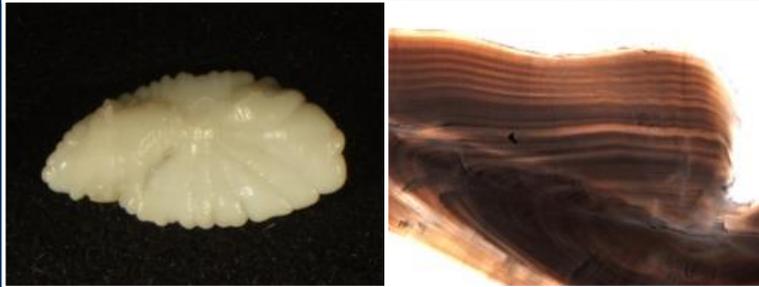# 2021 Stock Assessment Report

# Patagonian toothfish (*Dissostichus eleginoides*)



**Skeljo F**
**Lee B**
**Winter A**

**Fisheries Department**
**Directorate of Natural Resources**
**Falkland Islands Government**
**Stanley, Falkland Islands**

**31 May 2022**

SA – 2021 – TOO

**Participating Scientific Staff**

Frane Skeljo (PhD, Stock Assessment Scientist)
Brendon Lee (PhD, Fisheries Scientist)
Andreas Winter (PhD, Senior Stock Assessment Scientist)

Comments provided by: Alexander Arkhipin

**For citation purposes this publication should be referenced as follows:**

Skeljo F, Lee B, Winter A. 2022. 2021 Stock assessment report for Patagonian toothfish (*Dissostichus eleginoides*). Fisheries Report SA-2021-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands. 44 p.

Distribution: Public Domain

**Reviewed and approved by:**

Andrea Clausen                                              Date: 31 May 2022
Director of Natural Resources

# Table of Contents

## Summary

1. This report provides an updated Bayesian age-structured stock assessment of *Dissostichus eleginoides* in Falkland Islands waters, using data through the year 2021. Several changes were introduced in the 2021 model regarding both data treatment and model assumptions, following the recommendations of the Falkland Islands toothfish fishery external review (Bergh 2018) and the best practices in toothfish stock assessment around the world.

2. The initial spawning stock biomass $SSB_0$ was estimated at 25,692 tonnes and the current spawning stock biomass $SSB_{2021}$ at 12,432 tonnes, both higher than in the previous year's assessment (by 10.9% and 12.4% respectively).

3. The ratio of current spawning stock biomass to initial spawning stock biomass ($SSB_{2021}/SSB_0$) was estimated at 0.484, almost identical to the previous year's assessment. According to the established harvest control rules (HCR), the $SSB_{2021}/SSB_0$ ratio places the stock in the *expansion range*.

4. Projection from the current model indicated that the $SSB/SSB_0$ ratio will likely remain in the HCR expansion range, on a levelled-off trend throughout the projection period.

5. Maximum sustainable yield (MSY) was estimated at 1,728 tonnes, 6.6% lower than in the previous year's assessment, but still well above the current total allowable catch (TAC).

6. Based on the HCR, the recommendation for the toothfish longline fishery is to maintain the TAC at 1,040 tonnes, same as the previous year.

## 1. Introduction

Patagonian toothfish (*Dissostichus eleginoides*) is a large notothenioid fish found on the southern sea shelves and slopes of South America and around the sub-Antarctic islands of the Southern Ocean. It is a long-lived species (>50 years), which initially grows rapidly on the shallow shelf areas, before undertaking an ontogenetic migration into deeper waters (Collins *et al.* 2010). In Falkland Islands waters, Patagonian toothfish spawn on the slopes of Burdwood Bank at ca. 1000 m depth with a minor peak in May, and a major peak from July to August (Laptikhovsky *et al.* 2006). The eggs, larvae, and small juveniles (<10 cm TL) develop and grow in epipelagic layers of the Falkland Current, with early juveniles of 10-12 cm TL (<1 year old; Lee 2017) occurring on the Patagonian shelf at depths <100 m (Arkhipkin and Laptikhovsky 2010). Immature toothfish remain there for 3-4 years, and then, on reaching 60-70 cm TL, they migrate into deeper water over the Patagonian slope (Arkhipkin and Laptikhovsky 2010).

The Falkland Islands toothfish longline fishery began in 1992 as an exploratory fishery and became an established fishery in 1994 (Laptikhovsky and Brickle 2005). Fishing was traditionally conducted using the Spanish system of longlining (although in the beginning a few vessels used the *Mustad Autoline* system), until the 'umbrella' system was introduced in 2007. The latter system was developed to reduce the loss of hooked toothfish to depredation by cetaceans, with hooks set in clusters and an umbrella of buoyant netting set above each cluster. The umbrella floats above the hooks whilst the gear is on the seabed, but when the gear is recovered, it folds over the hooks and hooked fish, protecting it from depredation (Brown *et al.* 2010). Following initial trials in 2007, since 2008 the umbrella system has been adopted by all vessels operating in the Falkland Islands longline fishery.

Although longlining is the only fishery targeting toothfish in Falkland Islands waters, notable quantities are taken as a bycatch in finfish and calamari trawl fisheries. In the finfish fishery toothfish is a commercially valuable bycatch, while in the calamari fishery it is usually discarded, due to the small size of the specimens (20-30 cm TL). These fisheries exploit different parts of the toothfish population in different areas: longlining occurs on the slope and in deep water, finfish trawling on the shelf

primarily north and west of the Falkland Islands, and calamari trawling on the shelf south and east of the Falkland Islands (Figure 1).

This report provides an updated Bayesian age-structured stock assessment of *D. eleginoides* in Falkland Islands waters, using data through the year 2021.
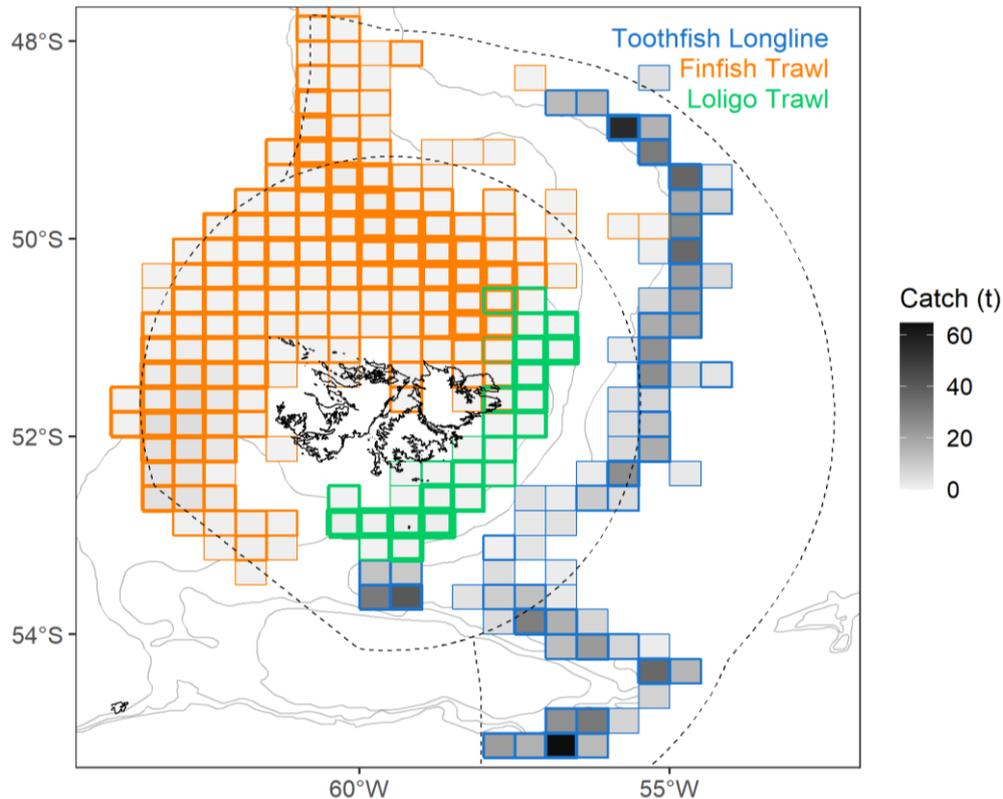


Figure 1. Spatial distribution of toothfish catch and effort by fisheries in 2021. Thickness of grid lines is proportional to the number of vessel days; greyscale is proportional to the toothfish catch biomass in tonnes.

## 1.1. Stock structure and assumptions

The stock structure of Patagonian toothfish in the southwest Atlantic is still poorly understood. On a larger spatial scale, there is well-documented genetic differentiation between toothfish found on the Patagonian Shelf and around South Georgia and South Sandwich Islands (Shaw *et al.* 2004; Rogers *et al.* 2006; Canales-Aguirre *et al.* 2018, Arkhipin *et al.* 2022). However, toothfish population structure across the Patagonian Shelf is less certain in the context of where it fits into a possible stock structure continuum, extending across possible sympatric discrete, spatially complex (e.g. metapopulation or source-sink) and panmictic population structures (Ciannelli *et al.* 2013, Hawkins *et al.* 2016, Cadrin 2020).

The existence of separate spawning populations south of Diego Ramirez Islands in Chilean waters and the eastern Burdwood Banks in Falkland Islands waters have been identified (Laptikhovsky *et al.* 2006, Arana 2009); with otolith microchemistry analysis suggesting that larvae settling on the Falkland Shelf originate from a combination of these two spatially distinct areas (Ashford *et al.* 2012). Annual spawning migrations have been inferred for Patagonian toothfish from foraging areas on the Patagonian shelf and slope located in the north and east of the Falkland Islands to spawning areas on the Burdwood Bank (Laptikhovsky *et al.* 2006). However, early satellite-tagging work undertaken in Falkland Islands waters showed high site fidelity and limited movement of adult toothfish (Brown *et al.* 2013), leading to the conclusion that the part of the stock targeted by the longline fishery (primarily older, adult individuals) is most likely confined to Falkland Islands waters.

Results from a recent genetic study suggests two independent evolutionary lineages within the '*D. eleginoides*' complex and their differentiation into two distinct taxonomic entities as separate species based on genomic and morphometric data (Arkhipkin *et al.* 2022). Results of Arkhipkin *et al.* (2022) also implied a complex level of restricted connectivity between local populations of Patagonian toothfish, specifically between fish captured from (1) southern Chile and the Falkland Islands; and (2) Falkland Islands and the high seas (North of Falkland Islands), illustrating the Falkland Islands as an intermediate zone of mixing.

Otolith shape analyses revealed high site fidelity across their distribution, and possible stock separation among early adult fish within localised areas on the Patagonian Shelf (Lee *et al.* 2018). Based on these results limited mixing appeared to occur between southern Chile and Burdwood Bank spawning areas; and feeding grounds off the continental slope and deep-sea plains to the north-east of the Falkland Islands (Lee *et al.* 2018).

Analyses of abundance distributions were undertaken to assess the spatial-temporal persistence (stability) of toothfish nursery area hotspots around the Falkland Islands and to describe their subsequent ontogenetic migration pathways into their adult deep-water habitats (Lee *et al.* 2021). Results indicate spatially discrete hotspots exhibiting high temporal variability. This variability is defined through oceanographic influence that drives larval dispersal and survival on the Shelf. Juvenile toothfish appear to follow persistent ontogenetic migrations, linking distinct recruitment areas with their respective component of the adult population on the Patagonian slope. Evidence of mixing among cohort groups during their ontogenetic migration into deep-sea adult habitats was limited, reflecting a mixed population based on oceanographically defined egg and larval dispersal.

A large-scale tag-recapture program using conventional and satellite tags was established in June 2016 (Randhawa and Lee 2016) to improve understanding of the movement patterns of toothfish within the region; and to quantify the exchange between adults on the northern and eastern slope, and the spawning grounds on the Burdwood Bank (and possible southern Chile). Analyses based on results over the first 6 years of data were used to elucidate the movement patterns as well as the extent that these are driving connectivity during the adult life-history stages of Patagonian toothfish in the region (Lee *et al.* 2022). Tagging and recapture results indicated that the majority of individuals (78%) displayed high site fidelity (<50 km). However, 10% of individuals undertook large-distance movements across oceanographic and physical boundaries. These were characterised by large (>120 cm) fish inhabiting the slope and deep-sea plains (north of 52°S) undertaking southward (direction = 150 - 240°) home-range relocations to spawning areas on the Burdwood Bank, North Scotia Ridge and southern Chile.

Otolith microstructure and trace element analysis were applied to infer time-resolved elemental profiles that reflect the early ontogeny of juvenile Patagonian toothfish from key nursery areas on the Patagonian Shelf (Lee 2022). Results revealed significant ontogenetic shifts in elemental profiles across the early life-history of Patagonian toothfish. From these, key biological benchmarks were identified, including (i) the hatch date distribution (mid-September to late October; mean = 3 October); (ii) the dispersal period of pelagic larvae until their entry onto the Patagonian shelf (0 to 40 days post-hatch); (iii) the transition phase from the shelf extent to inshore waters (<100 m depth; 40 to 75 days post-hatch); (iv) settlement into a demersal habitat (100 days post-hatch); and (v) subsequent migration into juvenile nursery areas (<150 m depth; 120 days post-hatch). Finally, these benchmarks were shown to occur significantly earlier in fish sampled in discrete nursery areas to the west of the Falkland Islands.

Overall, the results provide important considerations in terms of the complexity and protracted nature of early life-history stages, the reliance of recruitment upon stable environmental patterns and the potential for a mixed stock origin on the Patagonian Shelf. Results highlighted further monitoring and research priorities with a focus on the identification of reproductive potential, egg and larval dispersal pathways and settlement patterns of stock contingents to inform the dynamics of mixed stock origins across the Patagonian region.

Research on the stock structure of Patagonian toothfish on the Patagonian Shelf reveals complex patterns. High levels of uncertainty remain in terms of understanding the extent of connectivity during the early life-history phases of egg and larval dispersal. Further, evidence of connectivity across the region through the active migration of adults appears to occur on a relatively small scale (<8%). Current results demonstrate that the stock structure arising from the retention of mixed contingents across the Falklands Shelf remains discrete (within the Falkland Islands Conservation Zone) until adult life-history stages. Therefore, considering the currently available information, for the purpose of this assessment we assumed that there is one discrete toothfish stock present in Falkland Islands waters.

## 2. Methods

In this assessment we use an integrated statistical catch-at-age model implemented in CASAL (Bull *et al.* 2012), a general stock assessment software capable of integrating a variety of different types of input data in parameter estimation. The model assumes a single area with four distinct fisheries: Spanish-system longline, umbrella-system longline, finfish trawl and calamari trawl. Information from these fisheries covers varying time periods and areas and gives us an insight into the variety of issues that need to be addressed in toothfish stock assessment.

### 2.1. Model updates

The current assessment incorporates new data collected in 2021, including (a) catch and effort data for the umbrella-system longline fishery, (b) catch data for the finfish and calamari trawl fisheries, (c) ageing data, and (d) length frequencies and maturity data.

