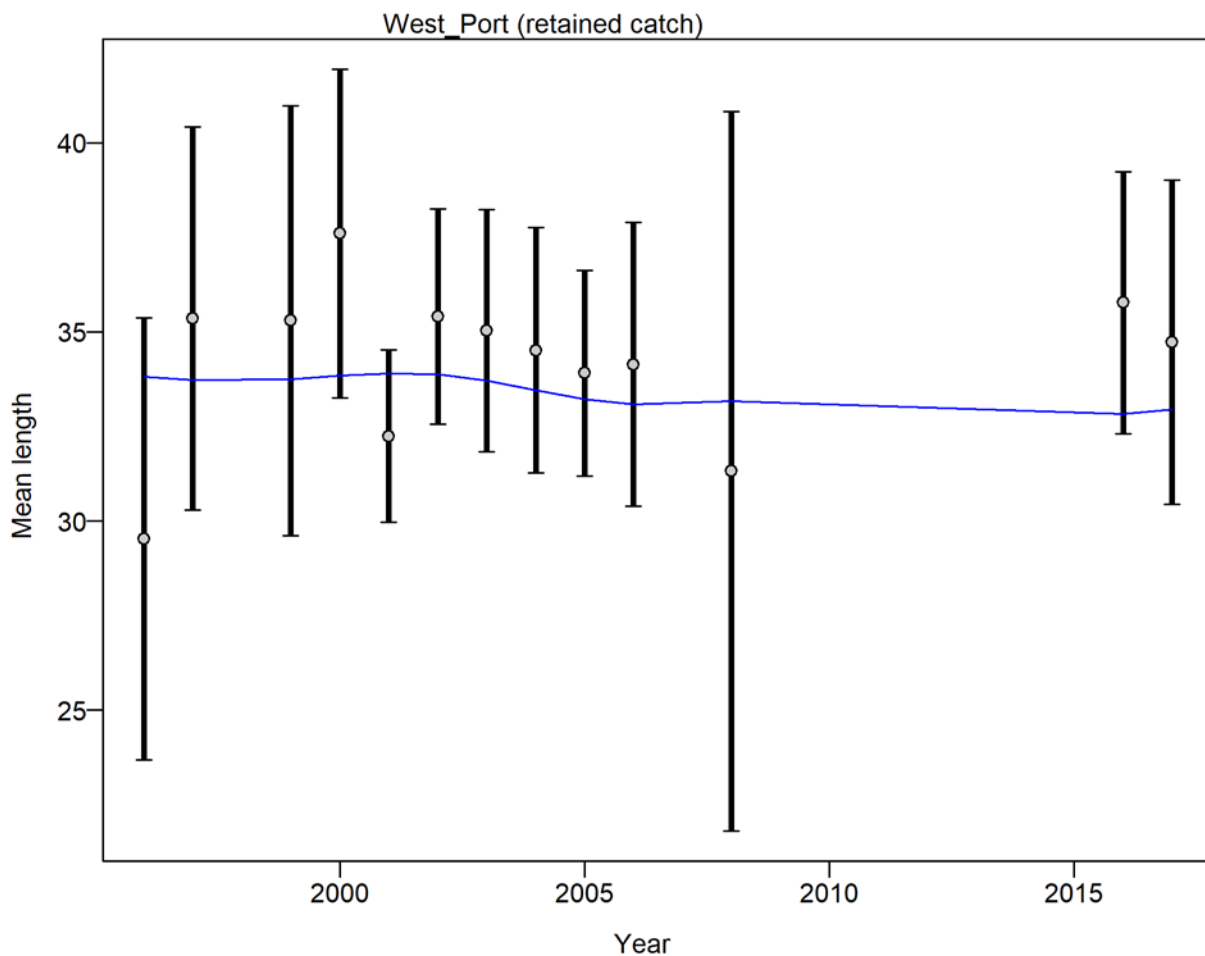


Figure A 8.6. Mean length for jackass morwong from western trawl port with 95% confidence intervals based on current samples sizes. Francis data weighting method TA1.8: Thin capped lines matching thick lines indicate this is well balanced.

Figure A 8.7. Implied fits to age compositions for jackass morwong western trawl onboard (retained).

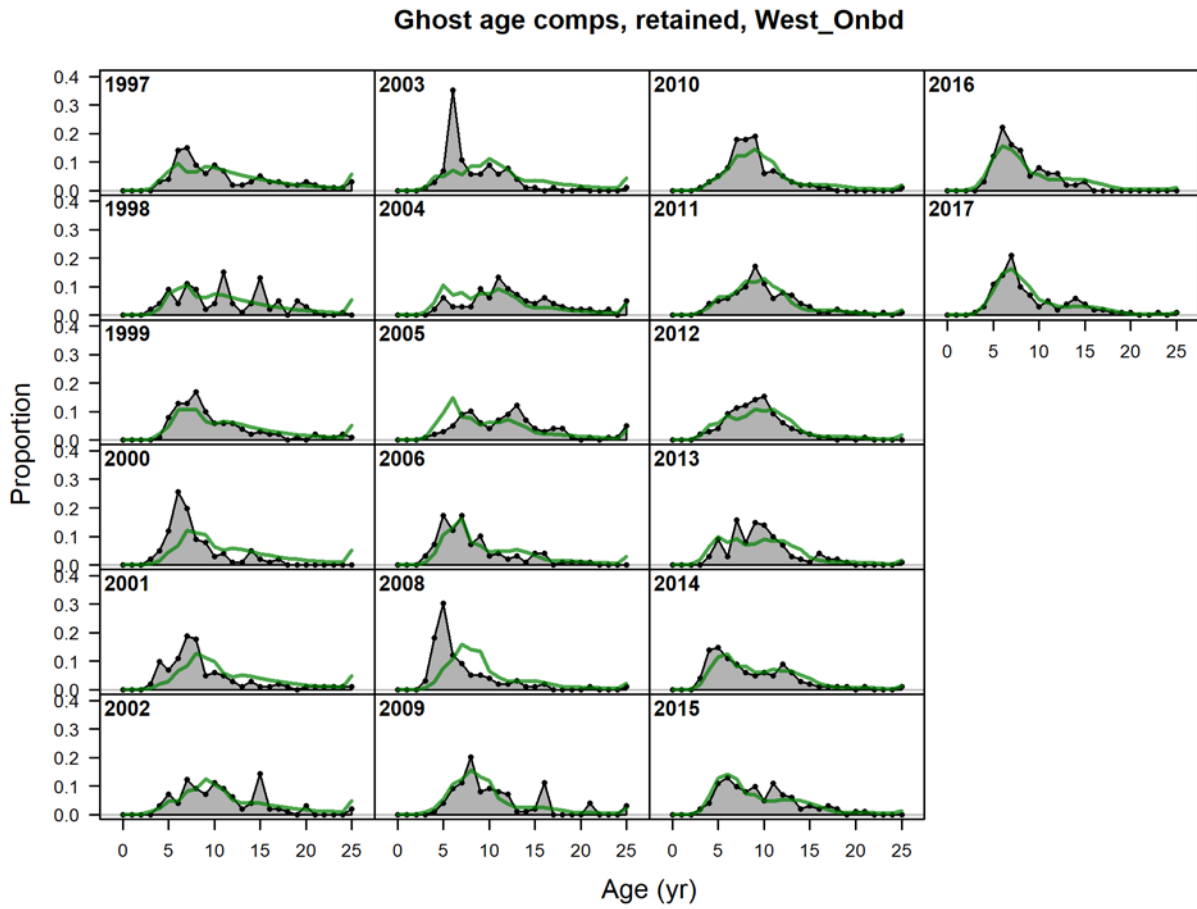


Figure A 8.8. Implied fits to age compositions for jackass morwong western trawl onboard (retained).

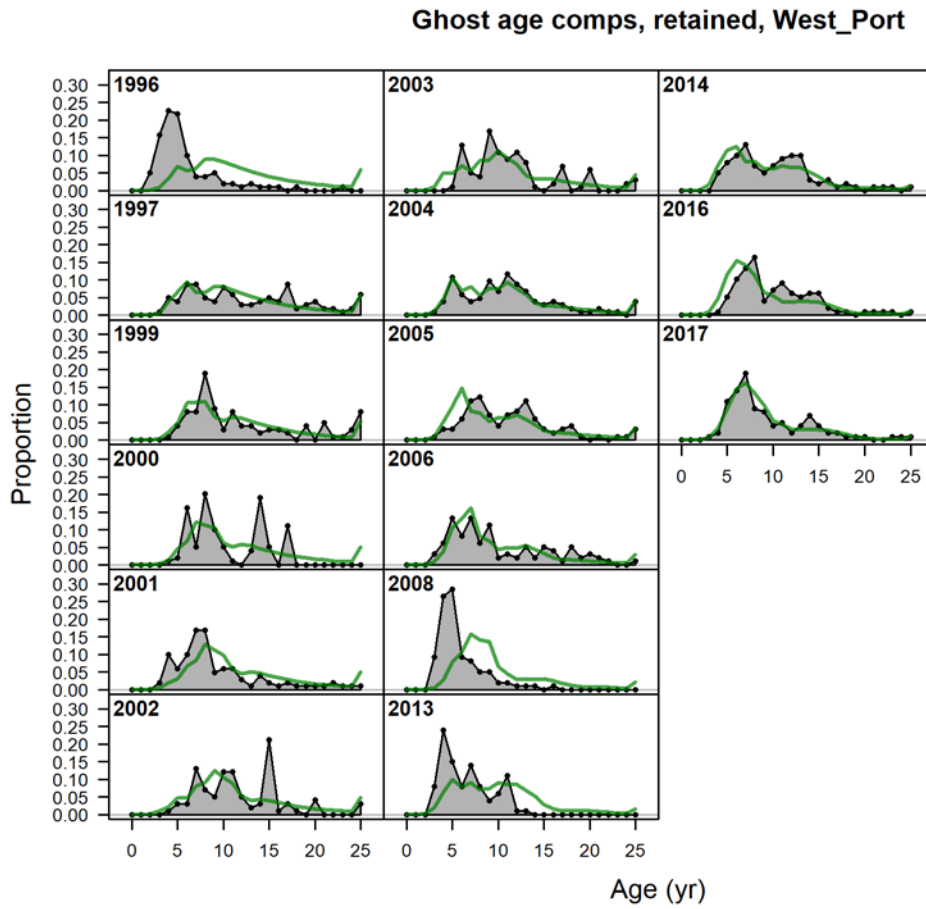


Figure A 8.9. Implied fits to age compositions for jackass morwong western trawl port (retained).

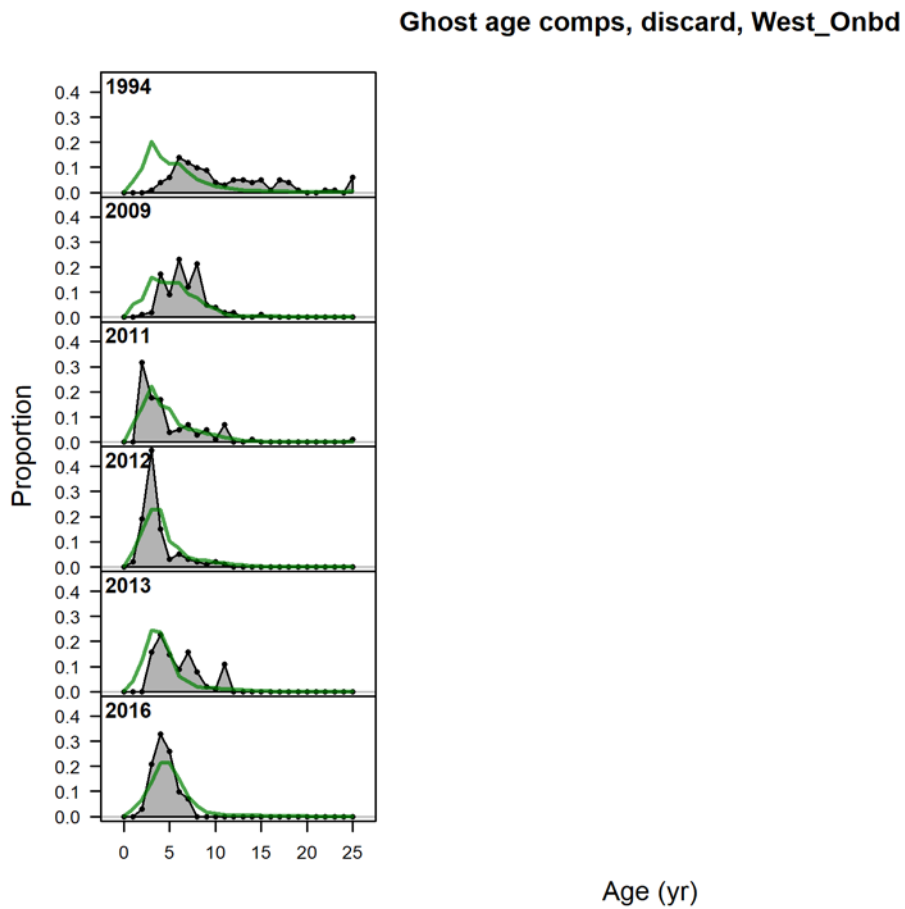


Figure A 8.10. Implied fits to age compositions for jackass morwong western trawl (discarded).

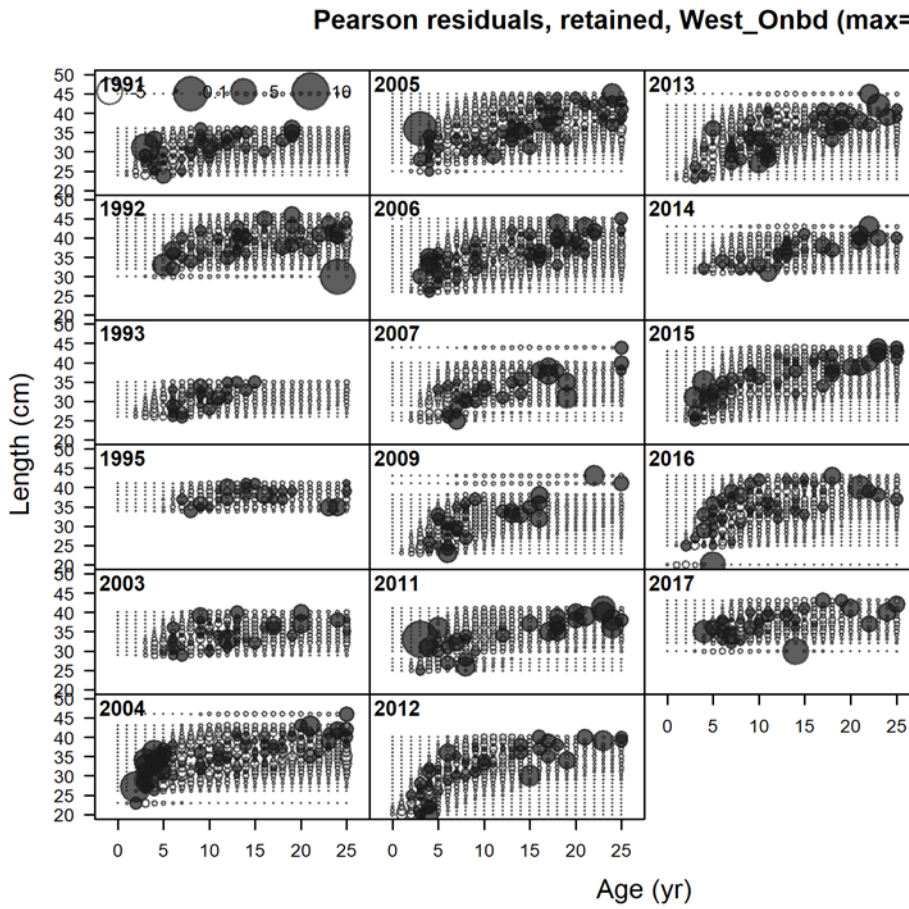


Figure A 8.11. Residuals from the fits to conditional age-at-length for jackass morwong western trawl onboard. This plot gives some indication of the variability in the age samples from year to year.

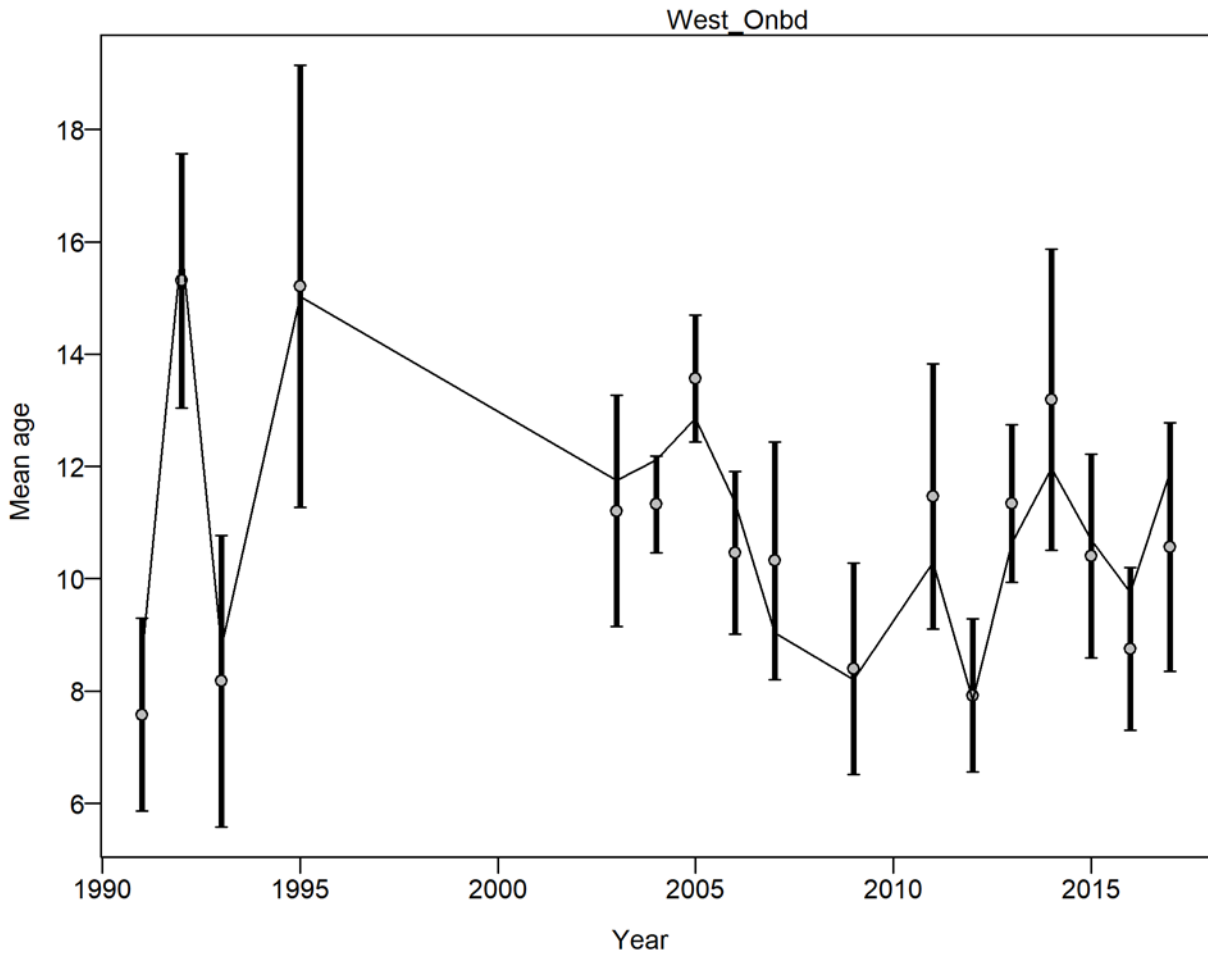


Figure A 8.12. Mean age (aggregated across length bins) for jackass morwong from western trawl with 95% confidence intervals based on current samples sizes. Punt data weighting method TA1.8: Thin capped lines matching thick lines indicate this is well balanced. Yearly variation in the data is shown in changes in mean age, which can be large over a short period (e.g. 1991-1996).

9. Blue grenadier (*Macruronus novaezelandiae*) stock assessment based on data up to 2017 – development of a preliminary base case

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9.1 Executive Summary

This document presents the preliminary base case for an updated quantitative Tier 1 assessment of blue grenadier (*Macruronus novaezelandiae*) for presentation at the first SERAG meeting in 2018. The last full assessment was conducted during 2013 (Tuck, 2013). Relative to the 2013 assessment, this preliminary base case reflects updates by the inclusion of data to the end of 2017, which entails an additional five years of catch, discard, CPUE, length-composition and conditional age-at-age data and ageing error. This document describes the process used to develop a preliminary base case for blue grenadier through the sequential updating of recent data in the stock assessment, using the stock assessment package Stock Synthesis (SS-V3.30.12.00-safe).

The base case specifications agreed by the SlopeRAG in 2013 were generally maintained into the preliminary base case presented here. The main differences are: separating length-composition into onboard- and port- collected components, assigning stage-1 weights to length-compositions by shots (onboard) and trips (port); and using the latest methods for assigning final weights to the various data sources and the extent of variation in recruitment.

The estimated time series of recruitment under the base-case parameter set shows the typical episodic nature of blue grenadier recruitment, with strong year-classes in 1979, the mid-1980s, 1994, and 2003, with very little recruitment between these years. However, the recent recruitments are more stable than has been observed before. The trajectories of spawning biomass show increases and decreases in spawning biomass as strong cohorts move into and out of the spawning population.

Results show reasonably good fits to the length-composition data, implied age compositions, egg survey and acoustic survey. The fit to the discard mass has improved compared to the 2013 assessment result. As has been noted in previous blue grenadier assessments, the fit to the standardized non-spawning catch-rate index is generally poor; the model is unable to fit to the high early catch rates and over-estimates catch rates during the early 2000s.

The estimated virgin female spawning biomass (B_0) is 57,638t tonnes (SD 7,943t) and the projected 2019 spawning stock biomass will be 138% (SD 28.5%) of virgin female spawning biomass.

Further development and sensitivity testing should include the addition of the FIS data, time blocking of the discard mass data, an exploration of the observed differences between port and onboard length-compositions and testing the sensitivity of the model to not estimating recruitment for the most recent years.

9.2 Introduction

An integrated analysis model, implemented in the generalized stock assessment software package, Stock Synthesis (SS) (Methot, 2011; Methot and Wetzel, 2013), was applied to the stock of blue grenadier in the Southern and Eastern Scalefish and Shark Fishery (SESSF), with data updated by the inclusion of data up to the 2017 calendar year (length-composition and conditional age-at-length data; age reading-error matrices, standardized catch rate series; landings and discard catch weight) and information from acoustic surveys of spawning biomass (series from 2003-2010, pertaining to total spawning biomass), with an assumption of 2-times turnover on the spawning ground (Russell and Smith, 2006). The base-case egg survey estimates of female (only) spawning biomass for 1994 and 1995 are included. The model fits directly to length-composition data (by sex where possible) and conditional age-at-length data by fleet. Retained length-composition data from port and onboard samples are separated (a change from the last assessment following current protocols).

The assessment model presented in 2011 (Tuck, Whitten and Punt 2001; Tuck 2011) was the first for blue grenadier to be implemented using SS. The 2013 assessment updated this assessment using SS-V3.22a (Tuck, 2013). Considerable changes to both the software and the tuning methods have occurred since the last assessment five years ago. As such, changes to key model outputs, such as the estimates of depletion and of the trajectory of spawning biomass, should be expected. The first bridging exercise (Bridge 1) will highlight changes that have occurred since 2013 simply through changes to software and assessment practices. The subsequent bridging exercise (Bridge 2) then sequentially updates the assessment model with new data through to 2017.

The use of SS allows for multiple fishing fleets and can fit simultaneously to several data sources and types of information. The population dynamics model, and the statistical approach used in the fitting of the model to the various types of data, is outlined fully in the SS user manual (Methot, 2005; 2011; Methot et al. 2018) and is not reproduced here. This document updates the assessment presented in 2013.

9.3 The fishery

Blue grenadier are found from New South Wales around southern Australia to Western Australia, including the coast of Tasmania. Blue grenadier is a moderately long-lived species with a maximum age of about 25 years. Age at maturity is approximately 4 years for males and 5 years for females (length-at-50% maturity for females is 57cm and 64cm respectively) based upon 32,000 blue grenadier sampled between February 1999 and October 2001 (Russell and Smith, 2006). There is also evidence that availability to the gear on the spawning ground differs by sex, with a higher proportion of small males being caught than females. This is most likely due to the arrival of males on the spawning ground at a smaller size (and younger age) than females. This was also noted by Russell and Smith (2006) who state that “young males entered the fishery one year earlier than females” and is consistent with information for hoki from New Zealand (Annala et al., 2003). Large fish arrive earlier in the spawning season than small fish. Spawning occurs predominantly off western Tasmania in winter (the peak spawning period based upon mean GSIs calculated by month was estimated to be between June and August according to Russell and Smith (2006)). There is some evidence that a high proportion of fish remain spawning in September. Variations in spawning period noted by Gunn et al (1989) may occur due to inter-annual differences in the development of coastal current patterns around Tasmania. Adults disperse following the spawning season and while fish are found throughout the south east region during the non-spawning season, their range is not well defined. Spawning fish have been caught off

the east coast of Australia, and larvae from a likely eastern spawning area have been described by Bruce et al. (2001).

Blue grenadier are caught by demersal trawling. The global agreed TAC for the 2017/18 fishing season was 8,810 tonnes. The annual TACs are shown in Table 9.2. There are two defined sub-fisheries: the spawning (Zone 40, months June, July and August) and non-spawning fisheries (all other months and zones).

9.4 Data

The assessment has been updated since the previous assessment (Tuck, 2013) by including recent length-composition and conditional age-at-length data from the spawning and non-spawning fisheries; updated standardized CPUE series (Sporcic and Haddon, 2018), the total mass landed and discarded, and updated age-reading error matrices. Acoustic estimates of spawning biomass (2003-2010) and estimates of the female spawning biomass in 1994 and 1995 from egg surveys (Bulman et al., 1999) are included as before. Data were formulated by calendar year (i.e. 1 Jan to 31 Dec) as in previous models.

9.4.1 Catch data

9.4.1.1 Landings

The landings from the logbook data were used to apportion catches to the spawning and non-spawning fisheries (Table 9.1). The logbook landings have been adjusted upwards to the CDRs to take account of differences between logbook and landings data (multiple of 1.4 for the non-spawning fishery, based on 40% conversion from headed and gutted to whole, since 1986 and up to and including 1997 (reliable CDR data were available from 1998); 1.2 for the spawning fishery from 1986 up to and including 1996 (when factory vessels entered the spawning fishery) (D. Smith, pers. comm.). As stated by Thomson and He (2001), the factor is lower for the spawning fleet than the non-spawning fleet because some fish in the spawning fishery, landed headed and gutted, were recorded as being landed whole. These factors were chosen by the Blue Grenadier Assessment Group (BGAG) (Chesson and Staples (1995), as cited by Punt (1998)). The adjusted logbook catches were then scaled up to the SEF2 data (CDR). As historical CDR data were only available from 1992, the average scaling factor from 1992 to 1996 (1.07) was used to scale the data for years between 1986 and 1991. Note that in years 2008 to 2013 logbook data were greater than landings from the CDR. In these cases, the tonnage from the CDR was used as the total catch (AFMA, pers. comm. 2011). Table 9.2 lists the annual catches used in the assessment and the annual TAC. The annual logbook catches by sub-fishery and the adjustments made to determine the catches used in the assessment are shown in Table 9.1.

9.4.1.2 Discards

Discard rates were estimated from on-board data which gives the weight of the retained and discarded component of those shots that were monitored (Thomson and Klaer, 2011). The discard values from 1995 to 2002 are based on estimates calculated from ISMP data by MAFRI and reported in He et al (1999) and Tuck, Smith and Talman (2004). As agreed by Slope RAG (2011), since 2003 discard rates are estimated using the methods described in Thomson and Klaer (2011). The mass of the discard is calculated from the annual discard rate and the retained catch from the non-spawning fishery. The MAFRI estimates of discards were made accounting for differences in sampling and discard rates according to the ISMP zones. The more recent estimates are simple ratios of total discards to (retained

+ discard) catch (N. Klaer, pers comm.). Information in support of the historical values was not able to be obtained and further exploration of the methods and data used to estimate these values should be encouraged. The discard data are provided in Table 9.2. The discard data were assumed to have standard error (on the log-scale) of 0.3.

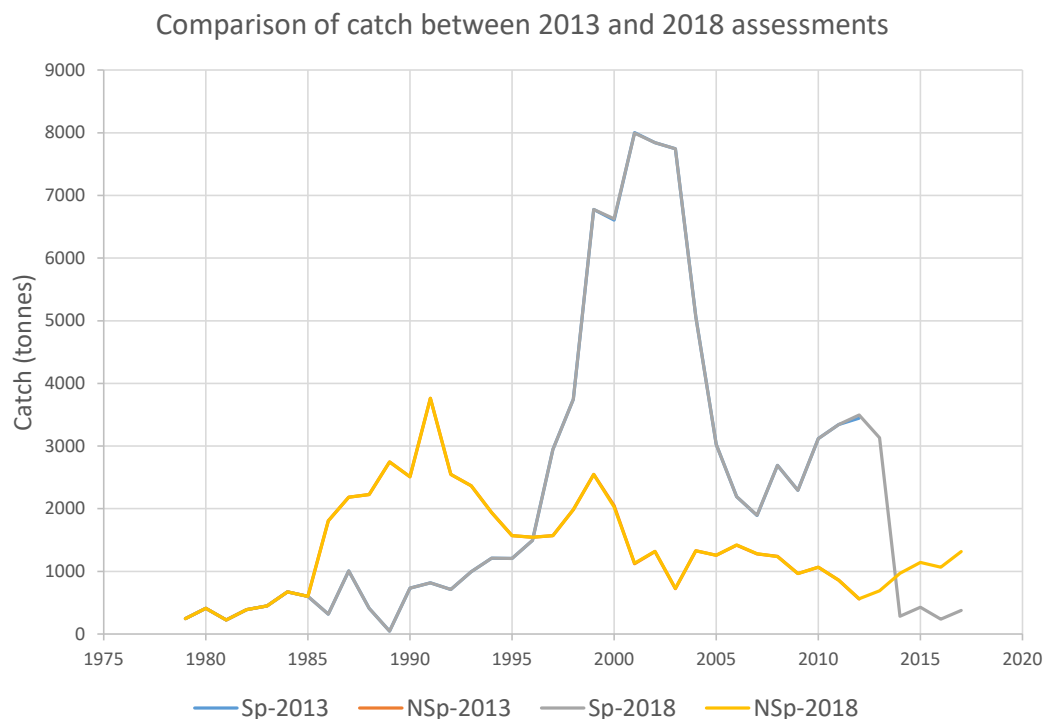


Figure 9.1. A comparison of total annual catches from the 2013 base case assessment and the updated catch used in the 2018 assessment for the spawning (Sp) and non-spawning (NSp) fisheries.

Table 9.1. Logbook and CDR landings for the spawning and non-spawning sub-fisheries by calendar year and adjustments made to account for logbooks being less than landings and incorrect reporting process code. Shaded CDR are historical landings values. ¹ average of CDR/logbook ratio from 1992 to 1996.

| Year | Logbook | | CDR | H&G Multiplier | | Adjusted Logbook | | | CDR / logbook | Catch for assessment | |
|------|----------|--------------|------|----------------|--------------|------------------|--------------|-------|-------------------|----------------------|--------------|
| | Spawning | Non-spawning | | Spawning | Non-spawning | Spawning | Non-spawning | Total | | Spawning | Non-spawning |
| 1979 | 245 | 245 | | 1 | 1 | 245 | 245 | 490 | 1.00 | 245 | 245 |
| 1980 | 410 | 410 | | 1 | 1 | 410 | 410 | 820 | 1.00 | 410 | 410 |
| 1981 | 225 | 225 | | 1 | 1 | 225 | 225 | 450 | 1.00 | 225 | 225 |
| 1982 | 390 | 390 | | 1 | 1 | 390 | 390 | 780 | 1.00 | 390 | 390 |
| 1983 | 450 | 450 | | 1 | 1 | 450 | 450 | 900 | 1.00 | 450 | 450 |
| 1984 | 675 | 675 | | 1 | 1 | 675 | 675 | 1350 | 1.00 | 675 | 675 |
| 1985 | 600 | 600 | | 1 | 1 | 600 | 600 | 1200 | 1.00 | 600 | 600 |
| 1986 | 246 | 1204 | | 1.2 | 1.4 | 295 | 1685 | 1981 | 1.07 | 317 | 1806 |
| 1987 | 782 | 1455 | | 1.2 | 1.4 | 939 | 2036 | 2975 | 1.07 | 1006 | 2183 |
| 1988 | 319 | 1485 | | 1.2 | 1.4 | 383 | 2079 | 2461 | 1.07 | 410 | 2228 |
| 1989 | 36 | 1829 | | 1.2 | 1.4 | 43 | 2560 | 2604 | 1.07 | 46 | 2745 |
| 1990 | 570 | 1671 | | 1.2 | 1.4 | 684 | 2340 | 3023 | 1.07 | 733 | 2508 |
| 1991 | 637 | 2508 | | 1.2 | 1.4 | 764 | 3511 | 4275 | 1.07 ¹ | 819 | 3764 |
| 1992 | 509 | 1565 | 3259 | 1.2 | 1.4 | 610 | 2191 | 2802 | 1.16 | 710 | 2549 |
| 1993 | 812 | 1659 | 3362 | 1.2 | 1.4 | 975 | 2323 | 3298 | 1.02 | 994 | 2368 |
| 1994 | 974 | 1338 | 3151 | 1.2 | 1.4 | 1169 | 1873 | 3042 | 1.04 | 1211 | 1940 |
| 1995 | 911 | 1017 | 2775 | 1.2 | 1.4 | 1093 | 1424 | 2517 | 1.10 | 1205 | 1570 |
| 1996 | 1200 | 1061 | 3040 | 1.2 | 1.4 | 1439 | 1485 | 2925 | 1.04 | 1496 | 1544 |
| 1997 | 2623 | 997 | 4516 | 1 | 1.4 | 2623 | 1396 | 4019 | 1.12 | 2947 | 1569 |
| 1998 | 2739 | 1452 | 5733 | 1 | 1 | 2739 | 1452 | 4191 | 1.37 | 3746 | 1986 |
| 1999 | 5460 | 2054 | 9324 | 1 | 1 | 5460 | 2054 | 7514 | 1.24 | 6775 | 2549 |
| 2000 | 5735 | 1755 | 8655 | 1 | 1 | 5735 | 1755 | 7490 | 1.16 | 6627 | 2028 |
| 2001 | 7309 | 1032 | 9124 | 1 | 1 | 7309 | 1032 | 8340 | 1.09 | 7995 | 1129 |
| 2002 | 6825 | 1148 | 9161 | 1 | 1 | 6825 | 1148 | 7973 | 1.15 | 7842 | 1319 |
| 2003 | 7239 | 679 | 8471 | 1 | 1 | 7239 | 679 | 7918 | 1.07 | 7745 | 726 |
| 2004 | 4647 | 1219 | 6392 | 1 | 1 | 4647 | 1219 | 5865 | 1.09 | 5064 | 1328 |
| 2005 | 2880 | 1199 | 4283 | 1 | 1 | 2880 | 1199 | 4079 | 1.05 | 3024 | 1259 |
| 2006 | 2058 | 1332 | 3614 | 1 | 1 | 2058 | 1332 | 3390 | 1.07 | 2193 | 1420 |
| 2007 | 1815 | 1228 | 3176 | 1 | 1 | 1815 | 1228 | 3044 | 1.04 | 1894 | 1282 |
| 2008 | 2838 | 1304 | 3931 | 1 | 1 | 2838 | 1304 | 4141 | 0.95 | 2693 | 1237 |
| 2009 | 2723 | 1145 | 3259 | 1 | 1 | 2723 | 1145 | 3868 | 0.84 | 2295 | 965 |
| 2010 | 3384 | 1158 | 4185 | 1 | 1 | 3384 | 1158 | 4541 | 0.92 | 3118 | 1067 |
| 2011 | 3554 | 914 | 4201 | 1 | 1 | 3554 | 914 | 4467 | 0.94 | 3342 | 859 |
| 2012 | 3838 | 620 | 4060 | 1 | 1 | 3838 | 620 | 4458 | 0.91 | 3495 | 565 |
| 2013 | 3443 | 759 | 3821 | 1 | 1 | 3443 | 759 | 4201 | 0.91 | 3131 | 690 |
| 2014 | 271 | 928 | 1251 | 1 | 1 | 271 | 928 | 1200 | 1.04 | 283 | 968 |
| 2015 | 393 | 1054 | 1570 | 1 | 1 | 393 | 1054 | 1447 | 1.08 | 426 | 1144 |
| 2016 | 216 | 968 | 1305 | 1 | 1 | 216 | 968 | 1184 | 1.10 | 238 | 1068 |
| 2017 | 354 | 1237 | 1693 | 1 | 1 | 354 | 1237 | 1591 | 1.06 | 376 | 1316 |

Table 9.2. Landed and discarded catches for the spawning and non-spawning sub-fisheries by calendar year. These estimates have been scaled up to the landings data. Standardised CPUE (Sporcic and Haddon, 2018) for the non-spawning sub-fisheries by calendar year are shown, along with the TAC. ¹ a voluntary industry reduction to 4,200 t was implemented in 2005. ² This was a 16 month TAC. ³ The TACs cover the fishing year 1 May to 30 April. In the table below, 2008 refers to 2008/09. ⁴ This is an estimate of retained catch based on the 2017/2018 TAC and relative split of catch between the spawning and non-spawning fisheries of 2017.

| Year | Spawning (t) | Non-spawning (t) | Discards (t) | TAC | CPUE |
|------|------------------|-------------------|--------------|-------------------|--------|
| 1979 | 245 | 245 | | | |
| 1980 | 410 | 410 | | | |
| 1981 | 225 | 225 | | | |
| 1982 | 390 | 390 | | | |
| 1983 | 450 | 450 | | | |
| 1984 | 675 | 675 | | | |
| 1985 | 600 | 600 | | | |
| 1986 | 317 | 1807 | | | 1.5611 |
| 1987 | 1006 | 2183 | | | 1.994 |
| 1988 | 410 | 2228 | | | 2.1709 |
| 1989 | 46 | 2745 | | | 2.1776 |
| 1990 | 733 | 2508 | | | 2.166 |
| 1991 | 819 | 3764 | | | 1.545 |
| 1992 | 710 | 2549 | | | 1.252 |
| 1993 | 994 | 2368 | | | 0.9511 |
| 1994 | 1211 | 1940 | | 10000 | 0.8586 |
| 1995 | 1205 | 1570 | 80 | 10000 | 0.5937 |
| 1996 | 1496 | 1544 | 975 | 10000 | 0.5361 |
| 1997 | 2947 | 1569 | 3716 | 10000 | 0.5574 |
| 1998 | 3746 | 1986 | 1329 | 10000 | 0.901 |
| 1999 | 6775 | 2549 | 123 | 10000 | 0.9466 |
| 2000 | 6627 | 2028 | 69 | 10000 | 0.6815 |
| 2001 | 7995 | 1129 | 10 | 10000 | 0.3927 |
| 2002 | 7842 | 1319 | 2 | 10000 | 0.391 |
| 2003 | 7745 | 726 | 8 | 9000 | 0.3258 |
| 2004 | 5064 | 1328 | 34 | 7000 | 0.5474 |
| 2005 | 3024 | 1259 | 294 | 5000 ¹ | 0.6594 |
| 2006 | 2193 | 1420 | 175 | 3730 | 0.8803 |
| 2007 | 1894 | 1282 | 72 | 4113 ² | 0.782 |
| 2008 | 2693 | 1237 | 18 | 4368 ³ | 0.8643 |
| 2009 | 2295 | 965 | 57 | 4700 ³ | 0.8004 |
| 2010 | 3118 | 1067 | 13 | 4700 ³ | 0.7975 |
| 2011 | 3342 | 859 | 169 | 4700 ³ | 0.6511 |
| 2012 | 3495 | 565 | 277 | 5208 ³ | 0.5187 |
| 2013 | 3131 | 690 | 469 | 5208 ³ | 0.9243 |
| 2014 | 283 | 968 | 680 | 6800 ³ | 1.1316 |
| 2015 | 426 | 1144 | 1032 | 8796 ³ | 1.2303 |
| 2016 | 238 | 1068 | 512 | 8810 ³ | 1.0448 |
| 2017 | 376 | 1316 | 718 | 8765 ³ | 1.1656 |
| 2018 | 378 ⁴ | 1323 ⁴ | | 8810 ³ | |

9.4.2 Catch rates

Sporcic and Haddon (2018) provide the updated standardised catch rate series for the non-spawning fishery of blue grenadier (Table 9.2; Figure 9.2). The catch rate generally follows the fluctuations of stock size driven by large, but sporadic, recruitments. The standard deviation of log-CPUE is assumed to be 0.25, but an extra variance component is estimated for the CPUE index.

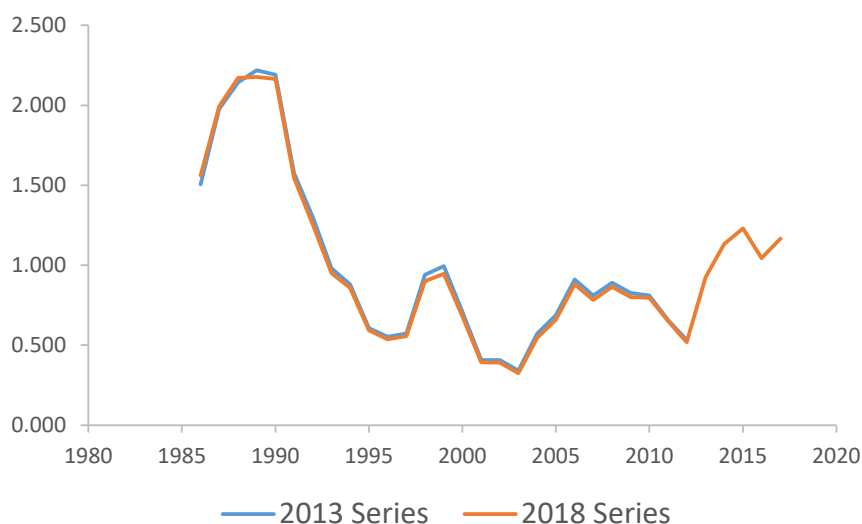


Figure 9.2. A comparison of the annual catch rates series for blue grenadier between the 2013 assessment (2013 Series) and 2018 (2018 Series).

9.4.3 Length-composition and age data

Length and age data are been included in the assessment as length-composition data and conditional age-at-length data by fleet and sex (the latter when available). Age-composition data are included in diagnostic plots but are not used directly when estimating the parameters of the population dynamics model. On-board and port length-compositions, when available, are used separately. This is a change in data protocol from the last assessment, where lengths from port and onboard measurements were combined. Prior to 2013, only port samples had been used to create the length-compositions. Plots of the observed length and age data are shown in later figures, with the corresponding model predicted values.

There had to be at least 100 measured fish for a retained and/or discard onboard and port length-composition data to be included in the assessment. For onboard samples, numbers of shots were used as the sampling unit (i.e. the stage-1 weights), with a cap of 200. For port samples, numbers of trips were used as the sampling unit, with a cap of 100. The number of fish measured is not used as the sample size because the appropriate sample size for length-composition data is probably more closely related to the number of shots (onboard) or trips (port) sampled, rather than the number of fish measured (Table 9.3).

Table 9.3. The years for which length data were available for the sub-fleets (spawning onboard = 1; spawning port = 3; non-spawning onboard = 2; non-spawning port = 4), sex (0 = no gender specified; female =1; male =2), partition (discard = 1; retained = 2). N is the number of shots (onboard) or trips (port). Red length data were excluded due to low sample sizes. ¹ the average number of fish from years 1984 and 1988. ² as no shot data were available, these estimates were based upon the average number of fish per shot for un-sexed fish for Fleet 1 (84.4). ³ the average number of fish from years 1984 and 1987-89. ⁴ as no shot data were available, these estimates were based upon the average number of fish per shot for Fleet 2 (40.7). ⁵ the average of 1980s samples, as no fish numbers or shot data were available.

| Year | Nfish | Fleet | Sex | Part | N |
|------|--------------------|-------|-----|------|-----------------|
| 1984 | 1,046 | 1 | 0 | 2 | 12 ² |
| 1985 | 1,090 ¹ | 1 | 0 | 2 | 12 ² |
| 1988 | 1,133 | 1 | 0 | 2 | 12 ² |
| 1998 | 1,948 | 1 | 0 | 2 | 29 |
| 1999 | 4,147 | 1 | 1 | 2 | 49 |
| 1999 | 5,929 | 1 | 2 | 2 | 70 |
| 2000 | 2,672 | 1 | 1 | 2 | 32 |
| 2000 | 2,956 | 1 | 2 | 2 | 35 |
| 2001 | 3,620 | 1 | 1 | 2 | 43 |
| 2001 | 4,256 | 1 | 2 | 2 | 50 |
| 2002 | 760 | 1 | 0 | 2 | 3 |
| 2003 | 2,700 | 1 | 1 | 2 | 32 |
| 2003 | 2,853 | 1 | 2 | 2 | 34 |
| 2004 | 1,307 | 1 | 1 | 2 | 15 |
| 2004 | 1,370 | 1 | 2 | 2 | 16 |
| 2005 | 198 | 1 | 1 | 2 | 2 |
| 2005 | 141 | 1 | 2 | 2 | 2 |
| 2006 | 3,184 | 1 | 1 | 2 | 38 |
| 2006 | 3,081 | 1 | 2 | 2 | 36 |
| 2007 | 2,957 | 1 | 1 | 2 | 35 |
| 2007 | 1,897 | 1 | 2 | 2 | 22 |
| 2008 | 3,073 | 1 | 1 | 2 | 36 |
| 2008 | 2,177 | 1 | 2 | 2 | 26 |
| 2009 | 3,868 | 1 | 1 | 2 | 46 |
| 2009 | 3,374 | 1 | 2 | 2 | 40 |
| 2010 | 2,488 | 1 | 1 | 2 | 29 |
| 2010 | 1,453 | 1 | 2 | 2 | 17 |
| 2011 | 4,207 | 1 | 1 | 2 | 50 |
| 2011 | 3,266 | 1 | 2 | 2 | 39 |
| 2012 | 3,939 | 1 | 1 | 2 | 47 |
| 2012 | 3,060 | 1 | 2 | 2 | 36 |
| 2013 | 6,371 | 1 | 0 | 2 | 76 |
| 2014 | 927 | 1 | 0 | 2 | 27 |
| 2015 | 1,861 | 1 | 0 | 2 | 19 |
| 2017 | 1,020 | 1 | 0 | 2 | 16 |

| Year | Nfish | Fleet | Sex | Part | N |
|------|--------------------|-------|-----|------|------------------|
| 1993 | 207 | 2 | 0 | 1 | 2 |
| 1995 | 2,216 | 2 | 0 | 1 | 21 |
| 1996 | 5,225 | 2 | 0 | 1 | 73 |
| 1997 | 6,504 | 2 | 0 | 1 | 159 |
| 1998 | 2,212 | 2 | 0 | 1 | 97 |
| 1999 | 940 | 2 | 0 | 1 | 45 |
| 2000 | 132 | 2 | 0 | 1 | 4 |
| 2004 | 1,077 | 2 | 0 | 1 | 21 |
| 2005 | 5,139 | 2 | 0 | 1 | 51 |
| 2006 | 1,225 | 2 | 0 | 1 | 81 |
| 2007 | 16 | 2 | 0 | 1 | 3 |
| 2008 | 106 | 2 | 0 | 1 | 17 |
| 2009 | 97 | 2 | 0 | 1 | 10 |
| 2010 | 16 | 2 | 0 | 1 | 2 |
| 2011 | 792 | 2 | 0 | 1 | 47 |
| 2012 | 1,261 | 2 | 0 | 1 | 80 |
| 2013 | 1,450 | 2 | 0 | 1 | 119 |
| 2014 | 864 | 2 | 0 | 1 | 57 |
| 2015 | 500 | 2 | 0 | 1 | 51 |
| 2016 | 1,323 | 2 | 0 | 1 | 100 |
| 2017 | 531 | 2 | 0 | 1 | 12 |
| 1981 | NA | 2 | 0 | 2 | 100 ⁵ |
| 1982 | NA | 2 | 0 | 2 | 100 ⁵ |
| 1984 | 3,035 | 2 | 0 | 2 | 75 ⁴ |
| 1985 | 4,046 ³ | 2 | 0 | 2 | 99 ⁴ |
| 1987 | 4,063 | 2 | 0 | 2 | 100 ⁴ |
| 1988 | 6,660 | 2 | 0 | 2 | 164 ⁴ |
| 1989 | 2,424 | 2 | 0 | 2 | 60 ⁴ |
| 1996 | 829 | 2 | 0 | 2 | 40 |
| 1997 | 2,501 | 2 | 0 | 2 | 128 |
| 1998 | 7,771 | 2 | 0 | 2 | 146 |
| 1999 | 8,768 | 2 | 0 | 2 | 117 |
| 2000 | 8,036 | 2 | 0 | 2 | 65 |
| 2001 | 6,293 | 2 | 0 | 2 | 48 |
| 2002 | 5,325 | 2 | 0 | 2 | 43 |
| 2003 | 2,558 | 2 | 0 | 2 | 27 |
| 2004 | 5,499 | 2 | 0 | 2 | 46 |
| 2005 | 5,698 | 2 | 0 | 2 | 62 |
| 2006 | 6,098 | 2 | 0 | 2 | 117 |
| 2007 | 219 | 2 | 0 | 2 | 14 |
| 2008 | 575 | 2 | 0 | 2 | 29 |
| 2009 | 1,944 | 2 | 0 | 2 | 80 |
| 2010 | 1,801 | 2 | 0 | 2 | 45 |
| 2011 | 1,643 | 2 | 0 | 2 | 84 |
| 2012 | 1,707 | 2 | 0 | 2 | 85 |
| 2013 | 1,785 | 2 | 0 | 2 | 125 |
| 2014 | 1,358 | 2 | 0 | 2 | 72 |
| 2015 | 1,525 | 2 | 0 | 2 | 79 |
| 2016 | 2,822 | 2 | 0 | 2 | 121 |
| 2017 | 951 | 2 | 0 | 2 | 17 |

| Year | Nfish | Fleet | Sex | Part | N |
|------|--------|-------|-----|------|-----|
| 1992 | 774 | 3 | 0 | 2 | 6 |
| 1994 | 1,038 | 3 | 0 | 2 | 9 |
| 1995 | 465 | 3 | 0 | 2 | 4 |
| 1996 | 927 | 3 | 0 | 2 | 7 |
| 1997 | 851 | 3 | 0 | 2 | 7 |
| 1998 | 1,648 | 3 | 0 | 2 | 9 |
| 1999 | 1,079 | 3 | 0 | 2 | 9 |
| 2000 | 339 | 3 | 0 | 2 | 2 |
| 2014 | 82 | 3 | 0 | 2 | 1 |
| 2016 | 74 | 3 | 0 | 2 | 1 |
| 1991 | 927 | 4 | 0 | 2 | 10 |
| 1992 | 3,832 | 4 | 0 | 2 | 31 |
| 1993 | 1,487 | 4 | 0 | 2 | 10 |
| 1994 | 8,604 | 4 | 0 | 2 | 78 |
| 1995 | 6,938 | 4 | 0 | 2 | 61 |
| 1996 | 5,397 | 4 | 0 | 2 | 51 |
| 1997 | 11,191 | 4 | 0 | 2 | 85 |
| 1998 | 16,234 | 4 | 0 | 2 | 100 |
| 1999 | 13,286 | 4 | 0 | 2 | 100 |
| 2000 | 13,613 | 4 | 0 | 2 | 91 |
| 2001 | 11,959 | 4 | 0 | 2 | 87 |
| 2002 | 9,416 | 4 | 0 | 2 | 77 |
| 2003 | 5,023 | 4 | 0 | 2 | 37 |
| 2004 | 4,392 | 4 | 0 | 2 | 41 |
| 2005 | 6,310 | 4 | 0 | 2 | 48 |
| 2006 | 2,874 | 4 | 0 | 2 | 30 |
| 2007 | 809 | 4 | 0 | 2 | 7 |
| 2008 | 1,320 | 4 | 0 | 2 | 11 |
| 2009 | 1,035 | 4 | 0 | 2 | 18 |
| 2010 | 698 | 4 | 0 | 2 | 25 |
| 2011 | 1,678 | 4 | 0 | 2 | 54 |
| 2012 | 999 | 4 | 0 | 2 | 29 |
| 2013 | 1,457 | 4 | 0 | 2 | 35 |
| 2014 | 1,611 | 4 | 0 | 2 | 30 |
| 2015 | 1,799 | 4 | 0 | 2 | 24 |
| 2016 | 1,790 | 4 | 0 | 2 | 27 |
| 2017 | 1,808 | 4 | 0 | 2 | 27 |

9.4.4 Acoustic survey estimates

Estimates of spawning biomass for 2003-2010 are provided in Ryan and Kloser (2012). There are no acoustic estimates for 2011 (not funded) and 2012 (technical issues). Table 9.4 shows the estimates of spawning biomass with their corresponding cv's used in the assessment. Sampling cv's less than 0.3 were increased to 0.3 to account for process error. Low sampling cvs (of 0.19 for example) were considered too low for an acoustic survey and a minimum of 0.3 should be used to reflect the total uncertainty (D. Smith, pers comm., Tuck et al., 2004; Slope RAG 2011). Of 22 acoustic cv's used for hoki in New Zealand, none are lower than 0.3 (Francis, 2009). It is assumed that the spawning ground experiences a turnover rate equal to 2 (i.e. for the model applied here, the spawning biomass estimates are doubled) (Russell and Smith, 2006; Punt et al., 2015). The acoustic survey selectivity is matched

to the maturity ogive, as it is assumed the acoustic survey observes mature fish on the spawning ground.

Table 9.4. The estimated biomass (tonnes) of blue grenadier on the spawning grounds in years 2003 to 2010 (Ryan and Kloser, 2012).

| | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 |
|--------------------------|--------|--------|--------|--------|--------|--------|--------|--------|
| biomass (t) | 24,690 | 16,295 | 18,852 | 42,882 | 56,330 | 24,450 | 24,787 | 20,622 |
| c.v. in assessment model | 0.30 | 0.46 | 0.30 | 0.30 | 0.52 | 0.30 | 1 | 0.33 |
| Sampling cv | 0.16 | 0.46 | 0.14 | 0.14 | 0.52 | 0.22 | 1 | 0.33 |

9.4.5 Egg survey estimates

Egg survey estimates of female spawning biomass are available for 1994 and 1995 (Bulman et al., 1999). The egg-estimates (cv) for 1994 and 1995 respectively are: 57,772 (0.18) and 41,409 (0.29) tonnes. For the analysis considered here, the base-case egg estimates were used.

9.4.6 The Fishery Independent Survey (FIS)

Abundance indices for blue grenadier for the FIS surveys conducted between 2008 and 2016 are provided in Table 9.5 (Knuckey et al., 2017; J. Day, pers comm.). The length-composition data from the FIS are shown in Figure 9.3. These data have not been included in the preliminary base case model presented here but will be considered as a sensitivity in a subsequent report.

Table 9.5. FIS-derived abundance indices for blue grenadier with corresponding coefficient of variation (cv).

| | 2008 | 2010 | 2012 | 2014 | 2016 |
|----------------------|-------|-------|-------|-------|--------|
| Blue grenadier (all) | 15.83 | 3.38 | 10.75 | 19.65 | 58.20 |
| c.v. | 0.30 | 0.28 | 0.23 | 0.21 | 0.23 |
| Spawning | 65.06 | 17.97 | 15.12 | 44.52 | 211.29 |
| c.v. | 0.59 | 0.35 | 0.34 | 0.32 | 0.26 |
| Non-spawning | 30.26 | 9.25 | 10.57 | 50.26 | 10.39 |
| c.v. | 0.57 | 2.31 | 0.93 | 2.19 | 0.34 |

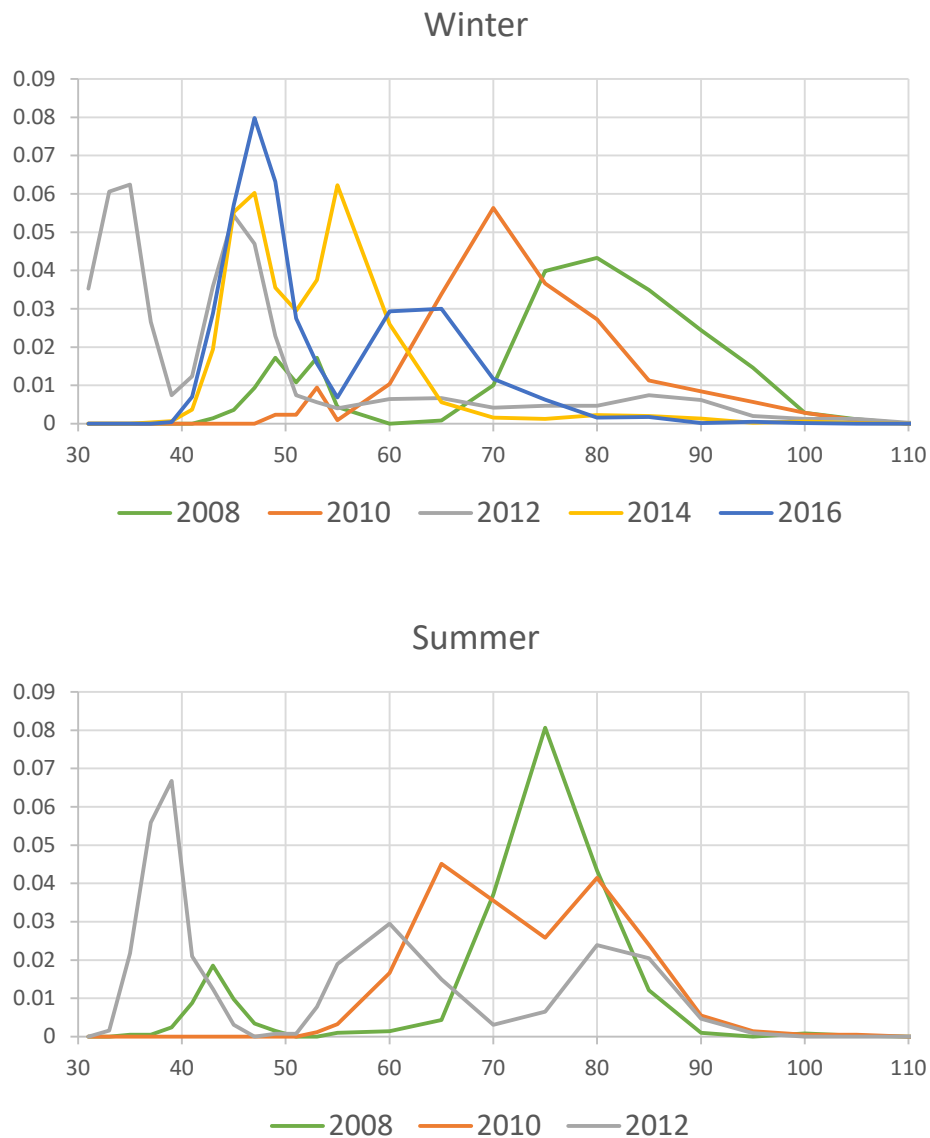


Figure 9.3. The length-compositions for blue grenadier from the FIS from the winter (top) and summer (bottom) surveys.

9.4.7 Biological parameters and stock structure assumptions

The assessment assumes that the proportion of females that spawn in each year is 0.84 and a length at 50% maturity of 63.7cm for females (Russel and Smith, 2006). The female maturity ogive is shown in Figure 9.4.

The length weight-relationship for males and females was estimated from spawning fishery data over years 1999 to 2008 (Figure 9.4). Natural mortality for females is estimated when fitting the model and male natural mortality is assumed to be 20% greater than the value for females based upon assumptions made for hoki in New Zealand (McAllister et al., 1994).

Francis (2009) reviews the values of steepness used in New Zealand hoki assessments, where a value of $h=0.9$ had been used since 1994. This value of steepness was derived from work of Punt et al. (1994) using 45 stocks of gadiform species (0.9 is the median). Following an analysis of the profile likelihood,

the effect of steepness on the 2007 assessment and additional information of Myers et al. (1999; 2002) beyond that used by Punt et al. (1994), Francis (2009) concludes that steepness should be reduced to $h=0.75$. This value of steepness was assumed in the previous blue grenadier assessments in 2011 and 2013 (Tuck, 2011; 2013) and in this assessment.

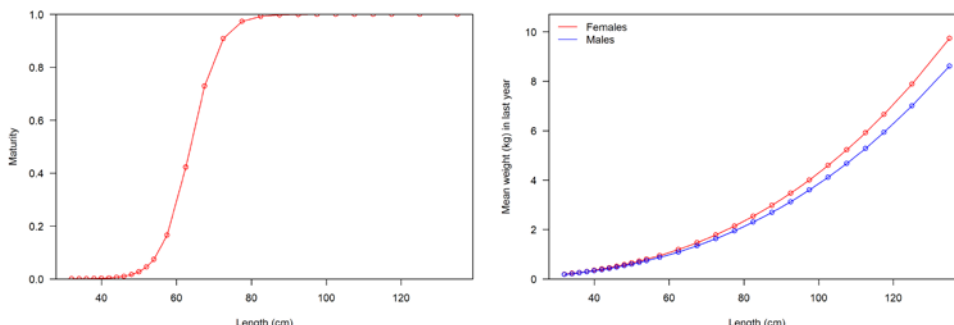


Figure 9.4. The maturity ogive by length for female blue grenadier (parameters from Russell and Smith (2006)) and the length-weight relationship for males and females.

9.4.8 Age-reading error

Updated standard deviations for aging error by reader (A and B) have been estimated, producing the age-reading error matrix of Table 9.6 (A. Punt, pers. comm.). Reader A applied to years 1991-93 and 2007-17, and reader B to years 1984-90 and 1994-2006.

Table 9.6. The standard deviation of age reading error.

| Age | St Dev | |
|-----|--------|-------|
| | A | B |
| 0 | 0.223 | 0.282 |
| 1 | 0.223 | 0.282 |
| 2 | 0.266 | 0.299 |
| 3 | 0.301 | 0.318 |
| 4 | 0.331 | 0.338 |
| 5 | 0.357 | 0.359 |
| 6 | 0.378 | 0.383 |
| 7 | 0.396 | 0.408 |
| 8 | 0.412 | 0.435 |
| 9 | 0.424 | 0.464 |
| 10 | 0.435 | 0.495 |
| 11 | 0.444 | 0.529 |
| 12 | 0.452 | 0.565 |
| 13 | 0.459 | 0.604 |
| 14 | 0.464 | 0.646 |
| 15 | 0.469 | 0.692 |
| 16 | 0.473 | 0.741 |
| 17 | 0.476 | 0.793 |
| 18 | 0.479 | 0.850 |
| 19 | 0.481 | 0.911 |
| 20 | 0.483 | 0.976 |

9.4.9 Analytic approach

The 2018 preliminary base case assessment of blue grenadier uses an age- and size-structured model implemented in the generalized stock assessment software package, Stock Synthesis (SS) (Version 3.30.12.00-safe, NOAA 2018). The methods utilised in SS are based on the integrated analysis paradigm. SS can allow for multiple seasons, areas and fleets, but most applications are based on a single season and area. Recruitment is governed by a stochastic Beverton-Holt stock-recruitment relationship, parameterized in terms of the steepness of the stock-recruitment relationship (h), the expected average recruitment in an unexploited population (R_0), and the degree of variability about the stock-recruitment relationship (σ_r). SS allows the user to choose among a large number of age- and length-specific selectivity patterns. The values for the parameters of SS are estimated by fitting to data on catches, catch-rates, acoustic and egg surveys, discard mass, discard and retained catch length-compositions, and conditional age-at-length data. The population dynamics model and the statistical approach used in fitting the model to the various data types are given in the SS technical documentation (Methot, 2005).

This assessment follows the agreements made at the 2013 meetings of Slope RAG. These were: include gender-specific selectivity for the spawning fishery, estimate natural mortality for females, use historical discard tonnages estimated by MAFRI, include cohort-dependent growth, and set steepness at 0.75.

The base-case model includes the following key features:

- a) Two sub-fisheries are included in the model – the spawning sub-fishery that operates during winter (June – August inclusive) off western Tasmania (zone 40), and the non-spawning sub-fishery that operates during other times of the year and in other areas throughout the year.
- b) The selectivity pattern was assumed to be length-specific, logistic and time-invariant for the spawning fleet and dome-shaped for the non-spawning fleet. The parameters of the selectivity function for each fleet were estimated within the assessment. A change in selectivity from 2005 was considered as a sensitivity for the non-spawning fleet; however this did not substantially affect the fits nor management quantities of interest.
- c) Blue grenadier consists of a single stock within the area of the fishery.
- d) The model accounts for males and females separately.
- e) The population was at its unexploited biomass with the corresponding equilibrium (unexploited) age-structure at the start of 1960.
- f) The CVs of the CPUE indices were initially set at a value equal to the standard error from a loess fit (0.25; Sporic and Haddon, 2018), before being re-tuned to the model-estimated standard errors within SS. The acoustic estimates were tuned through the estimation of an extra parameter that adds to the model input standard errors. This is done within SS.
- g) Discard tonnage was estimated through the assignment of a retention function for the non-spawning fleet. This was defined as a logistic function of length, and the inflection and slope of this function were estimated where discard information was available.
- h) The rate of natural mortality, M , is assumed to be constant with age, and also time-invariant. The value for female M is estimated within the assessment. Following previous assessments, male natural mortality is assumed to be 20% greater than that of females.
- i) Recruitment to the stock is assumed to follow a Beverton-Holt type stock-recruitment relationship, parameterised by the average recruitment at unexploited spawning biomass, R_0 , and

the steepness parameter, h . Steepness for the base-case analysis is set to 0.75. Deviations from the average recruitment at a given spawning biomass (recruitment residuals) are estimated for 1974 to 2015. Deviations are not estimated before 1974 or after 2015 because there are insufficient data to permit reliable estimation of recruitment residuals outside of this time period.

- j) The initial value of the parameter determining the magnitude of the process error in annual recruitment, σ_r , is set to 1.0, reflecting the large variation in recruitment observed for blue grenadier. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Taylor and Method (2011).
- k) The population plus-group is modelled at age 20 years. The maximum age for age observations was 15 years, reflecting that used in previous assessments.
- l) Growth is assumed to follow a von Bertalanffy type length-at-age relationship, with the parameters of the growth function being estimated separately for females and males inside the assessment model. Growth is also assumed to vary through time and be cohort (year class) specific. Evidence for time-varying and cohort specific growth in blue grenadier has been accumulating for over several decades (see Punt and Smith 2001; Whitten et al., 2013). The 2018 base-case model treats conditional age-at-length information as data, and predicts the expected length-at-age for each year. This is achieved by estimating the parameters of a von Bertalanffy growth function where the expected annual growth increment is based on the von Bertalanffy growth function but with a growth rate parameter that is determined by an expected value and a cohort-specific deviation. Cohort-specific deviations from average growth are estimated in the base case model for year classes 1978 to 2015.
- m) Retained and discard onboard length sample sizes were capped at 200 and a minimum of 100 fish measured was required for length-composition data to be included in the assessment. For port samples, numbers of trips were used as the sampling unit, with a cap of 100. The number of fish measured is not used as the sample size because the appropriate sample size for length-composition data is probably more closely related to the number of shots (onboard) or trips (port) sampled, rather than the number of fish measured (Table 9.3).

The values assumed for some of the parameters of the preliminary base case model are shown in Table 9.7.

Table 9.7. Parameter values assumed for some of the non-estimated parameters of the base-case model (BC).

| Parameter | Description | BC |
|------------|---|-----------------------------|
| M_f | Natural mortality for females | Estimated |
| M_m | Natural mortality for males | 1.2* M_f |
| σ_r | Initial c.v. for the recruitment residuals | 1.0 |
| σ_g | Input standard deviation for the cohort growth deviations | 0.1 |
| h | “steepness” of the Beverton-Holt stock-recruit curve | 0.75 |
| x | age observation plus group | 15 years |
| μ | fraction of mature population that spawn each year | 0.84 |
| aa | Female allometric length-weight equations | 0.01502 g ⁻¹ .cm |
| bb | Female allometric length-weight equations | 2.728 |
| aa | Male allometric length-weight equations | 0.0168 g ⁻¹ .cm |
| bb | Male allometric length-weight equations | 2.680 |
| l_m | Female length at 50% maturity | 63.7cm |
| l_s | Parameter defining the slope of the maturity ogive | -0.261 |

9.4.9.1 *Tuning method*

Iterative rescaling (reweighting) of input CVs or input sample sizes is a repeatable method for ensuring that the expected variation of the different data streams is comparable to what is input (Pacific Fishery Management Council, 2018). Sampling standard deviations/ CVs and stage-1 effective sample sizes for most of the data (CPUE, survey indices, composition data) used in fisheries assessments underestimate their true variance by only reflecting measurement or estimation error and not including process (or model) error.

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size was equal to the effective sample size calculated within the model. In SS3.30 there is an automatic adjustment made to survey CVs (CPUE).

1. Set the standard error for the log of relative abundance indices (CPUE, acoustic abundance survey, or FIS) to their estimated standard errors for each survey or for CPUE (and FIS values) to the standard deviation of a loess curve fitted to the logs of original data (which will provide a more realistic estimate compared to that obtained from the original statistical analysis). SS3.30 then re-balances the relative abundance variances appropriately.
2. The initial value of the parameter determining the magnitude of the process error in annual recruitment, σ_r , is set to 1.0, reflecting the large variation in recruitment observed for blue grenadier. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Taylor and Method (2011).

An automated tuning procedure was used for the remaining adjustments. For the conditional age-at-length and length-composition data:

3. Multiply the stage-1 sample sizes for for the conditional age-at-length data by the sample size multipliers using the approach of Punt (2017).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length-composition data using the ‘Francis method’ (Francis, 2011).
5. repeat steps 2 and 3, until all are converged and stable (proposed changes are $< 1 - 2\%$).

This procedure may change in the future after further investigations, but this approach constitutes current best practice.

9.5 **Bridging**

9.5.1 **Bridging from the 2013 to 2018 assessments**

The previous full quantitative assessment for blue grenadier was conducted in 2013 (Tuck, 2017) using Stock Synthesis (version SS-V3.22a, Methot, August 2012). The 2018 assessment uses the current version of Stock Synthesis (version SS-V3.30.12.00-safe, Methot et al., 2018).

As a first step in the process of bridging to a new model (referred to as Bridge 1), minor refinements and corrections (based on current best practice) were made to input values, and the data from the 2013 assessment were used with the new software (SS-V3.30). Bridging then continued (Bridge 2) by the inclusion in the model of updated data (pre-2013) and new data from 2013-17. These additional data

included new catch, CPUE, length-composition and conditional age-at-length data, age-reading error and discard mass. The last year of recruitment estimation was extended to 2015 (from 2010 in the 2013 assessment). The usual process of bridging to a new model by adding new data piecewise and analysing which components of the data could be attributed to changes in the assessment outcome was conducted. Details of this process are provided below.

9.5.1.1 Bridge 1 – initial bridging steps from the 2013 assessment

The 2013 assessment was first updated and converted to the latest version of Stock Synthesis and the latest tuning method because the 2013 blue grenadier assessment used methods and software appropriate for that time (denoted “2013_Updated”). This allows a comparison of what the assessment would have produced in 2013 had current methods been applied. The most recent version of the software is Stock Synthesis version SS-V3.30.12.00-safe.

9.5.1.2 Bridge 2 – inclusion of new data

The data inputs to the assessment come from multiple sources: length-composition and conditional age-at-length data from the trawl fishery, updated standardized CPUE series (Sporcic and Haddon, 2018), the annual total mass landed and discarded, and age-reading error matrices. Data were formulated by calendar year (i.e. 1 Jan to 31 Dec) and separated by sub-fishery, being the spawning fleet (Zone 40; months 6, 7, 8) and the non-spawning fleet (all other periods and zones; excluding the GAB).

Starting from the converted and re-tuned 2013 base case model (2013_New_Tuned), additional and updated data to 2017 were added sequentially to develop a preliminary base case for the 2018 assessment:

1. Start with the re-tuned 2013 assessment (2013_Tuned).
2. Change final assessment year to 2017, add landed catch until 2017 (addCatch2017).
3. Add CPUE to 2017 (from Sporcic and Haddon (2018)) (addCPUE2017).
4. Update length-composition data, including both port and onboard length-compositions (addLength2017).
5. Add the updated age-reading error matrix and conditional age-at-length data to 2017 (addAge2017).
6. Change the final year for which recruitments are estimated from 2010 to 2015 (extendRecruitment).
7. Change the final year of cohort growth deviation estimation from 2009 to 2015 (extendCGD).
8. Add updated discard mass estimates to 2017 (addDiscard2017).
9. Retune using latest tuning protocols (2018_BC_Tuned).

9.6 Results

9.6.1 Bridging

9.6.1.1 Bridge 1 – initial bridging steps from the 2013 assessment

While the current year depletion has changed little, the magnitude of the spawning biomass has changed under the new tuning method (Figure 9.5). Changes to the absolute magnitude of spawning biomass are not uncommon in blue grenadier. Previous assessments illustrated by a retrospective analysis by Punt et al. (2017) have shown initial spawning biomass estimates of between 40,000t and 55,000t (Figure 9.5).

9.6.1.2 Bridge 2 – inclusion of new data

The inclusion of new catch and CPUE data does not influence the trajectory of female spawning biomass substantially (Figure 9.6). However, subsequent data sources (length, conditional age-at-length, recruitment, discards) all imply a reduction in relative and absolute biomass (Figure 9.6). Extending recruitment estimation to 2015 leads to a marked increase in recent spawning biomass. This is not surprising because this feature allows considerable additional flexibility to fit the available data, which then flow into the spawning and available biomass. Recruitment estimates over the last five to six years are generally large and stable (Figure 9.7). There was concern in 2013 (Tuck, 2013) regarding the actuality of the estimated recruitments for the 2009 and 2010 cohorts. With the addition of new data (lengths and ages in particular), it appears these recruitments have been maintained, and in fact have been followed by further strong annual cohorts, through to 2015. As before, the estimates magnitudes of the most recent recruitments should be treated with some caution. However, it is an encouraging sign for the stock that stable strong recruitment is evident (compared to historical period of recruitment failure that have been observed for this stock; Figure 9.7).

While current spawning depletion is estimated to be at or above virgin levels, the updated assessment illustrates that the spawning biomass trajectory dropped below the target from 2012 to 2015; a period when no assessment was conducted (Figure 9.6). The decline in biomass (and magnitude of the recruitments) can be seen through each addition of data, implying that it is not being driven by one data source (such as the discard mass) and is a consistent signal across data inputs.

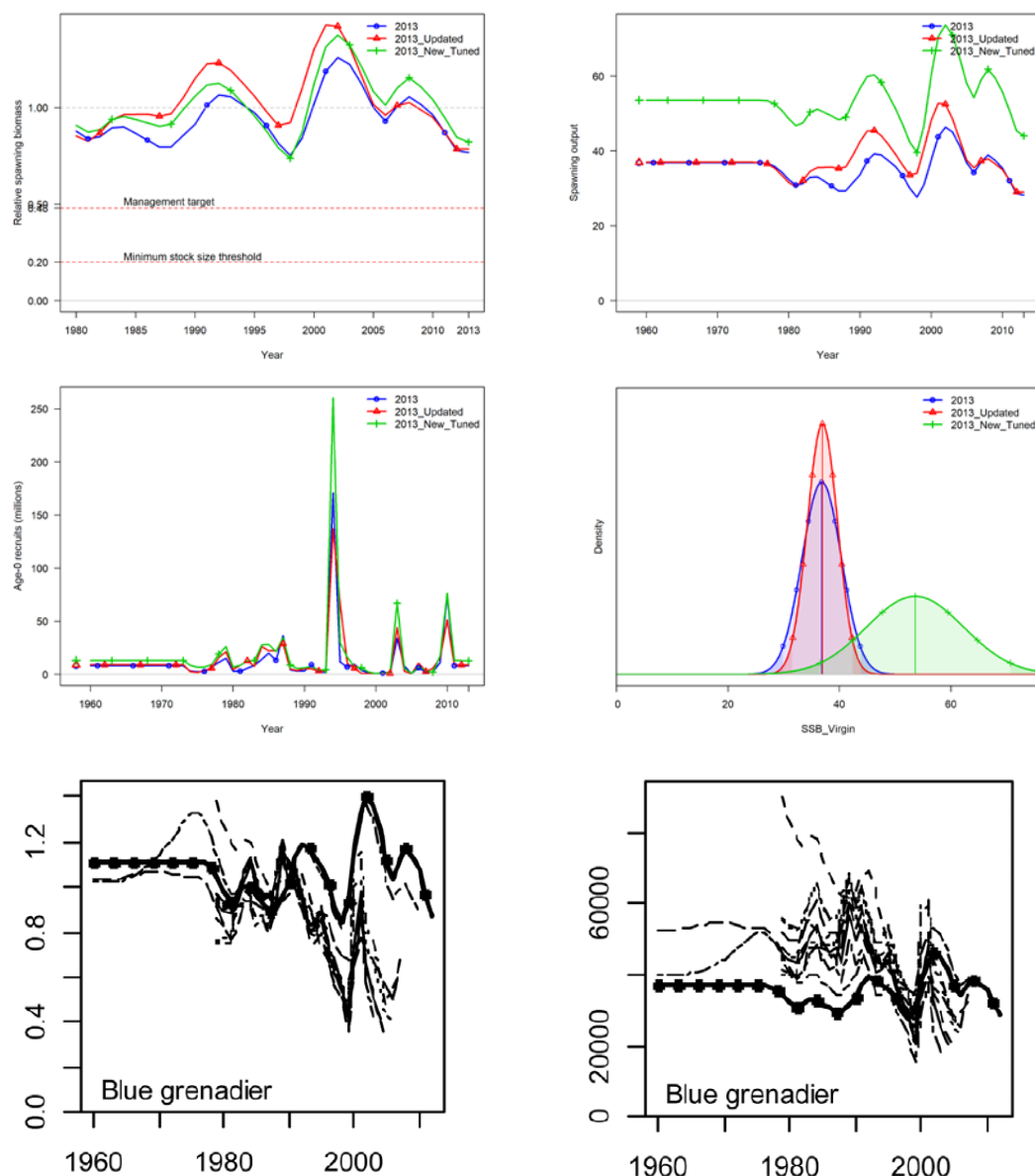


Figure 9.5. Comparison of the relative and absolute spawning biomass (top), recruitment time series (middle left) and virgin biomass estimate (middle right) for the 2013 assessment (2013), updates (2013_Updated, including the new version of SS) and the new tuning method applied (2013_New_Tuned). The lower figures show the relative (bottom left) and absolute (bottom right) spawning biomass trajectories for blue grenadier assessments dating back to the 1990s (from Punt et al. 2017). The solid line is the 2013 assessment.

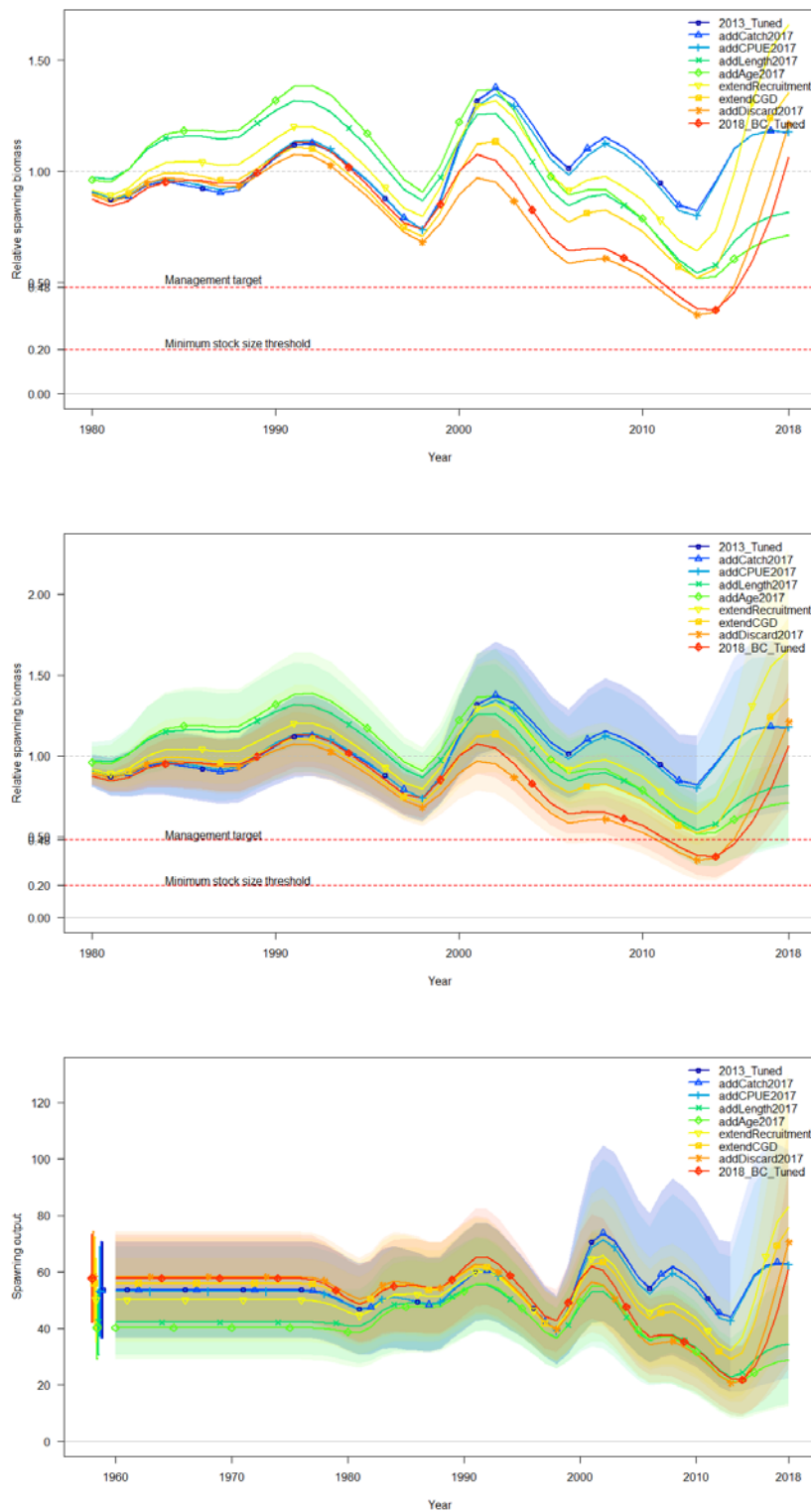


Figure 9.6. The relative (top, middle) and absolute (bottom) female spawning biomass trajectory for Bridge 2, moving from the re-tuned 2013 assessment (2013_Tuned), adding data sequentially through to the 2018 tuned base case (2018_BC_Tuned).

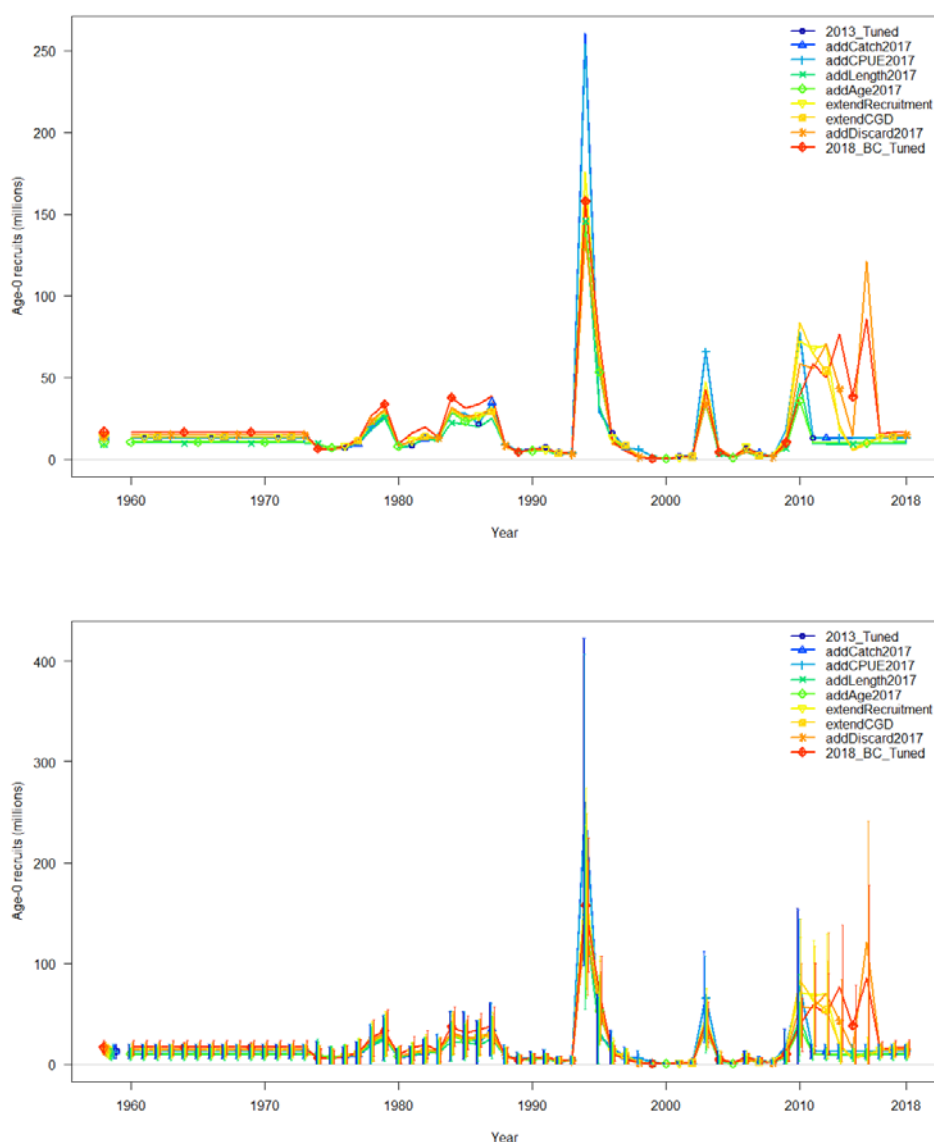


Figure 9.7. The time series of recruitments for blue grenadier for Bridge 2, moving from the re-tuned 2013 assessment (2013_Tuned), adding data sequentially through to the 2018 tuned base case (2018_BC_Tuned).

9.6.2 The 2018 preliminary base case

The base case specifications agreed by the SlopeRAG in 2013 were largely maintained into the 2018 preliminary base case presented here. The main differences are: separating length-compositions into onboard and port collected components, weighting length-compositions by shots (onboard) and trips (port) rather than fish measured; and using the latest new tuning methods.

The estimated time series of recruitment from the preliminary base-case assessment shows the typical episodic nature of blue grenadier recruitment, with strong year-classes in 1979, the mid-1980s, 1994, and 2003 with very little recruitment between these years. However, the more recent recruitments show a more stable level of annual recruitment than has been observed before. Noting there is now good evidence for strong recruitments in the early 2010s, the magnitude of the most recent recruitments (e.g.

2015) will remain somewhat poorly estimated until these fish move well into the available stock of the fishery (Figure 9.8).

The trajectories of spawning biomass and spawning biomass relative to the un-exploited level are shown in Figure 9.8. This shows the increases and decreases in spawning biomass as the strong cohorts move into and out of the spawning population. Results show reasonably good fits to the length-composition data, implied age compositions, egg survey and acoustic survey (Appendix). The fit to the discard mass has improved compared to the 2013 assessment. This is a result of a change in the shape of the non-spawning selectivity function that allows more small fish to be caught and subsequently discarded than the functions described in the 2013 assessment. As has been noted in previous blue grenadier assessments, the fit to the non-spawning CPUE is generally poor; the model is unable to fit to the high early catch rates and over-estimates catch rates in the early 2000s. However, fits improve from 2005 onwards. Fits to length compositions using onboard samples are generally very good. Port data show some mis-alignment, with the model expecting fish of a larger length. However, the implied age fits for both onboard and port samples are excellent across both fleets.

The estimated virgin female biomass is 57,638 tonnes (SD 7,943) (compared to 36,815 tonnes in the 2013 assessment). While a substantial increase in initial biomass compared to the 2013 assessment, this level of initial biomass has been observed before in blue grenadier stock assessments (see retrospective analysis in Figure 9.5). There is also clearly a degree of uncertainty regarding the initial biomass as can be seen from the likelihood profile for $\ln R_0$ (Figure 9.15) and the 95% confidence intervals of the biomass trajectory (Figure 9.8, middle). Also, the estimated biomass when fit to the 2018 egg survey, which influences the magnitude of biomass, has moved closer to the estimated point in 1994 and is larger (in terms of biomass) than the corresponding estimate in 2013 (see Appendix, comparison of 2013 and 2018 assessments), and in doing so has improved the fit to the egg survey points.

In the 2013 assessment, the estimated spawning biomass under the base-case scenario for 2012 was 77% of virgin stock biomass (SB_0) and the estimated spawning biomass in 2014, which was used in the harvest control rule, was approximately 94% SB_0 .

The 2018 preliminary assessment estimates that the projected 2019 spawning stock biomass will be 138% SB_0 . (SD 28.5%).

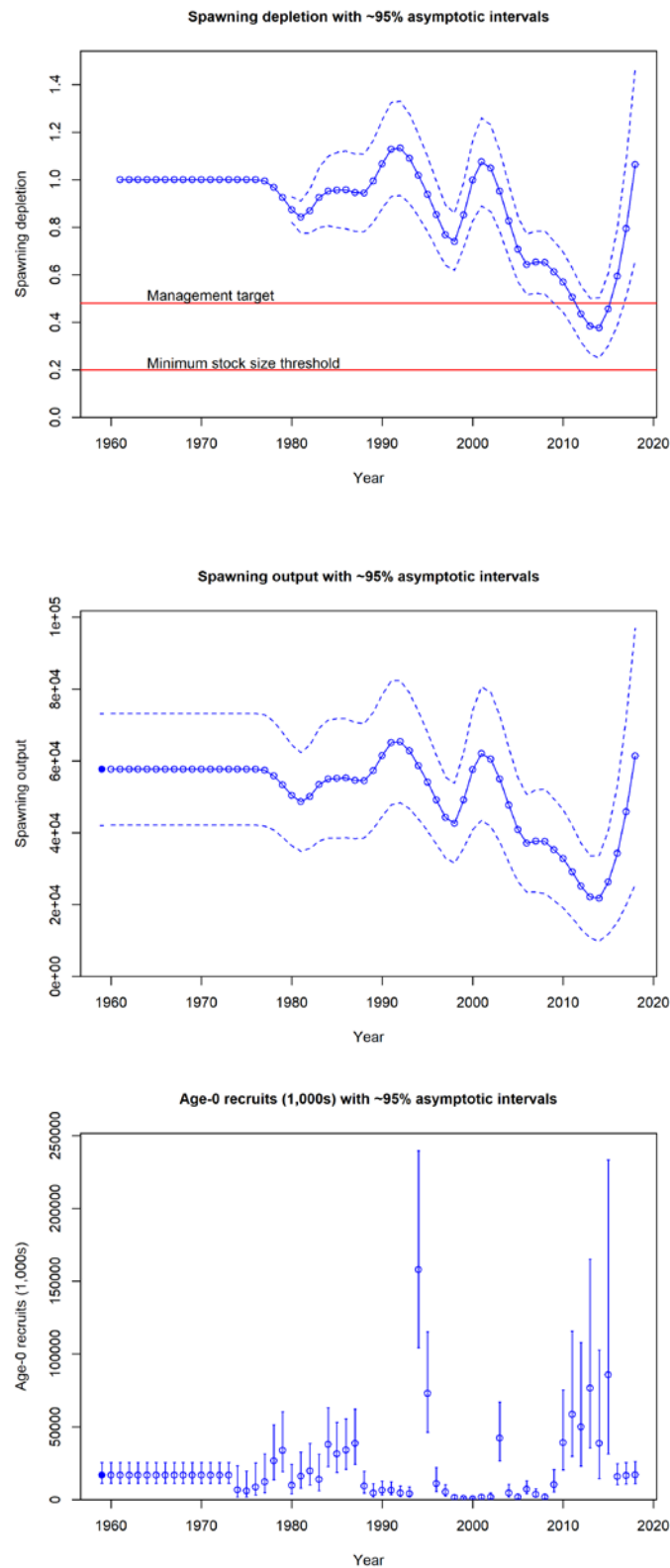


Figure 9.8. The estimated time-series of relative spawning biomass and annual recruitment for the 2018 preliminary base case assessment for blue grenadier.

9.6.3 Retrospective analysis

There are two types of analyses that are often called retrospective analysis. The first takes the base-case assessment and re-runs the assessment sequentially removing a year's data. The second type of retrospective analysis (historical analysis) displays the assessment results from previous years when an assessment was conducted (Punt et al., 2017). Both of these retrospective analyses are shown below.

9.6.3.1 Sequential removal of annual data

Sequentially removing each year's data from 2017 to 2012 (i.e. all data up to year 2012 are removed) illustrates that from 2012 to 2017, the addition of annual data sources all support evidence for a gradual decrease in the relative biomass trajectory and a gradual increase in initial biomass (Figure 9.9).

9.6.3.2 Previous assessment results

Punt et al. (2017) provided a retrospective investigation of assessment uncertainty for fish stocks in the SESSF. This involve reporting the spawning biomass trajectories of every previous assessment (Tier 1) conducted in the SESSF (Figure 9.5). This was reproduced in more detail for blue grenadier in Figure 9.10, showing each trajectory with associated year of the assessment. This shows that assessment outputs of biomass have varied considerably among assessments. Relative trends in spawning biomass show consistent fluctuations between assessments as recruitments enter and leave the fishery. However, the magnitude of the biomass varies.

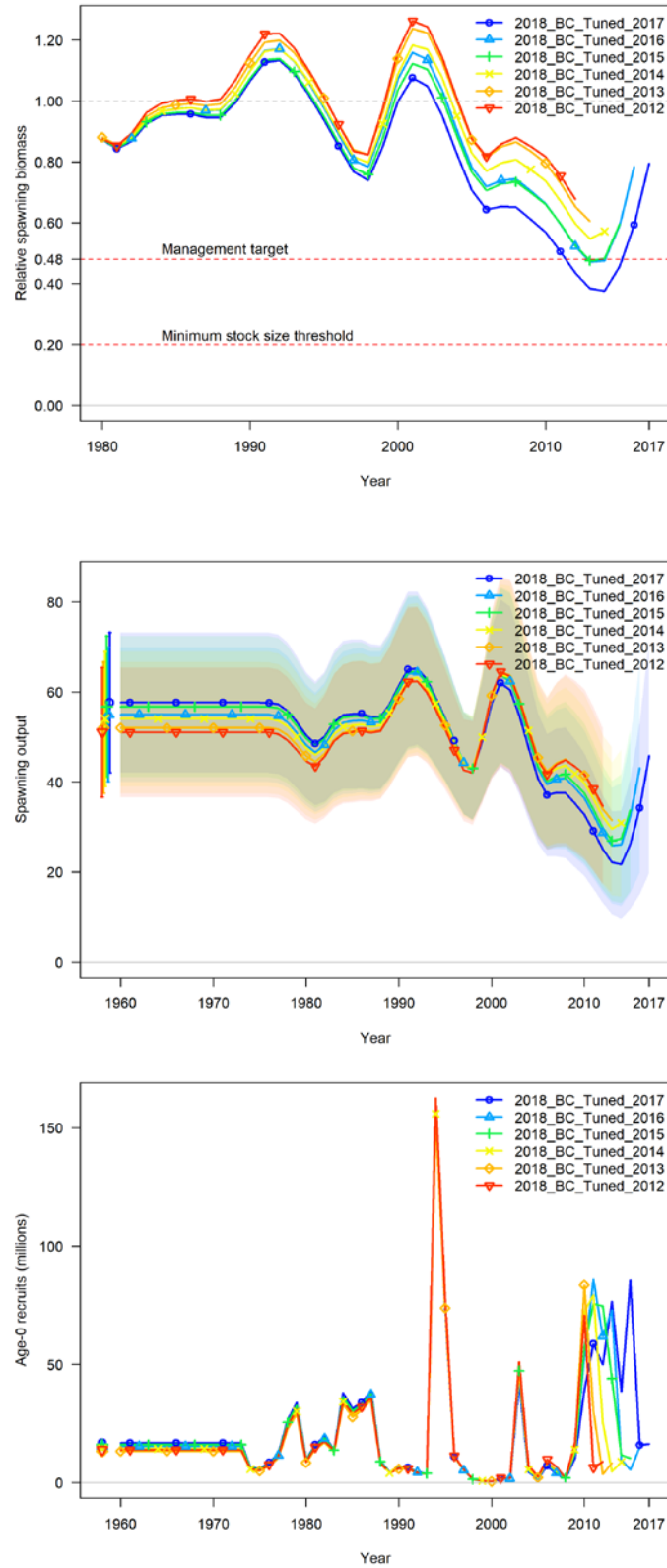


Figure 9.9. A retrospective analysis showing spawning biomass and recruitment time-series as each year’s data are removed from the assessment.

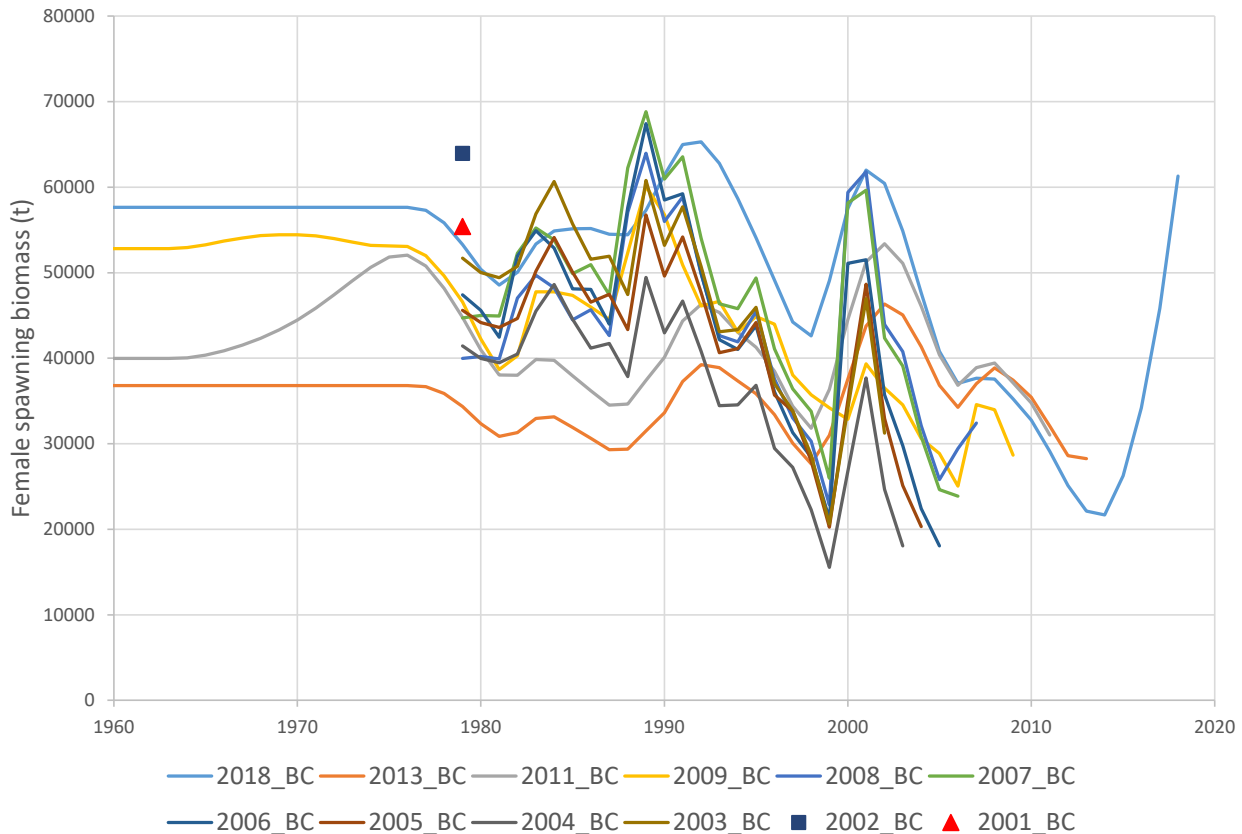


Figure 9.10. A retrospective of assessment outputs of female spawning biomass from each stock assessment from 2001 to 2018. Note that for 2001 and 2002 only values of biomass at 1979 were available.

9.6.4 Likelihood profiles

As stated by Punt (2018), likelihood profiles are a standard component of the toolbox of applied statisticians. They are most often used to obtain a 95% confidence interval. Many stock assessments “fix” key parameters such as M and steepness based on *a priori* considerations. Likelihood profiles can be used to evaluate whether there is evidence in the data to support fixing a parameter at a chosen value. If the parameter is within the entire range of the 95% confidence interval, this provides no support in the data to change the fixed value. If the fixed value is outside the 95% confidence interval it would be reasonable for a review panel to ask why the parameter was fixed and not estimated, and if the value is to be fixed, on what basis and why should what amounts to inconsistency with the data be ignored. Integrated stock assessments include multiple data sources (e.g., commonly catch-rates, length-compositions, and age-compositions) that may be in conflict, due for example to inconsistencies in sampling, but more commonly owing to incorrect assumptions (e.g., assuming that catch-rates are linearly related to abundance), i.e. model-misspecification. Likelihood profiles can be used as a diagnostic to identify these data conflicts (Punt, 2018).

Standard parameters to consider are natural mortality (M), steepness (h) and the logarithm of the unfished recruitment ($\ln R_0$). For blue grenadier, the likelihood profile for female natural mortality, M_f , an estimated parameter, is shown in Figure 9.11. This shows that M_f is generally well estimated with conditional age-at-length, length composition and the index data all consistent in terms of the relative support for the estimated value of M_f . Spawning biomass trajectories under alternative values for M_f

show a considerable spread of initial biomass of ~40,000t for $M_f=0.1$ through to 80,000t for $M_f=0.3$. Larger values for M_f lead to larger estimates of recruitment (not shown).