Besides the regular data updates, several model changes were introduced compared to the previous year's assessment following the recommendations of the Falkland Islands toothfish assessment external review (Bergh 2018) and the best practices in toothfish stock assessments found around the world. These are listed here for reference, and explained in more detail further in the text:

*Catch-per-unit-effort (CPUE) data*
- Corrected a number of erroneous database entries pertaining to longline catch reports from 2008-2011, after crosschecking them with logbook hard copies

*Age data*
- Used two separate age-length keys pertaining to longline and trawl fisheries respectively, as opposed to using a single pooled age-length key

*Model setup*
- Used a fixed input value for natural mortality, as opposed to estimating it within the model
- Decreased the process error associated with Spanish- and umbrella-system longline fishery CPUE datasets

*New model output*
- Included objective function contributions in the MPD model run to the report

## 2.2. Data

Several datasets were used as information in the assessment, either as observations or input parameters (Table 1). Observations appear in the objective function and are used to fit the model - in our case these include two CPUE and four catch-at-age time series from the commercial fisheries, and two catch-at-age time series from the research surveys. Input parameters were estimated outside the model, and then treated as fixed parameters within the model (e.g. von Bertalanffy growth coefficients). Input parameters were assumed known without error.

Table 1. Data used in the stock assessment.

| Data type | Data | Time series |
|---|---|---|
| Observations | **CPUE** | |
| | *Spanish-system longline* | 1996-2007 |
| | *umbrella-system longline* | 2007-2021 |
| | **Catch-at-age** | |
| | *Spanish-system longline* | 1992, 1994-2009 |
| | *umbrella-system longline* | 2007-2021 |
| | *finfish trawl* | 1988-1992, 1994, 1997-1999, 2002-2020 |
| | *calamari trawl* | 2008-2021 |
| | *groundfish survey* | 2015-2021 |
| | *calamari pre-season survey* | 2015-2021 |
| Input parameters | **Removals** | |
| | *Spanish-system longline* | 1992-2010, 2013 |
| | *umbrella-system longline* | 2007-2021 |
| | *finfish trawl* | 1987-2021 |
| | *calamari trawl* | 1987-2021 |
| | *groundfish survey* | 2015-2021 |
| | *calamari pre-season survey* | 2015-2021 |
| | **Length-weight relationship** | |
| | *all fisheries combined* | 1989-2021 |
| | **Von Bertalanffy growth** | |
| | *all fisheries combined* | 2014-2021 |
| | **Maturity** | |
| | *all fisheries combined* | 1988-2021 |

*Catch-per-unit-effort (CPUE)*
Although CPUE data were available for all four fisheries, only longline CPUE was used as a relative abundance index. This is motivated by the inconsistency of the toothfish CPUE in trawl fisheries, where this species is not targeted, and its bycatch may change due to factors other than stock abundance (e.g. fisheries are switching targets or areas). The longline CPUE data were treated separately for Spanish- and umbrella-system longline, according to the documented difference in the toothfish CPUE between these two fishing techniques (Brown *et al.* 2010). During the transition period from the Spanish- to umbrella-system (2007-2009), both techniques were used concurrently, sometimes by the same vessel on the same day. Catch reports from this period were inspected and showed a gradual transition between the two systems. The proportion of daily hooks set as an umbrella-system started low and gradually increased to ~50%, at which point there was a rapid switch to full (100%) umbrella-

system (however, timing differed between vessels). Since we use data aggregated by day in our analysis, daily catch reports with both types of lines set by the same vessel needed to be resolved; we decided to assign daily catch reports with >90% of hooks set in an umbrella-system to the corresponding fishery, and to exclude the remaining 'mixed' daily catch reports from the analysis (with ~10-50% of hooks set in an umbrella-system), as it was not clear how to correctly classify them.

For the Spanish-system longline, data were inspected and 95 daily catch reports pertaining to remote areas (outside the region 47° W - 70° W and 40° S - 57° S) were removed. These records belong exclusively to the early years of the fishery (1998-2002) when presumably more exploratory fishing took place. Also, in this period vessels that fished in Falkland Islands waters would sometimes report to FIFD their catches taken in other remote areas as well.

For the umbrella-system longline, data selection followed the same reasoning outlined in the previous year's assessment. In order to avoid introducing bias to the CPUE estimates, only the catch reports belonging to Falkland Islands flagged vessels were used. Since the onset of the umbrella-system the fishing was predominantly done by a single Falkland Islands vessel (*CFL Gambler*, replaced by *CFL Hunter* in 2017), assisted occasionally by one or two chartered Chilean vessels. None of the chartered vessels fished in Falkland Islands waters in more than two years since 2007, and their CPUE data were inconsistent. Moreover, at least one of these vessels had restrictions imposed on its fishing practice (e.g. limit on the number of fishing days in the 'best' fishing grounds), which were not in place for the Falkland Islands vessel. All of this led to a conclusion that the CPUE would be more representative as an index of abundance if only Falkland Islands vessels data were used. With a similar goal, data from dedicated 'tagging trips' and from longline sets at depths <600 m were removed from the analysis. Tagging trips were excluded because part of the actual catch was not reported (corresponding to the tagged and released fish), leading to a biased, lower estimates of CPUE. Fishing in shallow waters was excluded because commercial longlining is prohibited at depths <600 m, and the corresponding sets were experimental fishing aiming to collect brood stock for the toothfish rearing facility.

For the selected catch reports, CPUE data were calculated for each fishing day as reported toothfish catch in kg per hook (Spanish-system) or kg per umbrella (umbrella-system). Finally, CPUE was standardised using a generalised linear mixed modelling approach (GLMM), providing a time series of CPUE values which were assumed relative abundance indices (Appendix 1). Observation error of the CPUE indices was accounted for in the assessment model by using the coefficient of variation (CV) estimates obtained directly from a GLMM. To account for any additional variance on top of observation error, which may arise from the differences between model simplifications and real-world variation, a process error CV = 0.1 was added. It should be noted that this value is lower than used in the previous assessment (CV = 0.2); the new value up-weights the CPUE data and prioritises good model fit to the CPUE data, as advised by Francis (2011). The CPUE indices were assumed to be log-normally distributed about the model-predicted vulnerable biomass, via a catchability parameter.

*Catch-at-age (CAA)*

As in the previous assessment, CAA distributions were treated separately for each of the four fisheries and two research surveys. The longline CAA data had to be split between Spanish- and umbrella-system fishery in the same way as CPUE data (this is a model requirement), while the trawl data were split between finfish and calamari fisheries due to the differences in legal net mesh size and fishing grounds, leading to distinct CAA distributions. Same as in the previous year, pre-2008 CAA data for calamari trawl fishery was excluded from the analysis. The poor model fit to these data and the potential reasons behind it have already been reported (Skeljo and Winter 2020); in brief, pre-2008 data were few, with more toothfish sampled in 2008 alone then in the previous 15 years combined. We suspect this bias reflects different sampling protocols pre- and post-2008, i.e. different levels of attention given to accounting for the juvenile toothfish, which can be difficult to distinguish from certain other species in the juvenile stage. Therefore, we decided to treat pre-2008 data as questionable and exclude them from the model. The CAA data from two research trawl surveys

conducted in parallel by FIFD, groundfish survey and calamari pre-season survey, have been treated separately as they cover different (although partially overlapping) areas.

Toothfish ageing data used in the stock assessment were restricted to the otoliths sampled in 2014-2021. Otoliths collected in 2021 have been partially processed, with ~300 age readings available at the time of the assessment. All the otoliths from the period 2014-2021 were processed at FIFD, and the corresponding age readings are the most reliable toothfish age estimates available at the time of this assessment (Lee 2015, 2016, 2017, 2018, 2019, 2020). A total of 2,914 toothfish age estimates belonging to longline fisheries and 2,859 age estimates belonging to trawl fisheries were used to construct two corresponding age-length keys: $ALK_{LONGLINE}$ and $ALK_{TRAWL}$. Next, 162,888 toothfish length measurements (sampled randomly by the observers from commercial catches in 1988-2021 or during the research surveys in 2015-2021) were split between the four corresponding fisheries and two research surveys and age was assigned to each fish by conditional probability of the appropriate age-length key, i.e. $ALK_{LONGLINE}$ for Spanish- and umbrella-system longline fisheries and $ALK_{TRAWL}$ for calamari fishery, finfish fishery and both research surveys. Ages ≥31 years were assigned to a plus group. Finally, CAA datasets were constructed as fish counts per age class for each year and fishery, and then expressed as catch proportions-at-age. Ageing error was accounted for in the model by deriving an error misclassification matrix from a normal distribution with CV = 0.1. The CAA data were assumed independently multinomially distributed about the model-predicted CAA.

An important consideration in integrated models is to ensure that the observations are given appropriate weights in the objective function (Francis 2011), and for the CAA data this was achieved by estimating effective sample size for each fishery and year combination. The effective sample sizes were estimated by a two-stage weighting approach: in stage 1 the weights appropriate for the observation error are assigned outside the model, and in stage 2 those weights are adjusted within the model to allow for the process error (Francis 2011). In our assessment, in stage 1 the effective sample sizes were calculated based on the data fit to the multinomial distribution, using the function *neff.obs* from R package *DataWeighting* (Francis 2013). The initial model fit was then run, and the information from that run was used in the stage-2 adjustment of the effective sample sizes, multiplying them by a weighting factor calculated as:

$$w_j = 1/var_i\left[\left(O_{ij} - E_{ij}\right)/\left(v_{ij} - N_{ij}\right)^{0.5}\right]$$

where $N_{ij}$ is the number of multinomial cells, $O_{ij}$ is the observed proportions for age class $i$ in year $j$, $E_{ij}$ is the expected proportions, and $v_{ij}$ is the variance of the expected age distribution (Method TA1.8 in Table A.1, Francis 2011). The model was then run again with the adjusted effective sample sizes. The most important consequence of the described procedure was down-weighting of CAA data, as otherwise large sample sizes determined as the number of fish measured would give it disproportionate weight, potentially swamping CPUE data in the analysis (Francis 2011).

*Removals*

Total removals were calculated by adding three distinct catch components: (a) reported catches in Falkland Islands waters, (b) catches taken by *Illegal, Unreported and Unregulated* (IUU) fishing in Falkland Islands waters, and (c) catches lost to undetected whale depredation.

Catch reports from all available years for the four fisheries and two research surveys were used, going back to 1987. Catch reports that list the fishing effort as *trawl and jig time* (listed under various licenses until 1996) were considered trawls if the unit of effort was ≤1440, the number of minutes in 24 hours. Trawls catch reports without licence information were considered calamari trawls if the dominant species in the catch was *Doryteuthis gahi*. Otherwise, they were considered finfish trawls.

The IUU fishing is inherently difficult to estimate (Pitcher *et al.* 2002, Ainsworth and Pitcher 2005), and no reliable information specific to the Falkland Islands waters was found. Therefore, we utilized the data for the *Antarctic region* from *Table 2* in Agnew *et al.* (2009), which give estimates of IUU catches as a percentage of reported catch in 1980-2003. For years since 2003, we took grey-

literature estimations (e.g. CCAMLR 2010) that IUU fishing in the southern oceans has decreased significantly and assumed IUU catches to be 5% of the reported catch in Falkland Islands waters. The same IUU data was used in the previous year's baseline assessment model.

Whale depredations are included in longline catch reports when they are evident as toothfish hauled up damaged or destroyed by bite-marks. However, toothfish taken entirely by whales before hauling are not seen and not accounted for in the catch reports. In order to quantify this cryptic depredation, Winter and Pompert (2016) developed a model-differencing algorithm between catches predicted from all observer-monitored longlines, and catches predicted only from observer-monitored longlines without sign of whale depredation. Models included parameters longline position, fishing depth, year, month, numbers of hooks and soak time. The model-difference could then be projected onto all commercial longlines to estimate the amount of toothfish lost. The algorithm has recently been revised by modelling Spanish-system and umbrella-system longline fishing separately, as for stock assessment, and by projecting the depredation ratios of the models rather than the models themselves, which improved the avoidance of outlier extrapolations.

The above-mentioned catch components (reported catches, IUU catches and whale depredation) were added together into total removals and used in the assessment model run. Since removals are treated as input parameters and not as observations in CASAL, they were assumed known without error.

*Length-weight relationship*
The length-weight relationship was calculated as $W = aL^b$, based on the length and weight measurements of 37,931 toothfish sampled randomly by the observers from commercial catches in 1989-2021. Individual fish weights were expressed in tonnes (to be compatible with the removals in CASAL), lengths in cm, and parameters *a* and *b* are summarized in Table 2.

*Von Bertalanffy growth*
The length-at-age relationship was described by the von Bertalanffy growth model $L = L_{inf}\left(1 - e^{-k(age-t_o)}\right)$, based on age estimates and length measurements of 5,773 toothfish sampled randomly by observers from commercial catches in 2014-2021. Parameters $L_{inf}$, $k$ and $t_0$ are summarized in Table 2.

*Natural mortality*
Natural mortality (M) was assumed to be 0.165 (Payne *et al.* 2005) and assumed constant across all age classes. A fixed M value from the literature was used to prevent M from gaining exaggerated values unlikely for a long-lived species like toothfish. Over recent previous assessments, model-estimated M had kept increasing. Since using a fixed M value presents an important change to the model setup, model sensitivity to different assumed M values was explored and presented further in the report.

*Maturity*
Maturity-at-age vector was based on the maturity stage data estimated by the observers for 160,537 toothfish, sampled randomly from commercial catches in 1988-2021. Maturity was scored on an 8-point scale, and toothfish are considered mature from stage 3 (Laptikhovsky *et al.* 2006). However, mature toothfish occasionally enter a 'resting' stage, and they can skip annual spawning (Collins *et al.* 2010, Boucher 2018). While in this resting stage, the gonads look very similar macroscopically to stage 2 gonads that are considered immature. Analysis of the available maturity data strongly indicated that due to this, some older fish were erroneously assigned as immature (stage 2) when observed. To address this inaccuracy, a generalized additive model (GAM) was used to predict the expected number of older fish at stage 2, and the maturity data were corrected accordingly, as outlined in Farrugia and Winter (2018). Finally, instead of the more typical logistic function, the maturity ogive was fitted using

GAM, resulting in a maturity-at-age vector with proportion of mature fish in each age class from 1 to 31+ (plus group). Parameters of the maturity-at-age vector are summarized in Table 2.

Table 2. Biological input parameters assumed in the model.

| Relationship | Parameter | Value | | | | | |
|---|---|---|---|---|---|---|---|
| Length-weight | a (t·cm$^{-1}$) | 5.82e-9 | | | | | |
| | b | 3.128 | | | | | |
| Von Bertalanffy growth | $L_{inf}$ (cm) | 173.730 | | | | | |
| | k (y$^{-1}$) | 0.066 | | | | | |
| | $t_0$ (y) | -2.368 | | | | | |
| | CV | 0.151 | | | | | |
| Natural mortality | M (y$^{-1}$) | 0.165 | | | | | |
| Maturity *(proportion mature at age)* | Age 1 | 0 | Age 12 | 0.412 | Age 23 | 0.570 |
| | Age 2 | 0.007 | Age 13 | 0.436 | Age 24 | 0.583 |
| | Age 3 | 0.040 | Age 14 | 0.457 | Age 25 | 0.596 |
| | Age 4 | 0.079 | Age 15 | 0.477 | Age 26 | 0.610 |
| | Age 5 | 0.123 | Age 16 | 0.494 | Age 27 | 0.624 |
| | Age 6 | 0.172 | Age 17 | 0.508 | Age 28 | 0.639 |
| | Age 7 | 0.223 | Age 18 | 0.519 | Age 29 | 0.655 |
| | Age 8 | 0.272 | Age 19 | 0.529 | Age 30 | 0.671 |
| | Age 9 | 0.315 | Age 20 | 0.538 | Age 31+ | 0.688 |
| | Age 10 | 0.353 | Age 21 | 0.547 | | |
| | Age 11 | 0.385 | Age 22 | 0.558 | | |

## 2.3. CASAL model setup

*Population dynamics*

Toothfish population dynamics were described by an age-structured model, with age classes from 1 to 31+ years, the last one being a plus group. It is a single-sex, single-area model, with the annual cycle split into three time-steps. Recruitment, fishing mortality from all concurrent fisheries, and the first half of the year's natural mortality occur in time step 1; spawning and the second half of natural mortality in time step 2; and ageing in time step 3. Since both fishing and natural mortality occur in time step 1, the process was to apply half time step's natural mortality, then fishing mortality instantaneously, then the remaining half of the time step's natural mortality.

Recruitment to the population was calculated by multiplying average recruitment ($R_0$) with estimated year class strength multipliers (YCS) and a stock-recruitment relationship. Stock-recruitment was assumed to follow a Beverton-Holt relationship:

$$R = \frac{SSB}{SSB_0}\bigg/\left(1 - \frac{5h-1}{4h}\left(1 - \frac{SSB}{SSB_0}\right)\right)$$

where *R* is the recruitment, *SSB* is the spawning stock biomass, *SSB$_0$* is the pre-exploitation equilibrium spawning stock biomass, and *h* is the steepness parameter, defined as the fraction of recruitment from the unfished population when the spawning stock biomass declines to 20% of its unfished level (Mangel *et al.* 2013). Recruitment was fixed, rather than being estimated, as suggested for example by He *et al.* (2006) and Kenchington (2014), and the steepness parameter was set to the commonly used reference value *h* = 0.75 (Punt 2005, Brandão and Butterworth 2009, Dunn and Hanchet 2010, Mormede *et al.* 2014, Dunn and Parker 2019).

The initial year in the model was set to 1987, the first year of recorded data by the FIFD, and it was run up to 2021. Projections from the model extended for another 35 years, up to 2056. Conditions in the initial year were assumed to be an equilibrium age structure at an unexploited equilibrium biomass.

*Estimation method*

Model parameters were estimated by minimising the total objective function, which is the sum of the negative log-likelihoods from the observations, the negative-log Bayesian priors, and the penalties applied to constrain the parameterisations (see below). The estimated parameter values presented in the report are MPD (mode of the posterior density) point estimates (Bull *et al.* 2012).

To estimate the joint posterior distribution of the parameters in a Bayesian analysis, the Monte-Carlo Markov Chain (MCMC) method was used. Starting point of each chain was set to the corresponding MPD, length of the burn-in period was set to 100,000 iterations, and from the next 1,000,000 iterations every 1000$^{th}$ value was taken. The resulting 1,000 values represent a systematic sample from the Bayesian posterior distribution for the parameter of interest. Chains were investigated for evidence of non-convergence using trace plots, chain autocorrelation plots, and single-chain convergence tests of Geweke (1992) and the stationarity and half-width tests of Heidelberger & Welch (1983).

*Estimated parameters*

The parameters estimated by the model, their priors, starting values and bounds are given in Table 3, and detailed further in the text.

Table 3. Number of parameters (N), priors, starting values and bounds for the free parameters estimated by the model

| Estimated parameter/s | | N | Prior | Start value | Lower bound | Upper bound |
|---|---|---|---|---|---|---|
| SSB$_0$ | | 1 | uniform-log | 40,000 | 10,000 | 100,000 |
| YCS | | 34 | lognormal | 1 | 0.001 | 20 |
| Selectivity $_{LLH}$ | $a_{50}$ | 1 | uniform | 10 | 1 | 50 |
| | $a_{to95}$ | 1 | uniform | 5 | 0.05 | 50 |
| Selectivity $_{LLU}$ | $a_{50}$ | 1 | uniform | 10 | 1 | 50 |
| | $a_{to95}$ | 1 | uniform | 5 | 0.05 | 50 |
| Selectivity $_{FIN}$ | $a_1$ | 1 | uniform | 2 | 1 | 50 |
| | $S_L$ | 1 | uniform | 1 | 0.05 | 50 |
| | $S_R$ | 1 | uniform | 2 | 0.05 | 500 |
| Selectivity $_{LOL}$ | | 8 | uniform | 0.5 | 0 | 1 |
| Selectivity $_{RFIN}$ | | 6 | uniform | 0.5 | 0 | 1 |
| Selectivity $_{RLOL}$ | | 6 | uniform | 0.5 | 0 | 1 |
| q $_{LLH}$ | | 1 | uniform-log | - | 1e-9 | 0.1 |
| q $_{LLU}$ | | 1 | uniform-log | - | 1e-9 | 0.1 |

LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-recruitment survey.

SSB$_0$ is the estimated pre-exploitation equilibrium spawning stock biomass, defined as the spawning stock biomass that would exist with average recruitment in the absence of fishing. For SSB$_0$, a uniform-log prior was used (Hillary *et al.* 2006, Dunn and Hanchet 2010, Mormede *et al.* 2014, Dunn 2019). Year class strength multipliers (YCS) were estimated for the period 1986-2020 (34 parameters, one for each year), using the *Haist parameterisation* to make the YCS parameters average to 1 over the period 1986-2016 (for the Haist method description see Bull *et al.* 2012). For YCS, a lognormal prior with $\mu$ = 1 and CV = 1.1 was used (Constable *et al.* 2006a, 2006b). Catchability coefficients (q) were estimated

for the two CPUE series separately. They were treated as 'nuisance' parameters (default in CASAL), so no starting values had to be provided. For q's, log-uniform priors were considered appropriate (Hillary *et al.* 2006).

Selectivity-at-age was estimated separately for each fishery and survey to reflect the different age distributions of fish in the catch. Three types of selectivity ogives were used: logistic for longline fisheries, double-normal for finfish trawl fishery, and CASAL *allvalues* for calamari trawl fishery and both surveys. Logistic ogive is defined by two parameters: $a_{50}$ (age at 50% selectivity) and $a_{to95}$ (difference in age at 50% and 95% selectivity), where the value of selectivity at age *x* is given by

$$f(x) = 1/\left[1 + 19^{(a_{50}-x)/a_{to95}}\right].$$

Double-normal ogive is defined by three parameters: $a_1$ (the mode), $S_L$ (increasing left-hand limb shape parameter) and $S_R$ (decreasing right-hand limb shape parameter), where the value of selectivity at age *x* is given by

$$f(x) = 2^{-[(x-a_1)/S_L]^2}, \quad (x \leq a_1)$$
$$= 2^{-[(x-a_1)/S_R]^2}, \quad (x > a_1).$$

The *allvalues* ogive is defined by one selectivity parameter for each age class, meaning that for our CAA data we would have 31 parameters. Since negligible numbers of toothfish older than 8 years were recorded in the calamari fishery, selectivity parameters were estimated only for ages 1-8 and set to zero for the remaining age classes, to reduce the number of estimable parameters. The same procedure was applied to the research surveys, but for ages 1-6. The empirical *allvalues* ogive was used for calamari trawl fishery and research surveys because standard selectivity curves, such as logistic or double-normal, could not fit well the CAA patterns observed in the data, with the highest proportions in the catch corresponding to the lowest age classes (descending ogive).

Selectivities were assumed to remain constant throughout the modelled period. For all selectivity parameters uniform priors were used (Dunn and Hanchet 2010, Mormede *et al.* 2011, 2013, 2014). It is important to note that what we term 'selectivity' is a combination of *gear selectivity* and *availability* of the fish to the gear (Candy and Constable 2008). For example, trawl gear selectivity most likely doesn't decrease with toothfish age, but the fish availability does, as older individuals leave the trawling grounds for deeper waters. This is the reason toothfish selectivity in trawl fisheries was described by double-normal, instead of logistic ogive. In this report we use the term selectivity because it is consistent with CASAL terminology, but it should be interpreted as *vulnerability*.

*Penalties*
Besides the observations and priors, final components of the objective function are penalties. three types of penalties were included in the model: catch limit penalty, vector average penalty and ogive smoothing penalty. Catch limit penalty was applied to each fishery, to ensure that the model doesn't estimate abundances so low that the recorded removals could not have been taken. Vector average penalty was used to encourage YCS to average to 1. Ogive smoothing penalty was used to encourage *allvalues* selectivity ogives to be like a fourth-degree polynomial, i.e. smooth in appearance. Penalty multipliers were set to 100 for catch limits and 20 for YCS vector average and ogive smoothing penalty (for details on penalty calculations see Bull *et al.* 2012).

*Yield calculations*
MSY was calculated by projecting the estimated current stock status into the future, under a constant hypothetical catch split between the fisheries. For the yield calculations, recruitment for 2015-2056 was assumed to be log-normally distributed with standard deviation $\sigma_R = 0.6$ (Dunn and Hanchet 2006, Mormede *et al.* 2011, 2013, 2014). The future toothfish catch split between fisheries was assumed according to the recent catch history and the current longline catch quota: Spanish-system longline (0 t; 0%), umbrella-system longline (1,040 t; 75.9%), finfish trawl (300 t; 21.9%) and calamari trawl (30 t; 2.2%).

# 3. Results

*Model fits*

Diagnostics plots of the model fits to the different observation datasets are provided in [Appendix 2](). The model fit to the standardized CPUE data for the umbrella-system longline was good, with 95% CI of the observations and estimates overlapping in all analysed years. The fit to the Spanish-system data was slightly worse, underestimating observed CPUE in the earlier years and overestimating it in the later years of the fishery; however, the model adequately captured the declining overall trend (Figure A.3). Corresponding trends in normalised residuals for both longline fisheries are shown in Figure A.4.

The model fit to the catch proportion-at-age data was very good for all four fisheries and both research surveys (Figures A.5 - A.10). The corresponding residual bubble plots show no clear patterns, with the possible exception of longline fisheries, where the model tends to slightly overestimate the proportion of 1-3 year old fish and slightly underestimate the proportions of 4-6 year old fish (Figure A.11). The model fit to the observed mean toothfish age was good in all cases except the Spanish-system longline fishery (Figure A.12).

Likelihood profiles were carried out by fixing $SSB_0$ over a range of plausible values (15,000 - 60,000 t), while the remaining parameters were estimated. Only the Spanish-system CPUE observations strongly favoured the lower biomass estimates; CAA observations for umbrella-system longline and calamari trawl fishery suggested values close to the current estimate, while the remaining observations found higher biomass estimates more likely (Figures A.13, A.14).

MCMC trace plots showed no evidence of lack of convergence in most of the estimated parameters, with notable exception of selectivity parameters for calamari trawl fishery and both research surveys (all associated with the *allvalues* descending selectivity ogive) (Figure A.15). The convergence test of Geweke (1992) and the Heidelberger & Welch (1983) stationarity and half-width tests also suggested the failure to converge for some of these selectivity parameters. Autocorrelations in the MCMC samples for mentioned selectivity parameters were high, indicating slow mixing in MCMC chains (Figure A.16).

*Model estimates*

The key output parameters estimated by the stock assessment model are summarised in Table 4, and detailed further in the text. Contributions to the objective function of each dataset, prior and penalty, are provided in [Appendix 3]().

Table 4. Key output parameters estimated by the model.

| Parameter | MPD value | MCMC 95% CI |
|---|---|---|
| $SSB_0$ | 25,691 t | 22,729 - 35,422 t |
| $SSB_{2021}$ | 12,432 t | 9,485 - 21,947 t |
| $SSB_{2021}/SSB_0$ | 0.484 | 0.416 - 0.626 |
| MSY | 1,728 t | 1,529 - 2,383 t |

The MPD estimates of the initial spawning stock biomass ($SSB_0$) and the current spawning stock biomass ($SSB_{2021}$) were 10.9% and 12.4% higher than in the previous year's assessment ($SSB_{0\ in\ 2020}$ = 23,169 t, $SSB_{2020}$ = 11,056 t), while the SSB ratio in the final year ($SSB_{2021}/SSB_0$) remained almost the same ($SSB_{2020}/SSB_{0\ in\ 2020}$ = 0.477). According to the existing harvest control rules (HCR) (Farrugia and Winter 2018, 2019), the current $SSB_{2021}/SSB_0$ of 0.484 places the stock in the *expansion range*. The estimated historical SSB trend is shown in Figure 2, and the detailed HCR decision matrix used to manage Falkland Islands longline toothfish fishery is given in [Appendix 4]().
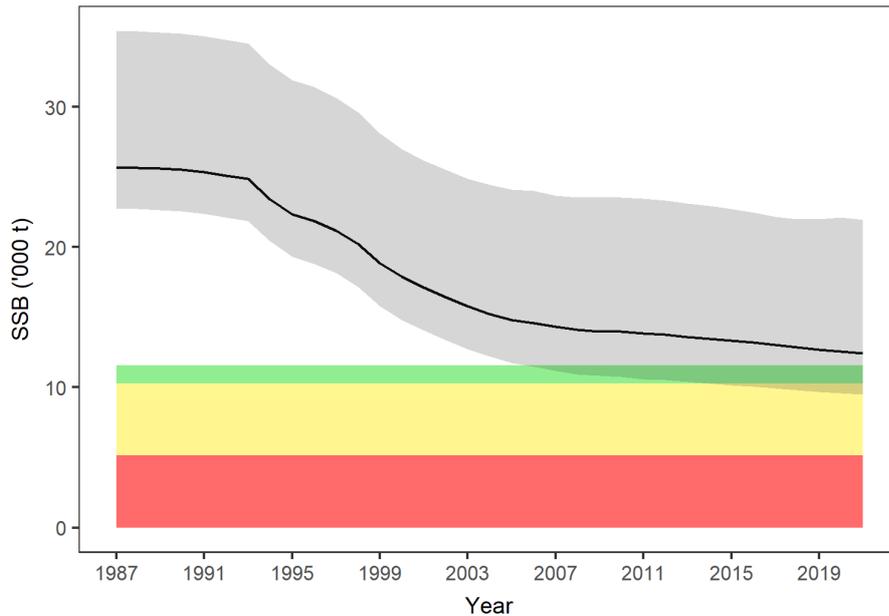
Figure 2. MPD estimate of the historical spawning stock biomass trajectory (black line); shaded area denotes MCMC 95% credible intervals of the model fit. Harvest control rule ranges are colour coded for reference: *target range* in green ($SSB/SSB_0$ = 0.45-0.40), *trigger range* in yellow ($SSB/SSB_0$ = 0.40-0.20) and *closure range* in red ($SSB/SSB_0 < 0.20$).