The likelihood profile on steepness confirms that it is not a well-defined parameter. The 95% confidence intervals are very broad, starting at 0.6 and going beyond what would be considered reasonable values for this parameter (Figure 9.13). The spawning biomass trajectory for alternative values of steepness is invariant to values greater than 0.5 (Figure 9.14).

The likelihood profile for $\ln R_0$ shows a 95% confidence interval between 9.3 and 10.2 (Figure 9.15). The estimated value is 9.73. This is a broad range and corresponds to values of initial female spawning biomass between 46,800t and 70,000t. This matches much of the variation already observed in estimated values of initial biomass from the retrospective analysis of previous stock assessments of blue grenadier (Figure 9.10). Conflicts in the data signals can be seen in Figure 9.15, with the length-composition and discard data showing opposite trends to recruitment and the survey data.

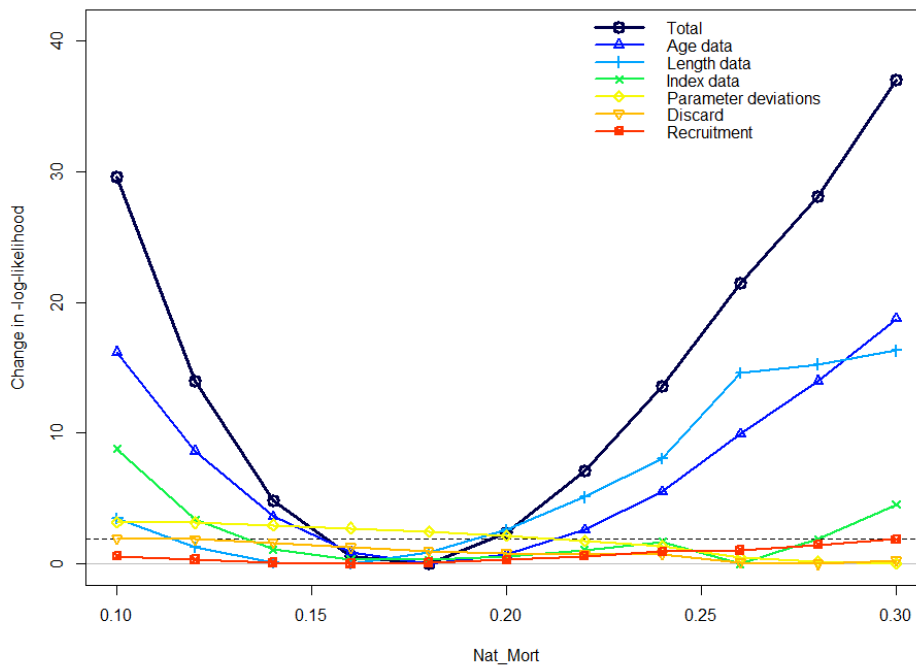


Figure 9.11. The likelihood profile for female natural mortality. The optimal value from the base case is $M=0.173\text{yr}^{-1}$.

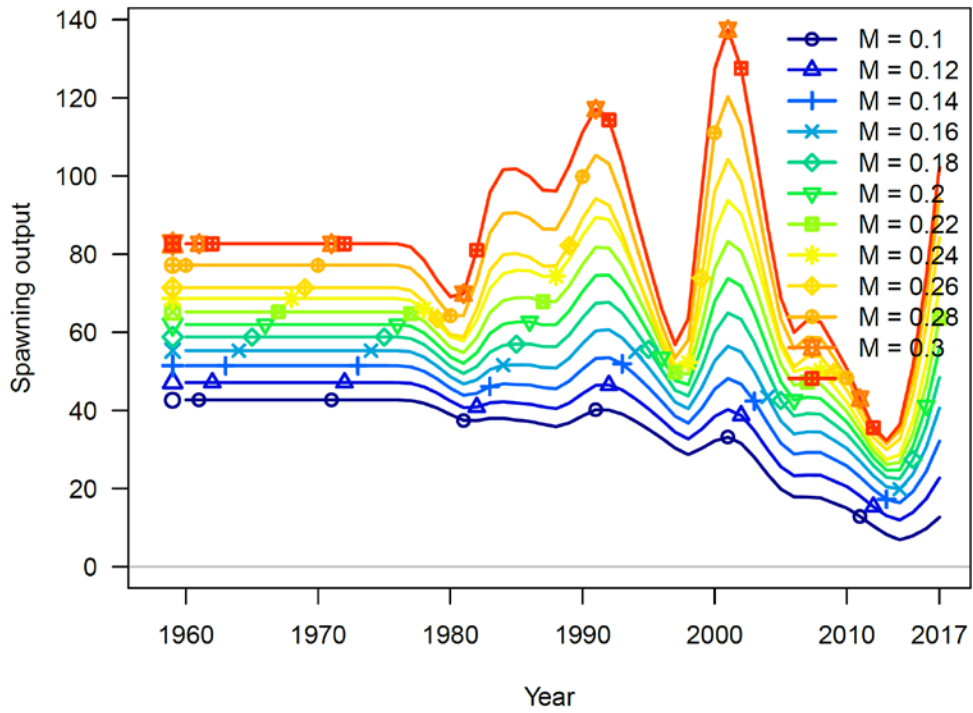


Figure 9.12. The female spawning biomass trajectories under alternative fixed values of M .

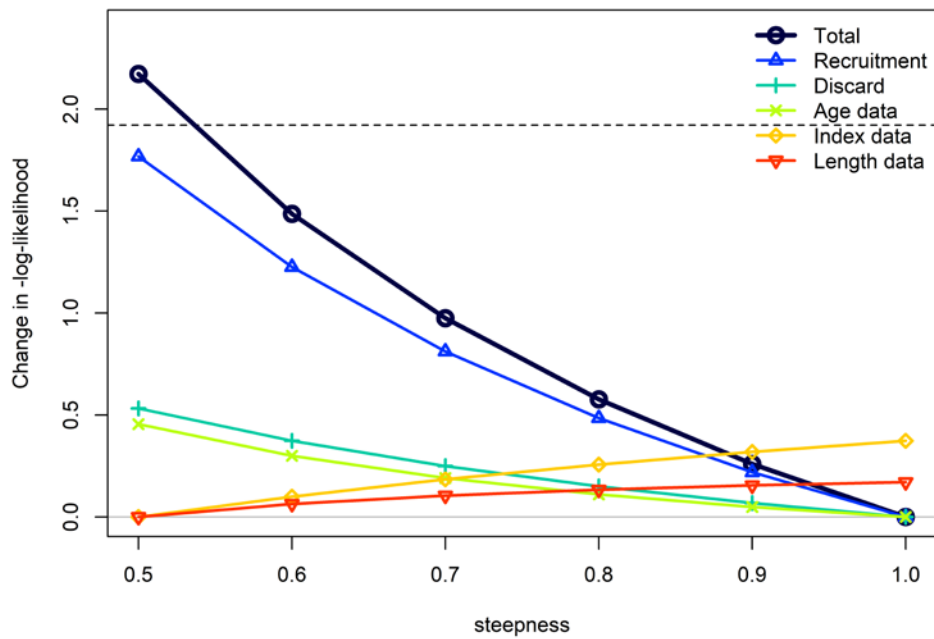


Figure 9.13. The likelihood profile for steepness. The fixed value used in the base case is $h=0.75$.

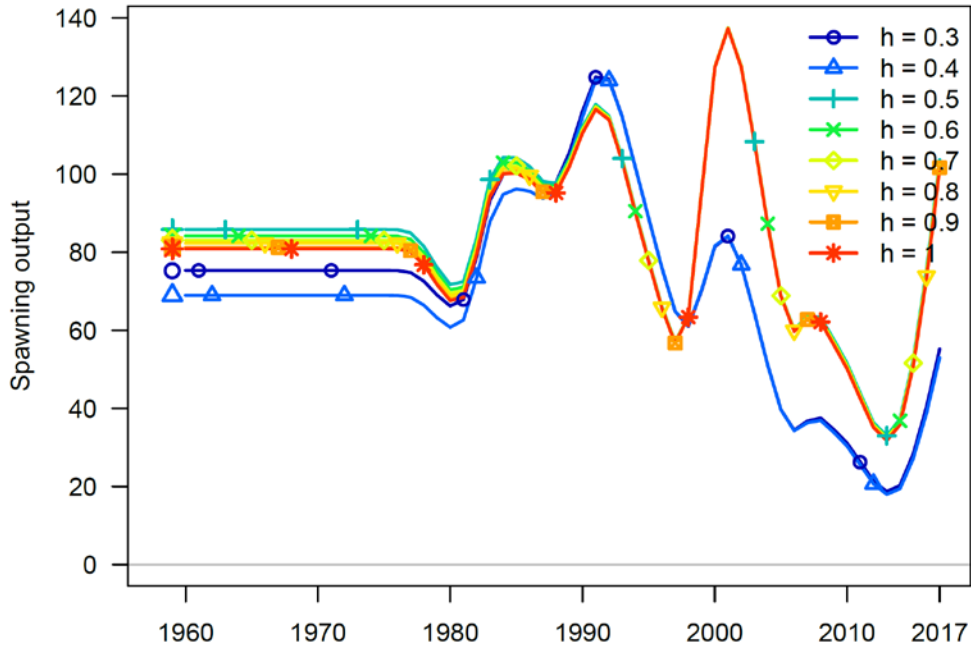


Figure 9.14. The female spawning biomass trajectories under alternative fixed values of steepness, h .

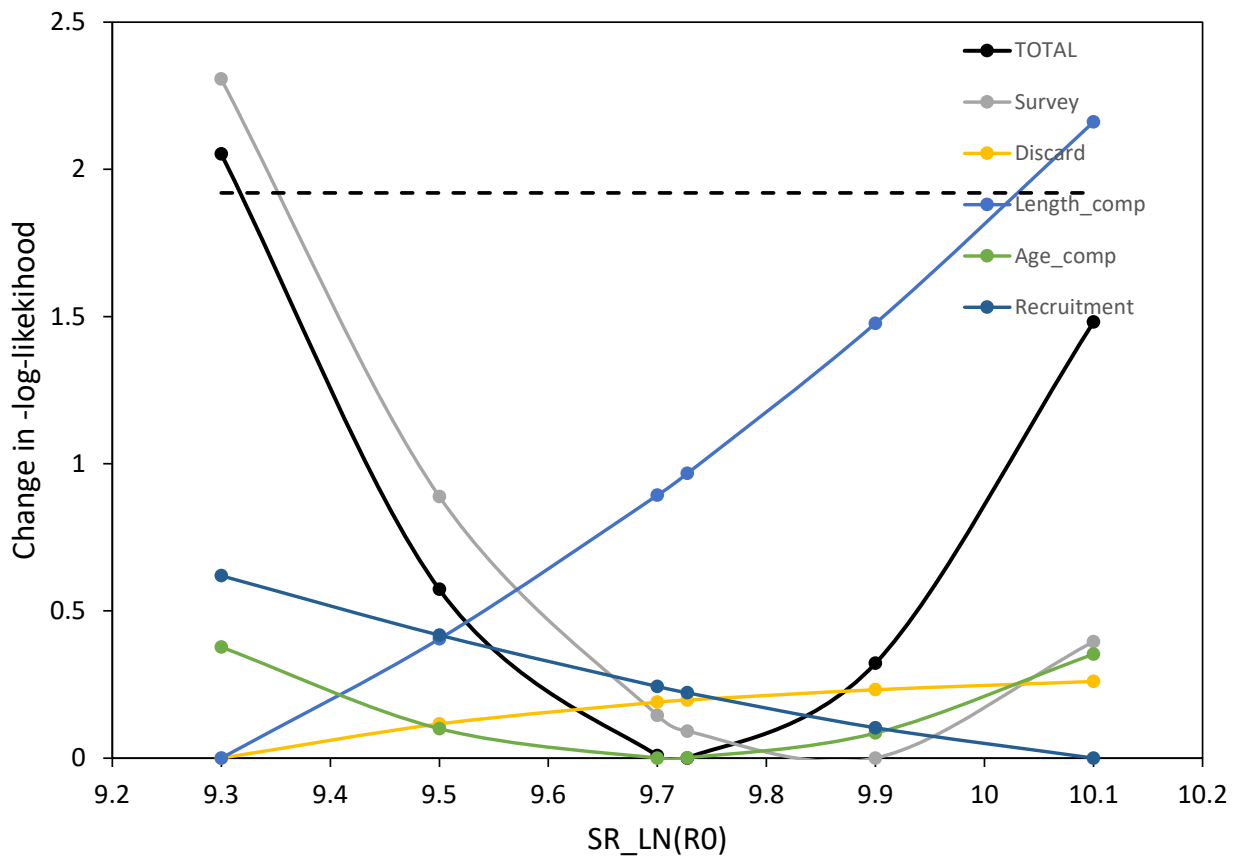


Figure 9.15. The likelihood profile for $\ln R_0$. The estimated value in the base case is $\ln R_0=9.73$.

9.7 Acknowledgements

Age data were provided by Kyne Krusic-Golub (Fish Ageing Services), ISMP and AFMA logbook and CDR data were provided by John Garvey (AFMA). Franzis Althaus, Mike Fuller, Roy Deng, and Paul Burch (CSIRO) pre-processed the data. Jemery Day, André Punt, Sandra Curin-Osorio, Paul Burch, Malcolm Haddon, Robin Thomson and Miriana Sporcic are thanked for helpful discussions on this work. Richard Methot, Chantel Wetzel and Ian Taylor (NOAA Fisheries) are thanked for all the Stock Synthesis support and advice. Malcolm Haddon provided useful code for auto-tuning, Athol Whitten provided useful R code for organising plots.

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9.9 Appendix A

A.1 Preliminary base case diagnostics

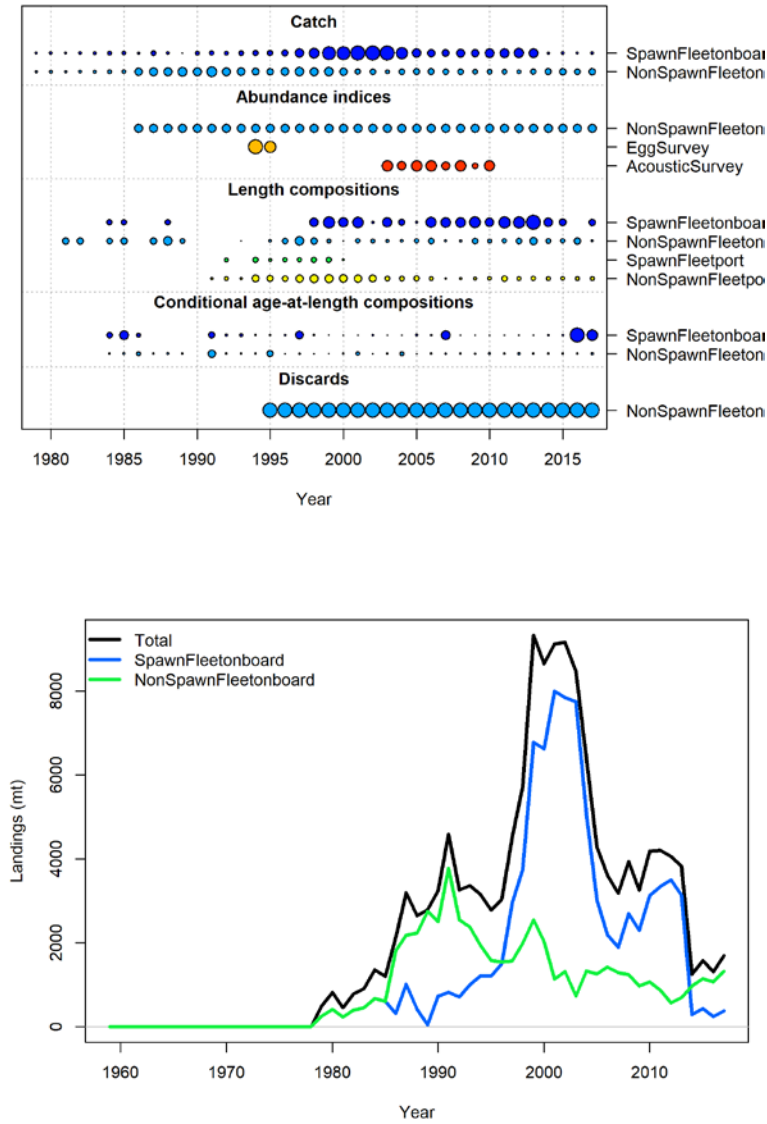


Figure A 9.1. Summary of data sources and the catch time-series for the preliminary base case assessment.

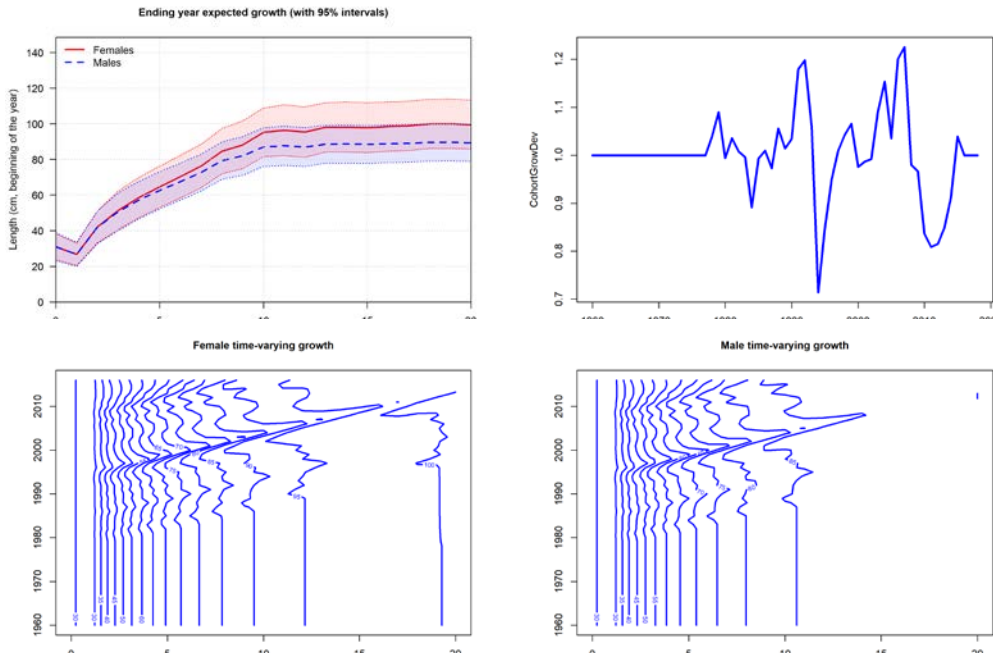


Figure A 9.2. Growth for blue grenadier.

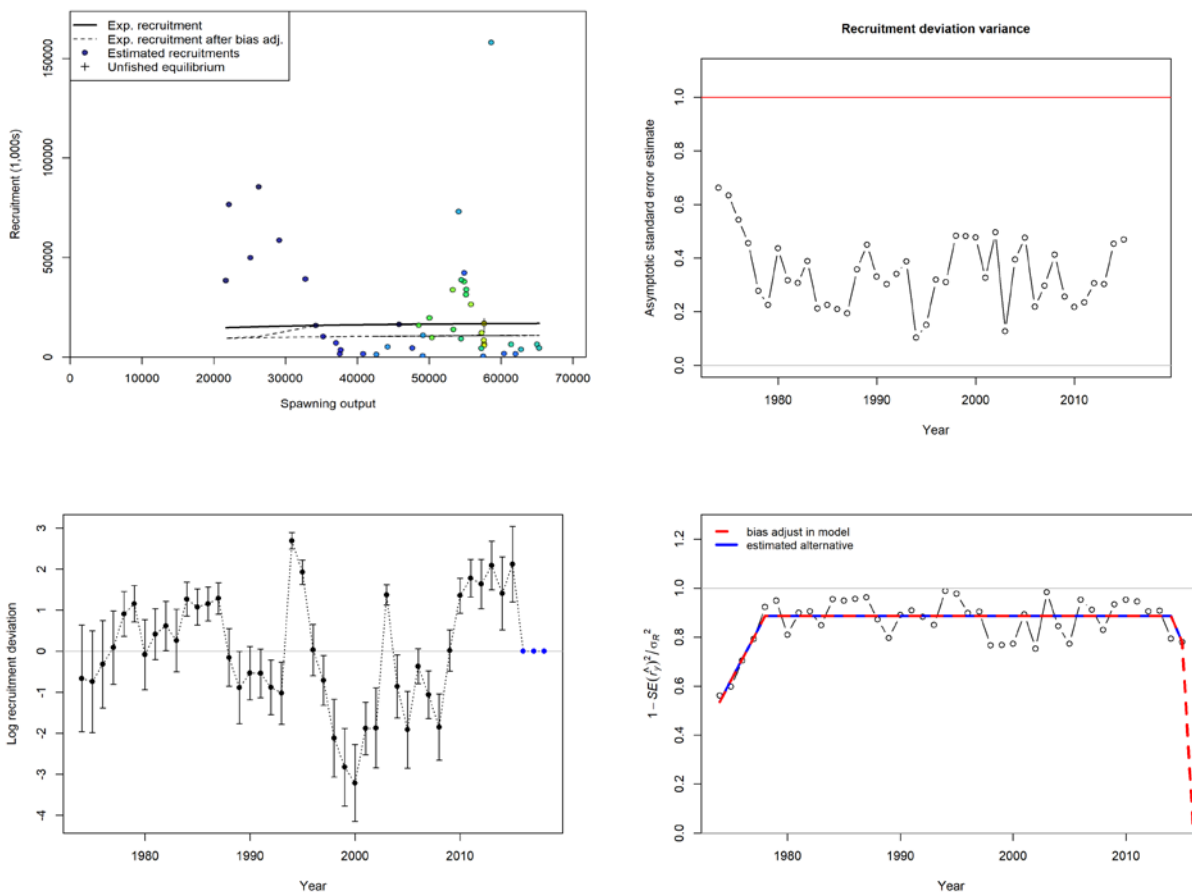


Figure A 9.3. Time series showing the stock recruitment curve, recruitment deviations and recruitment deviation variance check for blue grenadier.

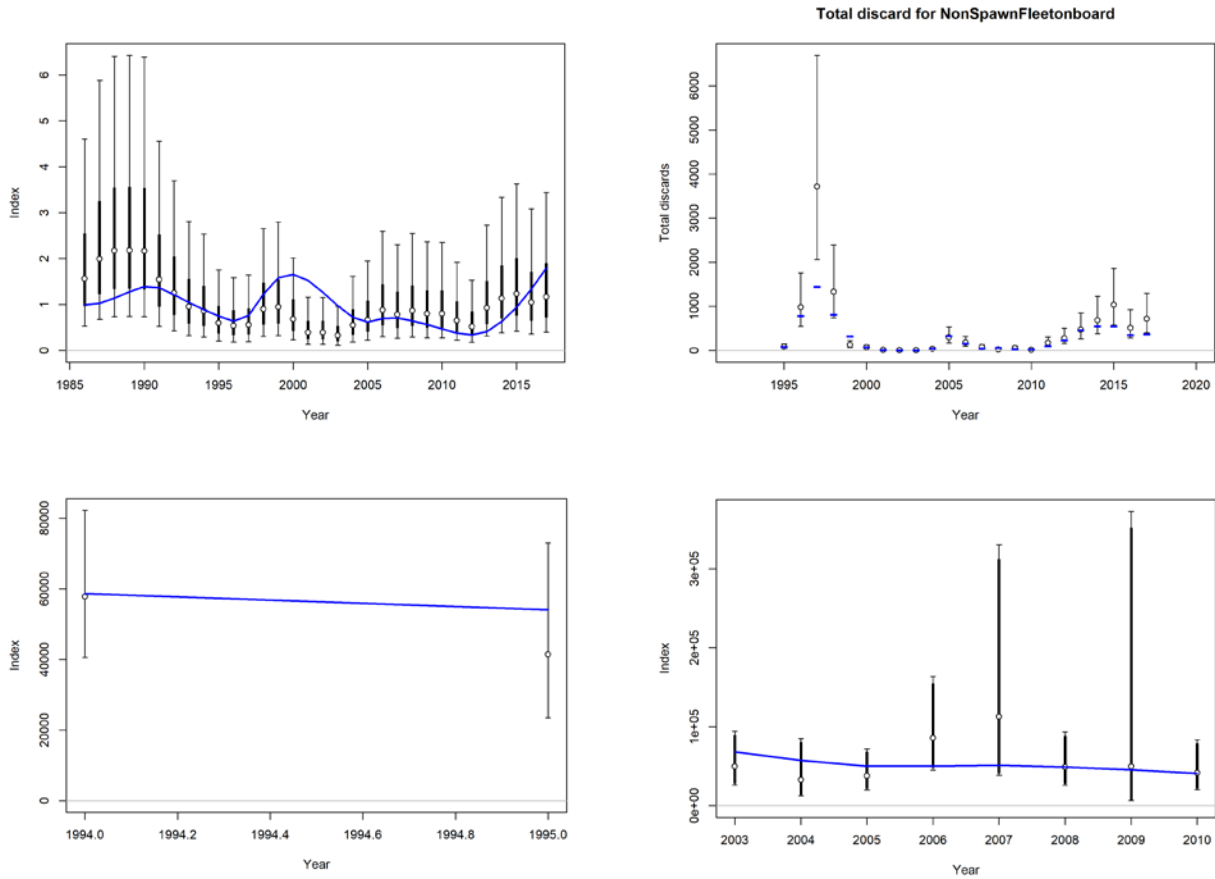


Figure A 9.4. Fits to the non-spawning CPUE index, discard mass, egg survey and acoustic survey.

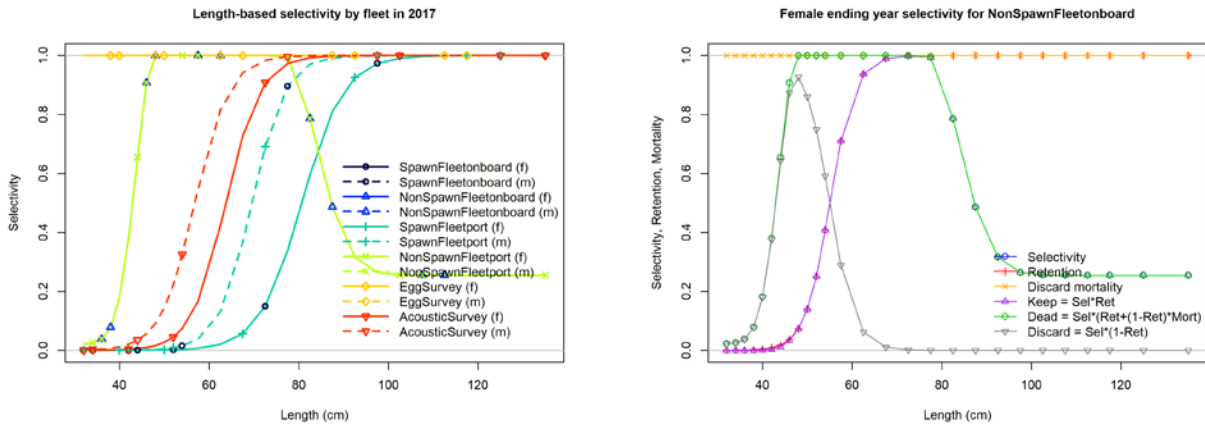


Figure A 9.5. Estimated selectivity for the spawning and on-spawning fleets using port and onboard samples and for males (m) and females (f) and the retention function.

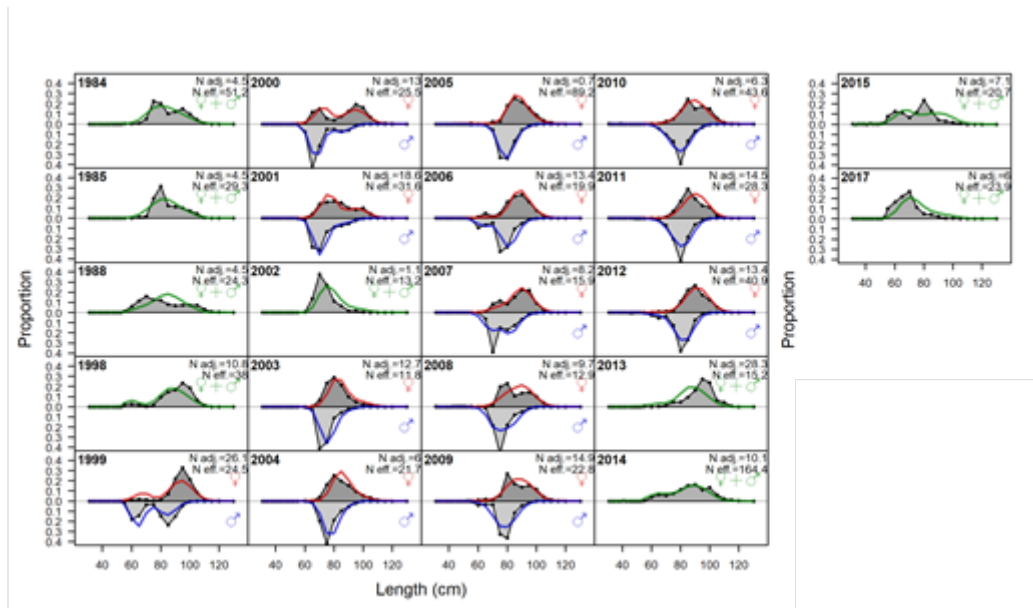


Figure A 9.6. Length composition fits: spawning fleet onboard retained.

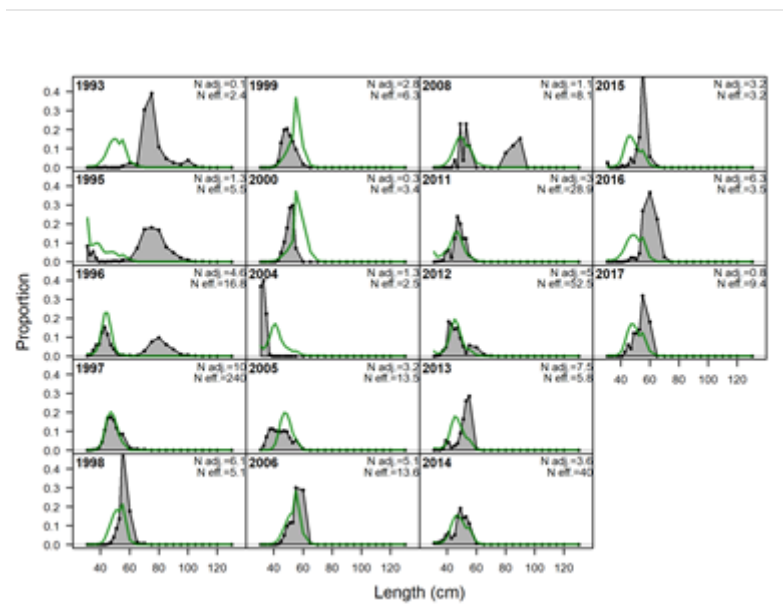


Figure A 9.7. Length composition fits: onboard non-spawning fleet discard.

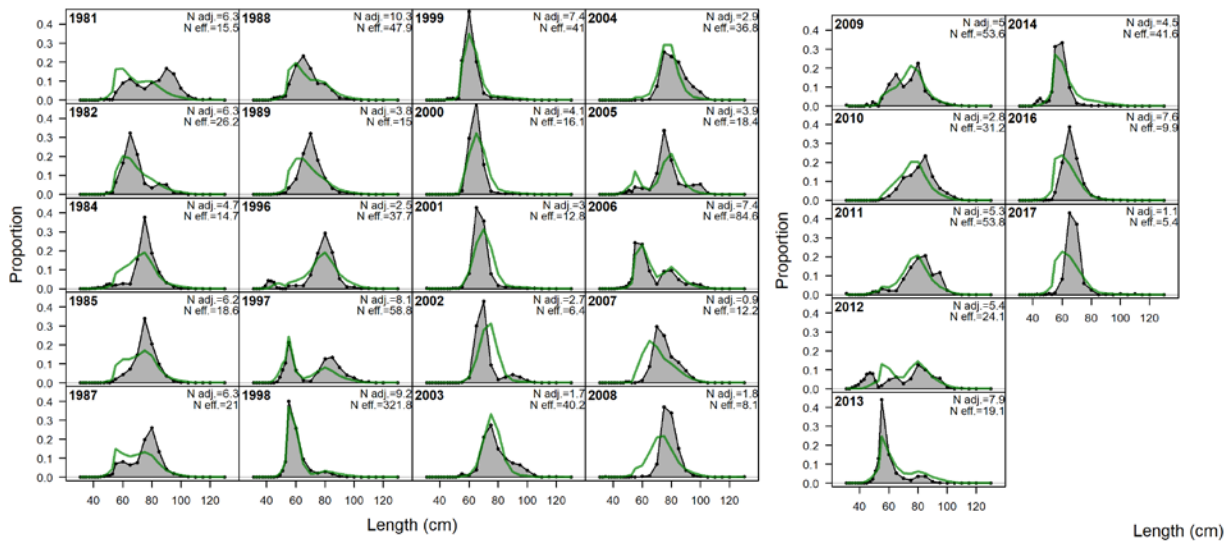


Figure A 9.8. Length composition fits: onboard non-spawning fleet retained.

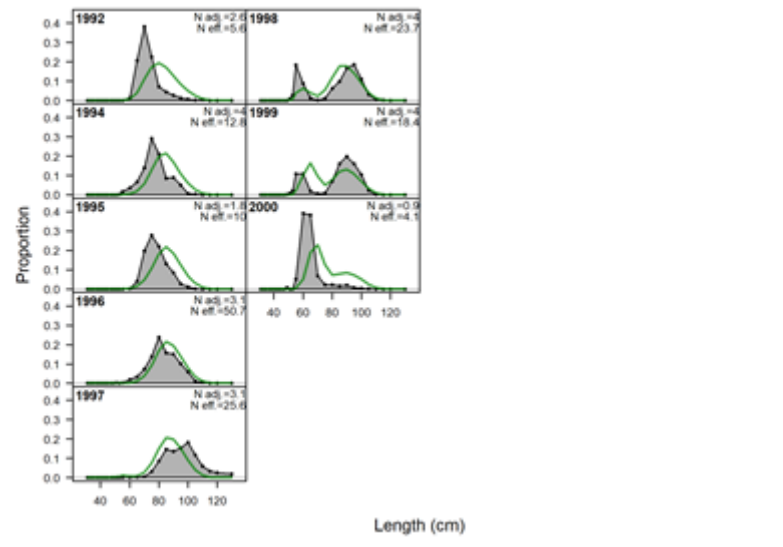


Figure A 9.9. Length composition fits: port spawning fleet retained.

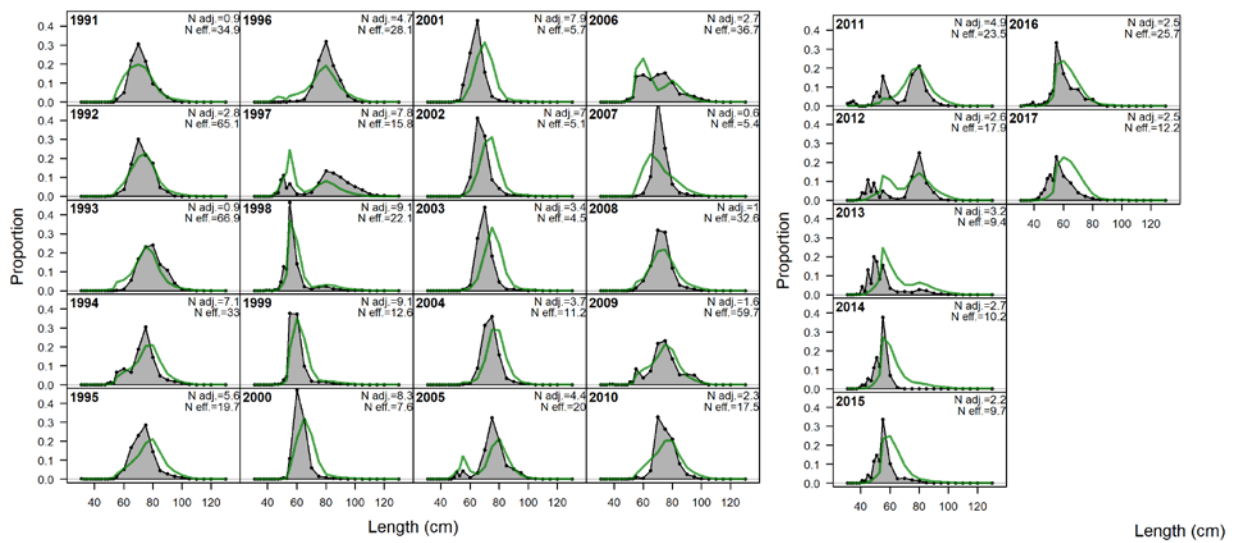


Figure A 9.10. Length composition fits: port non-spawning fleet retained.

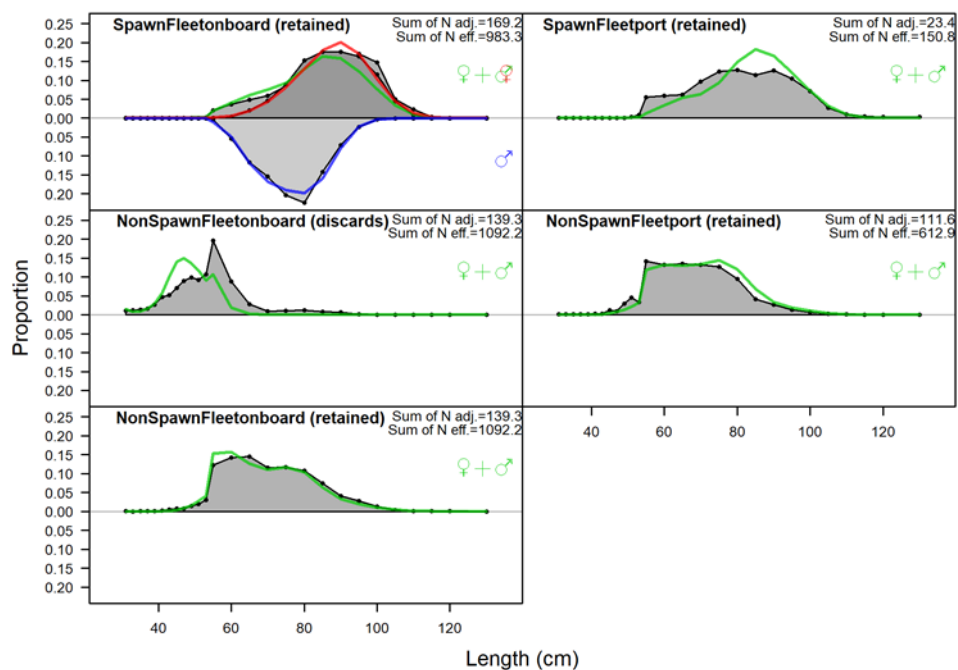


Figure A 9.11. Length composition fits aggregated across years.

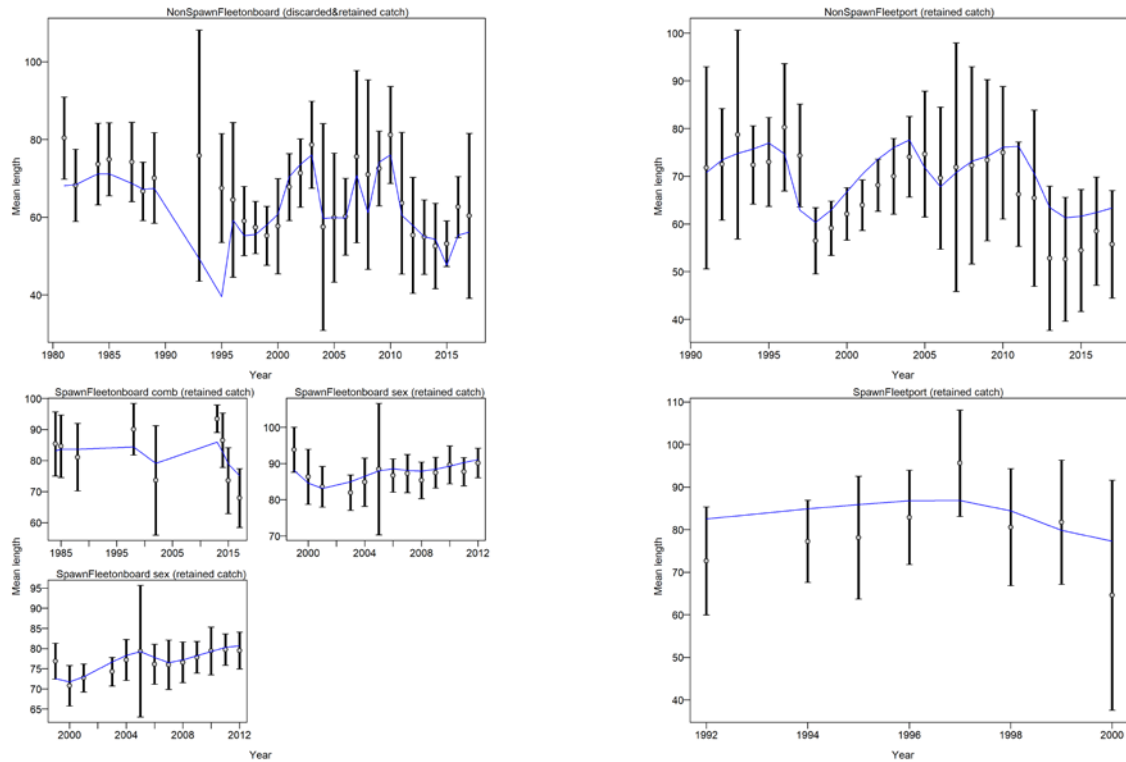


Figure A 9.12. Length composition fit diagnostics from tuning. Francis data weighting method TA1.8: thinner intervals (with capped ends) show result of further adjusting sample sizes based on suggested multiplier (with 95% interval) for length data.

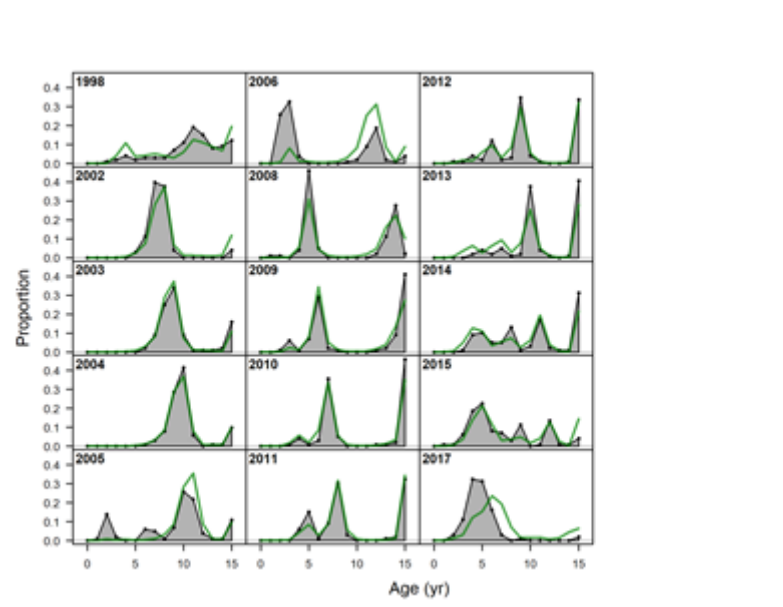


Figure A 9.13. Age composition fits: spawning fleet onboard retained.

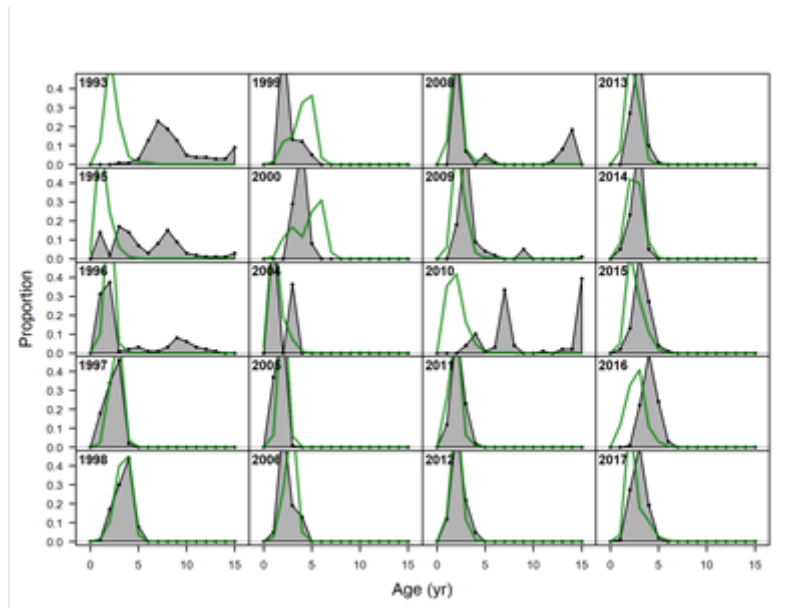


Figure A 9.14. Age composition fits: non-spawning fleet onboard discard.

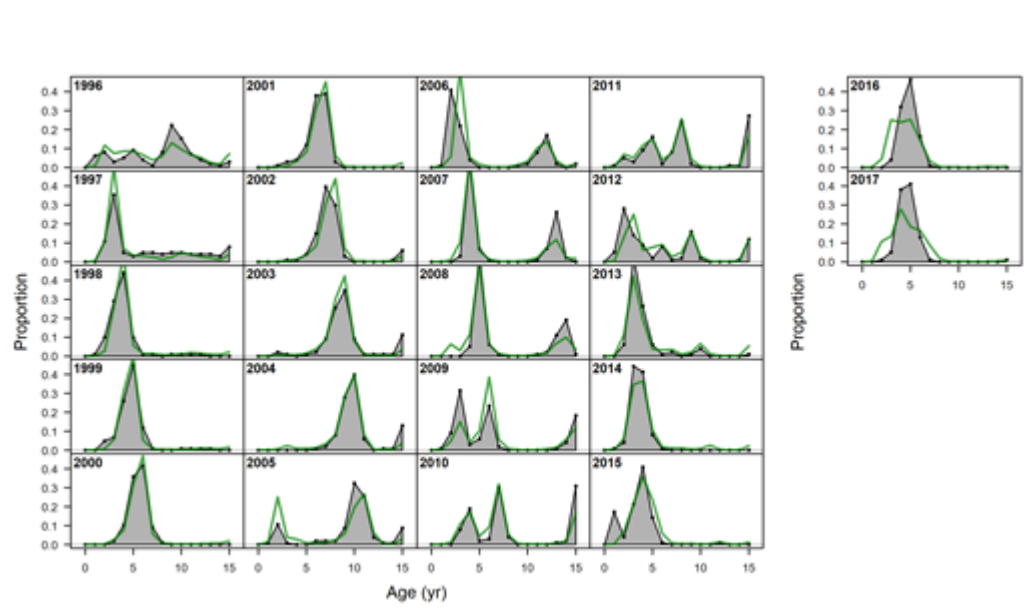


Figure A 9.15. Age composition fits: non-spawning fleet onboard retained.

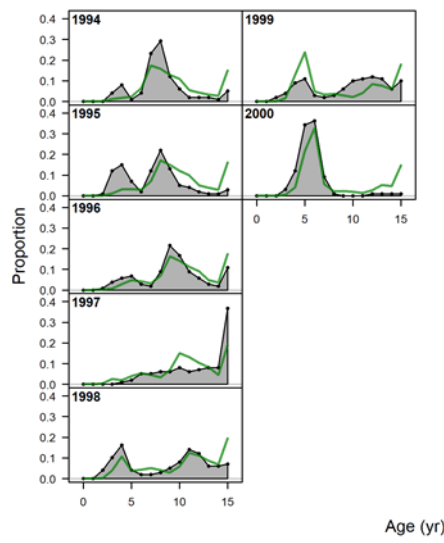


Figure A 9.16. Age composition fits: spawning fleet port retained.

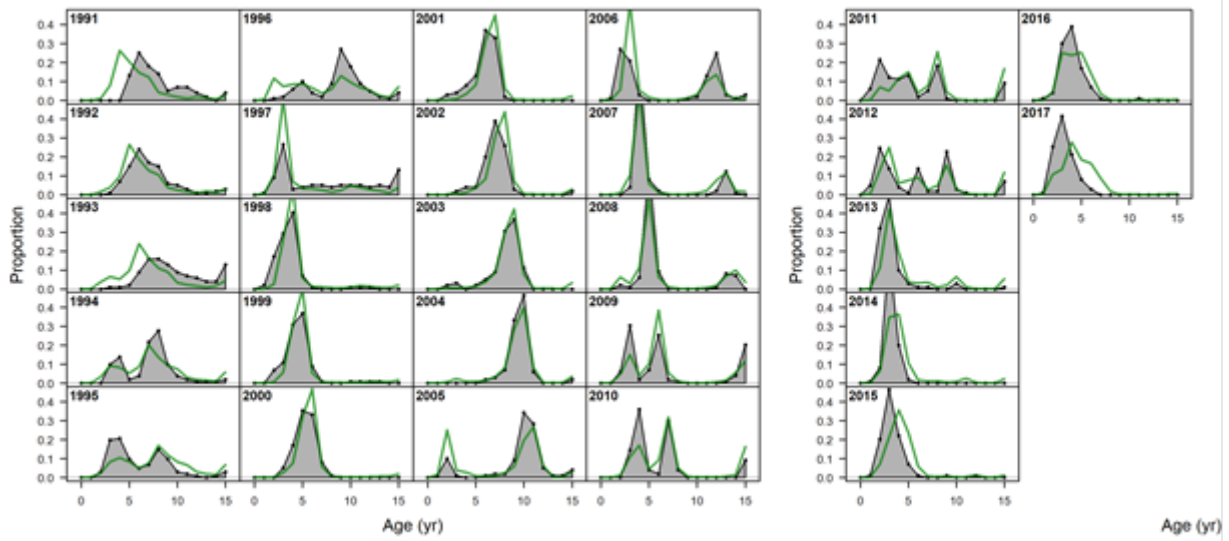


Figure A 9.17. Age composition fits: non-spawning fleet port retained.

A.2 Comparison between 2013 and 2018 assessment results

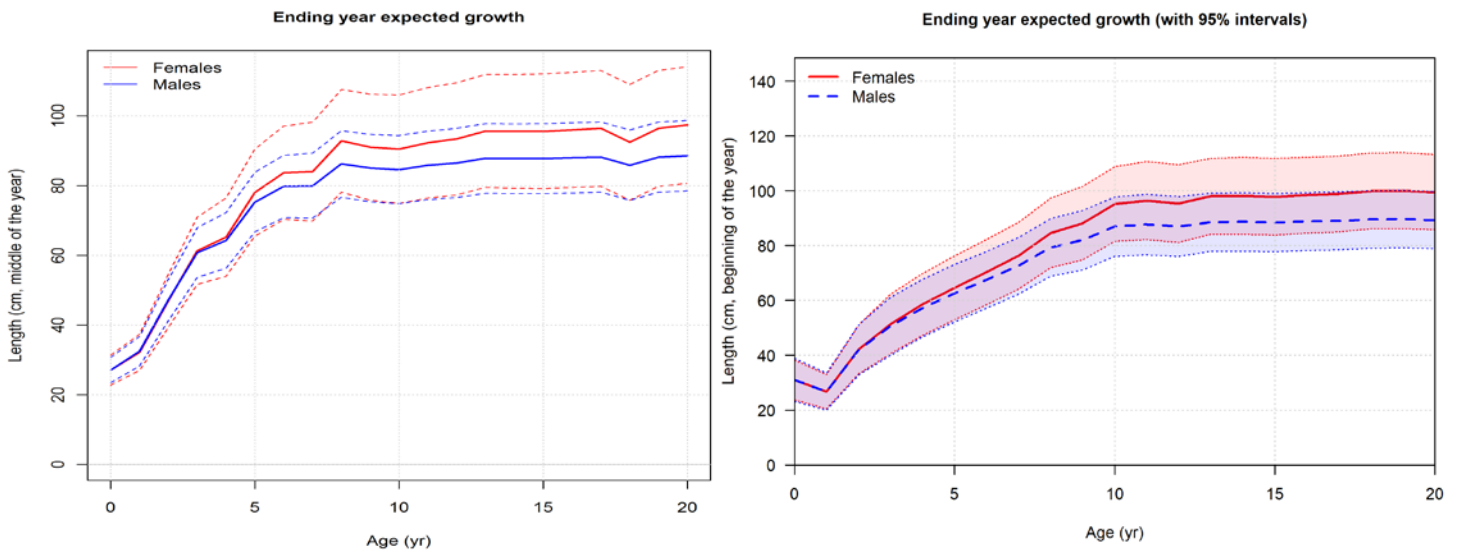


Figure A 9.18. Growth function for blue grenadier. 2013 (left) 2018 (right).

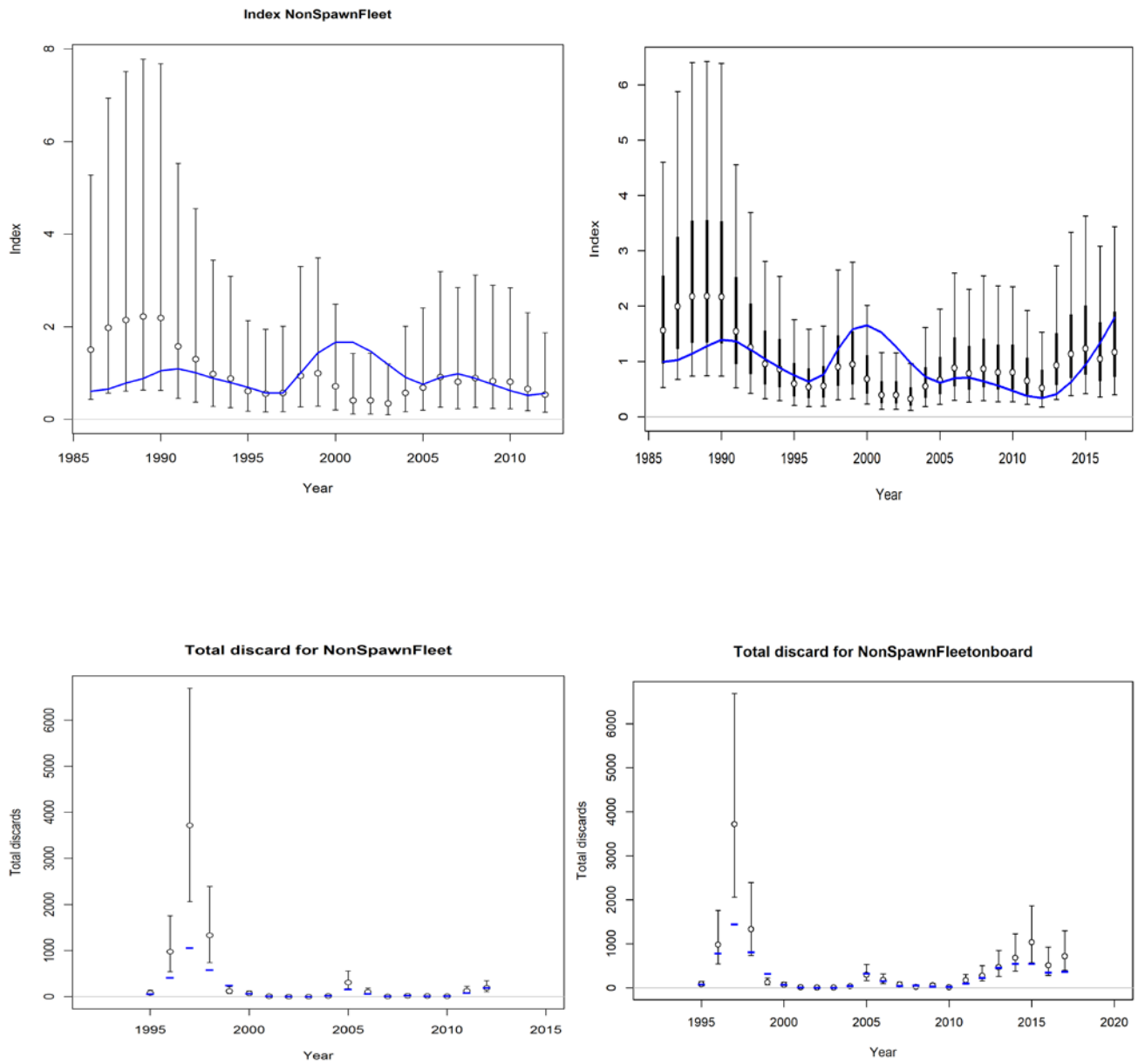


Figure A 9.19. Fits to the cpue and discard mass for blue grenadier. 2013 (left) 2018 (right).

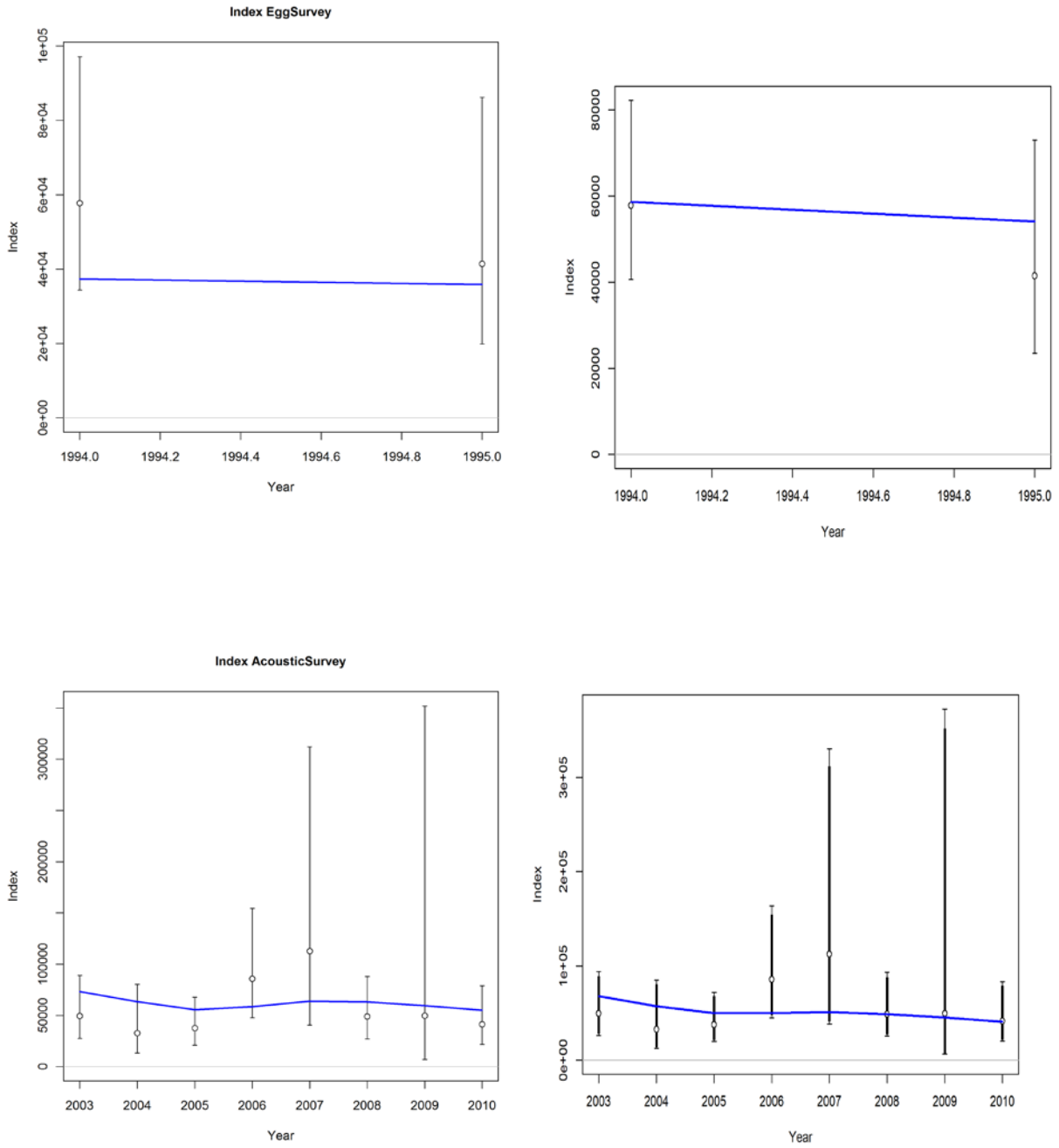


Figure A 9.20. Fits to the egg (top) and acoustic (bottom) surveys for blue grenadier. 2013 (left) 2018 (right).

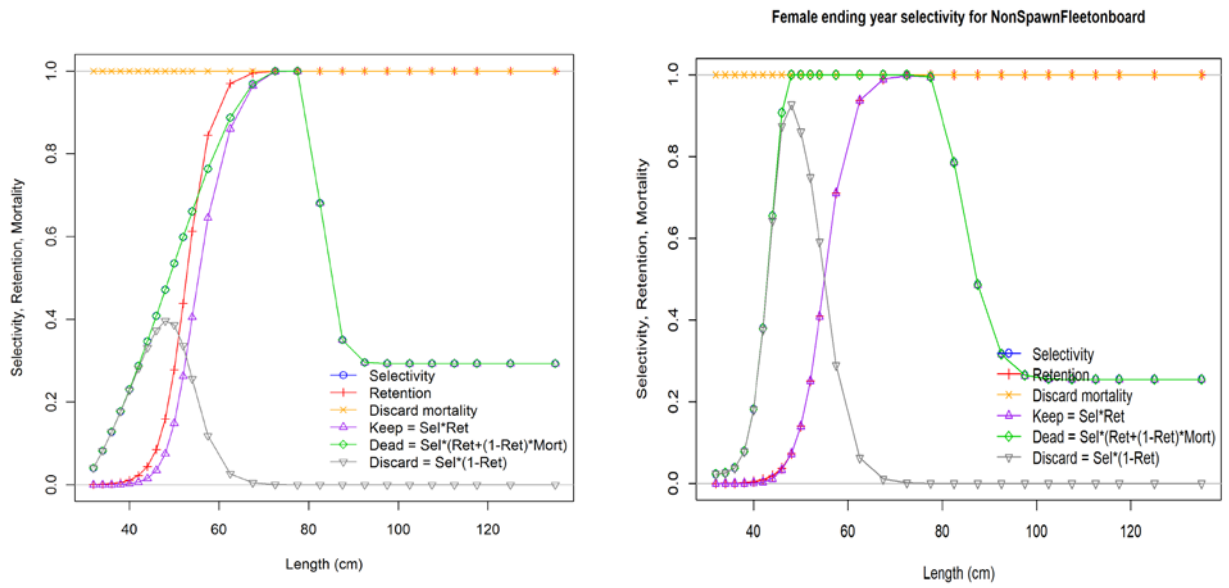


Figure A 9.21. The base case predicted selectivity-at-length for the non-spawning fleet (green) and the retention function for the non-spawning fleet (red/purple). The proportion discarded at length is grey. 2013 (left) 2018 (right).

10. Blue grenadier (*Macruronus novaezelandiae*) stock assessment based on data up to 2017 base case

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10.1 Executive Summary

This document presents the base case for an updated quantitative Tier 1 assessment of blue grenadier (*Macruronus novaezelandiae*) for presentation at the second SERAG meeting in 2018. The last full assessment was conducted during 2013 (Tuck, 2013). Relative to the 2013 assessment, the base case is updated by the inclusion of data to the end of 2017, which entails an additional five years of catch, discard, CPUE, length-composition and conditional age-at-age data and ageing error. This document describes the agreed base case from the September 2018 SERAG meeting and sensitivities to the base case.

The base case specifications agreed by the SlopeRAG in 2013 were generally maintained in the preliminary and the final base case presented here. The main differences are: separating length-composition into onboard- and port- collected components, assigning stage-1 weights to length-compositions by shots (onboard) and trips (port); and using the latest methods for assigning final weights to the various data sources and the extent of variation in recruitment. Differences between the preliminary base case (presented to the SERAG in September 2018) and the final base case include (a) the addition of the FIS abundance data for the non-Spawning area (and not the spawning area index), (b) the removal of discard lengths that were not believed to be representative from three early years and (c) the final assessment does not estimate recruitment for the last three years (instead of the last two years).

The estimated time series of recruitment under the base-case parameter set shows the typical episodic nature of blue grenadier recruitment, with strong year-classes in 1979, the mid-1980s, 1994, and 2003, with relatively low recruitment between these years. However, recent estimated recruitments are more stable than has been observed before. The trajectories of spawning biomass show increases and decreases in spawning biomass as strong cohorts move into and out of the spawning population. Results show reasonably good fits to the length-composition data, implied age compositions, egg survey and acoustic survey. The fit to the discard mass has improved compared to the 2013 assessment result. As has been noted in previous blue grenadier assessments, the fit to the standardized non-spawning catch-rate index is generally poor; the model is unable to fit to the high early catch rates and over-estimates catch rates during the early 2000s.

The estimated virgin female spawning biomass (B_0) from the base case assessment is 53,909t tonnes (SD 7,652t) and the projected 2019 spawning stock biomass will be 122% (SD 25.7%) of virgin female spawning biomass. Sensitivities are provided to test the influence of natural mortality estimation for males, excluding the port data, and removal of the cohort dependent growth, as was requested at the September 2018 SERAG meeting. Standard sensitivities are included to check how well the model fits if key parameters values are varied.

10.2 Introduction

An integrated analysis model, implemented in the generalized stock assessment software package, Stock Synthesis (SS) (Methot, 2011; Methot and Wetzel, 2013), was applied to the stock of blue grenadier in the Southern and Eastern Scalefish and Shark Fishery (SESSF), with data updated by the inclusion of data up to the end of the 2017 calendar year (length-composition and conditional age-at-length data; age reading-error matrices, standardized catch rate series; landings and discard catch weight) and information from acoustic surveys of spawning biomass (series from 2003-2010, pertaining to total spawning biomass), with an assumption of 2-times turnover on the spawning ground (Russell and Smith, 2006; Punt et al., 2015). The base-case egg survey estimates of female (only) spawning biomass for 1994 and 1995 are included, as is the FIS abundance series from the non-spawning area. The model fits directly to length-composition data (by sex where possible) and conditional age-at-length data by fleet. Retained length-composition data from port and onboard samples are separated (a change from the last assessment following current protocols).

The assessment model presented in 2011 (Tuck, Whitten and Punt 2011; Tuck 2011) was the first for blue grenadier to be implemented using SS. The 2013 assessment updated this assessment using SS-V3.22a (Tuck, 2013). Considerable changes to both the software and the tuning methods have occurred since the last assessment five years ago. As such, changes to key model outputs, such as the estimates of depletion and of the trajectory of spawning biomass, should be expected. The preliminary base case presented to SERAG in September 2018 (Castillo-Jordan and Tuck, 2018) illustrated the changes that have occurred since 2013 through changes to software, assessment practices and new data (bridging). Castillo-Jordan and Tuck (2018) also provided likelihood profiles of natural mortality, steepness and R_0 . The bridging analysis and likelihood profiles are not repeated here.

The use of SS allows for multiple fishing fleets and can fit simultaneously to several data sources and types of information. The population dynamics model, and the statistical approach used in the fitting of the model to the various types of data, is outlined fully in the SS user manual (Methot, 2005; 2011; Methot et al. 2018) and is not reproduced here. This document updates the assessment presented in 2013 and the preliminary assessment presented at SERAG in September 2018 (Castillo-Jordan and Tuck, 2018).

10.3 The Fishery

Blue grenadier are found from New South Wales around southern Australia to Western Australia, including the coast of Tasmania. Blue grenadier is a moderately long-lived species with a maximum age of about 25 years. Age at maturity is approximately 4 years for males and 5 years for females (length-at-50% maturity for females is 57cm and 64cm respectively) based upon 32,000 blue grenadier sampled between February 1999 and October 2001 (Russell and Smith, 2006). There is also evidence that availability to the gear on the spawning ground differs by sex, with a higher proportion of small males being caught than females. This is most likely due to the arrival of males on the spawning ground at a smaller size (and younger age) than females. This was also noted by Russell and Smith (2006) who state that “young males entered the fishery one year earlier than females” and is consistent with information for hoki (the same species) from New Zealand (Annala et al., 2003). Large fish arrive earlier in the spawning season than small fish. Spawning occurs predominantly off western Tasmania in winter (the peak spawning period based upon mean gonad somatic index (GSI) calculated by month was estimated to be between June and August according to Russell and Smith (2006)). There is some evidence that a high proportion of fish remain spawning in September. Variations in spawning period noted by Gunn et al. (1989) may occur due to inter-annual differences in the development of coastal

current patterns around Tasmania. Adults disperse following the spawning season and while fish are found throughout the south east region during the non-spawning season, their range is not well defined. Spawning fish have been caught off the east coast of Australia, and larvae from a likely eastern spawning area have been described by Bruce et al. (2001).

Blue grenadier are caught by demersal trawling. The global agreed TAC for the 2017/18 fishing season was 8,810 tonnes. The annual TACs are show in Table 10.2. There are two defined sub-fisheries: the spawning (Zone 40, months June, July and August) and non-spawning fisheries (all other months and zones).

10.4 Data

The assessment has been updated since the previous assessment (Tuck, 2013) by including recent length-composition and conditional age-at-length data from the spawning and non-spawning fisheries; updated standardized CPUE series (Sporcic and Haddon, 2018), the total mass landed and discarded, and updated age-reading error matrices. Acoustic estimates of spawning biomass (2003-2010) and estimates of the female spawning biomass in 1994 and 1995 from egg surveys (Bulman et al., 1999) are included, as is the FIS abundance estimates from the non-spawning area. Data were formulated by calendar year (i.e. 1 Jan to 31 Dec), as in previous models.

10.4.1 Catch data

10.4.1.1 Landings

The landings from the logbook data were used to apportion catches to the spawning and non-spawning fisheries (Table 10.1). The logbook landings have been adjusted upwards to match the CDR totals to take account of differences between logbook and landings data (multiple of 1.4 for the non-spawning fishery, based on 40% conversion from headed and gutted to whole, since 1986 and up to and including 1997 (reliable CDR data were available from 1998); 1.2 for the spawning fishery from 1986 up to and including 1996 (when factory vessels entered the spawning fishery) (D. Smith, pers. comm.). As stated by Thomson and He (2001), the factor is lower for the spawning fleet than the non-spawning fleet because some fish in the spawning fishery, landed headed and gutted, were recorded as being landed whole. These factors were chosen by the Blue Grenadier Assessment Group (BGAG) (Chesson and Staples (1995), as cited by Punt (1998)). The adjusted logbook catches were then scaled up to the SEF2 data (CDR). As historical CDR data were only available from 1992, the average scaling factor from 1992 to 1996 (1.07) was used to scale the data for years between 1986 and 1991. Note that in years 2008 to 2013 logbook data were greater than landings from the CDR. In these cases, the tonnage from the CDR was used as the total catch (AFMA, pers. comm. 2011). Table 10.2 lists the annual catches used in the assessment and the annual TAC. The annual logbook catches by sub-fishery and the adjustments made to determine the catches used in the assessment are shown in Table 10.1.

10.4.1.2 Discards

Discard rates were estimated from on-board data which gives the weight of the retained and discarded component of those shots that were monitored (Thomson and Klaer, 2011, Burch et al 2018). The discard values from 1995 to 2002 are based on estimates calculated from ISMP data by MAFRI and reported in He et al (1999) and Tuck, Smith and Talman (2004). The MAFRI estimates of discards were made accounting for differences in sampling and discard rates according to the ISMP zones. As agreed by Slope RAG (2011), since 2003 discard rates are estimated using the methods described in

Thomson and Klaer (2011). The Tier 1 discard estimates have been updated in 2018 to more closely match the discard calculations in Bergh et al (2008). These estimates use ratios of total discards to (retained + discard) catch on a per shot basis, rather than aggregated across a whole strata, which are then weighted up according to CDR landings within zone and season (N. Klaer, pers. comm.). Information in support of the historical values was not able to be obtained and further exploration of the methods and data used to estimate these values should be encouraged. The discard data are provided in Table 10.2. The discard data were assumed to have standard error (on the log-scale) of 0.3.

Discard rates for Tier 1 assessments are required by fishing fleet. This means that the discard estimates for TAC purposes used for Tier 3 and 4 assessments which are provided in the discard report (Burch et al, 2018) cannot be used in Tier 1 assessments. The discards from Burch et al. (2018) are produced using a set of rules to determine, for the entire quota fishery, whether sufficient data are available to make an annual fishery wide discard estimate. The discard rates calculated for and input to Tier 1 stock assessments are used to fit retention selectivity curves, so individual year values are not greatly influential on model estimated discard rates.

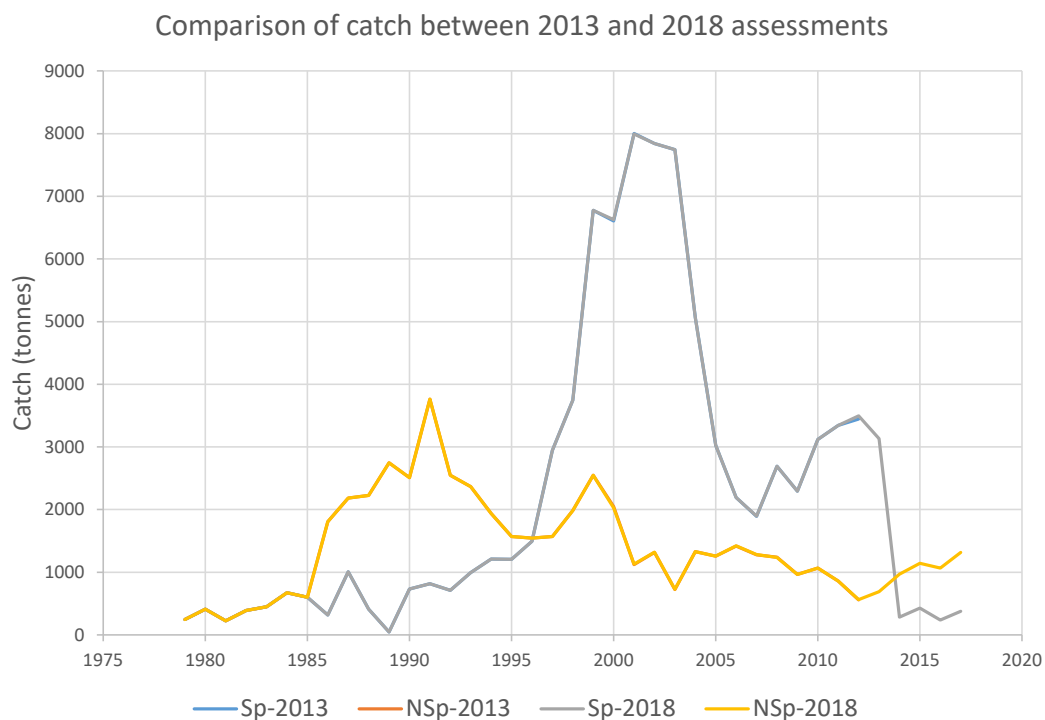


Figure 10.1. A comparison of total annual catches from the 2013 base case assessment and the updated catch used in the 2018 assessment for the spawning (Sp) and non-spawning (NSp) fisheries.

Table 10.1. Logbook and CDR landings for the spawning and non-spawning sub-fisheries by calendar year and adjustments made to account for logbooks being less than landings and incorrect reporting process code. Shaded CDR are historical landings values. ¹ average of CDR/logbook ratio from 1992 to 1996.

| Year | Logbook | | CDR | H&G Multiplier | | Adjusted Logbook | | Total | CDR / logbook | Catch for assessment | |
|------|----------|--------------|------|----------------|--------------|------------------|--------------|-------|-------------------|----------------------|--------------|
| | Spawning | Non-spawning | | Spawning | Non-spawning | Spawning | Non-spawning | | | Spawning | Non-spawning |
| 1979 | 245 | 245 | | 1 | 1 | 245 | 245 | 490 | 1.00 | 245 | 245 |
| 1980 | 410 | 410 | | 1 | 1 | 410 | 410 | 820 | 1.00 | 410 | 410 |
| 1981 | 225 | 225 | | 1 | 1 | 225 | 225 | 450 | 1.00 | 225 | 225 |
| 1982 | 390 | 390 | | 1 | 1 | 390 | 390 | 780 | 1.00 | 390 | 390 |
| 1983 | 450 | 450 | | 1 | 1 | 450 | 450 | 900 | 1.00 | 450 | 450 |
| 1984 | 675 | 675 | | 1 | 1 | 675 | 675 | 1350 | 1.00 | 675 | 675 |
| 1985 | 600 | 600 | | 1 | 1 | 600 | 600 | 1200 | 1.00 | 600 | 600 |
| 1986 | 246 | 1204 | | 1.2 | 1.4 | 295 | 1685 | 1981 | 1.07 | 317 | 1806 |
| 1987 | 782 | 1455 | | 1.2 | 1.4 | 939 | 2036 | 2975 | 1.07 | 1006 | 2183 |
| 1988 | 319 | 1485 | | 1.2 | 1.4 | 383 | 2079 | 2461 | 1.07 | 410 | 2228 |
| 1989 | 36 | 1829 | | 1.2 | 1.4 | 43 | 2560 | 2604 | 1.07 | 46 | 2745 |
| 1990 | 570 | 1671 | | 1.2 | 1.4 | 684 | 2340 | 3023 | 1.07 | 733 | 2508 |
| 1991 | 637 | 2508 | | 1.2 | 1.4 | 764 | 3511 | 4275 | 1.07 ¹ | 819 | 3764 |
| 1992 | 509 | 1565 | 3259 | 1.2 | 1.4 | 610 | 2191 | 2802 | 1.16 | 710 | 2549 |
| 1993 | 812 | 1659 | 3362 | 1.2 | 1.4 | 975 | 2323 | 3298 | 1.02 | 994 | 2368 |
| 1994 | 974 | 1338 | 3151 | 1.2 | 1.4 | 1169 | 1873 | 3042 | 1.04 | 1211 | 1940 |
| 1995 | 911 | 1017 | 2775 | 1.2 | 1.4 | 1093 | 1424 | 2517 | 1.10 | 1205 | 1570 |
| 1996 | 1200 | 1061 | 3040 | 1.2 | 1.4 | 1439 | 1485 | 2925 | 1.04 | 1496 | 1544 |
| 1997 | 2623 | 997 | 4516 | 1 | 1.4 | 2623 | 1396 | 4019 | 1.12 | 2947 | 1569 |
| 1998 | 2739 | 1452 | 5733 | 1 | 1 | 2739 | 1452 | 4191 | 1.37 | 3746 | 1986 |
| 1999 | 5460 | 2054 | 9324 | 1 | 1 | 5460 | 2054 | 7514 | 1.24 | 6775 | 2549 |
| 2000 | 5735 | 1755 | 8655 | 1 | 1 | 5735 | 1755 | 7490 | 1.16 | 6627 | 2028 |
| 2001 | 7309 | 1032 | 9124 | 1 | 1 | 7309 | 1032 | 8340 | 1.09 | 7995 | 1129 |
| 2002 | 6825 | 1148 | 9161 | 1 | 1 | 6825 | 1148 | 7973 | 1.15 | 7842 | 1319 |
| 2003 | 7239 | 679 | 8471 | 1 | 1 | 7239 | 679 | 7918 | 1.07 | 7745 | 726 |
| 2004 | 4647 | 1219 | 6392 | 1 | 1 | 4647 | 1219 | 5865 | 1.09 | 5064 | 1328 |
| 2005 | 2880 | 1199 | 4283 | 1 | 1 | 2880 | 1199 | 4079 | 1.05 | 3024 | 1259 |
| 2006 | 2058 | 1332 | 3614 | 1 | 1 | 2058 | 1332 | 3390 | 1.07 | 2193 | 1420 |
| 2007 | 1815 | 1228 | 3176 | 1 | 1 | 1815 | 1228 | 3044 | 1.04 | 1894 | 1282 |
| 2008 | 2838 | 1304 | 3931 | 1 | 1 | 2838 | 1304 | 4141 | 0.95 | 2693 | 1237 |
| 2009 | 2723 | 1145 | 3259 | 1 | 1 | 2723 | 1145 | 3868 | 0.84 | 2295 | 965 |
| 2010 | 3384 | 1158 | 4185 | 1 | 1 | 3384 | 1158 | 4541 | 0.92 | 3118 | 1067 |
| 2011 | 3554 | 914 | 4201 | 1 | 1 | 3554 | 914 | 4467 | 0.94 | 3342 | 859 |
| 2012 | 3838 | 620 | 4060 | 1 | 1 | 3838 | 620 | 4458 | 0.91 | 3495 | 565 |
| 2013 | 3443 | 759 | 3821 | 1 | 1 | 3443 | 759 | 4201 | 0.91 | 3131 | 690 |
| 2014 | 271 | 928 | 1251 | 1 | 1 | 271 | 928 | 1200 | 1.04 | 283 | 968 |
| 2015 | 393 | 1054 | 1570 | 1 | 1 | 393 | 1054 | 1447 | 1.08 | 426 | 1144 |
| 2016 | 216 | 968 | 1305 | 1 | 1 | 216 | 968 | 1184 | 1.10 | 238 | 1068 |

| | | | | | | | | | | | |
|------|-----|------|------|---|---|-----|------|------|------|-----|------|
| 2017 | 354 | 1237 | 1693 | 1 | 1 | 354 | 1237 | 1591 | 1.06 | 376 | 1316 |
|------|-----|------|------|---|---|-----|------|------|------|-----|------|

Table 10.2. Landed and discarded catches for the spawning and non-spawning sub-fisheries by calendar year. These estimates have been scaled up to the landings data. Standardised CPUE (Sporcic and Haddon, 2018) for the non-spawning sub-fisheries by calendar year are shown, along with the TAC. ¹ a voluntary industry reduction to 4,200 t was implemented in 2005. ² This was a 16 month TAC. ³ The TACs cover the fishing year 1 May to 30 April. In the table below, 2008 refers to 2008/09. ⁴ This is an estimate of retained catch based on the 2017/2018 TAC and relative split of catch between the spawning and non-spawning fisheries of 2017.

| Year | Spawning (t) | Non-spawning (t) | Discards (t) | TAC | CPUE |
|------|------------------|-------------------|--------------|-------------------|--------|
| 1979 | 245 | 245 | | | |
| 1980 | 410 | 410 | | | |
| 1981 | 225 | 225 | | | |
| 1982 | 390 | 390 | | | |
| 1983 | 450 | 450 | | | |
| 1984 | 675 | 675 | | | |
| 1985 | 600 | 600 | | | |
| 1986 | 317 | 1807 | | | 1.5611 |
| 1987 | 1006 | 2183 | | | 1.994 |
| 1988 | 410 | 2228 | | | 2.1709 |
| 1989 | 46 | 2745 | | | 2.1776 |
| 1990 | 733 | 2508 | | | 2.166 |
| 1991 | 819 | 3764 | | | 1.545 |
| 1992 | 710 | 2549 | | | 1.252 |
| 1993 | 994 | 2368 | | | 0.9511 |
| 1994 | 1211 | 1940 | | 10000 | 0.8586 |
| 1995 | 1205 | 1570 | 80 | 10000 | 0.5937 |
| 1996 | 1496 | 1544 | 975 | 10000 | 0.5361 |
| 1997 | 2947 | 1569 | 3716 | 10000 | 0.5574 |
| 1998 | 3746 | 1986 | 1329 | 10000 | 0.901 |
| 1999 | 6775 | 2549 | 123 | 10000 | 0.9466 |
| 2000 | 6627 | 2028 | 69 | 10000 | 0.6815 |
| 2001 | 7995 | 1129 | 10 | 10000 | 0.3927 |
| 2002 | 7842 | 1319 | 2 | 10000 | 0.391 |
| 2003 | 7745 | 726 | 8 | 9000 | 0.3258 |
| 2004 | 5064 | 1328 | 34 | 7000 | 0.5474 |
| 2005 | 3024 | 1259 | 294 | 5000 ¹ | 0.6594 |
| 2006 | 2193 | 1420 | 175 | 3730 | 0.8803 |
| 2007 | 1894 | 1282 | 72 | 4113 ² | 0.782 |
| 2008 | 2693 | 1237 | 18 | 4368 ³ | 0.8643 |
| 2009 | 2295 | 965 | 57 | 4700 ³ | 0.8004 |
| 2010 | 3118 | 1067 | 13 | 4700 ³ | 0.7975 |
| 2011 | 3342 | 859 | 169 | 4700 ³ | 0.6511 |
| 2012 | 3495 | 565 | 277 | 5208 ³ | 0.5187 |
| 2013 | 3131 | 690 | 469 | 5208 ³ | 0.9243 |
| 2014 | 283 | 968 | 680 | 6800 ³ | 1.1316 |
| 2015 | 426 | 1144 | 1032 | 8796 ³ | 1.2303 |
| 2016 | 238 | 1068 | 512 | 8810 ³ | 1.0448 |
| 2017 | 376 | 1316 | 718 | 8765 ³ | 1.1656 |
| 2018 | 378 ⁴ | 1323 ⁴ | | 8810 ³ | |

10.4.2 Catch rates

Sporcic and Haddon (2018) provide the updated standardised catch rate series for the non-spawning fishery of blue grenadier (Table 10.2; Figure 10.2). The catch rate generally follows the fluctuations of stock size driven by large, but sporadic, recruitments. The standard deviation of log-CPUE is assumed to be 0.25 (value equal to the standard error from a loess fit), but an extra variance component is estimated for the CPUE index during the tuning process.

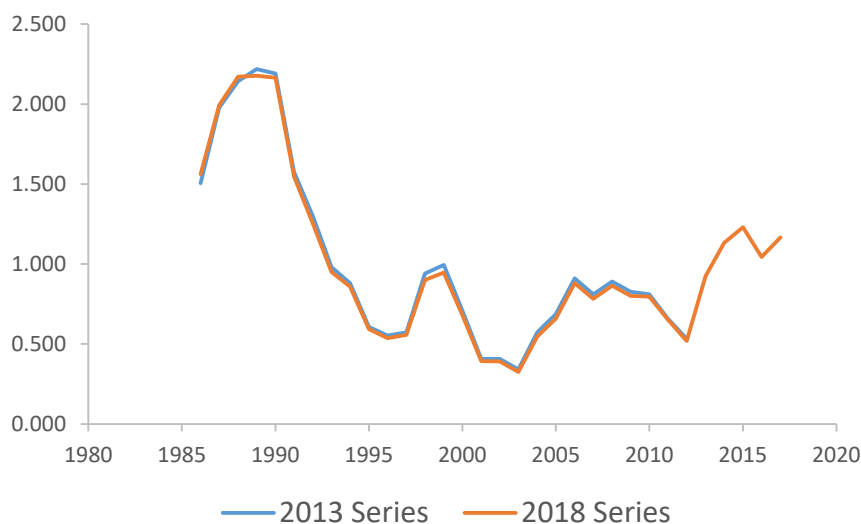


Figure 10.2. A comparison of the annual catch rates series for blue grenadier between the 2013 assessment (2013 Series) and 2018 (2018 Series).

10.4.3 Length-composition and age data

Length and age data are included in the assessment as length-composition data and conditional age-at-length data by fleet and sex (the latter when available). Age-composition data are included in diagnostic plots but are not used directly when estimating the parameters of the population dynamics model. On-board and port length-compositions, when available, are used separately. This is a change in data protocol from the last assessment, where lengths from port and onboard measurements were combined. Prior to 2013, only port samples had been used to create the length-compositions. Plots of the observed length and age data are shown in later figures, with the corresponding model predicted values.

There had to be at least 100 measured fish for a retained and/or discard onboard and port length-composition data to be included in the assessment. For onboard samples, numbers of shots were used as the sampling unit (i.e. the stage-1 weights; Francis, 2011), with a cap of 200. For port samples, numbers of trips were used as the sampling unit, with a cap of 100. The number of fish measured is not used as the sample size because the appropriate sample size for length-composition data is probably more closely related to the number of shots (onboard) or trips (port) sampled, rather than the number of fish measured (Table 10.3).

Table 10.3. The years for which length data were available for the sub-fleets (spawning onboard = 1; spawning port = 3; non-spawning onboard = 2; non-spawning port = 4), sex (0 = no gender specified; female =1; male =2), partition (part: discard = 1; retained = 2). N is the number of shots (onboard) or trips (port). Red length data were excluded due to low sample sizes. ¹ the average number of fish from years 1984 and 1988. ² as no shot data were available, these estimates were based upon the average number of fish per shot for un-sexed fish for Fleet 1 (84.4). ³ the average number of fish from years 1984 and 1987-89. ⁴ as no shot data were available, these estimates were based upon the average number of fish per shot for Fleet 2 (40.7). ⁵ the average of 1980s samples, as no fish numbers or shot data were available. ⁶ these years of discard lengths were removed due to spurious numbers of large fish.

| Year | Nfish | Fleet | Sex | Part | N |
|------|--------------------|-------|-----|------|-----------------|
| 1984 | 1,046 | 1 | 0 | 2 | 12 ² |
| 1985 | 1,090 ¹ | 1 | 0 | 2 | 12 ² |
| 1988 | 1,133 | 1 | 0 | 2 | 12 ² |
| 1998 | 1,948 | 1 | 0 | 2 | 29 |
| 1999 | 4,147 | 1 | 1 | 2 | 49 |
| 1999 | 5,929 | 1 | 2 | 2 | 70 |
| 2000 | 2,672 | 1 | 1 | 2 | 32 |
| 2000 | 2,956 | 1 | 2 | 2 | 35 |
| 2001 | 3,620 | 1 | 1 | 2 | 43 |
| 2001 | 4,256 | 1 | 2 | 2 | 50 |
| 2002 | 760 | 1 | 0 | 2 | 3 |
| 2003 | 2,700 | 1 | 1 | 2 | 32 |
| 2003 | 2,853 | 1 | 2 | 2 | 34 |
| 2004 | 1,307 | 1 | 1 | 2 | 15 |
| 2004 | 1,370 | 1 | 2 | 2 | 16 |
| 2005 | 198 | 1 | 1 | 2 | 2 |
| 2005 | 141 | 1 | 2 | 2 | 2 |
| 2006 | 3,184 | 1 | 1 | 2 | 38 |
| 2006 | 3,081 | 1 | 2 | 2 | 36 |
| 2007 | 2,957 | 1 | 1 | 2 | 35 |
| 2007 | 1,897 | 1 | 2 | 2 | 22 |
| 2008 | 3,073 | 1 | 1 | 2 | 36 |
| 2008 | 2,177 | 1 | 2 | 2 | 26 |
| 2009 | 3,868 | 1 | 1 | 2 | 46 |
| 2009 | 3,374 | 1 | 2 | 2 | 40 |
| 2010 | 2,488 | 1 | 1 | 2 | 29 |
| 2010 | 1,453 | 1 | 2 | 2 | 17 |
| 2011 | 4,207 | 1 | 1 | 2 | 50 |
| 2011 | 3,266 | 1 | 2 | 2 | 39 |
| 2012 | 3,939 | 1 | 1 | 2 | 47 |
| 2012 | 3,060 | 1 | 2 | 2 | 36 |
| 2013 | 6,371 | 1 | 0 | 2 | 76 |
| 2014 | 927 | 1 | 0 | 2 | 27 |
| 2015 | 1,861 | 1 | 0 | 2 | 19 |
| 2017 | 1,020 | 1 | 0 | 2 | 16 |

| Year | Nfish | Fleet | Sex | Part | N |
|-------------------|--------------------|-------|-----|------|------------------|
| 1993 ⁶ | 207 | 2 | 0 | 1 | 2 |
| 1995 ⁶ | 2,216 | 2 | 0 | 1 | 21 |
| 1996 ⁶ | 5,225 | 2 | 0 | 1 | 73 |
| 1997 | 6,504 | 2 | 0 | 1 | 159 |
| 1998 | 2,212 | 2 | 0 | 1 | 97 |
| 1999 | 940 | 2 | 0 | 1 | 45 |
| 2000 | 132 | 2 | 0 | 1 | 4 |
| 2004 | 1,077 | 2 | 0 | 1 | 21 |
| 2005 | 5,139 | 2 | 0 | 1 | 51 |
| 2006 | 1,225 | 2 | 0 | 1 | 81 |
| 2007 | 16 | 2 | 0 | 1 | 3 |
| 2008 | 106 | 2 | 0 | 1 | 17 |
| 2009 | 97 | 2 | 0 | 1 | 10 |
| 2010 | 16 | 2 | 0 | 1 | 2 |
| 2011 | 792 | 2 | 0 | 1 | 47 |
| 2012 | 1,261 | 2 | 0 | 1 | 80 |
| 2013 | 1,450 | 2 | 0 | 1 | 119 |
| 2014 | 864 | 2 | 0 | 1 | 57 |
| 2015 | 500 | 2 | 0 | 1 | 51 |
| 2016 | 1,323 | 2 | 0 | 1 | 100 |
| 2017 | 531 | 2 | 0 | 1 | 12 |
| 1981 | NA | 2 | 0 | 2 | 100 ⁵ |
| 1982 | NA | 2 | 0 | 2 | 100 ⁵ |
| 1984 | 3,035 | 2 | 0 | 2 | 75 ⁴ |
| 1985 | 4,046 ³ | 2 | 0 | 2 | 99 ⁴ |
| 1987 | 4,063 | 2 | 0 | 2 | 100 ⁴ |
| 1988 | 6,660 | 2 | 0 | 2 | 164 ⁴ |
| 1989 | 2,424 | 2 | 0 | 2 | 60 ⁴ |
| 1996 | 829 | 2 | 0 | 2 | 40 |
| 1997 | 2,501 | 2 | 0 | 2 | 128 |
| 1998 | 7,771 | 2 | 0 | 2 | 146 |
| 1999 | 8,768 | 2 | 0 | 2 | 117 |
| 2000 | 8,036 | 2 | 0 | 2 | 65 |
| 2001 | 6,293 | 2 | 0 | 2 | 48 |
| 2002 | 5,325 | 2 | 0 | 2 | 43 |
| 2003 | 2,558 | 2 | 0 | 2 | 27 |
| 2004 | 5,499 | 2 | 0 | 2 | 46 |
| 2005 | 5,698 | 2 | 0 | 2 | 62 |
| 2006 | 6,098 | 2 | 0 | 2 | 117 |
| 2007 | 219 | 2 | 0 | 2 | 14 |
| 2008 | 575 | 2 | 0 | 2 | 29 |
| 2009 | 1,944 | 2 | 0 | 2 | 80 |
| 2010 | 1,801 | 2 | 0 | 2 | 45 |
| 2011 | 1,643 | 2 | 0 | 2 | 84 |
| 2012 | 1,707 | 2 | 0 | 2 | 85 |
| 2013 | 1,785 | 2 | 0 | 2 | 125 |
| 2014 | 1,358 | 2 | 0 | 2 | 72 |
| 2015 | 1,525 | 2 | 0 | 2 | 79 |
| 2016 | 2,822 | 2 | 0 | 2 | 121 |
| 2017 | 951 | 2 | 0 | 2 | 17 |

| Year | Nfish | Fleet | Sex | Part | N |
|------|--------|-------|-----|------|-----|
| 1992 | 774 | 3 | 0 | 2 | 6 |
| 1994 | 1,038 | 3 | 0 | 2 | 9 |
| 1995 | 465 | 3 | 0 | 2 | 4 |
| 1996 | 927 | 3 | 0 | 2 | 7 |
| 1997 | 851 | 3 | 0 | 2 | 7 |
| 1998 | 1,648 | 3 | 0 | 2 | 9 |
| 1999 | 1,079 | 3 | 0 | 2 | 9 |
| 2000 | 339 | 3 | 0 | 2 | 2 |
| 2014 | 82 | 3 | 0 | 2 | 1 |
| 2016 | 74 | 3 | 0 | 2 | 1 |
| 1991 | 927 | 4 | 0 | 2 | 10 |
| 1992 | 3,832 | 4 | 0 | 2 | 31 |
| 1993 | 1,487 | 4 | 0 | 2 | 10 |
| 1994 | 8,604 | 4 | 0 | 2 | 78 |
| 1995 | 6,938 | 4 | 0 | 2 | 61 |
| 1996 | 5,397 | 4 | 0 | 2 | 51 |
| 1997 | 11,191 | 4 | 0 | 2 | 85 |
| 1998 | 16,234 | 4 | 0 | 2 | 100 |
| 1999 | 13,286 | 4 | 0 | 2 | 100 |
| 2000 | 13,613 | 4 | 0 | 2 | 91 |
| 2001 | 11,959 | 4 | 0 | 2 | 87 |
| 2002 | 9,416 | 4 | 0 | 2 | 77 |
| 2003 | 5,023 | 4 | 0 | 2 | 37 |
| 2004 | 4,392 | 4 | 0 | 2 | 41 |
| 2005 | 6,310 | 4 | 0 | 2 | 48 |
| 2006 | 2,874 | 4 | 0 | 2 | 30 |
| 2007 | 809 | 4 | 0 | 2 | 7 |
| 2008 | 1,320 | 4 | 0 | 2 | 11 |
| 2009 | 1,035 | 4 | 0 | 2 | 18 |
| 2010 | 698 | 4 | 0 | 2 | 25 |
| 2011 | 1,678 | 4 | 0 | 2 | 54 |
| 2012 | 999 | 4 | 0 | 2 | 29 |
| 2013 | 1,457 | 4 | 0 | 2 | 35 |
| 2014 | 1,611 | 4 | 0 | 2 | 30 |
| 2015 | 1,799 | 4 | 0 | 2 | 24 |
| 2016 | 1,790 | 4 | 0 | 2 | 27 |
| 2017 | 1,808 | 4 | 0 | 2 | 27 |

10.4.4 Acoustic survey estimates

Estimates of spawning biomass for 2003-2010 are provided in Ryan and Kloser (2012). There are no acoustic estimates since 2010. Table 10.4 shows the estimates of spawning biomass with their corresponding CV's used in the assessment. Sampling cv's less than 0.3 were increased to 0.3 to account for process error. Low sampling cvs (of 0.19 for example) were considered too low for an acoustic survey and a minimum of 0.3 should be used to reflect the total uncertainty (D. Smith, pers comm., Tuck et al., 2004; Slope RAG 2011). Of 22 acoustic cv's used for hoki in New Zealand, none are lower than 0.3 (Francis, 2009). It is assumed that the spawning ground experiences a turnover rate of 2 (i.e. for the model applied here, the spawning biomass estimates are doubled) (Russell and Smith, 2006; Punt et al., 2015). The acoustic survey selectivity is matched to the maturity ogive, as it is assumed the acoustic survey observes mature fish on the spawning ground.

Table 10.4. The estimated biomass (tonnes) of blue grenadier on the spawning grounds in years 2003 to 2010 (Ryan and Kloser, 2012).

| | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 |
|---------------------------------|--------|--------|--------|--------|--------|--------|--------|--------|
| biomass (t) | 24,690 | 16,295 | 18,852 | 42,882 | 56,330 | 24,450 | 24,787 | 20,622 |
| c.v. for assessment model | 0.30 | 0.46 | 0.30 | 0.30 | 0.52 | 0.30 | 1 | 0.33 |
| Sampling cv | 0.16 | 0.46 | 0.14 | 0.14 | 0.52 | 0.22 | 1 | 0.33 |

10.4.5 Egg survey estimates

Egg survey estimates of female spawning biomass are available for 1994 and 1995 (Bulman et al., 1999). The egg-estimates (CV) for 1994 and 1995 respectively are: 57,772 (0.18) and 41,409 (0.29) tonnes. For the analysis considered here, the base-case egg estimates were used.

10.4.6 The Fishery Independent Survey (FIS)

Abundance indices for blue grenadier for the FIS surveys conducted between 2008 and 2016 are provided in Table 10.5 (Knuckey et al., 2017; J. Day, pers comm.). The length-composition data from the FIS are shown in Figure 10.3. In the base case the FIS selectivity is mirrored to the non-spawning trawl fleet selectivity. Ideally the lengths would be used to independently estimate a FIS selectivity due to potential differences between survey selectivity and the non-spawning fleet selectivity. However, until there is an agreed set of weighted lengths, the length data have not been included in the base case model and should be considered as a sensitivity in subsequent reports.

Table 10.5. FIS-derived abundance indices for blue grenadier with corresponding coefficient of variation (CV).

| | 2008 | 2010 | 2012 | 2014 | 2016 |
|-------------------------|-------|-------|-------|-------|--------|
| Blue grenadier (all) | 15.83 | 3.38 | 10.75 | 19.65 | 58.20 |
| CV | 0.30 | 0.28 | 0.23 | 0.21 | 0.23 |
| Spawning | 65.06 | 17.97 | 15.12 | 44.52 | 211.29 |
| CV | 0.59 | 0.35 | 0.34 | 0.32 | 0.26 |
| Non-spawning | 30.26 | 9.25 | 10.57 | 50.26 | 10.39 |
| CV | 0.57 | 2.31 | 0.93 | 2.19 | 0.34 |

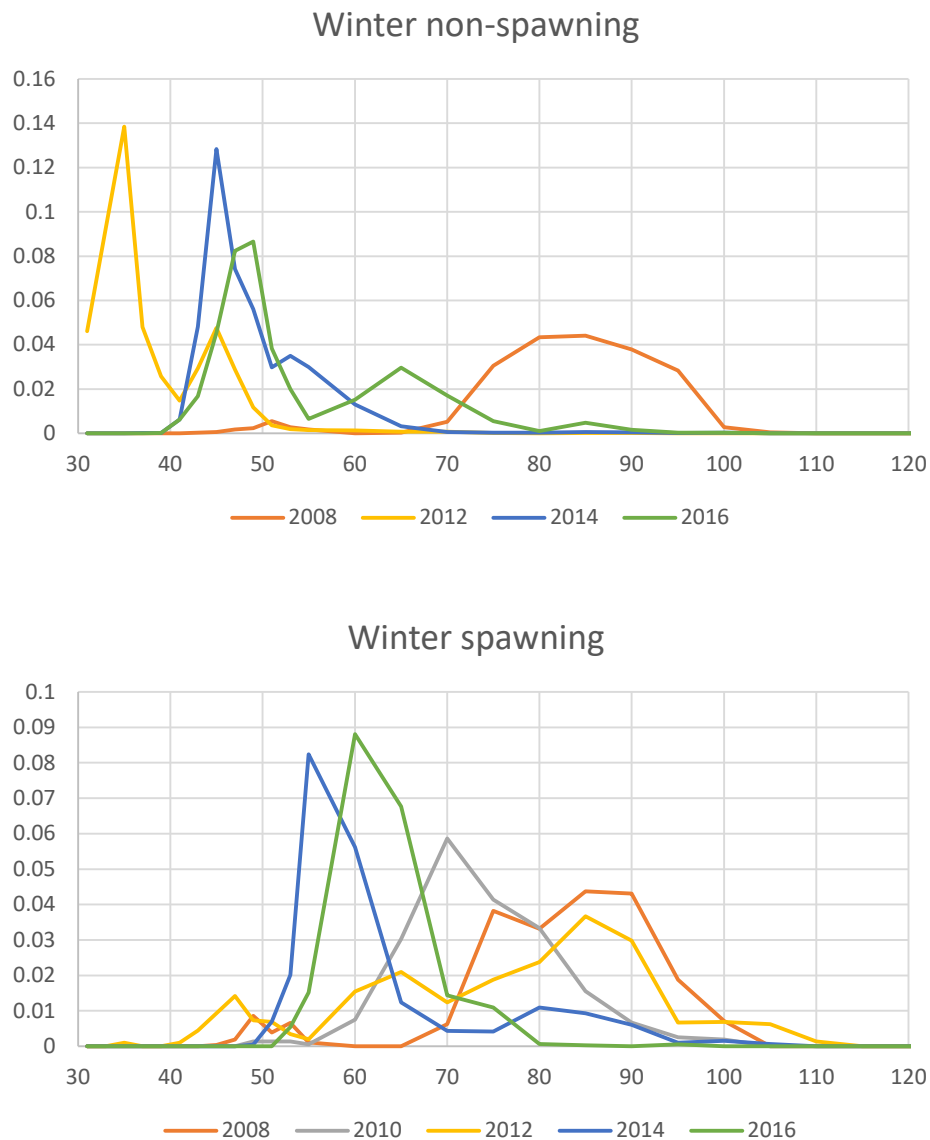


Figure 10.3. The length-compositions for blue grenadier from the FIS from the winter non-spawning (top) and spawning area (bottom) surveys.