Maximum sustainable yield (MSY), estimated under the assumption of a constant future catch partition, was 6.6% lower than in the previous assessment ($MSY_{2020}$ = 1,850 t).

MCMC posterior distributions of $SSB_0$ and $SSB_{2021}/SSB_0$ displayed positive skewness (Figure 3) suggesting that the model is more strongly constrained to the lower limit and resulting in asymmetrical 95% credible intervals.
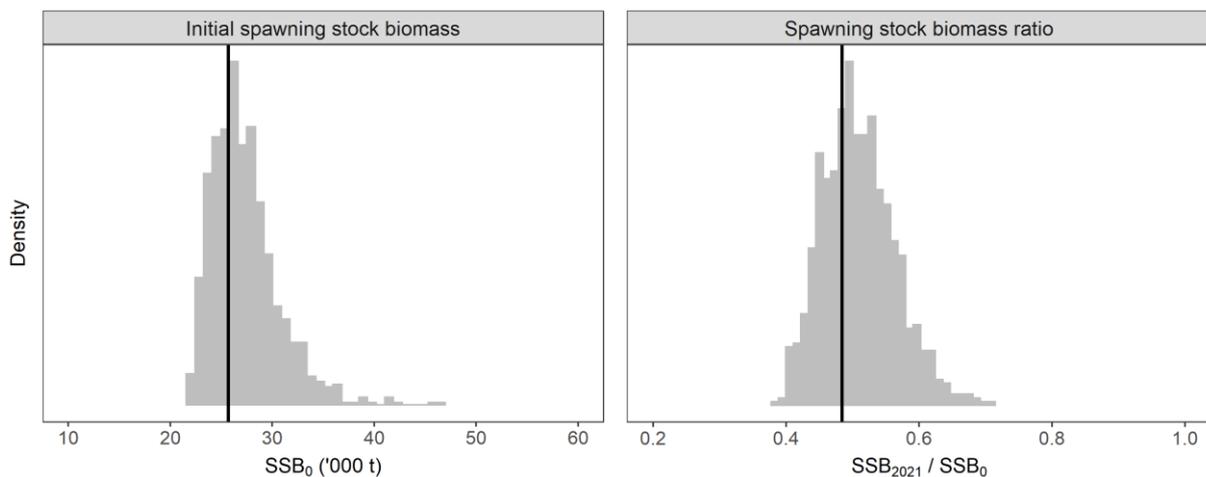


Figure 3. MCMC samples from posterior distribution of initial spawning stock biomass ($SSB_0$) and current spawning stock biomass ratio ($SSB_{2021}/SSB_0$); MPD point estimates are added as a reference (vertical black lines).

The estimated selectivity ogives appeared reasonable, showing the distinct differences in how the longline and trawl fisheries interact with the stock (Figure 4). The calamari trawl fishery catches the youngest fish, as a combination of fishing in shallower waters (=young fish is available) and using small mesh size (=low gear selectivity), which results in the descending right limb selectivity ogive with maximum selectivity for 1-year old fish. Finfish trawl fishery has domed selectivity with maximum for 2-year old fish, and lower selectivity for younger (=presumably escapes due to the gear selectivity) and older fish (=unavailable at trawling grounds). As could be expected, the two longline fisheries have

almost identical selectivity curves, catching predominantly older fish available in deeper waters. Selectivity of both research surveys closely resembles the calamari trawl fishery selectivity, which is expected as survey samples are collected using trawls with small cod-end mesh size, comparable to the commercial calamari fishery.

For the selectivity ogives with descending right limb (trawl fisheries and research surveys), MCMC 95% credible intervals were notably asymmetrical, and in the case of surveys, very wide. This is a consequence of previously mentioned MCMC convergence issues for *allvalues* selectivity ogive parameters. In general, right-hand limb descending ogives often cause convergence issues in CASAL (Dunn 2013, SAERI CASAL workshop, personal communication).
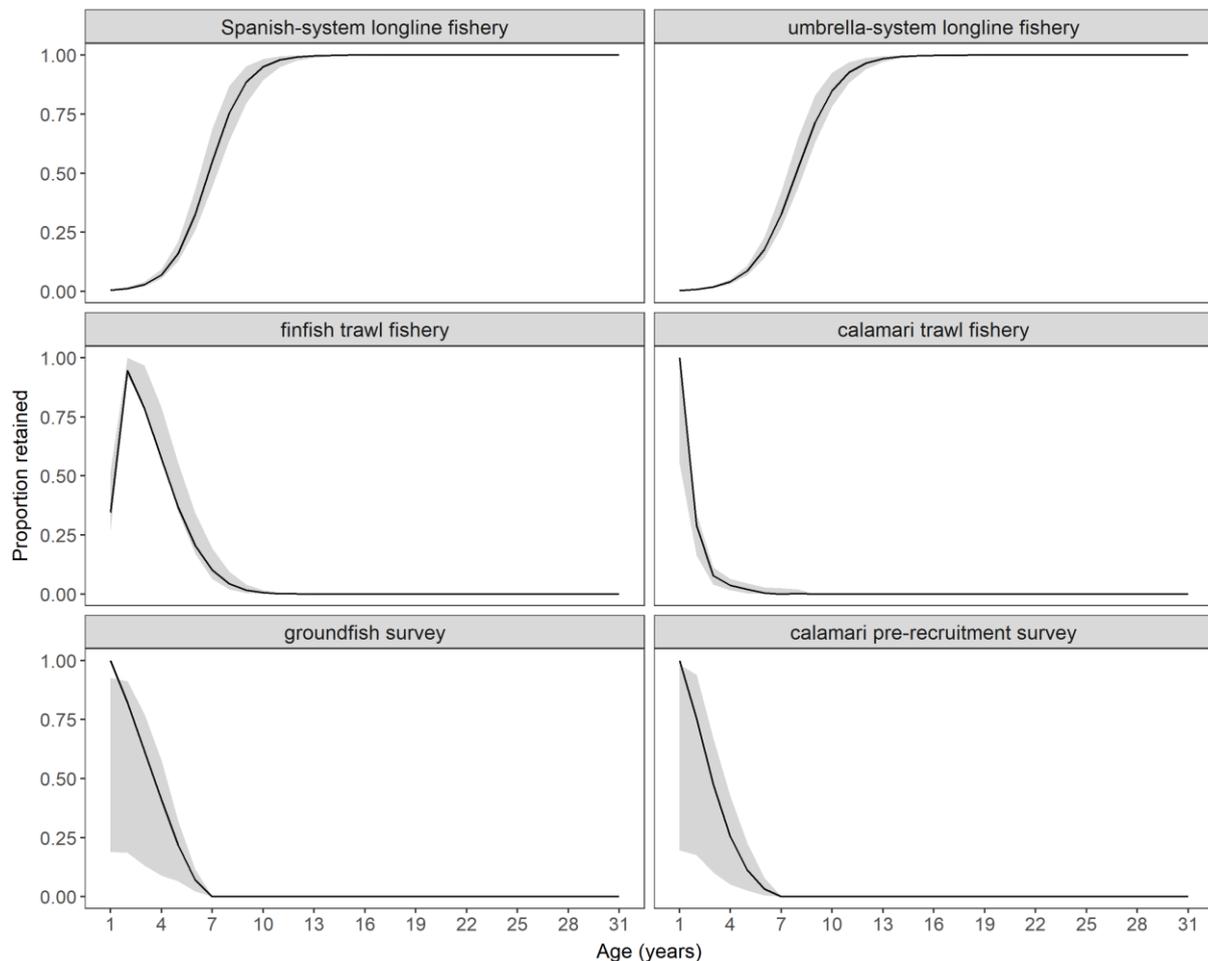


Figure 4. MPD estimates of selectivity ogives for four fisheries and two surveys (lines); shaded areas denote MCMC 95% credible intervals of the model fit.

Year class strength estimates for the most recent years (2015-2020) corresponded well to the CAA data from both research surveys introduced to the model, as well as recruitment estimates independent of the model (Lee, FIFD, personal communication), with strong recruitment peak in 2014/2015, followed by years of low recruitment (Figure 5). No independent survey data was available for the earlier years, making it difficult to confirm the model estimated YCS trend.
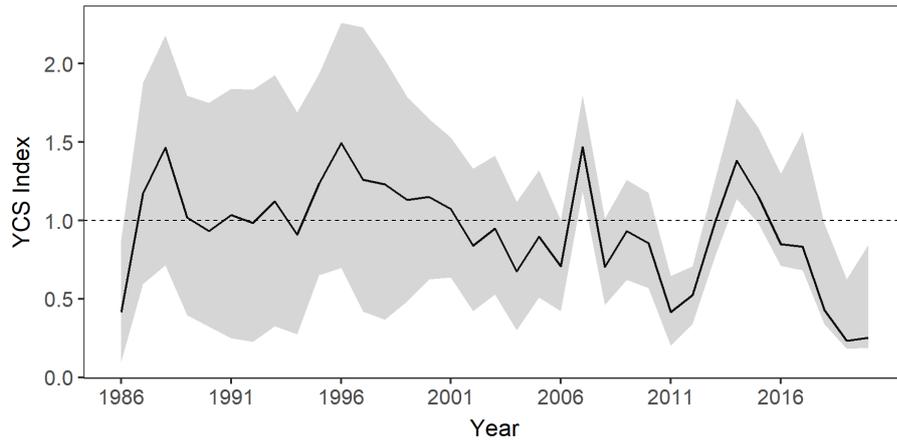
Figure 5. MPD estimates of year-class strengths in 1986-2020 (solid black line); shaded areas denote MCMC 95% credible intervals of the model fit.

*Model projections*

The future trend of $SSB/SSB_0$ was projected based on 5000 simulations, with random lognormal recruitment from 2015-2056 and constant annual catches from 2022-2056 (umbrella-system longline 1,040 t, finfish trawl 300 t, calamari trawl 30 t) (Figure 6). The median $SSB/SSB_0$ ratio was estimated to remain in the HCR *expansion range*, on a levelled-off trend throughout the projection period. The probability of $SSB/SSB_0$ ratio falling below existing management thresholds, corresponding to the upper bounds of HCR ranges, is shown in Figure 7; the highest probability of falling below 0.45, 0.40 and 0.20 thresholds during the projection period was estimated at ~39%, ~25% and <0.01%, respectively.
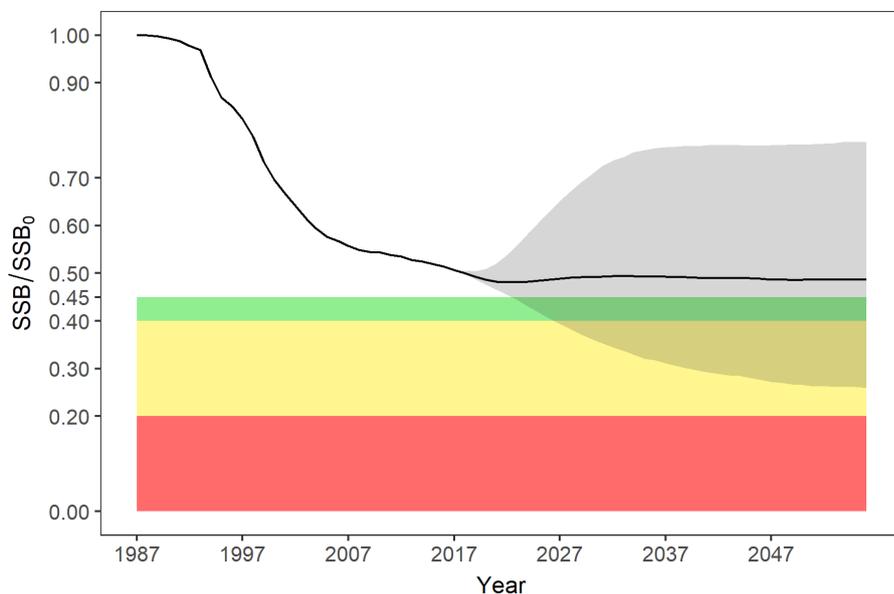


Figure 6. Projected $SSB/SSB_0$ trend under the assumption of random lognormal recruitment from 2015-2056 and constant annual catches from 2022-2056. Black line is the median, and shaded area denotes 95% confidence intervals of the projection. Harvest control rule ranges are colour coded for reference: *target range* in green ($SSB/SSB_0$ = 0.45-0.40), *trigger range* in yellow ($SSB/SSB_0$ = 0.40-0.20) and *closure range* in red ($SSB/SSB_0$ < 0.20).
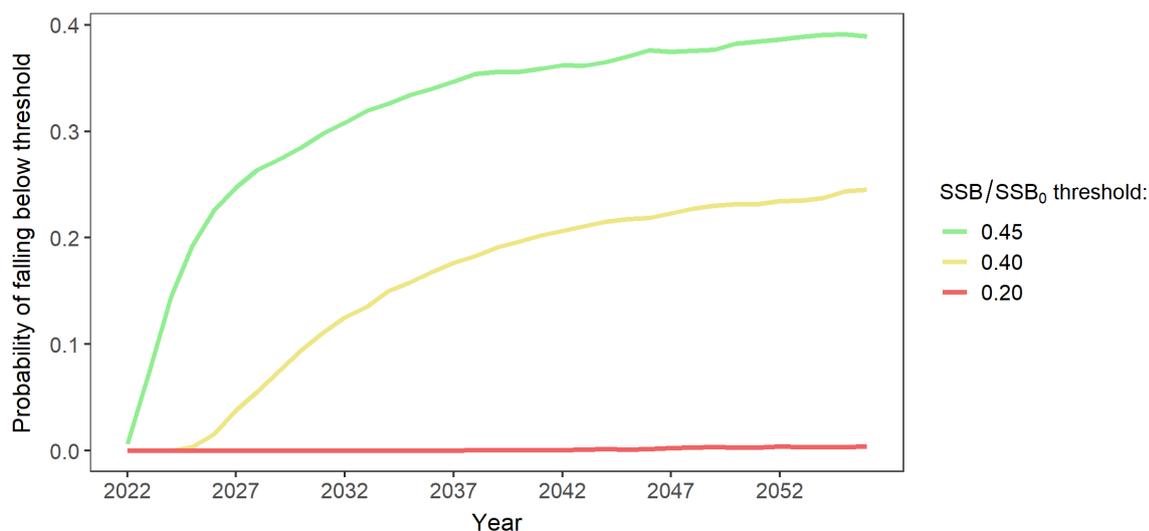
15

Figure 7. Probability of stock falling below designated $SSB/SSB_0$ management thresholds; based on 5000 projections.

*Sensitivity analyses*

The results of different sensitivity trials are summarised in Table 5, with the current model included as a base-case for the reference. Only the key estimated parameters are given here ($SSB_0$, $SSB_{2021}$ and $SSB_{2021}/SSB_0$), as they form the basis of harvest control rules and have a direct effect on the management decisions.

Table 5. MPD estimates of key assessment parameters ($SSB_0$, $SSB_{2021}$ and $SSB_{2021}/SSB_0$) obtained from different sensitivity trials. All biomass estimates are given in tonnes.

| Model run | $SSB_0$ | $SSB_{2021}$ | $SSB_{2021}/SSB_0$ |
|---|---|---|---|
| Base-case | 25,691 | 12,432 | 0.484 |
| **Alternative natural mortality** | | | |
| M = 0.130 | 35,421 | 16,995 | 0.480 |
| M = 0.145 | 29,113 | 13,154 | 0.452 |
| M = 0.155 | 26,921 | 12,400 | 0.461 |
| M = 0.175 | 25,040 | 12,905 | 0.515 |
| M = 0.185 | 25,096 | 13,935 | 0.555 |
| M = 0.192 | 25,384 | 14,832 | 0.584 |
| **Alternative maturity curve** | | | |
| FI $_{logit\_a}$ | 51,626 | 25,418 | 0.492 |
| FI $_{logit\_b}$ | 41,711 | 20,526 | 0.492 |
| HIMI | 31,666 | 14,819 | 0.468 |
| Kerguelen, Crozet | 43,599 | 21,088 | 0.484 |
| SG, SSI | 35,683 | 16,642 | 0.466 |

1) **Natural mortality**

In the current assessment we used a fixed input value for natural mortality (M = 0.165; Payne *et al.* 2005) as opposed to estimating it within the model; this approach is commonly used in toothfish stock assessments around the world. In order to test model sensitivity to assumed natural mortality,

a range of M values has been selected: 0.130 - currently used for South Georgia (Earl and Readdy 2021a) and South Sandwich Islands (Earl and Readdy 2021b); 0.155 - estimated by Candy *et al.* (2011) and currently used for Heard and McDonald Islands (Ziegler 2021), Kerguelen (Massiot-Granier *et al.* 2021a) and Crozet Island (Massiot-Granier *et al.* 2021c); 0.175 - model estimated for Falkland Islands in 2018 and 2019 assessments (Farrugia and Winter 2018, 2019); 0.185 - model estimated for Falkland Islands in 2020 assessment (Skeljo and Winter 2020); 0.192 - model estimated for Falkland Islands in 2021 assessment (Skeljo and Winter 2021).

The analysis showed that in the current model setup, higher M values lead to lower absolute $SSB_0$ estimates, but also a more optimistic $SSB/SSB_0$ trends and higher MSY estimates; the opposite is true of lower M values (Figure 8). As our harvest control rules are based on the $SSB/SSB_0$ ratio, the currently used M value (0.165) can be considered more precautionary than the higher values (0.175-0.192) and less precautionary than the lower values (0.130-0.155). However, the lower values resulted in very similar $SSB/SSB_0$ ratio in the final year (0.45-0.48) as did the current model, and would fall in the same harvest control rule range (expansion range). Similarly, lower M values resulted in lower MSY estimates (1,518-1,607 t) but even these conservative estimates are well above the current TAC (1040 t).
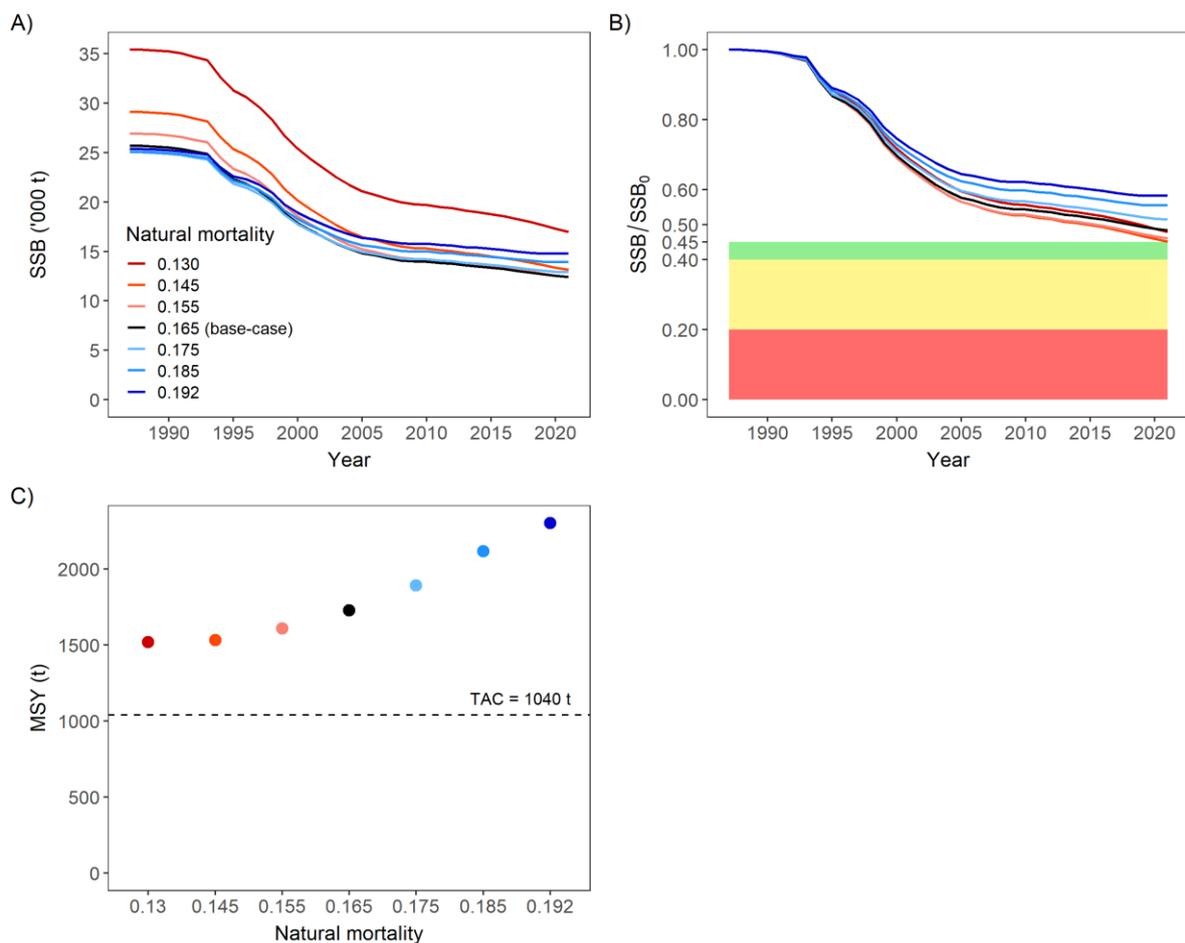


Figure 8. Estimated A) SSB trends, B) $SSB/SSB_0$ trends and C) MSY-s for the base-case model (black line/dot) and six alternative models with different assumed natural mortality. Note that the x-axis in plot C) is not linear.

### 2) Maturity curve

Maturity-at-age curve used in the current assessment includes an adjustment to the proportion of mature fish, as outlined in the 'Maturity' section of this report. The approach used for this adjustment has been described (Farrugia and Winter 2018) and found sensible by the external

review (Bergh 2018). Nevertheless, it was suggested that the maturity curve be compared with those used for other toothfish stocks and if there are marked differences that these are highlighted and explained. Following up on this recommendation we conducted a sensitivity analysis by re-running our model with alternative maturity curves, currently or recently used in toothfish assessments of Heard and McDonald Islands (Yates 2018), Kerguelen and Crozet Island (Massiot-Granier and Peron 2021b, 2021b) and South Georgia and South Sandwich Islands (Earl *et al.* 2015, Soeffker *et al.* 2015). In addition, we tested two alternative maturity curves fitted to subsets of Falkland Islands data: $FI_{logit\_a}$ - fish assigned to maturity stage 2 were excluded from the analysis and logistic curve was fitted to the remaining data; $FI_{logit\_b}$ - fish assigned to maturity stages 2 or 3 were excluded from the analysis and logistic curve was fitted to the remaining data. Stages 2 and 3 were excluded because they appear to have high number of erroneously assigned fish, i.e. very old fish assigned as immature and very young fish assigned as mature. Exclusion of these data produces more realistic maturity curves; the downsides being the lower overall sample size and the loss of variability leading to steep estimated maturity curves (Figure 9.A).

The sensitivity analysis showed that different maturity curves lead to very different estimates of absolute $SSB_0$ (25,691 - 51,626 t), but also to very similar SSB trends (Figure 9.B). Consequently, this leads to very similar $SSB/SSB_0$ trends and $SSB_{2021}/SSB_0$ ratios (0.468 - 0.492) that all fall within the harvest control rules *expansion range* (Figure 9.C). Finally, the MSY estimates were similar across all tested models (1,656 - 1,773 t) and well above the current TAC (1040 t) (Figure 9.D).

Although toothfish maturity-at-age in Falkland Islands waters warrants more research, sensitivity analysis shows that the main parameters used in the management ($SSB_{2021}/SSB_0$ ratio and, to a lesser extent, MSY) are robust to changes in assumed maturity curve.
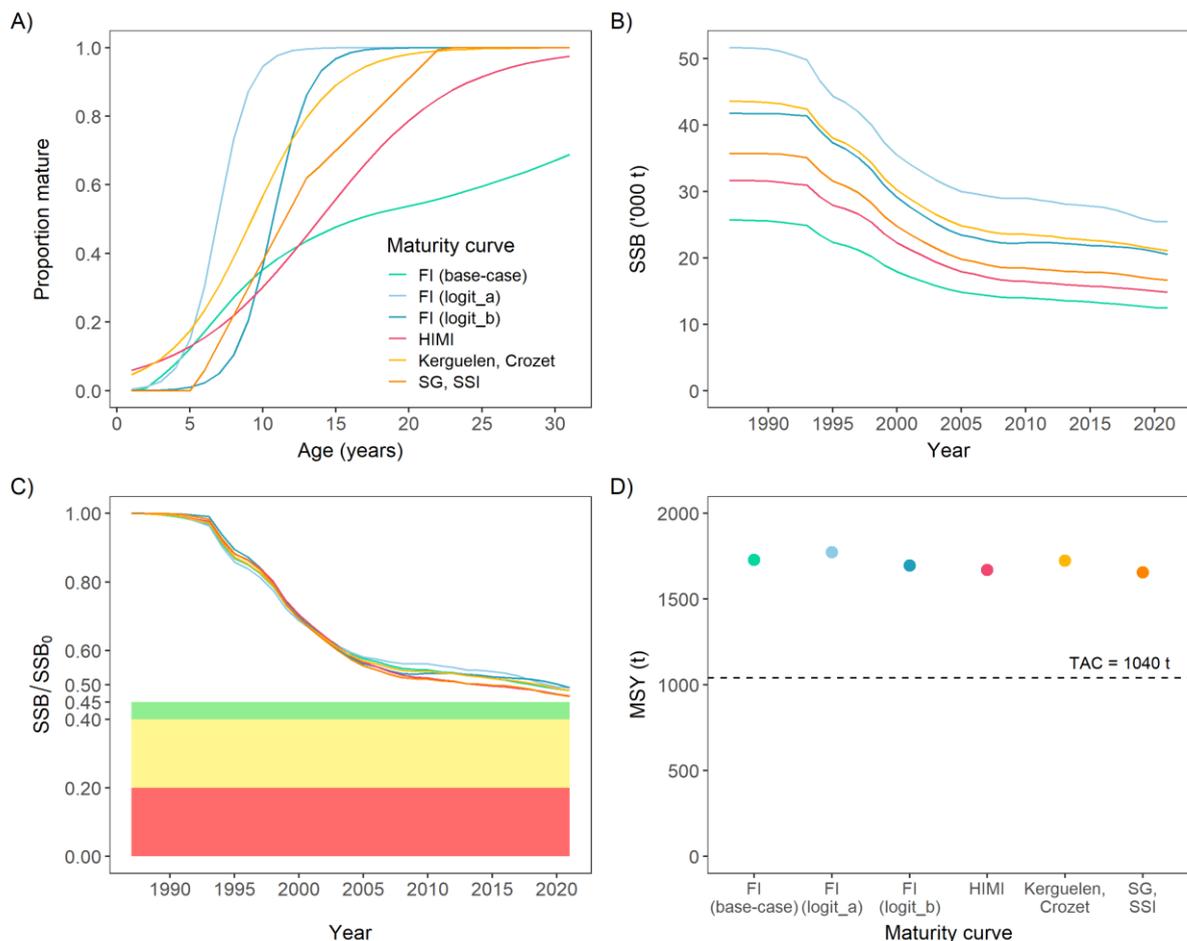


Figure 9. A) Maturity curves used in the base-case model (green line) and five alternative models; Estimated B) SSB trends, C) $SSB/SSB_0$ trends and D) MSY-s for the corresponding models.

## 4. Discussion

This report presents an updated assessment for Patagonian toothfish (*Dissostichus eleginoides*) in Falkland Islands waters, based on the catch and effort data reported by the fisheries, and toothfish age, length and maturity data collected by observers during commercial trips and research surveys. Compared to the 2020 assessment, this assessment incorporates (a) updated observations and ageing data for 2021, (b) revised longline fisheries CPUE time series and associated process errors, (c) separate age-length keys (ALKs) for longline and trawl fisheries and (d) fixed input value for natural mortality (M). While data updates and corrections are self-explanatory, the three remaining model changes (decreased CPUE process error, separate ALKs, and fixed M) are discussed below.

The poor model fit to the CPUE data, especially the failure to reflect the decline in the Spanish-system longline CPUE data, has been flagged by the external review and the suggestion was made to explore whether better fits can be obtained (Bergh 2018). This could be achieved by either up-weighting CPUE data or down-weighting catch-at-age (CAA) data in the model (Francis, 2011); since CPUE and CAA data are the only two types of observations available to our model, down-weighting one effectively gives more relative weight to the other. In current assessment we decided to up-weight CPUE data, by decreasing process error CV associated with CPUE datasets.

The use of two separate age-length keys was motivated by the fact that longline and trawl fisheries exploit different parts of the toothfish population in different areas: longlining occurs on the slope and in deep water (predominantly adult and mature fish), and trawling on the shelf (predominantly juvenile and immature fish). Separate ALKs allow for differences in age structure between these two areas, caused by differences in age-specific mortality rates, growth and availability to fishing due to age-dependent (ontogenetic) migration patterns (Ailloud and Hoenig 2019).

Regarding natural mortality, the model-estimated M kept increasing over the three most recent assessments; to prevent the estimates from gaining exaggerated values unlikely for a long-lived species like toothfish, we decided to fix M at a lower value previously used for toothfish stock assessment in Falkland Islands waters (Payne *et al.* 2005). Even though M would ideally be estimated within the model, there is likely not enough information in the available observations to do so reliably. This is not unique to our assessment; indeed, all toothfish assessments done in CCAMLR waters, i.e. South Georgia, South Sandwich Islands, Heard and McDonald Islands, Kerguelen, Crozet Island and Ross Sea, use a fixed value of M in the models, despite having access to additional information in the form of extensive tag-recapture datasets (Earl and Readdy 2021a, 2021b; Ziegler 2021; Massiot-Granier *et al.* 2021a, 2021c; Grüss *et al.* 2021).

Introduced changes resulted in an overall improved model fit, especially to CPUE data, and narrower credible intervals on most estimated parameters. The exception were selectivity parameters for both research surveys, which showed comparatively wide credible intervals, poor MCMC trace plots and high autocorrelation, possibly due to the nature of the *allvalues* right-hand limb descending selectivity ogive. However, this was considered of minor importance, since CAA data from research surveys have very little effect on the model estimates (Skeljo and Winter 2021).