10.4.7 Biological parameters and stock structure assumptions

The assessment assumes that the proportion of females that spawn in each year is 0.84 and a length at 50% maturity of 63.7cm for females (Russel and Smith, 2006). The female maturity ogive is shown in Figure 10.4.

The length weight-relationship for males and females was estimated from spawning fishery data over years 1999 to 2008 (Figure 10.4). Natural mortality for females is estimated when fitting the model and male natural mortality is assumed to be 20% greater than the value for females based upon assumptions made for hoki in New Zealand (McAllister et al., 1994). A sensitivity is conducted where male natural mortality is estimated, in line with current practice for hoki.

Francis (2009) reviews the values of steepness used in New Zealand hoki assessments, where a value of $h=0.9$ had been used since 1994. This value of steepness was derived from work of Punt et al. (1994) using 45 stocks of gadiform species (0.9 is the median). Following an analysis of the profile likelihood, the effect of steepness on the 2007 assessment and additional information of Myers et al. (1999; 2002) beyond that used by Punt et al. (1994), Francis (2009) concludes that steepness should be reduced to $h=0.75$. This value of steepness was assumed in the previous blue grenadier assessments in 2011 and 2013 (Tuck, 2011; 2013) and in this assessment.

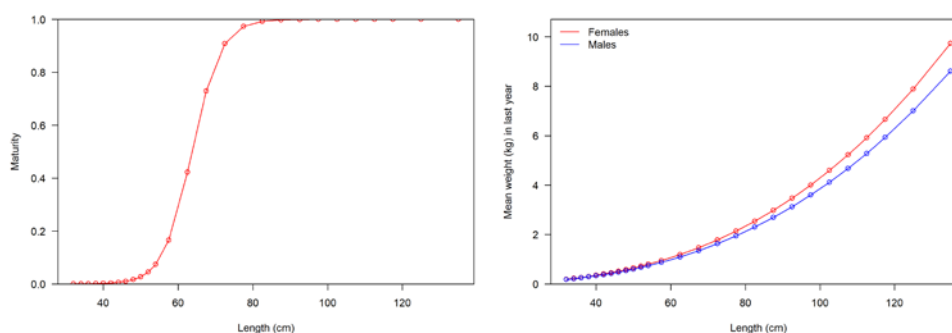


Figure 10.4. The maturity ogive by length for female blue grenadier (parameters from Russell and Smith (2006)) and the length-weight relationship for males and females.

10.4.8 Age-reading error

Updated standard deviations for aging error by reader (A and B) have been estimated, producing the age-reading error matrix of Table 10.6 (A. Punt, pers. comm.). Reader A applied to years 1991-93 and 2007-17, and reader B to years 1984-90 and 1994-2006.

Table 10.6. The standard deviation of age reading error.

| Age | St Dev | |
|-----|--------|-------|
| | A | B |
| 0 | 0.223 | 0.282 |
| 1 | 0.223 | 0.282 |
| 2 | 0.266 | 0.299 |
| 3 | 0.301 | 0.318 |
| 4 | 0.331 | 0.338 |
| 5 | 0.357 | 0.359 |
| 6 | 0.378 | 0.383 |
| 7 | 0.396 | 0.408 |
| 8 | 0.412 | 0.435 |
| 9 | 0.424 | 0.464 |
| 10 | 0.435 | 0.495 |
| 11 | 0.444 | 0.529 |
| 12 | 0.452 | 0.565 |
| 13 | 0.459 | 0.604 |
| 14 | 0.464 | 0.646 |
| 15 | 0.469 | 0.692 |
| 16 | 0.473 | 0.741 |
| 17 | 0.476 | 0.793 |
| 18 | 0.479 | 0.850 |
| 19 | 0.481 | 0.911 |
| 20 | 0.483 | 0.976 |

10.5 Analytic approach

The 2018 base case assessment of blue grenadier uses an age- and size-structured model implemented in the generalized stock assessment software package, Stock Synthesis (SS) (Version 3.30.12.00-safe, NOAA 2018). The methods utilised in SS are based on the integrated analysis paradigm. SS can allow for multiple seasons, areas and fleets, but most applications are based on a single season and area. Recruitment is governed by a stochastic Beverton-Holt stock-recruitment relationship, parameterized in terms of the steepness of the stock-recruitment relationship (h), the expected average recruitment in an unfished population (R_0), and the degree of variability about the stock-recruitment relationship (σ_r). SS allows the user to choose among a large number of age- and length-specific selectivity patterns. The values for the parameters of SS are estimated by fitting to data on catches, catch-rates, acoustic, FIS and egg surveys, discard mass, discard and retained catch length-compositions, and conditional age-at-length data. The population dynamics model and the statistical approach used in fitting the model to the various data types are given in the SS technical documentation (Methot, 2005).

This assessment follows the agreements made at the 2013 meetings of Slope RAG, and the 2018 meeting of SERAG. These were: include gender-specific selectivity for the spawning fishery, estimate natural mortality for females, use historical discard tonnages estimated by MAFRI, include cohort-dependent growth, set steepness at 0.75, use the non-spawning area of FIS abundances estimates, and separate port and onboard lengths.

The base–case model includes the following key features:

- a) Two sub-fisheries are included in the model – the spawning sub-fishery that operates during winter (June – August inclusive) off western Tasmania (zone 40), and the non-spawning sub-fishery that operates during other times of the year and in other areas throughout the year.
- b) The selectivity pattern was assumed to be length-specific, logistic and time-invariant for the spawning fleet and dome-shaped for the non-spawning fleet. The parameters of the selectivity function for each fleet were estimated within the assessment.
- c) The inclusion of the FIS is considered for the first time for the non-spawning area, and the selectivity mirrors the corresponding non-spawning fleet (Fleet 2).
- d) Blue grenadier consists of a single stock within the area of the fishery.
- e) The model accounts for males and females separately (growth, natural mortality, age at first breeding).
- f) The population was at its unfished biomass with the corresponding equilibrium (unfished) age-structure at the start of 1960.
- g) The CVs of the CPUE indices were initially set at a value equal to the standard error from a loess fit (0.25; Sporcic and Haddon, 2018), before being re-tuned to the model-estimated standard errors within SS. The acoustic estimates were tuned through the estimation of an extra variance component that is added to the model input standard errors. This is done within SS.
- h) Discard tonnage was estimated through the assignment of a retention function for the non-spawning fleet. This was defined as a logistic function of length, and the inflection and slope of this function were estimated where discard information was available. In addition, the discard length data from 1993, 1995 and 1996 were removed for the 2018 base case as recommended by SERAG (September, 2018) due to the existence of unusually large fish in the length distribution which is likely to be misreporting.
- i) The rate of natural mortality, M , is assumed to be constant with age, and also time-invariant. The value for female M is estimated within the assessment. Following previous assessments, male natural mortality is assumed to be 20% greater than that of females. A sensitivity was considered where both male and female M are estimated.
- j) Recruitment to the stock is assumed to follow a Beverton-Holt type stock-recruitment relationship, parameterised by the average recruitment at unexploited spawning biomass, R_0 , and the steepness parameter, h . Steepness for the base-case analysis is set to 0.75. Deviations from the average recruitment at a given spawning biomass (recruitment residuals) are estimated for 1974 to 2014. Deviations are not estimated before 1974 or after 2014 because there are insufficient data to permit reliable estimation of recruitment residuals outside of this time period.
- k) The initial value of the parameter determining the magnitude of the process error in annual recruitment, σ_r , is set to 1.0, reflecting the large variation in recruitment observed for blue grenadier. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Taylor and Methot (2011).
- l) The population plus-group is modelled at age 20 years. The maximum age for age observations was 15 years, reflecting that used in previous assessments.
- m) Growth is assumed to follow a von Bertalanffy type length-at-age relationship, with the parameters of the growth function being estimated separately for females and males inside the assessment model. Growth is also assumed to vary through time and to be cohort (year class)

specific. Evidence for time-varying and cohort specific growth in blue grenadier has been accumulating over several decades (see Punt and Smith 2001; Whitten et al., 2013). The 2018 base-case model treats conditional age-at-length information as data (i.e. to incorporate error), and predicts the expected length-at-age for each year. This is achieved by estimating the parameters of a von Bertalanffy growth function where the expected annual growth increment is based on the von Bertalanffy growth function but with a growth rate parameter that is determined by an expected value and a cohort-specific deviation. Cohort-specific deviations from average growth are estimated in the base case model for year classes 1978 to 2014.

- n) Retained and discarded onboard length sample sizes were capped at 200 and a minimum of 100 fish measured was required for length-composition data to be included in the assessment. For port samples, numbers of trips were used as the sampling unit, with a cap of 100. The number of fish measured is not used as the sample size because the appropriate sample size for length-composition data is probably more closely related to the number of shots (onboard) or trips (port) sampled, rather than the number of fish measured (Table 10.3).

The values assumed for some of the parameters of the preliminary base case model are shown in Table 10.7.

Table 10.7. Parameter values assumed for some of the non-estimated parameters of the base-case model (BC).

| Parameter | Description | BC |
|------------|---|--|
| M_f | Natural mortality for females | Estimated |
| M_m | Natural mortality for males | $1.2 * M_f$ |
| σ_r | Initial CV for the recruitment residuals | 1.0 |
| σ_g | Input standard deviation for the cohort growth deviations | 0.1 |
| h | “steepness” of the Beverton-Holt stock-recruit curve | 0.75 |
| x | age observation plus group | 15 years |
| μ | fraction of mature population that spawn each year | 0.84 |
| aa | Female allometric length-weight equations | $0.01502 \text{ g}^{-1} \cdot \text{cm}$ |
| bb | Female allometric length-weight equations | 2.728 |
| aa | Male allometric length-weight equations | $0.0168 \text{ g}^{-1} \cdot \text{cm}$ |
| bb | Male allometric length-weight equations | 2.680 |
| l_m | Female length at 50% maturity | 63.7cm |
| l_s | Parameter defining the slope of the maturity ogive | -0.261 |

10.5.1.1 Tuning method

Iterative rescaling (reweighting) of input CVs or input sample sizes is a repeatable method to ensure that the expected variation of the different data streams is comparable to what is input (Pacific Fishery Management Council, 2018). Sampling standard deviations/ CVs and stage-1 effective sample sizes for most of the data (CPUE, survey indices, composition data) used in fisheries assessments underestimate their true error by only reflecting measurement or estimation error and not including process (or model) error.

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size is equal to the effective sample size calculated within the model. In SS3.30 there is an automatic adjustment made to survey CVs (CPUE). The steps are:

1. Set the standard error for the log of relative abundance indices (CPUE, acoustic abundance survey, or FIS) to their estimated standard errors for each survey or for CPUE (and FIS values) to the standard deviation of a loess curve fitted to the logs of the original data (which will provide a more realistic estimate compared to that obtained from the original statistical analysis). SS3.30 then re-balances the relative abundance variances appropriately.
2. The initial value of the parameter determining the magnitude of the process error in annual recruitment, σ_r , is set to 1.0, reflecting the large variation in recruitment observed for blue grenadier. The magnitude of bias-correction depends on the precision of the estimate of recruitment and time-dependent bias-correction factors were estimated following the approach of Taylor and Methot (2011).

An automated tuning procedure was used for the remaining adjustments. For the conditional age-at-length and length-composition data:

3. Multiply the stage-1 sample sizes for for the conditional age-at-length data by the sample size multipliers using the approach of Punt (2017).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length-composition data using the 'Francis method' (Francis, 2011).
5. Repeat steps 2 and 3, until all are converged and stable (proposed changes are $< 1 - 2\%$).

This procedure may change in the future after further investigations, but this approach constitutes current best practice.

10.6 Calculating the RBC

The SESSF Harvest Strategy Framework (HSF) was developed during 2005 (Smith et al., 2008) and has been used as a basis for providing advice on TACs in the SESSF quota management system for fishing years 2006–2018. The HSF uses harvest control rules to determine a recommended biological catch (RBC) for each stock in the SESSF quota management system. Each stock is assigned to one of five Tier levels depending on the basis used for assessing stock status or exploitation level for that stock. Blue grenadier is assessed as a Tier 1 stock as it has an agreed quantitative stock assessment.

The Tier 1 harvest control rule specifies a target and a limit biomass reference point, as well as a target fishing mortality rate. Since 2005 various values have been used for the target and the breakpoint in the rule. Currently, the 20:40:40 (Blim:Bmsy:Ftarg) form of the rule will be used up to where fishing mortality reaches F48. Once this point is reached, the fishing mortality is set at F48. Day (2008) has determined that for most SESSF stocks where the proxy values of B40 and B48 are used for BMSY and BMEY this form of the rule is equivalent to a 20:35:48 strategy.

This document reports RBCs calculated under the 20:35:48 strategy.

10.7 Sensitivity tests and alternative models

A number of tests were used to examine the sensitivity of the results of the model to some of the assumptions and data inputs:

1. Estimation of M for males.
2. Exclusion of cohort dependent growth.
3. Exclusion of the port length data.
4. Exclusion of the FIS abundance series.
5. $h = 0.9$ (0.75 in the base case).
6. $M = 1.2 M_{est} \text{ yr}^{-1}$, where M_{est} is the estimated value for female natural mortality (0.174) in the base case.
7. $M = 0.8 M_{est} \text{ yr}^{-1}$.
8. Double and halve the weighting on the length composition data.
9. Double and halve the weighting on the age-at-length data.
10. $\sigma_r = 0.8$.
11. $\sigma_r = 1.2$.

The results of the sensitivity tests are summarized by the following quantities:

1. SB_0 the average equilibrium female spawning biomass.
2. SB_{2019} the female spawning biomass at the start of 2019.
3. SB_{2019}/SB_0 the depletion level at the start of 2019, i.e. the 2019 spawning biomass expressed as a fraction of the unexploited spawning biomass.
4. *2019 RBC* - the 2019 RBC, calculated using the 20:35:48 harvest rule.
5. *Long-term RBC* - the long-term RBC calculated using the 20:35:48 harvest rule.

10.8 Results and Discussion

10.8.1 The base case stock assessment

10.8.1.1 Parameter estimates

Figure A 10.2 shows how the expected mean length-at-age values change over time for the base case model. The ridges reflect the impact of some cohorts growing faster or slower than average. This figure also shows the expected mean length-at-age values for the end-year of the model. The impact of slower than average growth is visible by the decrease in expected size of 9 and 18 yo fish, corresponding to the larger than average recruitments in years 2003 and 1994 respectively. Natural mortality for females was estimated to be $M_f=0.174$ and males therefore was $M_m=0.204$.

The selectivity for the spawning and non-spawning fisheries and the retention function for the non-spawning fishery are shown in Figure A 10.5. Selectivity is assumed to be time-invariant, sex-specific and logistic for the spawning fleet and dome-shaped for the non-spawning fleet. Note that the estimated female length-specific selectivity for the spawning ground shows an ascending limb that includes much larger fish than the maturity ogive estimated by Russell and Smith (2006), which has an estimate of 50% maturity of 63.7cm. This result implies that, to a large extent, small mature females do not appear to be evident on the spawning ground. Russell and Smith (2006) present length frequencies during their study of blue grenadier reproductive biology showing that very few female fish less than 60cm

were caught (also see Figure 10.4). However, those that were caught were included in the study and a proportion of these fish were shown to be mature.

10.8.1.2 Fits to the data

Figure A 10.4 shows the model fit to the non-spawning catch rate series. The model fits intersect most of the 95% confidence intervals for the data, indicating that adjustments to the CV for the indices performed as expected. As has been seen in all previous assessment models for blue grenadier, the model is not able to fit the rise in catch rate following the large recruitment of the mid-1990s. The fit to the discard mass is able to replicate the increase in discarding through the late 1990s and mid 2000s, however the magnitude is under-estimated (as has been the case with previous assessments). Alternative models that time-blocked discarding, re-weighting discard CVs and including a discard fleet have all been unsuccessful in improving the fit to the discard and CPUE data. Further consideration should be perhaps be given to the GLM model structure used in the standardisation of CPUE. Fits to the biomass estimates from the acoustic surveys and egg surveys were reasonable. The predicted biomass trajectory intersects all of the 95% confidence intervals.

The model is able to replicate the length composition and implied age-composition data well (Figure A 10.6 – Figure A 10.17). Predicted age-compositions are able to track the strong cohorts typical of blue grenadier as they move through both the non-spawning fishery and the spawning fishery. Length composition data are also well estimated by the model.

10.8.2 Assessment outcomes

The estimated time series of recruitment under the base-case parameter set shows the typical episodic nature of blue grenadier recruitment, with strong year-classes in 1979, the mid-1980s, 1994, 2003, and from 2010 to 2014 (Figure 10.5). The magnitude of the recruitment of 2010 is consistent with the estimation in the previous assessment (Tuck, 2013) and is well estimated according to the current model results. As with the 2010 recruitment in Tuck (2013), the magnitude of the recent recruitment estimates will be better determined in the next assessment as they move well into the available stock of the fishery.

The trajectories of spawning biomass and spawning biomass relative to the un-exploited level are shown in Figure 10.5. This shows the increases and decreases in spawning biomass as the strong cohorts move into and out of the spawning population. The estimated virgin female biomass is 53,909 tonnes (compared to 36,815 t in the 2013 assessment). In the 2018 assessment, the estimated spawning biomass in 2019 which is used in the harvest control rule, is approximately 122% *SBo* (compared to 94% *SBo* in the 2013 assessment).

The optimistic outlook from this assessment is largely being driven by the addition of 5 further years of data and the substantial estimates of recruitment since 2010. While a promising sign for the fishery, some caution should be exercised with regard to these recruitment estimates and its implication on future stock status, until clear further indications of its existence (and magnitude) are evident in future years' data. But note that the recruitment estimates do appear to be well estimated (Figure A 10.3; top right).

For the base case model the 2019 recommended biological catch (RBC) under the 20:35:48 harvest control rule is 13,260t. The long-term retained catch is 4,899t. The retained portion of the RBC for 2019 is estimated to be 12,671t (Table 10.8).

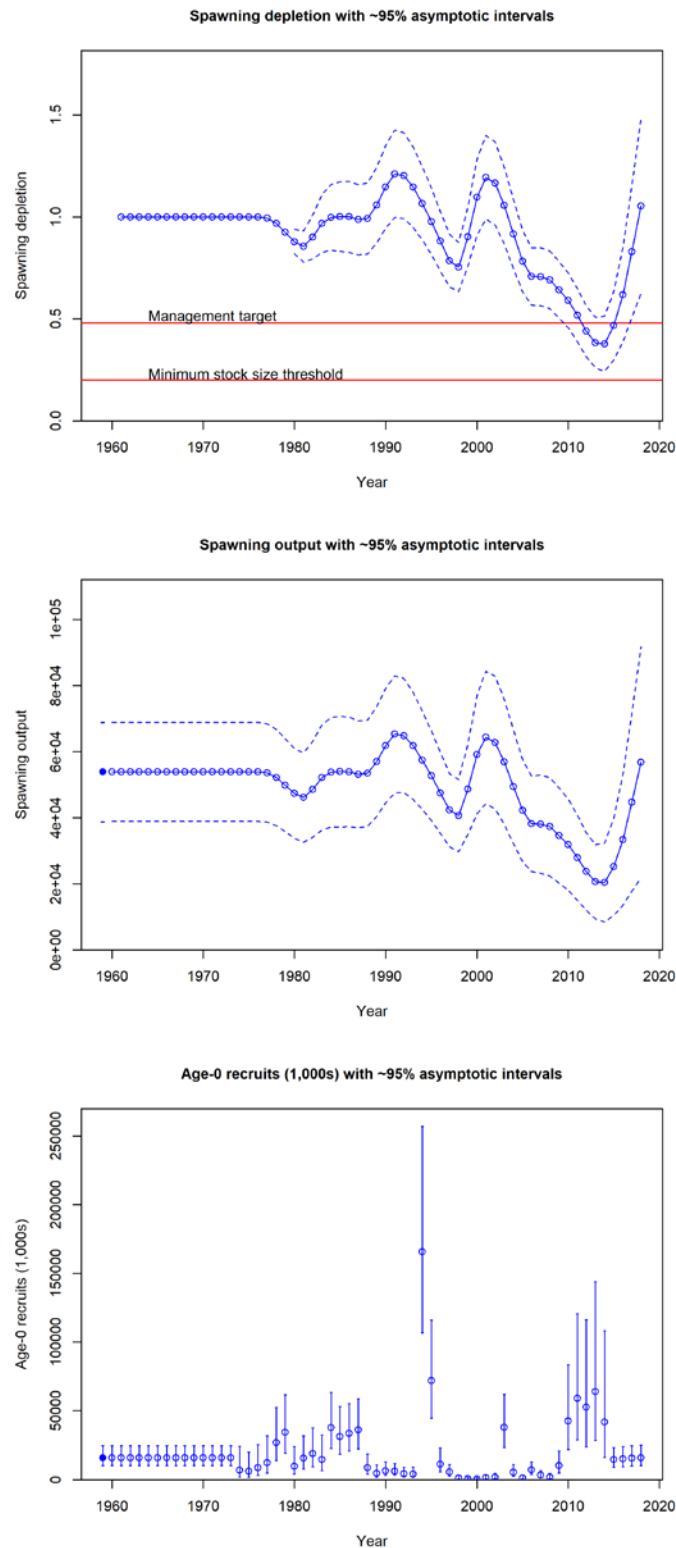


Figure 10.5. The estimated time-series of relative spawning biomass and annual recruitment for the 2018 base case assessment for blue grenadier.

Table 10.8. The estimated retained portion of the RBC and the RBC (that includes a model estimate of discarding) for blue grenadier under the base case model.

| Year | Retained | |
|------|----------|--------|
| | Catch | RBC |
| 2019 | 12,671 | 13,260 |
| 2020 | 11,706 | 12,238 |
| 2021 | 10,529 | 11,052 |
| 2022 | 9,422 | 9,943 |
| 2023 | 8,493 | 9,012 |
| 2024 | 7,748 | 8,264 |
| 2025 | 7,159 | 7,672 |
| 2026 | 6,693 | 7,202 |
| 2027 | 6,321 | 6,827 |
| 2028 | 6,023 | 6,525 |
| 2029 | 5,782 | 6,281 |
| 2030 | 5,586 | 6,081 |
| 2031 | 5,425 | 5,918 |
| 2032 | 5,293 | 5,783 |
| 2033 | 5,184 | 5,671 |
| 2034 | 5,093 | 5,578 |
| 2035 | 5,017 | 5,500 |
| 2036 | 4,953 | 5,434 |
| 2037 | 4,899 | 5,378 |

10.8.2.1 Sensitivity tests

Results of the sensitivity tests are shown in Table 10.9 and Table 10.10, and Figure 10.6. Steepness is not well estimated as the model estimated spawning biomass does not decrease to low enough magnitudes to inform the estimation of this parameter (as confirmed by the likelihood profile conducted on steepness in Castillo-Jordan and Tuck, 2018). All model sensitivities show relative spawning biomass levels well above the target biomass level (48% SBo). A model sensitivity with both female and male natural mortality (*Mmale*) estimated provided reasonable fits to the data (not shown), and predicted natural mortality values of $M_f=0.154\text{ y}^{-1}$ and $M_m=0.230\text{ y}^{-1}$ (noting that the male natural mortality is estimated to be ~1.5 times that of females). The inclusion of estimated male natural mortality could be considered in the base case for this stock (see Figure A 10.19 for a comparison of the base case and *Mmale* model spawning biomass and recruitment time series).

Table 10.9. Summary of results for the base case model BC and sensitivity tests. ^This is the retained catch at 2037. The long term catch had not yet stabilised by year 2037. Ret C = retained catch. Ret C 2019-21 is the average 3-year retained catch. Ret C 2019-23 is the average 5-year retained catch. Note that the upper two models are tuned.

| Model | SB0 | SB_Curr | CurrDepl | 2019 RBC | 2019 Ret C | Ret C 2019-2021 | Ret C 2019-2023 | Ret C Long-term |
|--|--------|---------|----------|----------|------------|-----------------|-----------------|-----------------|
| Base Case Model (Mf=est=0.174, h=0.75) | 53,909 | 65,993 | 1.22 | 13,260 | 12,671 | 11,635 | 10,564 | ^4,899 |
| Mmale (Mf=est=0.154, Mm=est=0.230) | 57,647 | 62,027 | 1.08 | | | | | |
| NoCGD | 48,036 | 56,589 | 1.18 | | | | | |
| NoPort data | 54,668 | 67,942 | 1.24 | | | | | |
| NoFIS | 54,146 | 67,895 | 1.25 | | | | | |
| <i>h</i> =0.90 | 52,581 | 64,403 | 1.22 | | | | | |
| <i>Mf</i> 20% more | 59,815 | 85,121 | 1.42 | | | | | |
| <i>Mf</i> 20% less | 47,831 | 45,210 | 0.95 | | | | | |
| Halve weight on LF data | 54,309 | 66,603 | 1.23 | | | | | |
| Double weight on LF data | 52,933 | 65,061 | 1.23 | | | | | |
| Halve weight on Age data | 57,021 | 67,820 | 1.19 | | | | | |
| Double weight on Age data | 51,259 | 65,464 | 1.28 | | | | | |
| Sigma R 0.8 | 47,284 | 59,876 | 1.27 | | | | | |
| Sigma R 1.2 | 64,337 | 69,950 | 1.09 | | | | | |

Table 10.10. Summary of likelihood components for the base-case BC and sensitivity tests. Likelihood components are unweighted, and sensitivities from the BC are shown as differences from the base case. A negative value indicates a better fit, a positive value a worse fit. Note that the upper two models are tuned and so likelihoods are not comparable.

| Model | TOTAL | Survey | Discard | Length comp | Age comp | Recruitment |
|--|-------|--------|---------|-------------|----------|-------------|
| Model BC (Mf=est=0.174, h=0.75) | 773 | -6 | 27 | 201 | 580 | 39 |
| Mmale (Mf=est=0.154, Mm=est=0.230) | 3 | 1 | 0 | 4 | -2 | 0 |
| NoCGD | 267 | 1 | 36 | 85 | 75 | 2 |
| NoPort data | -56 | -1 | -5 | -45 | -4 | 0 |
| NoFIS | -1 | -1 | 0 | 0 | 0 | 0 |
| <i>h</i> =0.90 | -1 | 0 | 0 | 0 | 0 | 0 |
| <i>Mf</i> 20% more | 4 | 0 | -1 | 4 | 1 | 0 |
| <i>Mf</i> 20% less | 5 | 1 | 1 | -1 | 4 | 0 |
| Halve weight on LF data | 10 | -1 | -13 | 31 | -5 | 0 |
| Double weight on LF data | 11 | 0 | 25 | -25 | 10 | 0 |
| Halve weight on Age data | 7 | -1 | -7 | -6 | 23 | -3 |
| Double weight on Age data | 5 | 1 | 4 | 10 | -13 | 3 |
| Sigma R 0.8 | 14 | 0 | 3 | 1 | 3 | 6 |
| Sigma R 1.2 | -7 | 0 | -3 | 0 | -2 | -3 |

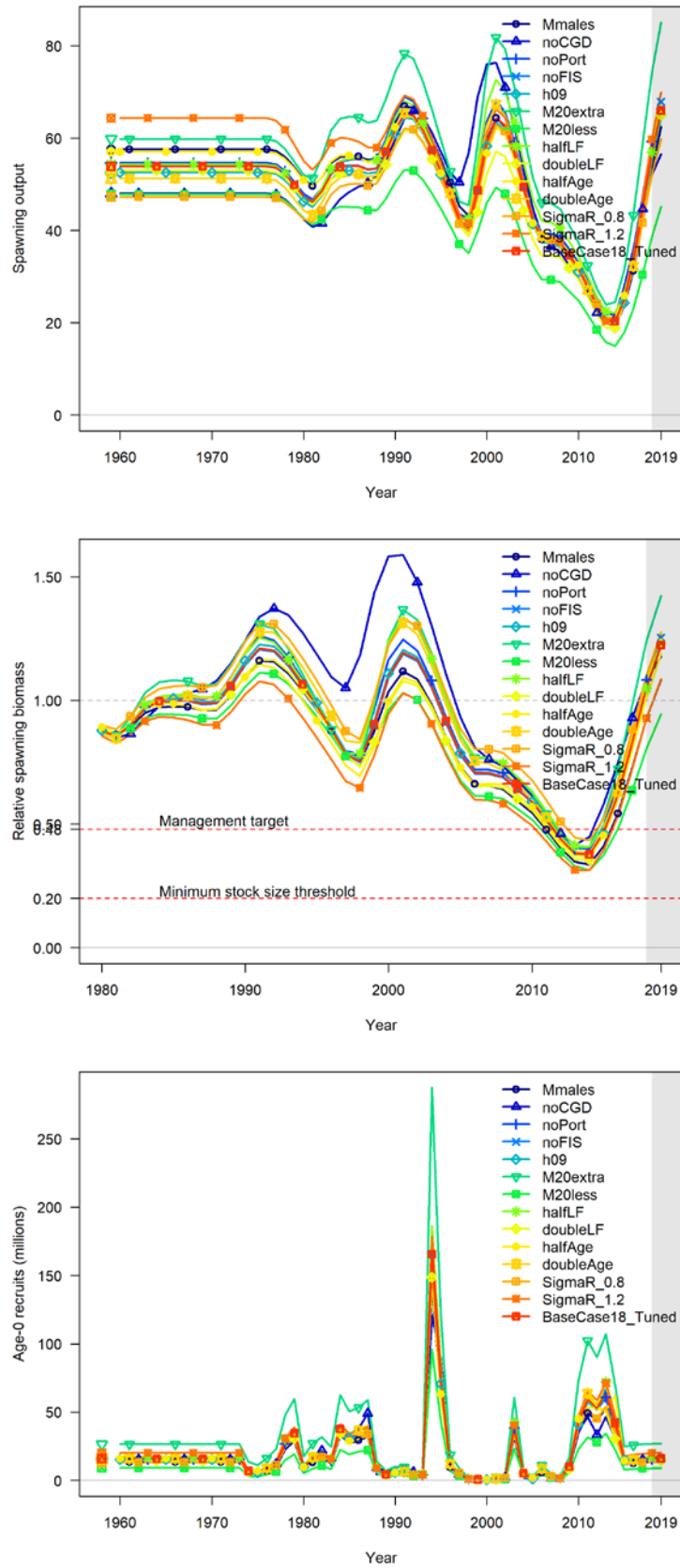


Figure 10.6. Comparison plot for the base case and sensitivities applied for the 2018 blue grenadier stock assessment for female spawning biomass (top), relative female spawning biomass (middle) and recruitment (bottom).

10.8.3 Further development

1. Explore the lack of fit to the catch rate series of the non-spawning fishery and whether the poor fit is a data issue or model structure issue.
2. Explore the utility of having age-dependent and sex-specific estimates of natural mortality.
3. Include updated FIS estimates of abundance and corresponding length frequencies.

10.9 Acknowledgements

Age data were provided by Kyne Krusic-Golub (Fish Ageing Services), ISMP and AFMA logbook and CDR data were provided by John Garvey (AFMA). Franzis Althaus, Mike Fuller, Roy Deng, and Paul Burch (CSIRO) pre-processed the data. Jemery Day, André Punt, Sandra Curin-Osorio, Paul Burch, Malcolm Haddon, Robin Thomson and Miriana Sporcic are thanked for helpful discussions on this work. Richard Methot, Chantel Wetzel and Ian Taylor (NOAA Fisheries) are thanked for all the Stock Synthesis support and advice. Malcolm Haddon provided useful code for auto-tuning, Athol Whitten provided useful R code for organising plots.

10.10 References

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10.11 Appendix A

A.1 Base case diagnostics

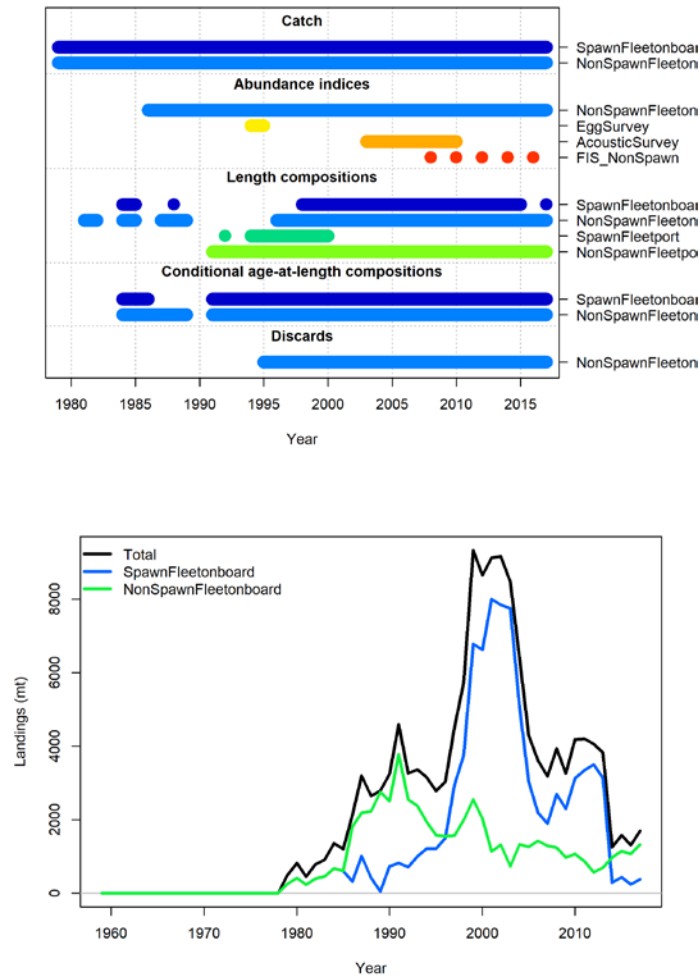


Figure A 10.1. Summary of data sources and the catch time-series for the base case assessment.

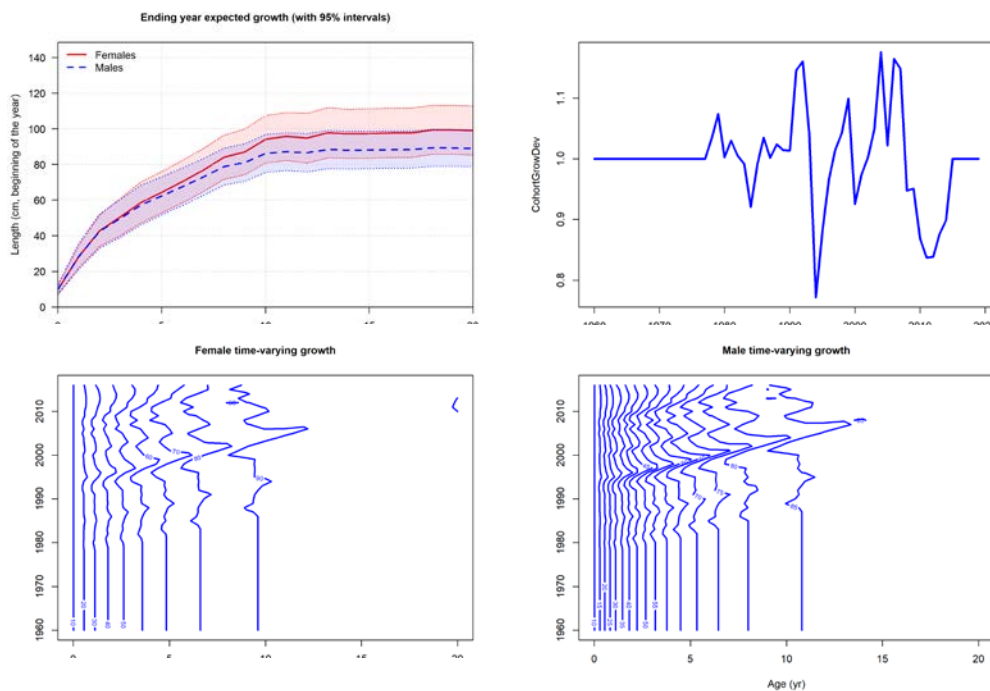


Figure A 10.2. Growth for blue grenadier.

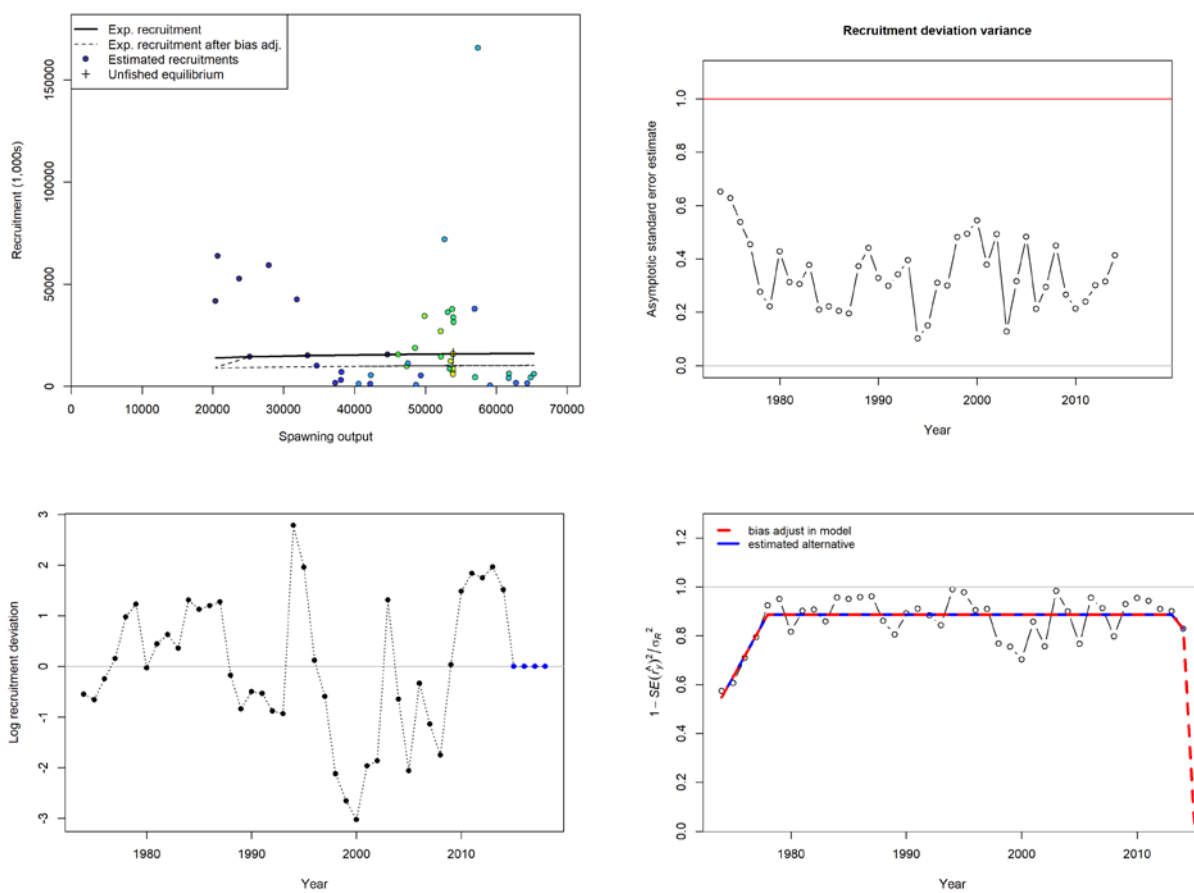


Figure A 10.3. Time series showing the stock recruitment curve, recruitment deviations and recruitment deviation variance check for blue grenadier.

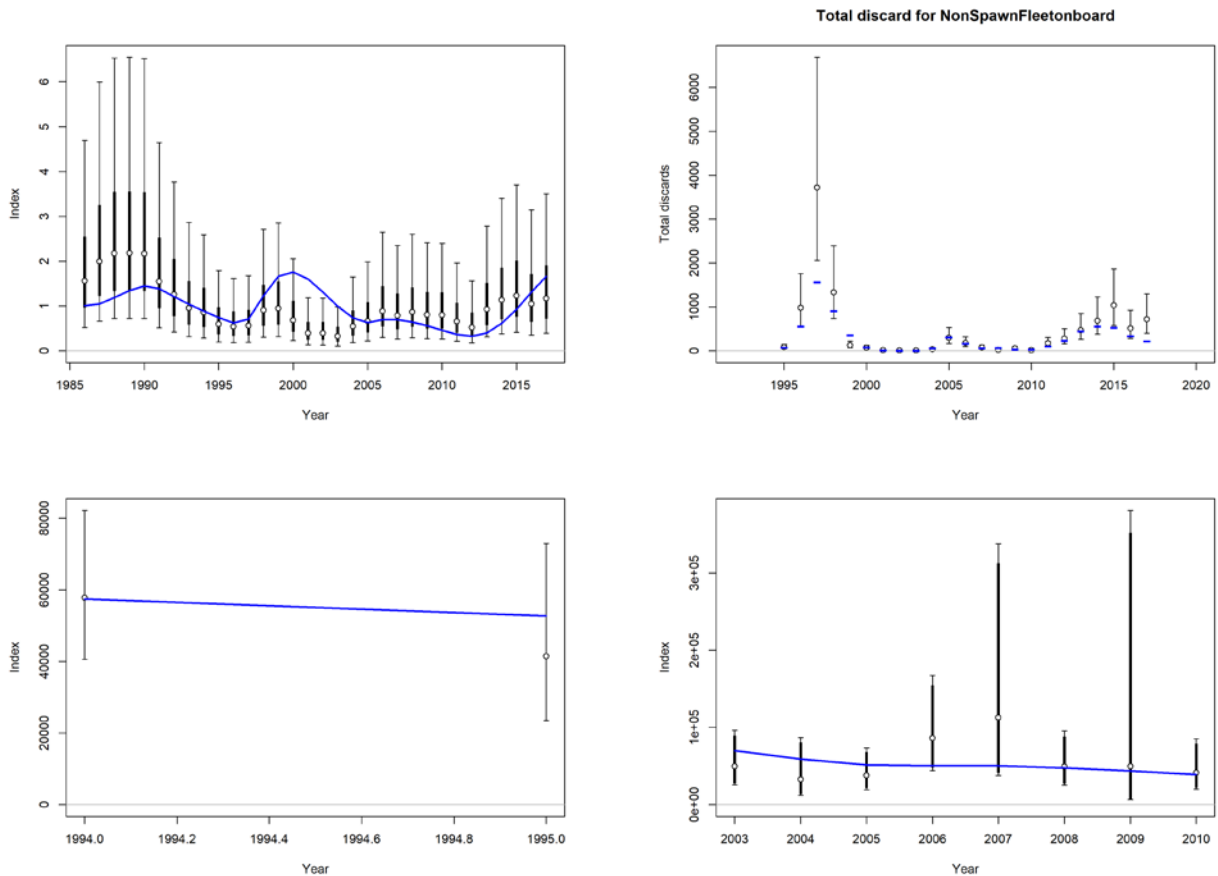


Figure A 10.4. Fits to the non-spawning CPUE index, discard mass, egg survey and acoustic survey.

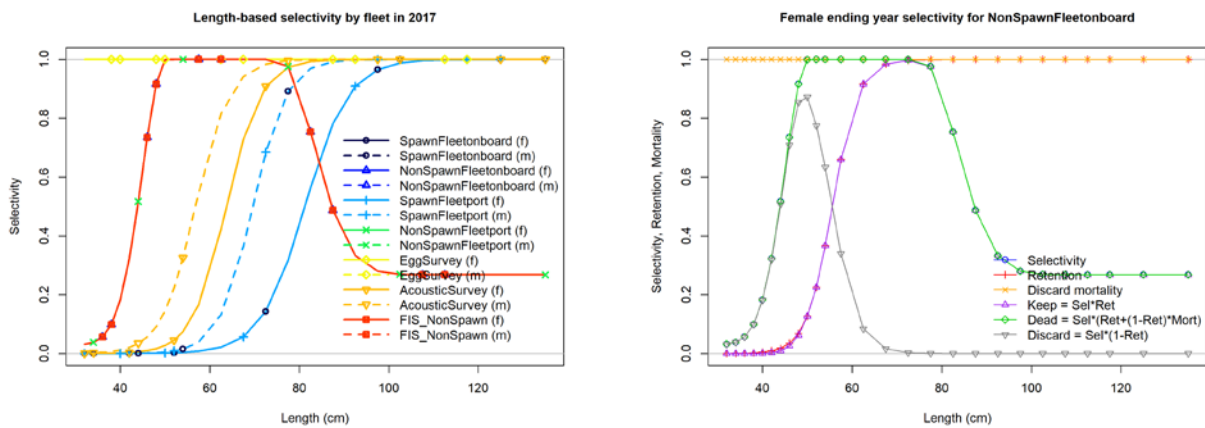


Figure A 10.5. Estimated selectivity for the spawning and on-spawning fleets using port and onboard samples and for males (m) and females (f) and the retention function.

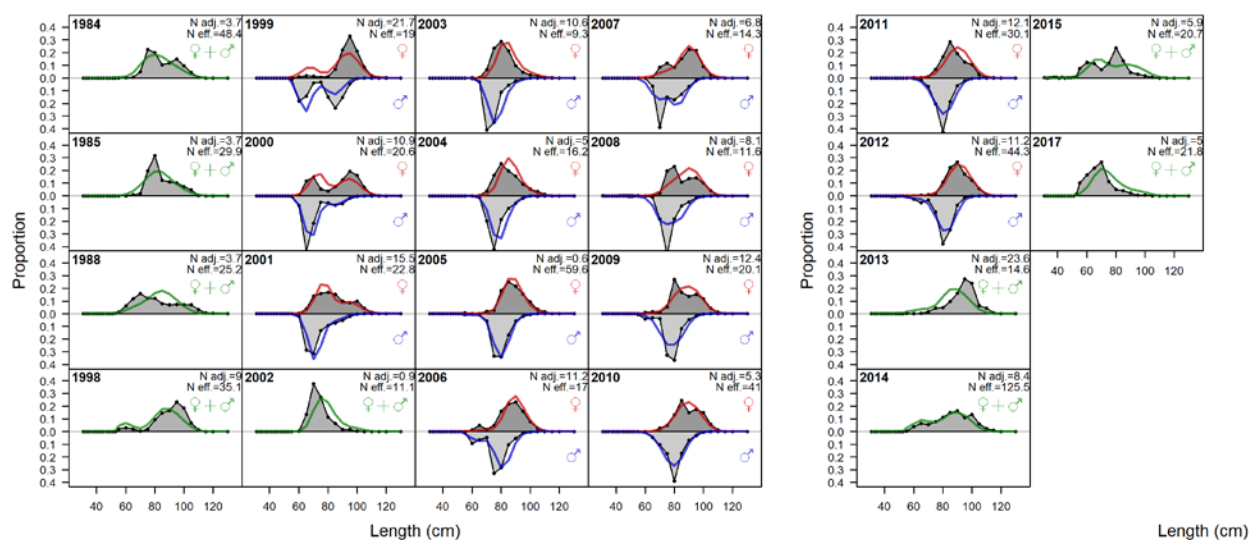


Figure A 10.6. Length composition fits: spawning fleet onboard retained.

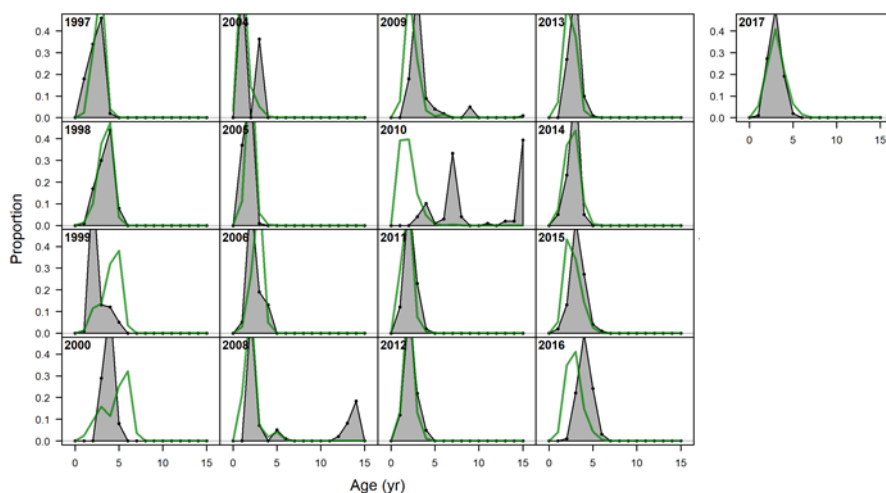


Figure A 10.7. Length composition fits: onboard non-spawning fleet discard.

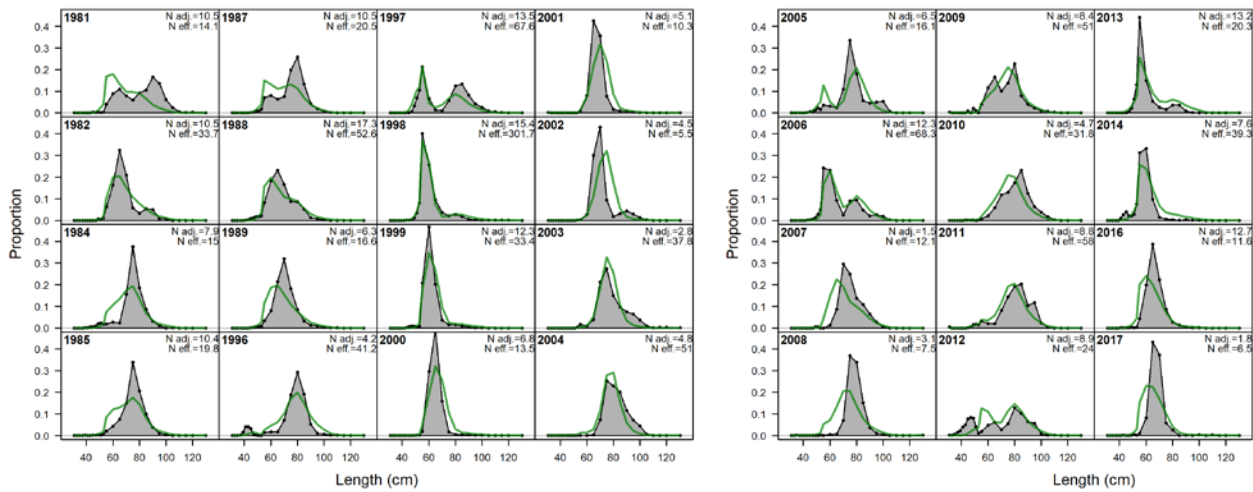


Figure A 10.8. Length composition fits: onboard non-spawning fleet retained.

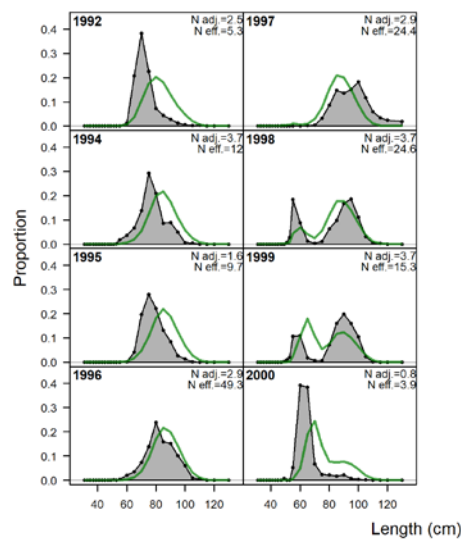


Figure A 10.9. Length composition fits: port spawning fleet retained

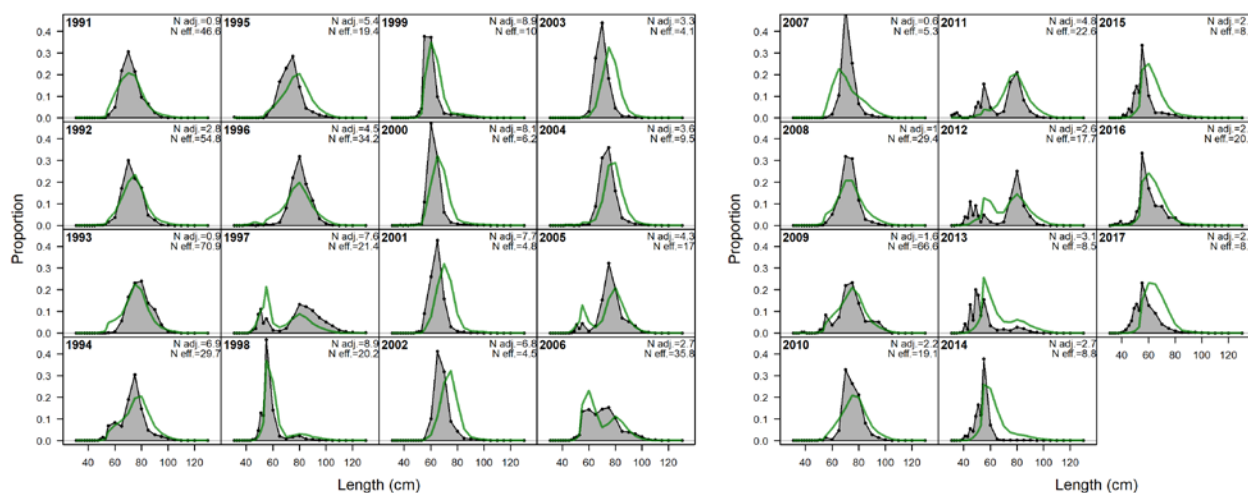


Figure A 10.10. Length composition fits: port non-spawning fleet retained.

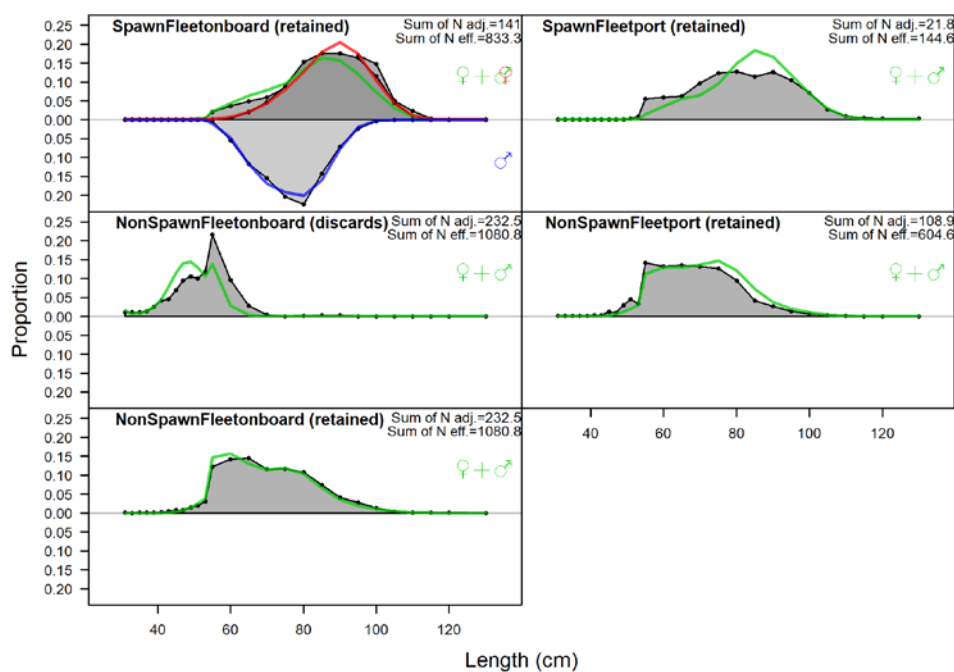


Figure A 10.11. Length composition fits aggregated across years.

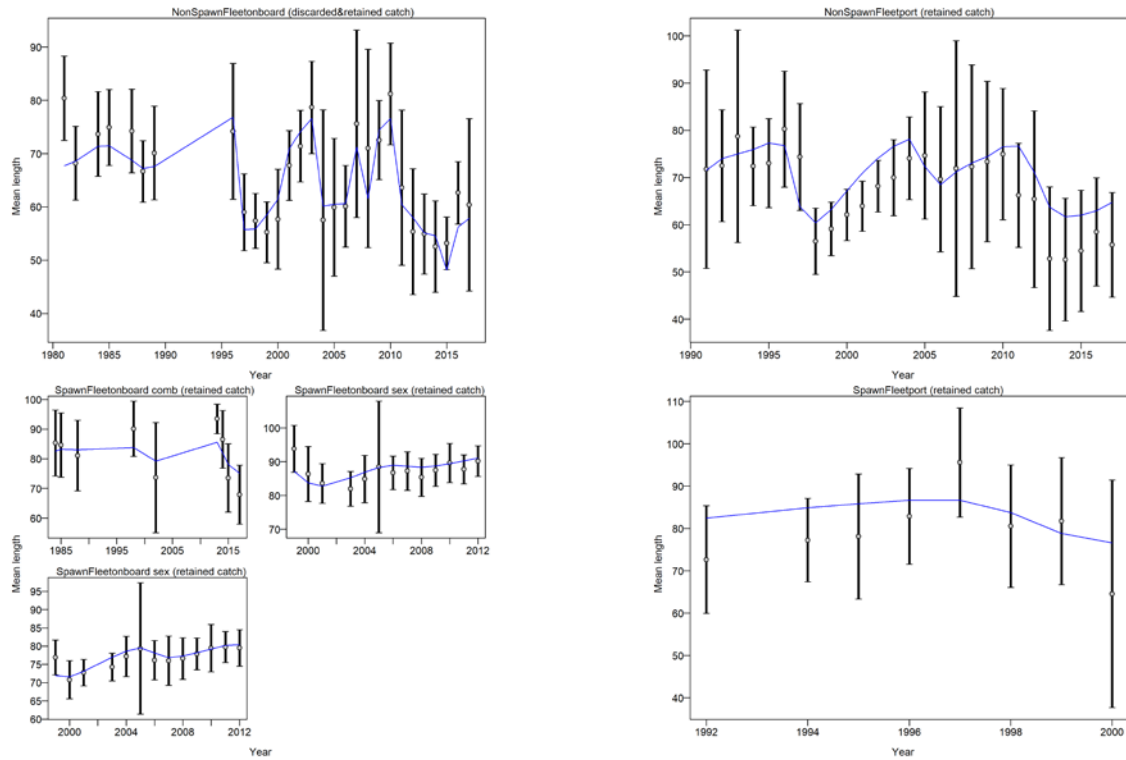


Figure A 10.12. Length composition fit diagnostics from tuning. Francis data weighting method TA1.8: thinner intervals (with capped ends) show result of further adjusting sample sizes based on suggested multiplier (with 95% interval) for length data.

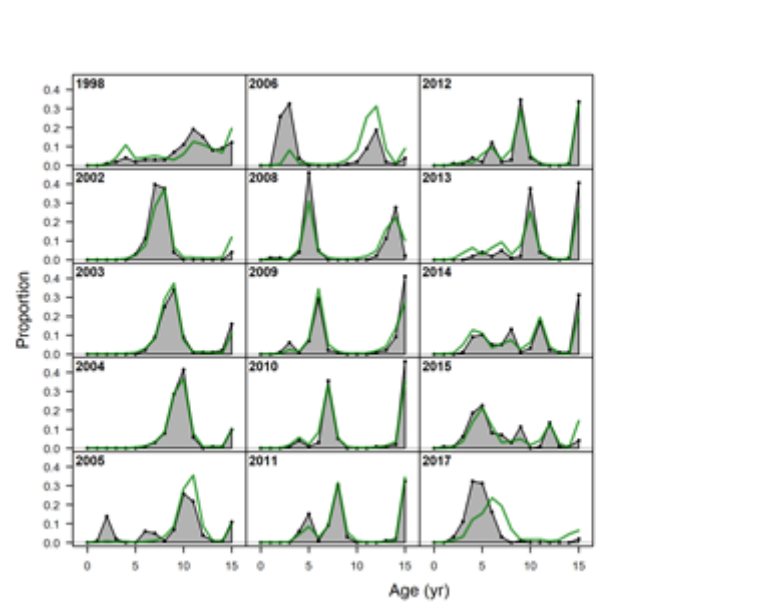


Figure A 10.13. Age composition fits: spawning fleet onboard retained.

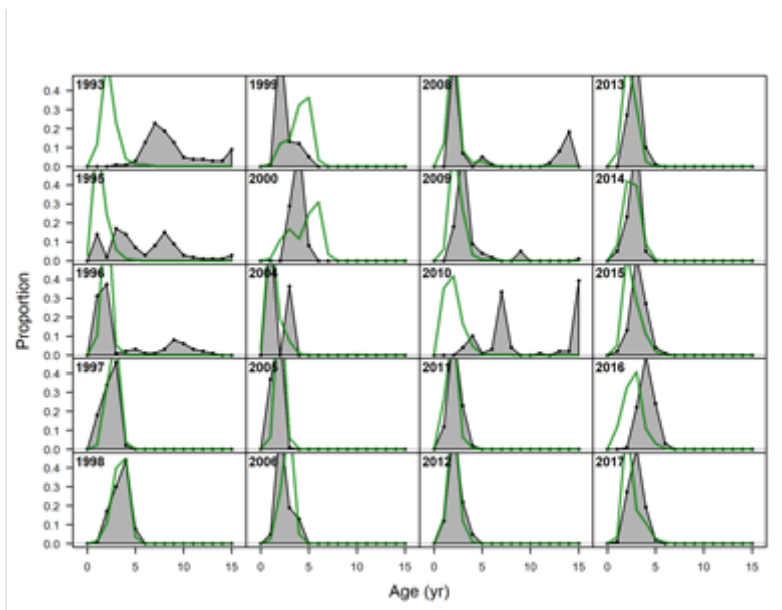


Figure A 10.14. Age composition fits: non-spawning fleet onboard discard.

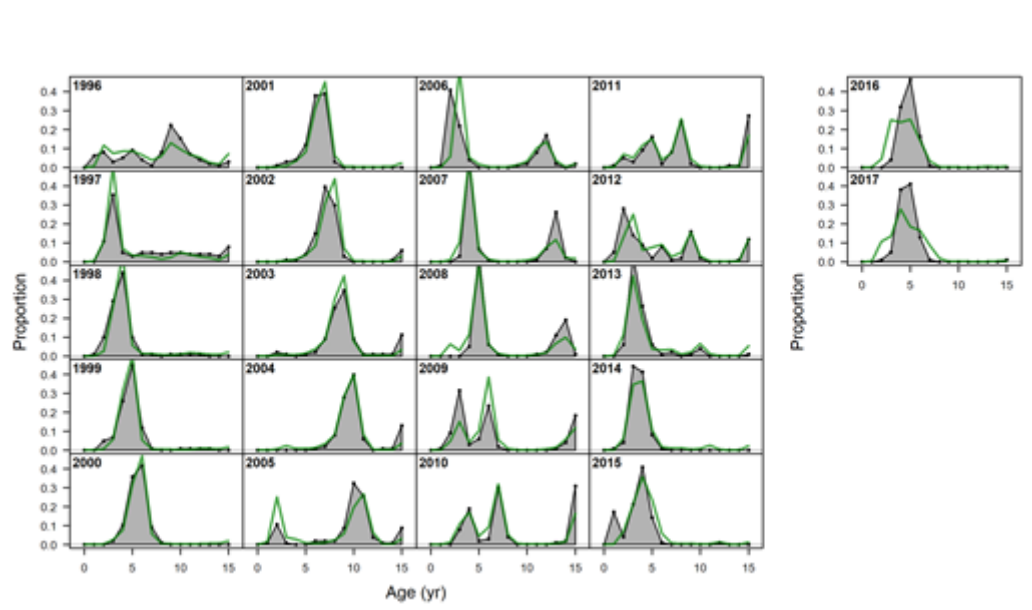


Figure A 10.15. Age composition fits: non-spawning fleet onboard retained.

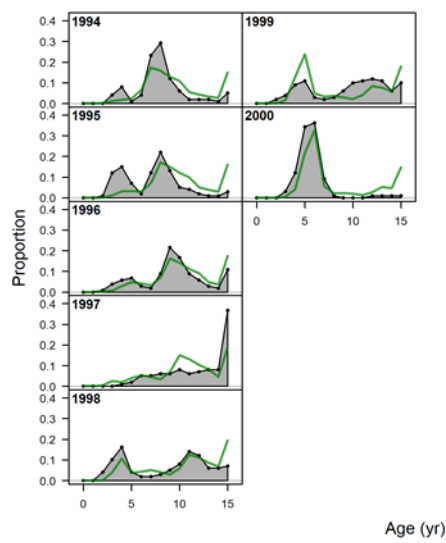


Figure A 10.16. Age composition fits: spawning fleet port retained.

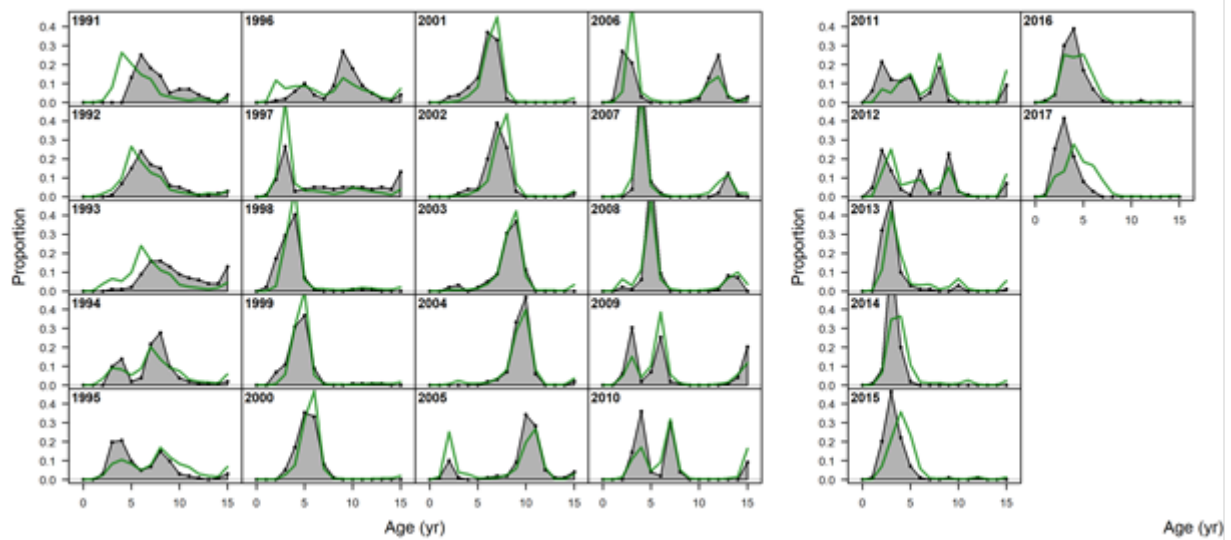


Figure A 10.17. Age composition fits: non-spawning fleet port retained.

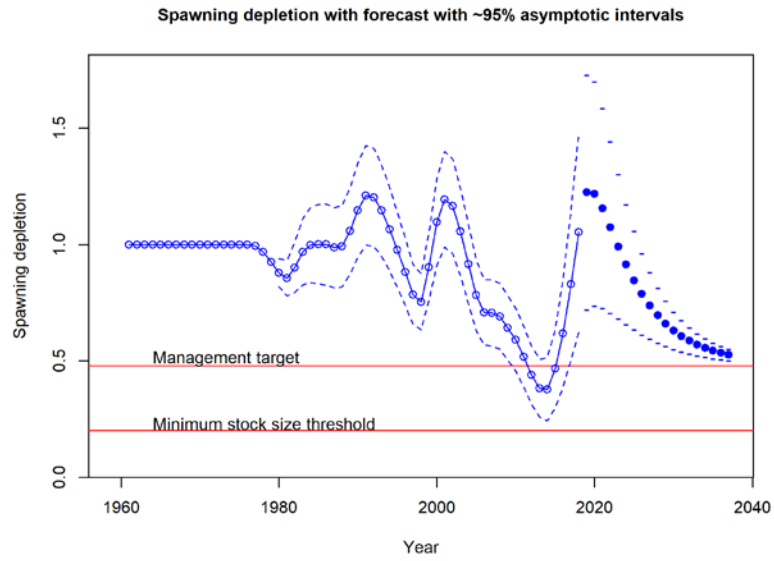


Figure A 10.18. The time-series of relative female spawning biomass with a projection to 2037.

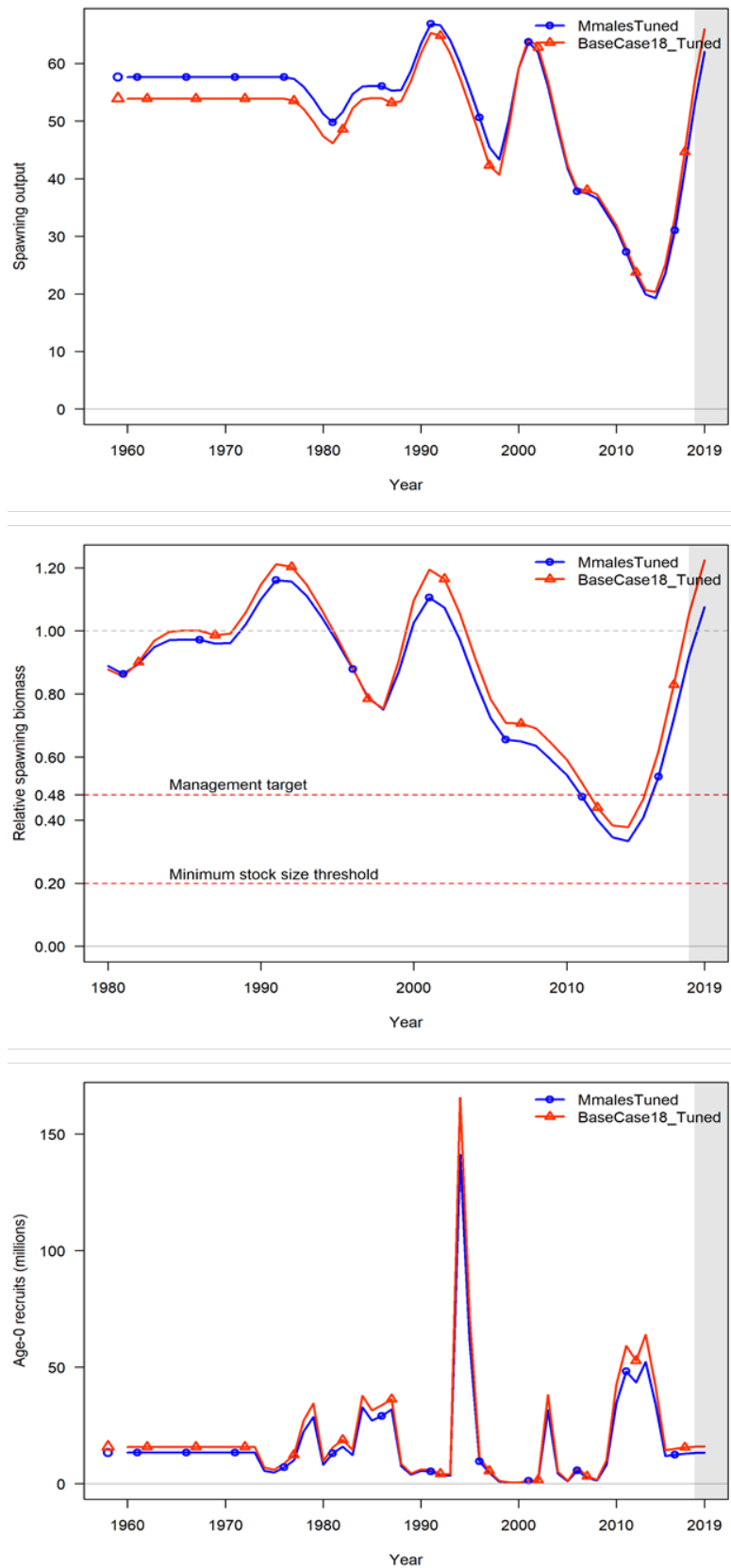


Figure A 10.19. Comparison plot for the base case and *Mmales* sensitivity applied for the 2018 blue grenadier stock assessment for female spawning biomass (top), relative female spawning biomass (middle) and recruitment (bottom).

11. Silver warehou (*Seriolella punctata*) stock assessment based on data up to 2017 – development of a preliminary base case

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11.1 Executive Summary

This document presents the preliminary base case for an updated quantitative Tier 1 assessment of silver warehou (*Seriolella punctata*) for presentation at the first SERAG meeting in 2018. The last full assessment was presented in Day et al. (2015). The preliminary base case has been updated by the inclusion of data up to the end of 2017, which entails an additional three years of catch, discard, CPUE, length-composition and conditional age-at-length data and ageing error updates and the inclusion of the 2016 east and west Fishery Independent Survey (FIS) indices. This document describes the process used to develop a preliminary base case for silver warehou through the sequential updating of recent data to the stock assessment, using the stock assessment package Stock Synthesis (SS-V3.30.12).

Changes to the last stock assessment include: the use of a re-estimated discards, the inclusion of age-at-length data for the western onboard trawl fleet and the use of an updated tuning method.

Results show good fits to the CPUE abundance index for the west trawl fleet. The fits to the CPUE of the eastern trawl fleet prior to 2001 are poor, however, from 2001 onwards they appear reasonable. The model fits are good to both the east and west FIS abundance indices, however, the width of the confidence intervals suggest this data provides little information to the model. The overall fits to the length and age-at-length data are reasonable with poor fits in some years.

This assessment estimates that the projected 2019 spawning stock biomass will be 33% of virgin stock biomass (projected assuming 2017 catches in 2018), however, it was predicted to be below 20% between 2013 and 2017. The previous assessment (Day et al., 2015) estimated depletion to be 40% at the start of 2016 and depletion had not dropped below 20% over the assessment period.

11.2 Introduction

11.2.1 Bridging from 2015 to 2018 assessments

The previous full quantitative assessment for silver warehou was performed in 2015 (Day et al., 2015) using Stock Synthesis (version SS-V3.24U; Methot 2015). The 2018 assessment uses the current version of Stock Synthesis (version SS-V3.30.12, Methot et al. 2018) which has had a number of changes from SS 3.24U. The main change to the assessment procedure and Stock Synthesis that relates to assessments in the SESSF is a revised tuning procedure. While the tuning procedure still uses Francis weighting for length and age data, the CPUE series are tuned within Stock Synthesis. There have also been improvements to the recruitment bias ramp adjustment.

As a first step in the process of bridging to a new model, the model was converted from version SS-V3.24U (Methot 2015) to version SS-V3.30.12 (Methot et. al, 2018) using the same data and model

structure used in the 2015 assessment. Once this translation was complete, improved features unavailable in SS-V3.24U were incorporated into the SS-V3.30.12 assessment, these included allowing smaller lower bounds on minimum sample sizes and estimating additional standard deviation to abundance indices. Following this step the model was re-tuned using the most recent tuning protocols. These changes to software and tuning practices are likely to lead to changes to key model outputs, such as the spawning biomass trajectory and depletion estimates. This initial bridging phase (Bridge 1) highlights changes that have occurred since 2015 simply through changes to software and assessment practices. The subsequent bridging exercise (Bridge 2) then sequentially updates the model with new data through to 2017.

The second part of the bridging analysis includes updating of some historical data (up to 2014), followed by including the data from 2015-2017 into the model. These additional data included new catch, discard, CPUE, length-composition, age-at-length data, an updated ageing error matrix and additional CPUE indices for the trawl fisheries. Additionally, age-at-length data for the western trawl fleet was included, which was omitted in the 2015 assessment, the reason for omission of the western trawl age-at-length is currently being investigated. The last year of recruitment estimation was extended to 2015 (from 2012 in the 2015 assessment).

The use of updated software and tuning procedure along with the inclusion of additional data resulted in some differences in the fits to CPUE, age and length data. The usual process of bridging to a new model by adding new data piecewise and analysing which components of the data could be attributed to changes in the assessment outcome was conducted with the details outlined below.

11.2.2 Update to Stock Synthesis SS-V3.30.12 (Bridge 1)

The 2015 silver warehou assessment (spot2015_24U) was initially translated to the most recent version of the software, Stock Synthesis version SS-V3.30.12 (spot2015_30_12) and there were negligible differences between the two models (Figure 11.1 to Figure 11.3). Re-tuning using the latest tuning protocol (Section 11.2.2.1) led to some minor changes in estimated recruitments (Figure 11.3) along with an increase in virgin spawning biomass and the estimated spawning biomass from 1980 to the late 1990s (Figure 11.2). After the late 1990s this trend reverses and the estimated spawning biomass of the re-tuned model is lower than the base case from the 2015 assessment.

This process demonstrates the consistency between Stock Synthesis version SS-V3.24U and the latest version used for the 2018 preliminary base case assessment. This initial bridging step, Bridge 1, does not incorporate any new data after 2014 or any structural changes to the assessment.

The results of Bridge 1 suggest that the stock may have been more depleted in 2016 than the 2015 assessment indicated. This is almost entirely due to changes in parameters that are being tuned, including variances that can be estimated internally and in the tuning procedure itself, rather than changes to the data or to the software.

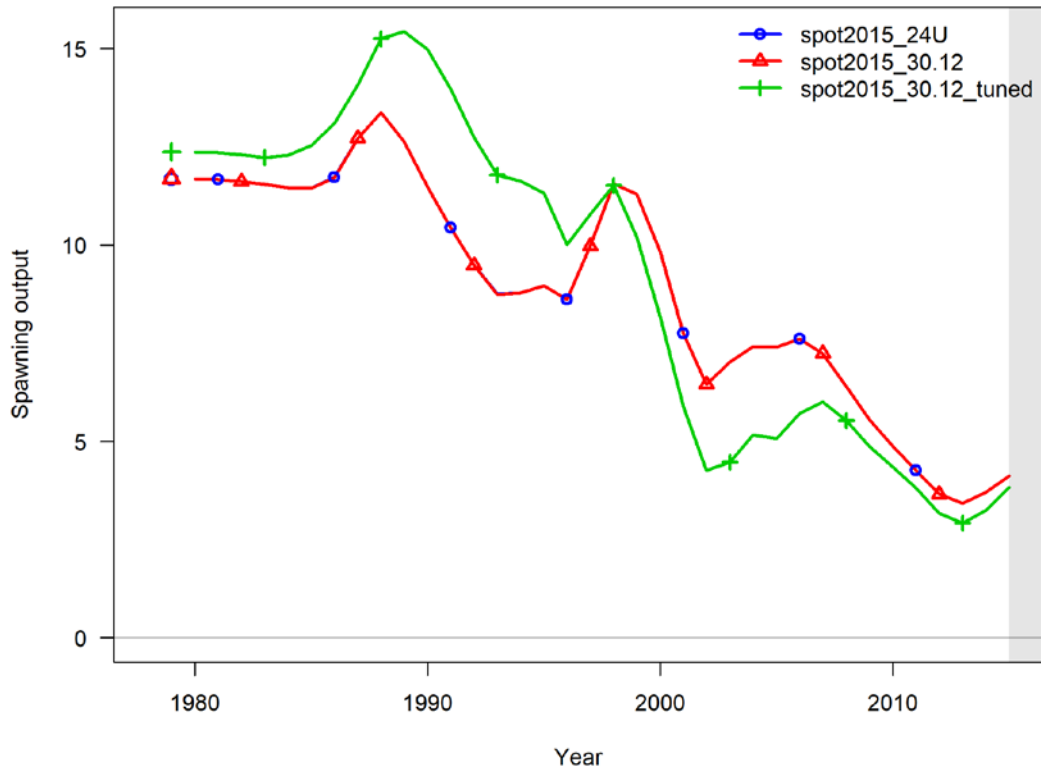


Figure 11.1. Comparison of the absolute spawning biomass time series for the 2015 assessment (spot2015_24U – in blue), and a model converted to SS-V3.30.12 (spot2015_30.12 in blue) and this same model tuned using the latest tuning procedures (spot2015_30.12_tuned – in green).

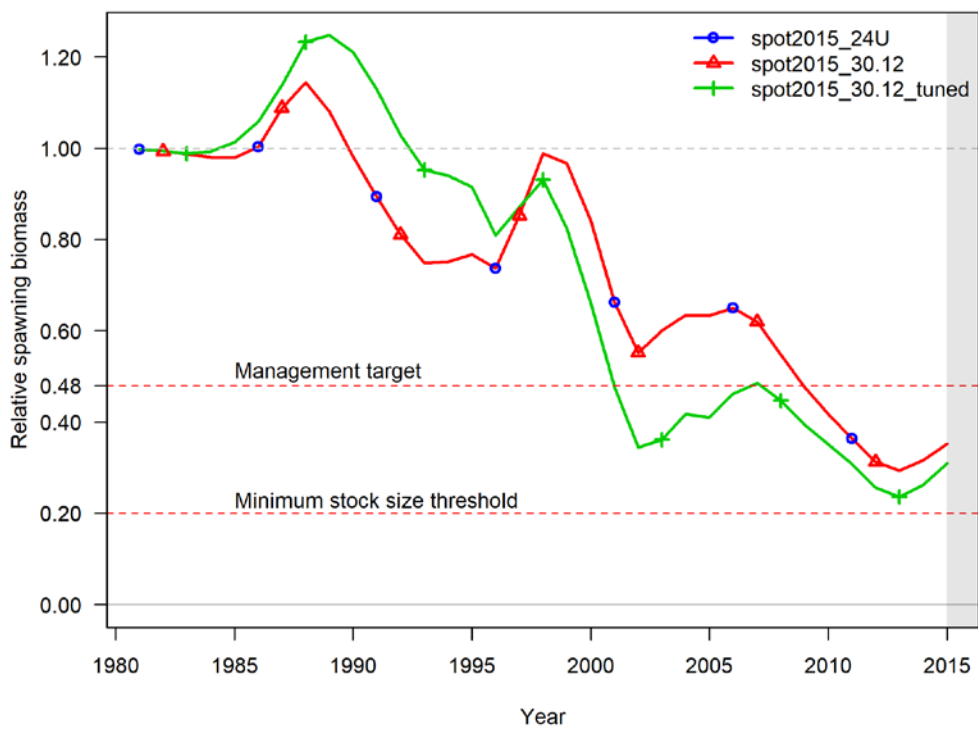


Figure 11.2. Comparison of the relative spawning biomass time series for the 2015 assessment (spot2015_24U – in blue), and a model converted to SS-V3.30.12 (spot2015_30.12 in blue) and this same model tuned using the latest tuning procedures (spot2015_30.12_tuned – in green).

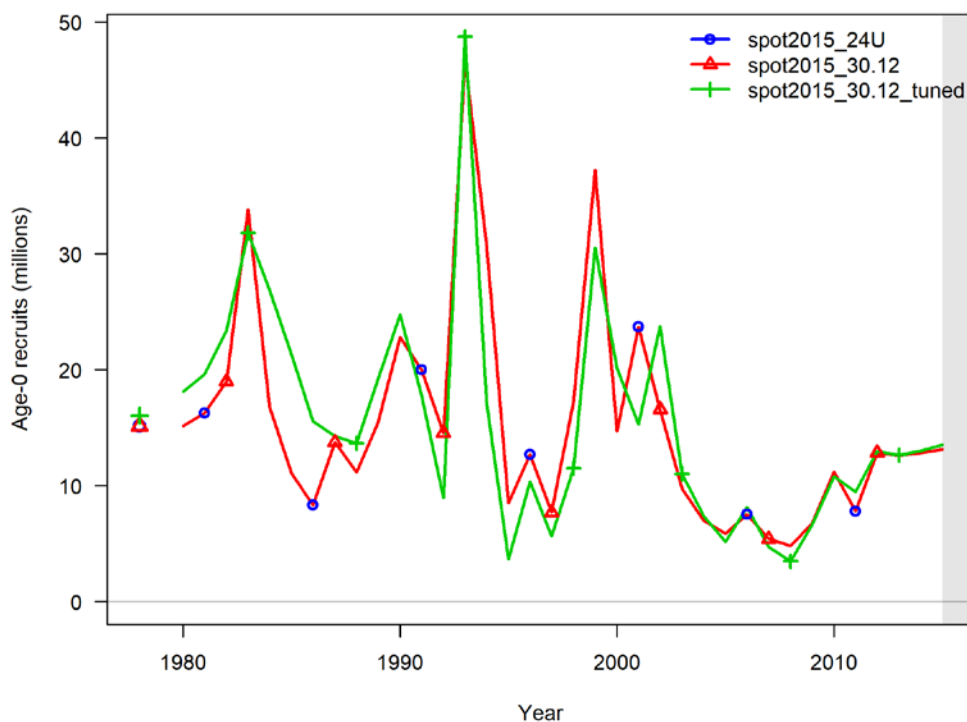


Figure 11.3. Comparison of the recruitment time series for the 2015 assessment (spot2015_24U – in blue), and a model converted to SS-V3.30.12 (spot2015_30.12 in blue) and this same model tuned using the latest tuning procedures (spot2015_30.12_tuned – in green).

11.2.2.1 Tuning method

Iterative rescaling (reweighting) of input and output CVs or input and effective sample sizes is a repeatable method for ensuring that the expected variation of the different data streams is comparable to what is input (Pacific Fishery Management Council, 2016). Most of the indices (CPUE, surveys and composition data) used in fisheries underestimate their true variance by only reporting measurement or estimation error and not including process error.

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size is equal to the effective sample size calculated by the model. In SS-V3.30 it is possible to estimate an additional standard deviation parameter to add to the input CVs for the abundance indices (CPUE).

1. Set the standard error for the relative abundance indices (CPUE or FIS) to their estimated standard errors to the standard deviation of a loess curve fitted to the original data - which will provide a more realistic estimate to that obtained from the original statistical analysis. SS-V3.30 then allows an estimate to be made for an additional adjustment to the relative abundance variances appropriately.

An automated iterative tuning procedure was used for the remaining adjustments. For the recruitment bias adjustment ramps:

2. Adjust the maximum bias adjustment and the start and finish bias adjustment ramps as predicted by SS-V3.30 at each step.

For the age and length composition data:

3. Multiply the initial sample sizes by the sample size multipliers for the age composition data using Francis weights (Francis, 2011).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length composition data.
5. Repeat steps 2 - 4, until all are converged and stable (with proposed changes $< 1 - 2\%$).

This procedure constitutes current best practice for tuning assessments.

11.2.3 Inclusion of new data: 2015-2017

Starting from the converted 2015 base case model, additional data from 2015-2018 were added sequentially to develop a preliminary base case for the 2018 assessment:

1. Change final assessment year to 2017, add trawl catches scaled to total CDR catch from 2015 to 2017 (add_Catch_2017). Changes in the catch time series before 2015 have not been examined here but will be considered as a sensitivity in the subsequent assessment report.
2. Add CPUE to 2017 (add_CPUE_2017) provided in Sporcic and Haddon (2018b).
3. Add the FIS index from 2016 (add_FIS_2016). The length frequency data from the FIS have not been included in the preliminary base case presented here, but will be considered as a sensitivity in the subsequent assessment report.
4. Add updated discard fraction estimates 1985 - 2017 (add_Discards_2017).
5. Add updated length frequency data to 2017 (add_Lengths_2017).
6. Add updated age error matrix and age-at-length data to 2017 (add_Ageing_2017).
7. Change the final year for which recruitments are estimated from 2012 to 2015 (extend_Rec_2015).
8. Re-tune using latest model tuning protocols, including Francis weighting on lengths and ages (preliminary_Basecase_2017).

Inclusion of three years of new catch data resulted in relatively small changes to the historical estimates of recruitment and the female spawning biomass (Figure 11.4 to Figure 11.6), with the most recent three years showing increases to spawning biomass. The inclusion of the new CPUE showed a large decline in spawning biomass prior to 2000, along with a moderate decline from the mid-2000s (Figure 11.4), while recruitment also fell prior to 1990 and after 2008 (Figure 11.6). The addition of the 2016 FIS biomass estimates had no impact on the assessment and are not shown. Adding the revised discard time series saw spawning biomass decline over the entire time series (Figure 11.4). Over most of the time series this decline was small, larger declines were seen between 1985 and 1998 and after 2014. Recruitment estimates were mostly unchanged (Figure 11.6), with small declines prior to 1988 and after 2009. Inclusion of the length-composition and conditional age-at-length data along with the retuning had minimal impact on either the magnitude or trend of spawning biomass or on recruitment (Figure 11.4 to Figure 11.6). Extending the period of estimated recruitment by 3 years, led to average recruitment in 2013 and 2014 and slightly above average recruitment in 2015 (Figure A 11.7). While the increase in estimated recent recruitment is welcome given the decade of below average recruitment for this species, they should be treated with some caution as recent recruitment events are often over-

estimated in stock assessment models. Model diagnostics and residual plots for the preliminary base case assessment are provided in Appendix A.

This preliminary base case assessment estimates that the projected 2019 spawning stock biomass will be 33% of virgin stock biomass (projected assuming 2017 catches in 2018), however, it was predicted to be below 20% between 2013 and 2017. The previous assessment (Day et al., 2015) estimated depletion to be 40% at the start of 2016 and the trajectory of depletion did not drop below 20%. The female equilibrium spawning biomass in 1980 is estimated to be 9,672 t and in 2019 the female spawning biomass is projected to be 3,210 t. It should also be noted that the increase in spawning stock biomass observed since 2017 may be overly optimistic because the last two years of recruitment are not estimated in the model and instead mean recruitment used.

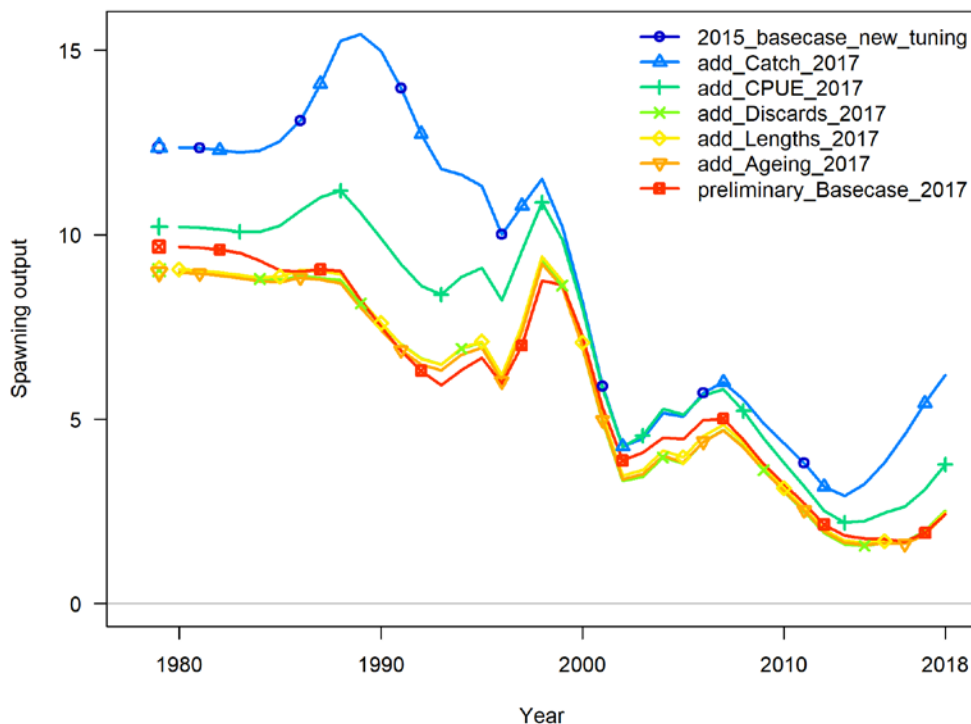


Figure 11.4. Comparison of the absolute spawning biomass time series for the 2015 assessment model converted to SS-V3.30.12 with various bridging models leading to a proposed 2018 tuned base case model (preliminary_Basecase_2017).

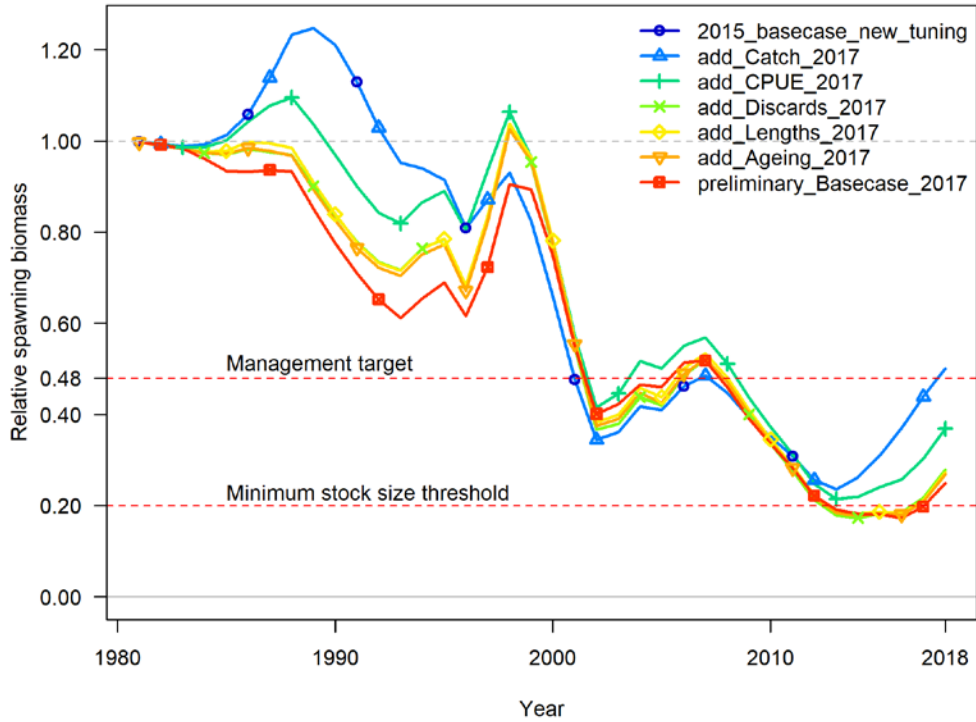


Figure 11.5. Comparison of the relative spawning biomass time series for the 2015 assessment model converted to SS-V3.30.12 with various bridging models leading to a proposed 2018 tuned base case model (preliminary_Basecase_2017).

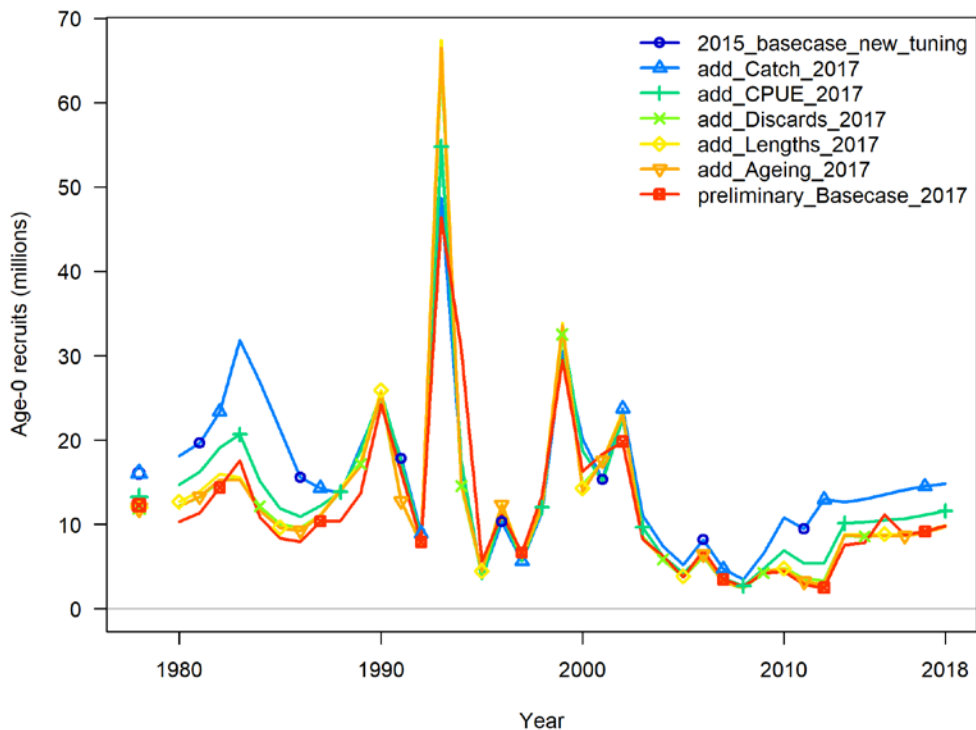


Figure 11.6. Comparison of the recruitment time series for the 2015 assessment model converted to SS-V3.30.12 with various bridging models leading to a proposed 2018 tuned base case model (preliminary_Basecase_2017).

11.2.4 Likelihood profiles

As stated by Punt (2018), likelihood profiles are a standard component of the toolbox of applied statisticians. They are most often used to obtain a 95% confidence interval. Many stock assessments “fix” key parameters such as M and steepness based on *a priori* considerations. Likelihood profiles can be used to evaluate whether there is evidence in the data to support fixing a parameter at a chosen value. If the parameter is within the entire range of the 95% confidence interval, this provides no support in the data to change the fixed value. If the fixed value is outside the 95% confidence interval, it would be reasonable for a review panel to ask why the parameter was fixed and not estimated, and if the value is to be fixed, on what basis and why should what amounts to inconsistency with the data be ignored. Integrated stock assessments include multiple data sources (e.g., commonly catch-rates, length-compositions, and age-compositions) that may be in conflict, due for example to inconsistencies in sampling, but more commonly owing to incorrect assumptions (e.g., assuming that catch-rates are linearly related to abundance), i.e. model-misspecification. Likelihood profiles can be used as a diagnostic to identify these data conflicts (Punt, 2018).

Standard parameters to consider are natural mortality (M), steepness (h) and the logarithm of the unfished recruitment ($\ln R_0$).

The silver Warehou assessment has used a fixed value for natural mortality (M) of 0.30 yr^{-1} since the 2007 assessment (Tuck and Punt 2017) when likelihood profiles showed that values for M larger than the base-case value of 0.25 yr^{-1} were preferred. In August 2007, SlopeRAG endorsed a move to a natural mortality value of 0.30 yr^{-1} for the base-case assessment. A likelihood profile on M was run for this assessment and is shown in Figure 11.7. While the optimal value of M from the profile likelihood is 0.38 yr^{-1} , the data sources provide little information about the most appropriate value of M so we suggest to keep M at 0.30 yr^{-1} for the base case assessment of silver warehou in 2018.

The likelihood profile for steepness, h , (Figure 11.8) suggests that there is little information in the model that can be used to inform this parameter, which is fixed at 0.75 in the model. The length and discard data suggest a higher steepness while the other data sources suggest a lower value of h . We suggest retaining a fixed value of 0.75 for steepness in the base case model.

The likelihood profile for the logarithm of the unfished recruitment ($\ln R_0$; Figure 11.9) shows how the different data sources contribute to the estimated $\ln R_0$ of 9.41. The CPUE indices (labelled ‘Survey’ in Figure 11.9), along with the length composition prefer a lower value of $\ln R_0 = 9$. The age composition and discards are consistent with the estimated value while the recruitment prefers a higher value.

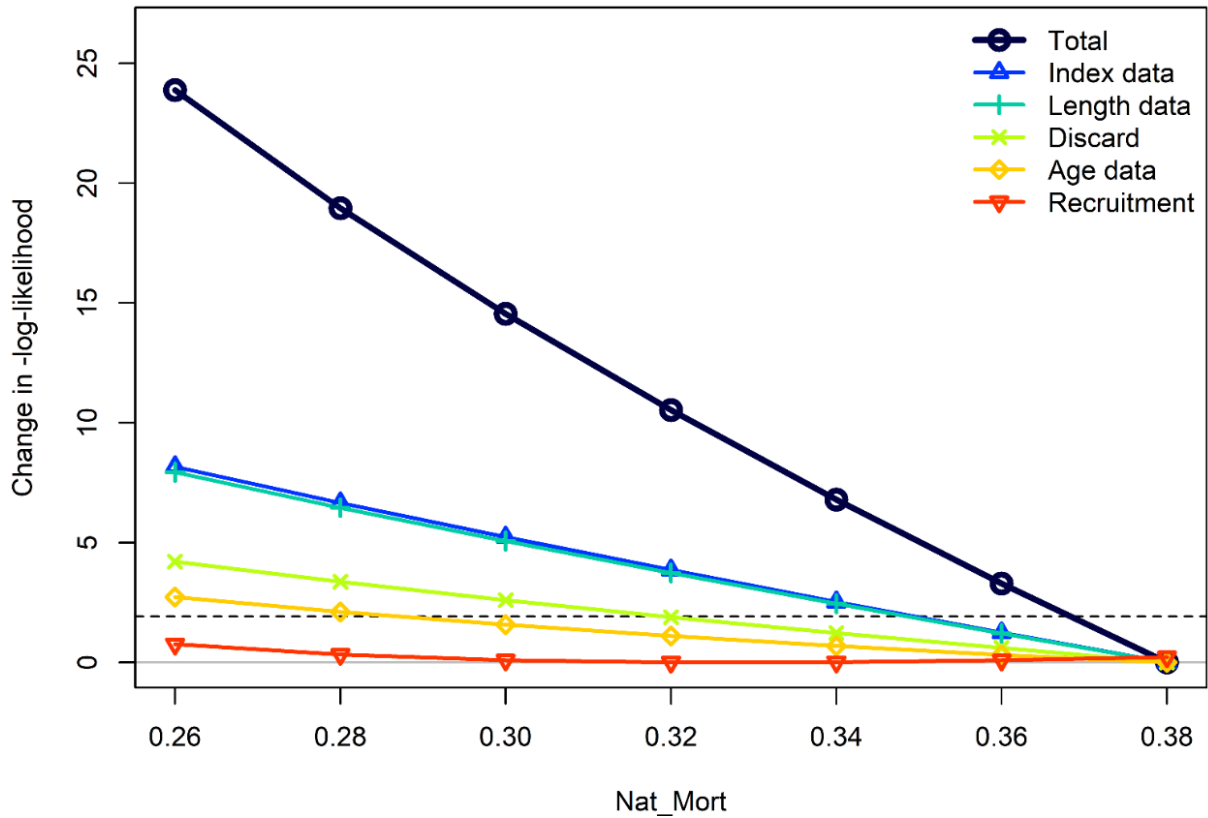


Figure 11.7. The likelihood profile for natural mortality. The fixed value for M is 0.30.

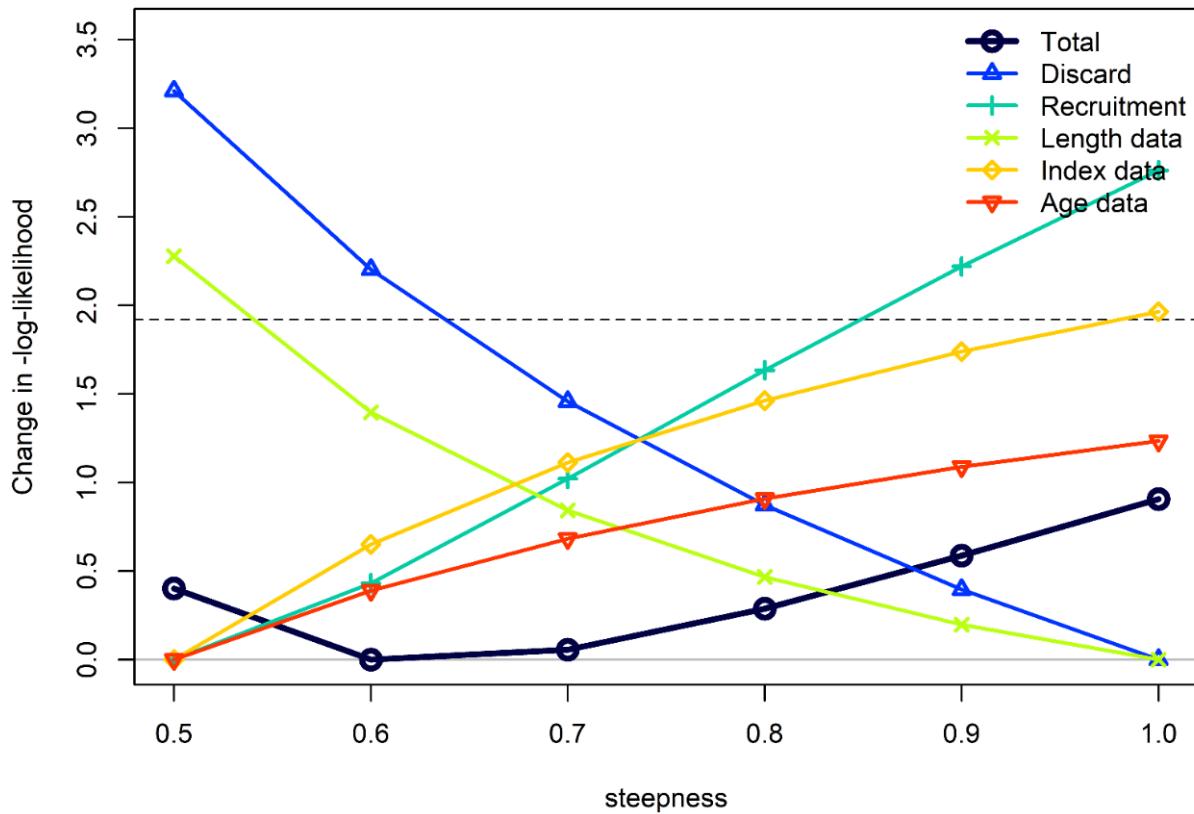


Figure 11.8. The likelihood profile for steepness. The fixed value for h is 0.75.

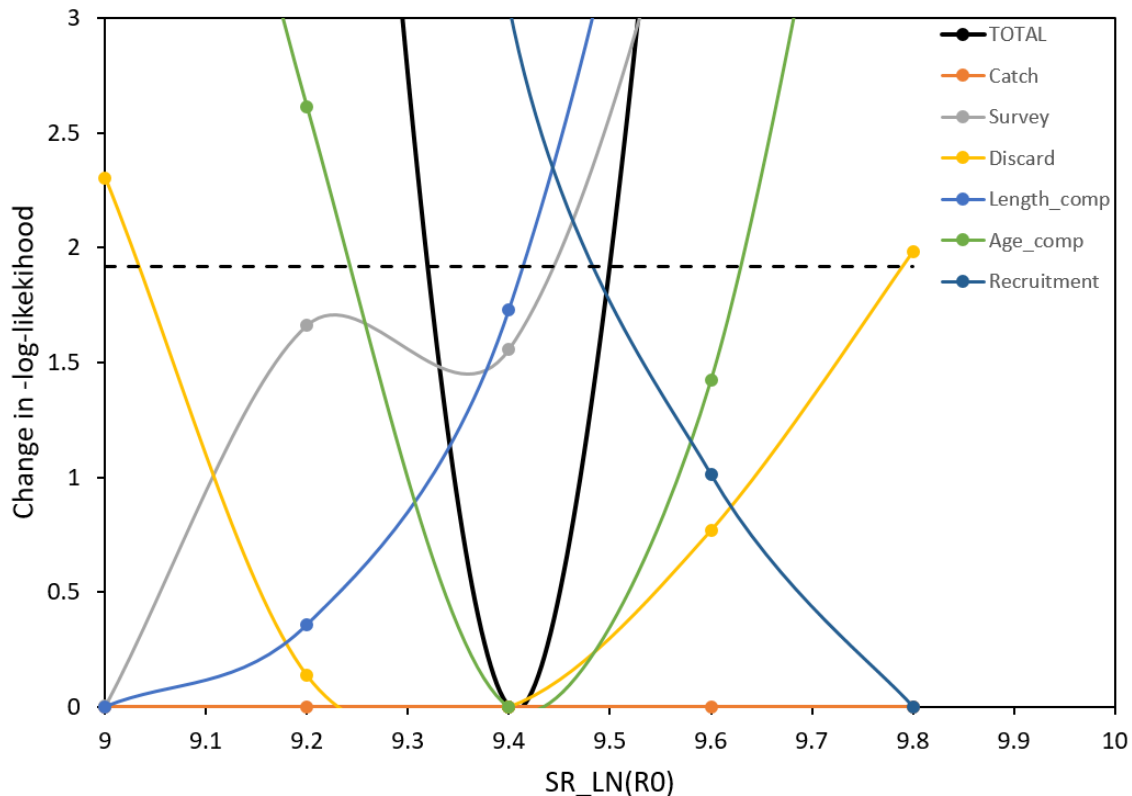


Figure 11.9. The likelihood profile for $\ln R_0$. This parameter is estimated by the model to be 9.41.

11.2.5 Future work and unresolved issues

There is still the need to investigate the impact of the change in the discard estimation method and examine the influence of the revised time series of catches prior to 2015 (Thomson et al., 2018). Two other sensitivities relating to the Fishery Independent Survey (FIS) would also be useful (i) excluding all FIS data and (ii) including FIS length frequency data and estimating selectivity for the FIS fleet.

Potential problems with the CPUE series that provide abundance indices have been identified by Sporcic and Haddon (2018a). While the preliminary base case model fits to the indices of abundance are good, particularly in recent years, there are serious concerns about the reliability of the CPUE time-series in both the east and west. Sporcic and Haddon (2018a) note that:

... there have been transitional periods in the time-series of CPUE. This urgently needs more attention because this may imply that CPUE may no longer be acting as a valid index of relative abundance through time.

Given the potential bias in the abundance indices, results from this assessment should be treated with caution until the impacts of the transitional period in the time series of CPUE on the assessment has been evaluated.

11.3 Acknowledgments

Age data was provided by Kyne Krusic-Golub (Fish Ageing Services), ISMP and AFMA logbook and CDR data were provided by John Garvey (AFMA). Franzis Althaus, Mike Fuller and Roy Deng Claudio Castillo-Jordán (CSIRO) pre-processed the data. Andre Punt, Geoff Tuck, Sandra Curin-Osorio, Malcolm Haddon, Robin Thomson and Miriana Sporcic are thanked for helpful discussions on this work. Ian Taylor, Richard Methot and Chantel Wetzel (NOAA Fisheries) are thanked for all the Stock Synthesis support and advice. Malcolm Haddon provided useful code for auto-tuning, and Athol Whitten provided useful R code for organising plots.

11.4 References

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http://www.pcouncil.org/wp-content/uploads/2017/01/Stock_Assessment_ToR_2017-18.pdf.
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11.5 Appendix A

A.1 Preliminary base case diagnostics

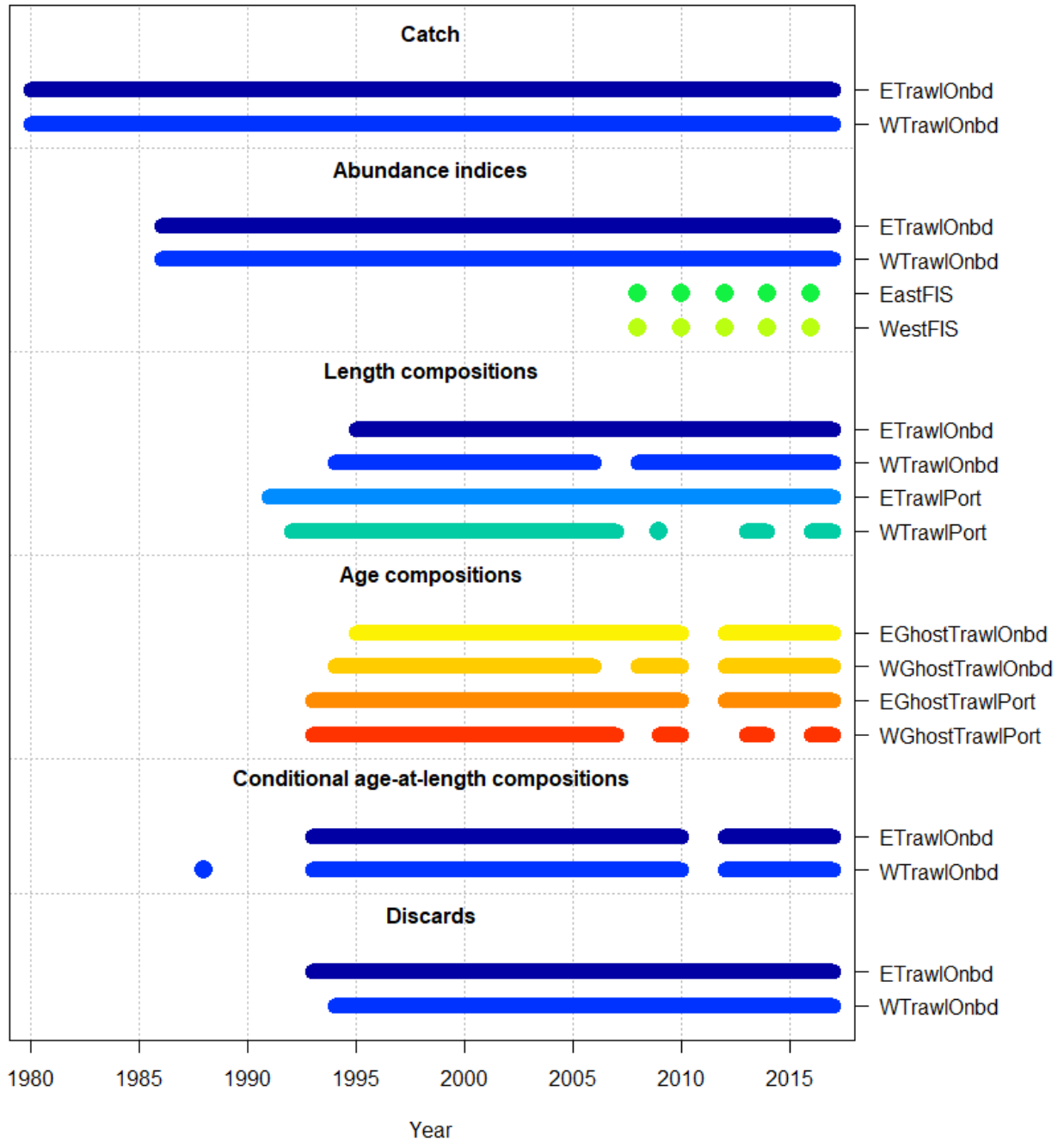


Figure A 11.1. Summary of data sources for silver warehou stock assessment.

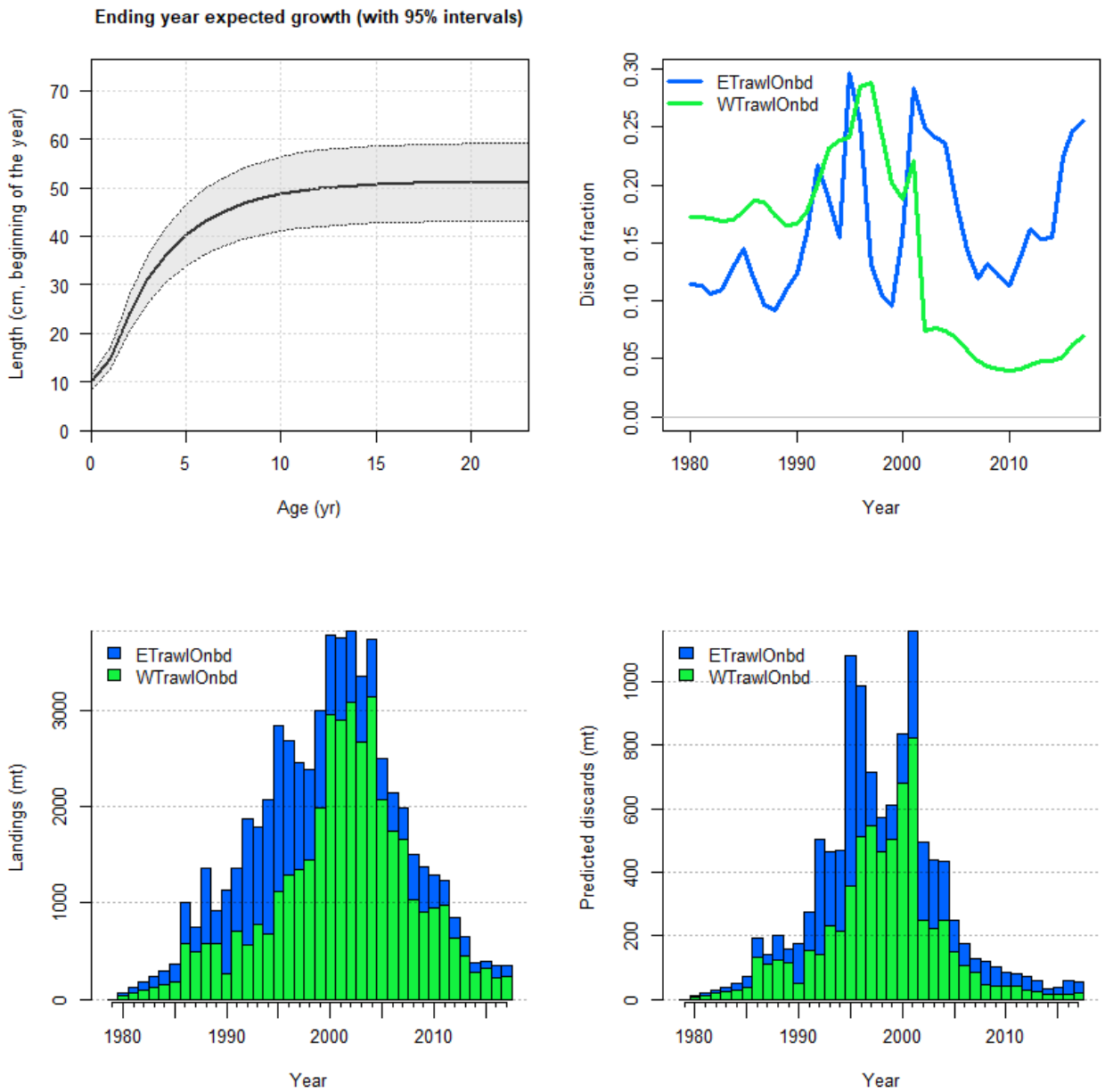


Figure A 11.2. Growth, discard fraction estimates, landings by fleet and predicted discards by fleet for silver warehou.

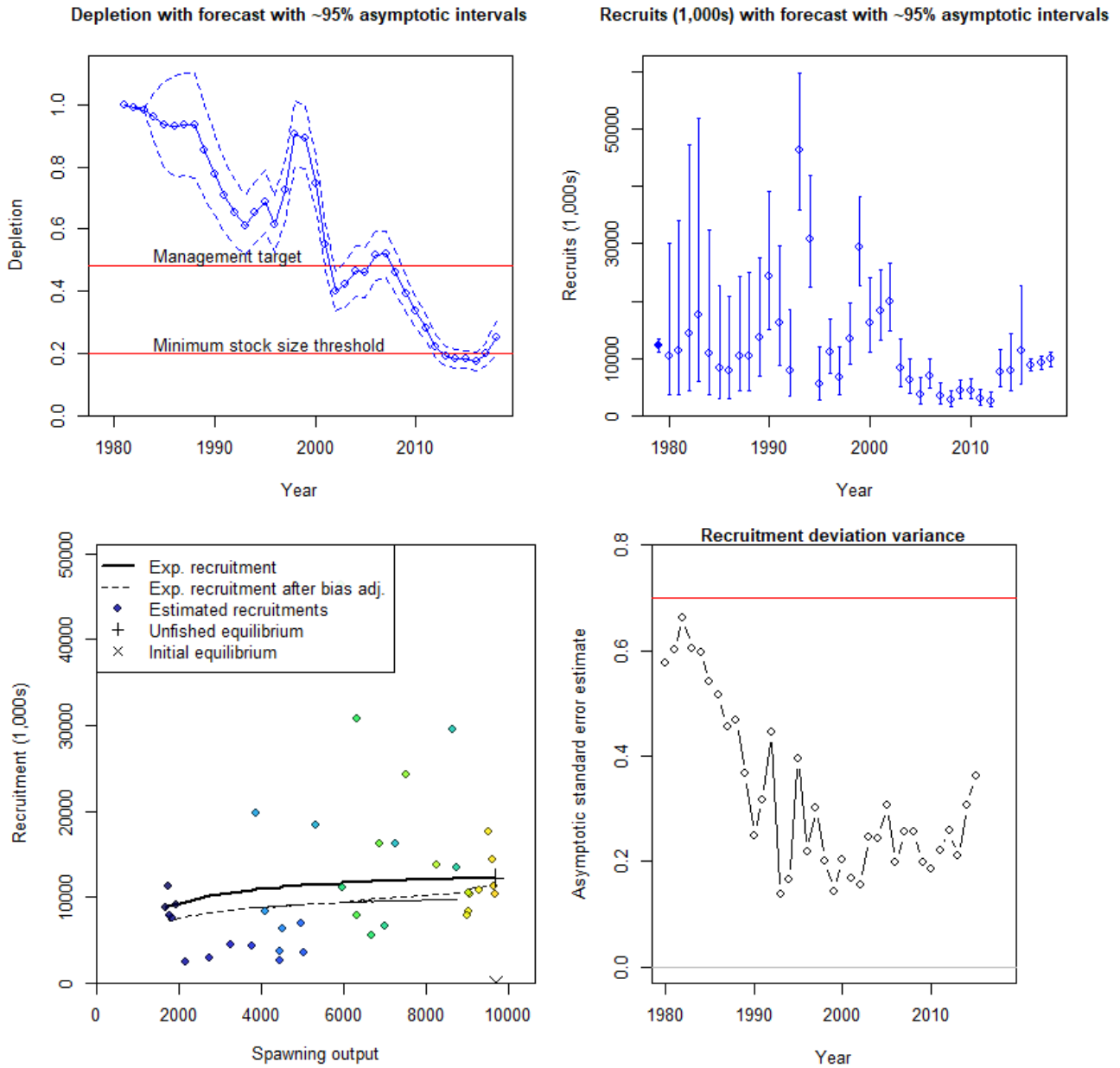


Figure A 11.3. Time series showing depletion of spawning biomass with confidence intervals, recruitment estimates with confidence intervals, stock recruitment curve and recruitment deviation variance check for silver warehou.

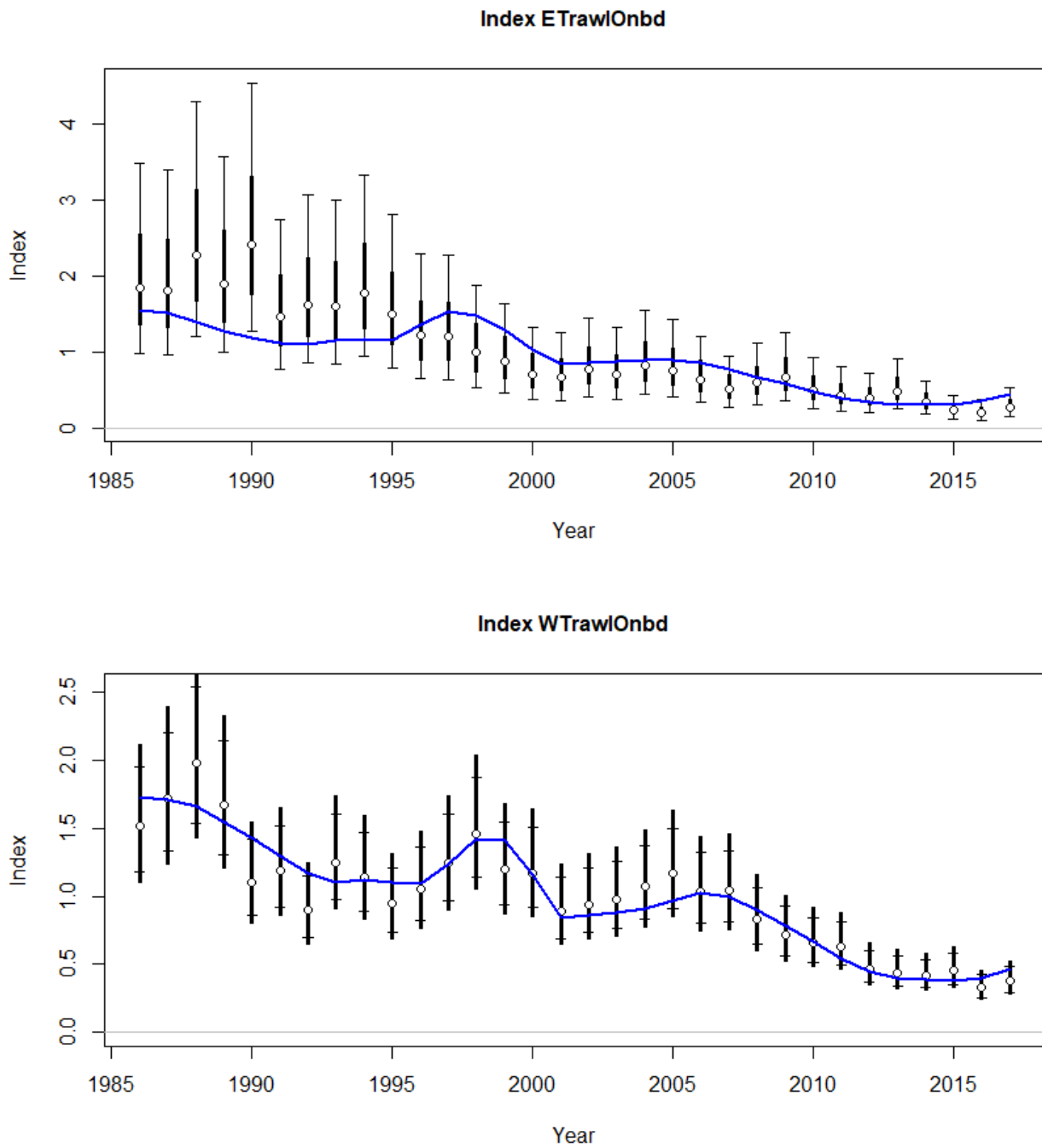


Figure A 11.4. Fits to CPUE by fleet for silver warehou: East and West trawl.

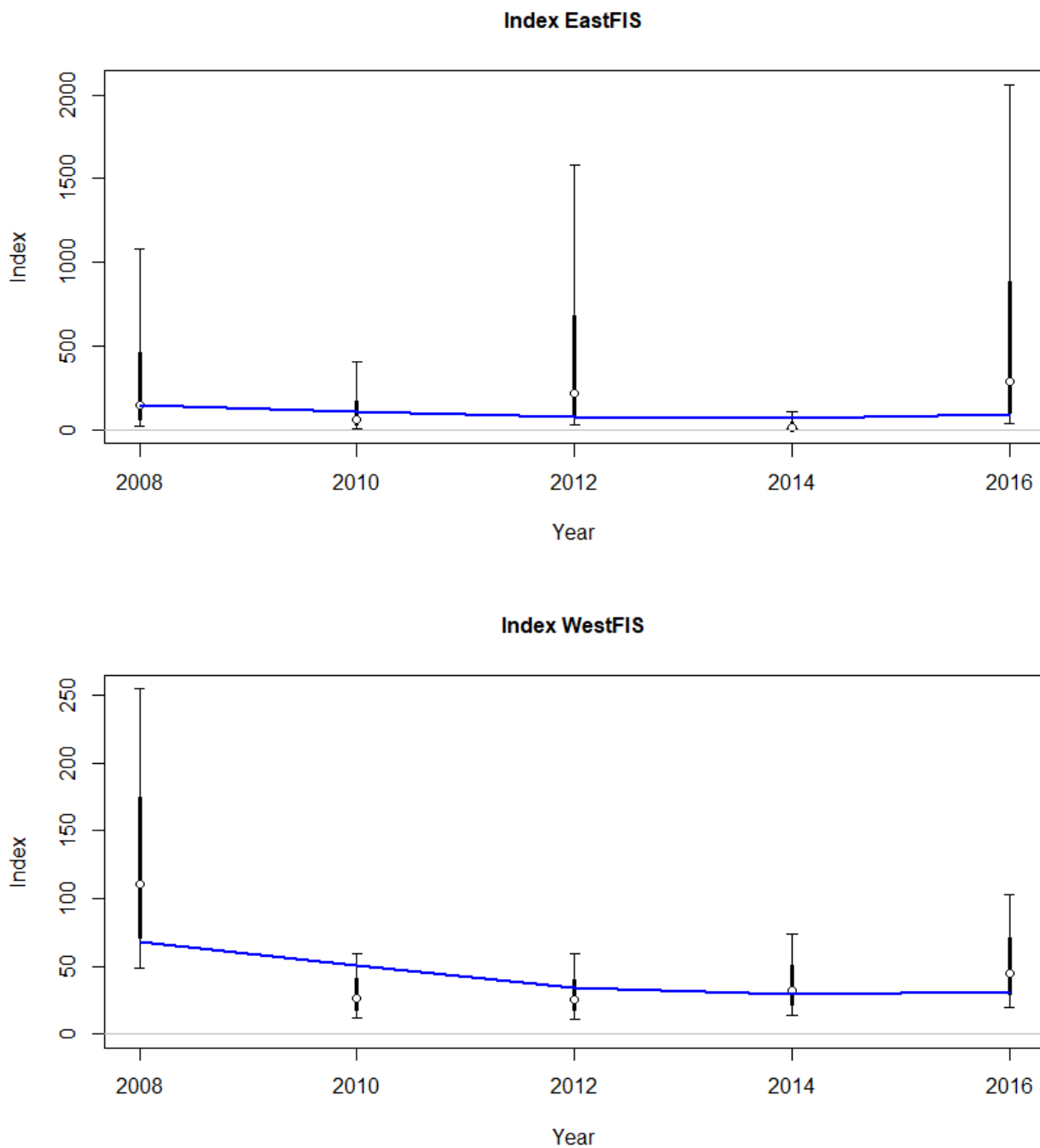


Figure A 11.5. Fits to CPUE from the FIS fleets for silver warehou.

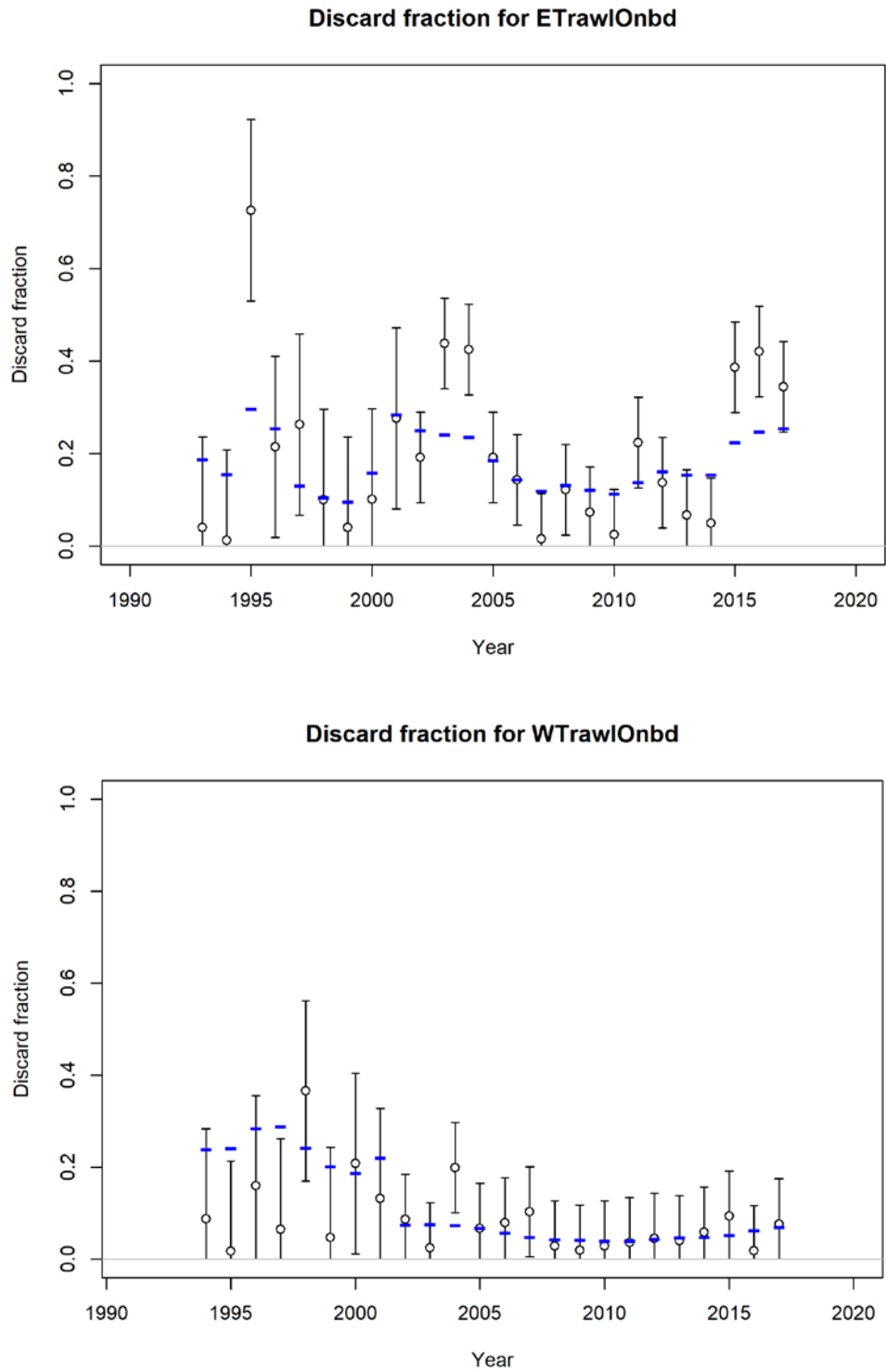


Figure A 11.6. Fits to discard rates for the east and west trawl fleets for silver warehou.

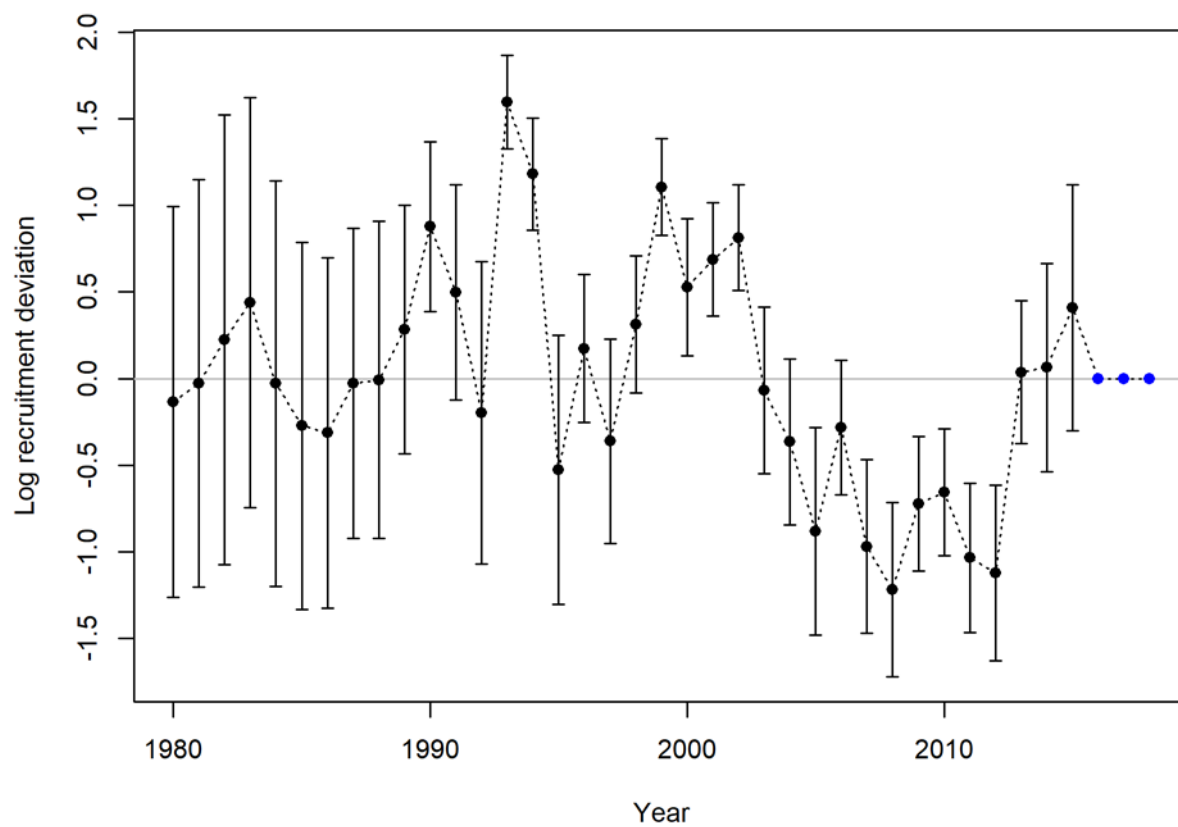


Figure A 11.7. Fits to discard rates (trawl) and recruitment deviations for silver warehou.

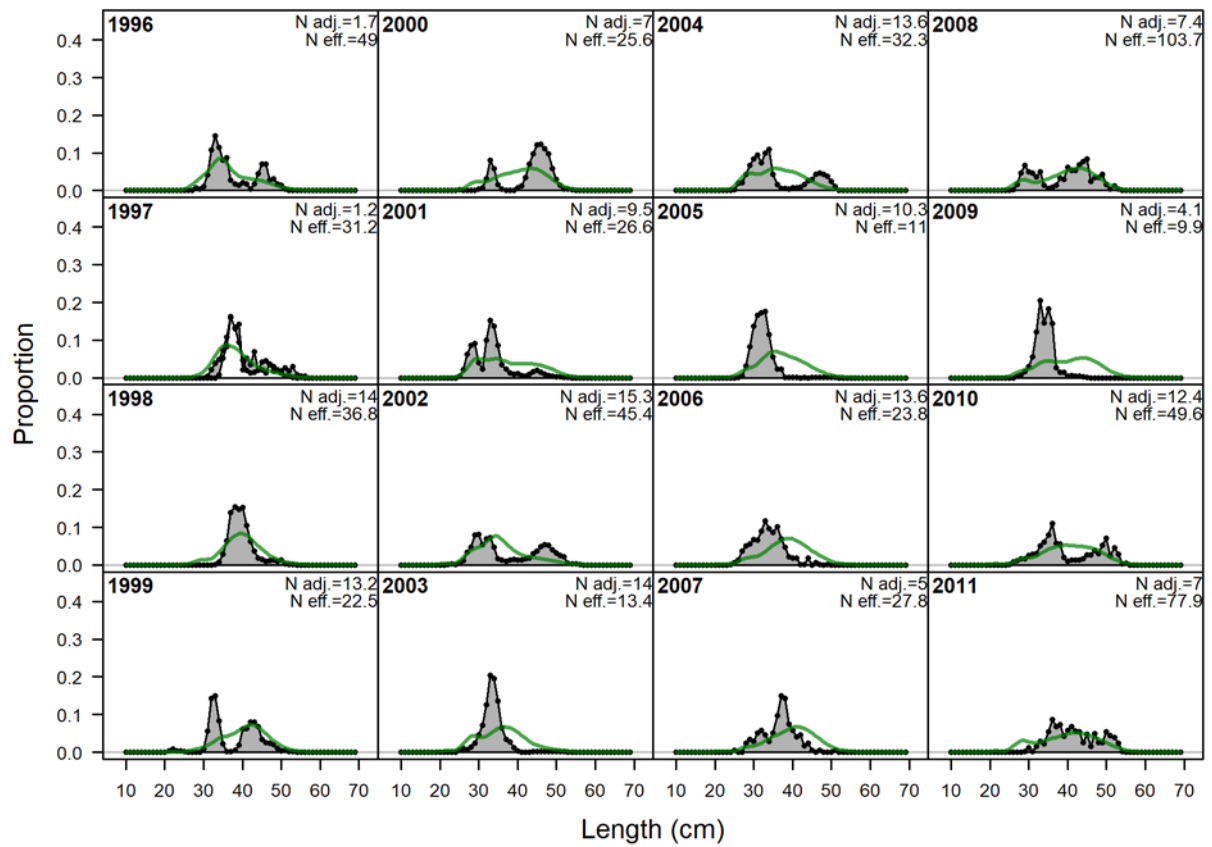


Figure A 11.8. Silver warehou length composition fits: east trawl onboard retained (page 1).

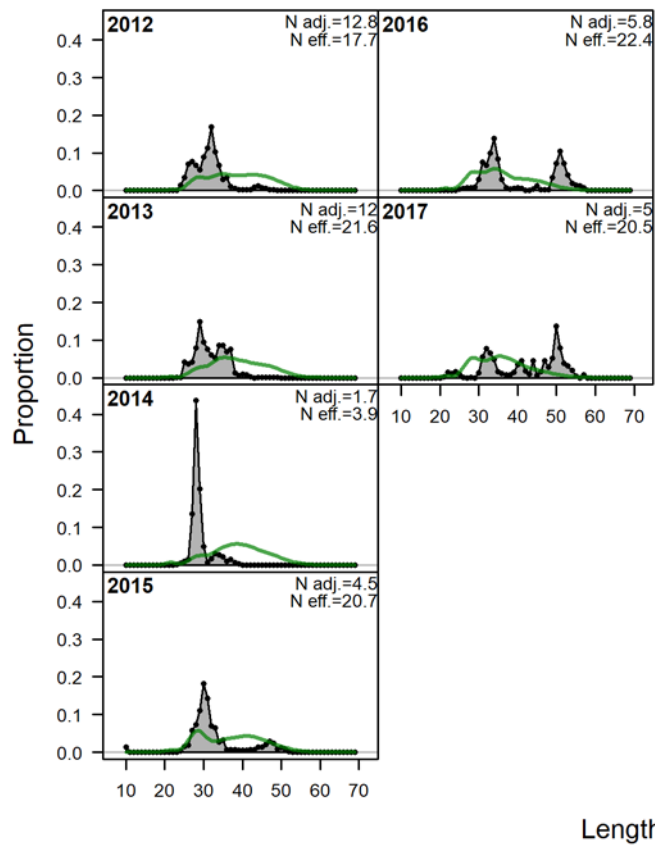


Figure A 11.9. Silver warehou length composition fits: east trawl onboard retained (page 2).

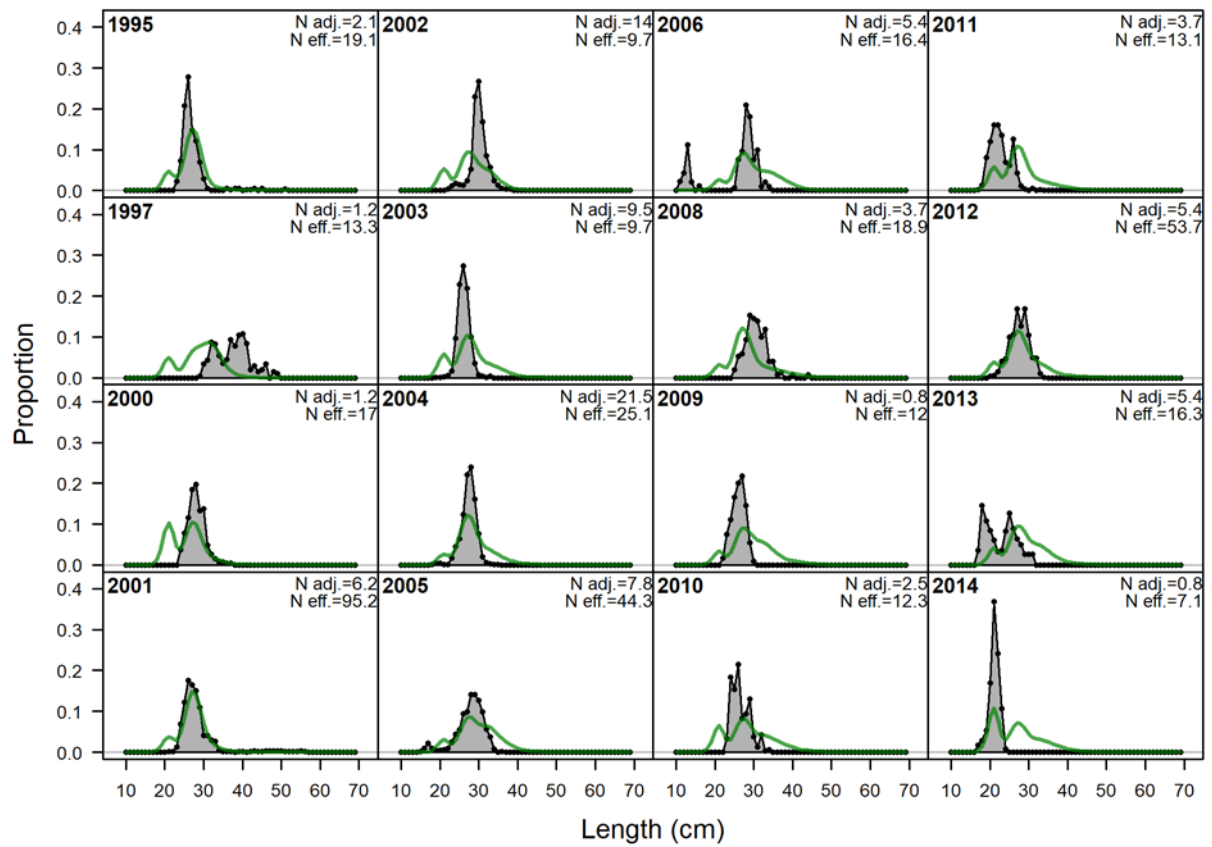


Figure A 11.10. Silver warehou length composition fits: east trawl onboard discarded (page 1).

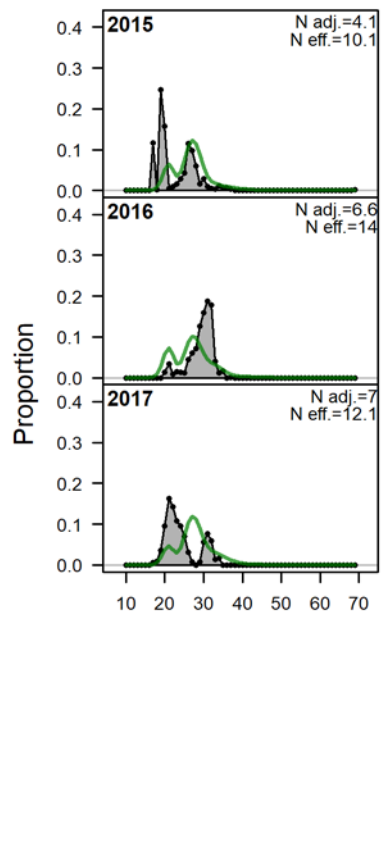


Figure A 11.11. Silver warehou length composition fits: east trawl onboard discarded (page 2).

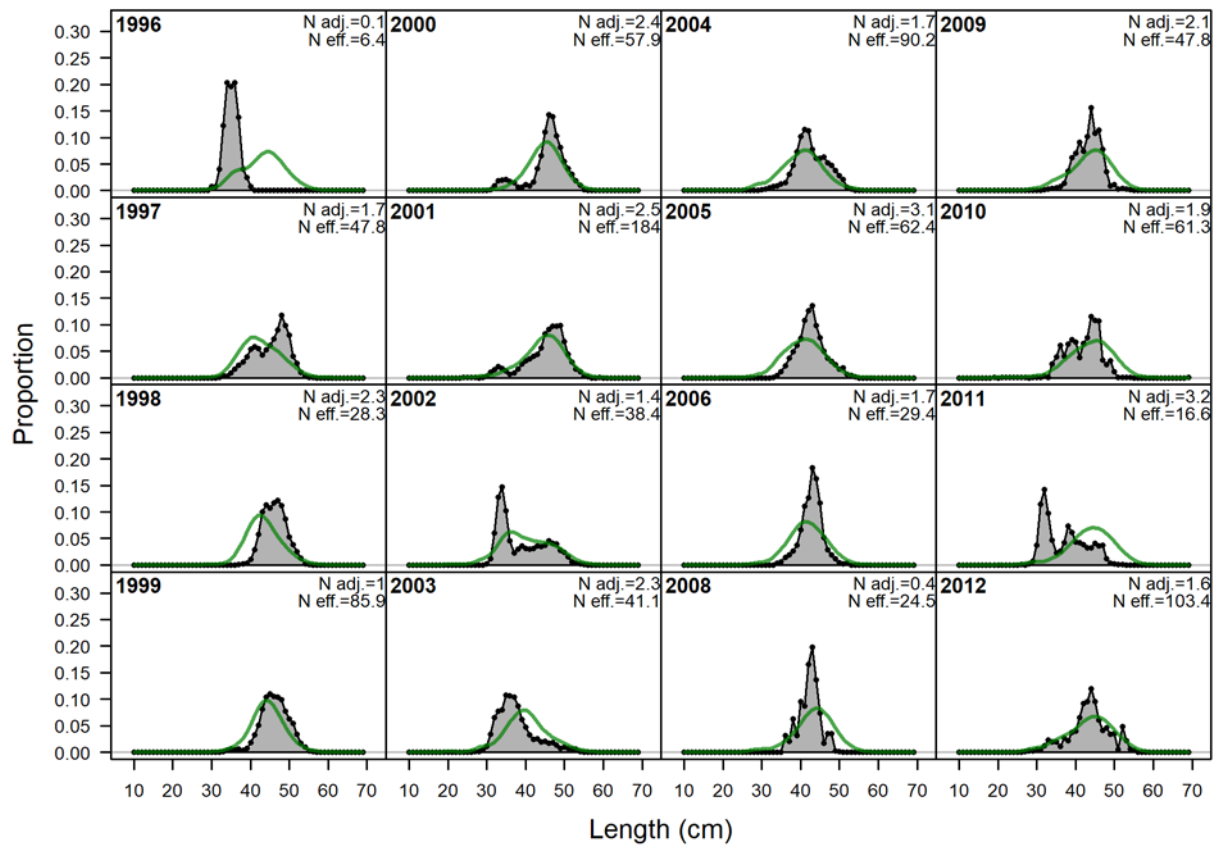


Figure A 11.12. Silver warehou length composition fits: west trawl onboard retained (page 1).

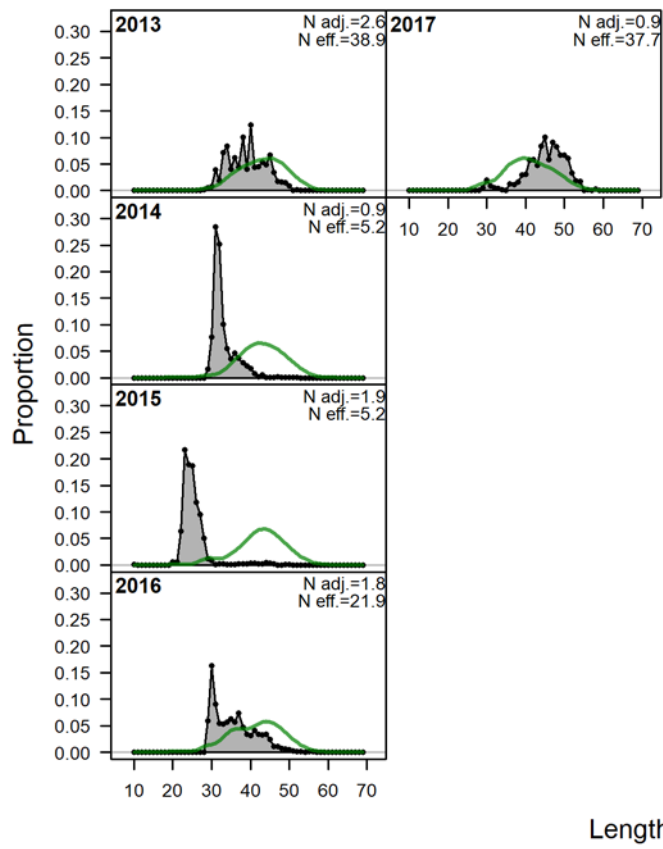


Figure A 11.13. Silver warehou length composition fits: west trawl onboard retained (page 2).

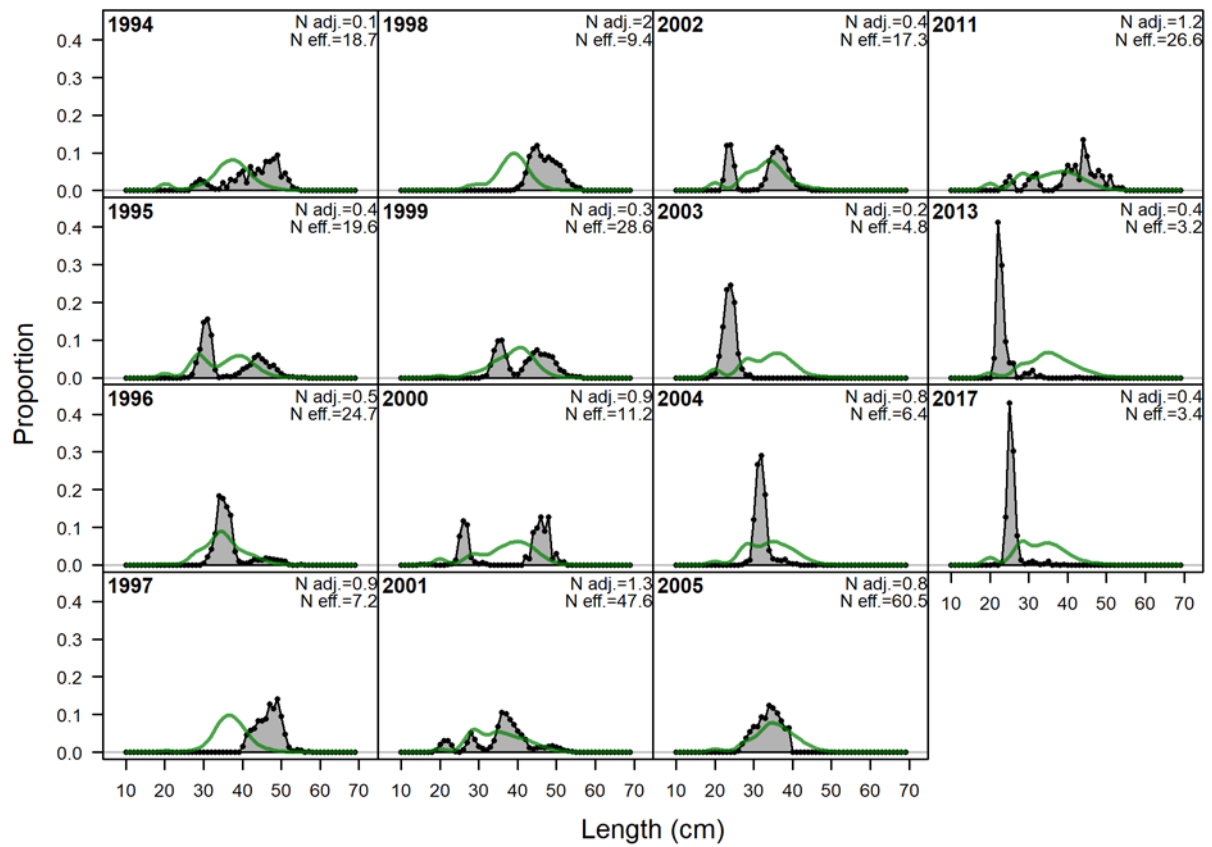


Figure A 11.14. Silver warehou length composition fits: west trawl onboard discarded.

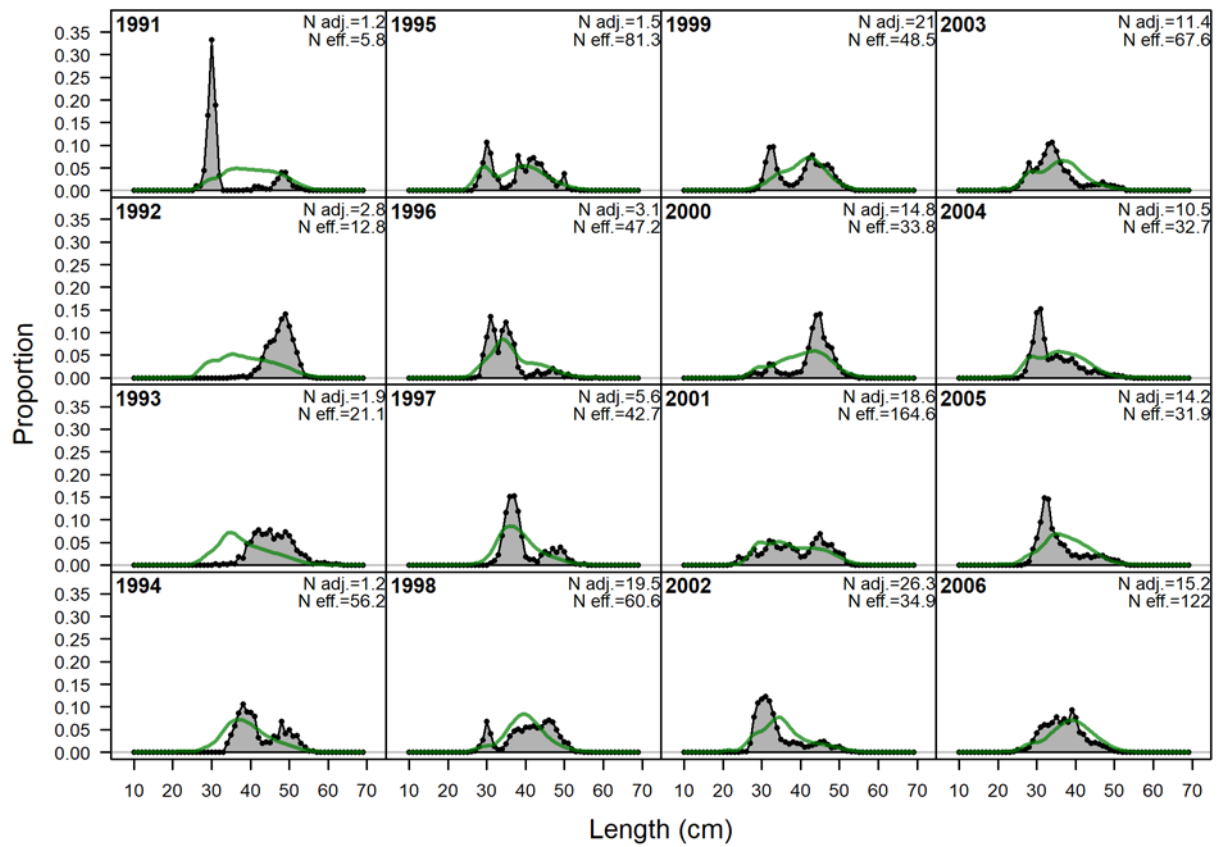


Figure A 11.15. Silver warehou length composition fits: east trawl port retained (page 1).

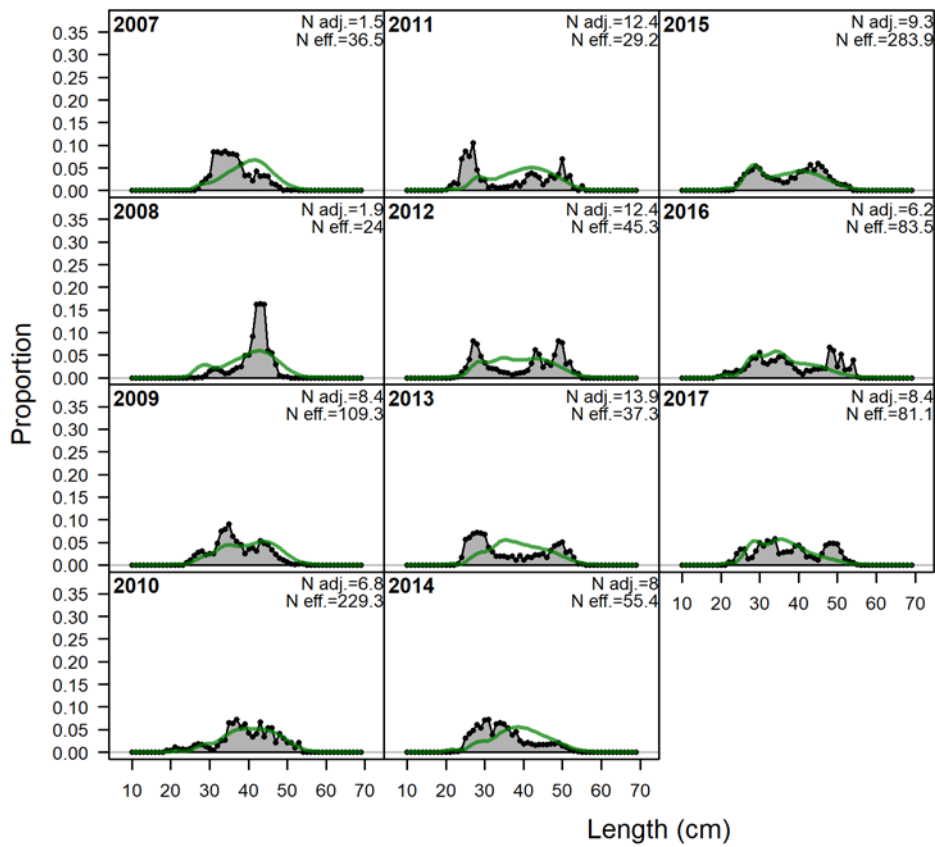


Figure A 11.16. Silver warehou length composition fits: east trawl port retained (page 2).

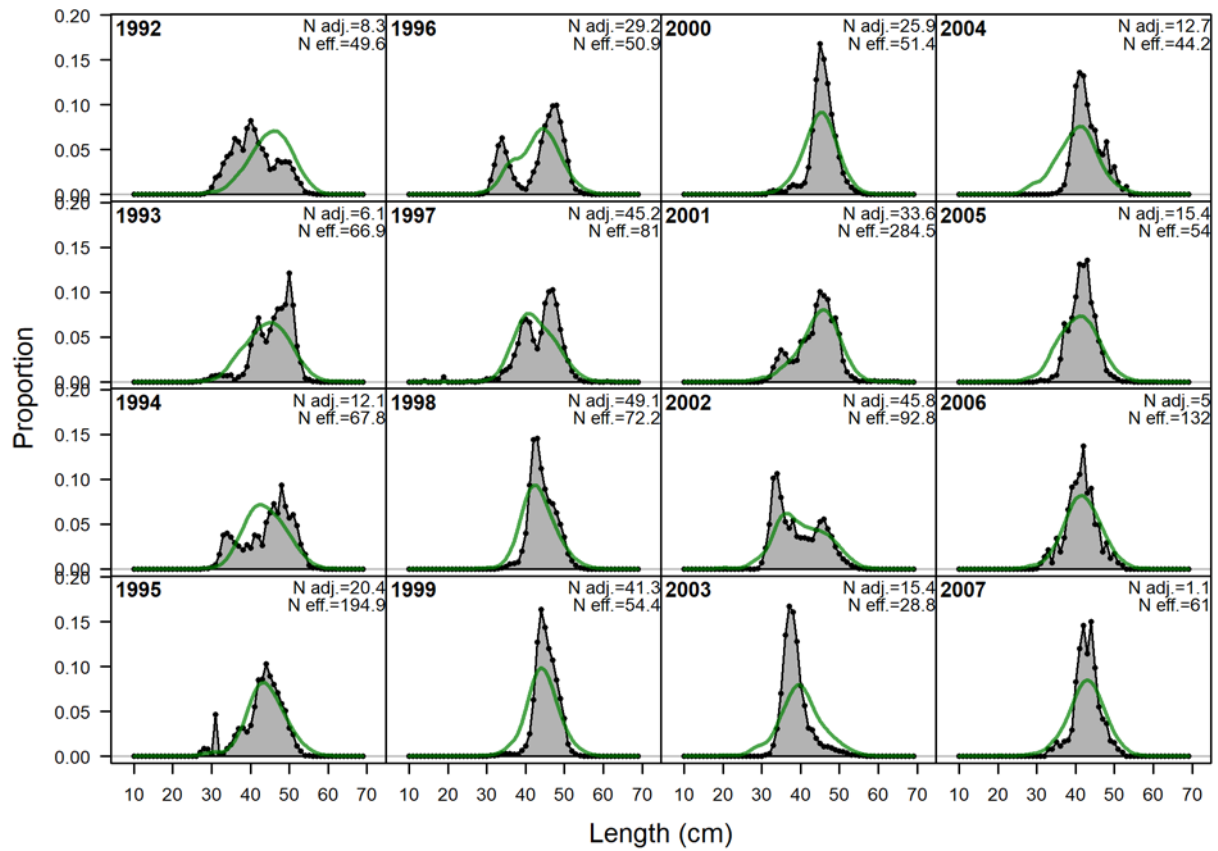


Figure A 11.17. Silver warehou length composition fits: west trawl port retained (page 1).

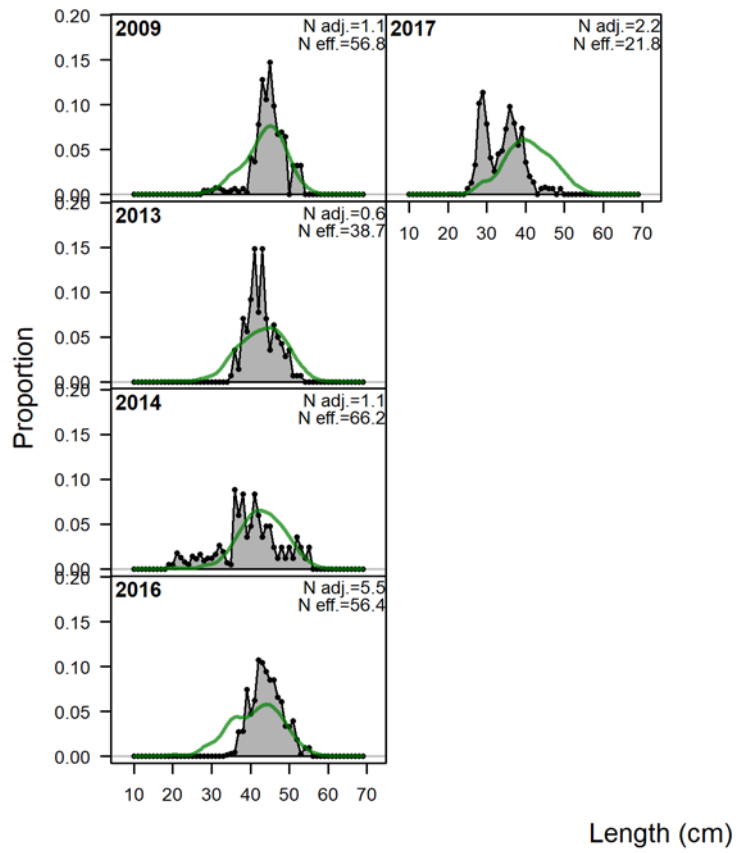


Figure A 11.18. Silver warehou length composition fits: west trawl port retained (page 2).

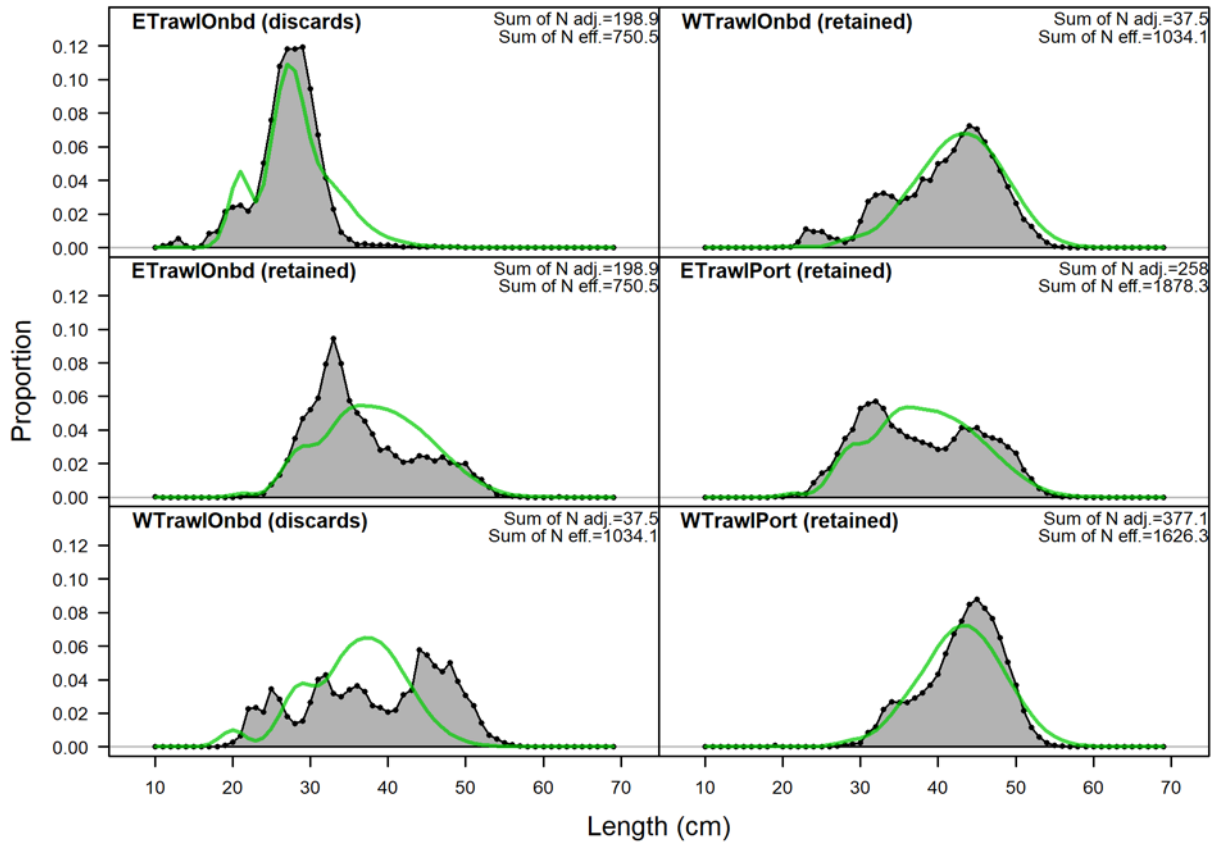


Figure A 11.19. Aggregated fits (over all years) to the length compositions for silver warehou displayed by fleet.

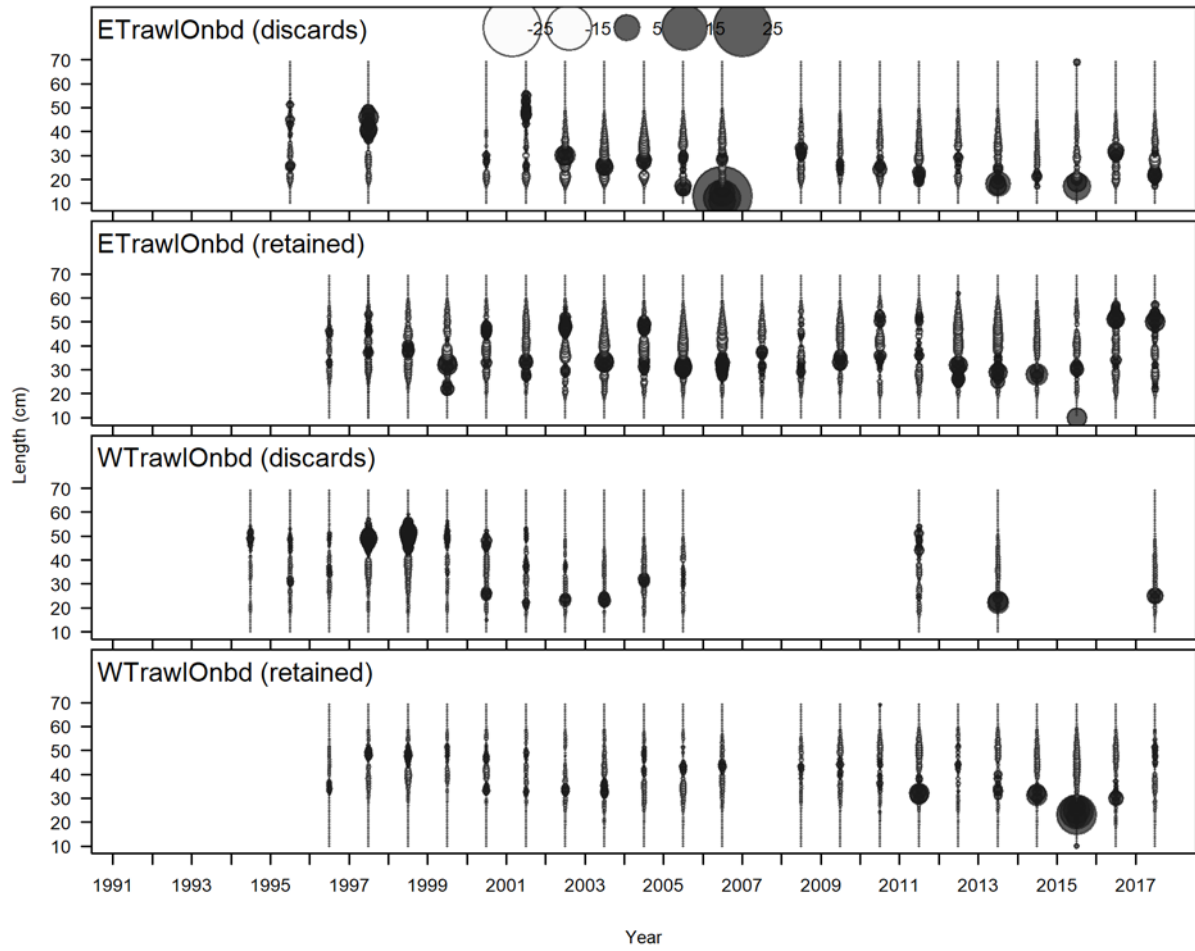


Figure A 11.20. Pearson residuals, comparing across fleets (page 1).

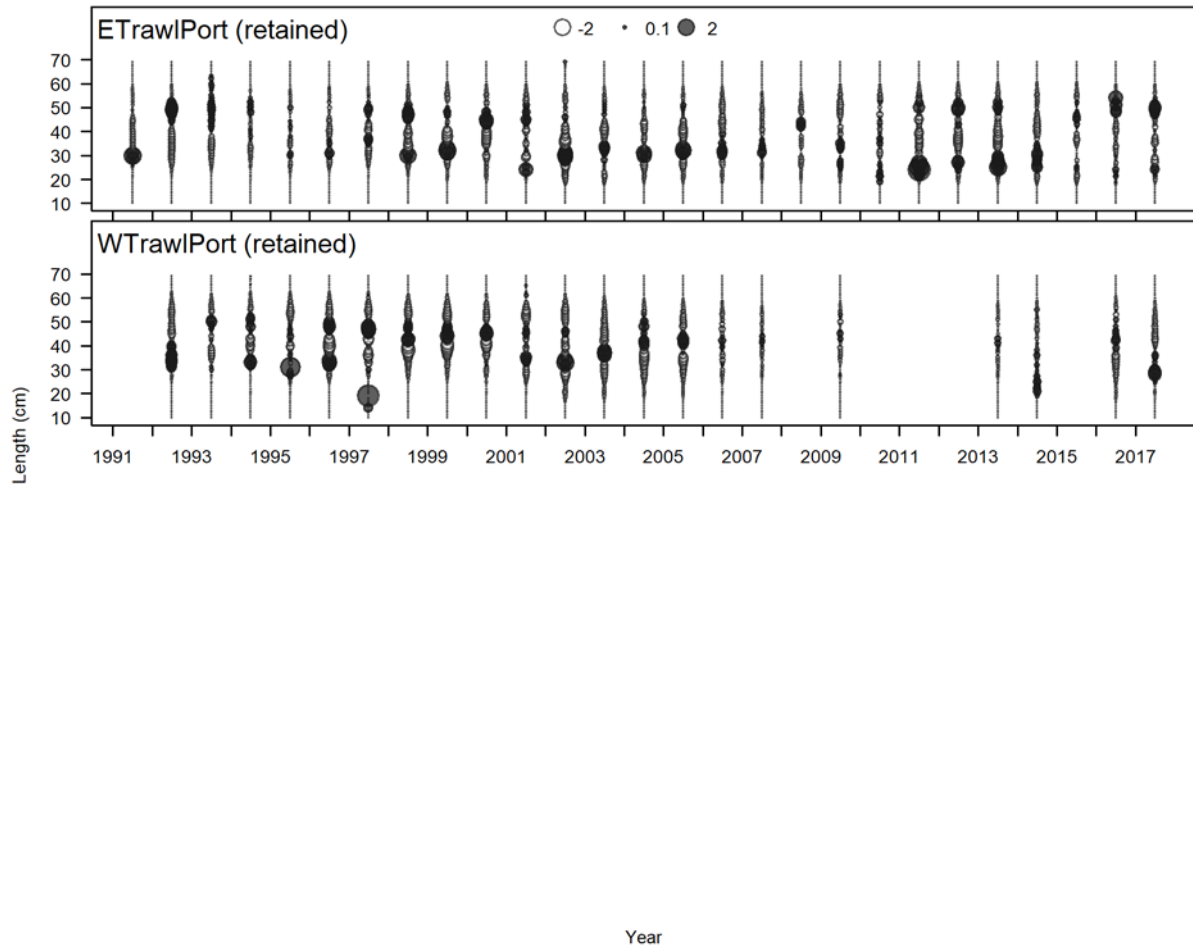


Figure A 11.21. Pearson residuals, comparing across fleets (page 2).

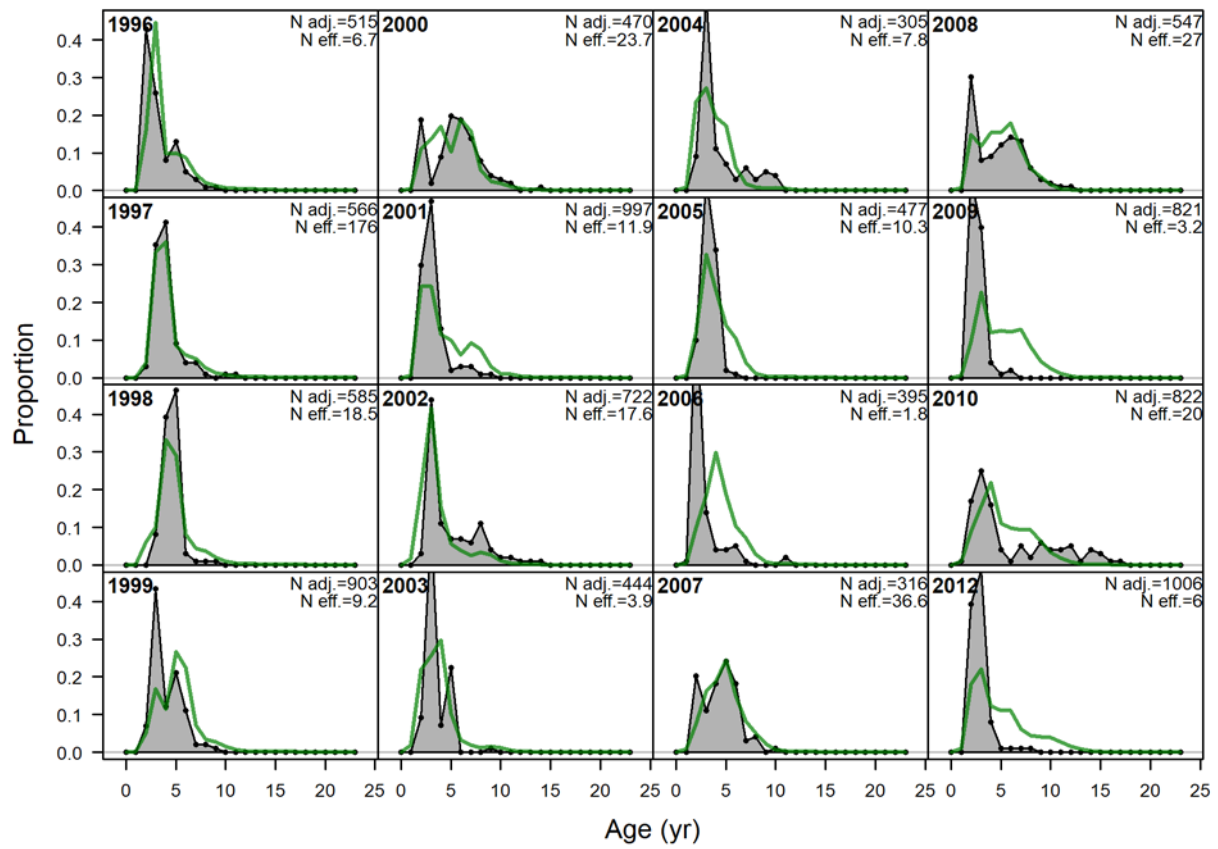


Figure A 11.22. Silver warehou implied fits to age: east ghost trawl onboard retained (page 1).

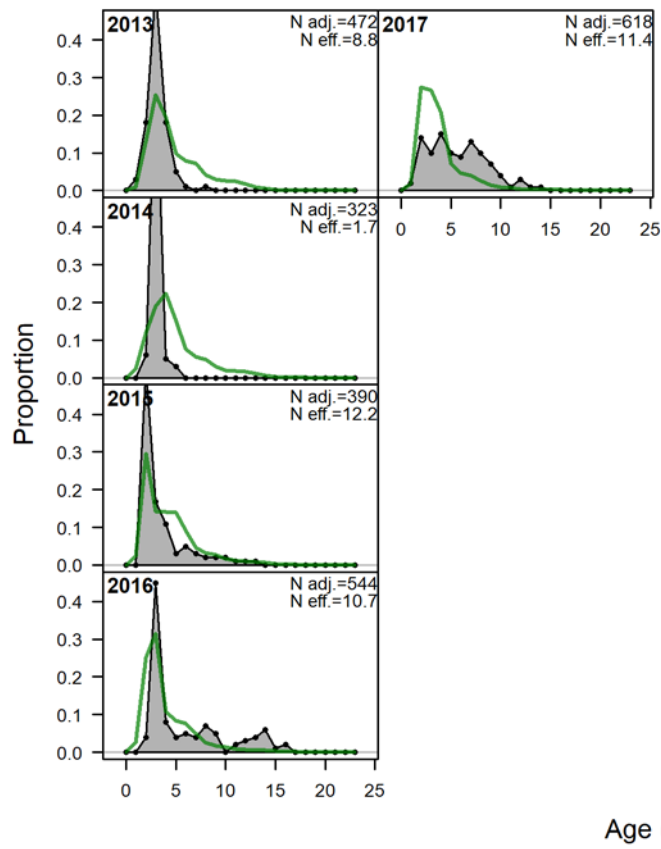


Figure A 11.23. Silver warehou implied fits to age: east ghost trawl onboard retained (page 2).

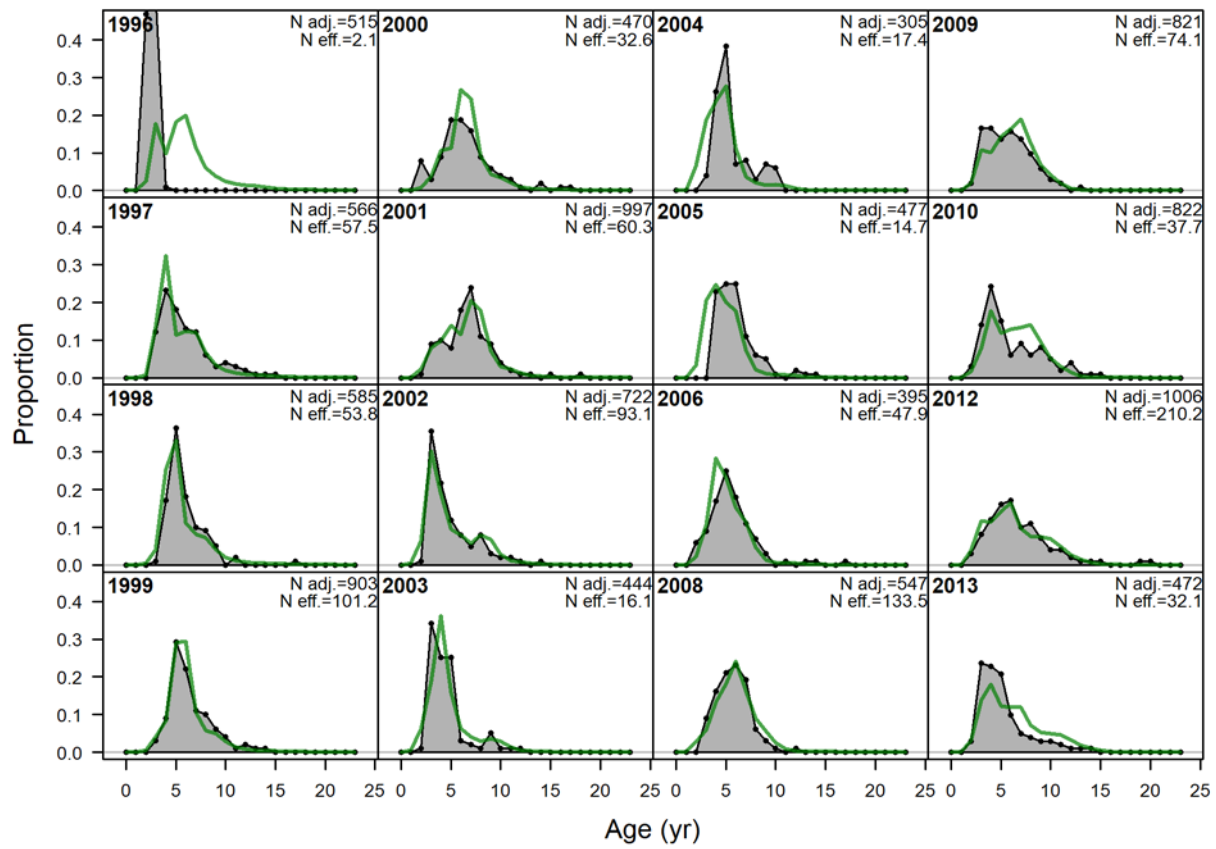


Figure A 11.24. Silver warehou implied fits to age: west ghost trawl onboard retained (page 1).

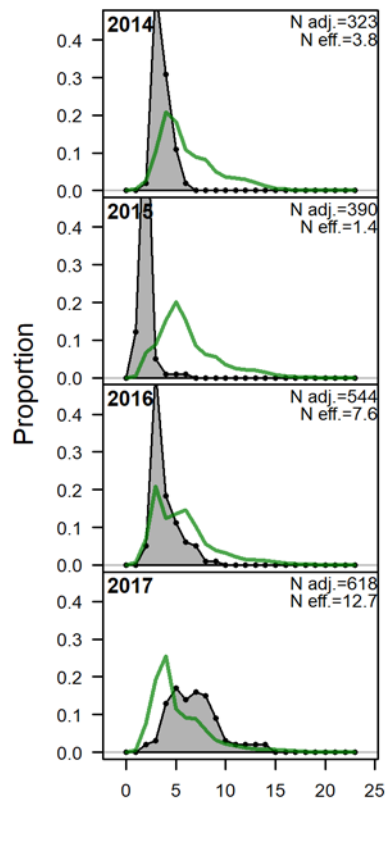


Figure A 11.25. Silver warehou implied fits to age: west ghost trawl onboard retained (page 2).

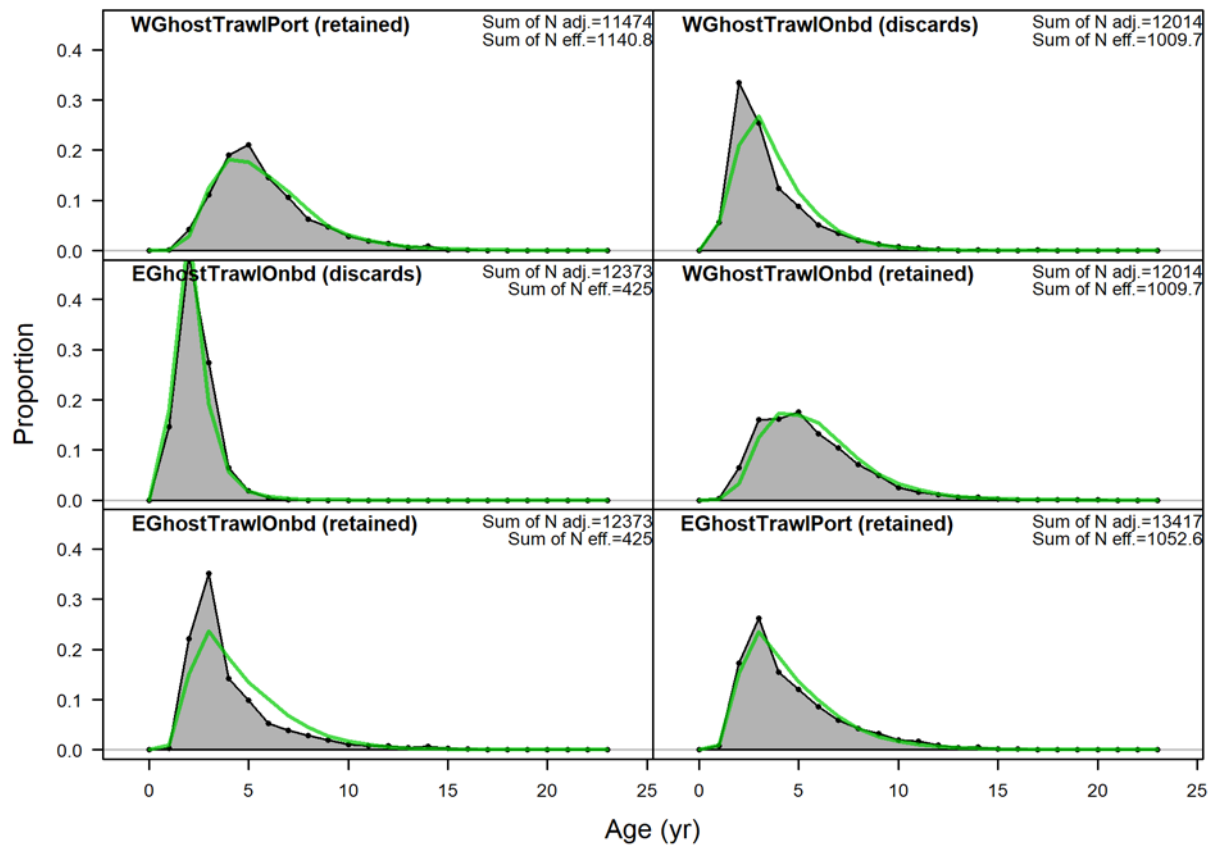


Figure A 11.26. Aggregated fits (over all years) to the implied age compositions for silver warehou displayed by fleet.

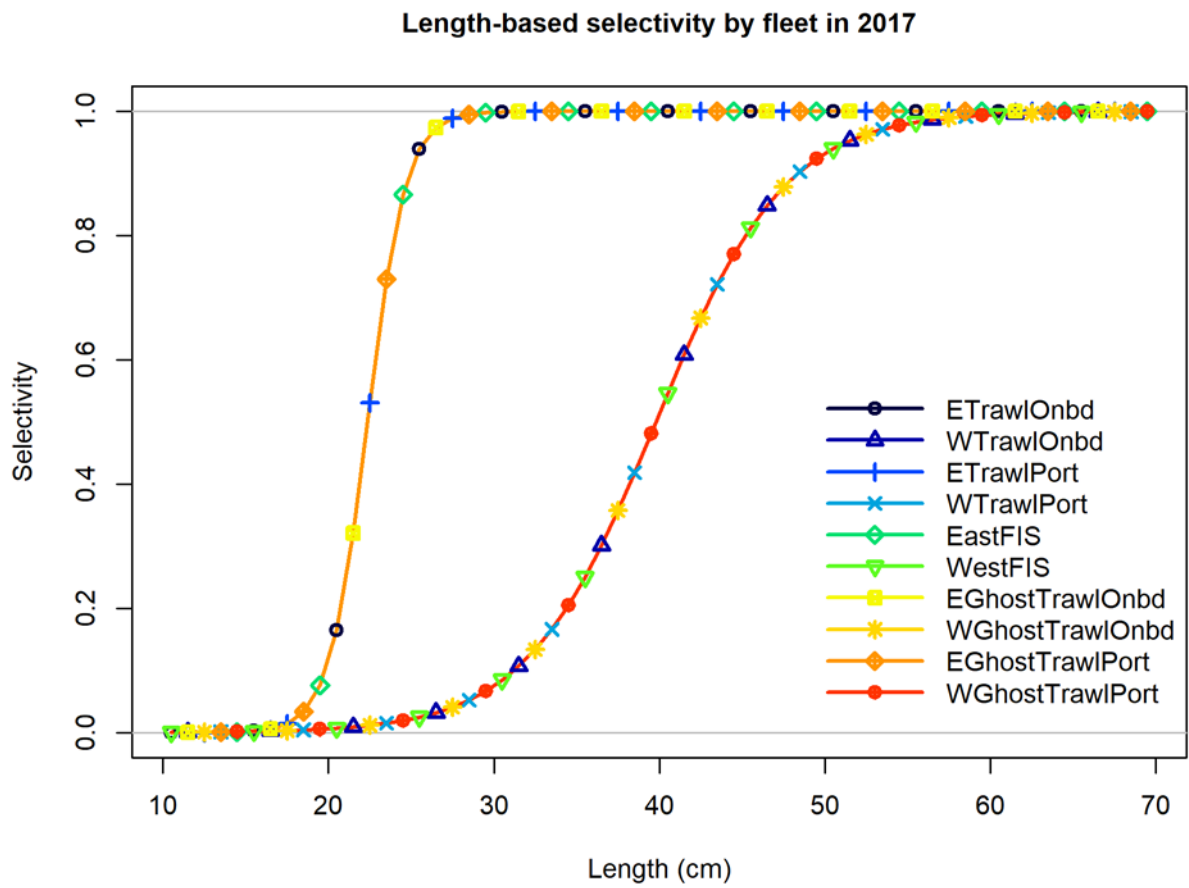


Figure A 11.27. Fits to selectivity for silver warehou fleets.

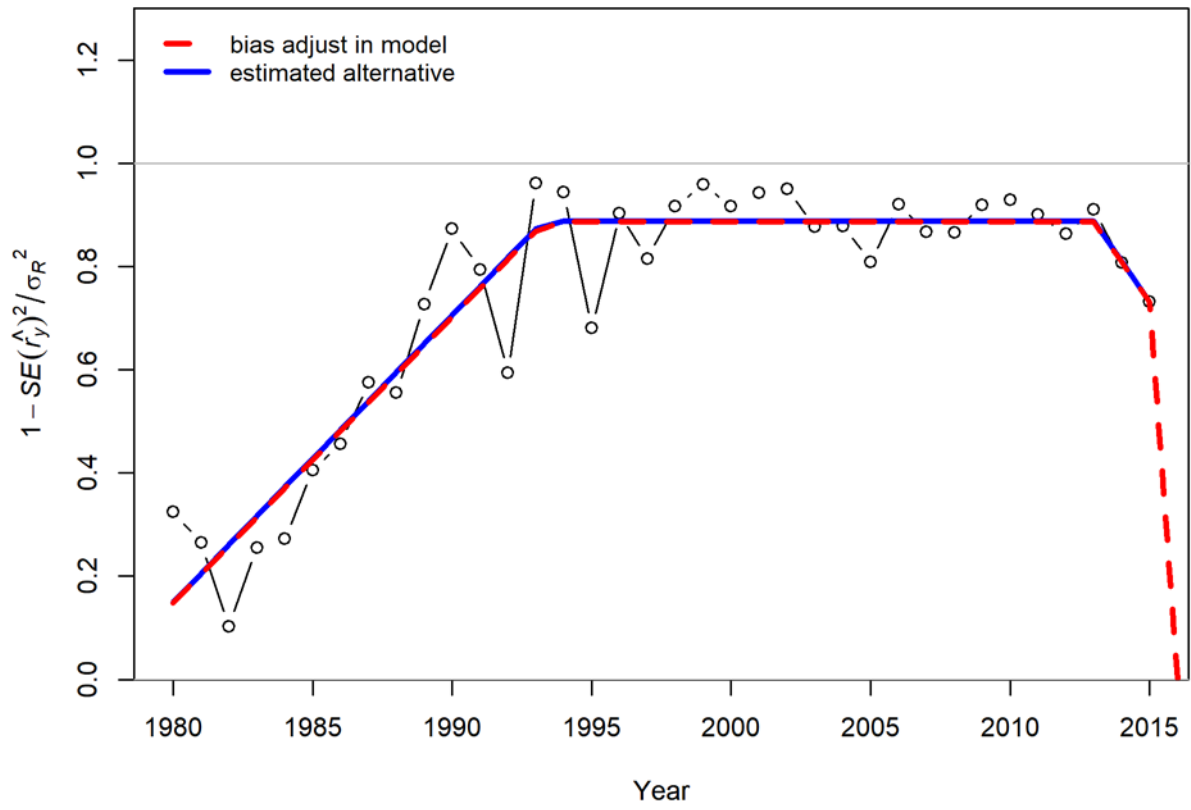


Figure A 11.28. Bias ramp adjustment for silver warehou.

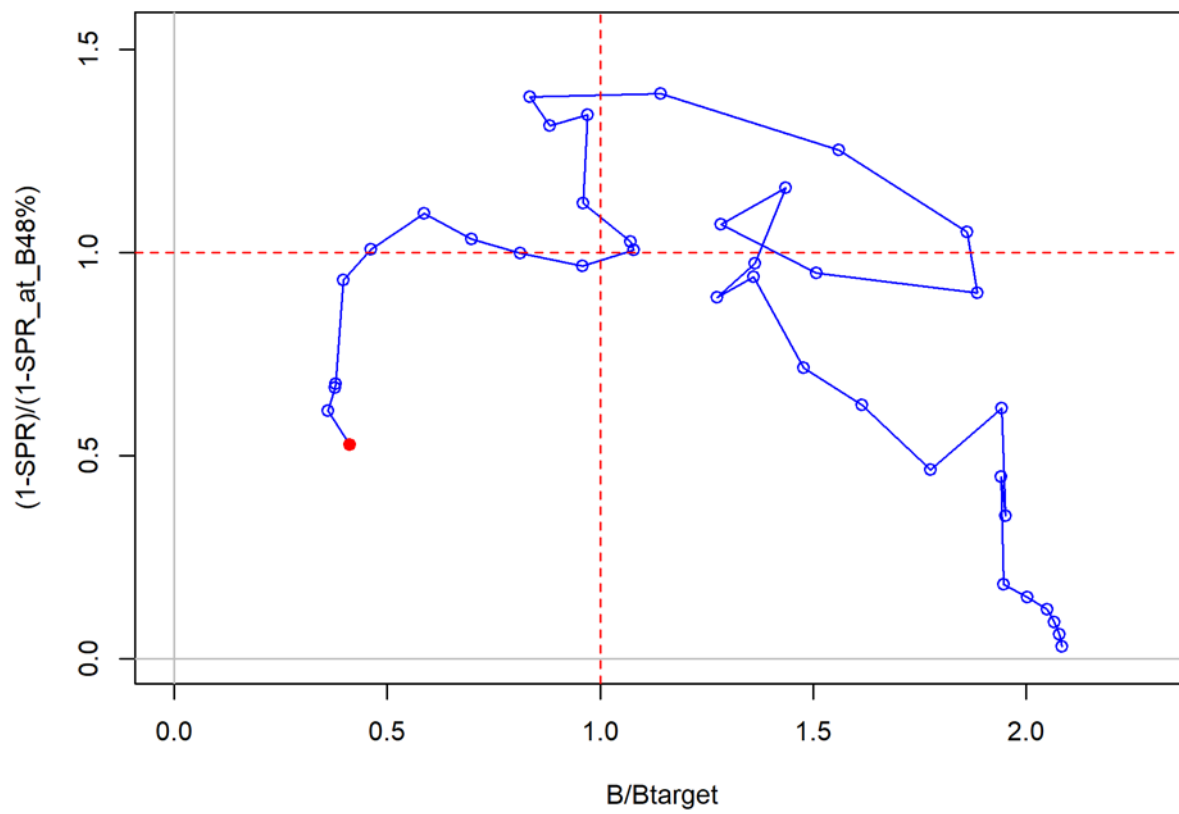


Figure A 11.29. Phase plot of biomass vs SPR ratio.

12. Silver Warehou (*Seriolella punctata*) stock assessment based on data up to 2017

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12.1 Executive Summary

This document presents a quantitative Tier 1 assessment of silver warehou (*Seriolella punctata*) in the Southern and Eastern Scalefish and Shark Fishery (SESSF) using data up to 31 December 2017. The last full assessment was presented in Day *et al.* (2015). The 2018 assessment has been updated by the inclusion of data up to the end of 2017, which entails an additional three years of catch, discard, CPUE, length-composition and conditional age-at-length data and ageing error updates. This document describes the 2018 assessment for silver warehou through the sequential updating of recent data to the stock assessment, using the stock assessment package Stock Synthesis (SS-V3.30.12).

Silver warehou catches peaked at 4,100t in 2002 and subsequently declined to less than 2,000t from 2007 onwards, with further declines to less than 1,000t since 2012. Since 2014 catches have remained relatively stable since 2014 at between 350t and 400t. Over the last decade catches have remained substantially lower than the agreed total allowable catch (TAC).

The preliminary base case was presented to the September 2018 meeting of the South and East Resource Assessment Group (SERAG; Burch *et al.* 2018). SERAG discussed the validity of the catch and discard data and requested that the fishery independent survey (FIS) be removed from the base case and the 2015 recruitment not be estimated. Following the meeting an investigation of the catch and discard data led to a number of changes to the 2018 base case that were communicated to AFMA and the SERAG chair in October. These changes were: the use of a re-estimated discard fractions split between the eastern and western trawl fleets, accounting for the observed discarding practices of factory trawlers, the inclusion of conditional age-at-length data for the western onboard trawl fleet, removal of length data from the small pelagic fishery (SPF) and inclusion of non-trawl catches in the existing eastern and western trawl fleets.

Results show reasonable fits to the CPUE abundance index for the western trawl fleet, however, fits to the eastern trawl fleet are poor prior to 2008. There are differences in the length distributions from onboard and port sampling in both the east and the west, resulting in poor fits to the aggregated length distributions for the eastern trawl fleet. In the west, the fits to the aggregated length distributions were reasonable. The overall fits to the retained length and conditional age-at-length data are reasonable, although there are poor fits in some years. Fits to the discard length data are poor, particularly in the west.

This assessment has seen a continuation of below average recruitment noted in the last two assessments (Day *et al.* 2012, 2015) with the last 11 years of estimated recruitment all below average. While the current assessment estimates that spawning biomass in 2019 will be 31% of unfished levels, previous assessments (Day *et al.* 2012, 2015) have shown that optimistic recent recruitments have been revised downwards in subsequent assessments. This may indicate that a regime shift has occurred for this stock.

Given the changes to the assessment structure, data and tuning methods since the 2012 assessment, a retrospective analysis was undertaken to determine whether the pattern of optimistic recent recruitments have been revised downwards in subsequent assessments was still present with the 2018 assessment structure. Increases in recruitment and spawning biomass in the most recent years from the 2014 and 2016 assessment scenarios that were revised downwards in subsequent assessments. This suggests that the increase in spawning biomass seen in the most recent years of the 2018 assessment may be overly optimistic and that the stock may currently be near the limit reference point.

This assessment estimates that the projected 2019 spawning stock biomass will be 31.3% of virgin stock biomass. The recommended biological catch (RBC) from the base case model for 2019 is 942t for the 20:35:48 harvest control rule, increasing to 1,353t in 2020 and 1,420t in 2021. The long-term yield is 1,772t. In comparison, the 2015 assessment estimated the 2016 spawning biomass to be 40% of the unfished level, with corresponding RBCs of 1,958t, with a long-term yield of 2,281t. However, these scenarios assume recruitment will return to average levels. Low and very low recruitment scenarios suggest that if current landed catches are maintained at around 350t per annum then the stock is likely to remain above the limit reference point.