The updated assessment for 2021 resulted in higher estimates of $SSB_0$ and $SSB_{2021}$, almost identical $SSB_{2021}/SSB_0$ estimate, and lower MSY estimate compared to the previous year. Assessing the contribution of different model updates to this change in model estimates is not straightforward due to complex interactions between different datasets (e.g., updating the ALK simultaneously affects CAA data, Von Bertalanffy parameters and selectivity estimates, and each of these in turn can have different effect on the model estimates). However, running multiple models with different combinations of most significant updates indicated that the change was mainly driven by the fixed M value and the decreased CPUE process error, with lower effect of ALKs and other model updates.

The large influence of CPUE data, especially Spanish-system longline CPUE, on model estimates has already been noted (Skeljo and Winter 2021); CPUE data are of particular importance in our assessment because these are currently the only data which provide an index

of resource abundance. Commercial fisheries CPUE data are often the most influential inputs to stock assessment models (Hoyle *et al.* 2014) and are widely used as an integral part of the stock assessment process (Campbell 2004, Maunder and Punt 2004, Maunder *et al.* 2006, 2020). However, shortcomings and limitations of this approach are well known and documented (Hilborn and Walters 1992, Harley *et al.* 2001, Ye and Dennis 2009, Bentley *et al.* 2011, Thorson *et al.* 2017). In order to provide additional information on the absolute stock abundance, we strongly recommend that the tag-recapture data be introduced into the model; this is a common approach, practiced by all toothfish stock assessments done in CCAMLR waters (referenced above). Although the existing tag-recapture data for Falkland Islands waters are still low (4690 releases and 334 recaptures since 2016), this should be enough for at least a preliminary model run, pending further data. A 4-year extension of the tagging programme has recently been recommended (Lee and Skeljo 2020); following this advice, targeted tagging efforts were renewed in 2021 after a two-year hiatus (Skeljo and Pearman 2021) with a goal of tagging ~1000 fish annually, roughly one fish per tonne of TAC.

The large influence of assumed natural mortality on model estimates is not surprising, given it is often considered one of the most important parameters in a fish stock assessment, influencing interpretation of both CPUE and CAA data and affecting productivity estimates for the population (Maunder and Piner 2015, Punt *et al.* 2021). However, it is also among the most difficult parameters to estimate, and the precision with which it can be estimated depends on the quality and contrast of the available data (Lee *et al.* 2011, Punt *et al.* 2021). The introduction of fixed M value into our assessment aimed to avoid overly high model estimates of M by capping it at a lower value available from the literature; bringing our model setup in line with the common approach to toothfish stock assessment around the world.

In summary, fixing M within the model and encouraging better model fit to CPUE data resulted in more consistent model behaviour, as evidenced by much narrower confidence intervals on key parameter estimates ($SSB_0$, $SSB_{2021}/SSB_0$, MSY). Even though the new model produced somewhat higher absolute SSB estimates, the $SSB_{2021}/SSB_0$ and MSY estimates were same or lower than before; another point of consideration is that upper 95% credible limits on all key parameters were severely reduced in the current model setup, suggesting that the high estimates are unlikely. Taking this into consideration, the current model can be considered precautionary compared to the previously used one; a preferred approach in face of uncertainty in model assumptions.

# 5. Management advice

Management advice is based on harvest control rules (HCR) established for the Falkland Islands toothfish longline fishery (Farrugia and Winter 2018, 2019) (Appendix 4). The estimated $SSB_{2021}/SSB_0$ ratio of 0.484 is above *upper target reference point* (0.45), i.e. in the *expansion range*, and the projection suggests it will remain above 0.45 in the future. This is the second consecutive year with SSB ratio within the expansion range; at least three consecutive years within the expansion range are required before considering TAC alterations, therefore no action is anticipated by HCR at this point.

The recommendation for the toothfish longline fishery is to maintain the annual total allowable catch (TAC) at its current level of 1,040 tonnes.

# 6. Future assessment requirements

Based on the external review recommendations for Falkland Islands toothfish assessment (Bergh 2018), the best practices in toothfish stock assessments around the world, and the findings of our current and previous stock assessments and sensitivity analyses, the following points for future consideration and model refinement were identified:

*Observations*
- Include the existing toothfish tag-recapture observations into the model, as additional information on stock abundance.

*Maturity*
- Further review the existing maturity-at-age data and the approach to modelling maturity in the assessment.


# 7. References

Agnew DJ, Pearce J, Pramod G, Peatman T, Watson R, Beddington JR, Pitcher TJ. 2009. Estimating the worldwide extent of illegal fishing. PLoS ONE 4, e4570. https://doi.org/10.1371/journal.pone.0004570

Ainsworth CH, Pitcher TJ. 2005. Estimating illegal, unreported and unregulated catch in British Columbia's marine fisheries. Fisheries Research 75, 40-55.

Arana P. 2009. Reproductive aspects of the Patagonian toothfish (*Dissostichus eleginoides*) off southern Chile. Latin American Journal of Aquatic Research 37, 381-394.

Arkhipkin AI, Laptikhovsky VV. 2010. Convergence in life-history traits in migratory deep-water squid and fish. ICES Journal of Marine Science 67, 1444-1451.

Arkhipkin AI, Brickle P, Lee B, Shaw PW, McKeown NJ. 2022. Taxonomic re-appraisal for toothfish (*Dissostichus:* Notothenioidea) across the Antarctic Polar Front using genomic and morphological studies. Journal of Fish Biology, 1-13.

Ailloud LE, Hoenig JM. 2019. A general theory of age-length keys: combining the forward and inverse keys to estimate age composition from incomplete data. ICES Journal of Marine Science 76(6), 1515-1523.

Ashford JR, Fach BA, Arkhipkin AI, Jones CM. 2012. Testing early life connectivity supplying a marine fishery around the Falkland Islands. Fisheries Research 121-122, 144-152.

Barton K. 2009. Mu-MIn: Multi-model inference. R Package Version 0.12.2/r18. http://R-Forge.R-project.org/projects/mumin/

Bentley N, Kendrick TH, Starr PJ, Breen PA. 2011. Influence plots and metrics: tools for better understanding fisheries catch-per-unit-effort standardizations. ICES Journal of Marine Science 69(1), 84-88.

Bergh M. 2018. A review of Falkland Islands toothfish stock assessment and management. Report prepared for the Falkland Islands Government. OLSPS Marine, Cape Town, 42 p.

Boucher EM. 2018. Disentangling reproductive biology of the Patagonian toothfish *Dissostichus eleginoides*: skipped vs. obligatory annual spawning, foraging migration vs residential lifestyle. Environmental Biology of Fishes 101, 1343-1356.

Brandão A, Butterworth DS. 2009. A proposed management procedure for the toothfish (*Dissostichus eleginoides*) resource in the Prince Edward Islands vicinity. CCAMLR Science 16, 33-69.

Brooks ME, Kristensen K, van Benthem KJ, Magnusson A, Berg CW, Nielsen A, Skaug HJ, Maechler M, Bolker BM. 2017. glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modelling. The R Journal 9(2), 378-400. https://journal.r-project.org/archive/2017/RJ-2017-066/index.html.

Brown J, Brickle P, Hearne S, French G. 2010. An experimental investigation of the 'umbrella' and 'Spanish' system of longline fishing for the Patagonian toothfish (*Dissostichus eleginoides*) in the Falkland Islands: Implications for stock assessment and seabird by-catch. Fisheries Research 106, 404-412.

Brown J, Brickle P, Scott BE. 2013. Investigating the movements and behaviour of Patagonian toothfish (*Dissostichus eleginoides* Smitt, 1898) around the Falkland Islands using satellite linked archival tags. Journal of Experimental Marine Biology and Ecology 443, 65-74.

Bull B, Francis RICC, Dunn A, McKenzie A, Gilbert DJ, Smith MH, Bian R, Fu D. 2012. CASAL (C++ algorithmic stock assessment laboratory): CASAL User Manual v2.30-2012/03/21. NIWA Technical Report 135, 275 p.

Cadrin SX. 2020. Defining spatial structure for fishery stock assessment. Fisheries Research 221, 105397.

Campbell RA. 2004. CPUE standardisation and the construction of indices of stock abundance in a spatially varying fishery using general linear models. Fisheries Research 70, 209-227.

Canales-Aguirre CB, Ferrada-Fuentes S, Galleguillos R, Oyarzun FX, Hernández CE. 2018. Population genetic structure of Patagonian toothfish (*Dissostichus eleginoides*) in the Southeast Pacific and Southwest Atlantic Ocean. PeerJ 6, e4173.

Candy SG, Constable AJ. 2008. An integrated stock assessment for the Patagonian toothfish (*Dissostichus eleginoides*) for the Heard and McDonald Islands using CASAL. CCAMLR Science 15, 1-34.

Candy SG, Welford DC, Lamb T, Verdouw JJ, Hutchins JJ. 2011. Estimation of natural mortality for the Patagonian toothfish at Heard and McDonald Islands using catch-at-age and aged mark-recapture data from the main trawl ground. CCAMLR Science 18: 29-46.

Candy SG. 2004. Modelling catch and effort data using generalised linear models, the Tweedie distribution, random vessel effects and random stratum-by-year effects. CCAMLR Science 11, 59-80.

CCAMLR. 2010. Estimation of IUU catches of toothfish inside the convention area during the 2009/10 fishing season. Document WG-FSA-10/6 Rev. 1, 12 p.

Ciannelli L, Fisher JAD, Skern-Mauritzen M, Hunsicker ME, Hidalgo M, Frank KT, Bailey KM. 2013. Theory, consequences and evidence of eroding population spatial structure in harvested marine fishes: A review. Marine Ecology Progress Series 480, 227-243.

Collins MA, Brickle P, Brown J, Belchier M. 2010. The Patagonian toothfish: biology, ecology, and fishery. Advances in Marine Biology 58, 227-300.

Constable AJ, Candy SG, Ball I. 2006a. An investigation of integrated stock assessment methods for the Patagonian toothfish (*Dissostichus eleginoides*) in Division 58.5.2 using CASAL. Document WG-FSA-SAM-06/14. CCAMLR, Hobart, Australia.

Constable AJ, Candy SG, Ball I. 2006b. An integrated stock assessment for the Patagonian toothfish (*Dissostichus eleginoides*) in Division 58.5.2 using CASAL. Document WG-FSA-06/64. CCAMLR, Hobart, Australia.

Dunn A, Hanchet SM. 2006. Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea including data from the 2005-06 season. Document WG-FSA-06/60, CCAMLR, Hobart, Australia.

Dunn A, Hanchet SM. 2010. Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea including data from the 2006–07 season. New Zealand Fisheries Assessment Report 2010/1, 28 p.

Dunn A, Parker S. 2019. Stock Annex for the stock assessment of Ross Sea region Antarctic toothfish (*Dissostichus mawsoni*). WG-FSA-2019/09. CCAMLR, Hobart, Australia, 13 p.

Earl T, Readdy L. 2021a. Assessment of Patagonian toothfish (*Dissostichus eleginoides*) in Subarea 48.3. CCAMLR working document WG-FSA-2021/59, CCAMLR, Hobart, Australia.

Earl T, Readdy L. 2021b. Assessment of Patagonian toothfish (*Dissostichus eleginoides*) in Subarea 48.4. CCAMLR working document WG-FSA-2021/61, CCAMLR, Hobart, Australia.

Earl T, Soeffker M, Darby C. 2015. Assessment of Patagonian toothfish (*D. eleginoides*) in Subarea 48.3. CCAMLR working document WG-FSA-15/59, CCAMLR, Hobart, Australia.

Farrugia TJ, Winter A. 2018. 2017 Stock Assessment Report for Patagonian toothfish, Fisheries Report SA-2017-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 35 p.

Farrugia TJ, Winter A. 2019. 2018 Stock Assessment Report for Patagonian toothfish, Fisheries Report SA-2018-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 38 p.

Francis RICC. 2011. Data weighting in statistical fisheries stock assessment models. Canadian Journal of Fisheries and Aquatic Sciences 68, 1124-1138.

Francis RICC. 2013. DataWeighting: A set of R functions for evaluating and calculating data weights for CASAL. R package version 1.0.

Geweke J. 1992. Evaluating the accuracy of sampling-based approaches to calculating posterior moments. In: Bernardo JM, Berger JO, David AP, Smith AFM (Eds.). Bayesian Statistics, 4. Clarendon Press, Oxford, 169-194.

Grüss A, Dunn A, Parker S. 2021. Assessment model for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region to 2020/21. CCAMLR working document WG-FSA-2021/26, CCAMLR, Hobart, Australia.

Harley SJ, Myers RA, Dunn A. 2001. Is catch-per-unit-effort proportional to abundance? Canadian Journal of Fisheries and Aquatic Sciences 58, 1760-1772.

He X, Mangel M, MacCall AD. 2006. A prior for steepness in stock-recruitment relationships, based on an evolutionary persistence principle. Fishery Bulletin 104 (3): 428-433.

Heidelberger P, Welch P. 1983. Simulation run length control in the presence of an initial transient. Operations Research 31, 1109-1144.

Hilborn R, Walters CJ. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman and Hall, New York, 570 p.

Hillary RM, Kirkwood GP, Agnew DJ. 2006. An assessment of toothfish in subarea 48.3 using CASAL. CCAMLR Science 13, 65-95.

Hoyle SD, Langley AD, Campbell RA. 2014. Recommended approaches for standardizing CPUE data from pelagic fisheries. Document WCPFC-SC10-2014/ SA-IP-10. DOI: 10.13140/2.1.4433.2164

ICES. 2017. Report of the workshop on evaluation of the adopted harvest control rules for Icelandic summer spawning herring, link and tusk (WKICEMSE), 21-25 April 2017, Copenhagen, Denmark. ICES CM 2017/ACOM:45, 196 p.

Kenchington TJ. 2014. Natural mortality estimators for information-limited fisheries. Fish and Fisheries 15(4): 533-562.

Laptikhovsky VV, Arkhipkin AI, Brickle P. 2006. Distribution and reproduction of the Patagonian toothfish *Dissostichus eleginoides* Smitt around the Falkland Islands. Journal of Fish Biology 68, 849-861.

Laptikhovsky VV, Brickle P. 2005. The Patagonian toothfish fishery in Falkland Islands' waters. Fisheries Research 74, 11-23.

Lee B, Arkhipkin A, Randhawa H. 2021. Environmental drivers of Patagonian toothfish (*Dissostichus eleginoides*) spatial-temporal patterns during an ontogenetic migration on the Patagonian Shelf. Estuarine, Coastal and Shelf Science 259, 107473.

Lee B, Brewin PE, Brickle P, Randhawa H. 2018. Use of otolith shape to inform stock structure in Patagonian toothfish (*Dissostichus eleginoides*) in the south-western Atlantic. Marine and Freshwater Research 69, 1238-1247.