A Bayesian assessment was undertaken because the base case assessment showed that the maximum likelihood estimate of spawning biomass was near the limit reference point of 20% of its unfished level between 2013 and 2016. The Bayesian assessment estimated the probability that the spawning biomass was below the limit reference point was greater than 20% between 2013 and 2016. In 2017 the probability the spawning biomass was below the limit reference point was 8% and in 2018 and 2019 the probability the spawning biomass was below the limit reference point is <1%.

At its November 2018 meeting, SERAG agreed to recommend a TAC for silver warehou based on the assumption that recruitment will remain below average in the next few years. SERAG chose to assume that recruitment would remain at the mean of the last five years of estimated recruitments in the base case model (2010 – 2014). Projections assuming this low recruitment were run for scenarios of constant landed catch that were between the catch in the most recent year for which data is available (348 t) and the RBCs from the base case model which assumes average recruitment (942 t in 2019). Scenarios with constant annual catches of 750 t or more led to the estimated spawning biomass declining under the low recruitment scenario. Under the low recruitment scenario with constant annual catches between 348 t and 600 t, spawning biomass is predicted to increase, albeit more slowly than the base case which assumes average recruitment.

12.2 Introduction

12.2.1 The fishery

Silver warehou occur throughout the SESSF in depths to 600m and are predominantly caught by demersal trawl (Morison *et al.*, 2007, Sporcic *et al.* 2015, Thomson *et al.* 2015). Silver warehou have also been captured off western Tasmania as bycatch of the winter spawning blue grenadier fishery. In addition to demersal trawl, there have also been some gillnet catches (Morison *et al.*, 2007) and catches by the small pelagic fishery (SPF) using mid-water trawl. Annual catches (landings by fleet) of silver warehou by calendar year are shown in Table 12.2.

Large catches of silver warehou were first taken in the 1970's (Smith, 1994) and landed catches increased to around 2,000t in the early 1990s and peaked at 4,100t in 2002 (Table 12.2, Figure 12.2 and Figure 12.3). Catches declined to less than 2,000t from 2007 onwards, with further declines to less

than 1,000t since 2012. Catches have remained relatively stable since 2014 at between 350t and 400t. Catches are described in more detail in Section 12.3.1.3.

For 2016, 2017 and 2018 the agreed total allowable catches (TACs) were 1,209t, 605t and 600t respectively. These TACs were set following the last assessment in 2015 (Day *et al.*, 2015) assuming a low recruitment scenario.

12.2.2 Stock Structure

Prior to 2015 silver warehou was assessed as a single population using a single trawl fleet in SESSF zone 10 – 50 (Day *et al.* 2012; Figure 12.1). However, differences in standardised catch rates, length and age distribution east and west of longitude 147° E were identified by Sporcic *et al.* (2015). This led to the development of a preliminary assessment which split the data into two fleets, an eastern fleet (SESSF zones 10, 20 and 30) and a western fleet (SESSF zones 40 and 50; Thomson *et al.* 2015). This fleet structure was adopted as the base case for the 2015 assessment (Day *et al.* 2015) and has been retained as the base case for the 2018 silver warehou assessment (Burch *et al.* 2018).

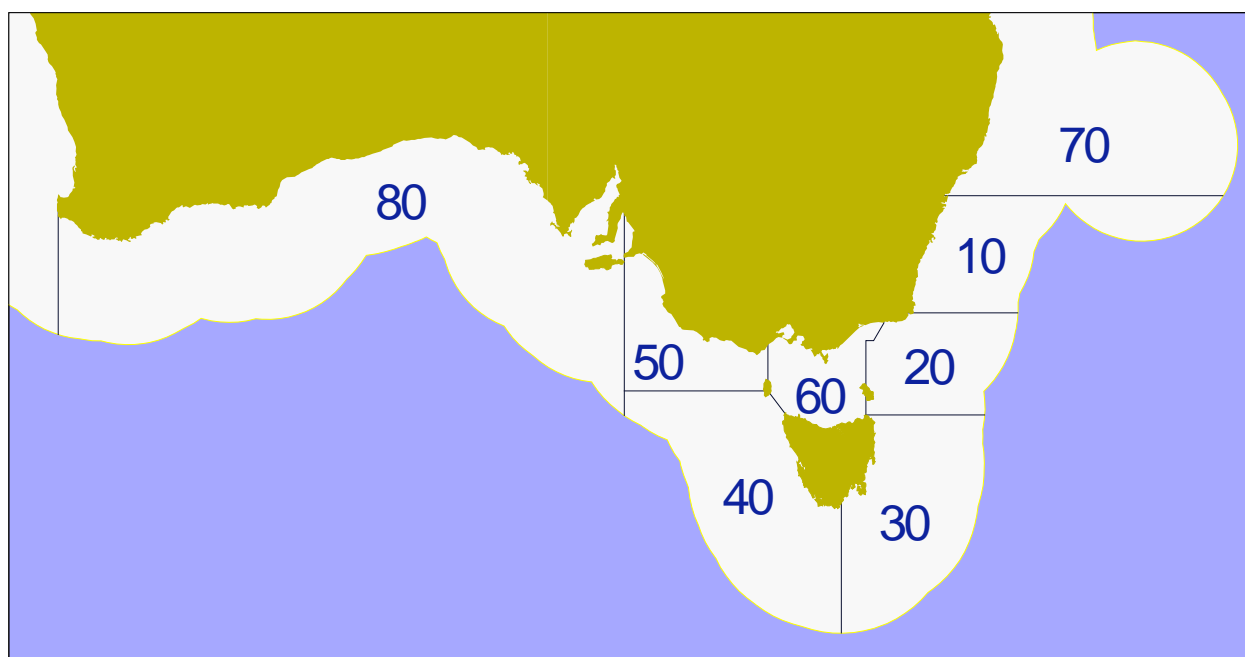


Figure 12.1. Map of the SESSF showing statistical zones.

12.2.3 Previous Assessments

The previous full quantitative assessment for silver warehou was performed in 2015 (Day *et al.*, 2015) using Stock Synthesis (SS-V3.24U, Methot 2015). The 2015 assessment indicated that the spawning stock biomass levels in 2016 were around 40% of virgin biomass, however, recruitment for nine out of the ten most recent years was estimated to be below average and the TACs for 2016 – 2018 were set assuming below average future recruitment.

The 2012 assessment (Day *et al.* 2012) modelled the stock using a single trawl fleet in SESSF zones 10 – 50, which continued the fleet structure from previous assessments (e.g. Tuck and Fay 2009). Prior

to the 2015 assessment identification of heterogeneity in standardised catch per unit effort (CPUE) between the eastern (SESSF zones 10 – 30) and the western (SESSF zones 40 – 50) areas (Sporcic *et al.* 2015) prompted a re-examination of the fleet structure in the assessment. Thomson *et al.* (2015) investigated the relationship between depth and silver warehou length frequency. They concluded there was a strong relationship between length frequencies of fish caught in the west and those caught in deeper water, and a similar relationship between those caught in the east and in shallow waters, with larger fish generally caught in the west and in deeper water. This led to the development of a preliminary base case assessment which split the single trawl fleet into eastern (SESSF zones 10-30) and western (SESSF zones 40-50) fleets (Thomson *et al.* 2015). Thomson *et al.* (2015) also identified evidence of changing discarding practices within the fishery with both size and market based discarding occurring up until 2001 and only size based discarding from 2002 onwards. This permitted discard rates to be estimated within the 2015 assessment using separate retention functions pre and post 2002 (Day *et al.* 2015). The changes to the fleet structure and how discards are accounted for led to improvements in the fits to the length and age composition data compared with previous assessments (e.g. Day *et al.* 2012, Tuck and Fay 2009).

Prior to 2012, an assessment for silver warehou was performed in 2009 (Tuck and Fay, 2009) using Stock Synthesis (version SS-V3.03a, Methot, 2009) and this assessment indicated that the spawning stock biomass levels in 2010 were around 48% of virgin biomass. Fits to the length, age, and catch-rate data were reasonable. The fit to the catch rate index was a substantial improvement compared to Tuck and Punt, (2007), with changes to the estimates of mortality and growth. Exploration of model sensitivity showed that the model outputs are sensitive to the value assumed for natural mortality, M .

Before the 2009 assessment, other Stock Synthesis based assessments for silver warehou were performed in: 2008 (Tuck, 2008) with a spawning biomass estimate for 2007/8 of 53% of the unfished level; 2007 (Tuck and Punt, 2007) with a spawning biomass estimate for 2007/8 of 49% of unfished levels. Even earlier assessments include Taylor and Smith, (2004) and Thomson, (2002).

12.2.4 Modifications to the 2015 assessment

The 2015 assessment made two substantial changes to structure of the silver warehou assessment, splitting a single trawl fleet into eastern and western fleets and estimating discarded catches within the assessment (Day *et al.* 2015). Both of these changes have been retained and we have made the following changes to the 2015 assessment:

1. CPUE, Catch, discard, length frequency, and age at length data for 2015, 2016 and 2017 have been added.
2. Catches from the Gillnet, Hook and Trap sector (GHAT) and the SPF are now included in the assessment.
3. Recruitment is estimated until 2014 (two more years than the last assessment).
4. The ageing error matrix has been updated.
5. Estimated annual discard rates that are fitted to by the assessment have been split into eastern and western components.
6. Discard estimates have been updated in 2018 to more closely match the discard calculations in Bergh *et al.* (2009). These estimates use ratios of total discards to (retained + discard) catch on a per shot basis, rather than aggregated across a whole strata, which are then weighted up according to CDR landings within zone and season (N. Klaer, pers. comm.).

7. Factory trawlers are now included in the estimation of annual discard rates when there is observer coverage.
8. The retention function remains time blocked with the second period extended to 2017, reflecting changes in the discarding practices in the periods 1980-2001 and 2002-2017.
9. FIS abundance indices for east and west fleets are removed from the base case assessment and are instead considered as a sensitivity.
10. A new tuning procedure has been used to balance the weighting of each of the data sources that contribute to the overall likelihood function.

These changes and their impact on the assessment are described in detail in Burch et al (2018) and Section 12.3.1.

12.2.5 The data and model input

12.2.5.1 Biological parameters

A single sex model (i.e. both sexes combined) was used, as the length composition data for silver warehou are not available by sex. A summary of the key biological parameters, including the values of the fixed parameters in the base case model is provided in Table 12.1.

Growth was assumed to follow the von Bertalanffy growth model with parameters estimated within the model-fitting procedure because Stock Synthesis accepts age-at-length data as an input. Estimating the parameters of the von Bertalanffy growth curve within the assessment is more appropriate because it better accounts for the impact of gear selectivity on the age-at-length data collected from the fishery and the impact of ageing error.

This assessment follows that of Day *et al.* (2012, 2015) and Tuck and Fay (2009) in using the base-case value of natural mortality of $M=0.3\text{yr}^{-1}$. Likelihood profiles of natural mortality undertaken for the preliminary base case (Burch *et al.* 2018) and presented to the September 2018 SERAG meeting suggested a value of natural mortality of $M=0.5\text{yr}^{-1}$ was preferred by the model. As such a high value of natural mortality is inconsistent with the biology of the species, natural mortality was retained at a fixed value of $M=0.3\text{yr}^{-1}$. The base case value of the steepness of the stock-recruitment relationship, h , is 0.75 Sensitivities to this value for M and h are considered.

Silver warehou become sexually mature at a length of about 37 cm. Fecundity is assumed to be proportional to spawning biomass. The parameters of the length-weight relationship are the same as those used in previous assessments ($a=6.5 \times 10^{-6}$, $b=3.27$). These values come from Taylor and Smith (2004) and were provided by David Smith (unpublished data).

Table 12.1. Summary of fixed and estimated parameters for the base case assessment.

| Parameter | Details | |
|--------------------------------|-----------|---------------------------|
| Natural mortality (M) | fixed | 0.3 |
| Steepness (h) | fixed | 0.75 |
| length-weight scale, a | fixed | 6.50E-06 |
| length-weight power, b | fixed | 3.27 |
| length at 50% maturity (cm) | fixed | 37 |
| maturity slope | fixed | -6 |
| Recruitment deviations | estimated | 1980-2014 |
| CV growth | estimated | 0.0808 |
| Growth K | estimated | Female 0.312 |
| Growth l_{min} | estimated | Female age 2 14.82 |
| Growth l_{max} | estimated | Female 51.21 |
| length at 50% selectivity (cm) | estimated | 22.82 (east) 39.87 (west) |
| selectivity spread (cm) | estimated | 3.48 (east) 11.24 (west) |
| $\ln(R_0)$ | estimated | 9.379 |

12.2.5.2 Fleets

The base case assessment for silver warehou is based on a trawl fleet split into an eastern trawl fleet (SESSF zones 10, 20 and 30) and a western trawl fleet (SESSF zones 40 and 50), with time-invariant logistic selectivity estimated separately for each fleet. Prior to the 2015 assessment, discards were added to the landed catch due to difficulties in distinguishing between size-based discarding and market-based discarding. This assumption ignored the size-related discarding of small fish that was occurring along with market-related discarding of fish of all sizes, as evidenced by the greater proportion of small fish in the discarded length frequencies from 2002 onwards relative to the retained LFs from 2002 onwards (Thomson *et al.* 2015). The 2015 assessment incorporated separate retention (discard) functions which were estimated for the 1980-2001 and 2002-2014 periods. This enabled a retention function to be fitted allowing for this apparent change in discarding practice from 2002 onwards and also resulted in improvements to the fits to the length residuals (Day *et al.* 2015).

Previous assessments excluded non-trawl catches because they were small and the assessments were insensitive to their inclusion (e.g. Thomson 2002). This assessment now includes non-trawl catches assigned to the eastern and western trawl fleets based on the location of the catches. This change was made because of increasing catches of silver warehou from the SPF in 2015 and 2016 and is described in more detail in Section 12.3.1.3.

12.2.5.3 Catches

The model uses a calendar year for all catch data. The catch history of silver warehou from 1994 onwards has been revised in the 2018 assessment to account for updates to the database made by AFMA (Thomson *et al.* 2018) and given by Castillo-Jordán *et al.* (2018).

The first model year is 1980, however, SEF1 record-keeping did not begin until 1985. Landings of silver warehou prior to 1985 are not considered to have been large and a linear increase in catch from 1980 to 1985 was assumed, following Punt *et al.* (2005). Silver warehou are closely related to blue

warehou and historically catches have often been reported mixed, or with all warehou species combined and referred to as Tassie trevally (Sporcic *et al.* 2015). This practice was most prevalent in the late 1980s with it unclear which species was caught and recorded in Commonwealth logbooks. For this reason, catches prior to 1994 have not been revised and are instead taken from Table 12.11 of Sporcic *et al.* (2015) and shown in the first column of Table 12.2. These catches differ slightly from those in the 2015 assessment because of an error in removing discards from the total catch (discarded and retained) when discarded catches were first estimated in the silver warehou assessment.

Previous assessments have excluded non-trawl catches as they were small and the assessment was insensitive to their inclusion. While the majority of the catch is still taken by demersal trawl, in 2015 and 2016 catches of silver warehou of 28t and 50t respectively were recorded by the SPF. While these catches were small compared to catches in the 1990s and early 2000s (Table 12.2, Figure 12.2 and Figure 12.3) with the decline in landings they constitute 7% and 14% of the silver warehou catches in 2015 and 2016 respectively. Catch disposal records (CDR) from the SPF, along with those from the Gillnet, Hook and Trap sector (GHAT) are combined with those from demersal trawling (Commonwealth Trawl Sector; CTS) to obtain total landed catches used in the assessment (Table 12.2). These catches were then split into eastern and western trawl fleets for the assessment based on the proportion of logbook catch east and west of 147° longitude (Table 12.2, Figure 12.2 and Figure 12.3). Prior to 1985, it was assumed 50% of the catch was taken east of 147° longitude and 50% was taken west of 147° longitude.

Table 12.2. Catch from the 2015 stock assessment (Day *et al.* 2015) with discards removed, catches by the Commonwealth Trawl Sector (CTS) the gillnet, hook and trap (GHAT) and small pelagic fishery (SPF). The total landed catch used in the 2018 assessment, the 2018 catch split into the eastern and western trawl fleets and the agreed TACs for silver warehou. Shaded columns represent catches used in the 2018 base case assessment. Grey cells denote the total landings used in the 2018 assessment.

| Year | 2015 | 2018 Catch (t) by Sector | | | 2018 Assessment Catch (t) | | | TAC |
|------|-----------|--------------------------|------|------|---------------------------|------------|------------|------|
| | Catch (t) | CTS | GHAT | SPF | Total | East trawl | West trawl | |
| 1980 | 59.0 | - | - | - | 59.0 | 29.5 | 29.5 | - |
| 1981 | 118.1 | - | - | - | 118.1 | 59.0 | 59.0 | - |
| 1982 | 177.1 | - | - | - | 177.1 | 88.6 | 88.6 | - |
| 1983 | 236.2 | - | - | - | 236.2 | 118.1 | 118.1 | - |
| 1984 | 295.2 | - | - | - | 295.2 | 147.6 | 147.6 | - |
| 1985 | 360.0 | 355.2 | - | - | 360.0 | 58.4 | 301.6 | - |
| 1986 | 1008.0 | 1147.4 | - | - | 1008.0 | 433.3 | 574.7 | - |
| 1987 | 748.8 | 781.8 | - | - | 748.8 | 261.0 | 487.8 | - |
| 1988 | 1365.6 | 1642.1 | - | - | 1365.6 | 781.6 | 584.0 | - |
| 1989 | 920.4 | 919.0 | - | - | 920.4 | 342.8 | 577.6 | - |
| 1990 | 1125.6 | 1339.3 | - | - | 1125.6 | 866.8 | 258.7 | - |
| 1991 | 1363.2 | 1259.4 | - | - | 1363.2 | 664.3 | 698.9 | - |
| 1992 | 1864.8 | 675.8 | 0 | 0 | 1864.8 | 1246.0 | 618.8 | 2000 |
| 1993 | 1969.2 | 1813.7 | 0 | 0 | 1969.2 | 1115.7 | 853.5 | 2000 |
| 1994 | 2054.3 | 2308.3 | 0 | 0 | 2308.3 | 1545.4 | 762.9 | 2500 |
| 1995 | 2213.9 | 2000.4 | 0 | 0 | 2000.4 | 1212.6 | 787.8 | 2500 |
| 1996 | 2735.7 | 2182.6 | 0 | 0 | 2182.6 | 1125.1 | 1057.4 | 2500 |
| 1997 | 2807.5 | 2378.7 | 0 | 0 | 2378.7 | 1043.6 | 1335.1 | 2500 |
| 1998 | 2434.0 | 2409.8 | 0 | 0 | 2409.8 | 918.6 | 1491.3 | 3500 |
| 1999 | 3255.2 | 3248.0 | 0 | 0 | 3248.0 | 1064.6 | 2183.4 | 4000 |
| 2000 | 3726.6 | 3726.1 | 0 | 0 | 3726.1 | 797.1 | 2929.0 | 4000 |
| 2001 | 3295.4 | 3296.2 | 0 | 0 | 3296.2 | 712.1 | 2584.2 | 4400 |
| 2002 | 4101.9 | 4101.4 | 0 | 0 | 4101.4 | 768.3 | 3333.1 | 4400 |
| 2003 | 3060.0 | 3041.0 | 12.6 | 3.5 | 3057.1 | 618.2 | 2438.9 | 4488 |
| 2004 | 3315.0 | 3311.0 | 0.2 | 0 | 3311.3 | 523.8 | 2787.5 | 4039 |
| 2005 | 2912.7 | 2907.6 | 0.1 | 0 | 2907.7 | 506.9 | 2400.8 | 4400 |
| 2006 | 2374.2 | 2373.5 | 0.1 | 0 | 2373.6 | 440.4 | 1933.3 | 4400 |
| 2007 | 1987.1 | 1998.4 | 0.1 | 0 | 1998.4 | 309.2 | 1689.2 | 3088 |
| 2008 | 1523.0 | 1522.8 | 0.1 | 0 | 1522.9 | 449.7 | 1073.2 | 3227 |
| 2009 | 1379.3 | 1378.2 | <0.1 | 0 | 1378.2 | 409.1 | 969.1 | 3000 |
| 2010 | 1288.7 | 1287.1 | 1.3 | 0 | 1288.4 | 311.8 | 976.6 | 2566 |
| 2011 | 1235.5 | 1228.8 | 0.1 | 0 | 1228.9 | 252.4 | 976.5 | 2566 |
| 2012 | 853.4 | 847.7 | <0.1 | 0 | 847.7 | 209.3 | 638.4 | 2566 |
| 2013 | 583.5 | 645.6 | 0 | 0 | 645.6 | 181.3 | 464.3 | 2329 |
| 2014 | - | 381.5 | <0.1 | 0 | 381.5 | 95.9 | 285.6 | 2329 |
| 2015 | - | 359.0 | <0.1 | 27.7 | 386.6 | 71.3 | 315.4 | 2417 |
| 2016 | - | 301.6 | <0.1 | 48.9 | 350.5 | 128.2 | 222.4 | 1209 |
| 2017 | - | 348.0 | 0.1 | 0.1 | 348.1 | 105.7 | 242.4 | 605 |
| 2018 | - | - | - | - | 348.1* | 105.7* | 242.4* | 600 |

* Catch from 2017 used for 2018 in the silver warehou assessment

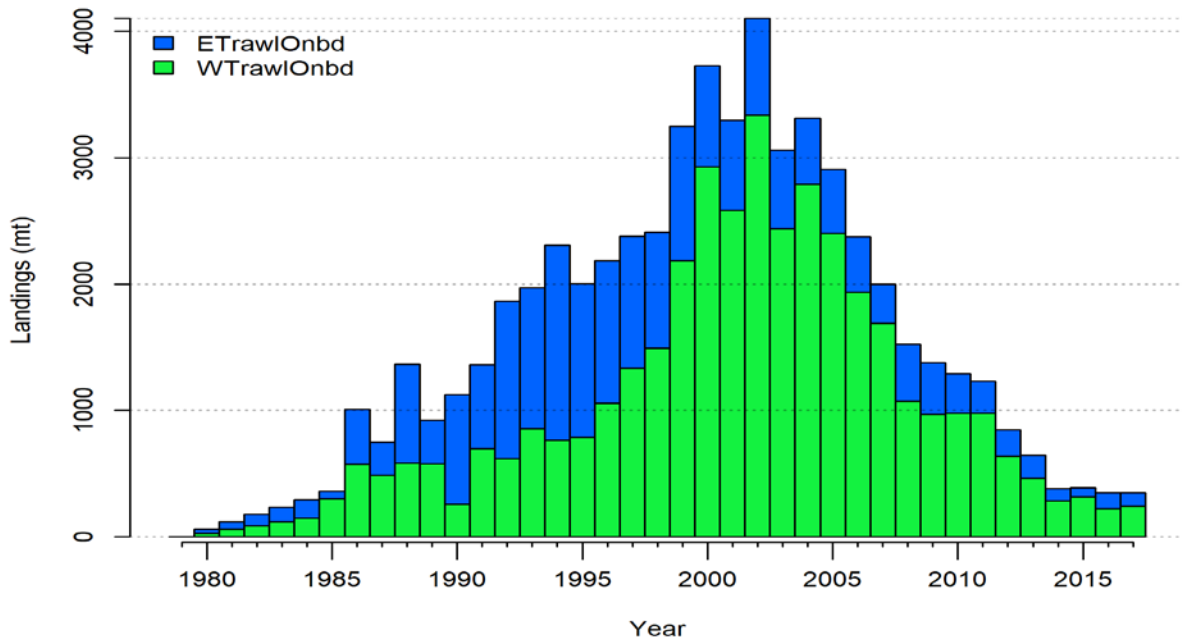


Figure 12.2. Total landed catch by fleet (stacked) for the eastern (blue) and western (green) fleets for silver warehou in the SESSF from 1980-2017 as used in this assessment.

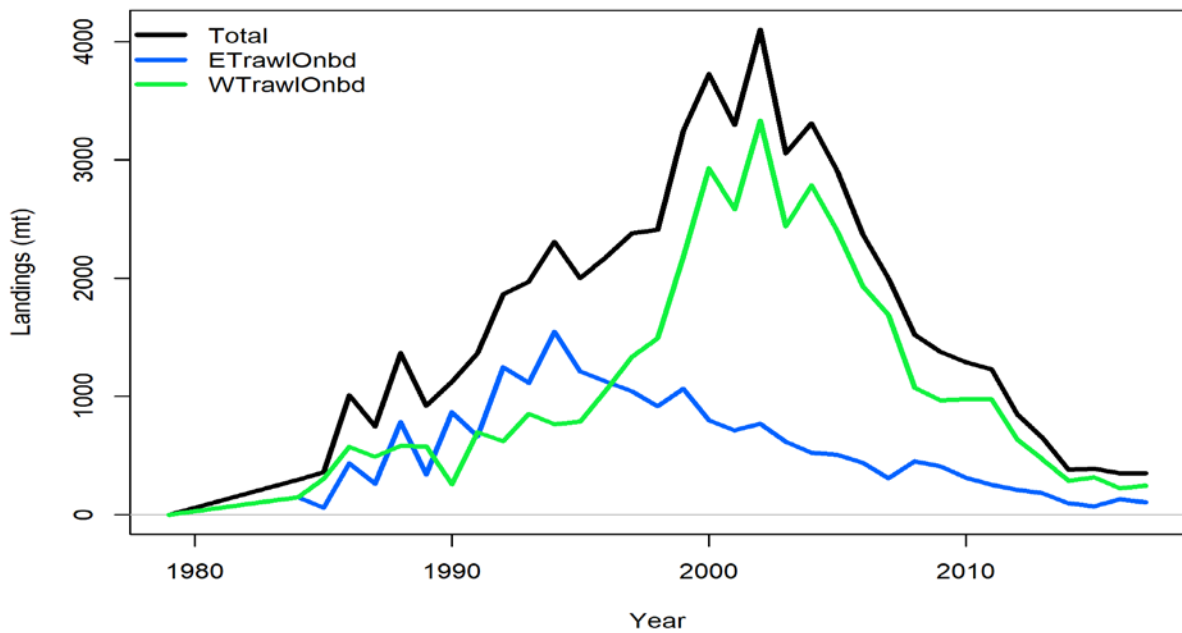


Figure 12.3. Total landed catch for the eastern (blue line) and western (green line) fleets for of silver warehou in the SESSF from 1979-2017 as used in this assessment.

12.2.5.4 Discard rates

Information on the discard catches of silver warehou is available from the integrated scientific monitoring program (ISMP) for 1993-2017. Prior to the 2015 assessment, there was no known pattern indicating when discarding was market-driven and when it was size-related, so the estimated discarded catch was added to the landed catch (Day *et al.* 2012). Thomson *et al.* (2015) provided evidence to support a change in discarding practice, from a mixture of market and sized based discarding prior to 2002, with only size based discarding occurring from 2002 onwards. The 2015 assessment estimated discarded catches separately for the eastern and western trawl fleets, both pre and post 2002, however, at that time the estimated fraction of discarded catch from ISMP data were only available for the stock as a whole (Upston and Thomson 2015).

Discard rates for Tier 1 assessments are required by fishing fleet. This means that the discard estimates for TAC purposes used for Tier 3 and 4 assessments which are provided in the discard report (Burch *et al.* 2018b) cannot be used in Tier 1 assessments. The discards from Burch *et al.* (2018b) are produced using a set of rules to determine, for the entire quota fishery, whether sufficient data are available to make an annual fishery wide discard estimate. The discard rates calculated for and input to Tier 1 stock assessments are used to fit retention selectivity curves, so individual year values are not greatly influential on model estimated discard rates.

The Tier 1 discard estimates have been updated in 2018 to more closely match the discard calculations in Bergh *et al.* (2009). These estimates use ratios of total discards to (retained + discard) catch on a per shot basis, rather than aggregated across a whole strata, which are then weighted up according to CDR landings within zone and season (N. Klaer, pers. comm.). This assessment separates the ISMP data east and west of 147° longitude and estimates the fraction of discarded catch separately for the eastern and the western trawl fleets. Discard fraction estimates are provided in Table 12.3.

Silver warehou are caught in the spawning fishery for blue grenadier and substantial catches have been taken in some years by factory trawlers that have operated since 1997. Previous assessments have adjusted the estimated discard fraction lower on the assumption that factory trawlers have fishmeal plants which apparently absorb all fish that might otherwise have been discarded. The September 2018 meeting of SERAG discussed the prevalence of the discarding of silver warehou by factory trawlers operating in the blue grenadier spawning fishery and recommended reviewing this assumption.

An investigation of the ISMP data identified records of factory vessels catching silver warehou exist in the ISMP database in 1998 and 1999 and between 2009 and 2013. In 1998, 1999, 2009 and 2013 there were no records of silver warehou being discarded in these years, however, there was some discarding of silver warehou from 2010 to 2012. For the years with observer coverage of the factory vessels we include the factory vessel data in the estimation of discarded silver warehou catches. For the years without observer coverage when factory vessels operated in this fishery (1997 and 2000 – 2008) we assume zero discarding by the factory vessels and adjust our discard rates lower by the proportion of factory vessel catch (Table 12.3).

Estimated discard fractions in the east were extremely variable with highs of 73% in 1995, 44% in 2003 and 43% in 2002 to <5% in 1993, 1994, 1999, 2007, 2010 and 2014 (Table 12.3). In the west estimated discard fractions were generally lower than those in the east with only the 1998 and 2000 being above 20%. The assessment did not fit the high discard fractions in either the east or the west with maximum discard rates in the east around 25% and <20% in the west (Figure 12.4). The discarded catches estimated within the assessment for years 1980 to 2017 are provided in Figure 12.5 and Figure 12.6.

Table 12.3. Estimated discarded catch fraction by fleet, factory vessel catch as a proportion of landed catch by fleet and discard rates adjusted for factory vessel catch proportions by fleet.

| Year | Estimated discard fraction | | Proportion factory vessel catch | | Observer coverage | Adjusted discard fraction | |
|------|----------------------------|------------|---------------------------------|-------|-------------------|---------------------------|------------|
| | East trawl | West trawl | East | West | | East trawl | West trawl |
| 1993 | 0.040 | - | - | - | - | 0.040 | - |
| 1994 | 0.012 | 0.087 | - | - | - | 0.012 | 0.087 |
| 1995 | 0.726 | 0.018 | - | - | - | 0.726 | 0.018 |
| 1996 | 0.214 | 0.159 | - | - | - | 0.214 | 0.159 |
| 1997 | 0.262 | 0.065 | 0 | 0.013 | No | 0.262 | 0.064 |
| 1998 | 0.099 | 0.366 | 0 | 0.034 | Yes | 0.099 | 0.366 |
| 1999 | 0.040 | 0.047 | 0.001 | 0.145 | Yes | 0.040 | 0.047 |
| 2000 | 0.100 | 0.208 | 0 | 0.288 | No | 0.100 | 0.148 |
| 2001 | 0.276 | 0.131 | 0.009 | 0.233 | No | 0.274 | 0.101 |
| 2002 | 0.192 | 0.086 | 0 | 0.415 | No | 0.192 | 0.050 |
| 2003 | 0.438 | 0.024 | <0.001 | 0.276 | No | 0.438 | 0.018 |
| 2004 | 0.425 | 0.199 | 0 | 0.043 | No | 0.425 | 0.190 |
| 2005 | 0.192 | 0.067 | 0 | 0.001 | No | 0.192 | 0.067 |
| 2006 | 0.143 | 0.080 | 0 | 0.025 | No | 0.143 | 0.078 |
| 2007 | 0.016 | 0.103 | 0 | 0.011 | No | 0.016 | 0.102 |
| 2008 | 0.121 | 0.028 | 0 | 0.088 | No | 0.121 | 0.026 |
| 2009 | 0.073 | 0.020 | 0 | 0.029 | Yes | 0.073 | 0.020 |
| 2010 | 0.025 | 0.029 | 0 | 0.056 | Yes | 0.025 | 0.029 |
| 2011 | 0.224 | 0.036 | 0 | 0.062 | Yes | 0.224 | 0.036 |
| 2012 | 0.137 | 0.046 | 0 | 0.028 | Yes | 0.137 | 0.046 |
| 2013 | 0.067 | 0.040 | 0 | 0.026 | Yes | 0.067 | 0.040 |
| 2014 | 0.049 | 0.058 | - | - | - | 0.049 | 0.058 |
| 2015 | 0.386 | 0.094 | - | - | - | 0.386 | 0.094 |
| 2016 | 0.420 | 0.019 | - | - | - | 0.420 | 0.019 |
| 2017 | 0.344 | 0.077 | - | - | - | 0.344 | 0.077 |

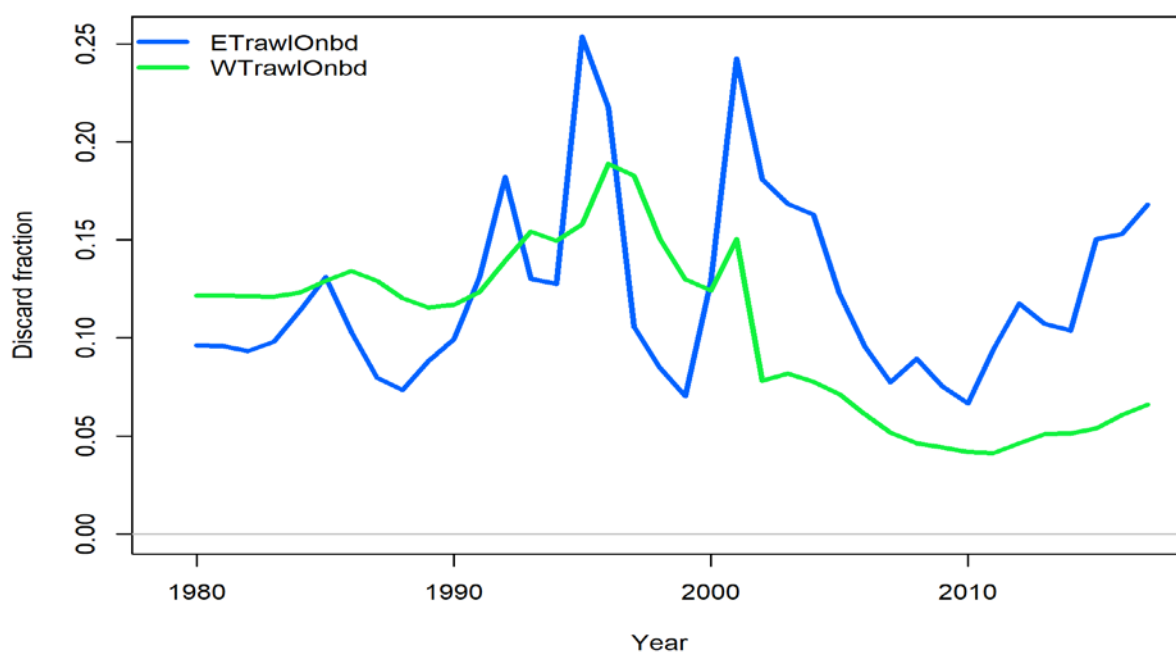


Figure 12.4. Estimates of discard fractions from the 2018 base case assessment for the eastern fleet (blue line) and the western fleet (green line) for silver warehou in the SESSF from 1980-2017.

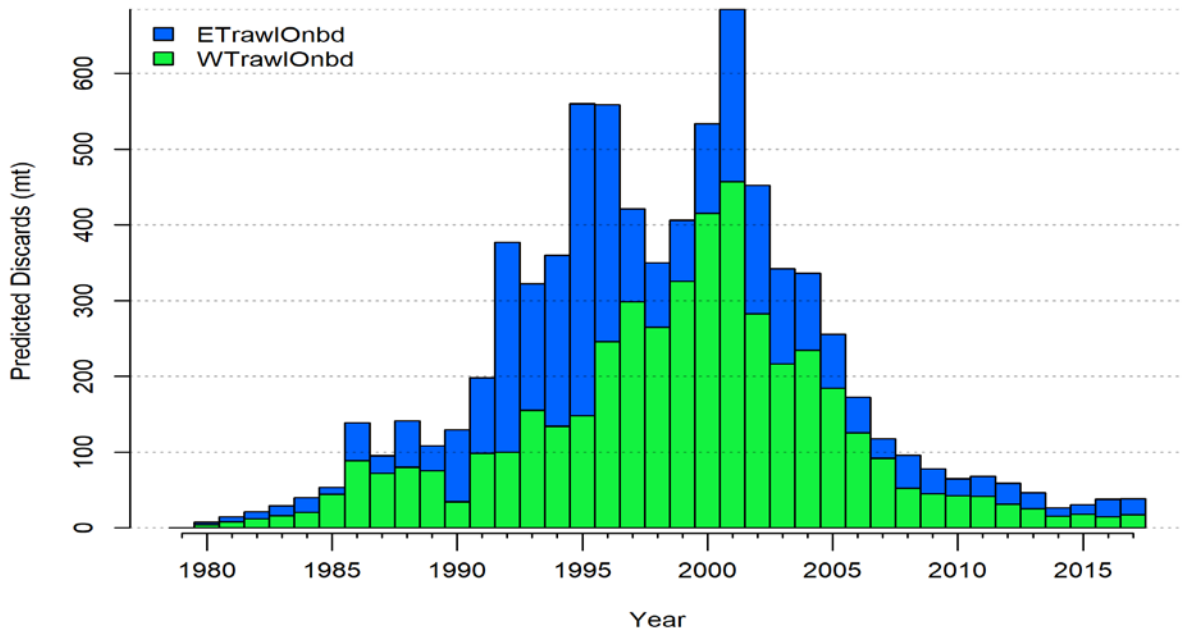


Figure 12.5. Estimated discards (stacked) of silver warehou from the 2018 base case assessment for the eastern (blue) and western (green) fleets 1980-2017.

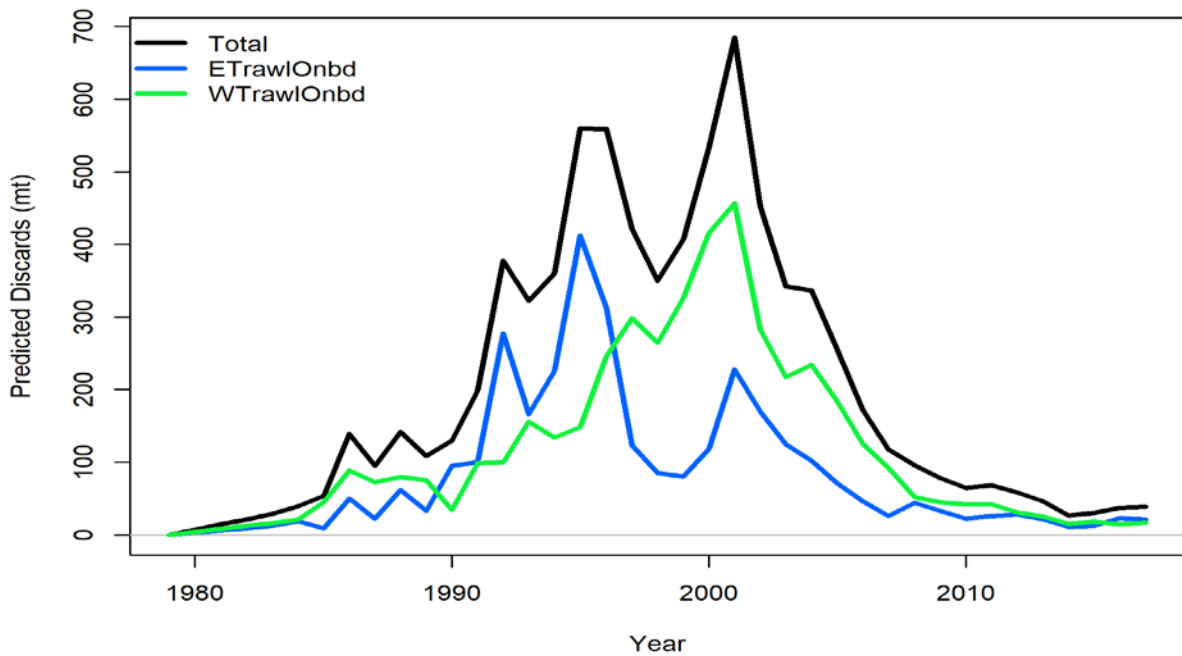


Figure 12.6. Estimated discards of silver warehou from the 2018 base case assessment 1980-2017.

12.2.5.5 Catch rate indices

Catch and effort data from the SEF1 logbook database from the period 1986 to 2017 were standardised using GLMs to obtain indices of relative abundance (Sporcic and Haddon 2018a, b) with the results listed in Table 12.4. Data used in this standardisation were restricted to trawl shots in depths between 0 and 600m from zones 10, 20 and 30 for the eastern trawl fleet and zones 40 and 50 for the western trawl fleet. Estimated standard deviations were obtained by fitting a loess smoother to each CPUE series which is standard practice for Tier 1 assessments in the SESSF.

Table 12.4. CPUE indices by fleet 1986-2017 with CVs used in the assessment.

| Year | East trawl | East trawl CV | West Trawl | West trawl CV |
|------|------------|---------------|------------|---------------|
| 1986 | 1.844 | 0.164 | 1.519 | 0.17 |
| 1987 | 1.799 | 0.164 | 1.719 | 0.17 |
| 1988 | 2.275 | 0.164 | 1.979 | 0.17 |
| 1989 | 1.887 | 0.164 | 1.670 | 0.17 |
| 1990 | 2.404 | 0.164 | 1.107 | 0.17 |
| 1991 | 1.454 | 0.164 | 1.186 | 0.17 |
| 1992 | 1.620 | 0.164 | 0.897 | 0.17 |
| 1993 | 1.589 | 0.164 | 1.250 | 0.17 |
| 1994 | 1.764 | 0.164 | 1.145 | 0.17 |
| 1995 | 1.488 | 0.164 | 0.945 | 0.17 |
| 1996 | 1.212 | 0.164 | 1.059 | 0.17 |
| 1997 | 1.200 | 0.164 | 1.248 | 0.17 |
| 1998 | 0.991 | 0.164 | 1.461 | 0.17 |
| 1999 | 0.868 | 0.164 | 1.203 | 0.17 |
| 2000 | 0.703 | 0.164 | 1.175 | 0.17 |
| 2001 | 0.659 | 0.164 | 0.887 | 0.17 |
| 2002 | 0.768 | 0.164 | 0.942 | 0.17 |
| 2003 | 0.700 | 0.164 | 0.978 | 0.17 |
| 2004 | 0.819 | 0.164 | 1.071 | 0.17 |
| 2005 | 0.757 | 0.164 | 1.171 | 0.17 |
| 2006 | 0.639 | 0.164 | 1.032 | 0.17 |
| 2007 | 0.502 | 0.164 | 1.043 | 0.17 |
| 2008 | 0.588 | 0.164 | 0.830 | 0.17 |
| 2009 | 0.666 | 0.164 | 0.721 | 0.17 |
| 2010 | 0.489 | 0.164 | 0.655 | 0.17 |
| 2011 | 0.424 | 0.164 | 0.631 | 0.17 |
| 2012 | 0.383 | 0.164 | 0.468 | 0.17 |
| 2013 | 0.479 | 0.164 | 0.438 | 0.17 |
| 2014 | 0.329 | 0.164 | 0.417 | 0.17 |
| 2015 | 0.228 | 0.164 | 0.451 | 0.17 |
| 2016 | 0.197 | 0.164 | 0.329 | 0.17 |
| 2017 | 0.276 | 0.164 | 0.375 | 0.17 |

12.2.5.6 Length composition data

In 2010 the RAGs decided to include both port and onboard retained length frequency data (for both historic and current years) in future assessments, whereas in previous assessments only port data have been used (Tuck and Fay 2009). In 2012, the port and onboard length composition data was combined to give one length distribution for each year of data. For the 2015 assessment, port and onboard length composition data were both used separately, with the gear selectivity estimated jointly from both port

and onboard data from each fleet (eastern and western trawl) and this decision was retained to the 2018 assessment. There were some length data available from the SPF in 2015 and 2016, however, this was excluded because of the SPF use mid-water trawl. Should silver warehou continue to be caught in the SPF it may be possible to incorporate the SPF as a separate fleet in the assessment and estimate selectivity from SPF length data.

For onboard data, the number of shots is used as the initial sample size before the length frequency data are re-weighted in the tuning process. This is considered to be more representative of the information content in the length frequencies than the number of fish measured. For port data, the number of shots is not available, but the number of trips is used instead. In the 2018 assessment, the initial sample size associated with each length frequency in the assessment is the number of shots or trips. However, data was excluded for years with less than 100 individual fish measured, as small samples are potentially unrepresentative. Sample sizes for retained length frequencies, including both the number of individuals measured and numbers of shots or trips, are listed in Table 12.5 for each fleet and year for 1991 to 2017.

Table 12.5. Number of retained lengths, shots and trips included in the base case assessment by fleet 1991-2017.

| Year | Onboard sampling | | | | Port sampling | | | |
|------|------------------|-------|------------|-------|---------------|-------|------------|-------|
| | East trawl | | West trawl | | East trawl | | West trawl | |
| | Lengths | Shots | Lengths | Shots | Lengths | Trips | Lengths | Trips |
| 1991 | - | - | - | - | 273 | 4 | - | - |
| 1992 | - | - | - | - | 1648 | 9 | 1769 | 15 |
| 1993 | - | - | - | - | 1087 | 6 | 1431 | 11 |
| 1994 | - | - | - | - | 215 | 4 | 1802 | 22 |
| 1995 | - | - | - | - | 500 | 5 | 4651 | 37 |
| 1996 | 293 | 4 | 122 | 1 | 1014 | 10 | 6023 | 53 |
| 1997 | 1815 | 22 | 1883 | 33 | 1762 | 18 | 8874 | 82 |
| 1998 | 2959 | 34 | 2671 | 43 | 6386 | 63 | 9704 | 89 |
| 1999 | 2449 | 32 | 1952 | 19 | 6347 | 68 | 7742 | 75 |
| 2000 | 1642 | 17 | 3584 | 46 | 8239 | 48 | 5424 | 47 |
| 2001 | 1446 | 23 | 4610 | 47 | 7958 | 60 | 6978 | 61 |
| 2002 | 2554 | 37 | 4047 | 26 | 12979 | 85 | 9064 | 83 |
| 2003 | 2005 | 34 | 5019 | 44 | 5431 | 37 | 3359 | 28 |
| 2004 | 2147 | 33 | 3679 | 33 | 4868 | 34 | 2638 | 23 |
| 2005 | 2028 | 25 | 6617 | 60 | 9007 | 46 | 3319 | 28 |
| 2006 | 1847 | 33 | 3763 | 32 | 7994 | 49 | 855 | 9 |
| 2007 | 173 | 12 | - | - | 728 | 5 | 491 | 2 |
| 2008 | 440 | 18 | 198 | 8 | 971 | 6 | - | - |
| 2009 | 370 | 10 | 853 | 41 | 2135 | 27 | 163 | 2 |
| 2010 | 1391 | 30 | 1285 | 37 | 1139 | 22 | - | - |
| 2011 | 371 | 17 | 1140 | 61 | 1288 | 40 | - | - |
| 2012 | 807 | 31 | 991 | 31 | 1252 | 40 | - | - |
| 2013 | 730 | 29 | 1523 | 49 | 1720 | 45 | 141 | 1 |
| 2014 | 142 | 4 | 900 | 17 | 1391 | 26 | 152 | 2 |
| 2015 | 282 | 11 | 934 | 25 | 1755 | 30 | - | - |
| 2016 | 452 | 14 | 656 | 34 | 1476 | 20 | 240 | 10 |
| 2017 | 404 | 12 | 549 | 17 | 1859 | 27 | 195 | 4 |

Discarded length frequencies were only available for onboard samples as discarded fish are not landed in port. Sample sizes for discarded length frequencies including both the number of individuals

measured and numbers of shots are listed in Table 12.6 for each fleet and year for the period 1994-2017.

Table 12.6. Number of discarded lengths and shots included in the base case assessment by fleet 1994-2017.

| Year | East trawl | | West trawl | |
|------|------------|-------|------------|-------|
| | Lengths | Shots | Lengths | Shots |
| 1994 | - | - | 224 | 2 |
| 1995 | 456 | 5 | 930 | 8 |
| 1996 | - | - | 1421 | 10 |
| 1997 | 234 | 3 | 232 | 18 |
| 1998 | - | - | 1998 | 39 |
| 1999 | - | - | 477 | 6 |
| 2000 | 210 | 3 | 296 | 18 |
| 2001 | 888 | 15 | 1371 | 25 |
| 2002 | 1805 | 34 | 1257 | 8 |
| 2003 | 1364 | 23 | 191 | 3 |
| 2004 | 3319 | 52 | 1111 | 16 |
| 2005 | 1332 | 19 | 658 | 15 |
| 2006 | 140 | 13 | - | - |
| 2008 | 150 | 9 | - | - |
| 2009 | 127 | 2 | - | - |
| 2010 | 131 | 6 | - | - |
| 2011 | 159 | 9 | 132 | 23 |
| 2012 | 471 | 13 | - | - |
| 2013 | 109 | 13 | 178 | 8 |
| 2014 | 163 | 2 | - | - |
| 2015 | 280 | 10 | - | - |
| 2016 | 499 | 15 | - | - |
| 2017 | 465 | 17 | 723 | 8 |

12.2.5.7 Age composition data

Age-at-length measurements, based on sectioned otoliths provided by the CAF, were available for the years 1988 and 1993 to 2017 east and west of 147° longitude (Table 12.7).

Table 12.7. Number of samples in the conditional age-at-length data in the base case assessment 1988-2017.

| Year | East | West | Total |
|-------|------|------|-------|
| 1988 | | 132 | 132 |
| 1993 | 171 | 163 | 334 |
| 1994 | 186 | 173 | 359 |
| 1995 | 157 | 294 | 451 |
| 1996 | 317 | 198 | 515 |
| 1997 | 443 | 123 | 566 |
| 1998 | 404 | 181 | 585 |
| 1999 | 220 | 562 | 782 |
| 2000 | 139 | 267 | 406 |
| 2001 | 366 | 631 | 997 |
| 2002 | 327 | 395 | 722 |
| 2003 | 122 | 302 | 424 |
| 2004 | 126 | 179 | 305 |
| 2005 | 125 | 352 | 477 |
| 2006 | 132 | 263 | 395 |
| 2007 | 237 | 69 | 306 |
| 2008 | 313 | 234 | 547 |
| 2009 | 494 | 327 | 821 |
| 2010 | 687 | 135 | 822 |
| 2012 | 775 | 214 | 989 |
| 2013 | 89 | 383 | 472 |
| 2014 | 153 | 153 | 306 |
| 2015 | 165 | 218 | 383 |
| 2016 | 206 | 273 | 479 |
| 2017 | 220 | 292 | 512 |
| Total | 6574 | 6513 | 13087 |

An estimate of the standard deviation of age-reading error for the entire fishery (east and west combined) was calculated by André Punt (pers. comm. 2018) using data supplied by Kyne Krusic-Golub of Fish Ageing Services Pty Ltd and a variant of the method of Richards *et al.* (1992) (Table 12.8). The implied age distributions are obtained by transforming length frequency data to age data by using the information contained in the conditional age-at-length data from each year.

Table 12.8. Estimated uncertainty in otolith age determinations by age class.

| Age | St. dev. |
|-----|----------|
| 0 | 0.1537 |
| 1 | 0.1537 |
| 2 | 0.2311 |
| 3 | 0.3074 |
| 4 | 0.3828 |
| 5 | 0.4571 |
| 6 | 0.5305 |
| 7 | 0.6028 |
| 8 | 0.6742 |
| 9 | 0.7446 |
| 10 | 0.8141 |
| 11 | 0.8827 |
| 12 | 0.9503 |
| 13 | 1.017 |
| 14 | 1.0828 |
| 15 | 1.1478 |
| 16 | 1.2118 |
| 17 | 1.2751 |
| 18 | 1.3374 |
| 19 | 1.3989 |
| 20 | 1.4597 |
| 21 | 1.5195 |
| 22 | 1.5786 |
| 23 | 1.6369 |

12.2.5.8 Fishery Independent Survey (FIS) estimates

A fishery independent trawl survey (FIS) has been undertaken biennially in winter between 2008 and 2016 (Knuckey *et al.* 2017). Indices from the FIS were re-estimated for the east (SESSF zones 10, 20 and 30) and the west (SESSF zones 40 and 50) with coefficients of variation calculated for each fleet (Table 12.9). At the SERAG meeting in September 2018 it was agreed that FIS would be removed from the base case and instead presented as a sensitivity. The utility of using the length composition data from the FIS to estimate FIS selectivity has yet to be investigated and we assumed the FIS has the same selectivity as the respective trawl fleets.

Table 12.9. FIS derived abundance indices for silver warehou with corresponding coefficient of variation (cv).

| Year | East abundance (CV) | West abundance (CV) |
|------|---------------------|---------------------|
| 2008 | 149.0 (0.576) | 110.7 (0.232) |
| 2010 | 55.6 (0.576) | 25.9 (0.232) |
| 2012 | 218.7 (0.576) | 25.6 (0.232) |
| 2014 | 14.7 (0.576) | 32.2 (0.232) |
| 2016 | 284.8 (0.576) | 44.8 (0.232) |

12.2.5.9 Input data summary

The data used in this assessment is summarised in Figure 12.7, indicating which years the various data types were available.

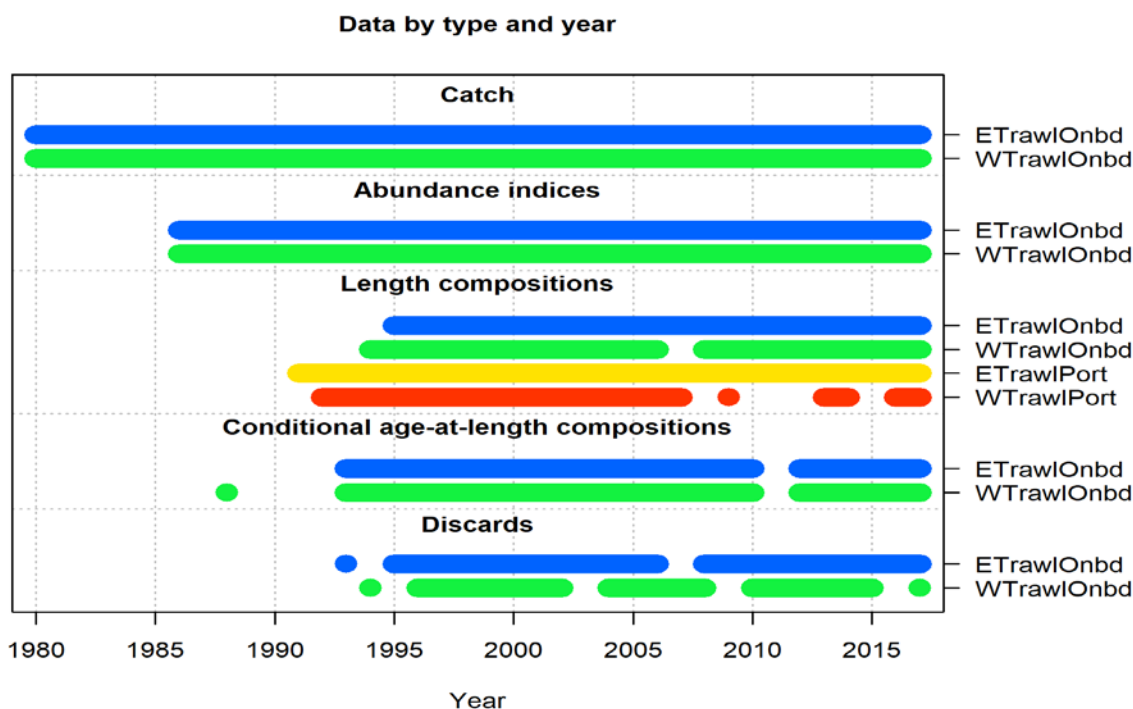


Figure 12.7. Summary of input data used for the silver warehou assessment.

12.2.6 Stock assessment method

12.2.6.1 Population dynamics model and parameter estimation

In 2015, a single-sex single-fleet stock assessment for silver warehou was conducted using the software package Stock Synthesis (version SS-V3.24U, Methot 2015). Stock Synthesis is a statistical age- and length-structured model which can allow for multiple fishing fleets and can be fitted simultaneously to the types of information available for silver warehou. The population dynamics model, and the statistical approach used in the fitting of the model to the various types of data, is outlined fully in the SS3 user manual (Methot *et al.* 2018) and is not reproduced here. This year, the model was translated to the latest version of Stock Synthesis (version SS-V3.30.12, Methot *et al.* 2018). A comparison of parameter estimates and population trajectories were visually indistinguishable between the two versions of Stock Synthesis (see Figure 12.1 to Figure 12.3 Burch *et al.* 2018a).

Some key features of the base case model are:

1. Silver warehou constitute a single stock within the area of the fishery.
2. The eastern (SESSF zones 10, 20 and 30) and western (SESSF zones 40 and 50) trawl fleets were modelled separately with separate catches, catch rates, length frequencies and selectivity.
3. The population was at its unfished (virgin) biomass with the corresponding equilibrium (unfished) age-structure at the start of 1979.
4. Selectivity for the trawl fleets is length-specific, logistic and time-invariant. The two parameters of the selectivity function were estimated within the assessment for each fleet.

5. Retention is estimated separately for two time blocks (1980-2001 and 2002-2014) for each fleet. The slope and intercept parameters were estimated for each fleet, but the asymptote was fixed at 100%. Separate discard time series were used for the east and west trawl fleets assuming a CV of 0.35.
6. The rate of natural mortality, M , is assumed to be constant with age, and also time-invariant. The base-case value for M is 0.30 yr^{-1} .
7. Recruitment to the stock is assumed to follow a Beverton-Holt type stock-recruitment relationship, parameterised by the average recruitment at virgin spawning biomass, R_0 , and the steepness parameter, h . Steepness for the base-case analysis is set to 0.75. Deviations from the average recruitment at a given spawning biomass (recruitment residuals) are estimated for 1980 to 2012. Deviations are not estimated prior to 1980 because there are insufficient data prior to 1980 to permit reliable estimation of recruitment residuals. Deviations are not estimated after 2014 as there would be insufficient numbers of fish recruited to the fishery or seen in the discards to reliably estimate recruitments from 2015 (the age at which 50% of fish have been recruited to the trawl fishery is approximately four). This final year for estimating recruitment deviations is confirmed by observing the increase in asymptotic standard error estimate of the recruitment deviation produced by Stock Synthesis.
8. A plus-group is modelled at age 23 years.
9. Any length frequency data with less than 100 individual fish measured in a year were excluded as unrepresentative. The number of shots was used as the sample size for onboard length frequencies and the number of trips for port based length frequencies, with a cap at 100 shots (although in practice this cap was not needed for this assessment). These sample sizes were then further tuned so that the input sample size was equal to the effective sample size calculated by the model. The tuning procedure is described in Section 12.2.6.2.
10. Onboard and port length frequencies were fitted separately, with a common selectivity estimated from these two sources of length frequency data.
11. Growth of silver warehou is assumed to be time-invariant, in that there is no change over time in mean size-at-age, with the distribution of size-at-age being estimated along with the remaining growth parameters within the assessment. No differences in growth related to gender are modelled, because the stock is modelled as a single-sex.

This forms the base case model for the 2018 silver warehou assessment.

12.2.6.2 Tuning method

Iterative rescaling (reweighting) of input CVs or input effective sample sizes is a repeatable method for ensuring that the expected variation of the different data streams is comparable to what is input (Pacific Fishery Management Council 2016). Sampling standard deviations/ CVs and stage-1 effective sample sizes for most of the data (CPUE, survey indices, composition data) used in fisheries assessments underestimate their true variance by only reflecting measurement or estimation error and not including process (or model) error.

In iterative reweighting, the effective annual sample sizes are tuned/adjusted so that the input sample size is equal to the effective sample size calculated within the model. In SS-V3.30 it is possible to estimate an additional standard deviation parameter to add to the input CVs for the abundance indices (CPUE).

1. Set the standard error for the log of relative abundance indices (CPUE or FIS) to their estimated standard errors to the standard deviation of a loess curve fitted to the original data - which will provide a more realistic estimate compared to that obtained from the original statistical analysis. SS-V3.30 then allows an estimate to be made for an additional adjustment to the relative abundance variances appropriately.

An automated iterative tuning procedure was used for the remaining adjustments. For the recruitment bias adjustment ramps:

2. Adjust the maximum bias adjustment and the start and finish bias adjustment ramps as predicted by SS-V3.30 at each step.

For the age and length composition data:

3. Multiply the initial (stage-1) sample sizes by the sample size multipliers for the conditional age-at-length data using the approach of Punt (2017).
4. Similarly multiply the initial samples sizes by the sample size multipliers for the length composition data using the 'Francis method' (Francis 2011).
5. Repeat steps 2 - 4, until all are converged and stable (with proposed changes $< 1 - 2\%$).

This procedure constitutes current best practice for tuning assessments.

12.2.6.3 Calculating the RBC

The SESSF Harvest Strategy Framework (HSF) was developed during 2005 (Smith *et al.*, 2008) and has been used as a basis for providing advice on TACs in the SESSF quota management system for fishing years 2006- 2012. The HSF uses harvest control rules to determine a recommended biological catch (RBC) for each stock in the SESSF quota management system. Each stock is assigned to one of four Tier levels depending on the quality and quantity of data for that stock. Silver warehou is assessed as a Tier 1 stock and it has an agreed quantitative stock assessment.

The Tier 1 harvest control rule specifies a target and a limit biomass reference point, as well as a target fishing mortality rate. Since 2005 various values have been used for the target and the breakpoint in the rule. For the 2013 TACs AFMA has directed that the 20:40:40 ($B_{lim}:B_{msy}:F_{targ}$) form of the rule will be used up to where fishing mortality reaches F_{48} . Once this point is reached, the fishing mortality is set at F_{48} . Day (2008) has determined that for most SESSF stocks where the proxy values of B_{40} and B_{48} are used for B_{MSY} and B_{MEY} this form of the rule is equivalent to a 20:35:48 strategy.

12.2.6.4 Sensitivity tests

A number of standard sensitivity tests are used to examine the sensitivity of the results of the 2018 base case to some of the assumptions and data inputs:

- a) Include the time series of winter FIS surveys with indices split east.
- b) $M = 0.25$ and 0.35 yr^{-1} .
- c) $h = 0.65$ and 0.85 .
- d) 50% maturity occurs at length 34 and 40cm.

- e) $\sigma_R = 0.6$ and 0.8 .
- f) Recruitment deviations estimated to 2013 and 2015.
- g) Double and halve the weighting on the CPUE series.
- h) Double and halve the weighting on the length composition data.
- i) Double and halve the weighting on the age-at-length data.
- j) Double the reported catch from 1998 to 2002.

The last sensitivity, doubling the reported catch from 1998 to 2002, came about following a suggestion from industry at the September 2015 Slope RAG meeting, to explore the impact of any possible mis-reporting of silver warehou landings in this period.

12.2.6.5 Low recruitment scenarios

To explore the potential impact of setting a multi-year TAC without updating this assessment, scenarios were run where the recruitment in the period from 2015-2023 was assumed to be below average. When the harvest control rules are applied and forward projections are made, recruitment deviations from 2015 onwards are set to zero, as there is insufficient information to estimate recruitment in this period. This essentially assumes average recruitment for the given level of spawning biomass for the period 2015 onwards.

Given that eleven of the last twelve recruitment events are estimated to be below average (Figure 12.16) and that the last two estimated recruitments (2013 and 2014) are just below average and could be revised down in the future with additional data, and given that catches and catch rates have been declining for the last ten years, it seems unlikely that catches will return to the projected RBC levels given in Table 12.12. Indeed, it seems plausible that recruitment may remain below average for the next few years. As such, the SERAG has requested scenarios be considered with below average future recruitment.

12.2.6.6 Retrospective analysis

The last two silver warehou assessments have shown below average recruitment, declines in CPUE and catch even though catches have been well below the TAC. The structure, data and tuning protocols of the silver warehou assessment has changed since Day *et al.* (2012). Because the stock may be on or near the limit reference point we undertake a retrospective analysis (Cadrin and Vaughan 1997, Mohn 1999) to identify whether below average recruitment and declining stock size would have been identified by previous assessments using the same assumptions, data and tuning as the 2018 assessment.

The retrospective analysis was undertaken using the following procedure:

1. Two years of data was removed from the 2018 base case assessment.
2. Time dependent model parameters (e.g. last year of recruitment) were changed to be two years earlier.
3. The model was retuned using the procedure described in Section 12.2.6.2 to create a base case assessment for 2016.
4. Steps 1 – 3 were repeated to create assessments for 2014 and 2012.

Trends in spawning biomass and estimated recruitment are then examined to help understand how reliable the most recent few years of estimated recruitments and spawning biomass are in the 2018 assessment.

12.2.6.7 MCMC analysis for the base case

A Bayesian assessment was undertaken as part of the 2018 silver warehou assessment because the base case assessment showed that the maximum likelihood estimate of spawning biomass was near the limit reference point of 20% of its unfished level between 2013 and 2016, with the lower 95th percent asymptotic confidence intervals being below the limit reference point in those years.

Bayesian frameworks for stock assessment better accommodate the uncertainties relating to a particular model structure and its parameter values, however, they still remain computationally intensive and require the specification of appropriate prior distributions (Punt and Hilborn 1997). Stock Synthesis version 3.30 is programmed using the software Auto Differentiation Model Builder (ADMB, Fournier et al. 2012) version 11.6 which implements Bayesian model frameworks using Markov chain Monte Carlo (MCMC) implemented using the Metropolis-Hastings algorithm (Gelman *et al.* 2013).

We ran seven MCMC chains of the base case model. The starting values of the parameters for each chain were jittered (slightly different between chains). Each chain was run for 2,000,000 iterations with the first 300,000 iterations discarded (the burn-in period). Each chain was thinned by saving every 2,000th iteration and the remaining values used to evaluate chain convergence and estimate model parameters with their associated uncertainty. We then calculated the probability the spawning biomass was above 20% of the unfished level (the limit reference point) between 2012 and 2019.

Convergence of MCMC chains was assessed using the Geweke test (Geweke 1992), the Heidelberger and Welch test (Heidelberger and Welch 1983) and by the examination of trace plots. The R packages r4ss (Taylor et al. 2018) and CODA (Plummer et al. 2006) were used to undertake the tests and create the trace plots presented in Appendix B.

12.2.6.8 Summary statistics

The results of the base-case analysis and the sensitivity tests are summarized using the following quantities:

- | | |
|---------------------------|--|
| a) SB_0 | the average unexploited spawning biomass |
| b) SB_{2019} | the spawning biomass at the start of 2019 |
| c) SB_{2019}/SB_0 | the 2019 spawning biomass expressed as a percentage of the virgin spawning biomass |
| d) $-\ln L$ | the negative of the logarithm of the likelihood function (this is the value minimised when fitting the model, thus a lower value implies a better fit to the data) |
| e) 2019 RBC 20:35:48 | the 2019 RBC calculated using the 20:35:48 harvest rule |
| f) Long term RBC 20:35:48 | the long term RBC calculated using the 20:35:48 harvest rule |

12.3 The 2018 assessment of silver warehou

12.3.1 The base case

12.3.1.1 Transition from the 2015 base case to the 2018 base case

The assessment models presented in Day *et al.* (2015) used data up to 2014. The major changes in the 2018 assessment are: updating the version of Stock Synthesis to version 3.30.12 (Methot *et al.* 2018); the addition of new data for 2015, 2016 and 2017 (including new catch, discard, CPUE, length frequency and age-at-length data); separating catch, catch rate and length data into eastern and western trawl fleets (each with their own selectivity pattern); increasing the second period of size based discarding to end in 2017; adding the 2016 abundance index from the east and west FIS; the estimation of three more years of recruitment; and the implementation of a new tuning procedure. The main change to the assessment procedure and Stock Synthesis that relates to assessments in the SESSF is a revised tuning procedure.

These revisions, with a bridging analysis, were considered by Burch *et al.* (2018a) and showed changes in the time series of spawning biomass including a decline in the estimated 2016 spawning biomass when compared to the 2015 assessment (Day *et al.* 2015). At the September 2018 meeting of SERAG there was discussion of continued below average recruitment estimates, the discarding practices of factory trawlers and the robustness of the silver warehou CPUE and catch data in regard to historical mixed reporting of blue warehou and silver warehou. SERAG accepted the preliminary base case assessment with two modifications: removal of the FIS from the base case (instead including it as a sensitivity) and not estimating the 2015 recruitment.

12.3.1.2 Corrections to the 2018 preliminary base case

Following the September 2018 SERAG meeting investigations into the catch and discard data identified a number of issues that resulted in modifications to the catch, discard and length frequency presented in preliminary base case assessment (Burch *et al.* 2018a). These changes are outlined below and described in more detail elsewhere in this report.

- a) Correction of an error in the catch time series that arose from the removal of discarded catches in the 2015 assessment (Section 12.3.1.3).
- b) Updating the catches from 1994 onwards to account for updates to the database made by AFMA. Catches prior to 1994 were retained due to problems in distinguishing silver warehou from blue warehou in logbooks (Section 12.3.1.3).
- c) Inclusion of catches from the GHAT and the SPF in the assessment (Section 12.3.1.3).
- d) Removal of length frequency data from the SPF in 2015 and 2016.
- e) Separating the estimates of discarded catch into eastern and western trawl fleets (the 2015 assessment used combined series) and updating time series.
- f) Incorporating discarded catch estimates from factory trawlers into overall discard estimates where these vessels had ISMP observer coverage.
- g) Assuming a lognormal error structure when fitting to the estimated discard fractions in the assessment (previously Normal errors were incorrectly assumed).

Along with the removal of the FIS from the base case and not estimating recruitment in 2015, these changes resulted in slight differences in the time series of relative spawning biomass (Figure 12.8). One difference between the final base case and the preliminary assessment presented to the September

2018 SERAG meeting is that the 2014 recruitment is now below average, when previously it was average (Figure 12.9). Note that the 2015 recruitment is not estimated in the final base case.

These results were communicated to AFMA and the SERAG chair in October and this model has been used as the base case for the 2018 silver warehou stock assessment.

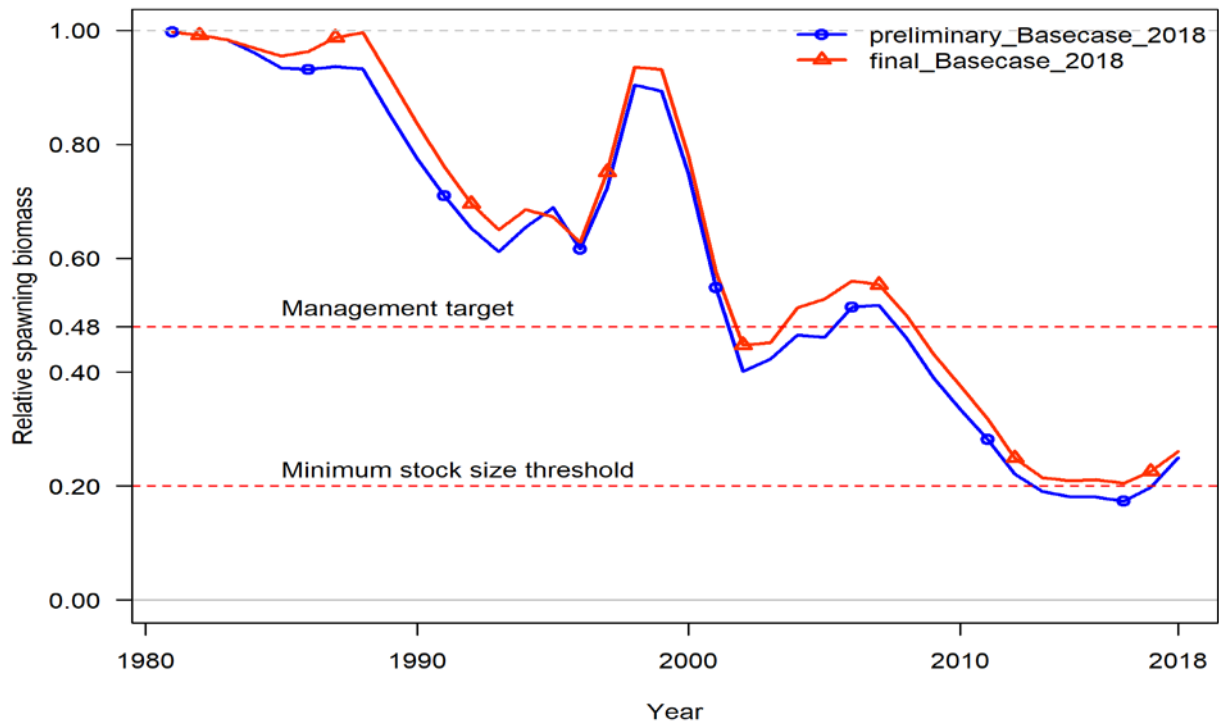


Figure 12.8. Comparison of the time-series of relative spawning biomass for the 2018 preliminary base case assessment from Burch *et al.* (2018a) and the final 2018 base case.

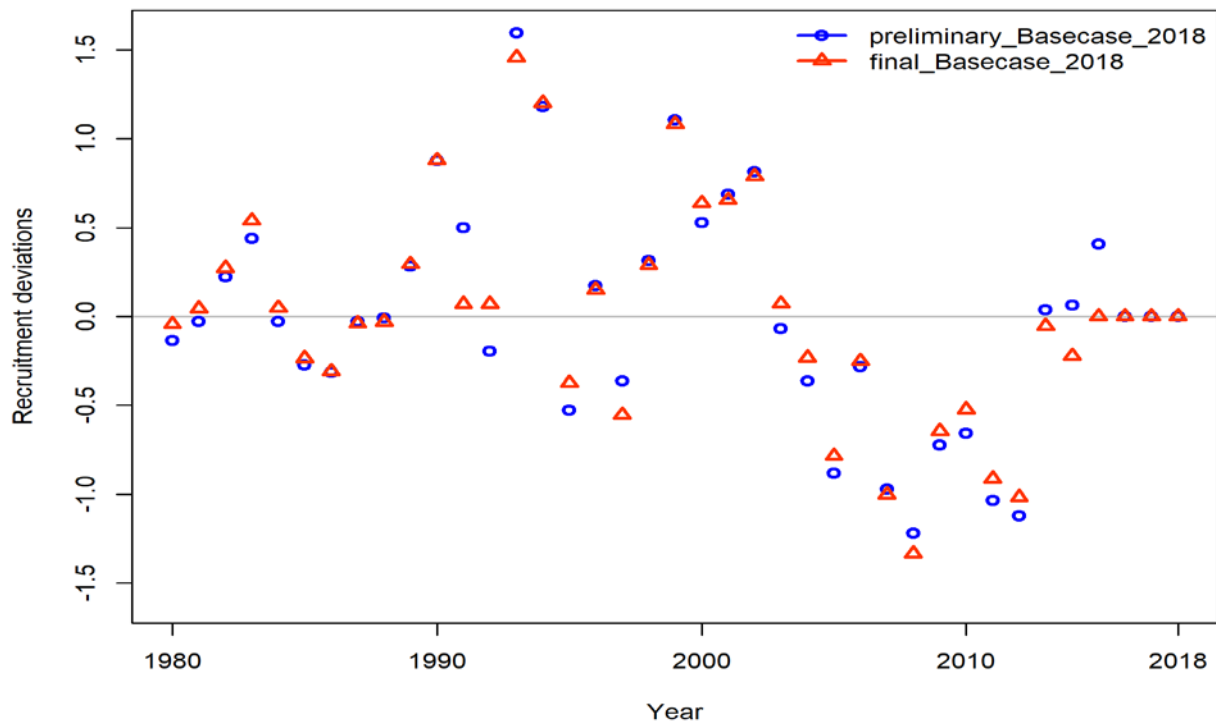


Figure 12.9. Comparison of the recruitment time series for the 2018 preliminary base case assessment from Burch *et al.* (2018a) with revision presented to SERAG in September and the final 2018 base case.

12.3.1.3 Parameter estimates of the base case model

Figure 12.10 shows the estimated growth curve for silver warehou. All growth parameters are estimated. The estimates of the growth parameters are: (a) $L_{min}=14.82\text{cm}$, (b) $L_{max}=51.21\text{cm}$, (c) $K=0.312\text{ yr}^{-1}$, and (d) $\text{cv of growth} = 0.0808$. This growth curve is very similar to the growth curve estimated in the 2015 assessment.

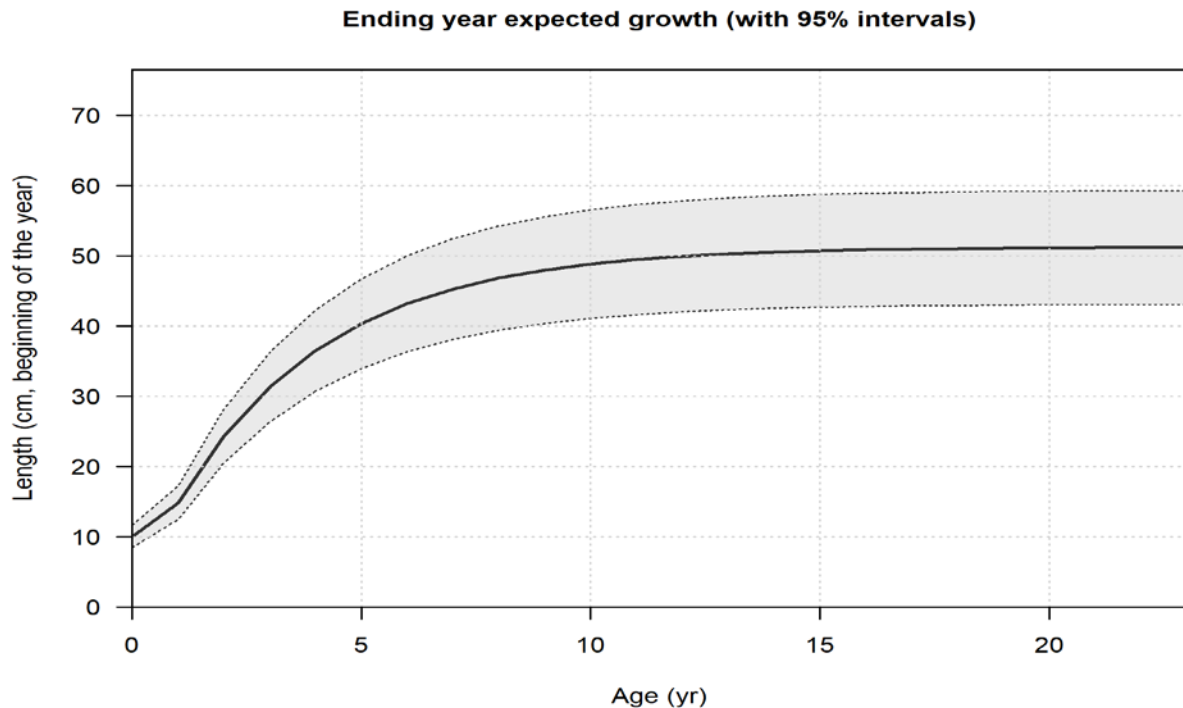


Figure 12.10. The model estimated growth function for silver warehou for the base case.

Figure 12.11 shows the estimated time varying retention and selectivity curves for the east and west trawl fleets for silver warehou. The parameters that define this selectivity function including the length at 50% selection and the spread. The estimates of these parameters for the base-case analysis are 22.82cm and 3.48cm respectively for the east and 39.87cm and 11.24cm for the west (Table 12.1). While length at 50% selection has remained stable in the west compared to the 2015 assessment (39.59cm) it has declined almost 2cm in the east (from 24.66cm).

The estimates for the parameters that defines in the time block 2002-2017 are 29.29cm and 3.29cm respectively for the east and 28.72cm and 4.84cm for the west. The estimate of the parameter that defines the initial numbers (and biomass), $\ln(R_0)$, is 9.379 for the base case.

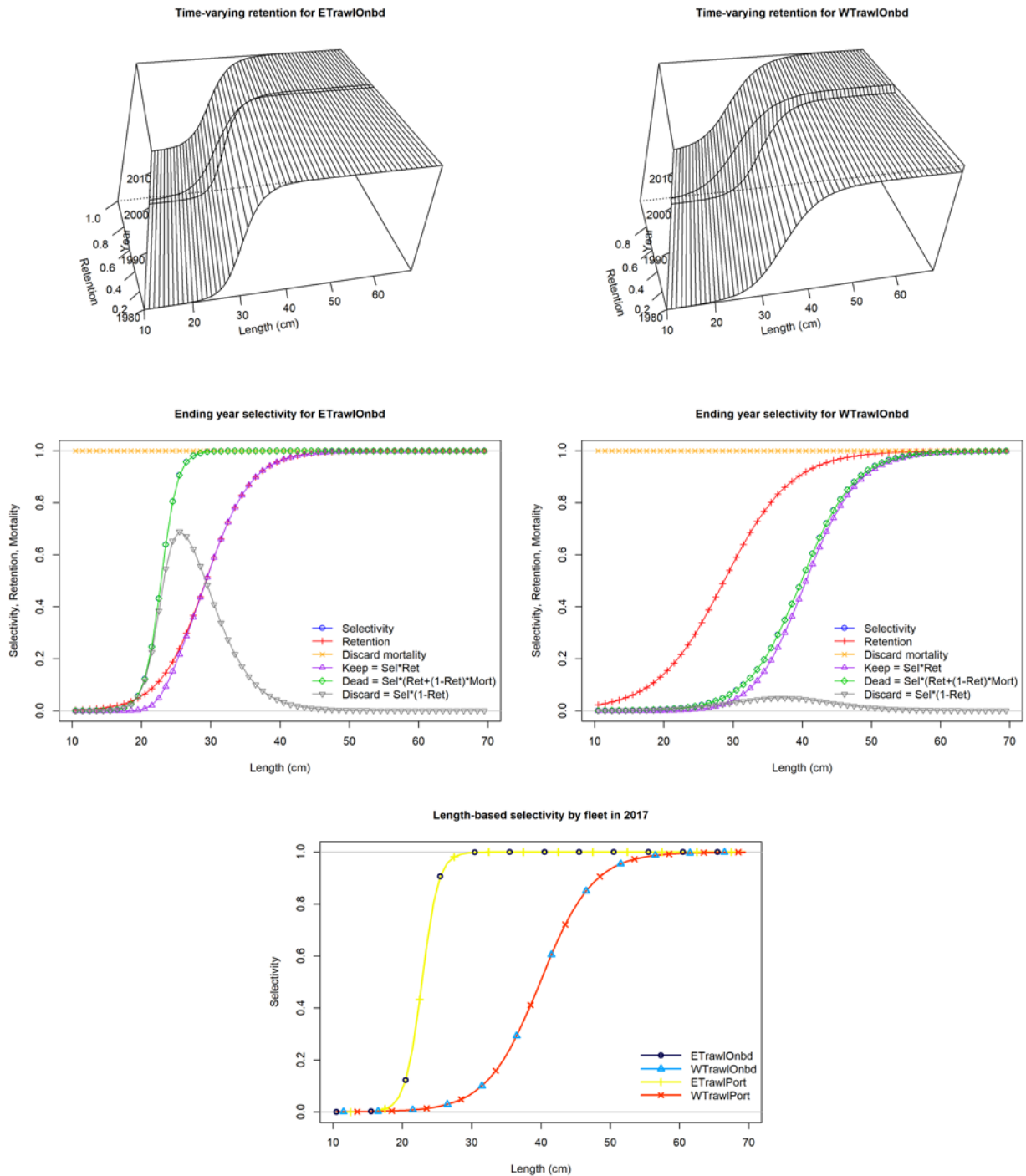


Figure 12.11. Estimated retention function (discard pattern) with two time blocks (1980-2001, and 2002-2017) for east (top left) and west (top right) trawl fleets. Selectivity and discard patterns for east (middle left) and west (middle right) fleets, and east and west selectivity patterns plotted together (bottom).

12.3.1.4 Fits to the data for the base case model

The fits to the catch rate indices for the base case are good for the west trawl fleet and poor for the east trawl fleet (Figure 12.12). For the east trawl fleet, the model underestimates CPUE between 1986 and 1995, then overestimates CPUE from 1996 to 2008, between 2009 and 2014 the fit is good and then the model overestimates the last three data points (2015, 2016, 2017). Fits to CPUE for the west trawl

fleet are good, with the model following trends in CPUE with the exception of 2002 – 2005 and the two most recent years (2016 and 2017) which it overestimates. Overestimation of the most recent CPUE data points was also seen in previous assessments (Day *et al.* 2012, 2015).

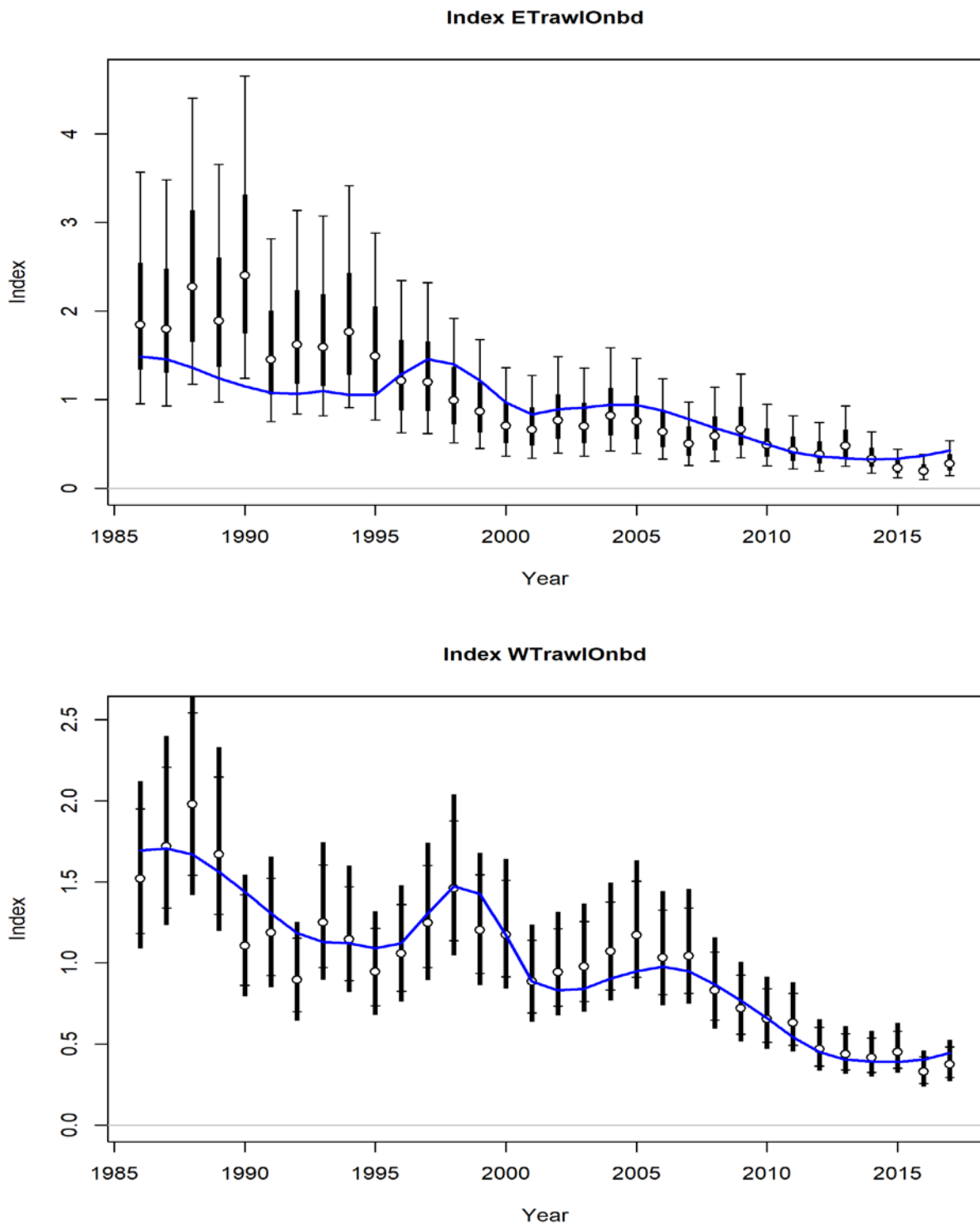


Figure 12.12. Observed (circles) and model-predicted (blue line) catch-rates for silver warehou for the east (top) and west (bottom) trawl fleet versus year for the base case analysis. The vertical lines indicate approximate 95% confidence intervals for the data.

Difference in the length frequency distributions between port and onboard samples, particularly in east, means that it is difficult for the assessment to fit the aggregated retained length frequency (Figure 12.13). The eastern trawl retained fits are not as good as the western trawl, and the western trawl discard length frequencies are quite variable and hence difficult to fit well. The discard length frequencies in the east fit quite well (Figure 12.13). These patterns are very similar to those in the last assessment (Day *et al.* 2015).

Annual fits and residuals are included in Appendix A. While the annual fits are not as good as the aggregated fits, the length frequency data appears to be very variable, especially for the eastern trawl fleet. This may reflect spatial and temporal differences in collection of this data between years and hence this length frequency data may not be as representative as we would like, similar to those of the 2015 assessment. The implied fits to age show similar patterns to the length data, with better fits in the west, than the east, but given this implied fit to age is derived from length frequency data, it is not surprising to see similar trends.

The onboard sampling data contains some fish < 20cm for the eastern discards in 2006, 2013 and 2015 and in the eastern and western retained catch in 2015. The 2006 and 2013 discards also appeared in the 2015 assessment and, although they are not fitted to by either assessment, should be investigated for future assessments.

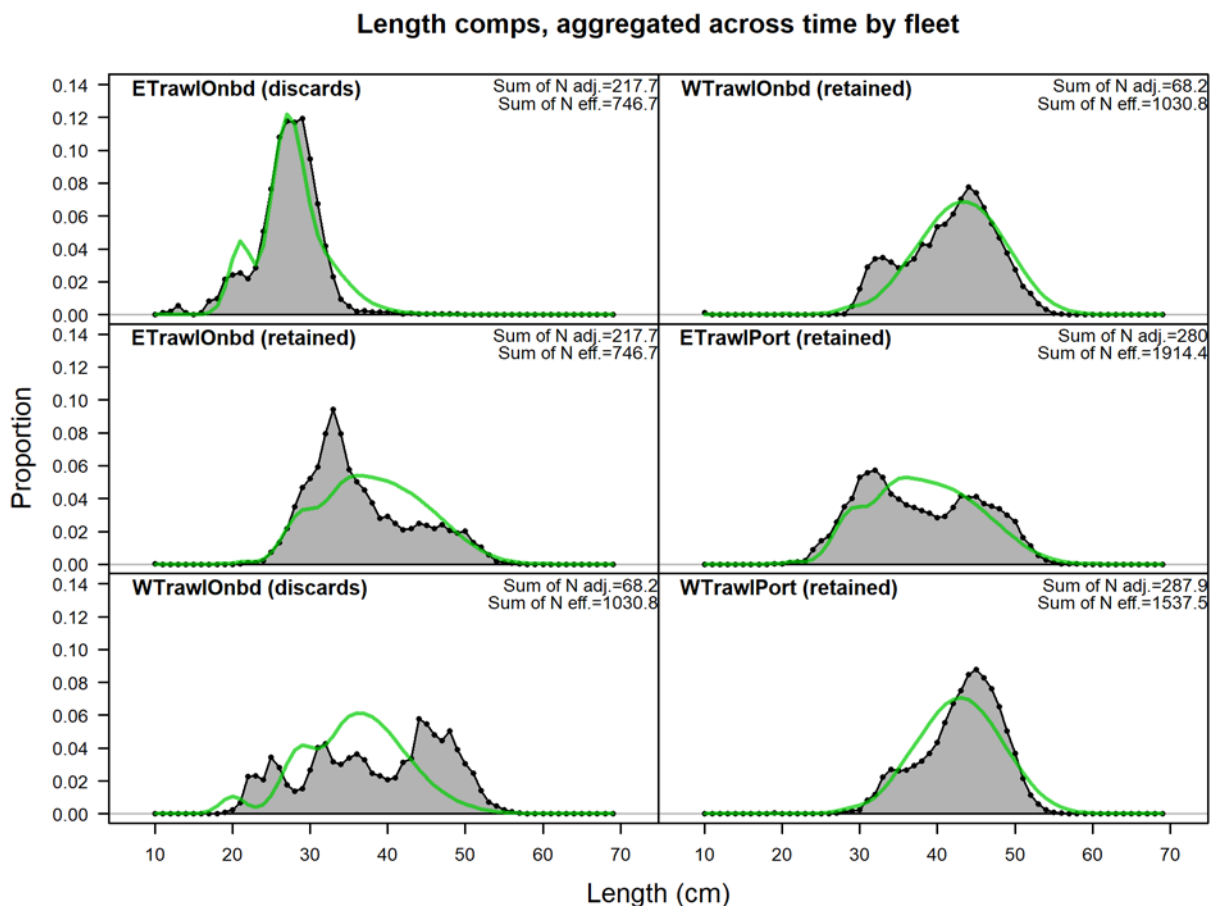


Figure 12.13. Aggregated length compositions for the onboard and port data sources in both the east and west. Observed data are grey and the fitted value is the green line.

The fits to the discard fractions are poor in the east where the model has not fitted to annual discard rates above 25% (Figure 12.14). In the west, fits to discard fractions are somewhat better, particularly after 2002, however the assessment still fails to fit the two annual discard fractions above 20%. Inclusion of the discard data split into eastern and western trawl fleets resulted in difficulties with model convergence which suggests a conflict between the discard data and other data sources within the model. The difficulties with convergence were resolved by increasing the CV on the discards from 0.25 to 0.35 and retuning the model after each model or data change to the preliminary base case assessment presented to the September 2018 SERAG meeting.

Given the extreme variability in annual discard fractions, particularly in the east, it is perhaps not surprising the assessment fails to fit the annual discard fractions >25% on the east and 20% in the west. This, along with the apparent increase in discarding in the east from 2015 – 2017 suggests that discarding of silver warehou and its impact on the assessment requires further exploration in future assessments.

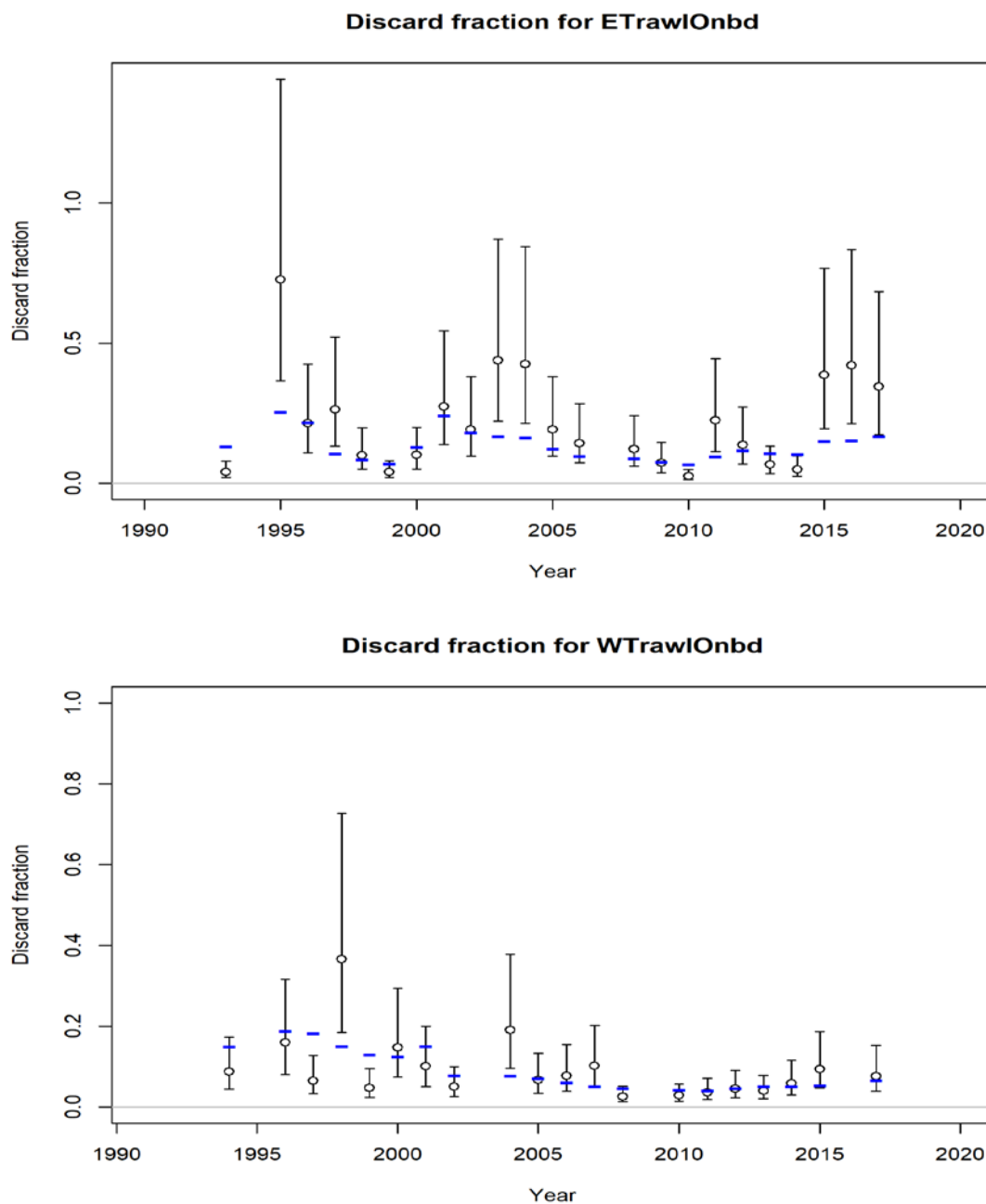


Figure 12.14. Observed (circles) and model-estimated (blue lines) discard estimates versus year, with approximate 95% asymptotic intervals for the Eastern trawl fleet (top) and western trawl fleet (bottom).

12.3.1.5 Assessment outcomes for the base case model

Figure 12.15 shows the relative spawning stock biomass with the limit and target reference points at 20% and 48% respectively. Stock size began to decline in the late 1980s concurrent with the increase in catches. Above average recruitment in the early 1990s (Figure 12.18) saw stock size increase towards the end of the 1990s even as catches reached 2000t per annum. Declines in stock size in the early 2000s are associated with catches peaking at between 3000 and 4000t and saw the stock drop below the target reference point in 2002 and 2003. Another period of above average recruitment in the late 1990s / early 2000s saw stock size increase in the mid-2000s. From 2007 – 2013 stock size

declined even though catches were also declining over this period. Between 2013 and 2016 the stock has remained just above the limit reference point before increasing in 2017. This increase in stock size towards the end of the series should be treated with some caution as this is a result of the model imposed average recruitment from 2014 onwards, when recruitment is unable to be estimated reliably. As data becomes available to inform these recruitment events in future assessments, the increase in stock size from 2017 may be revised.

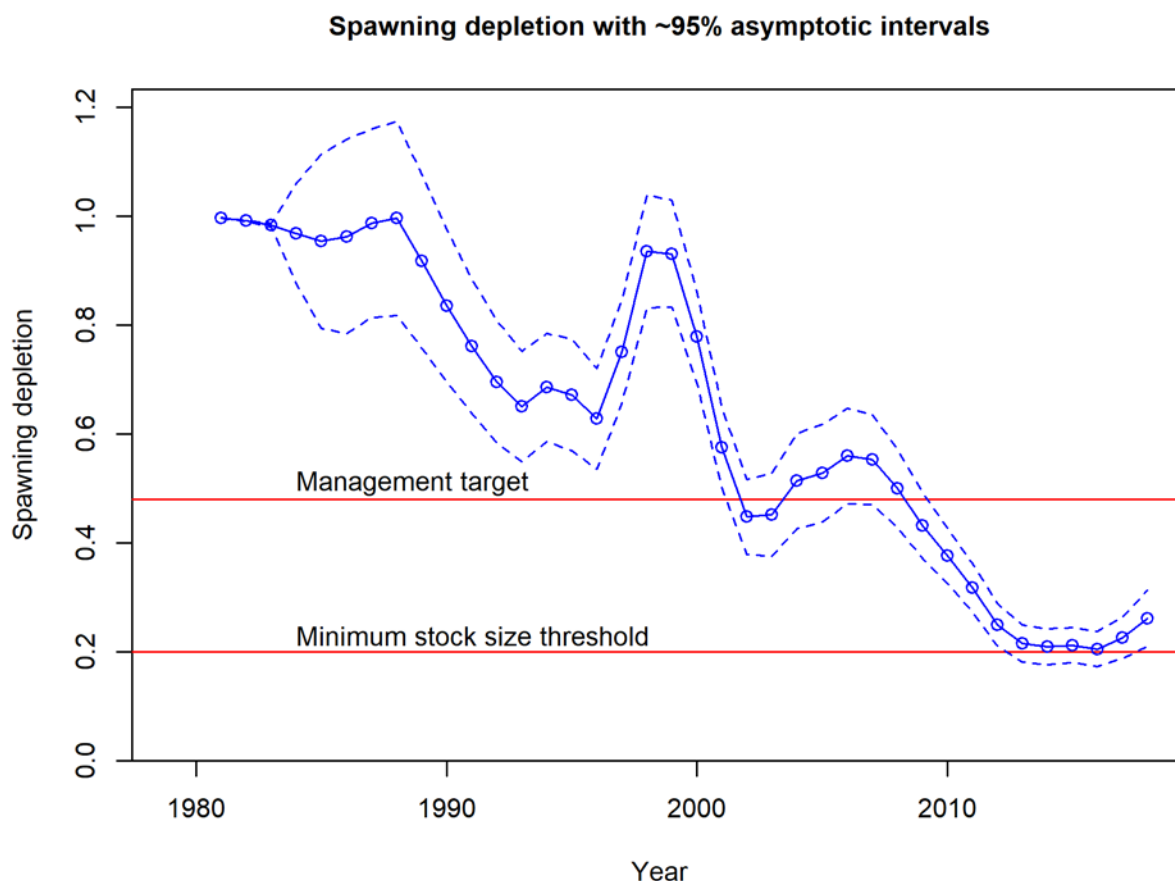


Figure 12.15. Time-trajectory of spawning biomass expressed as a percentage of virgin (with 95% confidence intervals) corresponding to the MPD estimates for silver warehou.

The time-trajectories of recruitment deviations are shown in Figure 12.16 and the bias adjustment and standard errors of recruitment deviation estimates are shown in Figure 12.17. Note that the last 11 estimated recruitment events have been below average. While the most recent two estimated recruitments are higher than those in preceding years, recent recruitments have been revised downwards in subsequent assessments (Day *et al.* 2012, 2015).

The current (2019) spawning stock biomass is estimated to be 31% of unfished stock biomass (i.e. 2019 spawning biomass relative to unfished spawning biomass).

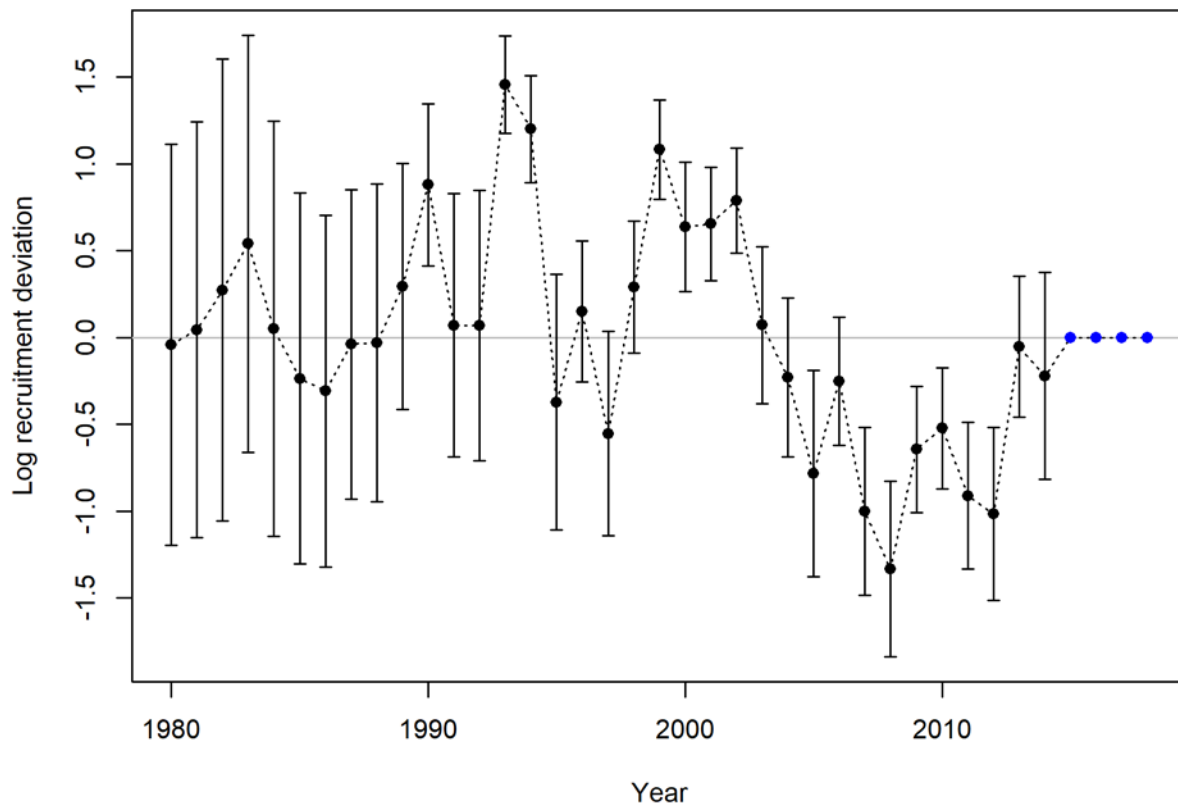


Figure 12.16. Recruitment estimates for the base case for silver warehou. Time trajectories of estimated log recruitment deviations, with approximate error distributions.

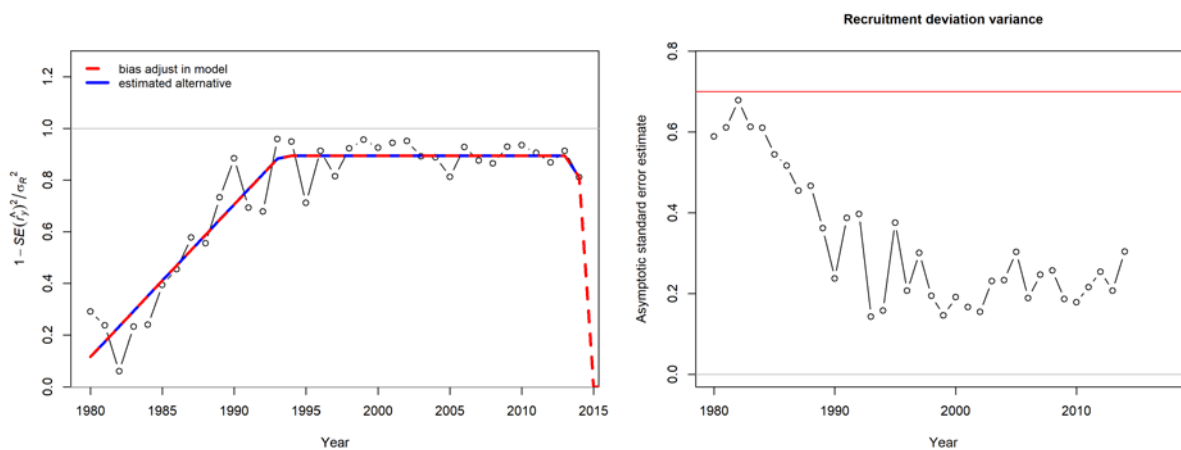


Figure 12.17. Bias adjustment (left) and standard errors of recruitment deviation estimates (right) for the base case for silver warehou.

12.3.2 Sensitivities and alternative models

Results of the sensitivity tests are shown in Table 12.10 and Table 12.11. None of the sensitivities gave relative spawning biomass estimates below the limit reference point or above the target reference point. For the fixed parameters, the results are most sensitive to the assumed value for natural mortality (M). However, even with $M=0.35$, the improved fits to the survey, discard and age data give an improvement to the overall likelihood of only nine units. Changing the size at 50% maturity changes the spawning biomass relative to the unfished level but has no impact on the model fit ($\Delta LL=0$). Including the FIS results in a slightly poorer fit and a relative spawning biomass level 1% higher than the base case. While the 2015 assessment was sensitive to changing σ_R it had little effect on the 2018 assessment.

Changing the weighting on various data sources has relatively large impacts on the relative spawning biomass estimates. The likelihood cannot be compared directly in these cases, but Table 12.10 shows the relative differences between the different components of the total likelihood, attributable to these changes. The estimated relative spawning biomass is 38% of virgin when catches from 1998 – 2002 are doubled, 36% when the weight on the CPUE is halved and 35% when recruitment is only estimated to 2013.

Table 12.10. Summary of results for the base case and sensitivity tests (log-likelihood (LL) values that are comparable are in bold face). Spawning stock biomass includes both male and female biomass in the total.

| Model | LL | ΔLL | SB ₀ | SB ₂₀₁₉ | SB ₂₀₁₉ /SB ₀ | 2019 RBC (t) | long term RBC (t) |
|------------------------------|------------|-------------|-----------------|--------------------|-------------------------------------|--------------|-------------------|
| base case ($M=0.3, h=0.7$) | 625 | - | 18949 | 5930 | 0.31 | 942 | 1773 |
| FIS | 627 | 2 | 18996 | 6021 | 0.32 | - | - |
| $M=0.25$ | 636 | 11 | 19368 | 5127 | 0.26 | - | - |
| $M=0.35$ | 616 | -9 | 20290 | 7340 | 0.36 | - | - |
| $h=0.65$ | 625 | 0 | 19690 | 5731 | 0.29 | - | - |
| $h=0.85$ | 625 | 0 | 18438 | 6120 | 0.33 | - | - |
| 50% maturity at 34cm | 625 | 0 | 20652 | 7170 | 0.35 | - | - |
| 50% maturity at 40cm | 625 | 0 | 16670 | 4545 | 0.27 | - | - |
| $\sigma_R = 0.6$ | 626 | 1 | 18222 | 5999 | 0.33 | - | - |
| $\sigma_R = 0.8$ | 625 | 0 | 19908 | 5919 | 0.3 | - | - |
| est. recruitment to 2013 | 626 | 1 | 19058 | 6682 | 0.35 | - | - |
| est. recruitment to 2015 | 625 | 0 | 19038 | 5920 | 0.31 | - | - |
| double weight on CPUE | 550 | -75 | 18848 | 5404 | 0.29 | - | - |
| halve weight on CPUE | 657 | 32 | 19426 | 6910 | 0.36 | - | - |
| double weight on lengths | 846 | 221 | 18658 | 6180 | 0.33 | - | - |
| halve weight on lengths | 507 | -118 | 19562 | 6031 | 0.31 | - | - |
| double weight on age | 921 | 296 | 19376 | 5922 | 0.31 | - | - |
| halve weight on age | 471 | -154 | 18667 | 5969 | 0.32 | - | - |
| double catch 1998-2002 | 643 | 18 | 26990 | 10390 | 0.38 | - | - |

Table 12.11. Summary of likelihood components for the base case and sensitivity tests. Likelihood components are unweighted and all cases below the primary base case are shown as differences from the base case. A negative value either in the total or individual components of likelihood indicates an improvement in fit compared to the primary base case. A positive value indicates deterioration in the fit.

| Model | Likelihood | | | | | | |
|------------------------------------|------------|-------------|--------|---------|--------|--------|-------------|
| | TOTAL | Δ LL | Survey | Discard | Length | Age | Recruitment |
| base case ($M=0.3$, $h=0.7$) | 625.08 | 0 | -68.66 | 23.69 | 362.17 | 302.11 | 5.39 |
| FIS | 626.76 | 1.68 | 1.45 | -0.07 | 0.87 | -0.48 | -0.09 |
| $M=0.25$ | 635.77 | 10.69 | 4.61 | 1.07 | 2.09 | 2.45 | 0.34 |
| $M=0.35$ | 616.41 | -8.67 | -4.07 | -0.99 | -2 | -1.78 | 0.33 |
| $h=0.65$ | 624.77 | -0.31 | -0.33 | 0.62 | 0.18 | -0.11 | -0.66 |
| $h=0.85$ | 625.46 | 0.38 | 0.27 | -0.51 | -0.07 | 0.09 | 0.6 |
| 50% maturity at 34cm | 625.04 | -0.04 | 0 | -0.03 | -0.02 | 0.01 | 0.01 |
| 50% maturity at 40cm | 625.03 | -0.05 | 0.03 | 0.05 | -0.07 | 0.02 | -0.07 |
| $\sigma_R = 0.6$ | 626.48 | 1.4 | 0.9 | 0.32 | -0.12 | -0.06 | 0.36 |
| $\sigma_R = 0.8$ | 624.91 | -0.17 | -0.53 | -0.26 | 0.08 | 0.11 | 0.44 |
| est. recruitment to 2011 | 626.39 | 1.31 | 1.73 | -1.79 | -0.5 | 1.64 | 0.23 |
| est. recruitment to 2013 | 624.85 | -0.23 | -0.01 | 0.03 | -0.02 | 0.01 | -0.22 |
| double weight on CPUE | 629.56 | 4.48 | -10.47 | 5.13 | 9.93 | -2.35 | 2.28 |
| halve weight on CPUE | 627.95 | 2.87 | 10.6 | -2.52 | -7.97 | 4.34 | -1.6 |
| double weight on lengths | 565.29 | -59.80 | 10.81 | 2.88 | -81.31 | 8.44 | -0.67 |
| halve weight on lengths | 761.02 | 135.94 | -5.19 | -1.69 | 146.73 | -4.12 | 0.28 |
| double weight on age | 629.59 | 4.505 | -3.22 | 5.35 | 13.24 | -10.72 | -0.1 |
| halve weight on age | 629.50 | 4.42 | 4.34 | -4.47 | -10.73 | 14.97 | 0.28 |
| double catch 1998- 2002 | 643.17 | 18.09 | 18.06 | -1.13 | 0.46 | 0.91 | -0.34 |

12.3.3 Application of the harvest control rules in 2018

An estimate of the catch for the 2018 calendar year is needed to run the model forward to calculate the 2019 spawning biomass and percentage of the unfished spawning biomass. Given that recent TACs have been under-caught and catches have been stable in the most recent four years, the catch in 2018 is assumed to equal that of 2017 (348.1t). The assessment estimates that percentage of the unfished spawning biomass was just above the 20% limit reference point between 2013 and 2016 with the lower 95% asymptotic confidence intervals being below the limit reference point between 2013 and 2017 (Figure 12.18).



Figure 12.18. The projection of relative spawning biomass under the 20:35:48 rule for silver warehou.

The percentage of the unfished spawning biomass in 2019 under the base-case parameterisation is estimated to be 31.3%.

An application of the Tier 1 harvest control rule with a target spawning biomass of 48% of unfished levels leads to the 2019 and long-term RBCs of 942t and 1,772t (Table 12.10). An example of the time-series of RBCs and corresponding spawning biomass corresponding to the calculated RBCs for the 20:35:48 harvest control rule is shown in Figure 12.18. Table 12.12 shows the annual RBCs and percentage of the unfished spawning biomass estimates under the 20:35:48 harvest control rule.

Model estimated discard rates for 2019-2021 are required for calculation of the TAC from the RBC, and these can be obtained from Stock Synthesis output files. Under the assumption of average recruitment from 2015 onwards and assuming that the RBC is caught in full each year, the estimated discard mass for these years follow: 88t in 2019; 122t in 2020; and 126t in 2021 (Table 12.12).

Table 12.12. Summary of the annual percentage of the unfished spawning biomass, RBCs and estimated discard mass for the base case under the 20:35:48 harvest control rule.

| Year | Percentage $SSB_{current}/SSB_0$ | RBC (t) | Discard mass (t) |
|------|-------------------------------------|---------|---------------------|
| 2019 | 31.3 | 942.1 | 88.1 |
| 2020 | 35.4 | 1352.8 | 121.9 |
| 2021 | 37.4 | 1419.6 | 125.8 |
| 2022 | 39.0 | 1477.0 | 129.5 |
| 2023 | 40.4 | 1528.4 | 132.6 |

12.3.4 Scenarios with low recruitment for 2015-2023

12.3.4.1 Poor and very poor recruitment scenarios

To explore the possible impact of continued poor recruitment, two additional recruitment scenarios were examined where recruitment was assumed to be poor in the period 2015-2023. In this case, the standard forward projections, assuming average recruitment, could produce RBCs that, if caught, could result in a lower spawning biomass than the target level. The first recruitment scenario, referred to as “poor recruitment” took the mean of the log recruitment deviations in the base case estimated from 2010-2014, giving a value of -0.545. This represented a recent period of five poor recruitment events. The second recruitment scenario, referred to as “very poor recruitment” took the mean of the log recruitment deviation from the worst three of these years, 2010-2012, giving a value of -0.817. The recruitment estimates from the poor and very poor recruitment scenarios are shown in Figure 12.19.

The same scenarios were explored for the 2015 assessment (Day *et al.* 2015) with the mean log recruitment deviations from the last five estimated recruitments was -0.576 while the mean of the lowest three of the last five estimated recruitments was -0.799.

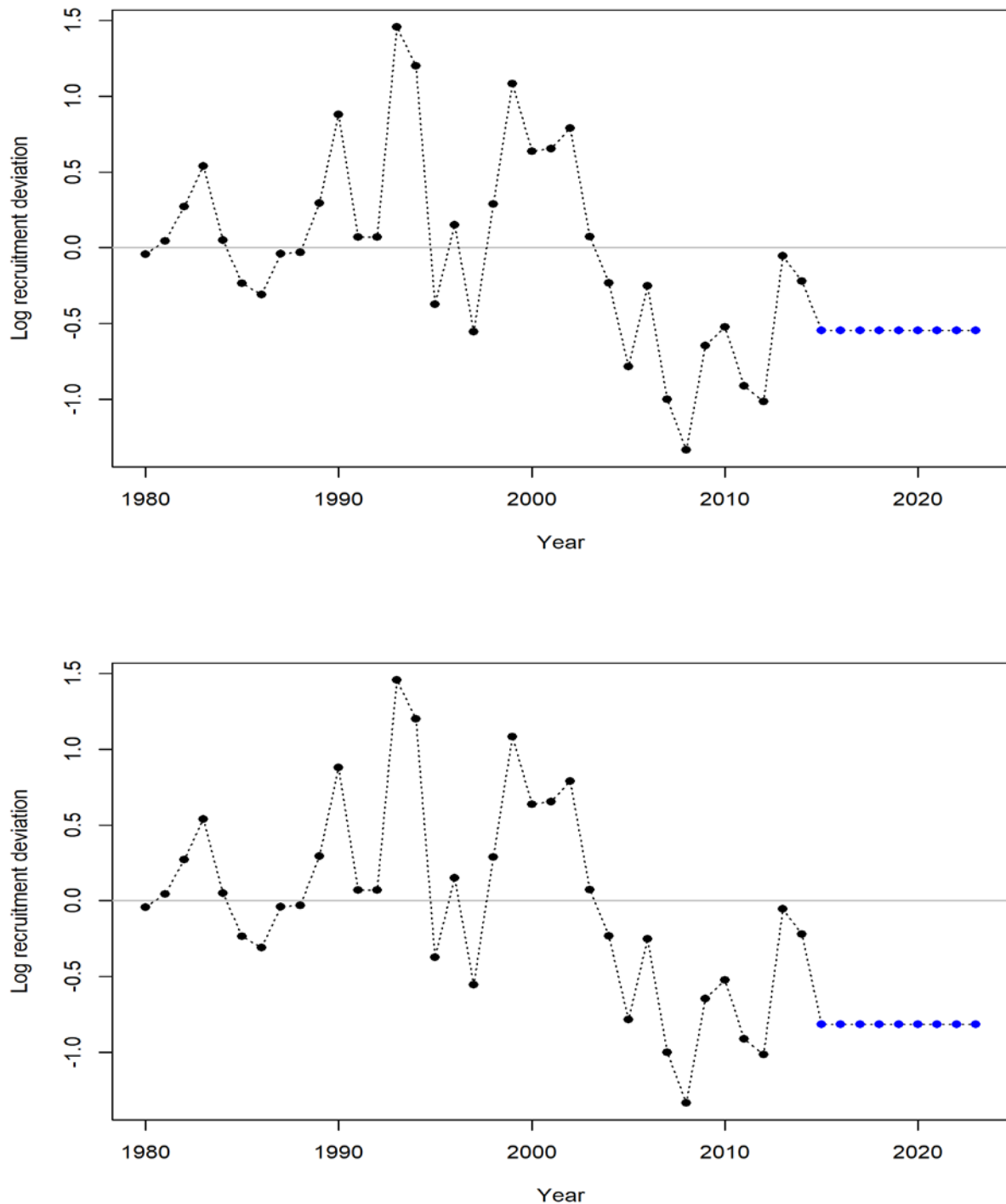


Figure 12.19. Time trajectories of log recruitment deviations estimates for the scenario with poor recruitment (top) and very poor recruitment (bottom).

12.3.4.2 Fixed catch projection to 2023

For the two poor recruitment scenarios, the dynamics were projected forward for five additional years, initially with a fixed catch level, set at the 2017 catch, 348t. Note that with discards being estimated, there are additional removals, and while the forecast landed catch is set to 348t, the actual total catch

is somewhat higher due to the inclusion of discards. The 2017 catch has been chosen for these scenarios rather than the RBC because the RBC has not been caught for a number of years (Table 12.1). Scenarios and percentages of the unfished spawning biomass are shown in Figure 12.20, Table 12.13 and Table 12.14.

At its November 2018 meeting, SERAG agreed to recommend a TAC for silver warehou based on the assumption that recruitment will remain below average in the next few years. SERAG chose to assume that recruitment would remain at the mean of the last five years of estimated recruitments in the base case model (2010 – 2014). SERAG requested additional constant catch projections for the poor recruitment scenario of between 400 t and 750 t as well as the RBCs from the base case model which assumes average recruitment (942 t in 2019).

Under the poor recruitment scenario, constant annual catches of 750 t or more led to the estimated spawning biomass declining under the low recruitment scenario (Table 12.13). For constant annual catches between 348 t and 600 t spawning biomass is predicted to increase, albeit more slowly than the base case which assumes average recruitment (Table 12.13). In the very poor recruitment scenario with a constant annual catch of 348 t the estimated spawning biomass remains around 27% between 2019 and 2023 (Table 12.14).

Table 12.13. Estimated percentage of virgin spawning biomass assuming poor recruitment from 2015-2023 from a series of fixed landed catches between 348t and the catches from the base case model assuming average recruitment (denoted with *). Estimates of total and discarded catch are provided for each scenario.

| Catch scenario | Year | Percentage | | |
|----------------|------|--|-----------------|---------------------|
| | | SSB _{current} /SSB ₀ | Total catch (t) | Discarded catch (t) |
| 348 | 2019 | 28.4 | 376 | 28 |
| 348 | 2020 | 30.1 | 375 | 27 |
| 348 | 2021 | 31.4 | 375 | 27 |
| 348 | 2022 | 32.6 | 374 | 26 |
| 348 | 2023 | 33.9 | 374 | 26 |
| 400 | 2019 | 28.4 | 432 | 32 |
| 400 | 2020 | 29.9 | 431 | 31 |
| 400 | 2021 | 31.0 | 431 | 31 |
| 400 | 2022 | 32.0 | 431 | 31 |
| 400 | 2023 | 33.1 | 430 | 30 |
| 450 | 2019 | 28.4 | 486 | 36 |
| 450 | 2020 | 29.6 | 486 | 36 |
| 450 | 2021 | 30.5 | 485 | 35 |
| 450 | 2022 | 31.4 | 485 | 35 |
| 450 | 2023 | 32.3 | 484 | 34 |
| 500 | 2019 | 28.4 | 540 | 40 |
| 500 | 2020 | 29.4 | 540 | 40 |
| 500 | 2021 | 30.1 | 539 | 39 |
| 500 | 2022 | 30.7 | 539 | 39 |
| 500 | 2023 | 31.5 | 539 | 39 |
| 550 | 2019 | 28.4 | 594 | 44 |
| 550 | 2020 | 29.2 | 594 | 44 |
| 550 | 2021 | 29.6 | 594 | 44 |
| 550 | 2022 | 30.1 | 594 | 44 |
| 550 | 2023 | 30.7 | 593 | 43 |
| 600 | 2019 | 28.4 | 649 | 49 |
| 600 | 2020 | 29.0 | 648 | 48 |
| 600 | 2021 | 29.2 | 648 | 48 |
| 600 | 2022 | 29.5 | 648 | 48 |
| 600 | 2023 | 29.9 | 648 | 48 |
| 750 | 2019 | 28.4 | 811 | 61 |
| 750 | 2020 | 28.3 | 811 | 61 |
| 750 | 2021 | 27.9 | 812 | 62 |
| 750 | 2022 | 27.6 | 812 | 62 |
| 750 | 2023 | 27.5 | 813 | 63 |
| 854* | 2019 | 28.4 | 924 | 70 |
| 1231* | 2020 | 27.8 | 1333 | 102 |
| 1294* | 2021 | 25.3 | 1408 | 114 |
| 1348* | 2022 | 22.7 | 1474 | 127 |
| 1396* | 2023 | 20.4 | 1535 | 139 |

Table 12.14. Estimated percentage of virgin spawning biomass assuming very poor recruitment from 2015-2023 assuming a fixed annual catch of 348t. Estimates of total and discarded catch are provided.

| Catch scenario | Year | Percentage | | |
|----------------|------|--|-----------------|---------------------|
| | | SSB _{current} /SSB ₀ | Total catch (t) | Discarded catch (t) |
| 348 | 2019 | 27.5 | 373 | 25 |
| 348 | 2020 | 27.6 | 373 | 24 |
| 348 | 2021 | 27.4 | 373 | 24 |
| 348 | 2022 | 27.2 | 373 | 25 |
| 348 | 2023 | 27.2 | 373 | 25 |

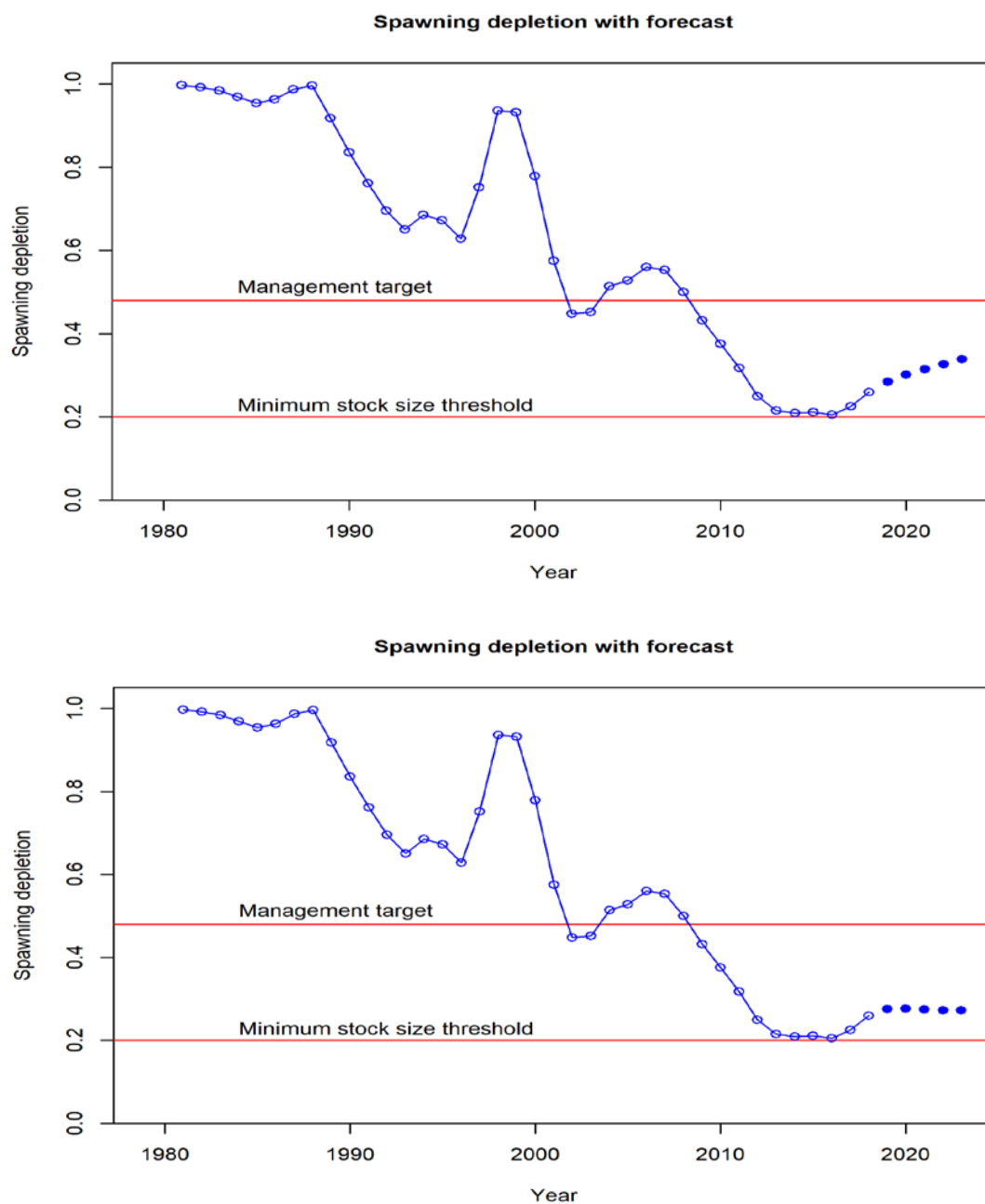


Figure 12.20. The poor (top) and very poor (bottom) recruitment scenario projections of relative spawning biomass with fixed catch of 348t (2017 catch).

12.3.4.3 Retrospective analysis

Given the changes to the silver warehou assessment since 2012, a retrospective analysis was undertaken to identify whether the downward revision of recent recruitments and upward trends in the percentage of the unfished spawning biomass seen in previous assessments (Day *et al.* 2012, 2015) was present with the 2018 assessment structure, data and tuning methods. For the 2016 and 2014 assessment scenarios, an increase in estimated percentage of the unfished spawning biomass in the final two or three years of the assessment is observed (Figure 12.21), however, this pattern was not present in the 2012 assessment scenario. It is interesting to note that the assessment appears to have

shifted to a lower productivity state between the 2014 and 2016 scenarios. Figure 12.22 shows a similar pattern with the estimated recruitment deviations, with recruitments from the 2014 and 2016 scenarios being revised downwards in subsequent assessments, while recruitments from the 2012 scenario changed little in subsequent assessments.

This analysis corroborates the pattern of overly optimistic recent recruitments and trends in the percentage of the unfished spawning biomass seen in previous assessments of silver warehou (Day *et al.* 2012, 2015) under the 2018 assessment structure, data and tuning methods.

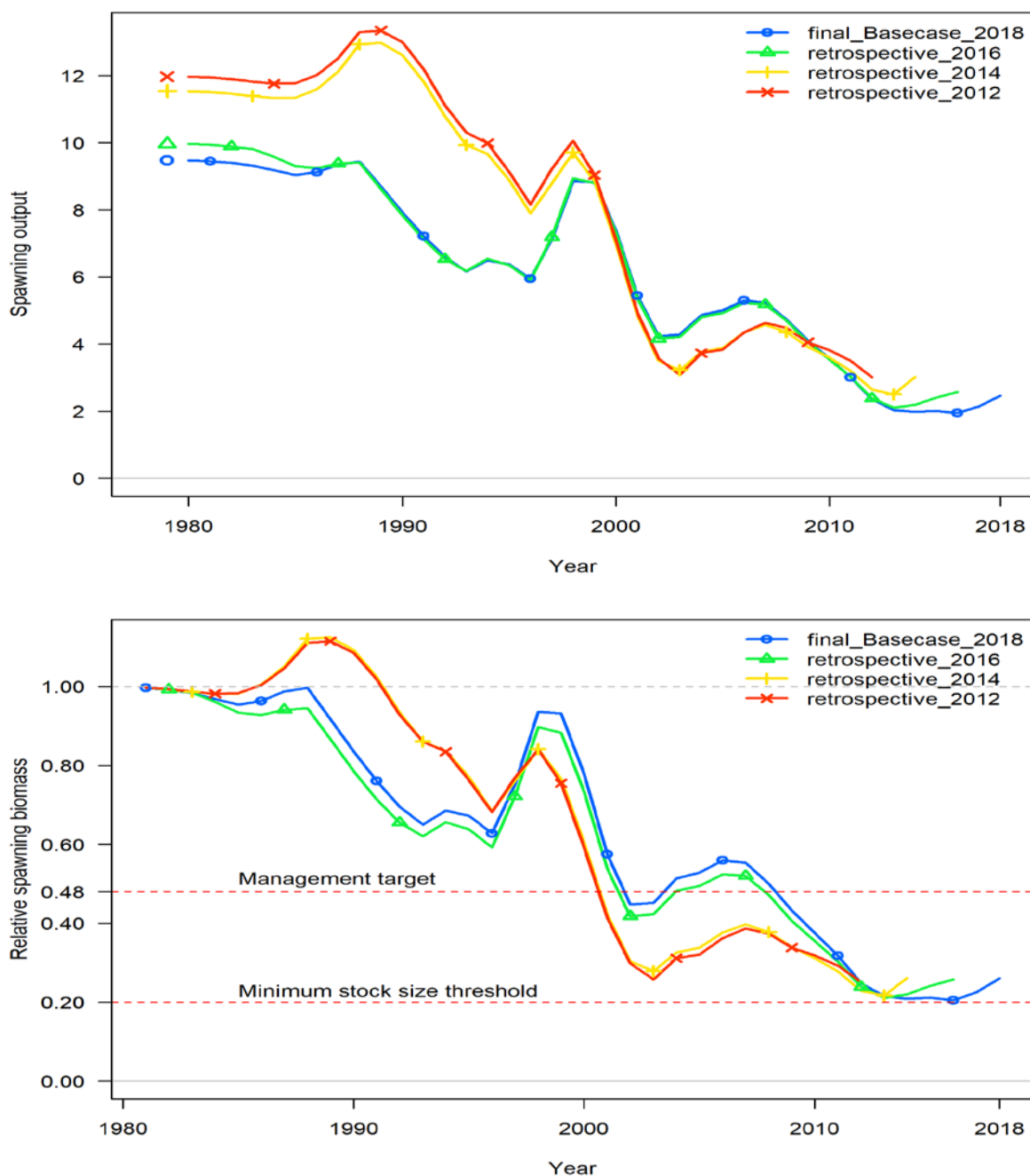


Figure 12.21. Retrospective analysis of absolute (top) and relative (bottom) spawning biomass. Two years of data were removed from the base case and the model returned to produce the assessments for 2016, 2014 and 2012 using the same model structure at the 2018 base case.

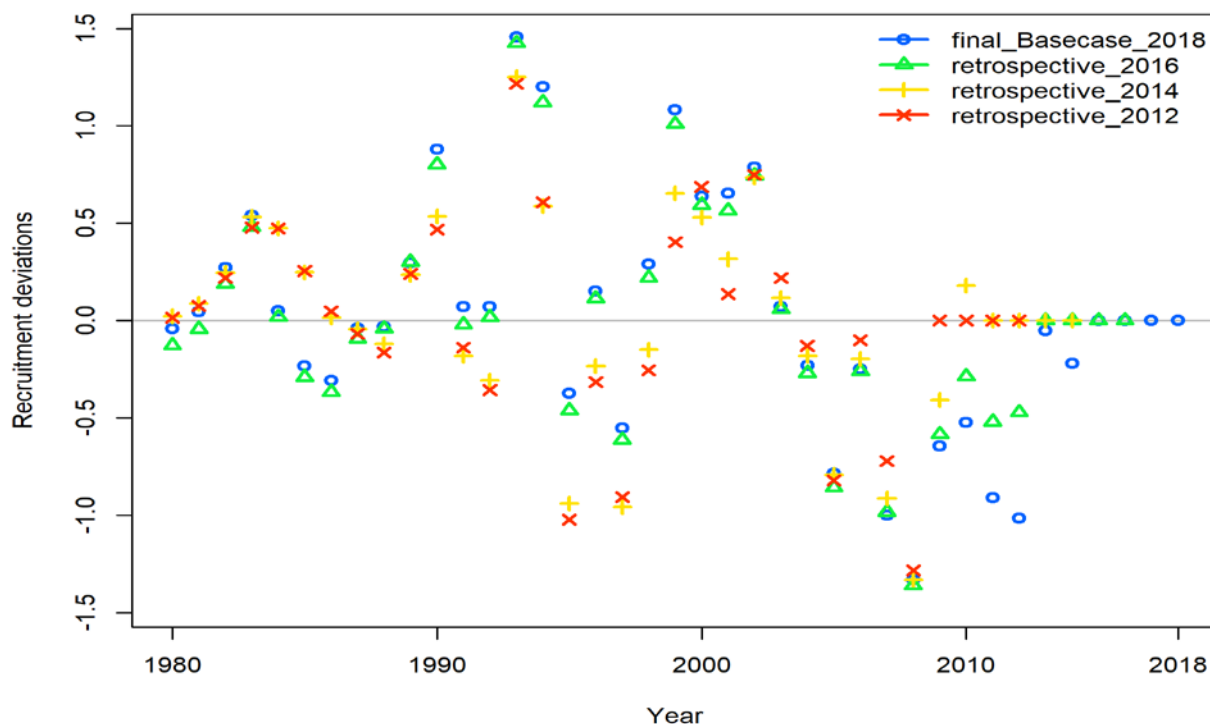


Figure 12.22. Retrospective analysis of estimated recruitment deviations. Two years of data were removed from the base case and the model returned to produce the assessments for 2016, 2014 and 2012 using the same model structure at the 2018 base case.

12.3.4.4 MCMC analysis for the base case

All seven MCMC chains passed standard diagnostic tests (Appendix B2). There was very little variability among the seven MCMC chains for time series of estimated percentage of spawning biomass (Figure 12.23). Estimated recruitment, recruitment deviations and growth parameters similarly showed little variability among the seven MCMC chains (Appendix B1). The Bayesian implementation is almost identical to the Maximum Likelihood implementation (Appendix B3). The only discernible difference was the Maximum Likelihood estimates of spawning biomass prior to 2000 are lower than the MCMC estimates (Figure 12.24). This results in the estimated percentage of virgin spawning biomass from the MCMC in 2019 of 30.4% being slightly lower than the MLE estimate of 31.3% (Table 12.15).

Credible intervals from the MCMC analysis show that the probability that the spawning biomass was below the limit reference point was greater than 20% between 2013 and 2016 (Table 12.15). In 2017 the probability the spawning biomass was below the limit reference point was 8% and in 2018 and 2019 the probability the spawning biomass was below the limit reference point is <1%.

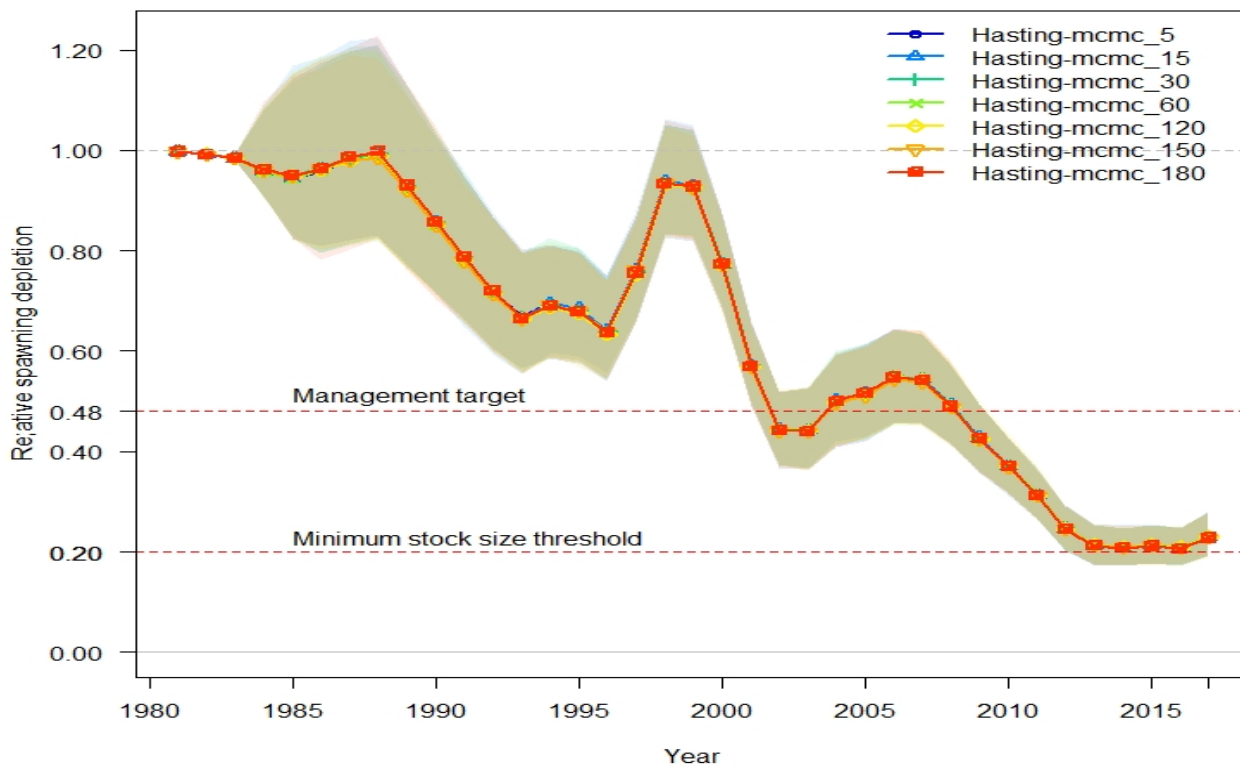


Figure 12.23. Time series of estimated spawning biomass relative to virgin levels from seven MCMC chains of the base case model. Shaded area represents 95% credible interval. Note estimates and credible intervals from all chains overlap.

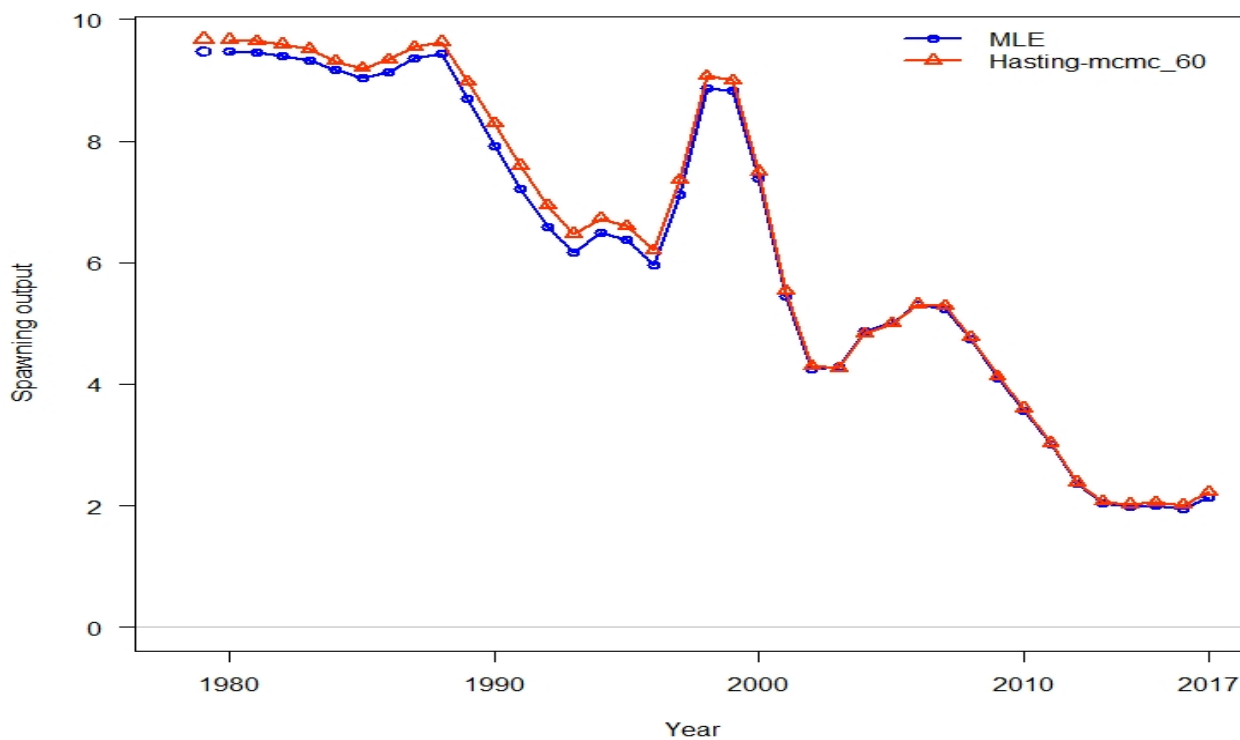


Figure 12.24. Time series of absolute spawning biomass estimates from the maximum likelihood estimate (MLE) and one MCMC chain of the base case model.

Table 12.15. Estimated percentage of virgin spawning biomass and percentage chance of being below the limit reference point from MCMC.

| Year | Percent | |
|------|--|--------------------------|
| | SSB _{current} /SSB ₀ | Percent chance below 20% |
| 2012 | 24.6 | 2.3 |
| 2013 | 21.3 | 26.0 |
| 2014 | 20.9 | 32.4 |
| 2015 | 21.3 | 24.8 |
| 2016 | 20.8 | 32.2 |
| 2017 | 23.1 | 8.0 |
| 2018 | 26.5 | 0.3 |
| 2019 | 30.4 | <0.1 |

12.3.5 Future work and unresolved issues

There is still the need to investigate the impact of separating the discarded catch estimates into eastern and western trawl fleets which caused difficulties in convergence. Difficulties in tuning were observed for the preliminary base case model and the likelihood profile for natural mortality estimated $M=0.5$ (Burch *et al.* 2018a), which suggests some inconsistencies between the data and assumed fixed parameters. Additionally, the question of discarding from factory trawlers could be investigated further to determine whether discarded silver warehou catches from factory trawlers were recorded in the CDR data.

The differences in the length composition data between onboard and port samples, particularly in the east, may indicate that the east / west split in the assessment may not be capturing all of the depth related variability in population structure. If depth related variability in length does occur in the east then a depth structure model may need to be considered and the utility of port sampling in the east should be examined because depth of fishing is not available for port samples that are aggregated over many shots.

Potential problems with the CPUE series that provide abundance indices have been identified by Sporcic and Haddon (2018a). While the base case model fits to the western trawl CPUE are good, particularly in recent years, the fits to the eastern trawl CPUE are poor and show signs of autocorrelation. An examination of the CPUE for evidence of changes in targeting practices is also recommended. An additional sensitivity of including FIS length-composition data and attempting to estimate selectivity for the FIS fleets would be useful.

This study confirmed that the retrospective pattern of optimistic recent recruitments being revised downwards in subsequent assessments (Day *et al.* 2012, Day *et al.* 2015) is still present with the 2018 assessment structure and is a serious concern. As this retrospective pattern may overestimate the current spawning biomass and hence stock status, understanding why the retrospective pattern occurs and correcting it if possible is urgently needed.

12.4 Conclusion

This document presents an updated assessment of silver warehou (*Seriolella punctata*) in the SESSF using data up to 31 December 2017. A full stock assessment for silver warehou was last performed in

2015 by Day *et al.* (2015) using the stock assessment package Stock Synthesis version SS-V3.24U (Methot 2015). Changes from the 2015 assessment include: (a) migration to the latest version of Stock Synthesis (SS-V3.30.12, Methot *et al.* 2018), (b) updates of all catch, discard, length, age and catch rate data and the last year of estimation of recruitment (2014), three years prior to the last year of data (2017), (c) including silver warehou catches from the GHAT and the SPF (d) separating the discard series separated into east and west components and incorporating factory trawler discard information where available, (e) removal of FIS abundance estimates from the preliminary base case and (i) adopting the new tuning methods.

Results show reasonable fits to the CPUE abundance index for the western trawl fleet, however, fits to the eastern trawl fleet are poor prior to 2008. There are differences in the length distributions from onboard and port sampling in both the east and the west, resulting in poor fits to the aggregated length distributions for the eastern trawl fleet. In the west, the fits to the aggregated length distributions were reasonable. The overall fits to the retained length and conditional age-at-length data are reasonable, although there are poor fits in some years. Fits to the discard length data are poor, particularly in the west. Inclusion of the fishery independent survey (FIS) data as a model sensitivity has little impact on either the trends in spawning stock biomass or the estimated percentage of the unfished spawning biomass.

The assessment fits to the most recent three CPUE data points in the east and two data points in the west suggest that the model may again be overly optimistic at the end of the time series (Day *et al.* 2015). This pattern is further highlighted by the retrospective analysis which shows increases in the estimated percentage of the unfished spawning biomass in the last few years of the 2014 and 2016 assessment scenarios are not realised in the following assessment scenario two years later. Given that the current assessment estimates that percentage of the unfished spawning biomass was only just above the 20% limit reference point between 2013 and 2016, with the lower 95% asymptotic confidence intervals being below the limit reference point (LRP) between 2013 and 2017 there is a possibility the stock is currently below the LRP.

Since the 2015 assessment the stock has seen a continued decline in CPUE, although catches have been stable since 2014 at between 350 and 400t. The last 11 years of estimated recruitment are all below average suggesting that relying on average recruitment for the stock to recover may be overly optimistic. Poor future recruitment scenarios illustrate the potential dangers to the stock if the calculated RBC is actually caught, although these impacts are reduced if the current low catch levels are maintained. This sustained period of below average estimated recruitments may be indicative of a regime shift in productivity for this stock.

Given the changes to the assessment structure, data and tuning methods since the 2012 assessment, a retrospective analysis was undertaken to determine whether the pattern of optimistic recent recruitments that have been revised downwards in subsequent assessments was still present with the 2018 assessment structure. Increases in recruitment and spawning biomass in the most recent years from the 2014 and 2016 assessment scenarios were revised downwards in subsequent assessments. This suggests that the increase in spawning biomass seen in the most recent years of 2018 assessment may be overly optimistic and that the percentage of the unfished spawning biomass may currently be around the limit reference point.

This assessment estimates that the projected 2019 spawning stock biomass will be 31% of virgin stock biomass. The RBC from the base case model for 2019 is 942t for the 20:35:48 harvest control rule, with a long-term yield of 1,773t. In comparison, the 2015 assessment estimated the 2016 percentage of the unfished spawning biomass to be 40%, with corresponding RBCs of 1,958t, with a long-term

yield of 2,281t. However, these scenarios assume recruitment will return to average levels. Low and very low recruitment scenarios suggest that if current landed catches are maintained at around 350t then the stock is likely to remain above the LRP.

A Bayesian assessment was undertaken because the base case assessment showed that the maximum likelihood estimate of spawning biomass was near the limit reference point of 20% of its unfished level between 2013 and 2016, with the lower 95th percent asymptotic confidence intervals being below the limit reference point in those years. The probability that the spawning biomass was below the limit reference point was greater than 20% between 2013 and 2016. In 2017 the probability the spawning biomass was below the limit reference point was 8% and in 2018 and 2019 the probability the spawning biomass was below the limit reference point is <1%.

At its November 2018 meeting, SERAG agreed to recommend a TAC for silver warehou based on the assumption that recruitment will remain below average in the next few years. SERAG chose to assume that recruitment would remain at the mean of the last five years of estimated recruitments in the base case model (2010 – 2014). Projections assuming this low recruitment were run for scenarios of constant landed catch that were between the catch in the most recent year for which data is available (348 t) and the RBCs from the base case model which assumes average recruitment (942 t in 2019). Scenarios with constant annual catches of 750 t or more led to the estimated spawning biomass declining under the low recruitment scenario. Under the low recruitment scenario with constant annual catches between 348 t and 600 t spawning biomass is predicted to increase, albeit more slowly than the base case which assumes average recruitment.

12.5 Acknowledgments

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12.7 Appendix A Base case fits

A.1 Length fits

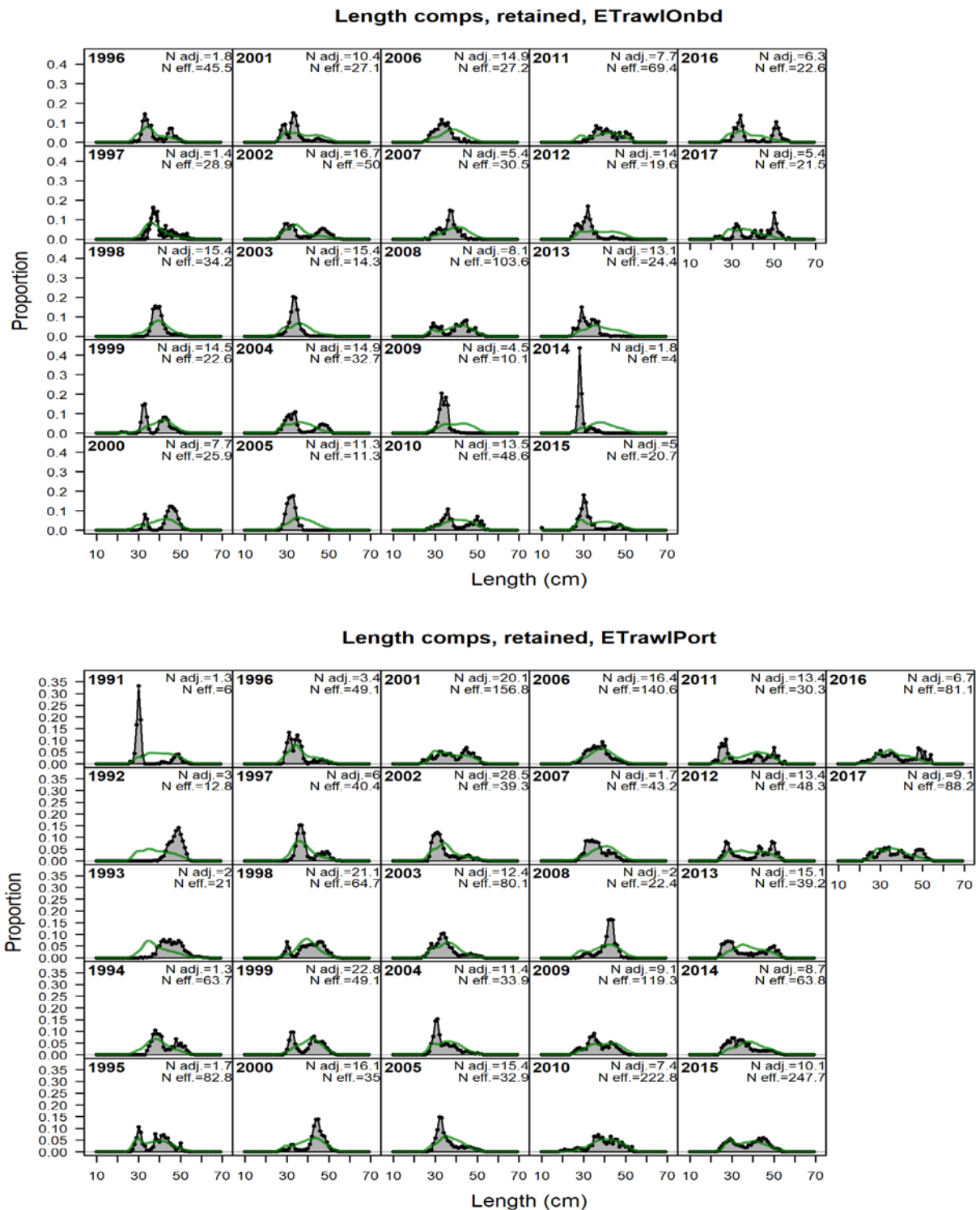


Figure A 12.1. The observed (shaded) and model-predicted (green line) fits to the retained length composition data for silver warehou for the eastern trawl fleet onboard (top) and port (bottom).

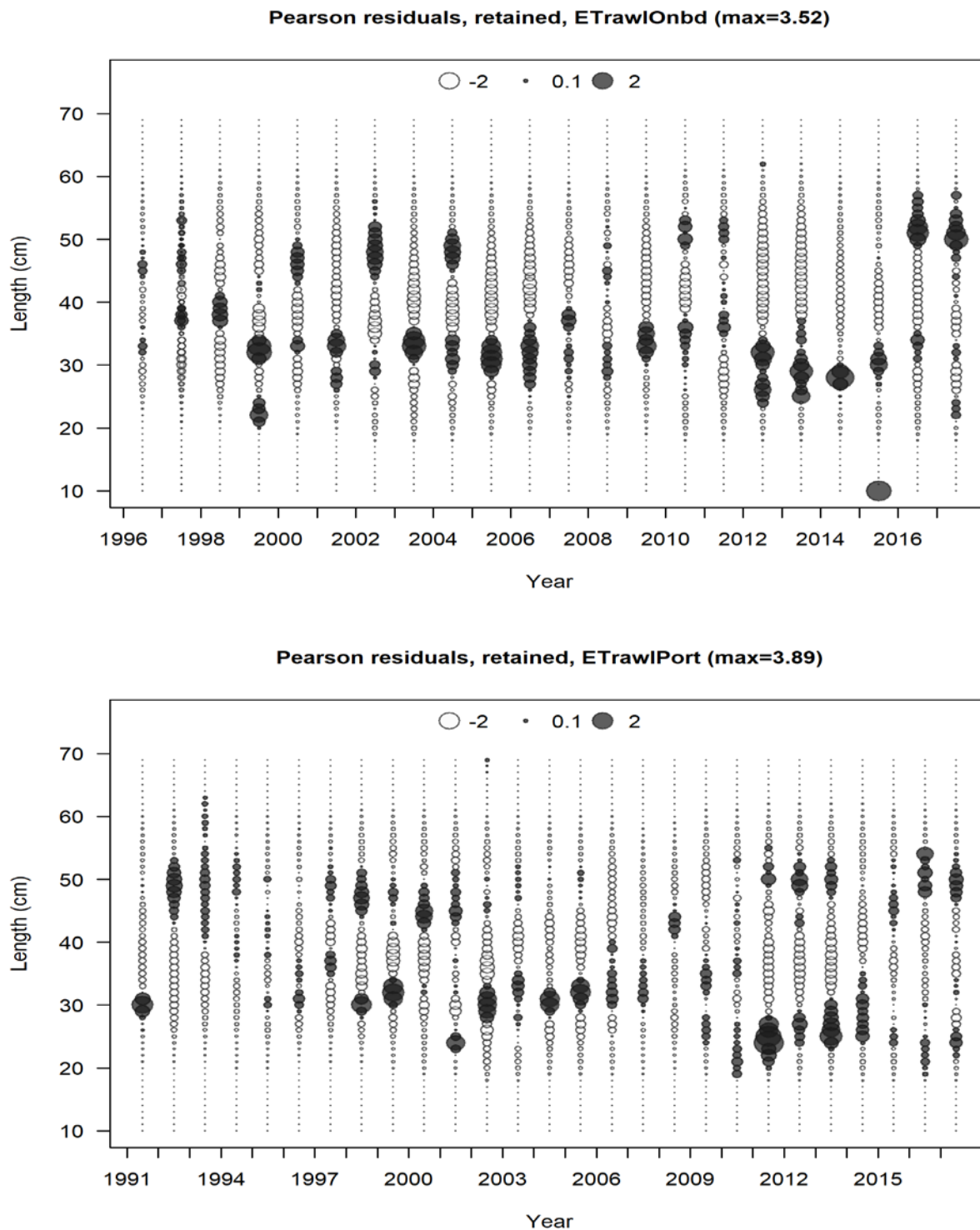


Figure A 12.2. The residual pattern for the retained length composition data for silver warehou for the eastern trawl fleet onboard (top) and port (bottom).

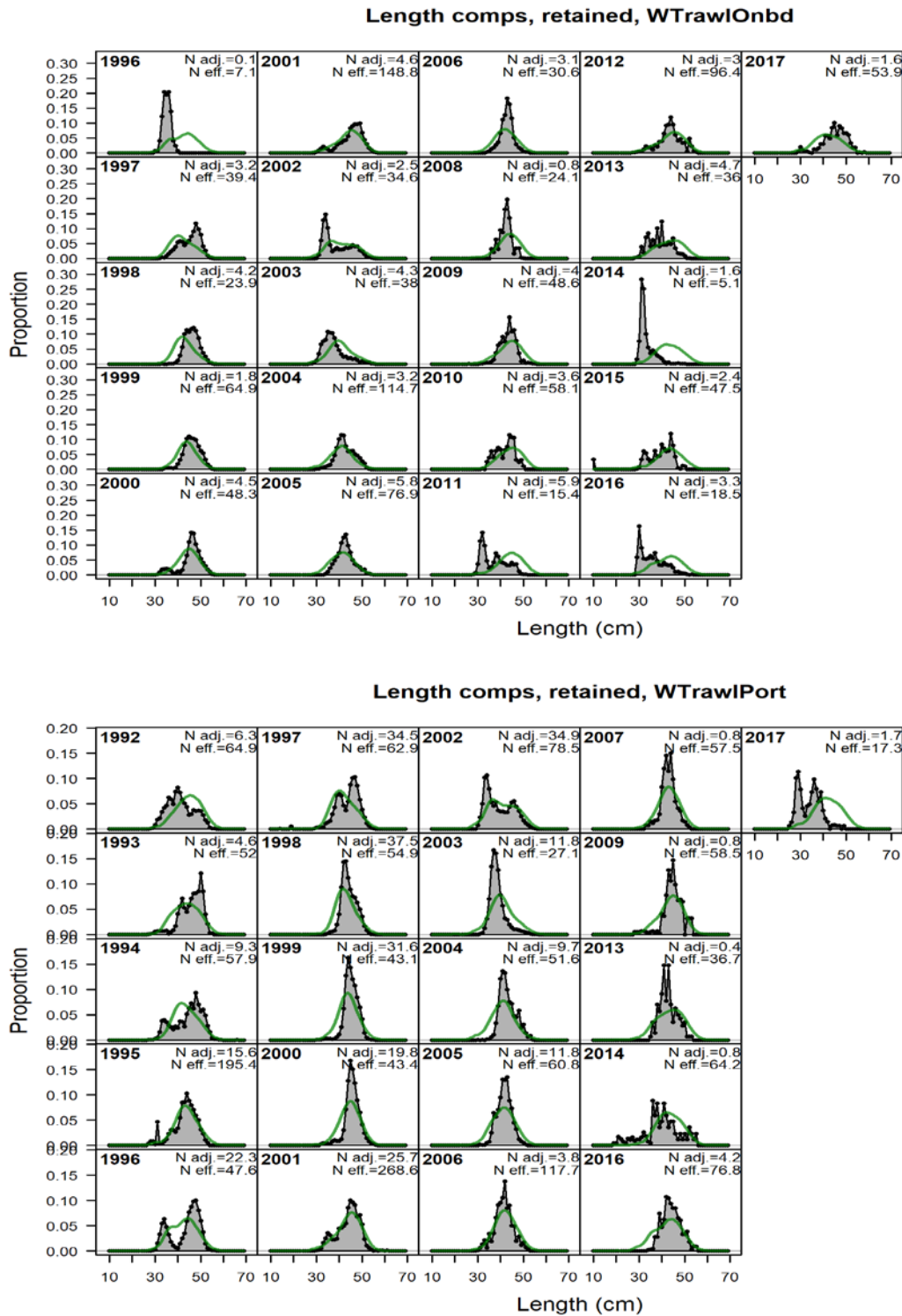


Figure A 12.3. The observed (shaded) and model-predicted (green line) fits to the retained length composition data for silver warehou for the western trawl fleet onboard (top) and port (bottom).

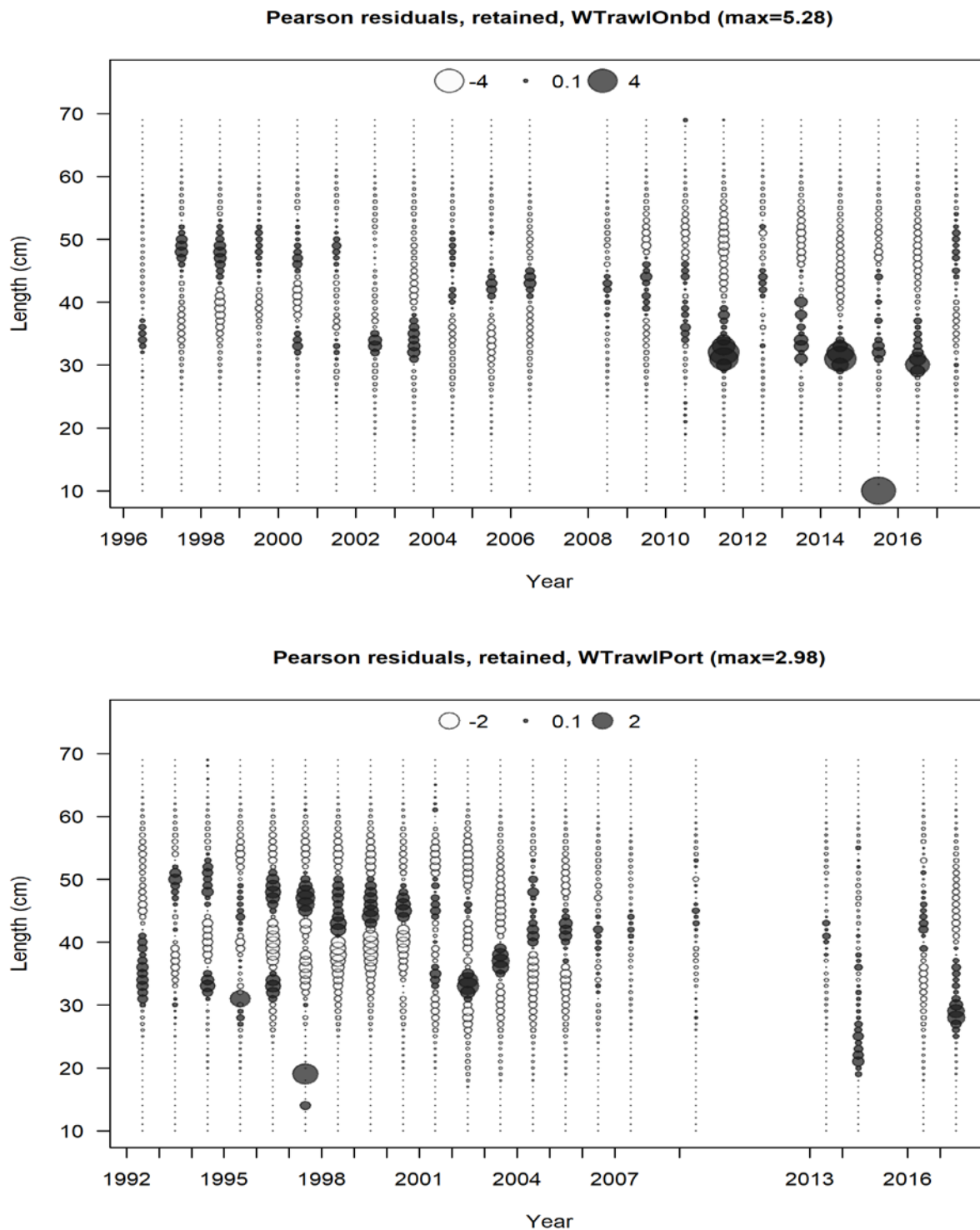


Figure A 12.4. The residual pattern for the retained length composition data for silver warehou for the western trawl fleet onboard (top) and port (bottom).

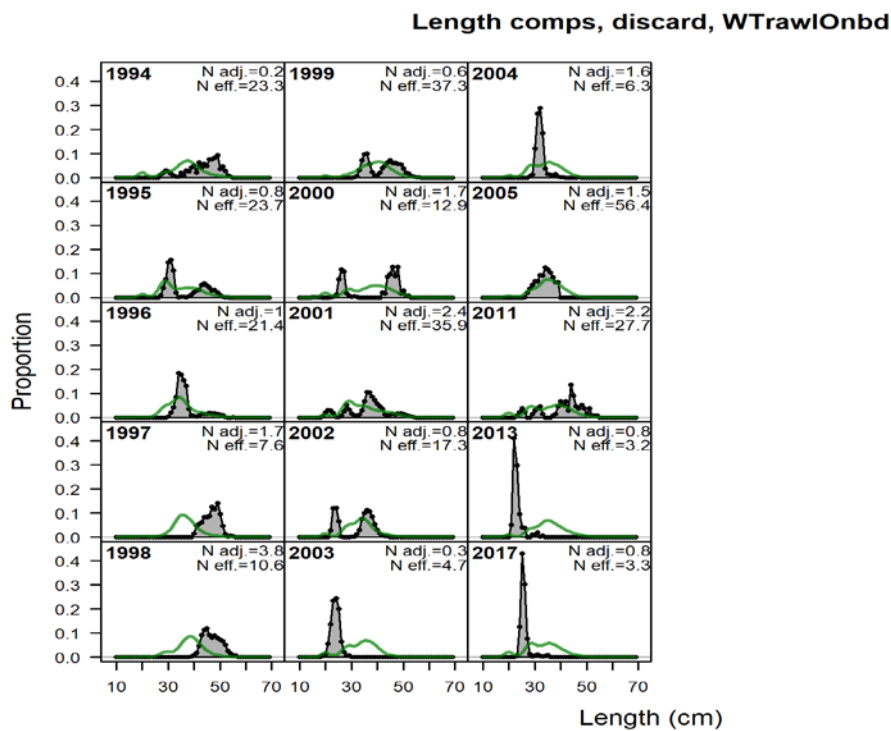
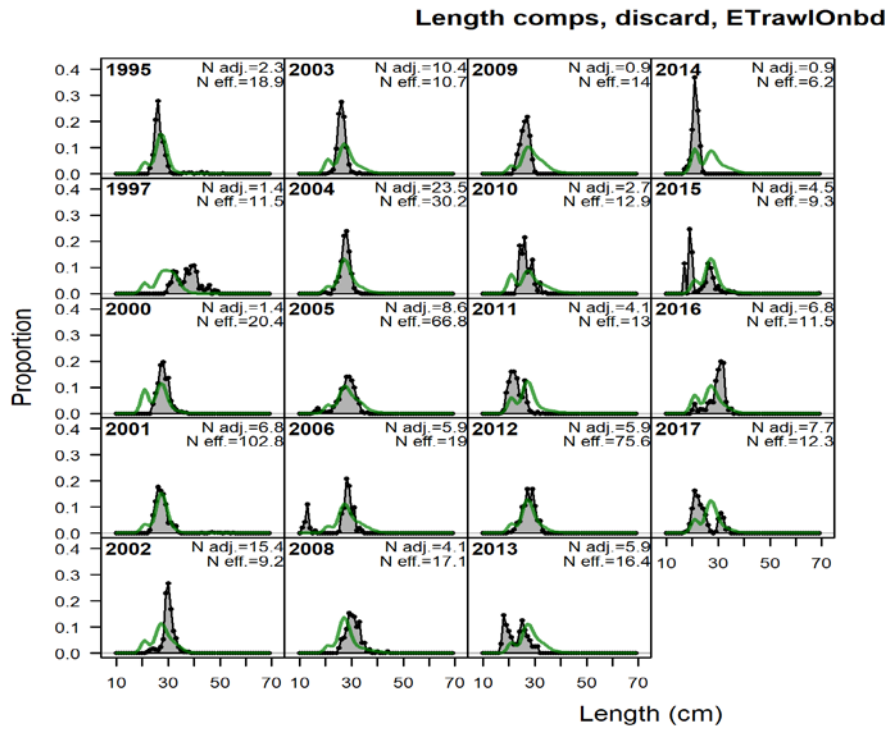


Figure A 12.5. The observed (shaded) and model-predicted (green line) fits to the discarded length composition data for silver warehou for the eastern trawl fleet onboard (top) and western trawl fleet onboard (bottom).

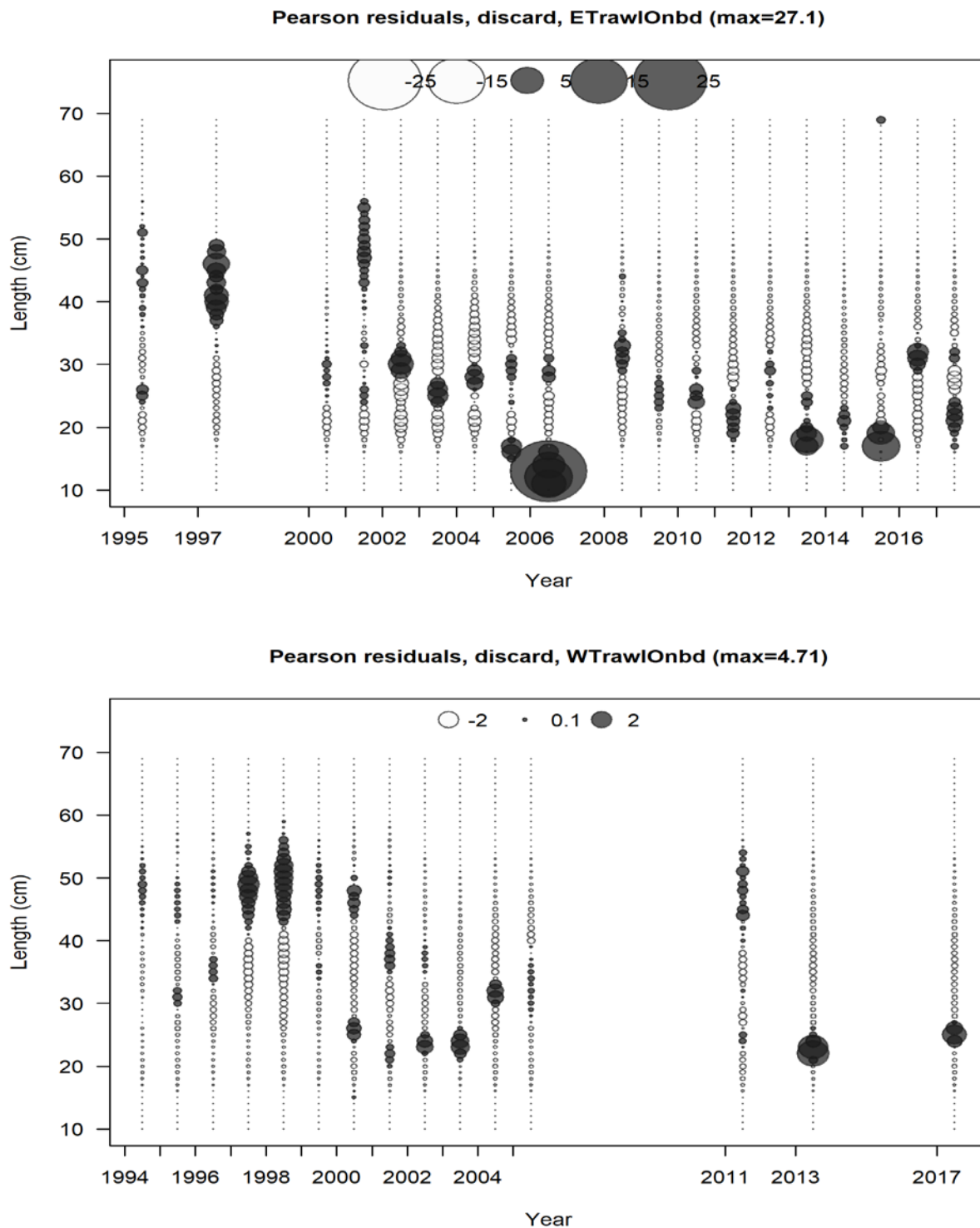


Figure A 12.6. The residual pattern for the discarded length composition data for silver warehou for the eastern trawl fleet onboard (top) and western trawl fleet onboard (bottom).

A.2 Age fits

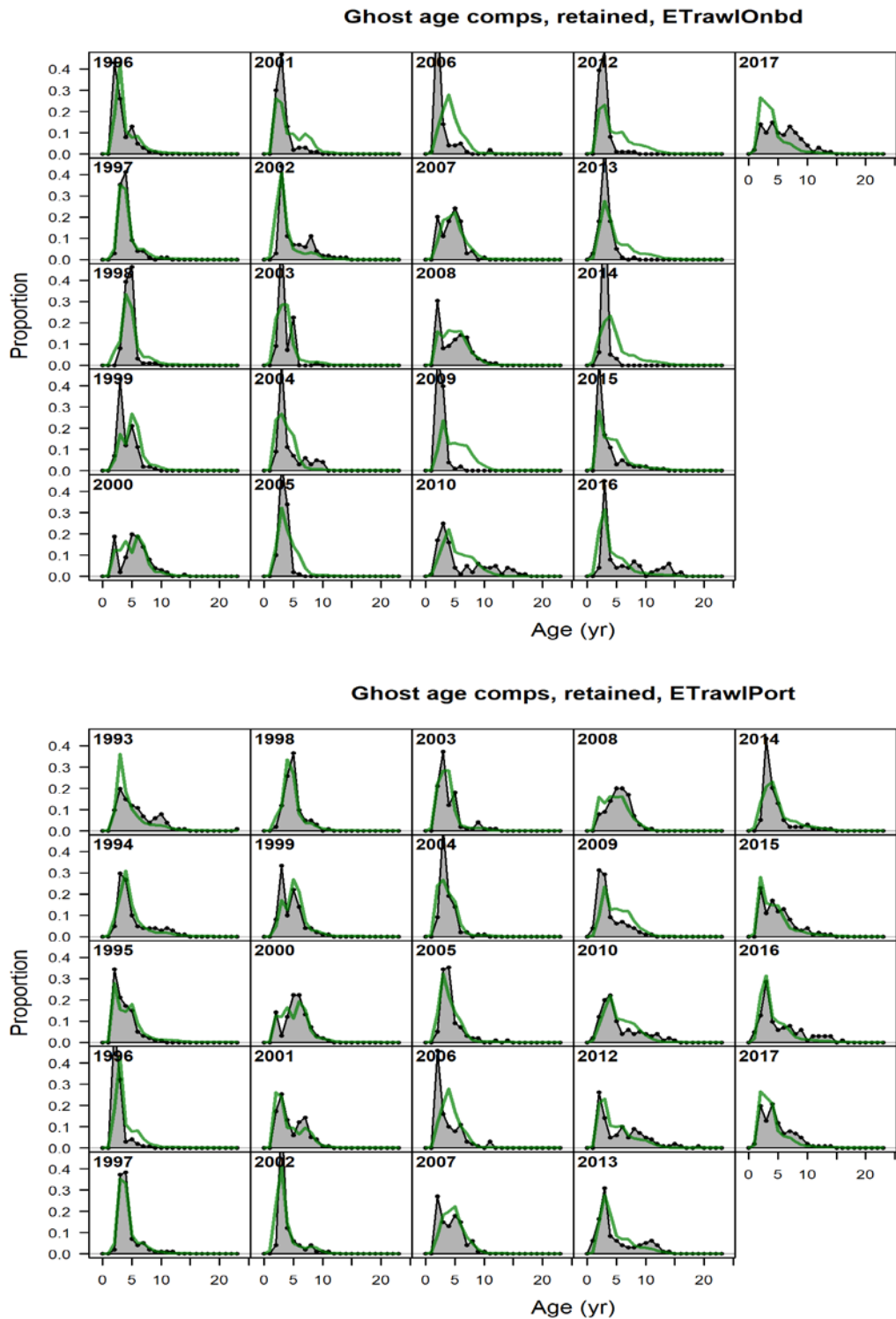


Figure A 12.7. The observed (shaded) and model-predicted (green line) implied fits to the age composition data for silver warehou for the eastern trawl fleet onboard (top) and port (bottom).

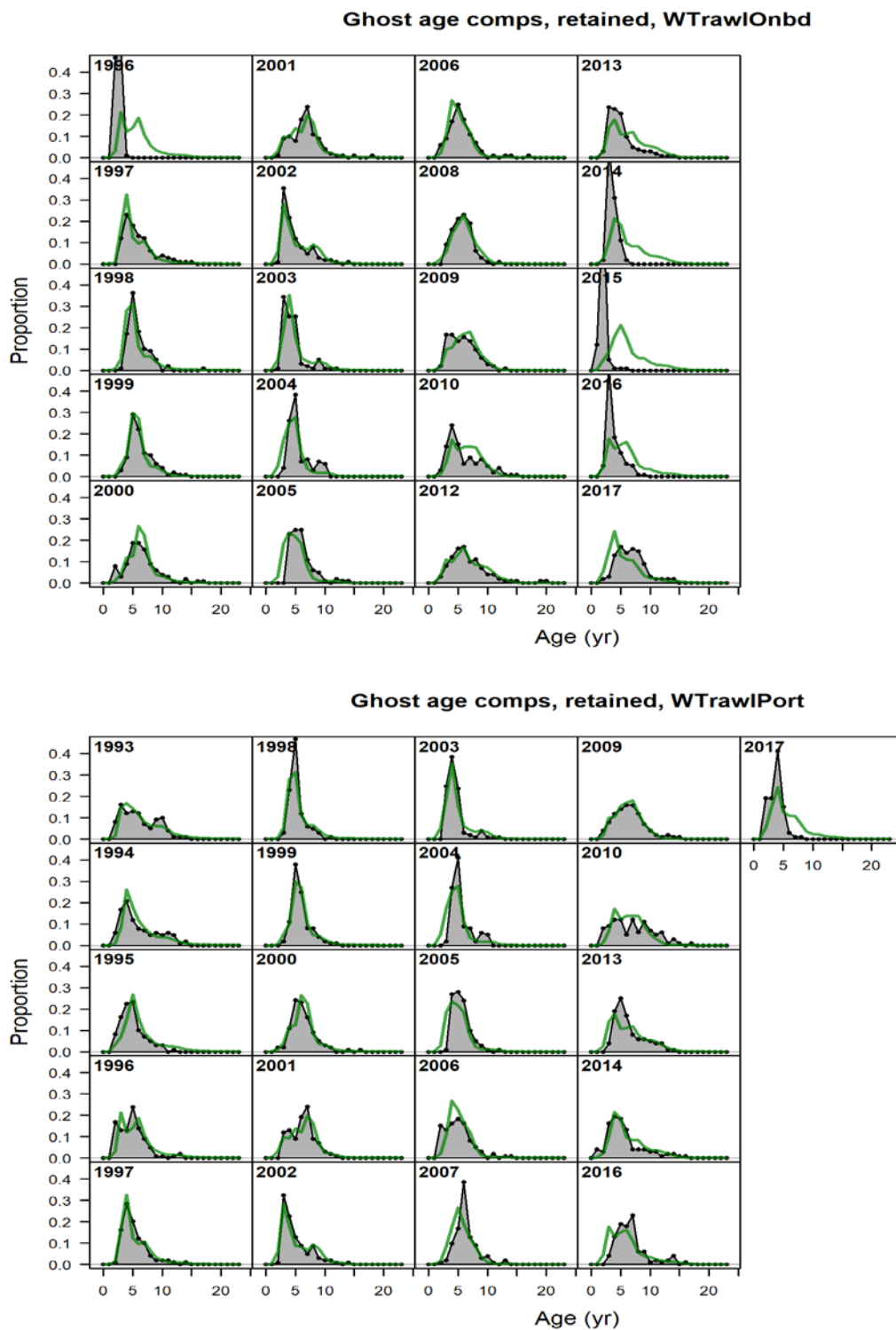


Figure A 12.8. The observed (shaded) and model-predicted (green line) implied fits to the age composition data for silver warehou for the western trawl fleet onboard (top) and port (bottom).

A.3 Age-at-length fits

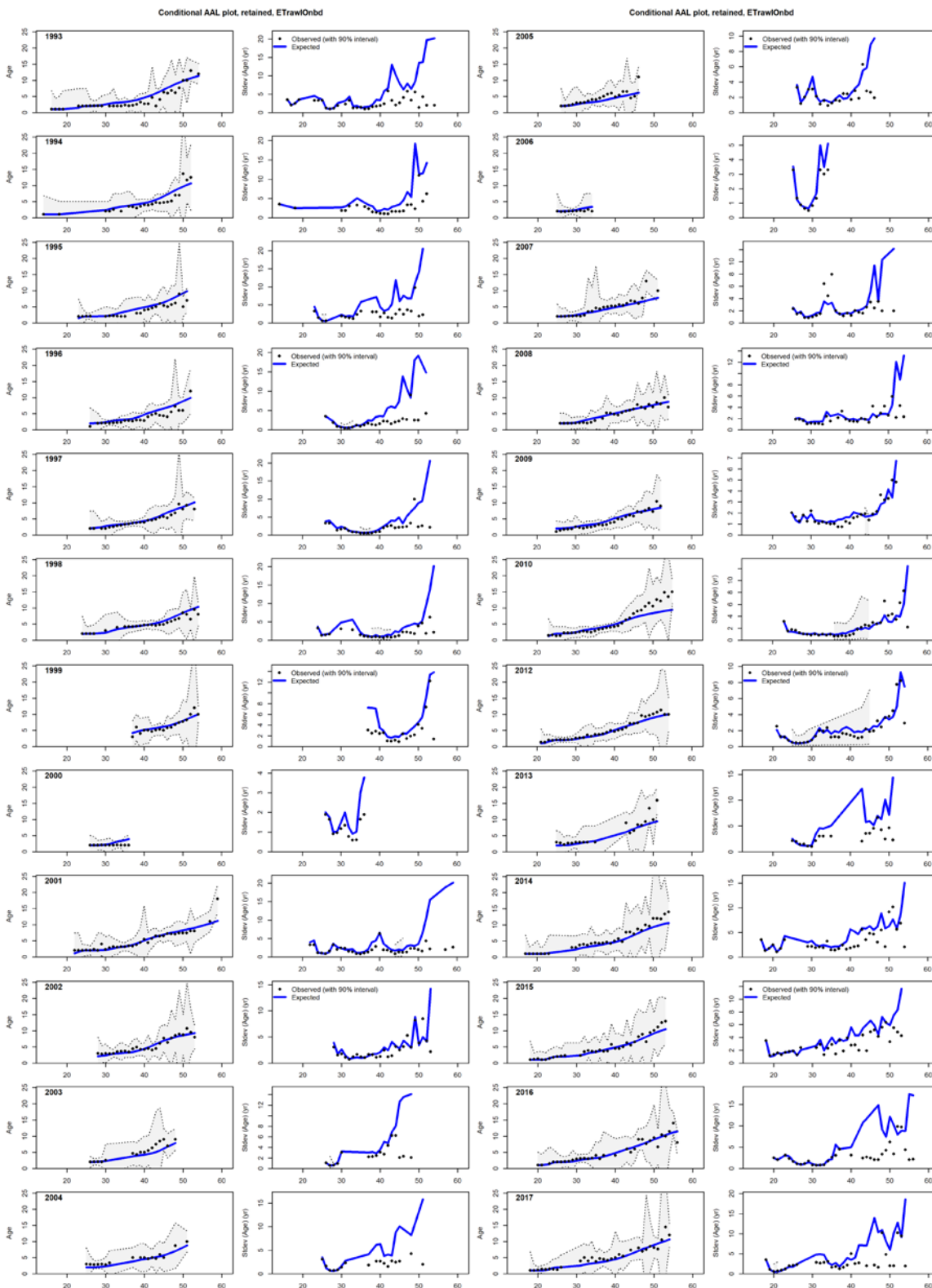


Figure A 12.9. Fits to the conditional age-at-length for the eastern fleet using method TA1.8 of Punt (2017). Observed in black, expected in blue lines. Second and fourth columns are standard deviations.

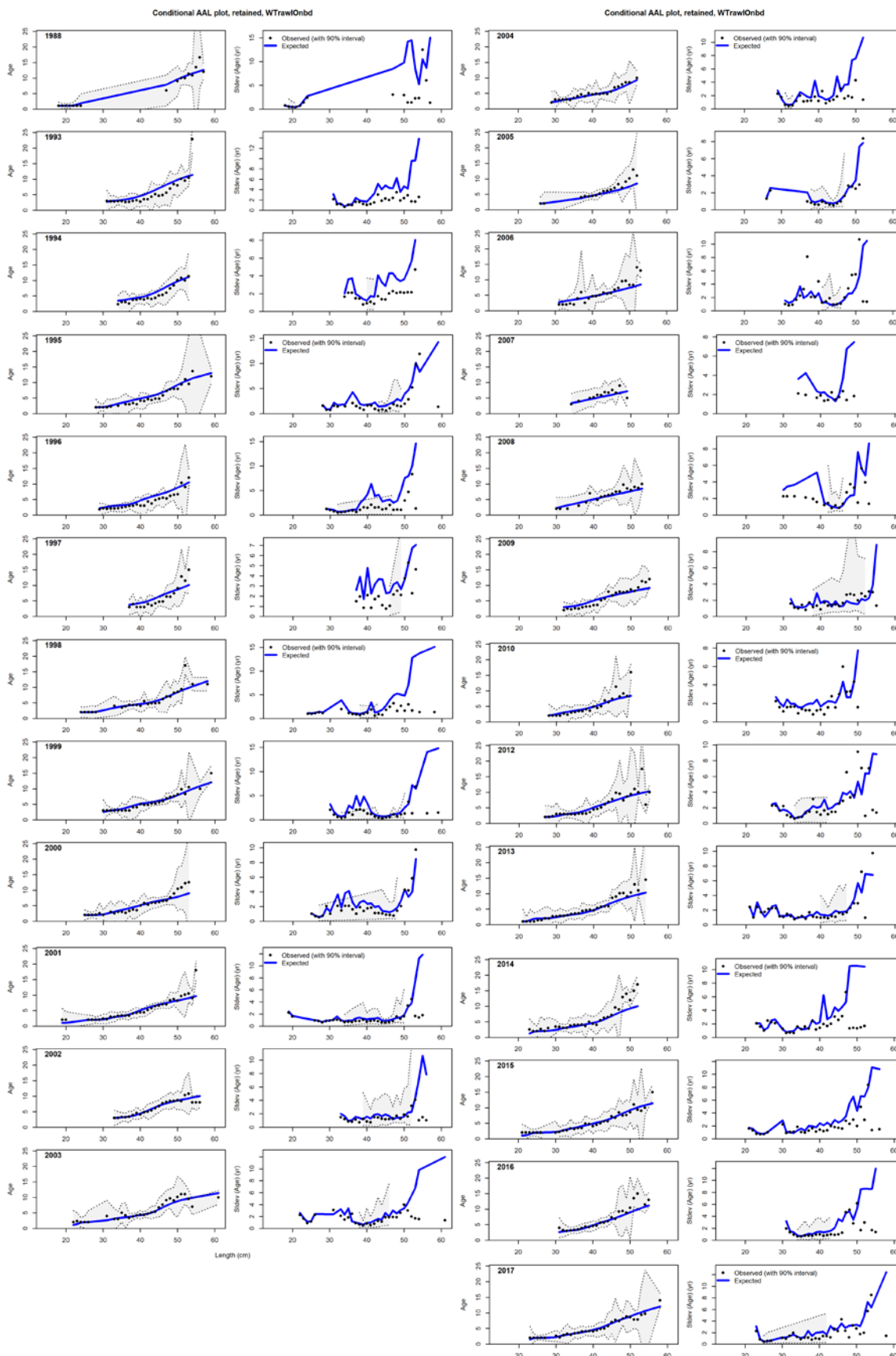


Figure A 12.10. Fits to the conditional age-at-length for the western fleet using method TA1.8 of Punt (2017). Observed in black, expected in blue lines. Second and fourth columns are standard deviations.

A.4 Length fit diagnostics (method TA1.8 of Punt (2017))

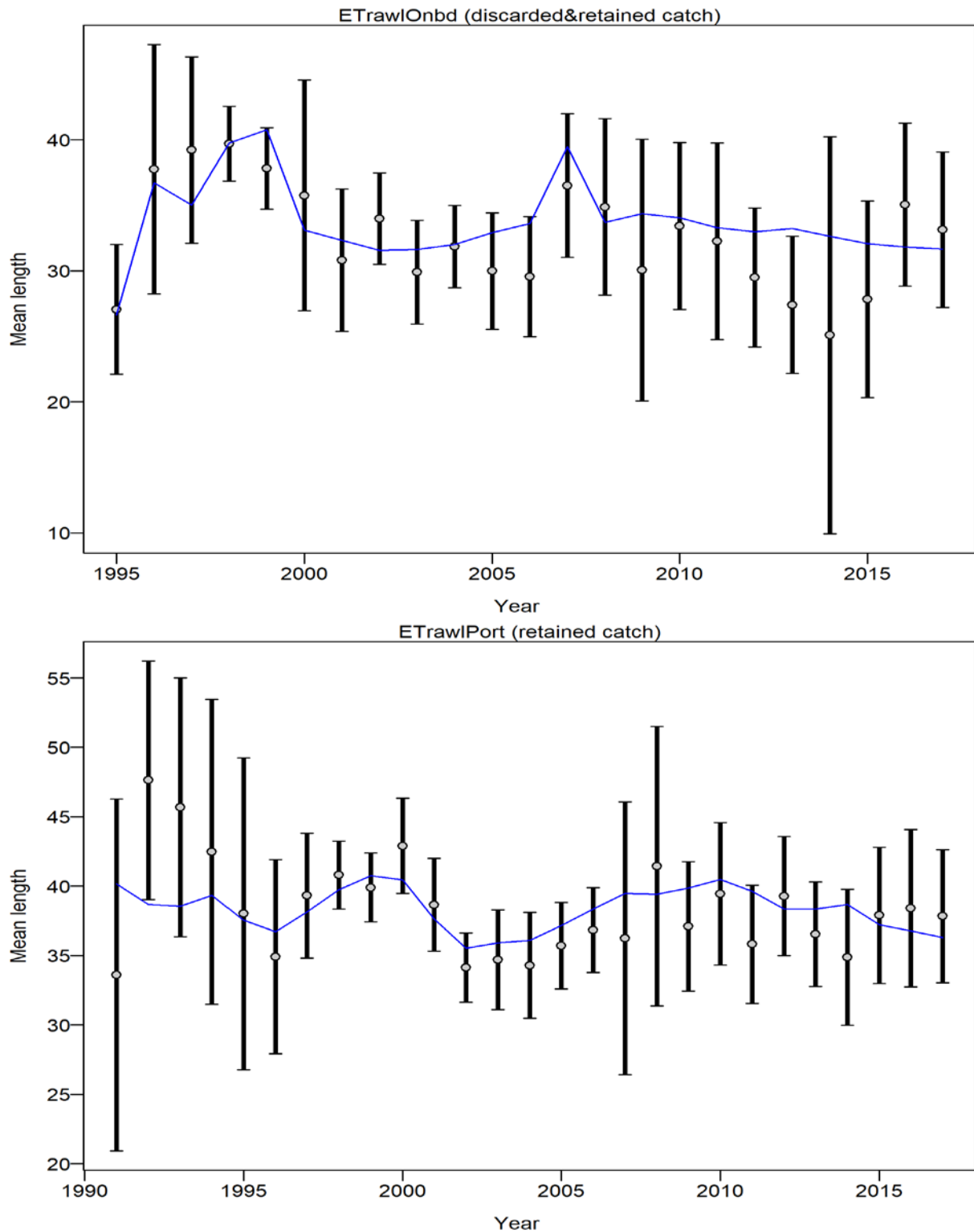


Figure A 12.11. Length fit diagnostics obtained from conditional age-at-length weighting using method TA1.8 of Punt (2017). Eastern trawl fleet onboard (top) and port (bottom).

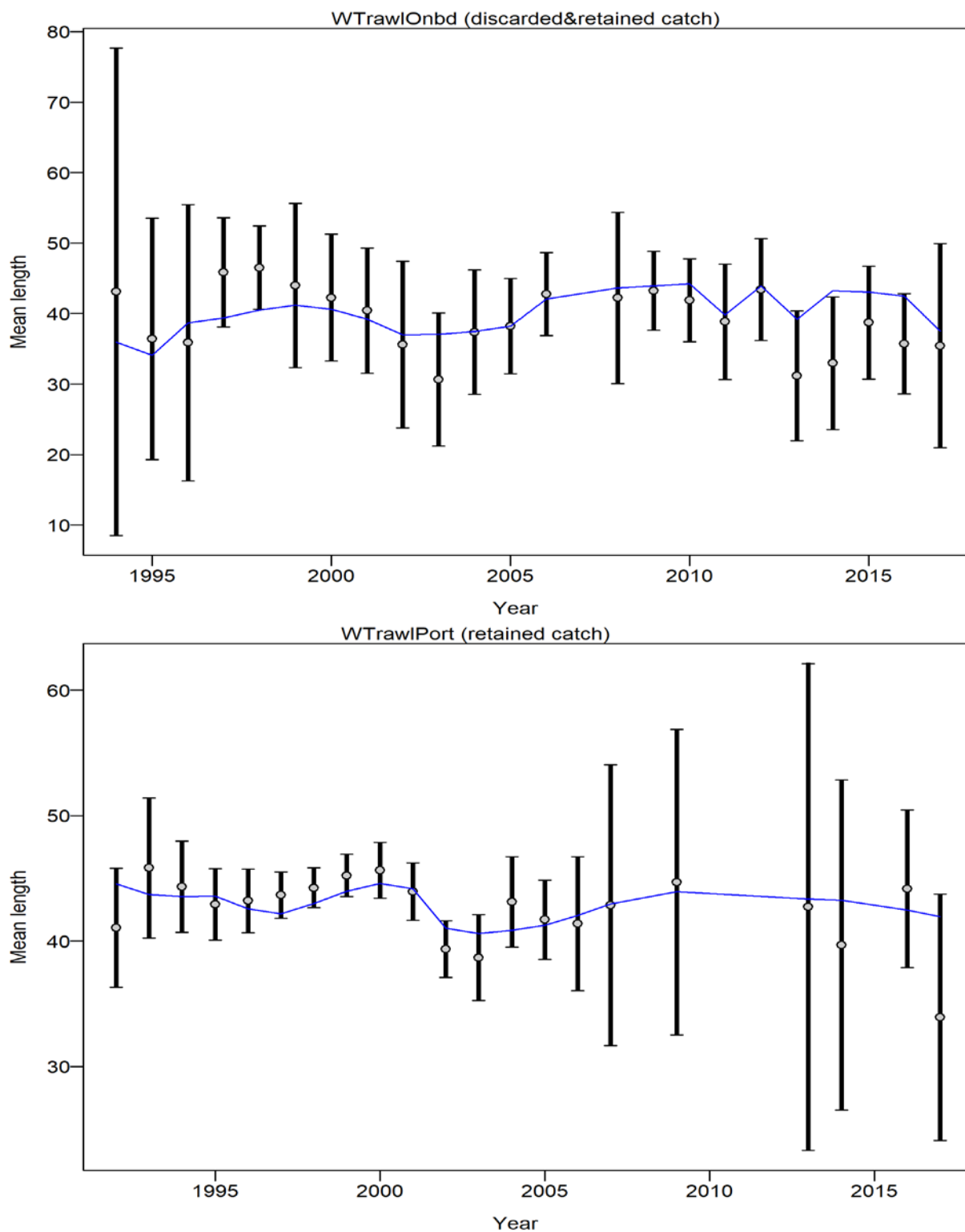


Figure A 12.12. Length fit diagnostics obtained from conditional age-at-length weighting using method TA1.8 of Punt (2017). Western trawl fleet onboard (top) and port (bottom).

A.5 Age fit diagnostics (method TA1.8 of Punt (2017))

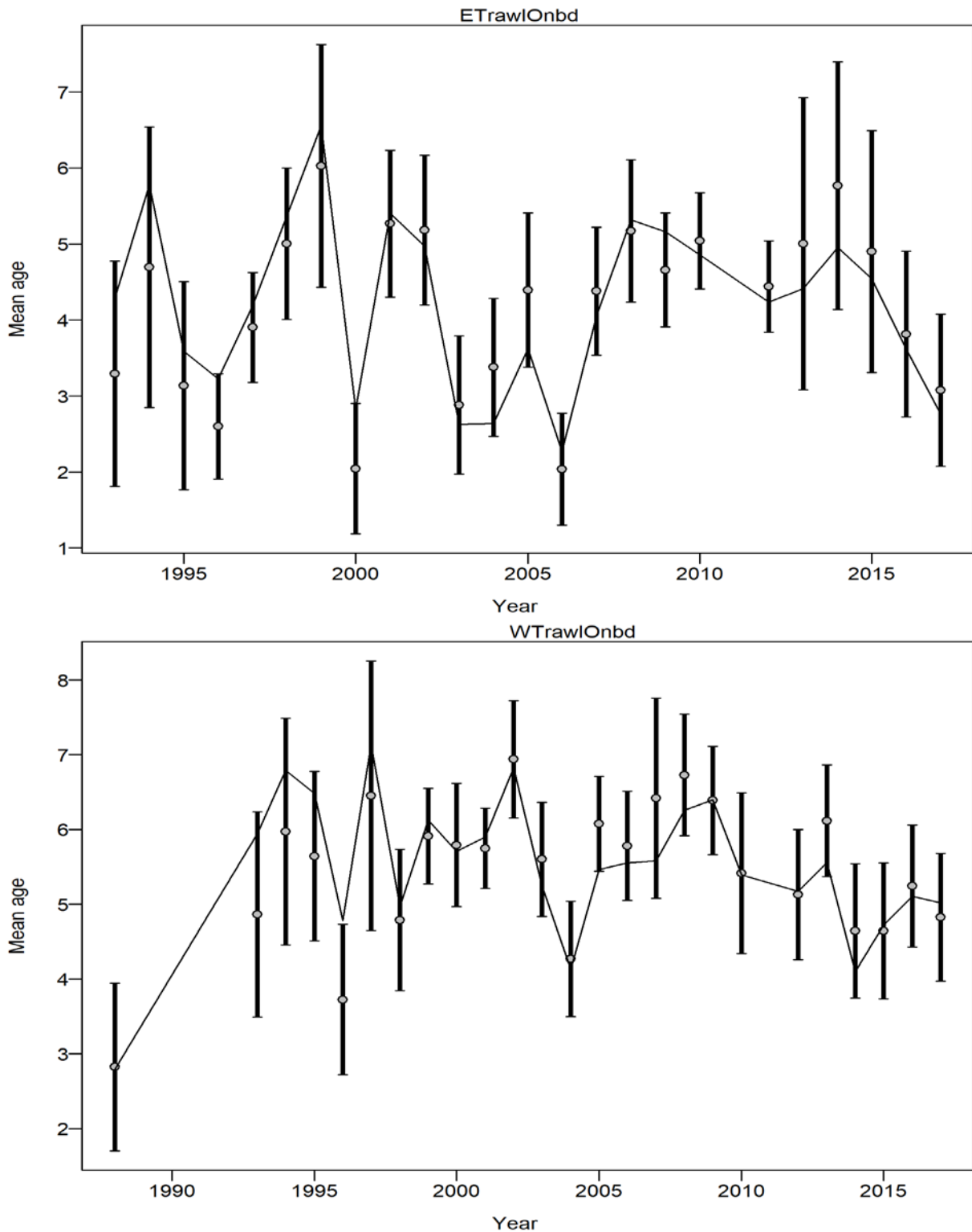


Figure A 12.13. Age fit diagnostics obtained from conditional age-at-length weighting using method TA1.8 of Punt (2017) for eastern trawl onboard (top) and western trawl onboard (bottom).