Lee B, Skeljo F, Randhawa HS, Arkhipkin A. 2022. Deep-sea movement patterns of the Patagonian toothfish *Dissostichus eleginoides* Smitt in the Southwest Atlantic. Marine and Freshwater Research, 1-13.

Lee B, Skeljo F. 2020. Patagonian toothfish tag recapture program update: June 2016 - July 2020. Scientific Report TAG-2020-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands, 15 p.

Lee B. 2014. Age structure of Patagonian toothfish *Dissostichus eleginoides* in the Falkland Islands: 2014 - 2015. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 11 p.

Lee B. 2015. Age structure for Patagonian toothfish *Dissostichus eleginoides* from Falkland Island waters: January - December 2015. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 11 p.

Lee B. 2016. Age structure for Patagonian toothfish *Dissostichus eleginoides* from Falkland Island waters: January - December 2016. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 12 p.

Lee B. 2017. Age structure for Patagonian toothfish *Dissostichus eleginoides* from Falkland Island waters: January - December 2017. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 16 p.

Lee B. 2018. Age structure for Patagonian toothfish *Dissostichus eleginoides* from Falkland Island waters: January - December 2018. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 16 p.

Lee B. 2019. Age structure for Patagonian toothfish *Dissostichus eleginoides* from Falkland Island waters: January - December 2019. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 15 p.

Lee B. 2020. Age structure for Patagonian toothfish *Dissostichus eleginoides* around the Falkland Islands: January - December 2020. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 14 p.

Lee B. 2022. Stock structure of Patagonian toothfish *Dissostichus eleginoides* (Smitt 1898, family Nototheniidae) in the southwest Atlantic. PhD Thesis, Rhodes University.

Lee HH, Maunder MN, Piner KR, Methot RD. 2011. Estimating natural mortality within a fisheries stock assessment model: an evaluation using simulation analysis based on twelve stock assessments. Fisheries Research 109, 89-94.

Magnusson A, Skaug HJ, Nielsen A, Berg CW, Kristensen K, Maechler M, van Bentham KJ, Bolker BM, Brooks ME. 2017. glmmTMB: Generalized Linear Mixed Models using Template Model Builder. R package version 0.1.3. https://github.com/glmmTMB

Mangel M, MacCall AD, Brodziak J, Dick EJ, Forrest RE, Pourzand R, Ralston S. 2013. A perspective on steepness, reference points, and stock assessment. Canadian Journal of Fisheries and Aquatic Sciences 70, 930-940.

Massiot-Granier F, Landru S, Peron C. 2021a. Updated stock assessment model for the Kerguelen Island EEZ Patagonian toothfish (*Dissostichus eleginoides*) fishery in Division 58.5.1 for 2021. CCAMLR working document WG-FSA-2021/46, CCAMLR, Hobart, Australia.

Massiot-Granier F, Péron C. 2021b. Stock Annex for the 2021 assessment of the Patagonian toothfish (*Dissostichus eleginoides*) population of Kerguelen. CCAMLR working document WG-FSA-2021/47, CCAMLR, Hobart, Australia.

Massiot-Granier F, Péron C. 2021c. An integrated stock assessment for the Crozet Islands Patagonian toothfish (*Dissostichus eleginoides*) fishery in Subarea 58.6. CCAMLR working document WG-FSA 2021/45, CCAMLR, Hobart, Australia.

Maunder MN, Piner KR. 2015. Contemporary fisheries stock assessment: many issues still remain. ICES Journal of Marine Science 72, 7-18.

Maunder MN, Punt AE. 2004. Standardizing catch and effort data: A review of recent approaches. Fisheries Research 70, 141-159.

Maunder MN, Sibert JR, Fonteneau A, Hampton J, Kleiber P, Harley SJ. 2006. Interpreting catch per unit effort data to assess the status of individual stocks and communities. ICES Journal of Marine Science 63, 1373-1385.

Maunder MN, Thorson JT, Xu H, Oliveros-Ramos R, Hoyle SD, Tremblay-Boyer L, Lee HH, Kai M, Chang SK, Kitakado T, Albertsen CM, Minte-Vera CV, Lennert-Cody CE, Aires-da-Silva AM, Piner KR. 2020. The need for spatio-temporal modelling to determine catch-per-unit-effort based indices of abundance and associated composition data for inclusion in stock assessment models. Fisheries Research 229, 105594.

Mormede S, Dunn A, Hanchet SM. 2011. Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea for the years 1997–98 to 2010–11. Document WG-FSA-11/42. CCAMLR, Hobart, Australia, 40 p.

Mormede S, Dunn A, Hanchet SM. 2013. Assessment models for Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea for the years 1997–98 to 2010–13. Document WG-FSA-13/51. CCAMLR, Hobart, Australia, 36 p.

Mormede S, Dunn A, Hanchet SM. 2014. A stock assessment model of Antarctic toothfish (*Dissostichus mawsoni*) in the Ross Sea region incorporating multi-year mark-recapture data. CCAMLR Science 21, 39-62.

Parker SJ. 2015. Stock discrimination tools for Falkland Islands toothfish. NIWA Client Report NEL2015-002, 28 p.

Pinheiro JC, Bates DM. 2000. Mixed-effects Models in S and S-plus. Springer, New York, 528 p.

Pitcher TJ, Watson R, Forrest R, Valtýsson HÞ, Guénette S. 2002. Estimating illegal and unreported catches from marine ecosystems: a basis for change. Fish and Fisheries 3, 317-339.

Punt AE, Smith DC, Koopman MT. 2005. Using information for "data-rich" species to inform assessments of "data-poor" species through Bayesian stock assessment methods. Final report to Fisheries Research and Development Corporation Project No. 2002/094. Primary Industries Research, Victoria Queenscliff, 243 p.

Punt AE, Castillo-Jordán C, Hamel OS, Cope JM, Maunder MN, Ianelli JN. 2021. Consequences of error in natural mortality and its estimation in stock assessment models. Fisheries research 233, 105759.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Randhawa HS, Lee B. 2016. Toothfish tagging cruise. Scientific Report ZDLC2-06-2016. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands, 20 p.

Rogers AD, Morley S, Fitzcharles E, Jarvis K, Belchier M. 2006. Genetic structure of Patagonian toothfish (*Dissostichus eleginoides*) populations on the Patagonian shelf and Atlantic and western Indian Ocean sectors of the Southern Ocean. Marine Biology 149, 915-924.

Shaw PW, Arkhipkin AI, Al-Khairulla H. 2004. Genetic structuring of Patagonian toothfish populations in the southwest Atlantic Ocean: the effect of the Antarctic polar front and deep-water troughs as barriers to genetic exchange. Molecular Ecology 13, 3293-3303.

Skeljo F, Pearman T. 2021. Patagonian toothfish (*Dissostichus eleginoides*) tagging and benthic habitat survey. Cruise Report ZDLK3-01-2021. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands, 13 p.

Skeljo F, Winter A. 2020. 2019 Stock assessment report for Patagonian toothfish (*Dissostichus eleginoides*). Fisheries Report SA-2019-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands, 36 p.

Skeljo F, Winter A. 2021. 2020 Stock assessment report for Patagonian toothfish (*Dissostichus eleginoides*). Fisheries Report SA-2020-TOO. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, Stanley, Falkland Islands. 38 p.

Soeffker M, Laptikhovsky V, Earl T, Darby C. 2015. An integrated stock assessment of Patagonian toothfish (*Dissostichus eleginoides*) in CCAMLR Subarea 48.4. CCAMLR working document WG-FSA-15/28, CCAMLR, Hobart, Australia.

Thorson JT, Fonner R, Haltuch MA, Ono K, Winker H. 2017. Accounting for spatiotemporal variation and fisher targeting when estimating abundance from multispecies fishery data. Canadian Journal of Fisheries and Aquatic Sciences 74, 1794-1807.

Winter A, Pompert J. 2016. Initial analysis of whale depredation in the Falkland Islands toothfish longline fishery. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 18 p.

Yates P, Ziegler P, Welsford D, McIvor J, Farmer B, Woodcock E. 2018. Spatio-temporal dynamics in maturation and spawning of Patagonian toothfish *Dissostichus eleginoides* on the sub-Antarctic Kerguelen Plateau. Journal of Fish Biology 92, 34-54.

Ye Y, Dennis D. 2009. How reliable are the abundance indices derived from commercial catch–effort standardization? Canadian Journal of Fisheries and Aquatic Sciences 66, 1169-1178.

Ziegler P. 2021. Draft integrated stock assessment for the Heard Island and McDonald Islands Patagonian toothfish (*Dissostichus eleginoides*) fishery in Division 58.5.2. CCAMLR working document WG-FSA-2021/21, CCAMLR, Hobart, Australia.

Zuur AF, Ieno EN, Elphick CS. 2010. A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution 1, 3-14.

CPUE data belonging to the commercial Spanish- and umbrella-system longline fisheries are the main source of information on stock abundance available to our stock assessment model. In order to provide unbiased indices of relative stock abundance, these CPUE data had to be standardized to remove the impact of explanatory variables other than abundance (Maunder and Punt, 2004). CPUE standardization employed a generalized linear mixed modelling approach (GLMM; Pinheiro and Bates 2000), an extension of the generalized linear modelling approach (GLM) used in the earlier assessments. GLMMs were fitted using package glmmTMB (Brooks *et al.* 2017, Magnusson *et al.* 2017) implemented in R (R Core Team, 2020).

Prior to modelling, data exploration was applied following the protocol described in Zuur *et al.* (2010). Variables where inspected for outliers and collinearity. Continuous explanatory variables were scaled, i.e. mean was subtracted from the individual values, and the values were divided by its standard deviation. Daily catch reports with zero toothfish catches were presumed to represent erroneous entries or broken sets and were excluded from the analysis.

The response variable in the model was daily longline CPUE, expressed as toothfish catch in kg-per-hook (Spanish-system) or kg-per-umbrella (umbrella-system). As the response variable was continuous and didn't include any zeroes, it was assumed gamma distributed around the mean, and the relationship between the linear predictor and the mean of the distribution was described by a canonical log link function. The explanatory variables considered in the model are given in table A.1.

Table A.1. Explanatory variables considered in the CPUE standardization GLMM, by fishery and type.

| Explanatory variables | | Variable type |
|---|---|---|
| Spanish-system | umbrella-system | |
| Year* | Year* | Categorical |
| Month* | Month* | Categorical |
| Region* | Region* | Categorical |
| Depth | Depth | Continuous |
| Soak-time | Soak-time* | Continuous |
| Vessel* | - | Categorical |
| - | Hooks-per-umbrella | Categorical |

\* Variables included in the final model.

*Year* effect is the quantity of interest so it must be a part of the final CPUE model (Maunder and Punt 2004). The remaining explanatory variables were added to the *Year* by forward stepwise selection and included in the final model only if they improved pseudo-$R^2$ by at least 0.5%. Pseudo-$R^2$ was calculated based on the likelihood-ratio test, as implemented in R package MuMIn (Barton 2009). The *Month* variable accounts for the seasonal variability in CPUE, and the *Region* variable attempts to capture the spatial distribution of CPUE, divided into two broad areas: Falklands conservation zone south of 53.5° S (Burdwood Bank spawning area) and Falklands conservation zone north of 53.5° S. *Depth* variable is the average fishing depth, and *Soak-time* the sum of soak times, of the lines pertaining to a single response CPUE value (usually multiple lines were set by a given vessel on a given day). *Vessel* variable was excluded from the umbrella-system longline CPUE standardization, as the only two vessels used in the assessment never fished concurrently in the same year, making the *Vessel* and *Year* effects indistinguishable. The umbrella-system had one additional variable, number of *Hooks-per-umbrella* (which was progressively decreased from 10 hooks initially to 8 hooks in December 2007, to 7 hooks in March 2014, to 6 hooks in June 2016).

The *vessel* and *month* variables were treated as random effects, thus imposing a correlation among CPUE values belonging to the same vessel or the same month. Random vessel effect accommodates variation between vessels in their ability to catch fish which will depend on the

attributes of the vessel, its crew, and the total extent of fishing grounds that they target (Candy 2004). The *Month* random effect was used to account for the temporal dependency.

Fitting GLMM to the Spanish-system data included the explanatory variables *Year*, *Month*, *Region* and *Vessel*, and the model explained 17.4% of the overall variation in CPUE. Standardized and unstandardized CPUE time series showed overall similar declining trend (Figure A.1). Fitting GLMM to umbrella-system data included the explanatory variables *Year*, *Month*, *Region* and *Soak-time*, and the model explained 14.0% of the overall variation in CPUE. Standardized and unstandardized CPUE time series were similar and showed no clear trend (Figure A.2). The *hooks-per-umbrella* variable wasn't significant, indicating that the gradual reduction in the number of hooks per umbrella from 8 to 7 to 6 didn't significantly affect the CPUE; this was expected, as the change was fishery driven, presumably to simplify the work and possibly reduce the amount of bait while maintaining the catches. This could be achieved because hooks are set in tight clusters, with each hook on a ~30 cm snood, and all snoods tied together at the free end; therefore, reducing the number of hooks doesn't necessarily reduce the catchability of the cluster as a whole.
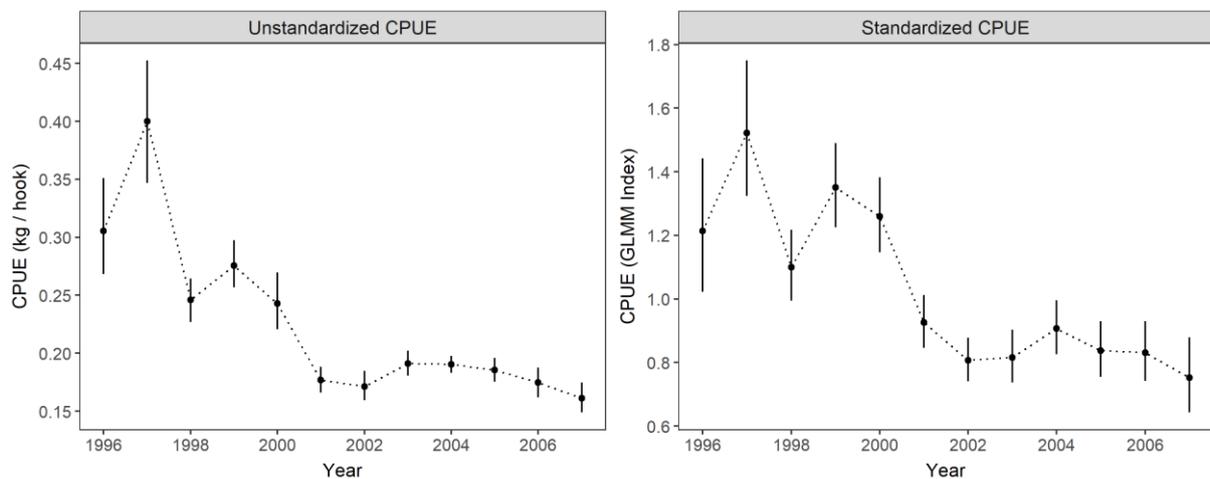


Figure A.1. Spanish-system longline unstandardized and standardized CPUE time series; black vertical lines correspond to 95% confidence intervals.
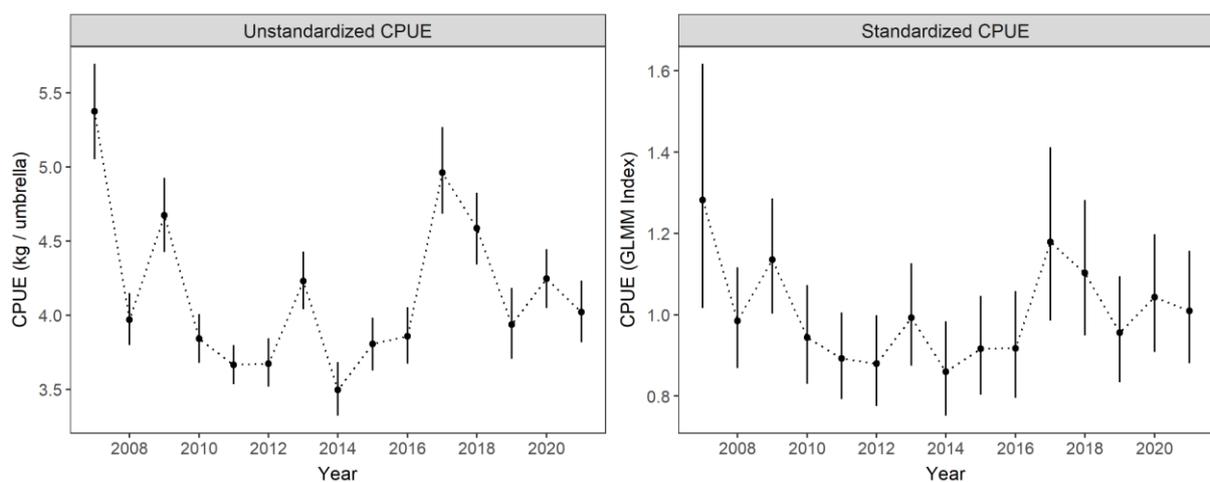


Figure A.2. Umbrella-system longline unstandardized and standardized CPUE time series; black vertical lines correspond to 95% confidence intervals.