12.8 Appendix B MCMC base case diagnostics

B.1 Comparison among chains

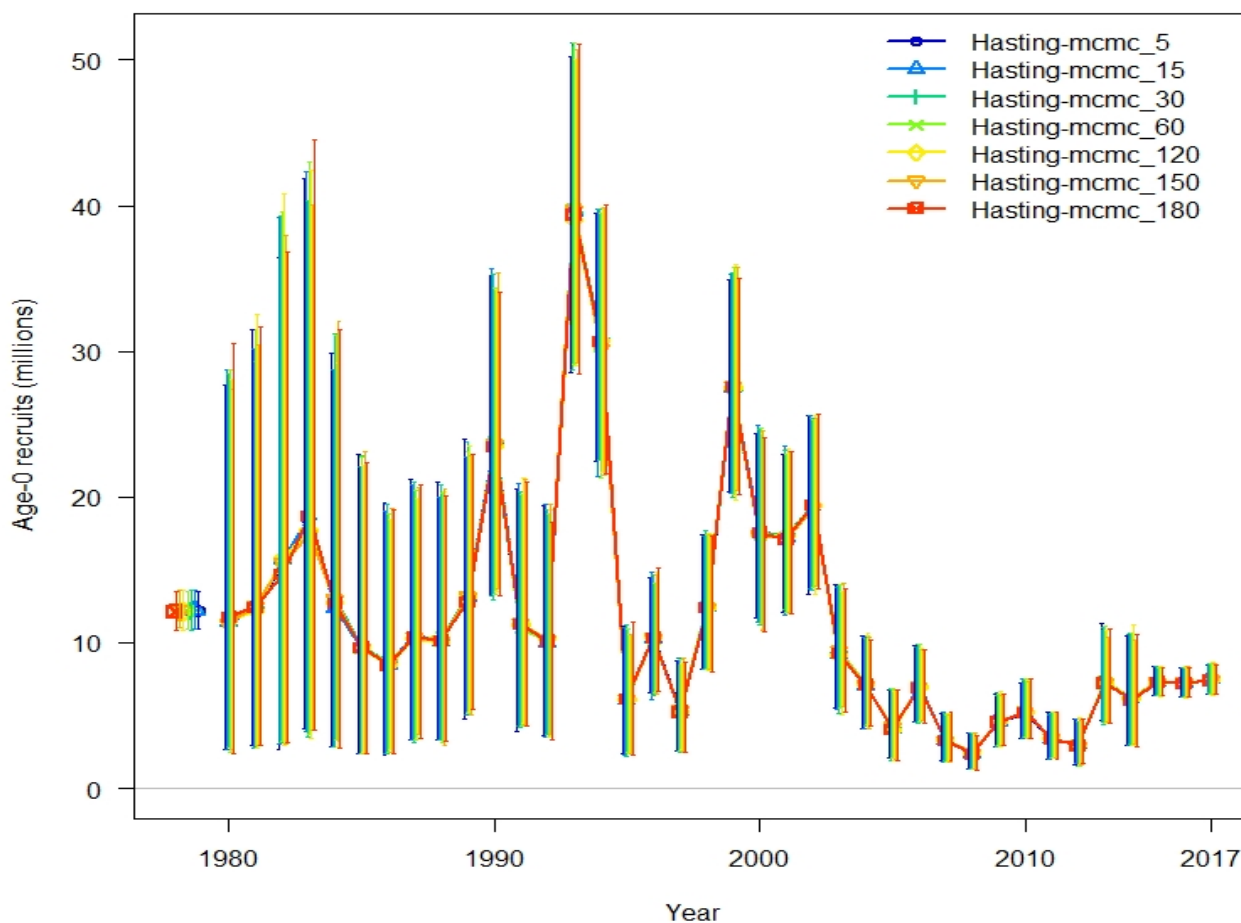


Figure B 12.1. Estimated absolute recruitment from seven MCMC chains run for the base case assessment.

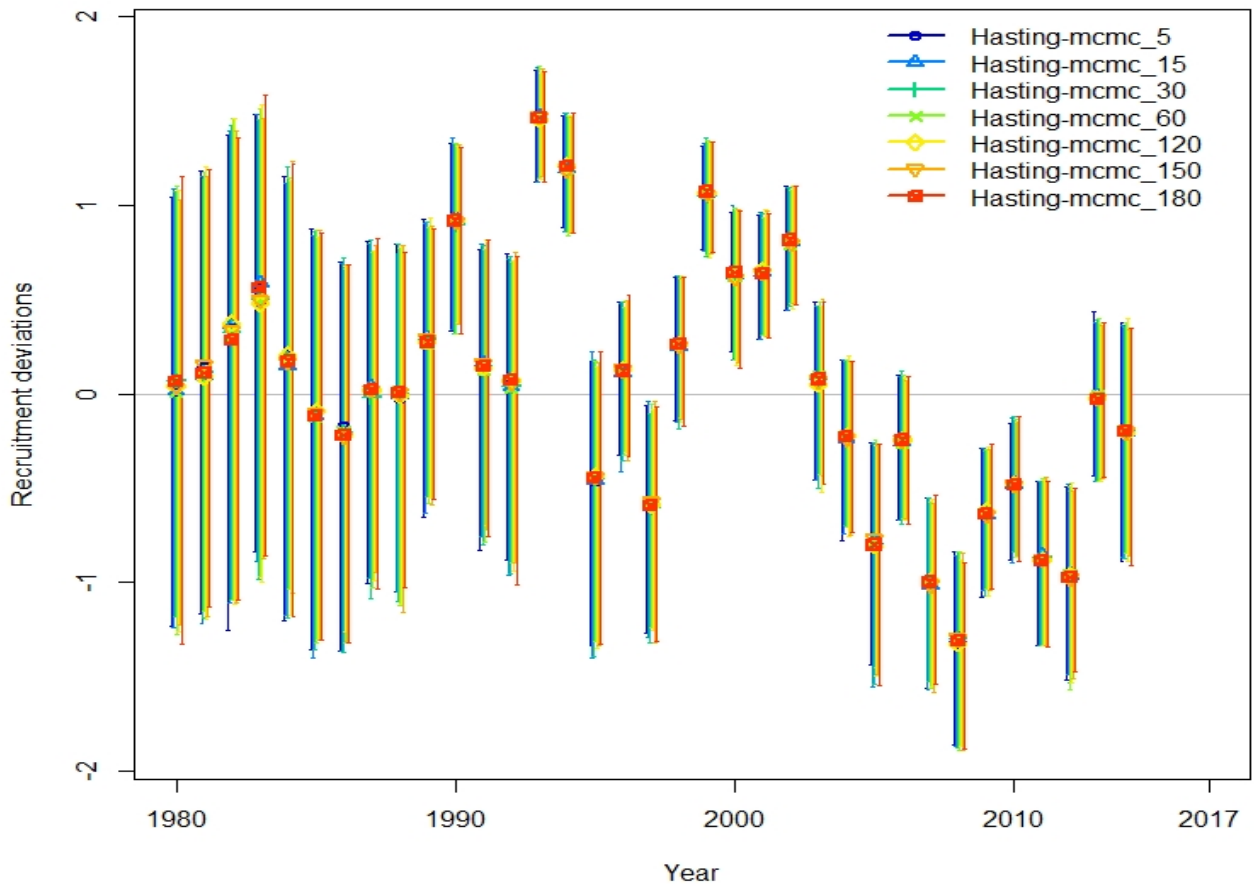


Figure B 12.2. Estimated recruitment deviations from seven MCMC chains run for the base case assessment.

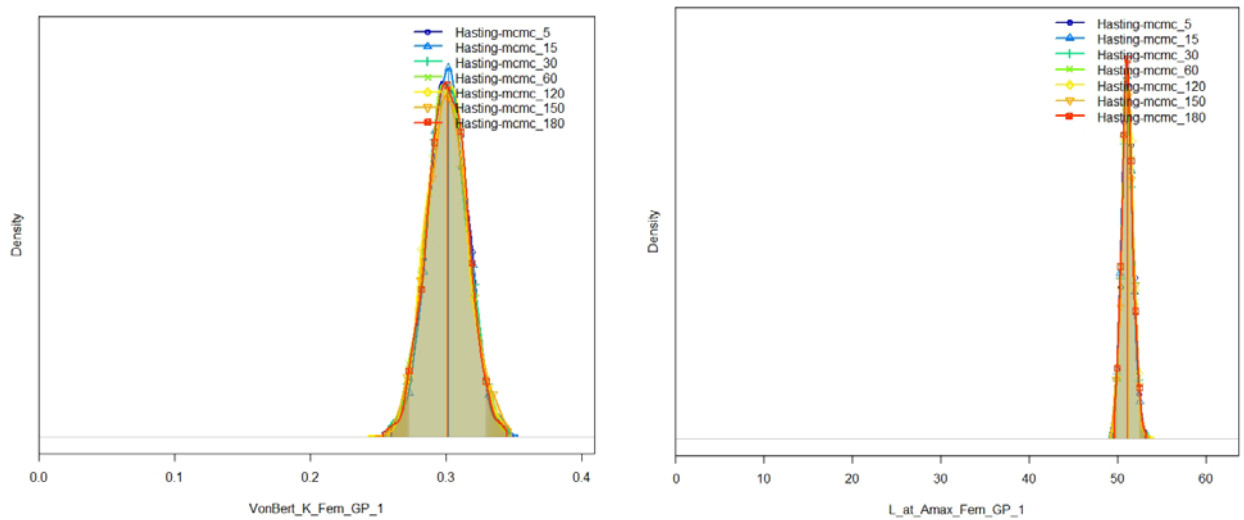


Figure B 12.3. Estimated von Bertalanffy growth parameters K (left) and mean length at maximum age (right) from seven MCMC chains run for the base case assessment.

B.2 MCMC Diagnostics

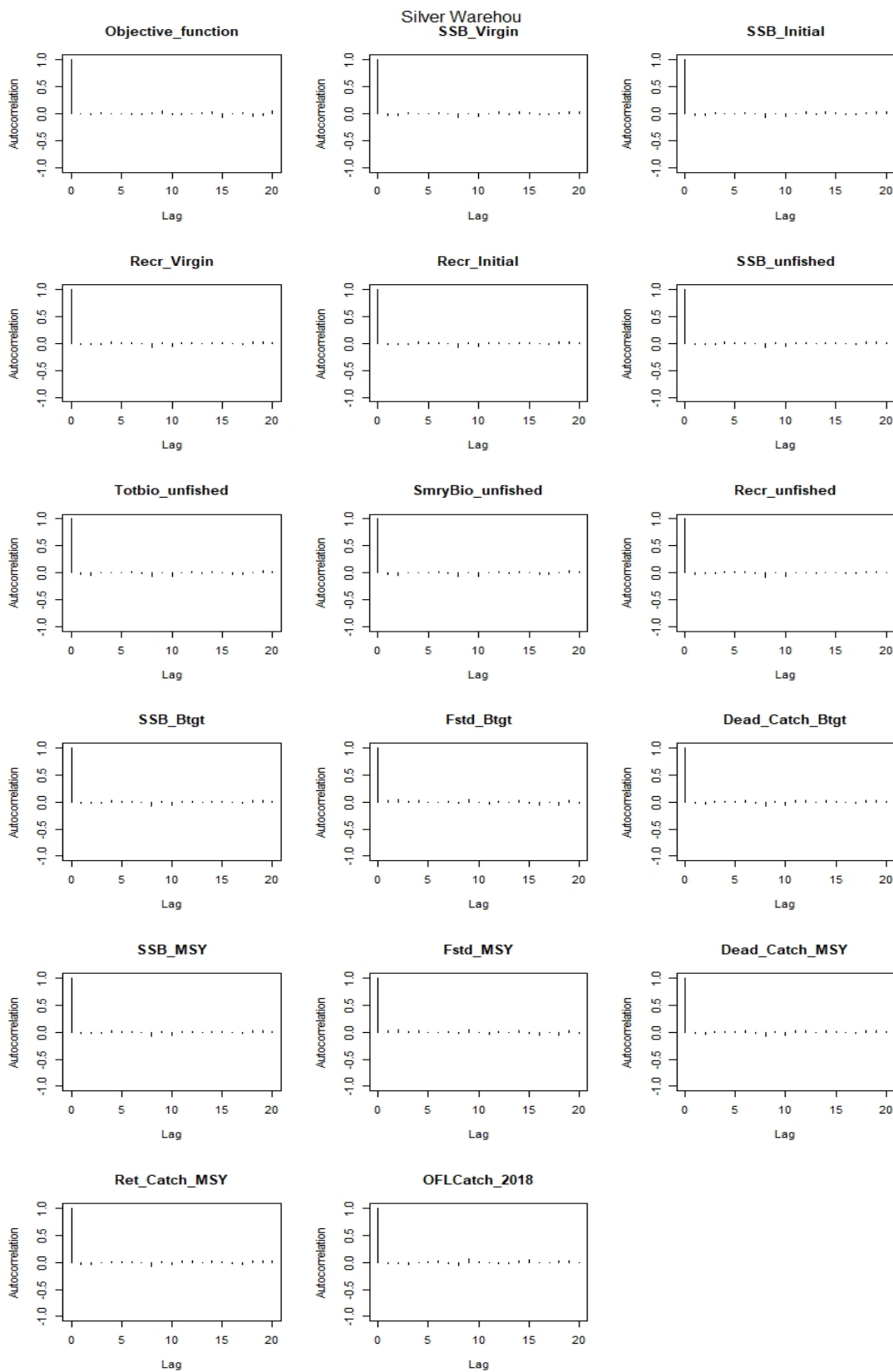


Figure B 12.4. Autocorrelation plots for time series from MCMC analysis.

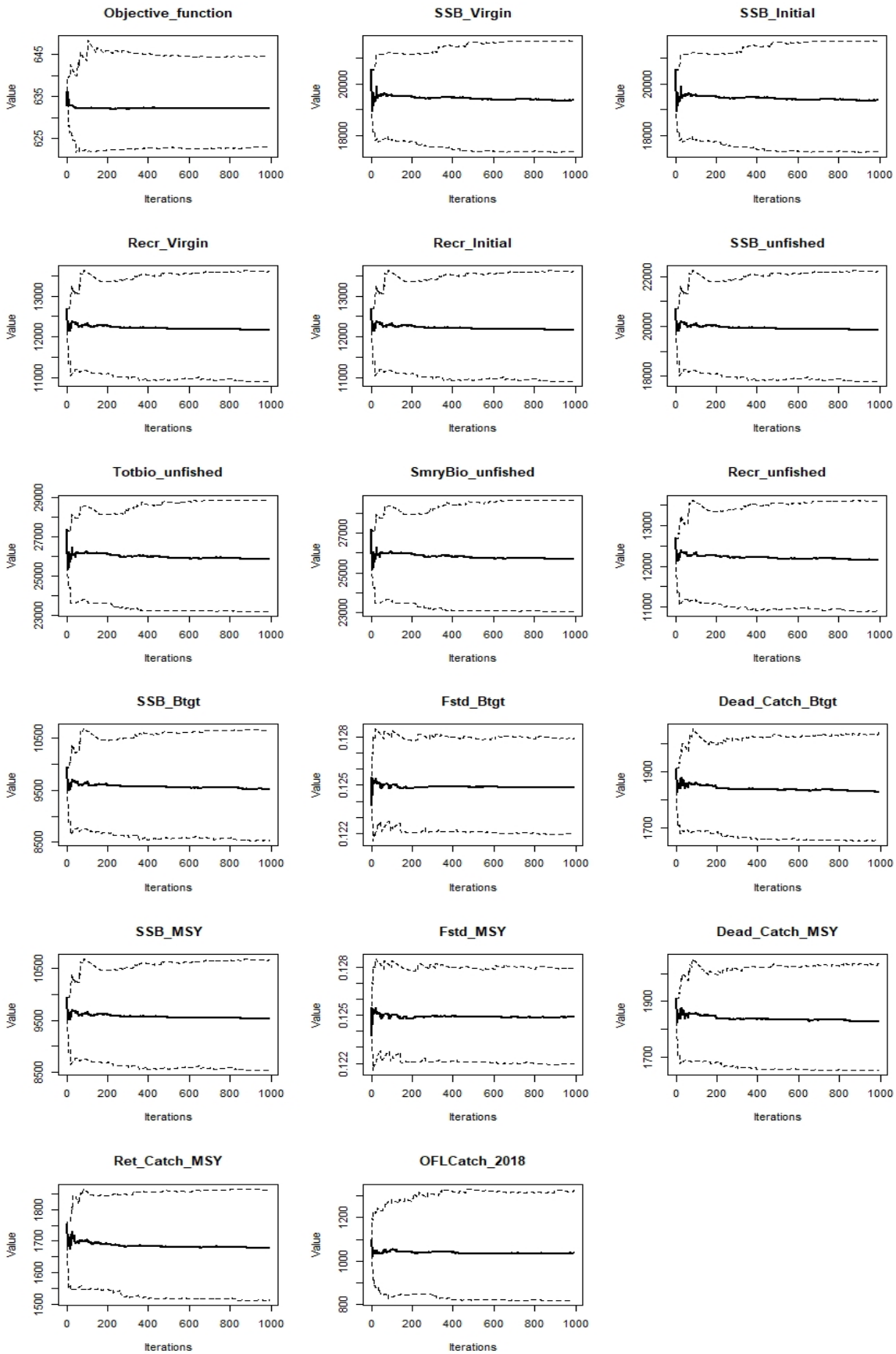


Figure B 12.5. Trace plot of time series for the base case from MCMC analysis.

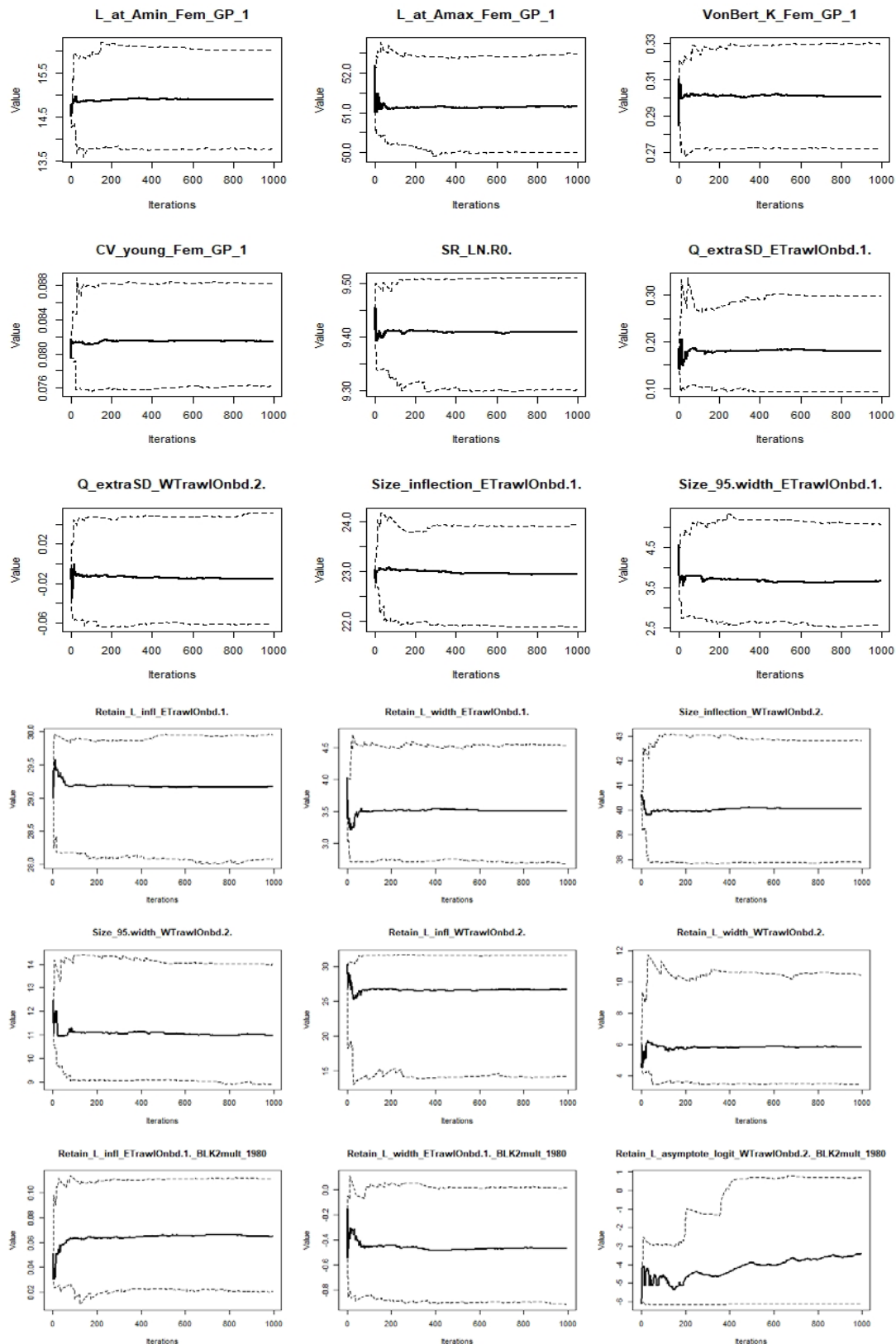


Figure B 12.6. Trace plots of growth, selectivity and retention parameters for the base case from MCMC analysis.

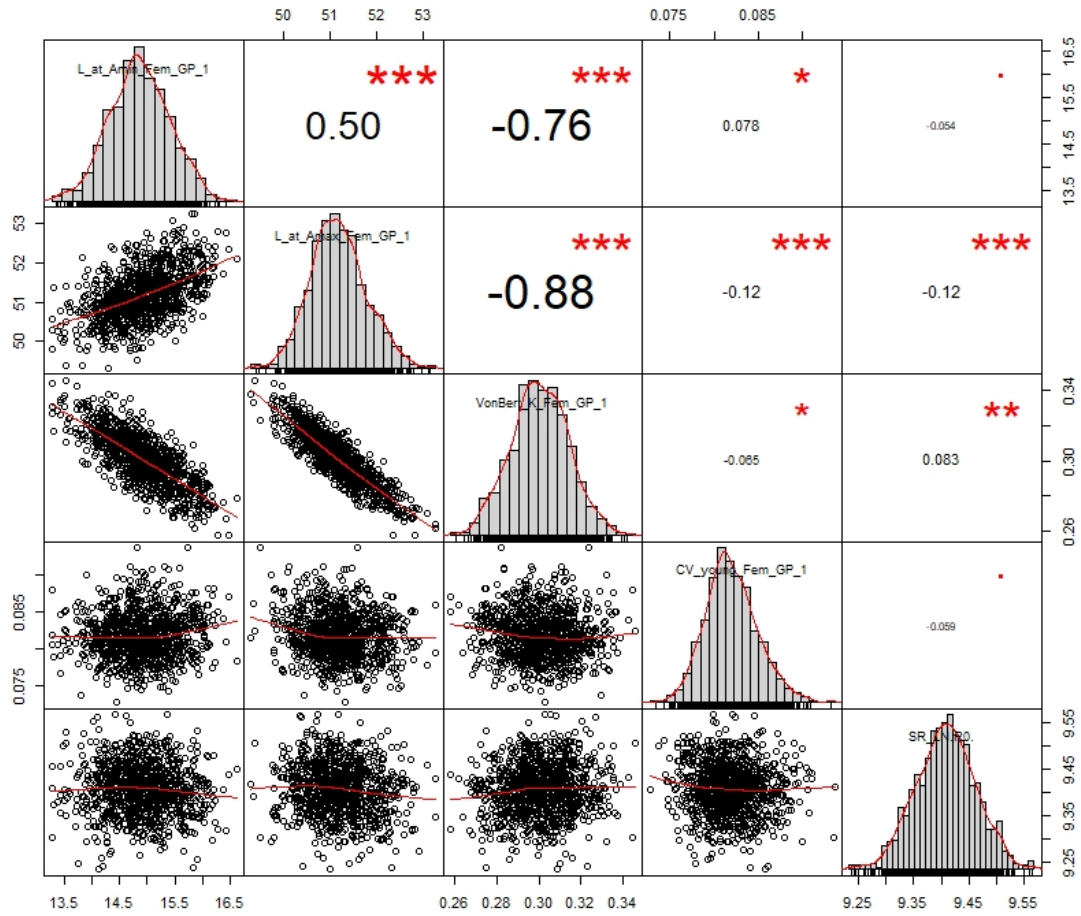


Figure B 12.7. Autocorrelation matrix and kernel density overlays of growth parameters estimated from the posterior distribution for base case assessment. Asterisks indicate statistical significance (0.05, 0.01 and 0.001).

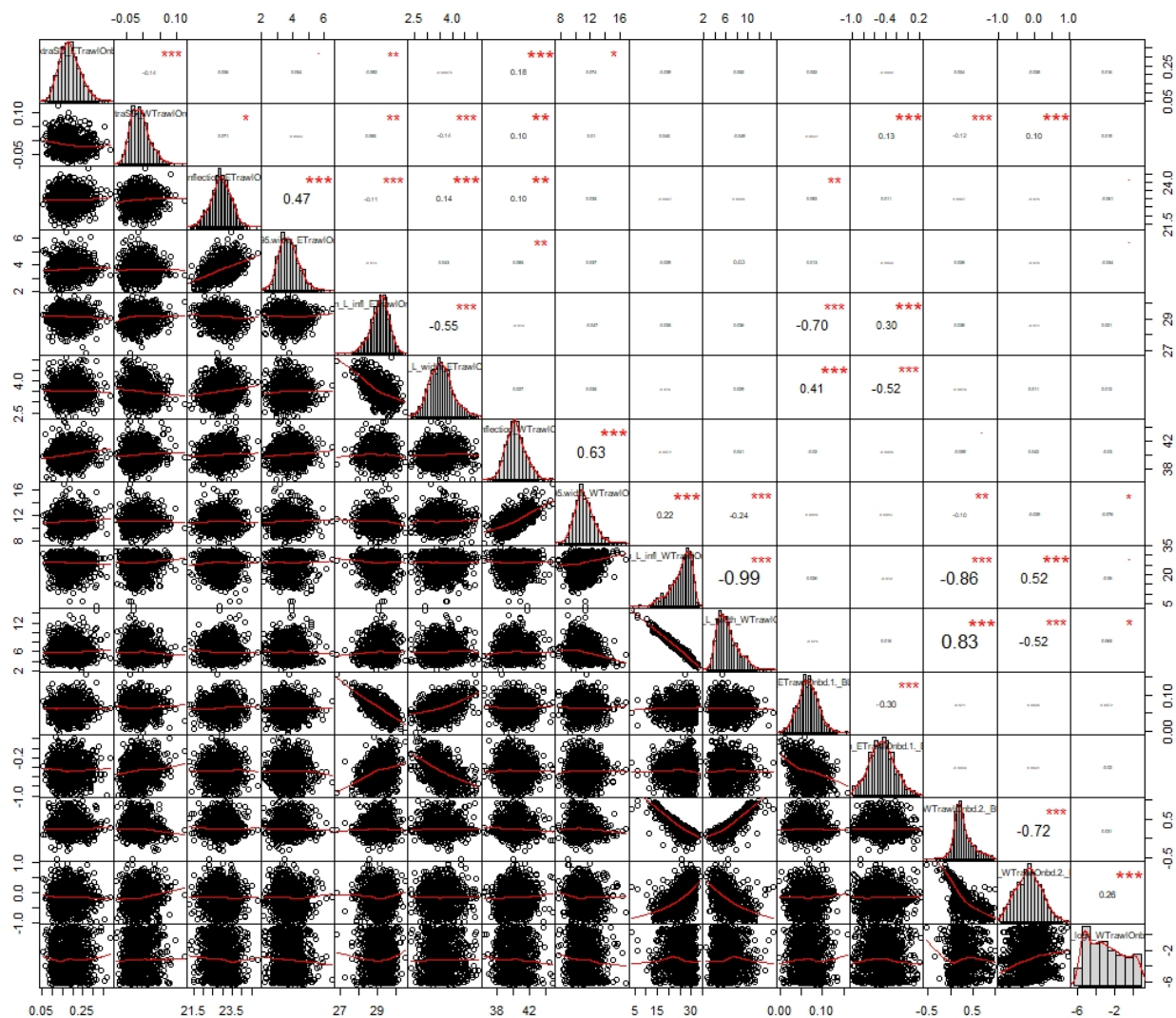


Figure B 12.8. Autocorrelation matrix and kernel density overlays of selectivity parameters estimated from the posterior distribution for base case assessment. Asterisks indicate statistical significance (0.05, 0.01 and 0.001).

Table B 12.1. Median estimates of key parameters with 95% credible intervals along with Geweke and Heidelberger and Welch tests derived from seven MCMC chains. Chain 4 (shaded grey) was used for results shown in this report.

| Chain | Parameter | Median (0.025, 0.975) | AC | | | Heidel-W |
|---------|-------------------------|--------------------------|--------|-------|----------|----------|
| | | | Lag1 | Eff.N | Geweke-Z | |
| Chain 1 | VonBert_K_Fem_GP_1 | 0.300 (0.27,0.32) | 0.036 | 795 | -0.207 | Passed |
| Chain 1 | SR_LN(R0) | 9.404 (9.31, 9.50) | 0.003 | 795 | 0.027 | Passed |
| Chain 1 | Main_RecrDev_2008 | -1.32 (-1.79, -0.908) | 0.017 | 795 | 0.484 | Passed |
| Chain 1 | Q_extraSD_ETrawlOnbd.1. | 0.173 (0.106, 0.274) | -0.03 | 795 | 0.369 | Passed |
| Chain 1 | Q_extraSD_ETrawlOnbd.2. | -0.015 (-0.05, -0.03) | 0.004 | 795 | -2.49 | Passed |
| Chain 2 | VonBert_K_Fem_GP_1 | 0.301 (0.278, 0.326) | -0.002 | 795 | -0.089 | Passed |
| Chain 2 | SR_LN(R0) | 9.40 (9.31, 9.49) | -0.007 | 664 | 2.175 | Passed |
| Chain 2 | Main_RecrDev_2008 | -1.30 (-1.80, - 0.936) | 0.06 | 510 | -0.791 | Failed |
| Chain 2 | Q_extraSD_ETrawlOnbd.1. | 0.17 (0.103, 0.275) | -0.029 | 795 | 1.227 | Passed |
| Chain 2 | Q_extraSD_ETrawlOnbd.2. | -0.016 (-0.05, 0.04) | 0.014 | 795 | 0.491 | Passed |
| Chain 3 | VonBert_K_Fem_GP_1 | 0.300 (0.276, 0.326) | -0.094 | 795 | 1.544 | Passed |
| Chain 3 | SR_LN(R0) | 9.403 (9.31, 9.50) | -0.044 | 795 | -0.54 | Passed |
| Chain 3 | Main_RecrDev_2008 | -1.29 (-1.78, -0.919) | -0.038 | 795 | 0.4 | Passed |
| Chain 3 | Q_extraSD_ETrawlOnbd.1. | 0.178 (0.10, 0.27) | -0.029 | 795 | -1 | Passed |
| Chain 3 | Q_extraSD_ETrawlOnbd.2. | -0.017 (-0.05, 0.04) | -0.026 | 795 | 0.771 | Passed |
| Chain 4 | VonBert_K_Fem_GP_1 | 0.300 (0.276, 0.326) | 0.041 | 795 | 0.793 | Passed |
| Chain 4 | SR_LN(R0) | 9.403 (9.31, 9.50) | 0.053 | 795 | 0.632 | Passed |
| Chain 4 | Main_RecrDev_2008 | -1.32 (-1.771, -0.921) | -0.006 | 795 | -0.553 | Passed |
| Chain 4 | Q_extraSD_ETrawlOnbd.1. | 0.181 (0.102, 0.44) | 0.018 | 795 | 1.829 | Passed |
| Chain 4 | Q_extraSD_ETrawlOnbd.2. | 0.016 (-0.05, 0.04) | -0.012 | 795 | 0.463 | Passed |
| Chain 5 | VonBert_K_Fem_GP_1 | 0.300 (0.277, 0.326) | -0.083 | 795 | -0.02 | Passed |
| Chain 5 | SR_LN(R0) | 9.40 (9.30 -9.499) | -0.073 | 795 | -1.099 | Passed |
| Chain 5 | Main_RecrDev_2008 | -1.327 (-1.767, - 0.941) | 0.025 | 795 | 0.179 | Passed |
| Chain 5 | Q_extraSD_ETrawlOnbd.1. | 0.177 (0.10, 0.275) | -0.043 | 795 | 1.253 | Passed |
| Chain 5 | Q_extraSD_ETrawlOnbd.2. | -0.01 (-0.05, 0.04) | 0.064 | 795 | 0.391 | Passed |
| Chain 6 | VonBert_K_Fem_GP_1 | 0.300 (0.277, 0.326) | -0.131 | 795 | 1.284 | Passed |
| Chain 6 | SR_LN(R0) | 9.40 (9.30, 9.499) | -0.005 | 795 | 0.698 | Passed |
| Chain 6 | Main_RecrDev_2008 | -1.299 (-1.794,- 0.91) | 0.042 | 795 | -0.018 | Passed |
| Chain 6 | Q_extraSD_ETrawlOnbd.1. | 0.177 (0.10, 0.275) | 0.004 | 795 | 0.778 | Passed |
| Chain 6 | Q_extraSD_ETrawlOnbd.2. | -0.01 (-0.05, 0.04) | -0.063 | 795 | -0.694 | Passed |
| Chain 7 | VonBert_K_Fem_GP_1 | 0.300 (0.277, 0.326) | -0.024 | 795 | 0.191 | Passed |
| Chain 7 | SR_LN(R0) | 9.40 (9.30, 9.499) | 0.0005 | 795 | -1.228 | Passed |
| Chain 7 | Main_RecrDev_2008 | -1.306 (-1.80, - 0.95) | 0.038 | 795 | -0.066 | Passed |
| Chain 7 | Q_extraSD_ETrawlOnbd.1. | 0.173 (0.10, 0.26) | 0.018 | 758 | 0.599 | Passed |
| Chain 7 | Q_extraSD_ETrawlOnbd.2. | -0.01 (-0.05, -0.04) | -0.002 | 795 | -1.07 | Passed |

B.3 Comparison with maximum likelihood estimate

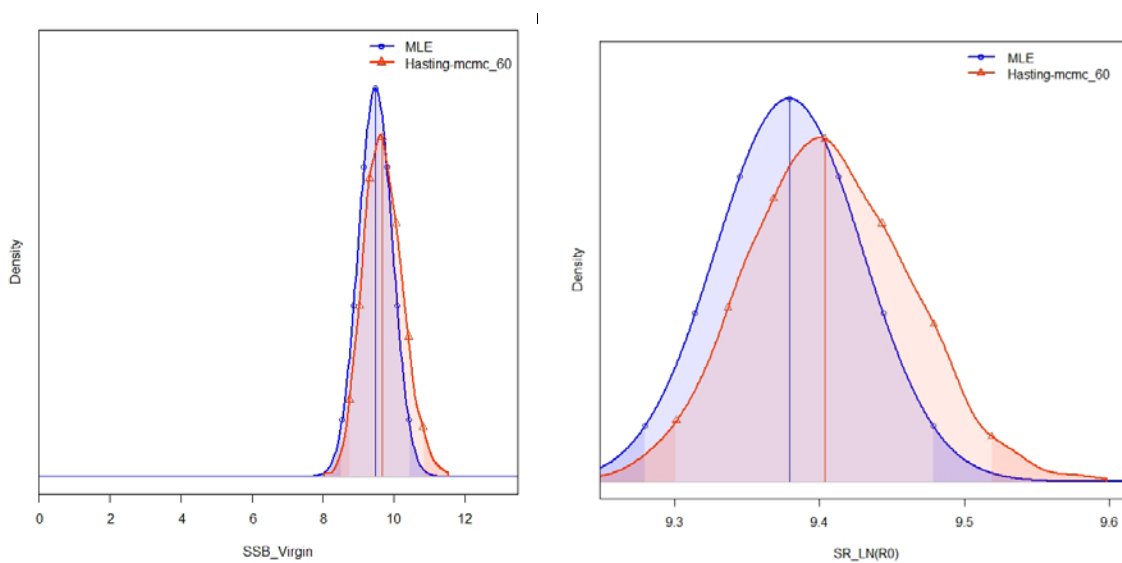


Figure B 12.9. Comparison of a single MCMC chain with the maximum likelihood estimate (MLE) estimate of virgin spawning biomass (left) and log of virgin recruitment (right).

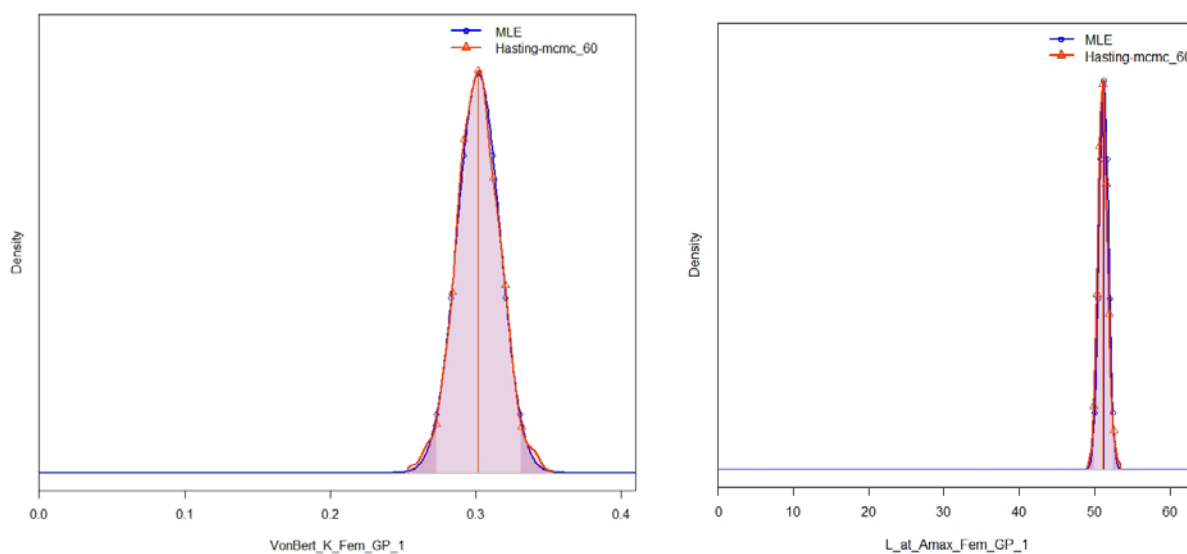


Figure B 12.10. Comparison of a single MCMC chain with the maximum likelihood estimate (MLE) estimate of von Bertalanffy growth parameters K (left) and mean length at maximum age (right).

13. Preliminary calculations towards a close kin model for school shark in the SESSF

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13.1 Abstract

Sampling for the school shark close kin project is complete, with approximately 3,000 sharks collected and genetically sequenced. A total of 3 parent offspring pairs (POPs, two mothers and one father) were found along with 34 full sibling pairs (FSPs) and 65 half sibling pairs (HSPs, i.e. two offspring with one parent in common) of which 27 were paternal and 38 maternal. The ratio of full to half siblings is relatively high, suggesting a large “litter effect” whereby some cohorts have unusually high survival due (possibly) to favourable environmental conditions (these are not expected to bias our estimates of abundance). There also seem to be a modest proportion of litters that have more than one father. All animals sequenced were also aged by counting vertebral “rings”. Relatively large ageing error was found (CV 0.08) and mature animals are known to have slower growth rates and to accumulate less than one vertebral ring per year of age.

Simple analyses of the proportion of half sibling pairs born since 2000, based on the facts that (1) each animal had exactly one mother and one father at birth, and (2) mothers and fathers may die over time, give a ballpark estimate for recent adult abundance. To accommodate some factors that are awkward to build into simple analyses - mainly, ageing error, catch history, and the very strong “stock-recruitment relationship” to be expected in depleted shark populations - we then constructed an age-structured population dynamics model that uses commercial catch and discard data, length frequencies from port measured gillnet catches (although these were given negligible weight), estimates of gear selectivity and several biological parameters used by the sharkRAG stock assessment model for school shark, as well as the close kin data. The model follows the same approach used for close kin mark recapture (CKMR) for Southern Bluefin Tuna (SBT) and several other species, whereby the probability that each pairwise comparison of two animals will prove to be a close kin pair is computed based on the working values of the population dynamics parameters, taking account of the ring counts, years of capture, and sex of the two animals concerned. The actual outcome of that comparison (e.g. that it was a maternal half sibling pair) is then compared with the computed probability, and parameters are adjusted (following the best statistical principles) to give the best fit between observed and expected values. Probability distributions were constructed for the age of each animal, given its ring count and accounting for ageing error and ring deposition rates at age.

Compared with the 2012 projection of the stock assessment model for school shark, which assumed catches of 225t after 2011, the simple analyses and the close kin model both estimate a substantially lower adult abundance. The assessment projection and the close kin model (as well as the simple approaches) both indicate an upward trend in abundance since 2000, of a similar rate (although the confidence interval on trend is quite wide).

This close kin model requires assumptions which may not hold far back into the history of this fishery, particularly those regarding density dependence. We therefore restricted attention (for now) to the 2000-2017 period, when most of the close kin sample were born and where the information content is

strongest. This was done by restricting the (estimated) age of included samples, leaving out the oldest. This did reduce the “sample size” (to 1,627 out of 2,438 original samples, and 29 out of 40 maternal half-sibling pairs, and a shorter window). The restriction led to satisfactory model fits, but more uncertainty about abundance than might be obtained with the complete dataset. In addition, because we had no prior estimates of whether male fecundity varies much through adulthood, and not enough POPs to estimate it, we took a conservative approach for the moment, of not considering the 27 paternal half siblings and the single father-offspring pair. If the model can be expanded to include the historical data adequately and include more of the samples, the CVs will improve.

The purpose of developing the close kin model was to better understand the new close kin data for school shark. The next step is to incorporate those data, and the close kin probability calculations and likelihood, into a full stock assessment model that (for example) allows density dependence to change over time, and allows for spatial segregation by size / age as well as other complexities identified in the past such as multiple stocks. On balance, we would also recommend including paternal kin pairs. This should allow the use of close to, or all, of the 102 kin pairs found and consequently yield lower CVs on quantities of interest.

The stock assessment model used by sharkRAG has been limited by the absence of an index of relative abundance after 1997 and has never been able to disentangle abundance from productivity without the use of a prior based on “expert opinion”. Close kin data does provide a fishery-independent estimate of absolute abundance, productivity, and spawning stock trend, and can thus obviate the need for the prior.

13.2 Introduction

The close kin mark recapture (CKMR) method for estimating abundance and other demographic parameters (Bravington et al 2016a) was first applied to Southern Bluefin Tuna (Bravington et al 2016b) with great success. It has since been applied to white shark in eastern Australia (Hillary et al 2018) and applications to western white sharks, two populations of an endangered freshwater shark (*Glyphis glyphis*) in northern Australia, and eastern Australian grey nurse shark, are nearing completion. A close kin project for school shark (*Galeorhinus galeus*) in south-eastern Australia, caught in the SESSF fishery, has been underway since 2015.

Whereas the close kin application to SBT uses both parent-offspring pairs (POPs) and half sibling pairs (HSPs) the bulk of the school shark catch is composed of juvenile animals, so that few POPs were expected (and indeed only three were found). There is minimal information from such low numbers, consequently we have concentrated entirely on the much more numerous HSPs in this report. The fundamentals of HSP-based CKMR are very simple. Here we describe the basic idea, and then list the factors that need to be allowed for.

Suppose all female adults are “reproductively similar” (i.e. expected to produce approximately the same number of surviving offspring per year). Now sample two fish, which for simplicity we will name Peter and Simon, born within a few years of one other (Peter is the elder). What is the probability that Peter and Simon have the same mother, i.e. are a maternal HSP (MHSP)? Simon's mother could have been any of the adult females alive at the time of Simon's birth (we will call that number N_{fem}). The chance that she is the same as Peter's mother is therefore “about” $1/N_{\text{fem}}$. Thus, by making pairwise comparisons amongst a large sample of juveniles and seeing what proportion of them yield an MHSP, we can “basically” estimate N_{fem} . Of course there will be some random variability in the

number of MHSPs actually found, and hence uncertainty in the estimate; but if the number of MHSPs actually found is fairly big, then the relative random variability in the proportion cannot be large. This argument would be exact, and there would be no need for "about" and "basically", except for the following four factors:

1. Peter's mother may have died before Simon was born. This reduces the probability of them being an MHSP, so mortality rates have to be allowed for.
2. Within-cohort comparisons tend to have a systematically higher proportion of MHSPs (and full sibling pairs, FSPs), because of random events that affect the survival rate of an entire litter. Same-cohort comparisons need to be excluded, or specifically allowed for, in any model using close kin data. The only reliable HSP signal comes from cross-cohort comparisons.
3. Adults of given sex may differ systematically in reproductive output. This is inevitable in species where body-size strongly affects fecundity, e.g. teleost fish. Variability between (female) adults will increase the proportion of (M)HSPs, as will somatic growth within any adult's lifespan (if Peter's mother doesn't die by the time Simon was born, then she would become more fecund, which would increase her relative probability of also being Simon's mother--- at least, if Peter and Simon were teleosts). This is why HSP-only CKMR cannot work for teleosts; it is necessary to have POPs as well (from which fecundity at size relationships can be estimated).
4. If there is a trend in adult abundance, then the probability depends on the total number of females alive at Simon's birth, not the "average" number of living adults. This is easy to build into a model.

In general, HSP-based CKMR is a little more delicate than POP-based CKMR, because all sources of systematic variation in reproductive output between female adults (or between male adults) needs to be accounted for in the HSP model to avoid bias. However, random variation from one breeding event to the next does not cause bias.

This report first presents a simple application of the basic principle that the number of kin pairs is related to inverse adult abundance, thus straightforwardly deriving an estimate of (absolute) abundance for school shark using close kin pair data alone. Second, we present a more sophisticated model (hereafter the *close kin model*) that accounts for the four complicating factors listed above. The first, straightforward, application was tackled in two ways: a very simple, essentially one-line calculation (hereafter termed the *one line calculation*) and a more sophisticated Generalised Linear Model (GLM) based calculation that allows for a trend over time in abundance, and for survival between years (*the GLM model*).

This report is an update on Thomson et al (2018) which was based on half the target sample size, much less powerful mitochondrial DNA information, incomplete ageing, and more cursory model development. This report uses approximately 3,000 samples, all of which were aged by the Fish Ageing Service (FAS) using vertebral counts. The full mitochondrial genome was sequenced for those sharks found to be in close kin pairs, thus indicating whether the shared parent was the mother or the father (see below).

In addition to the close kin data, the close kin model uses data relating to commercial catch, discards, length frequencies as well as gear selectivity, and known biological parameters such as relative pup production by (female) age, to estimate parameters that describe the stock. Several assumptions are adopted from the stock assessment model used in the past by AFMA and sharkRAG to manage school shark (Punt & Walker 1998; hereafter the '*stock assessment model*'). That model was last updated in 2009 using data to 2008 (Thomson & Punt 2009) but was used in 2012 for forward projections under

a range of scenarios regarding future line fishing and overall levels of catch (Thomson 2012). That version of the model freed a parameter that governs productivity of the stock, thereby allowing the data to inform the model regarding the selectivity of the stock. Landings, discards, and length frequency information collected between 2009 and 2017 was compiled for incorporation into the close kin model, along with the data to 2008 that was used in the 2009 stock assessment. The estimates of biological parameter values used in the 2009 stock assessment, along with several assumptions described in Thomson & Punt (2009), were also used in the close kin model presented here.

Bravington et al (2016a, 2016b) describe how to properly set up close kin mark recapture models for general situations, with genetically determined ‘marks’ and ‘recaptures’ (of closely-related animals) arising from commercial landings or surveys. Depending on the biology of the species and which types of kin can be found (POPs and/or HSPs), it may or may not be important to have time-series of age/length compositional data; species where adults (of given sex) do not vary much in expected reproductive output, such as whales and many sharks, have less stringent data requirements. Close kin mark recapture models do not require an index of relative abundance, nor does it need to account for e.g. seasonal movement details unless the latter affect the breeding or sampling probabilities underlying the close kin model. Catch (removals) data are useful (though not absolutely essential, unlike in conventional stock assessments), and do allow the separation of natural mortality (M) from fishing mortality.

A stark difference between the school shark stock assessment and the close kin model presented in this report, is that the stock assessment used CPUE from the fishery as an index of *relative* abundance, whereas the close kin model uses information on close kin pairs to give an *absolute* abundance signal that is not susceptible to changes in management and fishing practices. The stock assessment model also used conventional tag-recapture data, which is not incorporated in the close kin model (although it could be in future). The stock assessment model uses gillnet fishery CPUE data to 1996. After that date, management regulations including deliberate avoidance of school shark caused the CPUE series to break down as an index of abundance for school shark.

The purpose of the close kin model presented in this report is to explore the newly collected data, and to build an understanding of how those are best modelled. It is not yet intended for use as a management tool. Instead, the stock assessment model will be updated using commercial fishing data to 2017, retaining the tag-recapture data, adding the new age composition and ageing error information, and incorporating the close kin data into the likelihood. The close kin data will provide an index of absolute abundance; not only will this compensate for the absence of an index of relative abundance after 1996, but it will allow the model to better estimate the productivity parameter. Productivity and abundance are, typically, correlated in stock assessment models; the available data can often be equally well described by a model that has high productivity and low abundance, or by one that has low productivity and high abundance. Despite a dataset that shows great contrast due to overfishing in the past, the school shark stock assessment model is unable to estimate productivity, which therefore has to be informed by a prior that was established by asking the then sharkFAG members for their opinion on what productivity might be. Close kin data can do away with the reliance on that prior.

The stock assessment model incorporates many aspects of school shark biology (such as movement and consequent regional differences in availability at age, and the likely existence of a NSW stock that was depleted some time ago) that are ignored by (and potentially irrelevant to) the much simpler close kin model. Merging the close kin data into the existing stock assessment model (or possibly into a simpler assessment that nevertheless captures all the important aspects of the school shark stock(s)) provides the most defensible approach for managing school shark. In doing so, there are many issues to be considered by sharkRAG, such as the desirability or otherwise of including other CPUE series,

e.g. from the trawl fishery which does not target school shark. But before that can be done, it has been useful to construct a simpler close kin to explore the main (and some unexpected) features of the close kin data, without distraction from the many complexities of the stock assessment model.

13.3 Close kin data

13.3.1 Sample size distribution

With the help of the fishing industry, approximately 3,000 school shark samples were collected for the close kin study. Sample sizes markedly exceeded the collection targets in all locations except Tasmania (Table 13.1).

Table 13.1. Targets and collection totals for school shark close kin sample.

| <i>State</i> | <i>Target</i> | <i>Collected</i> |
|-----------------|---------------|------------------|
| South Australia | 700 | 1,318 |
| Bass Strait | 900 | 1,378 |
| Tasmania | 400 | 339 |
| TOTAL | 2,000 | 3,035 |

The close kin samples consisted of a section of the vertebral column taken from just behind the head, along with a lump (or sometimes, unfortunately, only a veneer) of tissue. Information was also supplied on the collection location, date, and the sex and length of the animal.

All samples that had adequate tissue quality and quantity (with the exception of some that came from catches where more than between 50 and 100 animals were sampled) were sent to Diversity Arrays Technology (DArT) in Canberra for sequencing. Of the 2,886 samples for which genetic sequences were obtained, 2,438 passed all quality control tests (see ‘Kin finding’ below).

Examination of the genetic sequences revealed 8 pairs of duplicated animals where the same animal was sampled twice on the same day. Such replication can easily happen (Chris Pitliangas pers comm) and is easily detected and overcome by randomly eliminating one of the duplicate samples. Another four pairs of duplicates arrived (both animals in each pair coming from one supplier) with reported sample dates that were between two and five days apart – these were also interpreted as accidental repeated sampling on the factory floor, as was one animal that was sampled twice on 8 Aug 2017 and a third time on 10 Aug 2017 by the same supplier. There were also 13 samples that were found to be a species other than school shark. These were most likely gummy sharks, sampled in error.

13.3.2 Ageing and age error

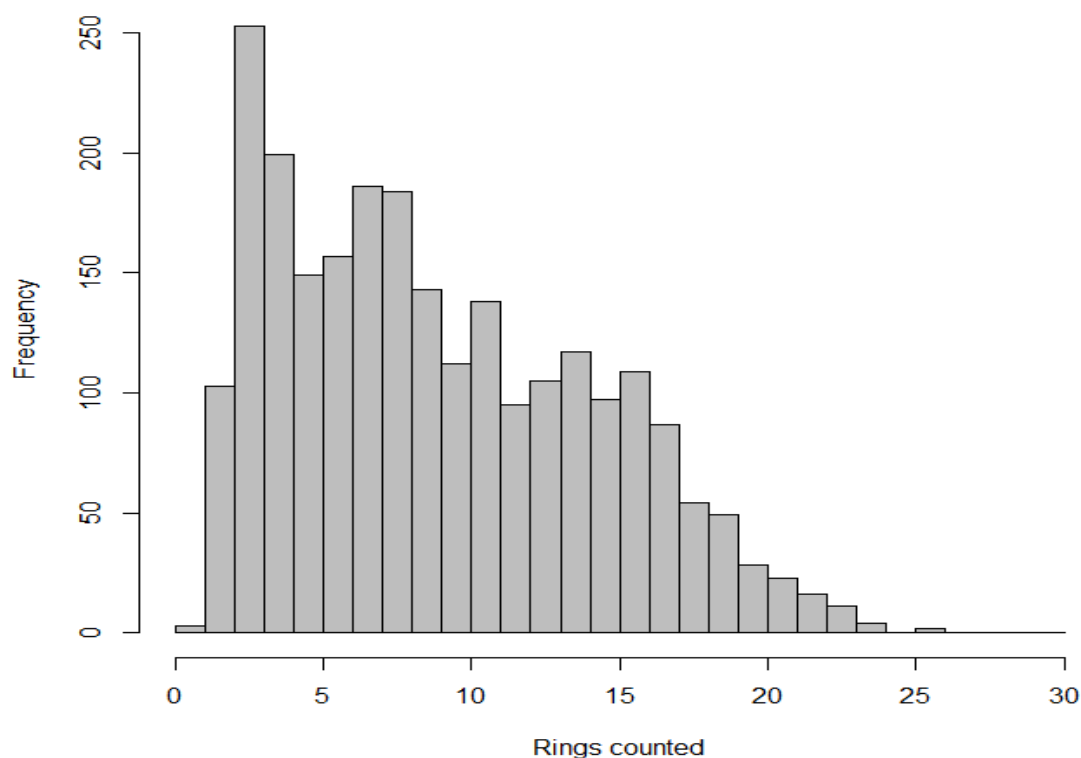


Figure 13.1. Frequency distribution of “rings” counted for the 2,438 animals used for kin finding.

The vertebrae were aged by the Fish Ageing Service (FAS) by counting mineralised zones (hereafter termed “rings”). The largest number of “rings” counted was 26 (Figure 13.1).

Vertebral ring counts correspond closely with actual age up to roughly age 11 (Moulton et al 1992), the age at which females mature and growth rate slows. After this age, zone deposition rate is lower than one per year (Walker et al 2001). Although Walker’s estimate of 0.36 rings laid down per year after the age of 11 is based on only five individuals, individuals (and it is not obvious whether it should apply equally to males as to females), theirs is the only measurement of this type available. They additionally trialled an alternative method for staining the vertebrae (which they deemed to be less reliable) that gave an estimate of only 0.25 rings per year. Sensitivity of the close kin model to using either value was examined. Kalish (2002) and Fenton (2001) used bomb radiocarbon dating to show that vertebral counts greatly underestimate the ages of older school shark but their work does not give estimates of annual ring deposition rate.

FAS randomly selected a set of vertebrae for recounts of deposition zones (“rings”) which showed that ageing error is not negligible (Figure 13.2). A CV of approximately 0.08 was found (Andre Punt, CSIRO, pers comm) when allowing for errors in both the ‘Age1’ and ‘Age2’ reads, and between individual readers using the method of Punt et al (2008).

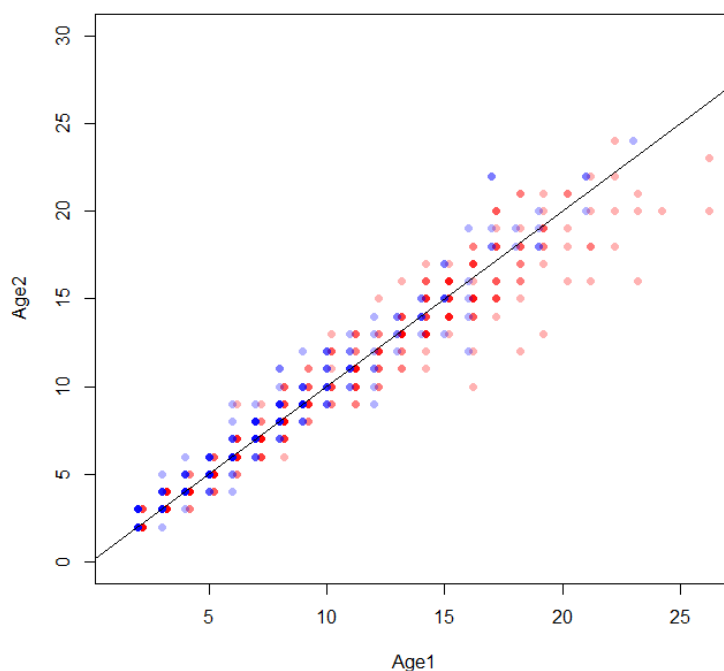


Figure 13.2. Number of vertebral zones counted (age) during a first (Age1) and second (Age2) reading of a random selection of school shark vertebrae. Ages for males sharks (blue dots) are offset slightly so that they do not overlie those for female sharks (red dots). Note that these are not true ages but zone (“ring”) counts.

13.3.3 Carcass length

The reported carcass length measurements were occasionally the total length, but most often the partial length or, equally often, the dressed length. Conversion of supplied length into total length was not always straightforward and some uncertainty surrounds the actual length measurement type of many of the samples. Further information is given in the Appendix. Uncertainty regarding the accurate conversion of the supplied carcass lengths, to total length, was such that the length measurements could not be used in the model.

13.3.4 Sex

The reported sex of the animal was verified (and in some cases corrected) using sex markers found in the genetic sequences. Of the 2,438 animals used in the close kin model, 99 were supplied without information on sex, 31 of which were found to be male and 68 female. Of those reported to be female, 90 out of 1,427 (6%) were corrected to male; and out of 912 reported males, 23 (3%) were corrected to female. The sex ratio in the final sample (of 2,438 animals) was 57% female versus 43% male.

13.4 Kin finding

The basis of CKMR is finding Parent-Offspring Pairs (POPs) and Half-Sibling Pairs (HSPs; where two animals have one parent in common). It is also sometimes necessary to deal with other types of kin, such as Full-Sibling Pairs (FSPs). To find HSPs, FSPs, and POPs for school shark, we applied the same genotyping and statistical techniques that have been successfully developed and used at CSIRO

on Southern Bluefin Tuna, white sharks, grey nurse sharks, and *Glyphis* spp (northern river sharks). The technical principles are explained in Bravington et al (2016) section 5; more details on the process for school shark are given in the Appendix. Here we summarize the outcomes.

13.4.1 Distinguishing HSPs

HSPs are harder to find than POPs or FSPs, because the degree of relatedness is less i.e. there is less shared genetic material. The key is to use plenty of loci, and to make sure that the genotyping process is highly reliable (i.e. few sequencing errors). For each pair that is being checked, a statistic (the "PLOD"—pseudo-log-odds-ratio) is calculated; the histogram of all the PLODs should show the HSPs as a distinct bump in a predictable location, largely separated from the vast majority of comparisons which will consist of UPs (Unrelated— or very weakly related— Pairs). Any FSPs and POPs will tend to have very high PLODs and will sit to the far right. This is exactly what we see for school shark (Figure 13.3); note that the enormous bump of UPs (about 4.5 million comparisons) mostly sits beyond the left-hand edge of the graph and has been excluded to make the HSP bump visible.

A small overlap between UP and HSP distributions is inevitable, genetically. To deal with this, the red line is chosen (visually) as a safe threshold such that very few UPs are likely to have PLODs to the right of the red line— i.e., to exclude false-positives. Pairs are only counted as definite HSPs if their PLOD is to the right of the red line. Since this will lead to some false-negatives (true HSPs that are rejected by having an accidentally low PLOD), an adjustment is made in the CKMR model to allow for the likely proportion of false-negatives, which is estimated to be about 12% in this case. (Note that the exact location of the red line does not bias the results; moving it will affect the number of observed kin-pairs, but will also affect the probability of detecting a kin-pair, by a similar amount.)

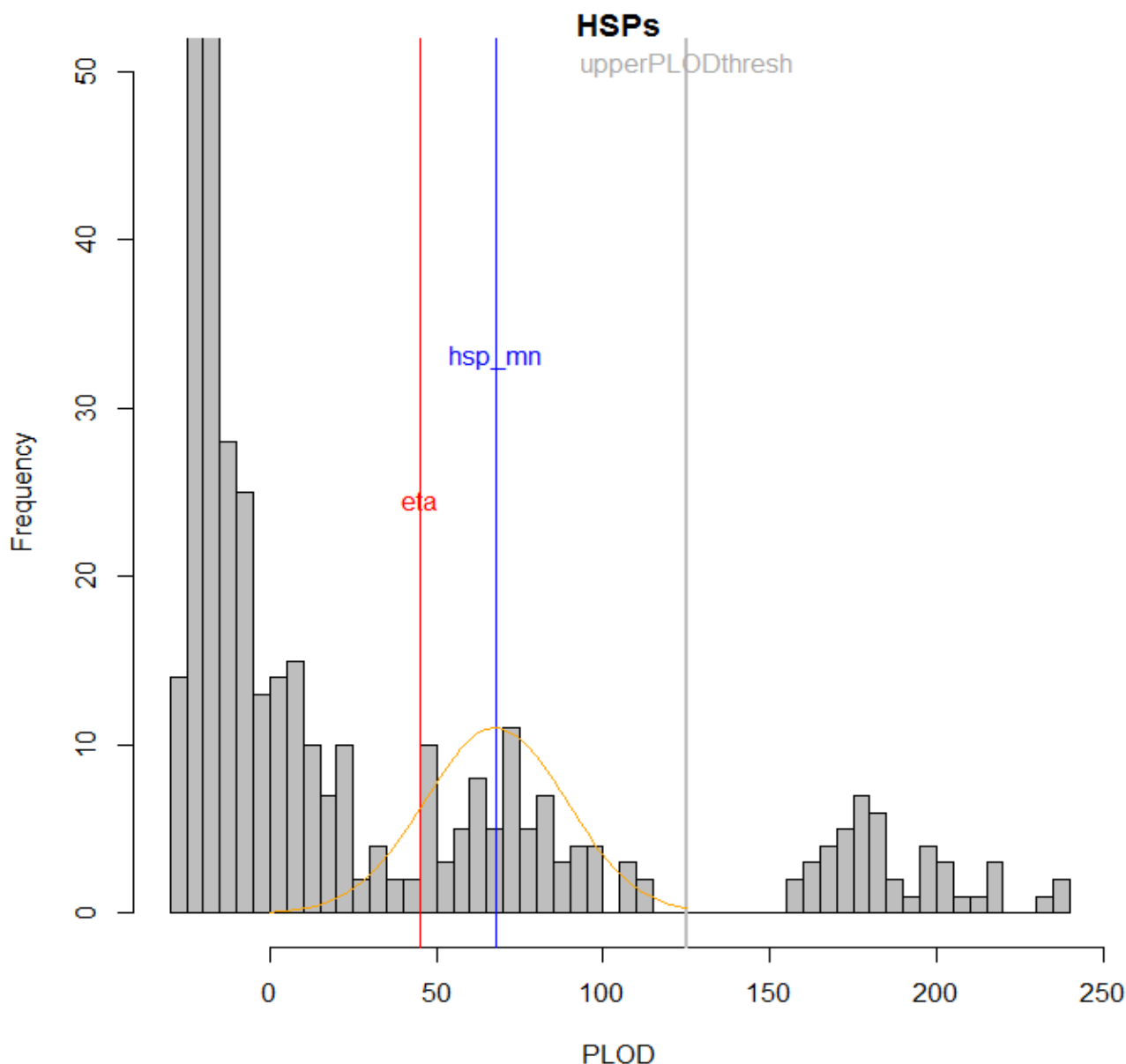


Figure 13.3. Numbers of pairs whose relatedness fell within the higher PLOD values. The theoretical mean value for HSPs is shown (blue line) as well as the threshold value, eta (red line) above which it was felt that no unrelated pairs (or less related than HSPs) would fall by chance. The pairs to the right of the grey line are POPs of FSPs. The orange curve shows a normal distribution centred on the theoretical HSP mean and with variance equal to that for observed HSPs above the theoretical mean i.e. the theoretical HSP distribution.

13.4.2 mtDNA and mother/father implications

The reproductive dynamics of male and female school sharks are different (e.g. earlier maturation of males) so it is important to decide which parent each HSP is related through: M(ateral)HSP or P(ateral)HSP. This can be done with mitochondrial DNA (mtDNA)—quite separate to the DNA used in identifying kin-pairs. mtDNA is always inherited from the mother only, so if the two animals have different mtDNA "signatures" (known as haplotypes), then they must be related through the father; i.e., they must be a PHSP. If they have the same haplotype, then they are probably related through the mother, but since there are only a modest number of different haplotypes at large in the population, they may by chance be a PHSP where the two mothers shared a haplotype by chance.

The entire mitochondrial genome (as opposed to just the ‘control region’ as per Thomson et al 2018) was sequenced for every animal found in a close kin pair. We found 122 mitochondrial haplotypes of which the most common was shared by just 5% of the sample. This gives very powerful information for discriminating maternal from paternal HSPs; we estimated a mere 3% chance of the HSPs in this sample sharing their mitochondrial haplotype by chance. This allowed us to distinguish half siblings that shared a mother (maternal HSPs, MHSPs) from those that shared a father (paternal HSPs, PHSPs). Among the 65 HSPs that we found, 38 had the same haplotype. This means two things. First, it backs up our HSP-finding; it is impossible that so many pairs would have the same haplotype if they were really unrelated. Second, it suggests there are substantially more "typical" adult males than "typical" adult females. The difference was found to be consistent with close kin model estimates—made only with the MHSPs—based on the later age of maturity, and progressive fecundity increase post-maturity, in females.

Since the chance of sharing a haplotype by chance is so low, we simplified the modelling by interpreting all shared-haplotype HSPs as MHSPs and all different-haplotype HSPs as PHSPs. This might mean that one or two of our nominal MHSPs are actually PHSPs, but the overall impact on the CKMR model should be small. (This is what we did for SBT, which has similarly high diversity of haplotypes. For some other, rare, shark species where there are very few distinct haplotypes, we have had to build more elaborate probabilistic models.)

13.4.3 FSPs and POPs

FSPs and POPs have the same average degree of relatedness, and are easy (collectively) to separate from HSPs. They are obvious on the right-hand side of Figure 13.3 (to the right of the grey vertical line), and indeed a formal statistical test identifies them easily from the HSPs.

FSPs are not much use in CKMR, because they are almost certain to be litter-mates (with an adult population of the order of 100,000, only a tiny proportion of repeat matings will occur) and same-cohort comparisons are explicitly excluded from CKMR calculations, for reasons described earlier.

Separating FSPs from POPs based purely on genetics can be done in principle, but is slightly tricky. In the case of school shark, though, separation is easy, based on the likely gap in birth-cohort; FSPs will be from the same cohort whereas POPs must be separated by at least the age of maturity. Male school shark are thought to be mature from 7 and females from 11 years old (Olsen 1954, Walker 2005).

13.4.4 Cohort-gaps in kin-pairs

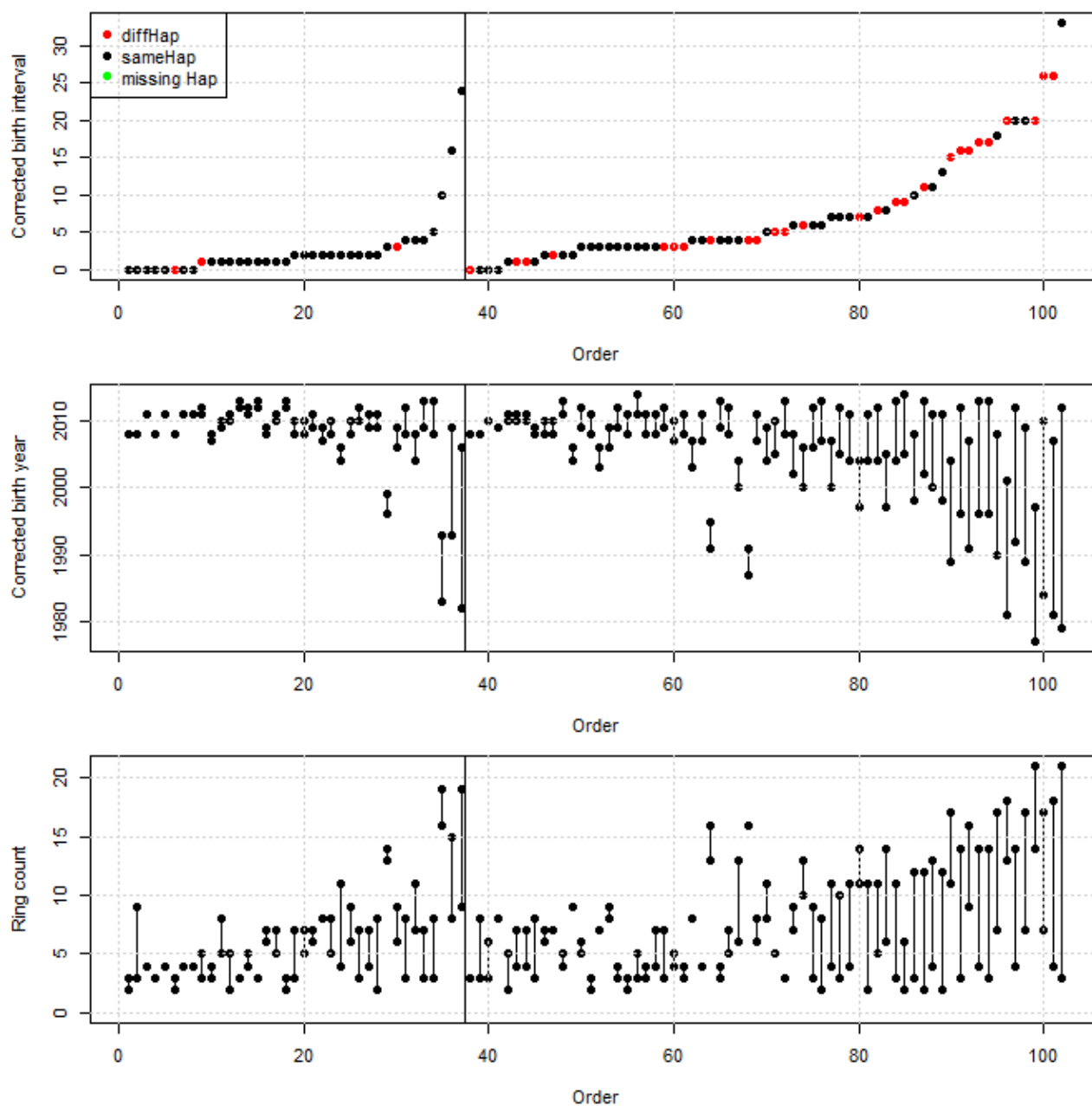


Figure 13.4. Gaps between estimated birth-years of each kin-pair (correcting for ring-count-rate in adults), sorted by increasing gap (upper plot). FSPs and POPs are shown on the left; HSPs on the right). Note 3 cases of FSPs with apparently different mtDNA haplotypes (red dots); these can only have come from mix ups in processing during mtDNA processing, a secondary process involving numerous lab steps. Also, corrected birth year (middle plot) and ring count (lower plot) for both animals.

Figure 13.4 shows, for each kin-pair, the gap between estimated birth-years in the pair (upper plot) along with correct birth year and ring count for each animal involved in the kin pairs. The birth intervals shown in Figure 13.4 are based on corrected ring counts, where the correction assumes no ageing error and no variability in the ring deposition rate (0.36 rings per year after age 11).

Among the FSPs/POPs (left panel), the three rightmost pairs, and only those pairs, have a gap large enough to be POPs. This very small number of POPs turns out to be roughly as expected, given the age distribution of the sampled animals. The remainder of the left-hand panel must be FSPs, and the maximum apparent gap is 5 years. Most of the FSPs have a gap of 0–2 years, which is entirely explicable in terms of ageing error on animals from the same cohort. The six FSPs with gaps of 3–5 years are either due to ageing error (certainly plausible), and / or conceivably to sperm-storage, whereby a female uses sperm from one mating to fertilize not just one litter but the next (known to occur in several shark species). If sperm storage does occur (and if it does, then it clearly does not often cover long gaps), then apparent cross-cohort FSPs (from successive matings) should be treated demographically as if they were MHSPs; if not, they can be basically ignored in the CKMR model. We have chosen, for now, to assume that all the FSPs are same-cohort, and hence that the up-to-5-year-gaps in apparent birth-cohort are due entirely to ageing error; assuming sperm storage instead would somewhat lower the estimated abundance.

The number of FSPs found (34) is surprisingly high compared to the number of HSPs, since there are many more "mating opportunities" for HSPs compared to FSPs, which must come from a single mating. The discrepancy suggests a substantial "lucky litter" effect (where some litters have unusually high survival because of favourable environmental conditions, and consequently generate a disproportionate number of within-cohort siblings). For this reason, additional parameters are estimated by the close kin model to quantify this "litter effect", as well as the proportion of full to half siblings within a litter.

Among the HSPs (right panel), there are some very distant gaps, which could be Grandparent-Grandchild-Pairs (GGPs) instead of HSPs; those two types of kin are genetically indistinguishable. In our CKMR estimates, we have assumed that all detected HSP-like pairs really are HSPs, i.e. we have not incorporated the small additional probability that they might be GGPs. Including some GGPs by accident would have some impact on the CKMR model, so to mitigate that issue (among others) we excluded the oldest animals from HSP comparisons in the close kin model(s), as described later.

Among the younger HSPs, there is some suggestion of more black dots (MHSPs) at a 3-year gap than at 0, 1, or 2-year gaps, perhaps suggesting a 3-year pupping interval. At least some of the MHSPs found at short gaps (0–2 years) may well be same-cohort, which would indicate some low level of multiple paternity within litters (based on the proportion of short-gap MHSPs to FSPs). Hernandez et al (2015) provide alternative evidence of multiple paternity in school shark, albeit by a much less direct route than mtDNA comparisons among same-cohort HSPs. No real patterns are evident in the red dots (PHSPs); there is of course nothing to stop males mating every year.

13.4.5 "Triad" families

Interestingly, we found 8 "family triads" in which at least one fish was the sibling of two others. Of those 8 families, 6 comprised one FSP and two HSPs (i.e. A and B share mother and father; C has either the same mother, or the same father), and two comprised two HSPs (i.e. A has same mother as B; A has same father as C; B and C are unrelated). The three fish involved in each family were caught in different fishing trips in all but one case. That case consisted of an FSP that were caught together, and a half-sibling that was caught in a different trip. The proportion of kin-pairs involved in triads should be very low in large populations, but increasingly common in small populations; triads are rife in grey nurse shark and white shark, for example, but scarce or totally absent among the 140 HSPs that we have for SBT. However, it would be unwise to over-interpret the modest number of triads that we have for school shark. Overall, 4% of the school shark sample is included in a kin-pair of some type,

so it is not particularly surprising to see that in some of the kin-pairs, the "second" animal happens to occur in some other pair.

Triads do not particularly lead to bias in CKMR, but large numbers of them would cause the CV to be under estimated, because pairwise comparisons become non-independent. Getting the CV exactly right is not a critical concern for now, and triads are not overwhelmingly common for school shark anyway; the variance issue will eventually be addressed in future research.

13.4.6 Summary of kin-finding

1. Genotyping and kin-finding processes worked well for school shark, and there is little ambiguity regarding the identification of the HSPs, FSPs, and POPs. We found 65 HSPs overall, which probably underestimates the true number by about 12% (and this is allowed for in subsequent modelling).
2. mtDNA data reinforces the HSP-finding conclusion, and reveals substantially more MHSPs than PHSPs, consistent with a larger number of adult males— which in turn is qualitatively consistent with males maturing four years earlier than females.
3. Birth intervals between cohorts (estimated from corrected ring counts) clearly separate three POPs from the 34 FSPs. Given random mating, the great majority, if not all, of the FSPs must really be same cohort pairs; however, most estimated gaps are 1–2 years or more, therefore ageing errors are clearly substantial (and this is shown by FAS's repeat age readings). The ratio of FSPs (same-cohort) to HSPs (mostly different cohorts) suggests a strong "lucky litter" effect.
4. There is a modest level of multiple paternity within litters.

The distribution of kin pairs shows no regionalization (Figure 13.5). The paucity of kin pairs that include an animal from south of Bass Strait is a function of the relatively small sample collection from the southern region (Table 13.1). Nevertheless, there are two HSPs that span Bass Strait.

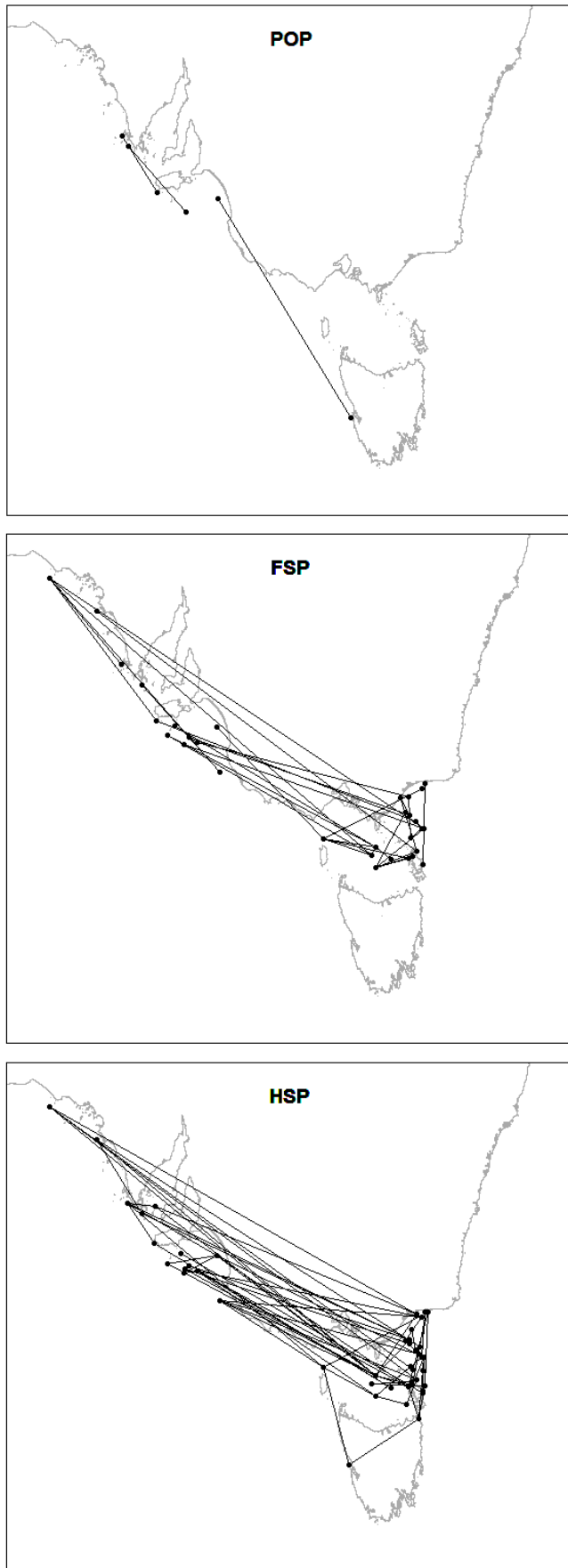


Figure 13.5. Approximate collection locations of the animals found to be parent offspring pairs (POP), full sibling pairs (FSP) or half sibling pairs (HSP).

13.5 Simple models

Using the “Simon and Peter” logic presented in the Introduction, it is possible to make a crude estimate of recent adult abundance directly from summaries of the close kin dataset. Some assumptions must be made: (1) that all adults of a given sex are equal in terms of fecundity; (2) animals that are born in the same year can be identified and, similarly, birth year can be accurately inferred from corrected “ring” counts; (3) mortality rates do not vary over the model time period; and (4) either there is no trend in abundance, or that the log trend is linear.

To minimise error resulting from the assumption that mortality rates do not vary during the model time period, and due to difficulty ageing animals above 11 years old, we used only those animals that had a ring count of 11 or fewer, and excluded those born before the year 2000.

To avoid bias due to the over-representation of same cohort siblings (i.e. the lucky litter effect) we excluded kin pairs whose nominal birth years were less than 4 years apart. Had ageing been perfectly accurate, we would only have excluded those born in the same year, but ageing error forced us to use a wider interval.

13.5.1 One line calculation

In this summarized subset, consisting only of "recent" cross-cohort comparisons:

- There are about 771,000 comparisons, of which 16 yielded MHSPs, and 10 yielded PHSPs.
- The mean difference in birth year within kin pairs is about 6.6 years.
- The mean year-of-birth of the *younger* animal is about 2011.

Since the mean birth year difference will be biased high because of errors in ageing, we might assume that the real mean difference is closer to 4 years than 6, and similarly that the mean birth-year is about 2009. Assuming an average adult mortality rate of $Z = 0.15$, and ignoring trends in abundance over the 2000–2016 period, the expected number of MHSPs is *roughly*:

$$771,000 * \frac{1}{N_{fem}} * e^{(-4*0.10)}$$

equating this to the observed total of 16 MHSPs:

$$16 = 771,000 * \frac{1}{\hat{N}_{fem}} * e^{(-4*0.10)}$$

therefore

$$\hat{N}_{fem} \approx 32,300$$

and similarly for males, we get:

$$\hat{N}_{mal} \approx 51,700$$

giving a total of around 84,000 "typical adults on average" across the 2000s.

13.5.2 GLM model

A more nuanced treatment, allowing for a linear trend in log abundance, can be obtained by fitting a simple GLM to the reduced dataset. Technical details are described in the Appendix. Like the one-line calculation above, the GLM assumed a constant mortality rate of $Z=0.10$ (Figure 13.6). Figure 13.3 also shows the results of the base case close kin model, which is discussed in the following section of the report.

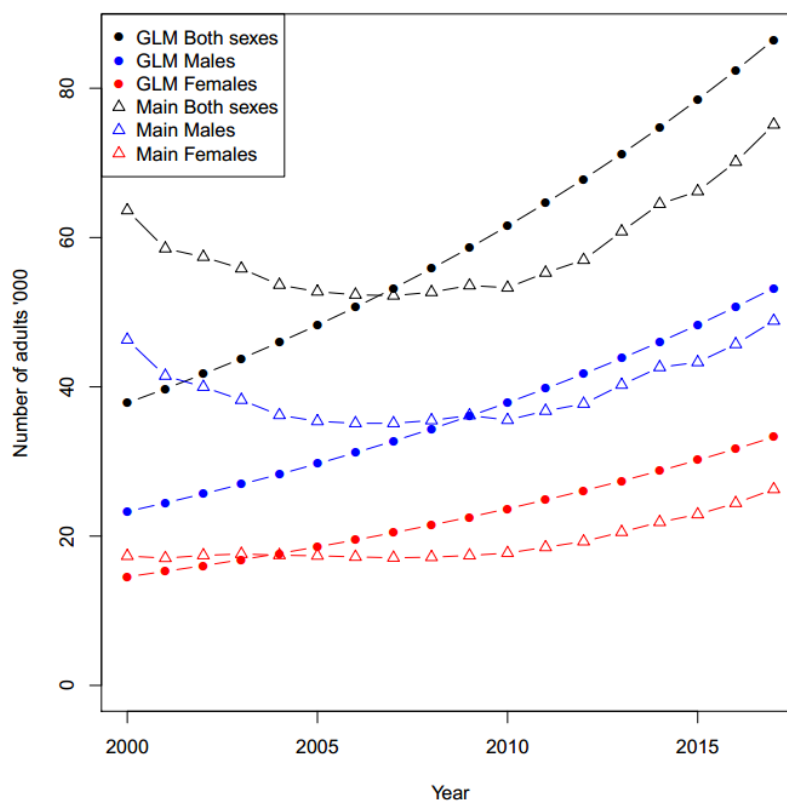


Figure 13.6. Estimated numbers of total (black), female (red) and male (blue) adult school shark from a simple GLM model (close circles) and the base case close kin model (open triangles).

13.5.3 Caveats for simple models

Note that these simple approaches assume a constant mortality rate (of $Z = 0.10$) that subsumes both natural and fishing mortality. This effectively assumes constant fishing pressure over the model period. Because the bulk of the catch is taken by gillnet gear, which largely does not catch adult fish (the part of the population on which the close kin data provides abundance information) that assumption is probably reasonable.

The simple model assumes that all adults are reproductively equal, which is not true for school shark, whose reproductive output varies from roughly 20 per litter to 30 per litter, so that a younger shark counts as only two thirds of an older adult from a close kin model perspective. This will lead to a slight under-estimation of abundance, but the variation in litter size, and therefore the bias in abundance, should not be huge (certainly not if compared with a teleost fish). The simple model also ignores ageing error, but the effect of that assumption (although complex) also seems unlikely to be huge.

13.6 Close kin model

We constructed a simple (relative to the stock assessment model for school shark) population dynamics model for school shark that makes none of the four assumptions made by the one line model, nor the three made by the GLM model. We (1) explicitly model the increase in fecundity with age for female sharks; (2) estimate extra parameters to account for same cohort comparisons i.e. the “litter effect”, and the proportion of full to half siblings within a litter, as well as modelling the distribution of true age as a function of ring count; (3) allow fishing mortality rate to vary during 2000 to 2017 as a function of observed catches (given ‘known’ gear selectivity), and (4) allow a trend in abundance that is driven by the observed catch data and not forced to be log-linear.

The close kin related data consisted of: the “ring” count (distinct from “age”, which is imputed within the model, accounting for ageing error and ring deposition rate), collection year, sex, and (as the response variable) the actual relationship (kin) type for each pair of animals. Only a subset of close kin samples were used, as explained previously. For all pairs of sampled animals (except within trip comparisons) the probability of the pair being a mother-offspring pair (MOP), full sibling pair (FSP), or maternal half sibling pair (MHSP) was calculated using the ideas of Expected Relative Reproductive Output, as explained in Bravington et al (2016a). The previous implementation of this work (Thomson et al 2018) also considered Grandparent-Grandchild Pairs (GGPs), but the scarcity of POPs in this study clearly suggests that GGPs are unlikely to be common, and our subsetting effectively eliminates the possibility altogether.

The close kin model is more realistic than the simple approaches, but is nevertheless simpler than the current stock assessment model. The close kin model considers only one region, one population, starts in 2000 (much later than the stock assessment model’s 1927 start), does not allow (or need to take account of) movement between regions (because there is only one region), and has an annual time step. The model is age structured, but it computes the length distribution of the population and (potentially) compares that with observed length frequencies. The main reason to introduce the complications and uncertainties around seasonal and annual movement, would be to improve the realism of the fit to length frequency data; if length frequency data are not required, and if there really is only one stock throughout the period considered, then the close kin model does not need those extra embellishments. However the regional length differences do (effectively) model availability, without a spatial component, the gear selectivity alone dictates which components of the population are model intensively fished.

13.6.1 Catches, selectivity, and biological parameters

The close kin population dynamics model incorporates catch data, biological parameters and gear selectivities that were used by the stock assessment model for school shark (as described by Punt & Walker 1998). The stock assessment model was most recently updated in 2009 (Thomson & Punt 2009) and was used in 2012 (Thomson 2012) to make forward projections under differing future catch scenarios, on which the current recovery strategy is based. That model used landed catches to the end of 2008 (the 2012 projection also used recorded catches to 2011) and assumed that discarding was negligible. Catches for 2010 to 2017 were compiled from logbooks (to be consistent with the original dataset). The stock assessment model assumed that discards were negligible. While this was probably true prior to 2009, subsequent reductions in the TAC for school shark are likely to have resulted in higher discard rates. A discard rate of 9% was calculated from ISMP data collected during 2011, with subsequent estimates of 12%, 14% and 15% during 2013-2015 respectively. No ISMP estimates are available after 2015 due to the replacement of onboard observers by camera systems, but work currently underway by ABARES suggest a similar discard rate during 2017 to that calculated from

2015 ISMP data (ABARES, in prep). Observed and imputed annual discard rates were used to convert the landed catches after 2009 into estimated total catches; further details are given in the Appendix.

Due to computational limitations in 2006, the stock assessment model does not currently incorporate full length frequency data but is instead conditioned on time series of the mean length and sex ratio of the catch calculated from the observed length frequencies. The close kin model includes (but gives very little weight to) length frequencies measured in port from mesh net catches. Attempting to condition the model on these data, using the gear selectivities used in the stock assessment model, resulted in expected length frequencies that included larger animals than those observed. This suggests that availability, and not just gear selectivity, is influencing the size of the animals captured. The spatial disaggregation of the stock assessment model might be able to account for that, but for the close kin model we set gear selectivity to zero for animals older than 1 years (roughly 150cm total length). Even so the model was unable to fit the length frequency data well and were given negligible weight. Further information on the compilation of length frequency data, and sample size for the data available, are given in the Appendix.

Sexes were combined when compiling the length frequencies and the catches were assumed to consist of 50% males and 50% females. This split is roughly that observed (see Appendix for details) but when the stock assessment model is update, length frequencies will be compiled separately for males and females. The addition of other gears, such as longline and trawl, will also be considered.

Recorded catches, by each of the five gears, from 2000 were incorporated into the close kin. The full catch time series is shown in the Appendix.

13.6.2 Close kin sample restriction

13.6.2.1 Fathers

In the current close kin model, we opted not to consider the paternal half sibling pairs or parent-offspring pairs, because the fecundity at age (or size) relationship for male school shark is unknown (whereas it is known for females). Since any source of systematic variation that is not allowed for in the close kin model, will lead to bias in a purely HSP-based close kin model, there is some risk in just assuming that “all males are equal” (and we do know that all female school shark are *not* equal). Having said that, for the other (non-commercial) shark species where CSIRO has fitted close kin models, we did assume that there was little variability between males in reproductive output, and the results were not surprising; but those species all have smaller litter sizes and are taxonomically quite different to school shark. Whether it is worth the perhaps rather small risk of assuming equal male reproductive output in school shark, in order to gain an appreciable improvement in “sample size” (effectively, the number of usable kin pairs), is something to be considered in future.

Male fecundity is of no relevance to stock assessment models and has therefore been little studied in fisheries science. If we had observed enough father-offspring pairs (FOPs), then we might have been able to directly estimate male fecundity as a function of body size (as has been done for SBT). However, as we found only one FOP, that will not work; we instead excluded those kin pairs whose use requires a male fecundity relationship. We assumed that sex ratio is 50:50 at birth, and that natural mortality rate was the same for male and female sharks. Fishing mortality was imposed by the observed catches, under the assumption that the catch was made up equally of males and females (see Appendix for justification), and we used separate growth curves for males and females to relate catches, via selectivity, to catch at age.

13.6.2.2 Trips

School shark show a tendency to school with individuals of the same size and sex, and Thomson et al (2018) found a tendency for relatives to be caught together. Mark recapture studies require that (re)captures must be independent of one another, so for a close kin study the capture of an animal must be independent of captures of its close relatives. For this reason, we assigned a trip ID to every fishing trip that was sampled for this study and disallowed within-trip comparisons of samples (i.e. no looking for close relatives amongst those animals that were caught together).

13.6.2.3 Rings ≤ 11

To keep the model simple (and the number of estimated parameters low) we allowed a single, time invariant, parameter that accounted both for mortality during the first year of life, and for the pupping interval (which is known to be 2 or 3 years; Walker 2001, see Appendix). In effect, this assumes that density dependence did not change over the model period (so that production of one-year-olds was directly proportional in any year to total female reproductive output, i.e. a fecundity-weighted sum of numbers at age over female adults.) - this is one reason we kept the model period short.

Our early attempts to construct a close kin model for school shark encountered great difficulty reconciling the close kin data (which support adult abundance of the order of 80,000 adults during 2000-2017) with the historical catches. Catches were very high during the 1980s and require a very large starting population to support them, which appears not to be compatible with the basic (and hard to argue with) signal from recent close kin data *if* per capita pup production has remained constant. Our close kin model, at least in its current form, assumes that per capita pup production is constant throughout the modelled period, which may not be true because biomass dropped greatly during the 1980s, so that density dependent changes in production may have been occurring at that time. Given the age of our samples, we also have rather little direct data to inform abundance pre-2000. It is therefore unwise to extend the current close kin model before the year 2000. We achieved this by excluding all samples for animals that were old enough to have been born before that year. The ageing error, and in particular the slow deposition of vertebral rings after age 11, mean that even sampled animals with as few as 15 rings have a non-negligible probability of having been born well before 2000. Therefore we restricted the sample to only those that were younger, and for which aging was unbiased: 11 rings or less. This allowed us to place a “plus group” at age 20, since there is little chance that an 11-ringer could be older than 20, and all the age-related fecundity changes in females are thought to have stabilized by age 20. We assumed that natural mortality was constant across age from 1+ up right through the plus-group.

13.6.3 Kin probabilities and likelihood

When considering whether a pair of animals (where the older one is female) might be a MOP, we must first work out in which year the younger animal was born, and whether the potential mother was mature in that year. If she was, then the probability that she is the mother of the younger animal is roughly $1/N_m$ where N_m is the number of mature females present in the population in the year that the younger animal was born. To be more exact, because female fecundity varies with age, the probability depends on her Expected Relative Reproductive Output compared to the total across adult females; i.e., her fecundity given her age, divided by the total fecundity across all living females that year.

Because the actual birth year for any school shark in our study is clouded by ageing error, we integrated over all possible birth years, weighted by the probability that each was the actual birth year for that animal (given the observed ring count and the degree of ageing error).

To calculate the probability that a pair of animals might be a maternal half sibling, the mother of the first animal must also be the mother of the second. Therefore she must have been mature when the older animal was born (y_1), and must have survived until the second animal was born (y_2). The probability that the mother of the first is also mother of the second (if all adult females were reproductively equal) would be the inverse of the number of mature females present in y_1 , multiplied by the survival rate for females of this age between y_1 and y_2 . However, since female fecundity increases with age in school sharks, this formula must be modified to account for the likely reproductive output of a female of given age in year y_1 and year y_2 . This is explained, albeit briefly, in section 3 of Bravington (2016a); further details will be added to the Appendix of future versions of this report. As with POPs, it is also necessary to integrate over all probable birth years, given ageing error and age uncertainty.

Because of ageing error, we cannot simply exclude same cohort comparisons based on “most likely” birth year. Instead, for animals born in the same year (some HSPs and, by assumption, all FSPs) we had to allow extra parameters to account for: (1) “litter effect”, the inflated number of surviving siblings pairs in certain litters where favourable conditions occurred, and (2) the proportion of animals within a litter that share a father. Female school sharks can mate with multiple males to produce a single litter consisting of both full and maternal half siblings (Hernandez et al 2014). This “multi-mate” parameter scales the number of full to half siblings observed. The “litter effect” parameter scales the numbers of same- versus different-cohort siblings observed.

For a pair of half siblings, the mitochondrial DNA (mtDNA) tells us whether they share a mother (identical mtDNA) or a father (different mtDNA). Paternal HSPs could share mtDNA by chance, but this study found 144 different haplotypes with only a 3% chance of sharing mtDNA by chance. As already stated, paternal HSPs were not considered further because the fecundity at age of male school sharks is unknown and could not be estimated within the model.

The log-likelihood for the close kin component of the model is straightforward in principle. Provided that sampling is fairly “sparse” compared to the population size (as will be the case when population size is fairly large; see Bravington et al (2016a), section 4), then it is statistically reasonable to treat all pairwise comparisons as approximately independent, with each comparison constituting a Bernoulli trial (i.e. a yes/no outcome) whose probability is determined by the demographic parameters (known and to be estimated). The log-likelihood follows directly. The complication of triplets and known FSPs does invalidate the independence assumption, strictly speaking, but (as explained elsewhere in that paper) does not generally cause bias.

The kin probability calculations are lengthy, particularly when age is uncertain, but the underlying biological principles are quite clear and transparent. Every animal has one mother and one father; the chance of the mother being one particular female (in particular, the parent of another specific animal) is that female’s expected reproductive output divided by the total reproductive output from all females at that time and place.

13.6.4 *Estimated parameters*

The close kin model had eight estimable parameters (Table 13.2).

Table 13.2. Description of the eight estimable parameters used in the close kin population dynamics model.

| Symbol | Description |
|--------------------|--|
| M | Natural mortality rate for all animals aged ≥ 1 year |
| N_0 | Number of animals in the first year modelled |
| δ | Pupping interval multiplied by mortality during the first year (which incorporates density dependence) |
| γ, F_0, F_1 | Parameters that govern age distribution during the first year |
| v_1 | Litter effect, allowing for 'lucky litters' |
| v_2 | Proportion of the litter that are likely to have different fathers (1 indicates that every litter has a just one father) |

13.6.5 Results

The sample (both in terms of kin pairs found and comparisons allowed) was restricted to animals aged 11 and over. This reduced the sample from 2,438 to 1,627 animals and removed the two MOPs as well as 9 of the 38 MHSPs. The model starts in the year 2000, with initial age composition that year being determined indirectly by three estimated parameters. The CV for ageing error is assumed to be 0.08 to age 11 and after that is doubled (0.16) to allow for uncertainty in ring deposition rate after the age of 11. Ring deposition rate is assumed to be 1 p.a. up to age 11 and 0.36 thereafter. Selectivity is set to zero after age 15. Length frequency data was given the very low effective sample size of 1, but a sensitivity test was done by raising this to 5. Another sensitivity allowed the CV on older animals to be equal to that of younger animals (0.08) which is equivalent to assuming that ring deposition rate is a steady 0.36 with no variation. Another sensitivity allowed ring deposition to be only 0.25 per year.

Table 13.3. Estimated parameters in the close kin, negative log likelihood (-lnL) and estimated numbers of POPs (nPOP), FSPs (nFSP) and maternal HSPs (nMHSP). Observed numbers of kin pairs are shown in parentheses in the first column. The symbols in the first column are described in Table 13.2.

| Quantity | Sensitivity | | | |
|----------------|-------------|-----------|------------|----------|
| | Base case | LF effn=5 | CVold=0.08 | Rpy=0.25 |
| M | 0.099 | 0.069 | 0.110 | 0.087 |
| N_0 ('000) | 111 | 85 | 212 | 92 |
| δ | 0.20 | 0.33 | 0.20 | 0.19 |
| v_1 (litter) | 5.0 | 2.8 | 4.3 | 5.3 |
| v_2 (mate) | 2.8% | 0.9% | <0.01% | 3.4% |
| -lnL | 816.8 | 844.7 | 832.4 | 815.8 |
| nPOP (0) | 1.0 | 1.1 | 0.7 | 1.1 |
| nFSP (33) | 33.0 | 33.0 | 33.0 | 33.0 |
| nMHSP (29) | 25.9 | 26.5 | 28.4 | 25.3 |

Unsurprisingly, the model was able to achieve close matches between the observed and expected numbers of kin pairs. Estimates of natural mortality are not dissimilar to the value of 0.1 which has been assumed (but not estimated) by the stock assessment model. The parameter δ seems rather high, given that it is made up of pupping interval (0.25 or 0.33) multiplied by survival during the first year of life (likely to be lower than that of older animals i.e. < 0.1). The litter effect parameter (v_1) is high, which seems reasonable given the large numbers of sibling pairs born close together, and the large number of FSPs observed. The proportion of litter-mates that have different fathers (v_2) is very low, which is credible given the number of FSPs observed.

The imposition of zero selectivity after the age of 15 effectively imposes an availability function on the model. This did not restrict the expected captures very much because relatively few animals of that age are alive in the population (Figure 13.7); it did, however, improve the general behaviour of the model, which otherwise tried hard to increase the number of older animals in order to match the upper tail of the length frequency. Alternative likelihood formulae that might be considered in future.

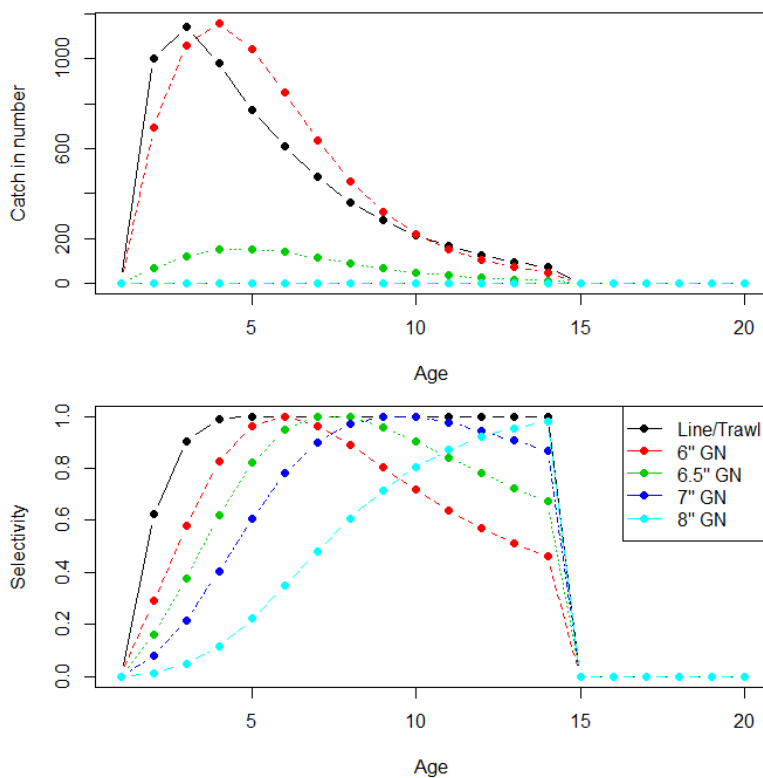


Figure 13.7. Number of animals captured at age in 2016, by gear type (upper plot) and selectivity curves for the five gear types (lower plot). Results are shown for the base case close kin model.

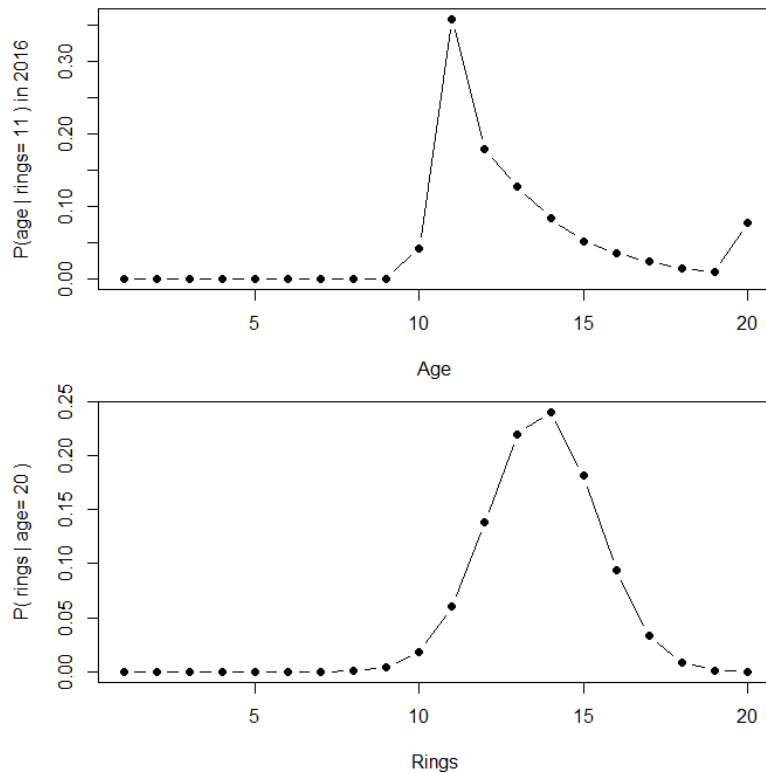


Figure 13.8. The probability distribution for an animal that has 11 vertebral rings counted (upper plot) and the probability distribution for the number of rings that an animal aged 20 (the plus group age) will be observed to have in 2016.

Even an upper limit of 11 vertebral rings resulted in a more than negligible probability that such an animal is aged 20 or more (Figure 13.11). This indicates that animals with more rings could not have been used without increasing the plus group age and therefore applying the model assumptions (chiefly that density dependence is unchanged) for a longer period of time.

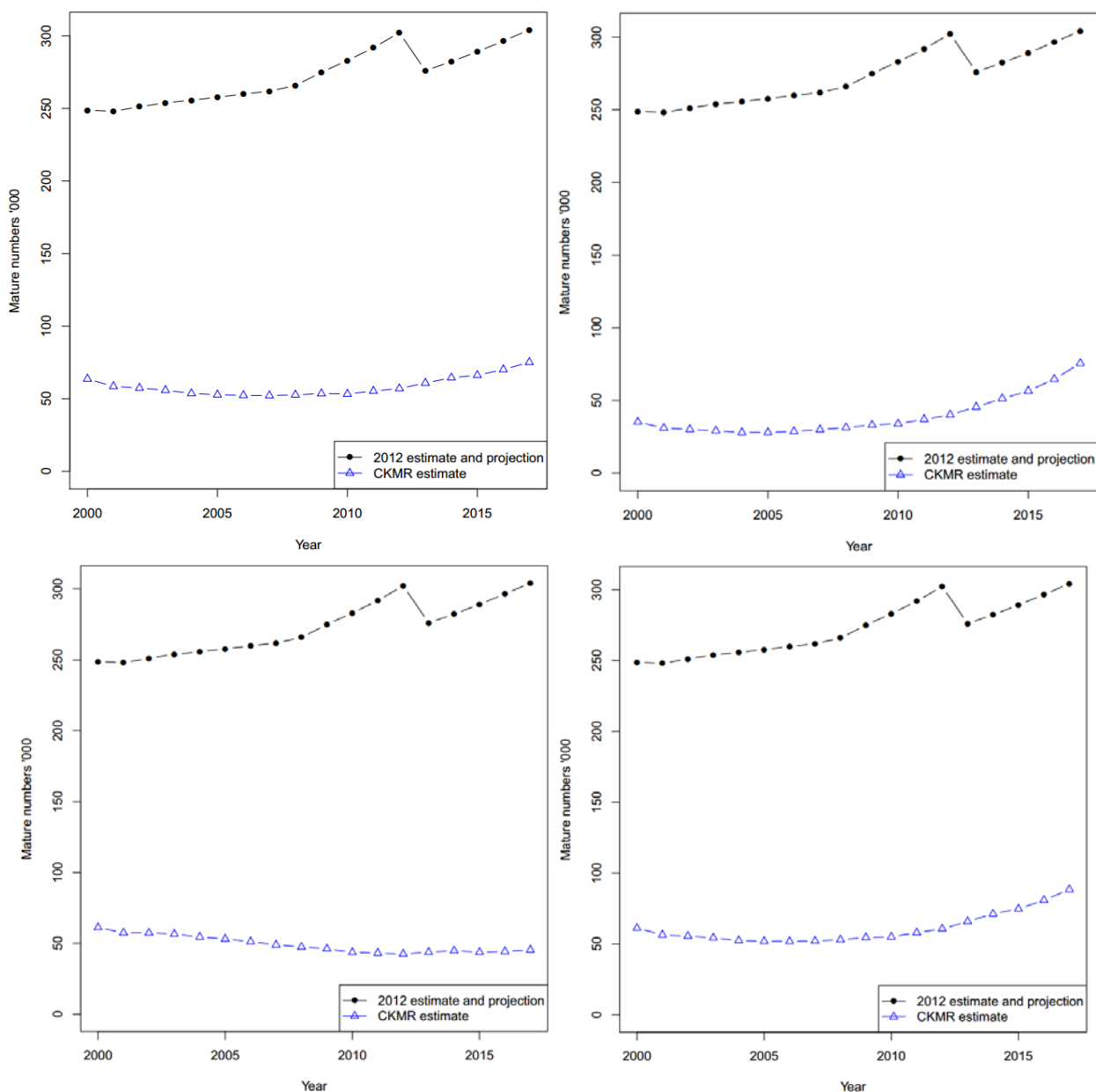


Figure 13.9. Numbers of mature school shark (females aged 11 and over, and males aged 7 and over) for the full stock assessment model's base case that assumed catches of 225t after 2011 (2012 estimate and projection; black line) and close kin model estimate (blue line). Results are shown for the base case scenario (upper left), $effn=5$ (upper right); $CV=0.08$ for all ages (lower left), and ring deposition rate after age 11 = 0.25 (lower right).

Estimated numbers of adult school shark are substantially lower than those for the most recent work using the full assessment model (Thomson 2012), which used data to 2008, catches to 2011, and assumed a constant 225t catch from 2012 onwards (Figure 13.9). Three of the four scenarios do show recovery since 2000, at a rate that is similar to (but no higher) than that of the full model projection.

Estimated CVs for the number of mature females in the population are lowest for 2004, which is roughly the average birth year for offspring in the close kin sample. The close kin data are unable to provide much direct information on the most recent years, where the length frequency data provide the most information. However, the length frequency data did not fit well (Figure 13.10) and were

therefore not allowed much weight. Consequently, the CV for recent years is high (Table 13.4). The CV for 1+ abundance is lower than that for mature female abundance and is lowest in 2008.

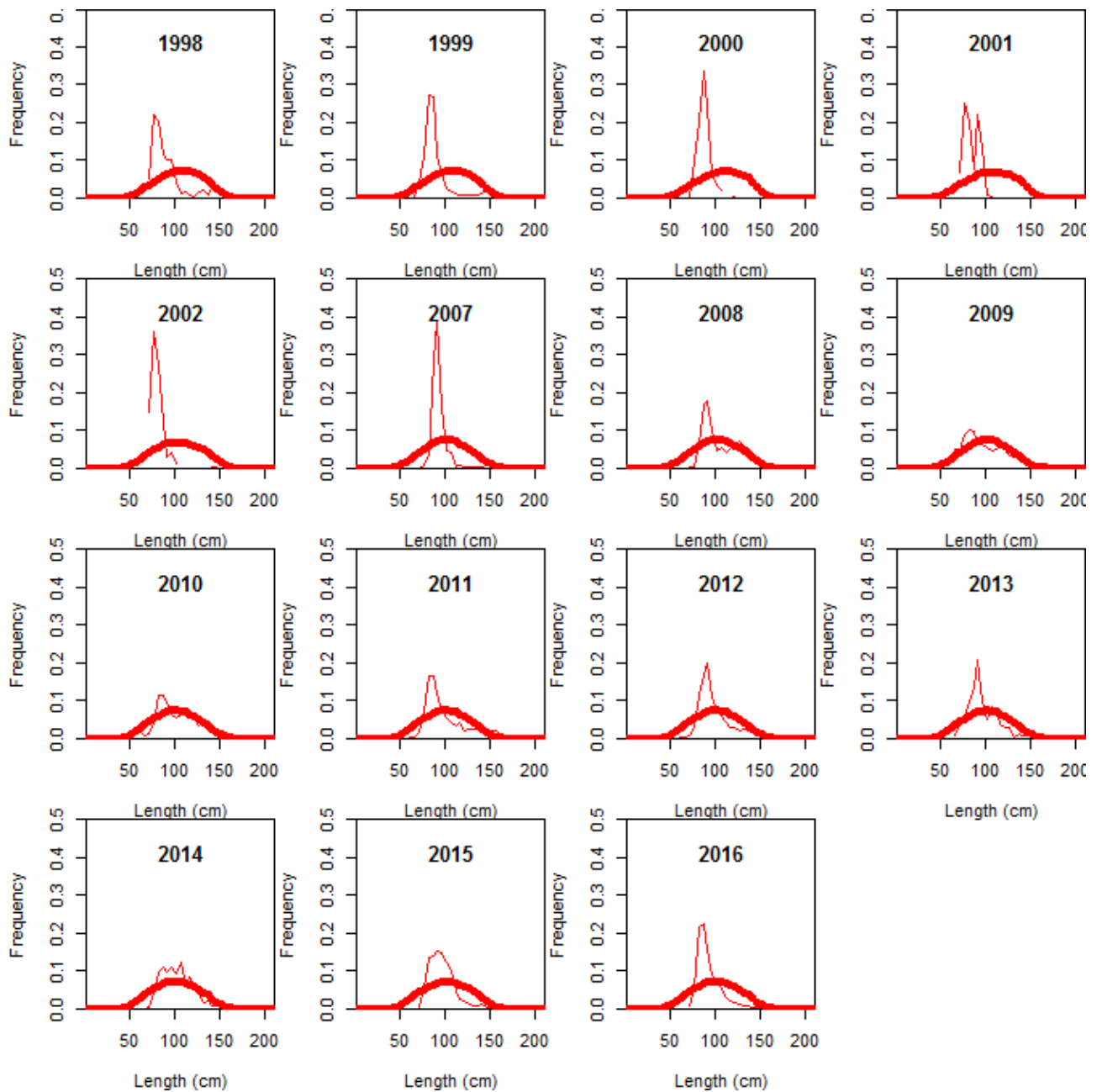


Figure 13.10. Observed (thin line) and estimated (thick line) length frequencies for port measured gillnet catches of school shark. The 1998 and 1999 data were not included in the likelihood of the close kin model, but are shown for interest. The base case results are shown.

Table 13.4 Point estimates of the numbers of school shark ('000s) aged 1 and over $N(1+)$, and of mature females $N_f(\text{mat})$, with associated CVs.

| Year | $N(1+)$ | CV | $N_f(\text{mat})$ | CV |
|------|---------|------|-------------------|------|
| 2000 | 367 | 0.29 | 14 | 0.38 |
| 2001 | 360 | 0.29 | 14 | 0.34 |
| 2002 | 369 | 0.28 | 14 | 0.31 |
| 2003 | 374 | 0.28 | 14 | 0.29 |
| 2004 | 379 | 0.27 | 15 | 0.28 |
| 2005 | 384 | 0.26 | 15 | 0.29 |
| 2006 | 388 | 0.25 | 15 | 0.30 |
| 2007 | 391 | 0.24 | 15 | 0.32 |
| 2008 | 395 | 0.24 | 15 | 0.34 |
| 2009 | 400 | 0.25 | 15 | 0.35 |
| 2010 | 396 | 0.27 | 15 | 0.36 |
| 2011 | 404 | 0.30 | 15 | 0.37 |
| 2012 | 413 | 0.34 | 16 | 0.37 |
| 2013 | 432 | 0.37 | 17 | 0.39 |
| 2014 | 450 | 0.42 | 18 | 0.44 |
| 2015 | 461 | 0.49 | 19 | 0.50 |
| 2016 | 487 | 0.55 | 20 | 0.57 |
| 2017 | 520 | 0.62 | 21 | 0.64 |

13.7 Discussion

For several other shark species, it has been possible to construct close kin models based on juvenile samples and HSPs, and with no extra quantitative biological data (e.g. Hillary et al 2018). That was possible because for those particular species there is biological evidence suggesting little variation in average annual fecundity between female adults, either within one individual's life or between individuals of the same sex. It was reasonable to assume, therefore, that all adults are reproductively equal. The same assumption was made for males, although there was never any direct evidence one way or the other for males; in all cases, the estimated abundance of adult males (basically, estimated independently from that of females) seemed plausible relative to female adult abundance estimates. It may be relevant to note that, at least in white sharks and grey nurse sharks, males mature at a younger age than females and do not grow as large, suggesting perhaps that male size is not an important driver for fecundity (otherwise there would presumably be selection pressure for larger male size, since the species are clearly physiological capable of it).

For Southern Bluefin Tuna, on the other hand, fecundity changes hugely during adulthood, so that the same "total reproductive output" (a key notion in calculating the probability of finding a close kin pair) could in principle come from many small adults or fewer big ones. However, there were ample POPs from which to estimate "true" fecundity in relation to body size (based, intuitively, on the average size of identified parents compared to the average size of sampled adults), and sufficient age composition data from the fishery to track changes in average adult size (see Bravington et al, 2016b for details).

School shark does not fit either of these paradigms. There is substantial (female) growth post maturity, and litter size is correlated with body size, so all adults are clearly *not* equal reproductively. On the other hand, we do not have enough adult school shark samples to find enough parent offspring pairs

(POPs) to infer the fecundity-size relationship directly inside the close kin model. Consequently, we took the female maturity and fecundity at age relationships from the stock assessment model as known inputs, along with the selectivity at age relationship for the five gear types used in the model. We do not yet have nearly enough POPs to estimate the male fecundity-size relationship directly, neither is it known from the literature. Instead, for the moment we took the rather conservative path of not considering PHSPs or Father-Offspring Pairs at all - which eliminates one risk but does reduce our ability to estimate abundance and trend. If close kin studies for school shark are continued, there may be sufficient POPs captured in future to allow direct estimation of fecundity-size relationships for both sexes. Meanwhile, given that school shark, like white sharks, also have lower maturation age and lower asymptotic size for males compared to females, it may be reasonable to simply assume that male adults do not vary much, reproductively speaking. Even if there is substantial reproductive variability among adult males, so that the absolute numbers of PHSPs do not inform directly about abundance level, the PHSP data may still be useful in estimate abundance trends.

School shark demonstrate several biological features that have not been encountered in our other CKMR analyses: high frequency of full sibling pairs (FSPs), ageing error, and ageing bias, and a long and complicated history of changing (but generally heavy) exploitation rates. These have presented some challenges in developing a suitable close kin model.

The work presented here differs from that presented to sharkRAG in February 2018 (Thomson et al 2018) in that

- the close to 3,000 sharks have been sampled (the previous work used 1,500 samples);
- several improvements have been made to the close kin population dynamics model, including but not limited to
 - relating pup production to number of females in the population (the previous model allowed number of pups to be a free parameter, estimated in a number of year ‘blocks’)
 - mitochondrial DNA results have been improved by more sensitive measurement, going from a relatively uninformative 7 haplotypes, to 122 This gives near certainty about whether a half sibling pair share a mother or a father;
 - ring counts have been made for all animals sampled and ageing error has been quantified;
- samples that were caught together in the same trip are not compared to determine whether they are close kin, thus preventing artificial inflation of the number of kin pairs found (and thereby under-estimation of the abundance of the stock) resulting from a tendency for siblings to school together.

Thomson et al (2018) speculated that the number of full siblings and their apparent separation in birth years indicated that sperm storage might be occurring, as is known to happen in many shark species. Now that ageing error has been quantified, it seems possible to attribute most, if not all, of the apparent separation in birth years to ageing error. It is nevertheless possible that sperm storage does occur – birth years were separated by as many as 5 years. Hernandez et al (2014) examined five pregnant school shark finding that three of those carried pups sired by a single male, the remaining two having multiple fathers. Hernandez’s work suggests a higher multiple paternity rate than the 2.8% estimated by the model. The close kin model presented here ignores fathers and is therefore unaffected by sperm storage. If paternal HSPs and FOPs are used in future, however, consideration will need to be given to this question.

Thomson et al (2018) indicated that better use might be made of the carcass length information provided with the close kin sample, after vessel by vessel examination of length frequencies to better attribute the lengths provided to either partial length or dressed length measurements. Vessel by vessel examination was performed (not shown) and some attributes were changed. However, the spatial segregation of school shark made it difficult to determine whether overall shifts in lengths relative to average were due to the size of the sharks caught, or to the type of length measurement that was taken. Bimodality in some of the plots examined suggested that more than one length measurement might have been taken. Also, it seems likely that deck crew differ in where they cut when they dress carcasses (Kyri Toumazos pers comm), so that the 'dressed length' measurement may not always be consistent. After careful examination of the length data we concluded that it could not be used. That is a pity, since age at length composition data (i.e. tables of observed ring counts, grouped into narrow bands of body length) constitute a largely fishery-independent data source (i.e. unaffected by selectivity or other variable aspects of fisher behaviour) that can carry useful information on stock trends - arguably the only such data, apart from close kin. If sampling continues in future, it would be worth making every effort to ensure consistent (or at least reliably inter-calibrated) length measurements.

The close kin data indicate that adult abundance of school shark is much lower than that suggest by the stock assessment model and its 225t p.a. catch projection. This is supported by the findings of the simple model approaches which found around 80,000 "typical adults on average" across the 2000s (one line approach) and roughly 40,000 to 80,000 adults (GLM model). While these are certainly crude, and really only suitable as a check that the more elaborate close kin model has been set up correctly, it is quite clear that estimates of adult abundance in the 200,000s (stock assessment model) are incompatible with the observed close kin data.

While (1) the close kin model's currently unsophisticated treatment of density dependence may be to blame for that model's inability to reconcile high catches in the past with current low biomass, and (2) the incorporation of the close kin data into the full assessment might provide a unifying theory for these two data sources, it is equally possible that (3) even a more complicated model might struggle too. Another possible explanation for such a discrepancy is that there was, in the past, at least one other school shark stock from which a large proportion of the past high catches was taken, and that the recent close kin sample is representative of, only, the remaining stock(s) of school shark. Braccini et al (2009) failed to find school shark pups in bays and inlets of Victoria in which they had previously been common, and Hernandez (per comm) also looked and failed to find Victorian school shark pups (in the summer of late 2009). It is possible that a Victorian school shark stock has been seriously depleted, leaving only South Australian and Tasmanian stocks represented in the close kin study. In other words, the HSPs tell us about the adults that gave rise to the (mostly still immature) school shark that are currently being caught and sampled, and which were mostly pupped during the 2000s. It is possible that there were other "groups" of adults, e.g. from other subpopulations, which in the past contributed to numbers in a historical stock assessment, but which are not now contributing to the stock that is now being fished.

The next update of the school shark stock assessment should include:

- calculation of close kin probabilities (for POPs, FSPs, MHSPs and possibly FOPs and PHSPs), and incorporation into the log-likelihood;
- time series of observed length frequencies for at least one gear type (instead of mean lengths and sex ratios);
- separate (if possible) line catches made in deep and shallow waters, as well as trawl catches, because their length frequencies have been shown to differ greatly;

- improved correspondence between observed and expected length frequencies (this might require the addition of an availability function); and
- available historical age composition data.

Historically, school shark management has been based on “pup production” as a proxy for spawning biomass. In this report we did not compare pup production estimates from the stock assessment model with those from the close kin model, because the treatment of fecundity, pupping interval, and pregnancy rates were too different in the two models to allow easy comparison. An updated stock assessment will provide a pup production time series, and current depletion estimate, on which future management can be based.

The CVs calculated in this study are relatively high (24% or greater for 1+ abundance and 28% or greater for mature female abundance). The target in the design of the close kin sampling was 15%, but of course the realized CV is always contingent on the true stock size and biology, rather than on the assessment based estimates from which designs have to be made. The CVs that were estimated are, in fact, underestimates because all kin pairs were assumed to be independent. We did not account for the kin pair “families”, and this remains an open question for long-term future work on the underlying statistical theory; however, it does not seem mission-critical in the short term. (Note that it applies even more to the other shark species where CSIRO has used close kin, where population sizes are much lower than for school shark.)

The high CVs are largely the result of the restrictions imposed on the sample, which reduced the overall number of kin pairs used, but is also a function of ageing error. The restrictions were necessary in order to reduce the time period modelled, so as not to violate the assumption of constant density dependence, nor to force the model to try to describe far-flung history on which it has little direct evidence (i.e. when few of the samples were even born) . When the close kin data is incorporated into a full assessment model, it should not be necessary to impose such strict rules, so that all (or at least a greater number of) kin pairs can be used. Paternal kin pairs could also be incorporated, either by assuming that all males are reproductively equal after the age at maturity, in which case they would inform both the trend and abundance estimates, or simply to estimate the trend (in which case an additional parameter would be estimated that compensates for variable fecundity of males).

13.8 Future work

Catch data used by the stock assessment model has been gathered from logbook data, not adjusted by verified landings records (CDRs). Prior to 2009 (the last time anyone checked), the logbook totals and CDR records were been remarkably similar, indicating that adjustment is not necessary. The correspondence between logbook and CDR catches in more recent years should be checked to see whether this correspondence continues to hold. State catches should also be considered because these have become more important since the reduction in the size of the commercial catch.

A different formulae for the likelihood component for length frequency should be explored so that the tails of the distribution have less influence on the overall estimation of fit.

More detail on the equations used to calculate close kin probabilities should be added to future versions of this report.

Walker et al (2001) showed that pupping frequency for females is at least two years, possibly three. The true interval would have been clear in the data from the maternal half siblings, had ageing error not obscured that signal. Examination of a larger number of pups, aged 0, 1 and possibly 2 years old (where growth is sufficiently rapid for age to be clearly apparent from length) could provide clear information on the pupping frequency. Furthermore, the presence (or absence) of cross cohort full siblings amongst these pups would help in understanding whether sperm storage is occurring, and if it is, to what degree it occurs.

The collection of consistently measured carcass length data with future school shark samples would allow modelling of fecundity as a function of size rather than age. Fecundity is more likely to be related to size than to age, so this would give a more accurate model. Such a model would be limited by the absence of reliable length information for the existing 3,000 samples, and therefore it might not be possible until sufficient future sampling had been done to allow the newer sample to overwhelm the older.

Collection of pup sample from South Australia (and from Victoria, if any can be found) would allow examination of both somatic and mitochondrial DNA for any signals that might indicate stock separation.

13.9 Acknowledgements

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13.11 Appendix

13.11.1 Tissue/Vertebral samples

Samples (consisting of a section of vertebrae, used for ageing, and a chunk of tissue for DNA extraction) were collected by fishers (Leigh Castle, Andy Joy, Kyriakos Toumazos), fish processors (The Fish Factory: Philious Toumazos; Pitliangas Foods: Nick and Chris Pitliangas). In addition, samples were provided from a collection held by the Fish Ageing Service (FAS) that consisted of samples collected by AFMA's Observer program, and historical collections sourced from Terry Walker (MAFFRI). Samples were transported to CSIRO, mainly by refrigerated truck. The contracted total for sample collection was 2,000 sharks, but a sample size of 3,000 was always considered best and savings from a reduction in genetic sequencing costs facilitated the collection and processing of 3,000 sharks (Table 13.1).

It was expected that samples would mainly come from the gillnet fishery, which catches predominantly smaller, younger animals, so that no parent offspring pairs (POPs) were anticipated, and the model was expected to rely instead on half sibling pairs only (HSPs). Usually (for a species where fecundity changes substantially through adulthood), both POPs and HSPs are needed to calculate all the parameters needed for a population dynamics model, but in the case of school shark, gear selectivity is known, as are the female maturity at length ogive and the number of pups per female as a function of length. Knowledge of these functions allows close kin mark recapture using HSPs only (or, at least, maternal HSPs for estimating parameters related to female adults). As it happens, a larger than expected proportion of the sample was caught using line gear, so that some POPs were found.

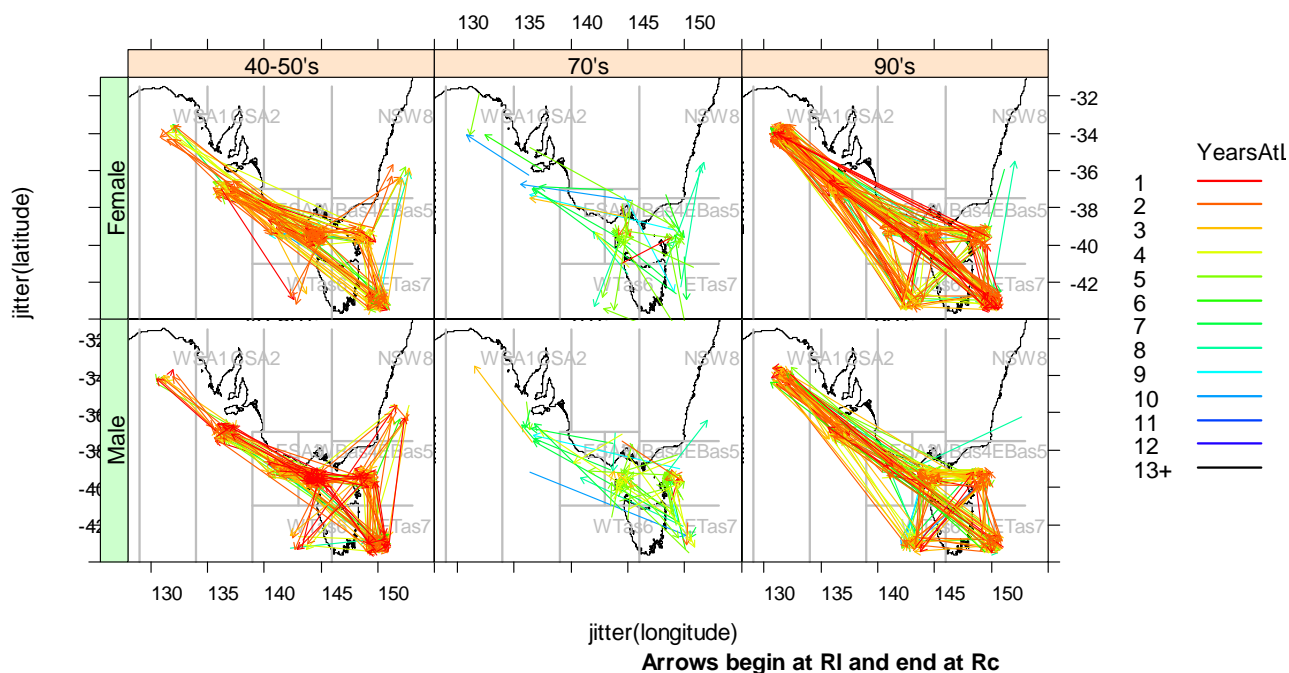


Figure 13.11. Movements of school shark from conventional tag and recapture data, organised by decade of release, and by gender of shark. Arrow colour indicates number of years at large.

School shark have been seen to move from every one of AFMA's shark zones to every other zone (Figure 13.11). This movement does not prove interbreeding – reproductively isolated stocks could exist – but it does suggest that school shark are highly mobile and therefore that sampling location is

not crucially important. Nevertheless, the target sample size was broken down in proportion to fishing activity, between three broad locations (South Australia, Bass Strait, and Tasmania) to guard against any unknown sub-structuring of the population. The target for Tasmania was inflated relative to actual landings to ensure a useable sample size from that state. Collections have aligned adequately with the targets (Table 13.1)

13.11.2 Ageing and birth year

School shark lay down hypermineralized growth-increment bands (or “rings”) in their vertebrae during winter (Walker et al 2001). Vertebral samples were sectioned and bands counted at Fish Ageing Services (FAS). Additional work was done for some of the animals found to be in close kin pairs, to increase the accuracy of their band counts.

Examination of sectioned vertebrae suggest that the most recent hypermineralised band is visible by November of the year in which it forms (Simon Roberston pers comm). School shark pupping takes place in summer (Nov to Feb in Pittwater, Tasmania; Stevens & West 1997) so a birth day of 1 January was assumed for all animals. Those collected before November, for which b bands were counted, were assumed to have been born b years prior to the collection year. However, those collected between Nov and Dec (inclusive) were assumed to have been born $(b-1)$ years before the collection year because they have lived through one additional winter and have a detectable band associated with that season.

Three investigations of the accuracy of school shark aging by counting bands have been conducted. Two of these used bomb radiocarbon dating, which is accurate enough to show that bands typically underestimate true age (Kalish 2002, Fenton 2001) but are not as accurate as the data obtained by Walker et al (2001) who used oxytetracycline injections to create a reference band in the vertebra of sharks that were then tagged and returned to the water. Once recaptured, Walker was able to count the number of bands laid down after the chemical layer and divide that by the number of winters at liberty (i.e. between marking and recapture) to provide an accurate estimate of the number of bands laid down per year. Unfortunately, their sample size was only 18 animals, of which only 5 were in the “large” poorly age-able category (over 11 years). Walker et al (2001) also drew on the findings of Moulton et al (1992), who compared von Bertalanffy growth curves fitted to tag-recapture data with those fitted to band counts, to conclude that ageing was precise for school shark up to age 11, but underestimated thereafter. Moulton et al (1992) found that the fitted growth curves were similar for sharks under the age of 11 but deviated for older animals. Band counts for those over 11 yo are biased downwards presumably due to “difficulties resolving growth-increment bands at the vertebra margin of large sharks” (Walker et al 2001) due to slowed growth causing bands to be more tightly compressed. Walker et al (2001) found that the mean number of bands per winter for the 5 “large” sharks was 0.36. Interestingly, that for the 7 “small” sharks (<100cm) was 1.22 – suggesting overestimation.

13.11.3 Mitochondrial DNA

Mitochondrial DNA (mtDNA) is distinct from the nuclear DNA used to find kin pairs and needs to be measured using different techniques. The reason for looking at mtDNA is that it can reveal whether the shared-parent of two half-siblings is the mother or the father; this is important for half-sibling-based CKMR because the two sexes can have different reproductive dynamics.

Individuals that were found to belong to kin pairs underwent sequencing of their mtDNA. The ‘control’ region of the mtDNA strand was extracted and sent to an independent contractor for sequencing. Earlier work on school shark (Thomson et al 2018) showed that relatively few haplotypes were found when only the control region was used: just seven from the sample of 1,500 sharks examined. Work

by Hernandez et al (2015) had suggested greater diversity in the control region, however, several of his haplotypes were represented by only 1 or 2 animals suggesting that those might have been sequencing errors rather than rare haplotypes. Roughly half of the sample shared a single, very common, haplotype, rendering the control region mtDNA very uninformative.

For this reason the mtDNA was re-sequenced using the full genome, which gave a much more informative 122 haplotypes of which the most common was found in only 5% of the sharks examined. To save expense, the mitochondrial DNA was sequenced for only those sharks found to be closely related. Haplotype frequency was calculated by excluding one animal (selected randomly) from each pair, because close relatives are more likely to share haplotypes than unrelated individuals.

13.11.4 Growth curve, weight, fecundity and gear selectivity

Sex specific von Bertalanffy growth curves, calculated using tag-recapture data, and therefore free of inaccuracy due to band counts, were used in the stock assessment model for school shark (Punt & Walker 1998).

Gear selectivity curves (dome-shaped for gillnets and knife-edged for a combined trawl and line fleet) were fixed (not estimated) in the stock assessment.

Female fecundity (the product of the proportion of females mature-at-age, and the mean number of pups produced-at-age, and a 3 year pupping interval) are also fixed in the stock assessment model. For the close kin model we used the linear relationship between number of pups and length of female given by Walker (2005) which rises from roughly 20 pups per female for younger animals to 30 for the largest animals. No pupping interval was assumed, neither was juvenile survival rate, instead the model was allowed to estimate a parameter that subsumes pupping interval and survival during the first year of life as well as any inaccuracy that might exist in the fixed fecundity relationship.

The stock assessment model assumes a first size at maturity for females of 140cm (11 years of age). However, the female maturity ogive developed by Walker et al (2005) allows maturity from a little over 130cm (9yo) and Olsen (1954) gave 135cm (10yo) as the minimum size at maturity for females.

Walker (2005) considered 3 indicators of male maturity: testis condition, seminal vesicle condition, and clasper condition. All measures suggest that males are mature from roughly 120cm (7 years old), although a very small proportion of animals were mature from all little as 100cm (4 years). The smallest mature male reported by Olsen (1954) was 121cm in length.

The model also uses fixed length-weight relationships for female school sharks but ignores male paternity (and therefore fecundity).

The close kin model shown here uses all of the abovementioned, fixed, parameter values, and estimates the number of pups produced at age 1 (thus absorbing the survival rate in the first year and scaling the fecundity relationship). No pupping interval is assumed.

13.11.5 Identifying sex from genetic data

Sex markers in the genetic sequence data were identified and used to check reported sex (which was generally very accurate, but occasionally incorrect) and to fill in missing sex data. Reported sex was used to identify the sex markers. The method used is surprisingly robust to errors in the reported data. A total of 15 markers (loci) were originally identified (not shown here) of which 5 were selected for

'baiting' (i.e. routine sequencing at the same time as the overall genotyping used to find kin-pairs). Multiple alleles from these loci were present in male but not female sharks, suggesting an XX-XY reproductive system. Baited loci were targeted for intensive sequencing whereas unbaited loci were not sequenced. Of the 5 loci, 3 proved reliable in identifying both males and females, whereas 2 had rather low read depth (number of sequences read) so that it was sometimes difficult to distinguish between a male with very low reads of those loci, and a female who lacked those alleles.

Even using 3 reliable loci, there is some uncertainty in the allocation of sex, with 10% of samples considered to be unclear due to difficulty distinguishing low reads from genuinely absent alleles. A low level of 'leakage' also occurs in the sequencing process, so that females can sometimes appear to have a small number of reads of alleles that are actually absent. On the other hand, variation in the number of reads of particular alleles can result in some males having very low counts of alleles that are actually present. Animals that had ambiguous counts of all 3 sex markers were not re-assigned; instead, their reported sex was assumed to be correct.

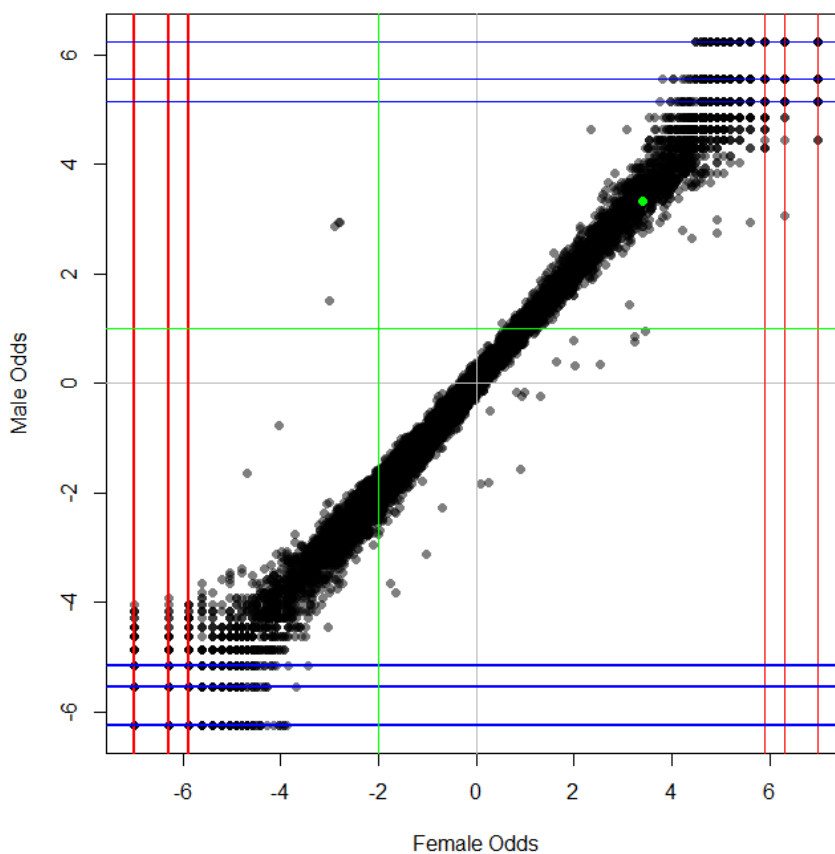


Figure 13.12. The odds ratio of having a particular genetic allele for male vs female school sharks. The red and blue lines show the odds corresponding with only 1,2, or 3 (thick lines) or all but 1, 2, or 3 (thin lines) animals having a particular allele and guard against spurious results due to low numbers of individuals. The green lines were used to select the 3 alleles (appearing in the top left quadrant) that proved to be reliable sex markers.

13.11.6 Missing meta-data for close kin sample

Along with the tissue/vertebral samples, information was requested on carcass length, sex, fishing gear, collection date, collection location, and vessel name.

Collection date (which was sometimes collection day, and sometimes landing day, and sometimes could not be unambiguously matched to trips reported in AFMA's logbook and CDR databases) and vessel name were used, where possible, to identify collection location and gear type. This was not always unambiguous. Some vessels, for example, report the use of multiple gear types (such as trawl and Danish seine, line and gillnet) in single trips.

Where collection day was missing (month and year were always provided) the first day of the month was assumed.

Several vessels appear in the logbook database much earlier than they do in the CDR database, and also provided samples (to the FAS sample collection) earlier than their appearance in the CDR record. For those vessels, the most recent fishing day reported in the logbook, on or immediately prior to the reported collection date, was assumed to give the location where the sample was taken.

13.11.7 Landed catches and discards

Landed catch data for school shark reported in logbooks was used to update the catch time series for school shark (Figure 13.13). Note the increased use of line gear from roughly 2010 (mainly in South Australia - location not shown) and its subsequent reduction in more recent years. Catches between 2009 and 2017 were not (yet) broken down by region (lower plot), for expediency, because regional information, while needed for the stock assessment model, is not needed for the close kin model.

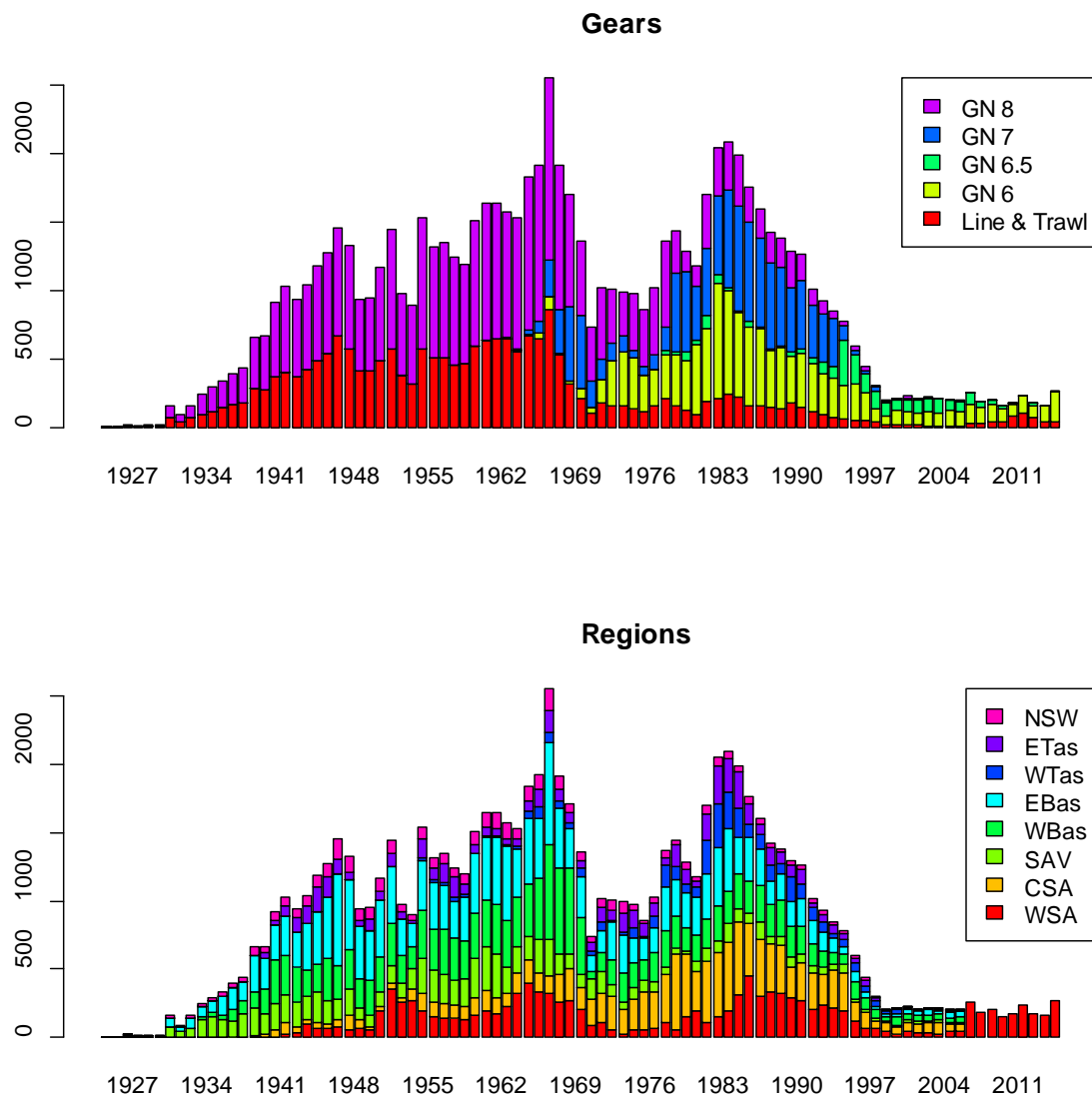


Figure 13.13. School shark catch by gear (upper plot) or region (lower plot) between 1927 and 2017. Note that the catches for 2009 to 2017 were not broken down by region.

Discards were not considered in the stock assessment model in the past. However, discard rates are likely to have increased since 2009 due to reduced TACs. Discard rates are available, from calculations using ISMP data, for 2011 to 2014 (Thomson et al 2016) and have been assumed for 2009, 2010 and 2015-2017 (Table 13.5). The discard rates were applied equally to all gears and were used to inflate the landings figures shown in Figure 13.13.

Table 13.5. Estimated and assumed discard rates for school shark calculated using ISMP data. An asterisk (*) denotes an assumed value.

| Year | Discard rate |
|------|--------------|
| 2009 | 0 * |
| 2010 | 4.5% * |
| 2011 | 9.0% |
| 2012 | 11.9% |
| 2013 | 14.4% |
| 2014 | 15.1% |
| 2015 | 15.1% * |
| 2016 | 15.1% * |
| 2017 | 15.1% * |

13.11.8 Length frequencies

The stock assessment model did not use length frequency data. This choice was made to reduce the overall computational complexity of the model (which uses 8 regions, a 12 monthly-time step, has 2 populations, and estimates movement separately for each age group, sex, and stock, and also keeps track separately of the main population and a tagged population). Instead of length frequencies, the model uses annual mean length by region and gear type (only 3 region-gear combinations were available) and sex ratio of the catch (taken from the length data).

The close kin model (and, it is planned, the next update of the stock assessment model) can use length frequencies. Because these have not been used before, and the collection of length frequency data has passed from Victoria (MAFRI) to AFMA since the mean lengths were last calculated, there is no existing code or precedents for using this data in the school shark model.

Length data are available from both port and onboard observations. These data are stored in AFMA's observer database. Sample sizes are shown for port (Table 13.6) and onboard (Table 13.7) collections, by year and gear type. Note that gillnet mesh size is not recorded in the observer database.

Table 13.6. All sample sizes (number of sharks measured by observers) by gear type (DS=Danish seine, HL=hand line, MN=mesh net, OT=otter trawl, UN=Unknown) recorded in the observer data base, collected in port.

| Year | DS | HL | MN | OT | UN |
|------|----|-----|------|-----|-----|
| 1998 | 5 | | 304 | | |
| 1999 | | | 650 | 15 | |
| 2000 | | | 322 | | |
| 2001 | 73 | | 110 | 183 | |
| 2002 | | | 150 | | |
| 2003 | | | 55 | 17 | |
| 2004 | | | 192 | 96 | |
| 2005 | | | 347 | 62 | |
| 2006 | | | 9 | | |
| 2007 | | | 696 | 4 | 12 |
| 2008 | | | 935 | | 351 |
| 2009 | | | 3925 | 56 | |
| 2010 | | 106 | 2950 | 508 | |
| 2011 | 7 | | 3925 | 429 | 7 |
| 2012 | | | 3094 | 412 | |
| 2013 | 14 | | 870 | 148 | 28 |
| 2014 | 4 | 77 | 1407 | 485 | |
| 2015 | | | 1928 | 108 | |
| 2016 | | | 1715 | 85 | |

Table 13.7. All sample sizes (number of sharks measured by observers) by gear type (DL=drop linr, DS=Danish seine, LL=long line, MN=mesh net, OT=otter trawl, setFIS=Fisheries Independent Survey trawl) recorded in the observer data base, collected onboard.

| Year | DL | DS | LL | MN | OT | setFIS |
|------|----|----|-----|------|-----|--------|
| 1993 | | | | | 27 | |
| 1994 | | 1 | | | 104 | |
| 1995 | | 1 | | | 84 | |
| 1996 | | | | | 21 | |
| 1997 | | | | | 239 | |
| 1999 | 1 | | | 1 | | |
| 2002 | | | 1 | 28 | | |
| 2003 | | | 31 | | 2 | |
| 2004 | | | 23 | | | |
| 2005 | | | 3 | | 27 | |
| 2006 | | | 1 | | 18 | |
| 2007 | | | | 395 | | |
| 2008 | | | | 62 | 3 | 2 |
| 2009 | | | 47 | 662 | 78 | |
| 2010 | | | 67 | 425 | | |
| 2011 | | | 36 | 890 | 21 | |
| 2012 | | | 102 | 1616 | 4 | 27 |
| 2013 | | | 420 | 980 | 35 | |
| 2014 | | | 942 | 1524 | 5 | 59 |
| 2015 | | | 1 | 337 | 3 | |
| 2016 | | | 5 | | 46 | 61 |

Typically, in the SESSF, port and onboard length frequencies differ, partly but not always because only the onboard sample contains measurements of smaller fish that are discarded. Other reasons for this difference are not as clear (Thomson et al 2017). Ideally, therefore, assessments incorporate both port and onboard length frequencies and estimate separate selectivity curves for both (e.g. Tuck 2017).

Sample sizes are sufficiently large for mesh nets in several years from both the port and onboard datasets, as well as trawl data from ports, and line data from onboard. There is a large dataset from unknown gear in 2008 port measurements which might, on closer examination, be possible to attribute to gear(s).

Due to time limitation, only mesh net data from ports have been used thus far (Figure 13.14). Furthermore, gear selectivity is fixed in the school shark models so any differences between port and onboard collections would have to be carefully considered.

All measurements (whether fork length or partial length) were converted to total length. Where possible, samples were scaled up using the ratio between the catch weight and the sample weight. Where catch and/or sample weight was unavailable, the samples were used but were not scaled. The length frequencies for 2003 and 2004 look peculiar and were therefore excluded from the model.

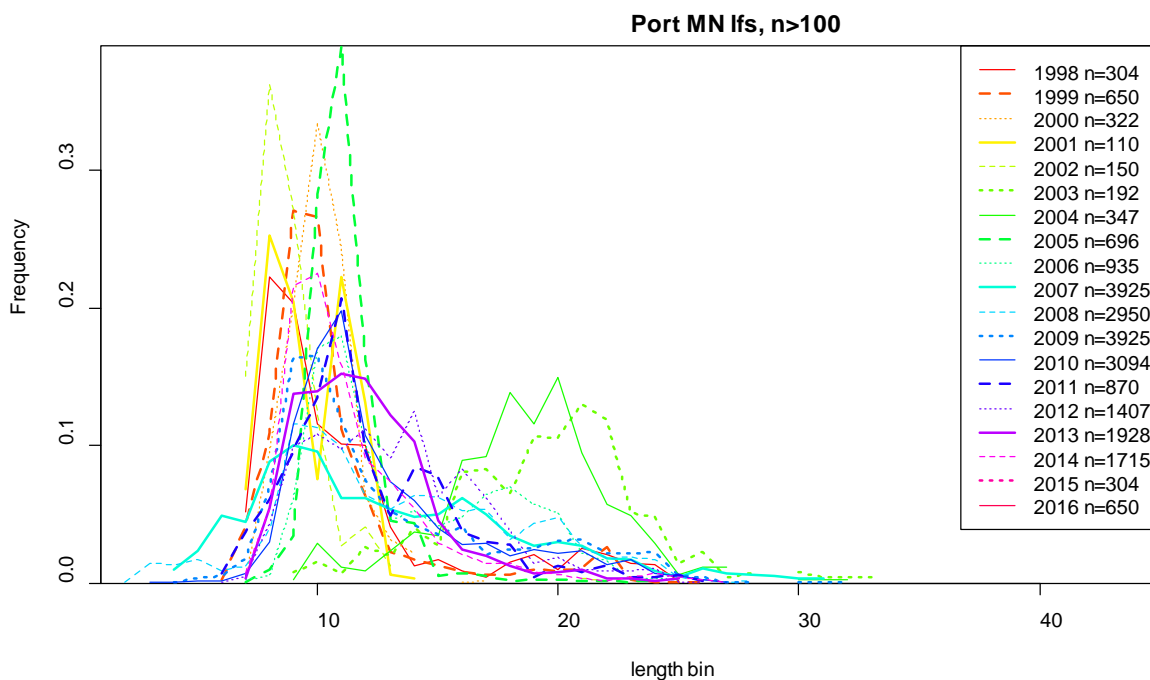


Figure 13.14. Proportional length frequency distributions for school shark from port observer measurements for all years for which at least 100 samples were measured. The distributions for 2004 and 2005 were not used.

13.11.9 Length composition in the close kin model

Length frequency data collected in port from commercial gillnet vessels were used to condition the close kin. Other data are available but were not considered, including from onboard collections (which had lower sample sizes, on the whole, than the port collections) and smaller collections of line and trawl caught fish. The mesh size associated with the gillnet catches was not known, so it was assumed that they gave a representative sample from the catch from all gillnet gears operating at the time. Unlike the full stock assessment model for school shark, no regionalisation was modelled and this could substantially affect the observed length frequencies. For example, the length composition of school shark from eastern Bass strait is much smaller than that from other areas. Too much emphasis must not, therefore, be placed on the length frequency data in this model. The weight given within the model likelihood is dictated by the “effective sample size” assigned to the length frequencies. We assumed an effective sample size of just 1 but perform a sensitivity test to assuming 5.

The stock assessment model does not use length frequency data, only mean length and sex ratios. Both the stock assessment and the close kin model presented here assume that selectivity for all gear types is known, and do not estimate those selectivities. Selectivities for gillnets were calculated from a multi-gear trial (selectivities were calculated for both school and gummy shark but only the gummy results were published, see Kirkwood & Walker 1986) but that for the combined trawl and line gear was assumed to be knife-edged at 54cm total length.

When attempting to fit to the length frequency data in the close kin model, given the assumed gear selectivities, we noted that sharks over the age of 15 (roughly 150cm in total length) were over-represented in the expected compared to the observed length frequencies. The assumed gear selectivity for gill nets is quite high even for the largest animals but, to the knowledge of the authors, these selectivities have not been tested against observed length data before now. It is possible that while the gear selectivity is high for larger animals, those animals are not available to the gear because of spatial

(including depth) segregation by size. Rather than try to estimate an availability function (or incorporate the 8 regions considered by the full assessment model) we set gear selectivity to zero for animals aged over 15.

Demographic parameters are estimated by maximizing the joint log-likelihood from all the pairwise close kin trials, plus the log-likelihood contribution from length-composition data which follows standard statistical principles used in other stock assessments.

13.11.10 Length type conversion

Observers measure sharks using either total length (TOT), fork length (LCF), or partial length (PAR). Partial length is measured from the hindmost gillslit to the base of the tail. Conversion factors have been calculated for converting both LCF and PAR to TOT using dual measurements recorded by fishery observers (Thomson 2015).

Length measurements recorded for animals sampled for the close kin project were initially assumed to be PAR, as that was the measurement requested, however it became clear that many samples were too large to be partial measurements. After consultation with Kyri Toumazos and Chris Pitliangas, it was concluded that samples originating in South Australia were partial length measurements but those from Victoria were dressed lengths (hereafter 'VIC'), measured from a point anterior to the hindmost gillslit, to a point posterior to the base of the tail. Chris Pitliangas kindly took dual measurements from 30 school sharks, to which a linear relation was fitted, allowing conversion between VIC and PAR measurements (Figure 13.15). The PAR-TOT conversion was then applied to give total length.

Length frequencies for each vessel were examined, which allowed a correction, in some cases, for that vessel's allocation to either the 'VIC' or 'PAR' classification, however some remained ambiguous. Length data for this sample must therefore be regarded with caution.

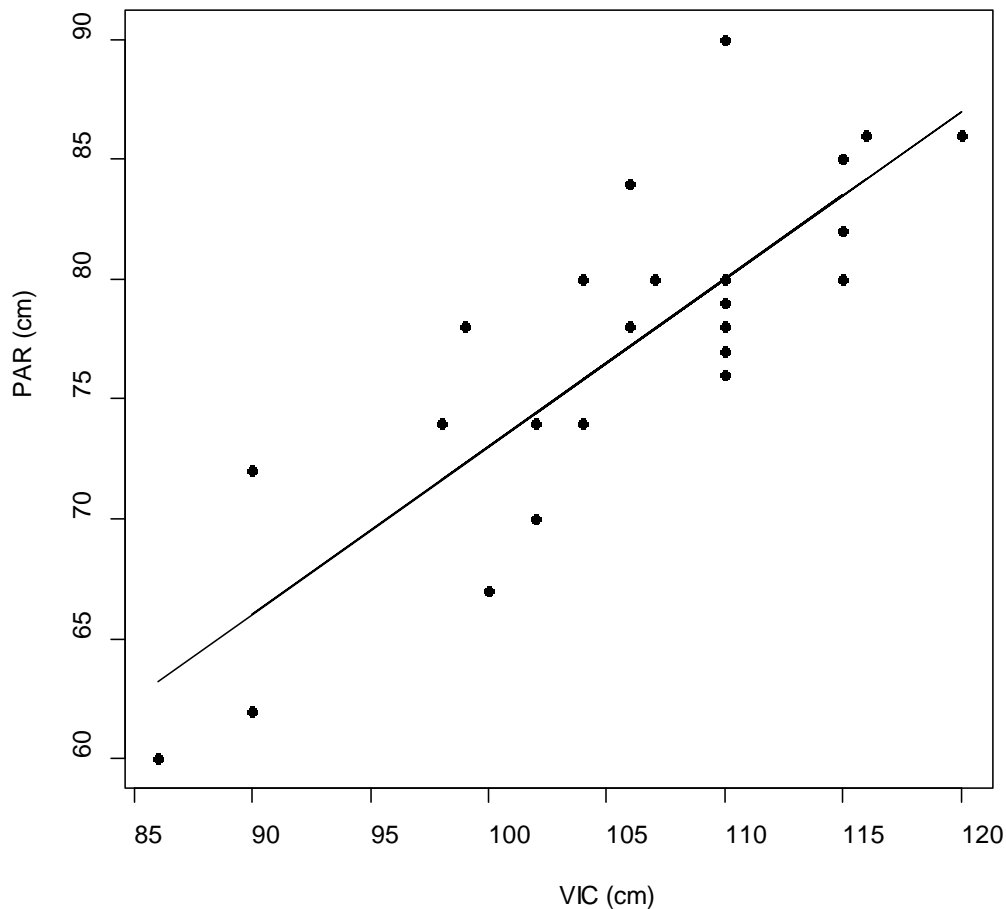


Figure 13.15. Dual measurements of partial length (PAR) and dressed length (VIC) for 30 school sharks (points) and a linear relationship fitted to these data (straight line). The intercept of the line was 3.1662 and the slope was 0.6984.

13.11.11 *Proportion female*

Information on the proportion of the catch that was female was used in the stock assessment model, this was calculated, where available, from length frequency information. Its purpose was most likely to provide additional information from the length frequency, thus partially compensating for the fact that only the mean length, not the whole frequency by sex, was used in the model.

Table 13.8. Proportion female used in the stock assessment model, available for shark regions Western South Australia (WSA) or Central South Australia (CSA) and 6" (MN6) or 7" mesh nets (MN7).

| Year | WSA | CSA | CSA |
|------|-----|-----|-----|
| | MN7 | MN6 | MN7 |
| 1970 | | | 47% |
| 1971 | | | 36% |
| 1972 | | | 47% |
| 1973 | | 71% | 65% |
| 1974 | | 64% | 61% |
| 1975 | | 57% | |
| 1976 | | 52% | |
| 1977 | | 55% | |
| 1978 | | 57% | |
| 1979 | | 61% | |
| 1980 | | 62% | |
| 1981 | | 55% | |
| 1982 | | 56% | |
| 1983 | | 57% | |
| 1984 | | | |
| 1985 | | 47% | |
| 1986 | 52% | 50% | |
| 1987 | 55% | 49% | |
| 1988 | 52% | 48% | |
| 1989 | | 47% | |
| 1990 | | 49% | |
| 1991 | 69% | 51% | |
| 1992 | 58% | 49% | |
| 1993 | 70% | 52% | |
| 1994 | 58% | 48% | |
| 1995 | 66% | 51% | |
| 1996 | 68% | 36% | |
| 1997 | 60% | | |
| 1998 | | 49% | |
| 1999 | | 44% | |
| 2000 | | 50% | |
| 2001 | | 60% | |
| 2002 | | | |
| 2003 | | 52% | |
| 2004 | | | |
| 2005 | | | |
| 2006 | | | |
| 2007 | | | |
| 2008 | | | |

For this report, annual proportion female, across regions for mesh net gear, were calculated from the ISMP port observer data, scaled to total catch where possible (Table 13.9).

Table 13.9. Proportion of the catch that was female, calculated from mesh nets from ISMP observer port data.

| Year | Scaled sample size | | Proportion female | Sample size | |
|------|--------------------|--------|-------------------|-------------|------|
| | Female | Male | | Female | Male |
| 1998 | 0 | 0 | | 0 | 0 |
| 1999 | 0 | 0 | | 0 | 0 |
| 2000 | 0 | 0 | | 0 | 0 |
| 2001 | 0 | 0 | | 0 | 0 |
| 2002 | 0 | 0 | | 0 | 0 |
| 2003 | 0 | 0 | | 0 | 0 |
| 2004 | 141.9 | 99.0 | 59% | 109 | 76 |
| 2005 | 190.0 | 157.0 | 55% | 190 | 157 |
| 2006 | 5.0 | 4.0 | 56% | 5 | 4 |
| 2007 | 1856.5 | 1844.1 | 50% | 310 | 386 |
| 2008 | 671.5 | 516.9 | 57% | 504 | 431 |
| 2009 | 2754.8 | 2447.5 | 53% | 2058 | 1860 |
| 2010 | 1884.3 | 1947.6 | 49% | 1410 | 1536 |
| 2011 | 2100.5 | 2385.6 | 47% | 1846 | 2076 |
| 2012 | 1956.5 | 1929.2 | 50% | 1523 | 1551 |
| 2013 | 415.0 | 452.0 | 48% | 415 | 452 |
| 2014 | 1065.0 | 1076.5 | 50% | 697 | 709 |
| 2015 | 931.2 | 1235.1 | 43% | 836 | 1092 |
| 2016 | 799.0 | 915.0 | 47% | 799 | 915 |

Note that most sex ratios shown in these tables are close to 50%, and given the tendency of school shark to swim in same sex schools (Olsen 1954, Walker 1999), CV is likely to be relatively high. For this reason it was decided to assume a 50:50 sex ratio for the catch for all gears.

13.11.12 Identifying close kin pairs

An extensive data cleaning and genetic locus selection exercise was performed. This identified 1,706 informative loci from 27,532 candidates available. Sequencing information was available for 2,886 samples, of which 244 were re-sequenced to allow the estimation of sequencing error rates. There were also several accidental duplicates and a number of (probably) gummy shark seemingly resulting from re-sampling of the same fish at the processor or species mis-identification. These 85 accidental duplicates were removed from the sample. An additional 36 samples were removed because of either excessive heterozygosity (an indication of DNA contamination i.e. the DNA of more than one fish is found in the sample) or too little heterozygosity (an indication of degraded DNA). Another 78 samples were removed because of their gene frequencies were outside of the norm. Some fish, for reasons yet to be understood, have a genetic make-up that causes them to show a high degree of statistical kinship with a wide range of fish. A total of 249 fish showing this tendency were also removed.

Once loci and fish had been selected, a range of statistical measures were applied to determine which pairs are close relatives, and what their relationship is. Figure 13.16 shows the results for three statistics

that were used, which were developed specifically for close kin projects and have yet to be documented. The blue cluster of points in Figure 13.16 are FSPs mixed with POPs. The green cluster are HSPs and the grey points are unrelated pairs, some HSPs that fall below the chosen threshold value for the PLOD statistic (η) by chance, and more distant relatives such as cousins and half uncle/aunt – niece/nephews. Figure 13.3 shows the distribution of the ‘PLOD’ statistic, which gives the pseudo-likelihood that a pair are HSPs. HSPs merge into unrelated and less related pairs (UPs). A cut-off PLOD value must be chosen (termed η), above which pairs are clearly HSPs but below which some ‘false negatives’ will occur. Theory applied to the observed allele frequencies for all the loci used in the study provides a mean value for PLOD for true HSPs, and the observed distribution of HSPs above this mean provides an estimate of the variance of the HSP distribution. Thus the proportion of HSPs ‘lost’ below the threshold (named η) can be calculated from the theoretical distribution of HSPs given this mean and variance (orange curve on Figure 13.3). For this study the false negative loss rate was 12%.

There was no clear genetic delineation between POPs and FSPs in this study. Figure 13.17 shows two statistics for separating out POPs (‘wpsex’ and ‘nABOO’) but no separation of a cluster of POPs is visible. Clear delineation has been observed for grey nurse shark (not shown). The white shark and *Glyphis* samples consisted of juveniles only, so no POPs were possible. The original SBT study searched for POPs only. In principle, it is possible to distinguish POPs from FSPs by using enough loci and carefully-designed statistical measures (whereas, by contrast, it is impossible to distinguish between half-sibling-pairs and grandparent-grandchild pairs). The statistic we used for FSP/POP delineation was fairly easy to calculate but not the most powerful possible, more powerful versions will be sought in future. In this particular dataset, though, we were able to separate POPs from FSPs based on the age differences between the animals in each pair (Figure 13.4).

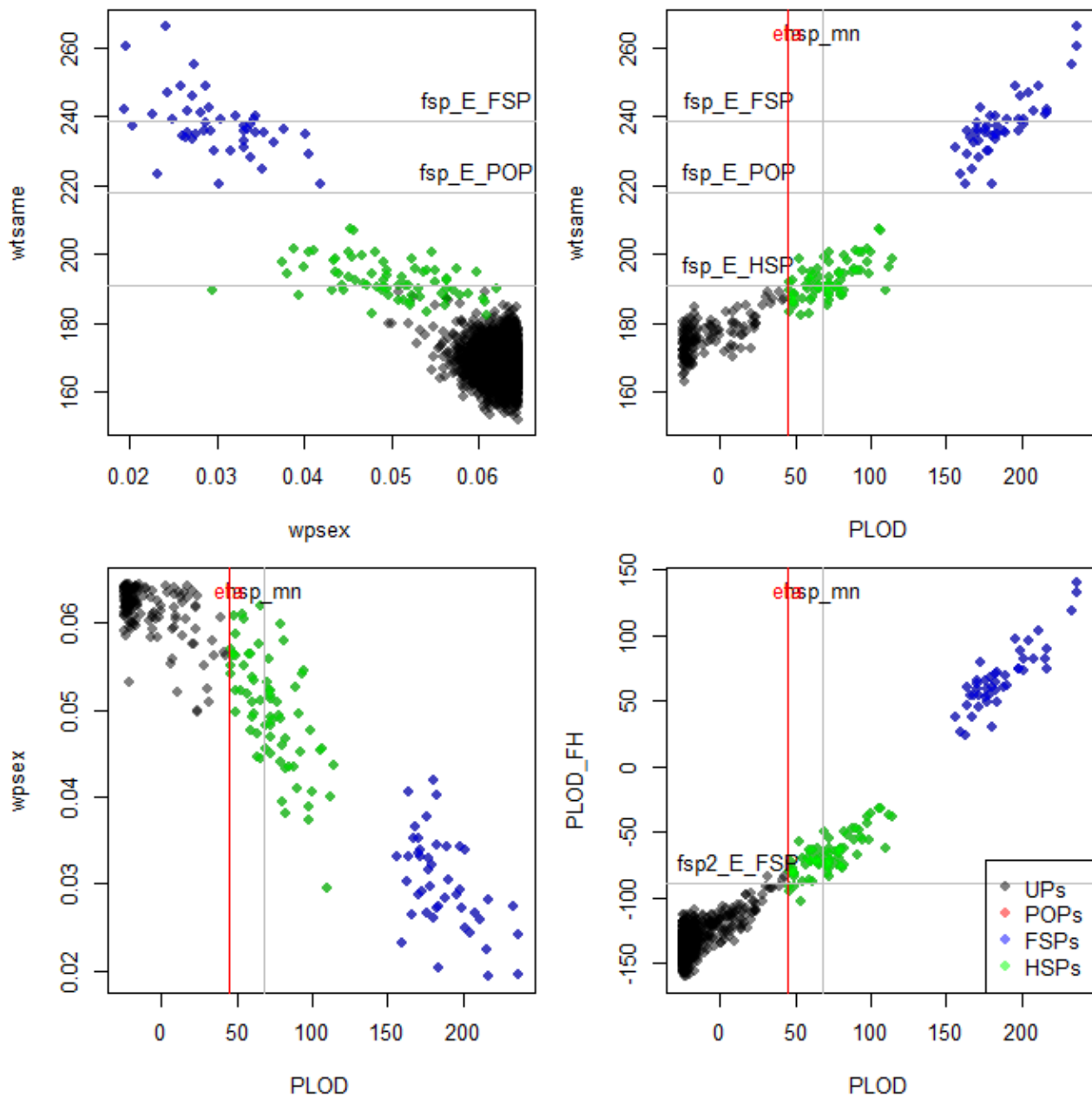


Figure 13.16. Scatter plots (right) showing three statistics for determining relatedness ‘wtsame’ is optimised for finding FSPs, ‘wpsex’ for POPs, and PLOD for HSPs. pairs of animals. Each dot represents a pair of animals, some less related pairs are not shown. The theoretical mean PLOD value for HSPs is shown (grey line) as well as the PLOD threshold value used for identifying clear HSPs (red line marked ‘eta’).

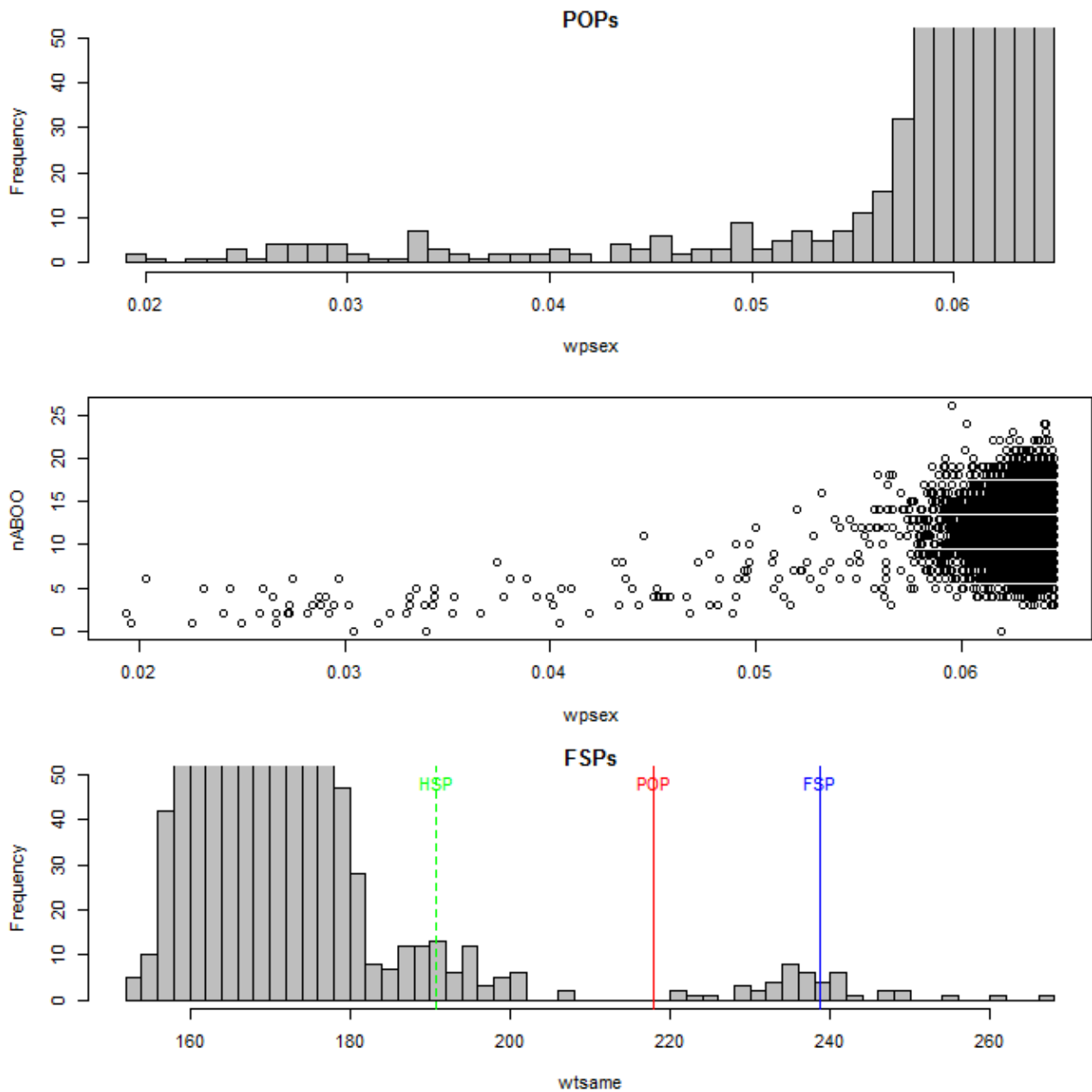


Figure 13.17. Histogram showing the frequency distribution of the more closely related pairs for the statistic 'wpsex' that is optimised for finding POPs (upper plot) and 'wpsex' along with another POP-finding statistic 'nABOO' (middle plot) and frequency distribution for the 'wtsame' statistic for finding FSPs showing theoretical values for HSPs (green line), POPs (red line) and FSPs (blue line).

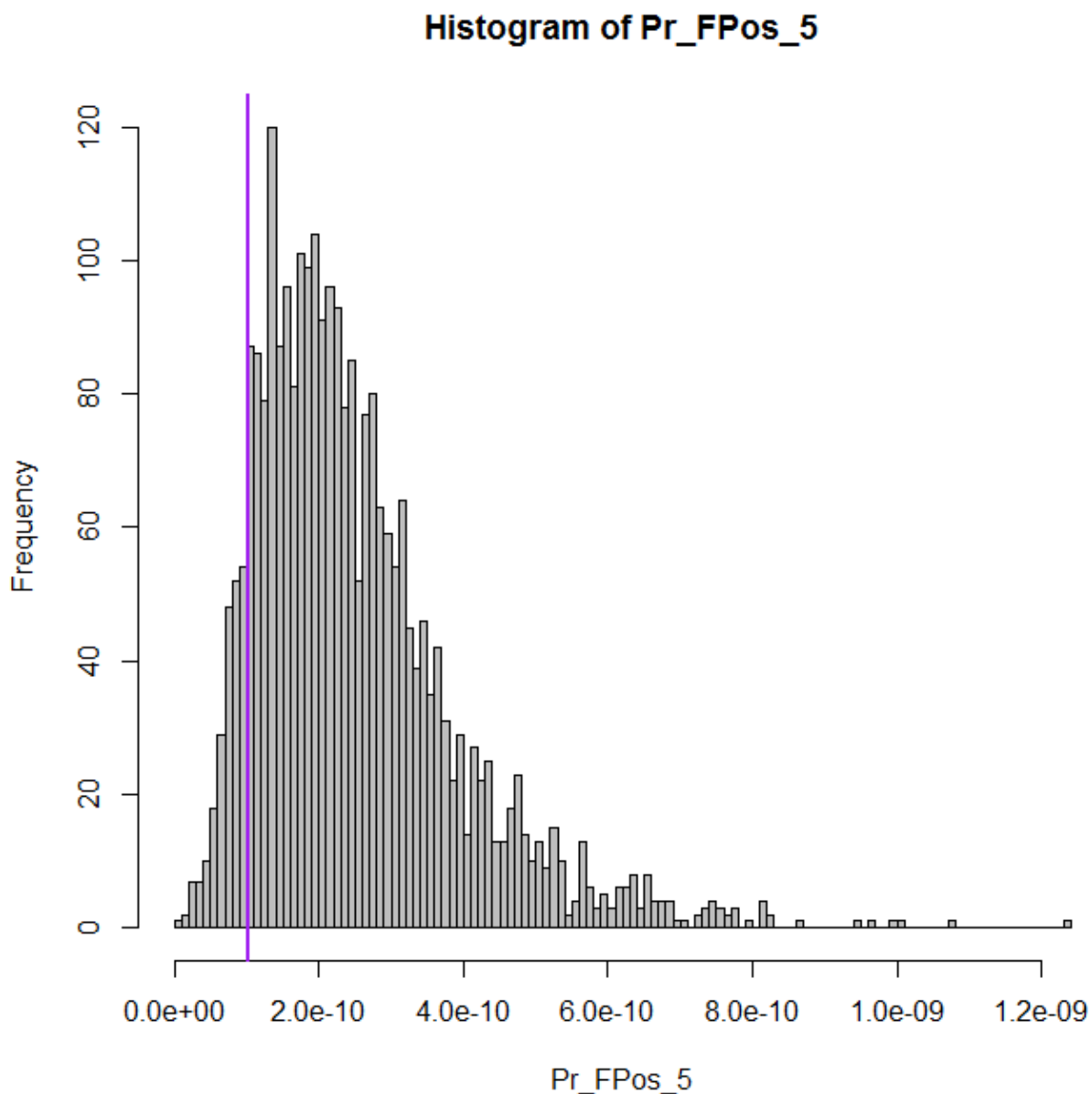


Figure 13.18. Histogram showing the frequency distribution for all animals sampled for a statistic that attempts to measure how likely that is animal is to appear, spuriously, to be related to other animals. Those to the left of the purple line were removed from the sample.

It was noted during the SBT study that some fish, because of their genetic makeup, happen to show a greater probability of being found to be related to other fish. These are likely to result in false positives that will over-estimate the true numbers of kin pairs and thereby lead to under estimation of abundance. A statistic (CLOD) that measures this propensity was applied and all fish below a threshold value were discarded. This resulted in the discarding of 249 school shark (Figure 13.18).

13.11.13 Simple GLM method

If some broad assumptions are made, it is possible to apply a very simple model to close kin data alone, yielding an estimate of population size and trend. Assume that (1) fecundity of mature animals does

not change with age (for female school shark it changes from roughly 20 to 30 pups per litter, and for males it is unknown); (2) although some cohorts can be unusually large due to favourable environmental effects, this effect can be eliminated by removing same-cohort kin pairs; (3) mortality rates do not vary over the model time period; (4) trend in abundance is linear; (5) birth year can be accurately inferred from corrected “ring” counts.

Given these assumptions, consider a particular maternal half sibling pair: the mother of the older animal is shared by the younger animal. The probability that the mother of the older animal is also that of the younger animal, provided she survives, is $1/N^f$ where N^f is the number of adult females in the population in the year that the younger animal was born.

If there is a gap of t years between the births of the older and younger animals, then the survival rate for the mother must be $\exp(-Z*t)$; note that Z is assumed to be constant. The overall probability that the mother of the older animal is also that of the younger animal is the probability that she survives, times the probability that she, out of all the living females in the population, is the mother of the younger animal. The probability, therefore that these two animals, born t years apart, is a maternal half sibling pair (MHSP) is:

$$P(MHSP|t) = 1/N^f e^{-Z t}$$

This approximation is only possible because the variation in fecundity for female school shark is relatively small (between 20 and 30) whereas for teleost fish the change in fecundity with body size is much more profound and could not be ignored.

Now assume that there is a trend in abundance over time, so that the number of females in the y th year of the model N_y^f is given by

$$N_y^f = N_0^f e^{r y}.$$

Now the overall probability that two animals are a maternal half sibling pair, given that the older was born in the y th year, and the other younger t years later, is:

$$P(MHSP|t) = e^{-r y} \left(\frac{1}{N_0^f} \right) e^{-Z t}$$

If samples are taken from a set of individuals, such that n unique pairings can be formed, each of which is a potential MHSP for animals born t years apart, then the expected number of MHSPs from those n pairings is n times the probability $P(MHSP|t)$ above.

This can be viewed as a series of Binomial probabilities where each pairing is a trial with “success” given by $P(mHSP|t)$, however these probabilities will be very low (for a population of the likely size of the school shark population) and can thus be approximated by a Poisson with expected value

$$n e^{-r y - Z t} \left(\frac{1}{N_0^f} \right)$$

Because the population size is relatively large, each pairwise comparison can be regarded as independent.

Similar formulae apply for paternal HSPs (PHSPs) but the number of mature males in the population N_0^m might differ from N_0^f even if mortality and birth rates are the same for both sexes (i.e. the number of males and females in the population is the same), simply because males mature at a younger age therefore a larger proportion of the total number of males will be adults. It is also possible the mortality rates might differ for the sexes as a result of variable fishing mortality rates due to spatial segregation or differing natural mortality rates. Nevertheless, the trend (r) in the male and female numbers ought to be similar, at least.

A GLM was used to estimate trend (r) and numbers of males and females in year 2000 (N_0^m and N_0^f) given the observed numbers of MHSPs, and PHSPs by birth years.

Corrected birth years were calculated by ignoring ageing error and variability in ring deposition rate and assuming that for ages 1 to 11 the most probable age is given by the “ring” count whereas from age 12 rings are deposited at a rate of 0.36 per year, therefore an additional “ring” will only be visible after roughly every 3 years.

Observed number of HSPs were corrected for the “false negative loss rate” as described elsewhere in this report.

Some litters might, by chance, have higher survival rates than the norm because of favourable conditions. These will be over-represented in the kin sample because those favourable conditions will prevail for both animals in the same-cohort sibling pair. To avoid having to estimate additional parameters to correct for this eventuality, same-cohort siblings must be removed from the sample. However, ageing errors make it difficult to identify these same cohort pairs.

To minimise the risk of same-cohort comparisons, we excluded all comparisons where the nominal difference in birth-year was between 0 and 4 years. Note that both the kin pairs, and the number of comparisons (n) were removed from the dataset.

To improve the accuracy of our corrected birth years, and to restrict the period over time over which our assumptions must apply, we removed animals with more than 15 “rings” and those born before 2000.

Note that ageing error is likely to lead to systematic over estimation of birth interval (t) corresponding under estimation of the mortality rate Z , but the estimates of trend and absolute population should not be badly affected. Nevertheless, the results of this very approximate method should not be over interpreted.

14. Orange roughy east (*Hoplostethus atlanticus*) cross-catch risk assessment based upon the 2017 stock assessment

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14.1 Executive Summary

This paper presents a cross-catch risk assessment for eastern orange roughy based upon the model structure of Haddon (2017). Two models are considered that differ only by the assumed value of natural mortality, M . The base-case model has $M=0.04$ and an alternative has $M=0.032$. The alternative value for natural mortality was chosen to define a low productivity model, and used the value with highest likelihood from the likelihood profile of Haddon (2017).

The catches input to the two model structures were the predicted projected catches from each model, and a fixed 3-year catch series proposed by industry; thus three projected catch scenarios associated with each natural mortality used. The purpose of the risk assessment was to identify if any of the catch series led to biomass trajectories that may be perceived as a risk to the long-term sustainability of the stock. The consequent six scenarios (2 models \times 3 catch series) were projected 55 years into the future.

Results showed that, as expected, the model with lower productivity (the $M=0.032$ model) and with the highest catches (from the $M=0.04$ model) had the lowest long-term biomass series (in terms of annual tonnage of female spawning biomass). This series stabilised at approximately 30% of virgin biomass. All other scenarios had biomass levels that were considerably greater than this. As far as short-term catches and depletion were concerned, the differences between biomass trajectories across catch series were minimal within a model structure (i.e. for a particular value of M). For example, by 2025, the depletion ranged between 0.40 and 0.42 for the $M=0.04$ models, whereas the depletion ranged between 0.31 and 0.34 for the $M=0.032$ model.

14.2 Introduction and Methods

This paper presents a cross-catch risk assessment for eastern orange roughy based upon the underlying stock assessment of Haddon (2017). The 2017 stock assessment provided a base-case assessment ($M=0.04$, $h = 0.75$) and an alternative based upon a likelihood profile that suggested the stock may be less productive ($M=0.036$, $h = 0.60$). A cross-catch risk assessment was conducted on these models, whereby the projected catch predicted by the usual HCR from one model structure was used as an input for deterministic projections in the other model (Haddon, 2017).

At the August 2018 SESSFRAG meeting (AFMA, 2018a) an alternative series of catches from industry was proposed. SERAG has now been asked to consider the industry catches in a cross-catch risk assessment using the base-case ($M=0.04$, $h=0.75$) and an alternative low productivity model ($M=0.032$, $h=0.75$). The choice of parameters for the alternative model was based upon a natural mortality value of $M=0.032$, being the most likely value from the likelihood profile analysis conducted by Haddon (2017). As steepness would only become important for catches in the future, only M is varied in the scenarios considered in the cross-catch risk assessment, with steepness fixed at the base-

case value of $h=0.75$. The model with $M=0.032$ has been tuned using the same method as the base-case $M=0.04$ model of Haddon (2017). Note that aspects of the model tuning process and platform have changed since this assessment was conducted, however it was not within the remit of this work to reassess the stock. As such the same model structure, platform and tuning methods were applied as those conducted on the base-case in 2017. Catches from 2017 were recalculated based on landings from the eastern roughy zone and logbook catches from Pedra Branca. Haddon (2017) used the TAC to estimate catches in 2017 for projections.

There are 6 scenarios to consider: 2 model structures \times 3 catch series (AFMA, 2018b). The scenario definitions are in Table 14.1 and the catch series for each scenario from 2017 to 2025 is given in Table 14.2. Note that the projected catches from the base-case model ($M=0.04$) are greater than 1.5 times the current TAC. As such, annual catches were set equal to 1.5 times the previous year's catch until projected catches were no longer greater than 1.5 times the previous year (invoking the 50% change rule). This only affected models 04w04 and 032w04 (shaded cells of Table 14.2).

Table 14.1. The definitions of each of the 6 scenarios where the name of the scenario is given by “Model” with “Catch”, eg “04w032” refers to the $M=0.04$ model structure and with catch from the model having $M=0.032$.

| Scenarios | Model | |
|-----------|------------------------|--------------------------------|
| | Base-case ($M=0.04$) | Low productivity ($M=0.032$) |
| Catch | Base-case HCR | 04w04 |
| | Low productivity HCR | 04w032 |
| | Industry proposal | 04wInd |
| | | 032w04 |
| | | 032w032 |
| | | 032wInd |

Table 14.2. The catches from years 2017 to 2025. Note the model was projected to 2071, but only catches to 2025 are shown here for brevity. The shaded cells were fixed in the projection model due to (i) known catches from 2017, (ii) as fixed input catches from one model into the alternative (04w032, 032w04), (iii) due to the influence of the 50% catch limit meta-rule (04w04, 032w04), (iv) due to fixed 3-year industry proposed catches (04wInd, 032wInd). The unshaded catch values come directly from the SESSF Tier 1 20:35:48 HCR.

| | Scenario | | | | | |
|------|----------|--------|--------|---------|--------|---------|
| | 04w04 | 04w032 | 04wInd | 032w032 | 032w04 | 032wInd |
| 2017 | 346 | 346 | 346 | 346 | 346 | 346 |
| 2018 | 698 | 537 | 709 | 537 | 698 | 709 |
| 2019 | 1046 | 615 | 900 | 615 | 1046 | 900 |
| 2020 | 1398 | 686 | 900 | 686 | 1398 | 900 |
| 2021 | 1423 | 750 | 1437 | 750 | 1423 | 722 |
| 2022 | 1443 | 806 | 1457 | 806 | 1443 | 779 |
| 2023 | 1459 | 855 | 1473 | 855 | 1459 | 828 |
| 2024 | 1472 | 896 | 1485 | 896 | 1472 | 871 |
| 2025 | 1481 | 931 | 1494 | 931 | 1481 | 907 |

14.3 Results

The trajectory of female spawning biomass and relative female spawning biomass are shown in Figure 14.1 to Figure 14.3, with relative values for years 2017 to 2025 in Table 14.3. As expected, the higher productivity model ($M=0.04$) with the lower catches (from the $M=0.032$ model) shows the largest biomass over time (04w032; blue). The higher productivity model with catches from this model (04w04) and with the industry proposed catches (04wInd) show little difference over time (both long and short term; Table 14.3). Likewise, the lower productivity model ($M=0.032$) with the higher catches

(from the model with $M=0.04$) shows the lowest biomass trajectory (032w04; yellow). The lower productivity model with catches from this model (032w032) and with the industry proposed catches (032wInd) show little difference over time (both long and short term; Table 14.3). This is because the industry proposed catches are only applied for 3 years before returning to the standard Tier 1 HCR of the particular model.

In terms of risk, the lowest biomass occurs with the low productivity model with the high productivity model catches (032w04). In this case the biomass stabilises at approximately 0.30Bo. All other scenarios stabilise at a relative biomass level well above this. Note that these are deterministic projections and the uncertainty surrounding these trajectories has not been estimated using methods such as MCMC (further work). However, the asymptotic confidence intervals are all above the limit reference point from approximately 2017 onwards (Figure 14.3). The greatest difference in realised biomass between these models is, not surprisingly, in the utilisation of one or other of the models with differing natural mortality.

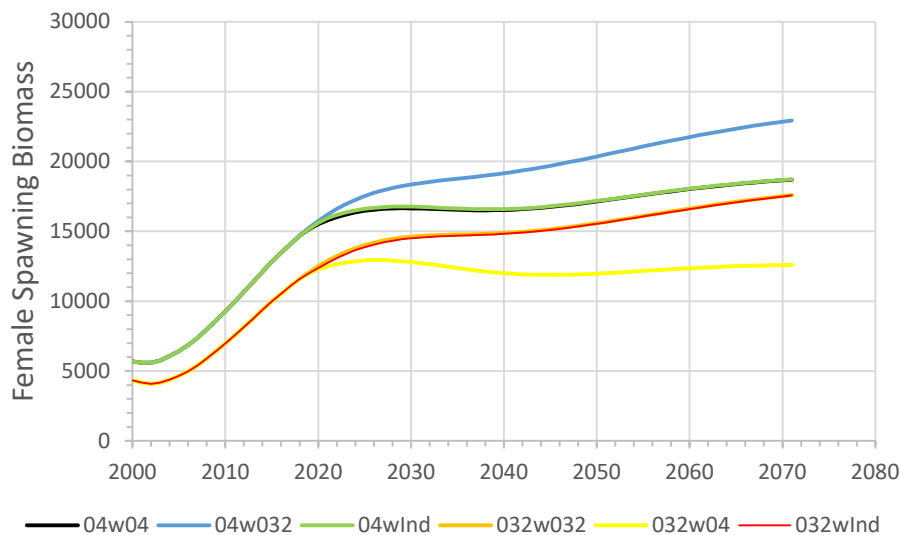


Figure 14.1. The female spawning biomass (t) for eastern orange roughy under each of the six scenarios described in Table 14.1.

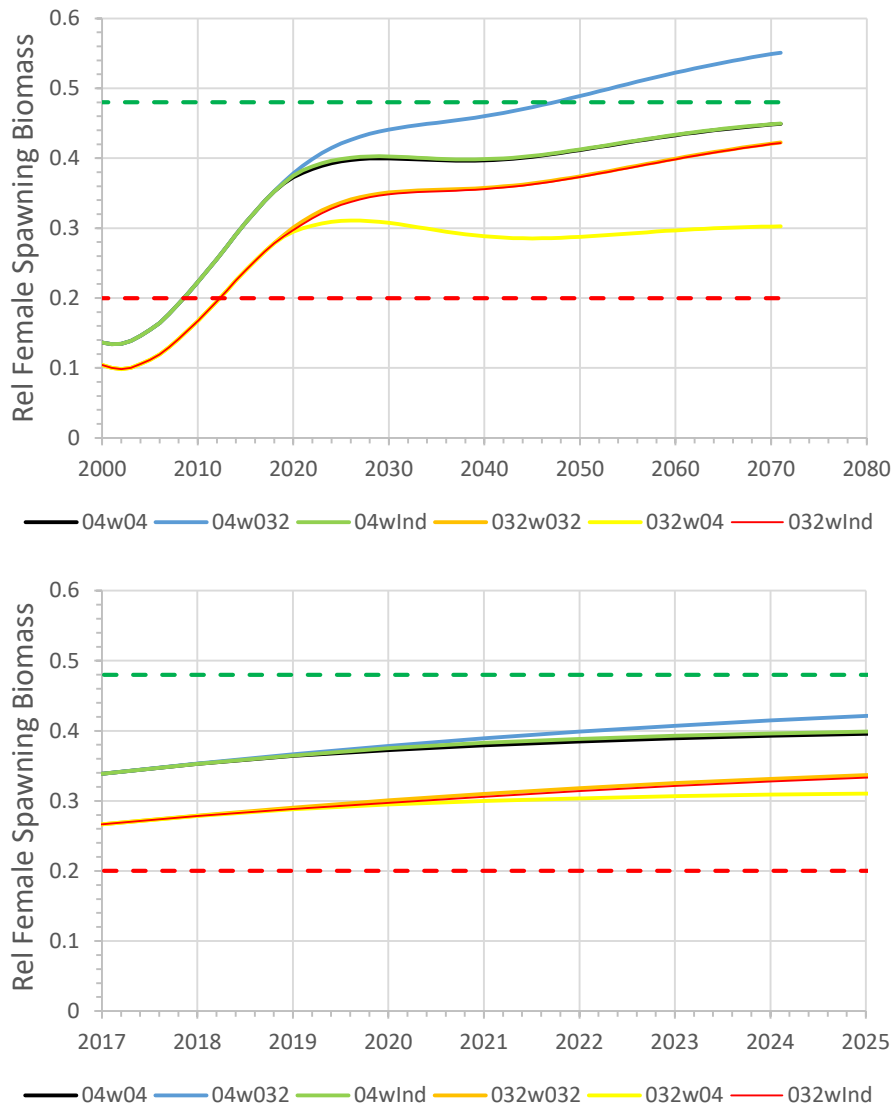


Figure 14.2. The relative female spawning biomass for eastern orange roughy under each of the six scenarios described in Table 14.1 (top) and with restricted years to better show detail of the trajectories (bottom).

Table 14.3. The relative female spawning biomass from 2017 to 2025 for each of the six scenarios of Table 14.1.

| | Scenario | | | | | |
|------|----------|--------|--------|---------|--------|---------|
| | 04w04 | 04w032 | 04wInd | 032w032 | 032w04 | 032wInd |
| 2017 | 0.34 | 0.34 | 0.34 | 0.27 | 0.27 | 0.27 |
| 2018 | 0.35 | 0.35 | 0.35 | 0.28 | 0.28 | 0.28 |
| 2019 | 0.36 | 0.37 | 0.36 | 0.29 | 0.29 | 0.29 |
| 2020 | 0.37 | 0.38 | 0.37 | 0.30 | 0.29 | 0.30 |
| 2021 | 0.38 | 0.39 | 0.38 | 0.31 | 0.30 | 0.31 |
| 2022 | 0.38 | 0.40 | 0.39 | 0.32 | 0.30 | 0.31 |
| 2023 | 0.39 | 0.41 | 0.39 | 0.33 | 0.31 | 0.32 |
| 2024 | 0.39 | 0.41 | 0.40 | 0.33 | 0.31 | 0.33 |
| 2025 | 0.40 | 0.42 | 0.40 | 0.34 | 0.31 | 0.33 |

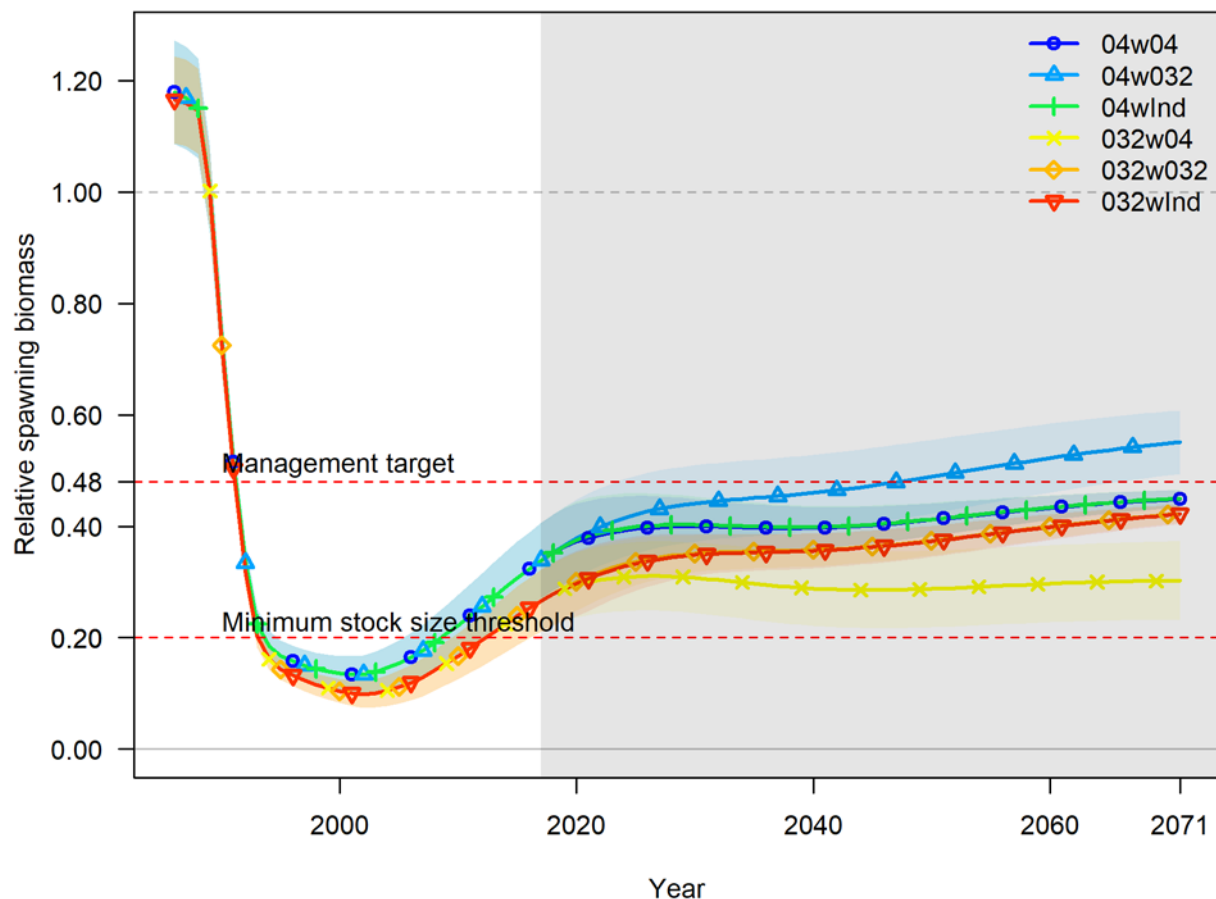


Figure 14.3. The relative female spawning biomass for eastern orange roughy under each of the six scenarios described in Table 14.1 with asymptotic 95% confidence intervals.

14.4 Discussion

This paper presents a cross-catch risk assessment for eastern orange roughy based upon the model structure of Haddon (2017). Two models are considered that differ only by the assumed value of natural mortality, M . The base-case model has $M=0.04$ and an alternative has $M=0.032$. The alternative value for natural mortality was chosen to define a low productivity model that then bounds a greater degree of uncertainty across parameterisations than the cross-catch risk assessment of Haddon (2017) that used $M=0.036$. The value of $M=0.032$ also was the most likely value of M for the given model structure and data inputs identified in the likelihood profile of Haddon (2017).

The catches input to the two model structures were the respective catches from each model, and a 3-year catch series proposed by industry. The purpose of the risk assessment was to identify if any of the catch series led to biomass trajectories that may be perceived as a risk to the long-term sustainability of the stock. The consequent six scenarios (2 models \times 3 catch series) were projected 55 years into the future.

Results showed that the model with lower productivity and with the highest catches (Model 032w04) had the lowest long-term biomass series (in terms of annual tonnage of female spawning biomass). This series stabilised at approximately 30% of virgin biomass. All other scenarios had biomass levels

that were considerably greater than this although only in model 04w04 did the depletion level rise above $0.48B_0$ after 55 years. As far as short term (out to 2025) catches and depletion are concerned, the differences between biomass trajectories across catch series were minimal within a model structure (i.e. for a particular chosen value of M). For example, by 2025, the depletion ranged between 0.40 and 0.42 for the $M=0.04$ models, whereas the depletion ranged between 0.31 and 0.34 for the $M=0.032$ models. These small differences are unlikely to be distinguishable within the natural stochastic variability of the stock and the uncertainty in the model. The greater uncertainty is with regard to the value of M used in the model. This is not unusual for stock assessment models, as M is known to be influential, but unfortunately in many cases difficult to determine. This uncertainty in natural mortality should be explored further and AFMA (2018b) proposes analyses to do this.

14.5 Acknowledgements

The authors wish to thank Malcolm Haddon (CSIRO Honorary Fellow) and all other members of the CSIRO SESSF assessment team for their contributions to a preliminary version of this document.

14.6 References

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- AFMA (2018b). Orange Roughy East RBC advice. Report to the SEFRAG, 19-21 September 2018.
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15. Benefits

The results of this project have had a direct bearing on the management of the Southern and Eastern Scalefish and Shark Fishery. Direct benefits to the commercial fishing industry in the SESSF have arisen from improvements to, or the development of, assessments under the various Tier Rules of the Commonwealth Harvest Strategy Policy for selected quota and non-quota species. Information from the stock assessments has fed directly into the TAC setting process for SESSF quota species. As specific and agreed harvest strategies are being developed for SESSF species (a process required by and agreed to under EPBC approval for the fishery), improvements in the assessments developed under this project have had direct and immediate impacts on quota levels or other fishery management measures (in the case of non-quota species).

Participation by the project's staff on the SESSF Resource Assessment Groups has enabled the production of critical assessment reports and clear communication of the reports' results to a wide audience (including managers, industry). Project staff's scientific advice on quantitative and qualitative matters is also clearly valued.

The stock assessments presented in this report have provided managers and industry greater confidence when making key commercial and sustainability decisions for species in the SESSF. These assessments have provided the most up-to-date information, in terms of data and methods, to facilitate the management of the Southern and Eastern Scalefish and Shark Fishery.

16. Conclusion

- Provide quantitative and qualitative species assessments in support of the four SESSFRAG assessment groups, including RBC calculations within the SESSF harvest strategy framework.

The 2018 assessment of the stock status of key Southern and Eastern Scalefish and Shark fishery species is based on the methods presented in this report. Documented are the latest quantitative assessments (Tier 1) for key quota species (school shark, jackass morwong (east and west), blue grenadier, silver warehou and an eastern orange roughy risk assessment, as well as cpue standardisations for shelf, slope, deepwater and shark species and Tier 4 analyses. Typical assessment outputs provide indications of current stock status and an application of the Commonwealth Harvest Strategy framework. This framework is based on a set of assessment methods and associated harvest control rules, with the decision to apply a particular combination dependent on the type and quality of information available to determine stock status (Tiers 1 to 4).

The assessment outputs from this project are a critical component of the management and TAC setting process for these fisheries. The results from these studies are being used by SESSFRAG, industry and management to help manage the fishery in accordance with agreed sustainability objectives.

Stock status and Recommended Biological Catch (RBC) conclusions (Tier 1):

The 2015 assessment for eastern and western jackass morwong was updated by the inclusion of data to the end of 2017. The base-case assessment for eastern jackass morwong estimates that current spawning stock biomass is 35% of unexploited stock biomass. Under the agreed 20:35:48 harvest control rule, the 2019 recommended biological catch (RBC) is 261 t, with the long-term yield (assuming average recruitment in the future) of 356 t. The average RBC over the three-year period 2019-2021 is 270 t. The base-case assessment for western jackass morwong estimates that current spawning stock biomass is 68% of unexploited stock biomass. Under the agreed 20:35:48 harvest control rule, the 2019 recommended biological catch (RBC) is 235 t, with the long-term yield (assuming average recruitment in the future) of 158 t. The average RBC over the three-year period 2019-2021 is 223 t

The base case Tier 1 assessment for blue grenadier was updated from the last full assessment in 2013. Recent estimated recruitments are more stable than has been observed before. While a promising sign for the fishery, some caution should be exercised regarding these recruitment estimates and its implication on future stock status, until clear further indications of its existence and magnitude are evident. The estimated spawning biomass in 2019 which is used in the harvest control rule, is approximately 122% SSB₀. The 2019 recommended biological catch (RBC) under the 20:35:48 harvest control rule is 13,260t. The long-term retained catch is 4,899t. The retained portion of the RBC for 2019 is estimated to be 12,671t.

A quantitative Tier 1 assessment of silver warehou using data to the end of 2017 was updated from the last full assessment in 2015. The assessment estimates that the projected 2019 spawning stock biomass will be 31% of virgin stock biomass. The recommended biological catch (RBC) from the base case model for 2019 is 942t for the 20:35:48 harvest control rule, increasing to 1,353t in 2020 and 1,420t in 2021. Scenarios with constant annual catches of 750 t or more led to the estimated spawning biomass declining under a low recruitment scenario. Under the low recruitment scenario with constant annual catches between 348 t and 600 t, spawning biomass is predicted to increase.

A cross-catch risk assessment for eastern orange roughy was presented based upon the model structure of the last full quantitative assessment in 2017. Two models are considered that differ only by the assumed value of natural mortality, M . The base-case model has $M=0.04$ and an alternative has $M=0.032$. The catches input to the two model structures were the predicted projected catches from each model, and a fixed 3-year catch series proposed by industry; thus three projected catch scenarios associated with each natural mortality were used. Results showed that the model with lower productivity (the $M=0.032$ model) and with the highest catches (from the $M=0.04$ model) had the lowest long-term biomass series (in terms of annual tonnage of female spawning biomass). As far as short-term catches and depletion were concerned, the differences between biomass trajectories across catch series were minimal within a model structure.

A close kin analysis was integrated within a stock assessment for school shark in 2018. Compared with the 2012 projection of the stock assessment model for school shark, which assumed catches of 225t after 2011, the analyses and the close kin model both estimate a substantially lower adult abundance. This is supported by the findings of the simple model approaches which found around 80,000 "typical adults on average" across the 2000s (one line approach) and roughly 40,000 to 80,000 adults (GLM model). While these are certainly crude, and only suitable as a check that the more elaborate close kin model has been set up correctly, it is quite clear that estimates of adult abundance in the 200,000s (stock assessment model) are incompatible with the observed close kin data.

17. Appendix: Intellectual Property

No intellectual property has arisen from the project that is likely to lead to significant commercial benefits, patents or licenses.

18. Appendix: Project Staff

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