## Appendix 2. Diagnostics plots
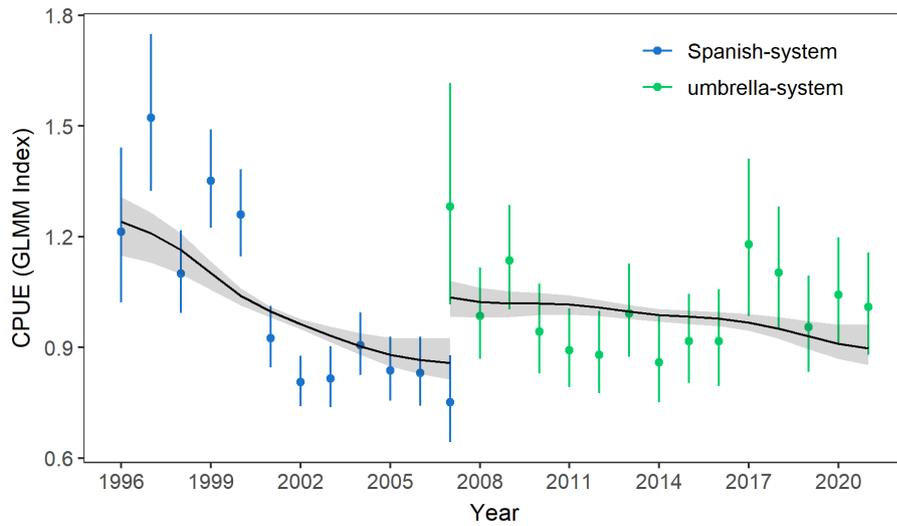
Figure A.3. MPD model fit (black line) to the standardised CPUE indices for Spanish-system (blue dots) and umbrella-system longline (green dots); Vertical blue and green lines denote 95% confidence intervals of the standardised CPUE indices; shaded areas denote MCMC 95% credible intervals of the model fit.
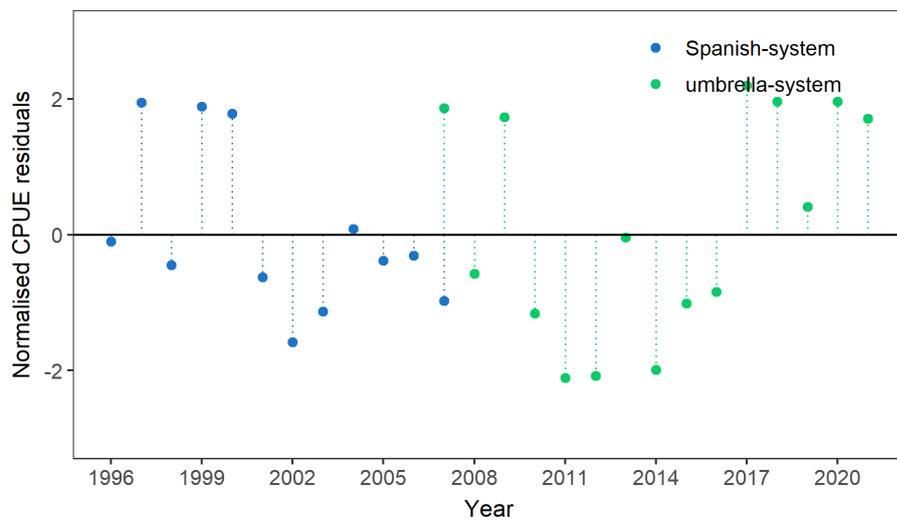


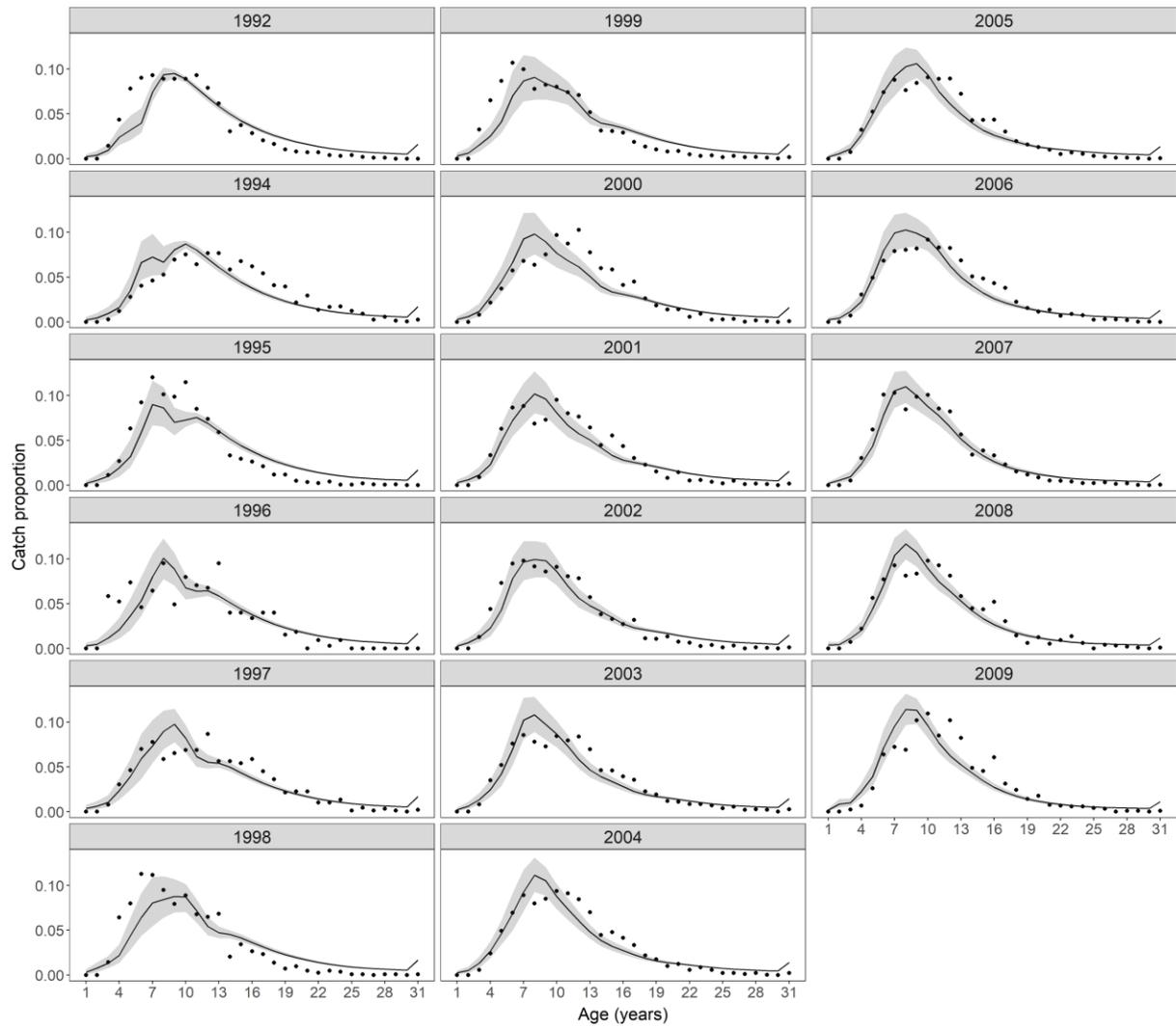Figure A.4. Normalised residuals from the model fit to the standardized CPUE time series; for Spanish-system (blue) and umbrella-system longline (green).

Figure A.5. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the Spanish-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.
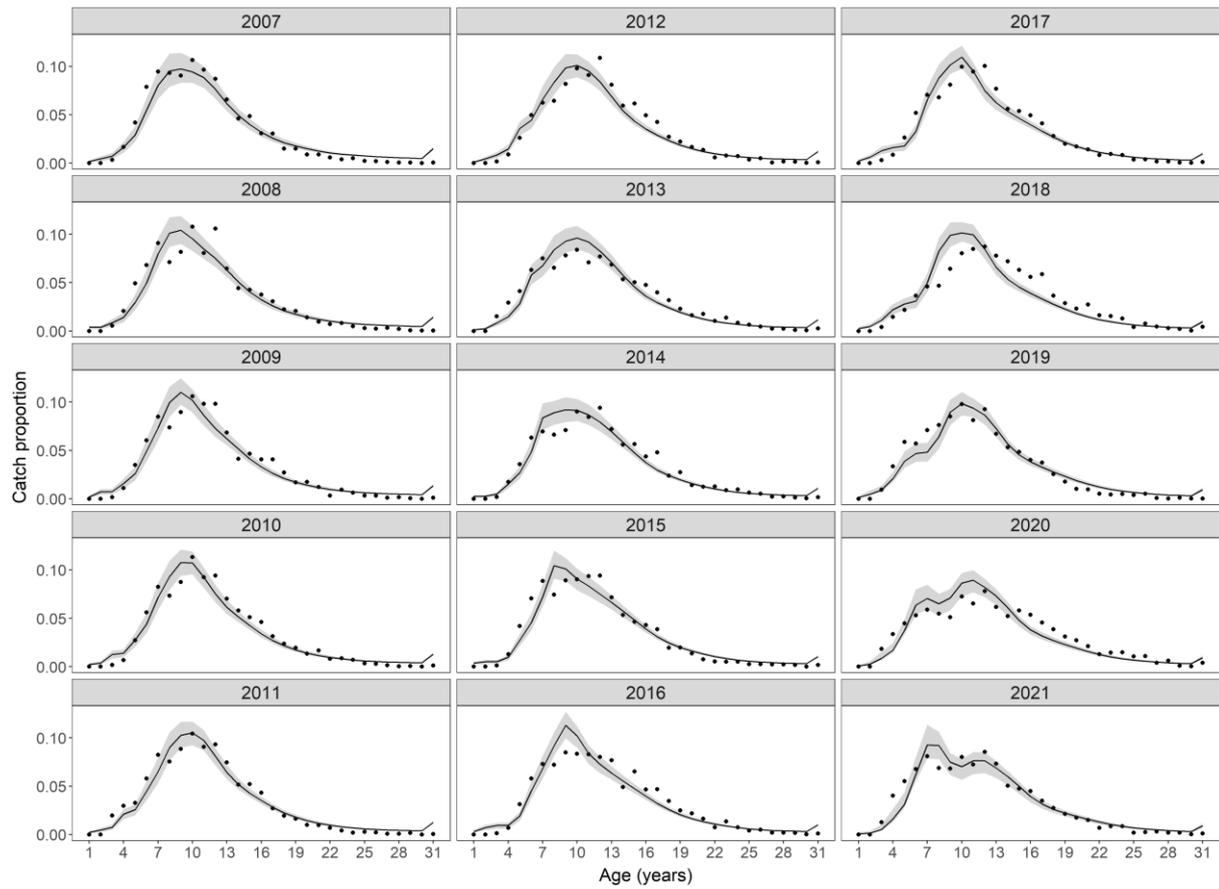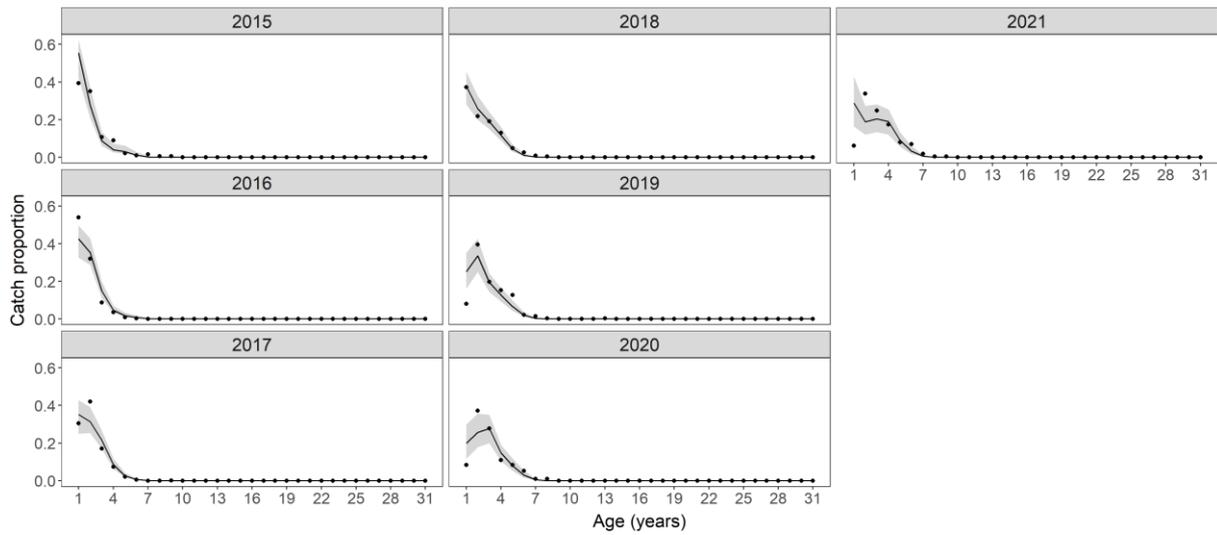
Figure A.6. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the umbrella-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.
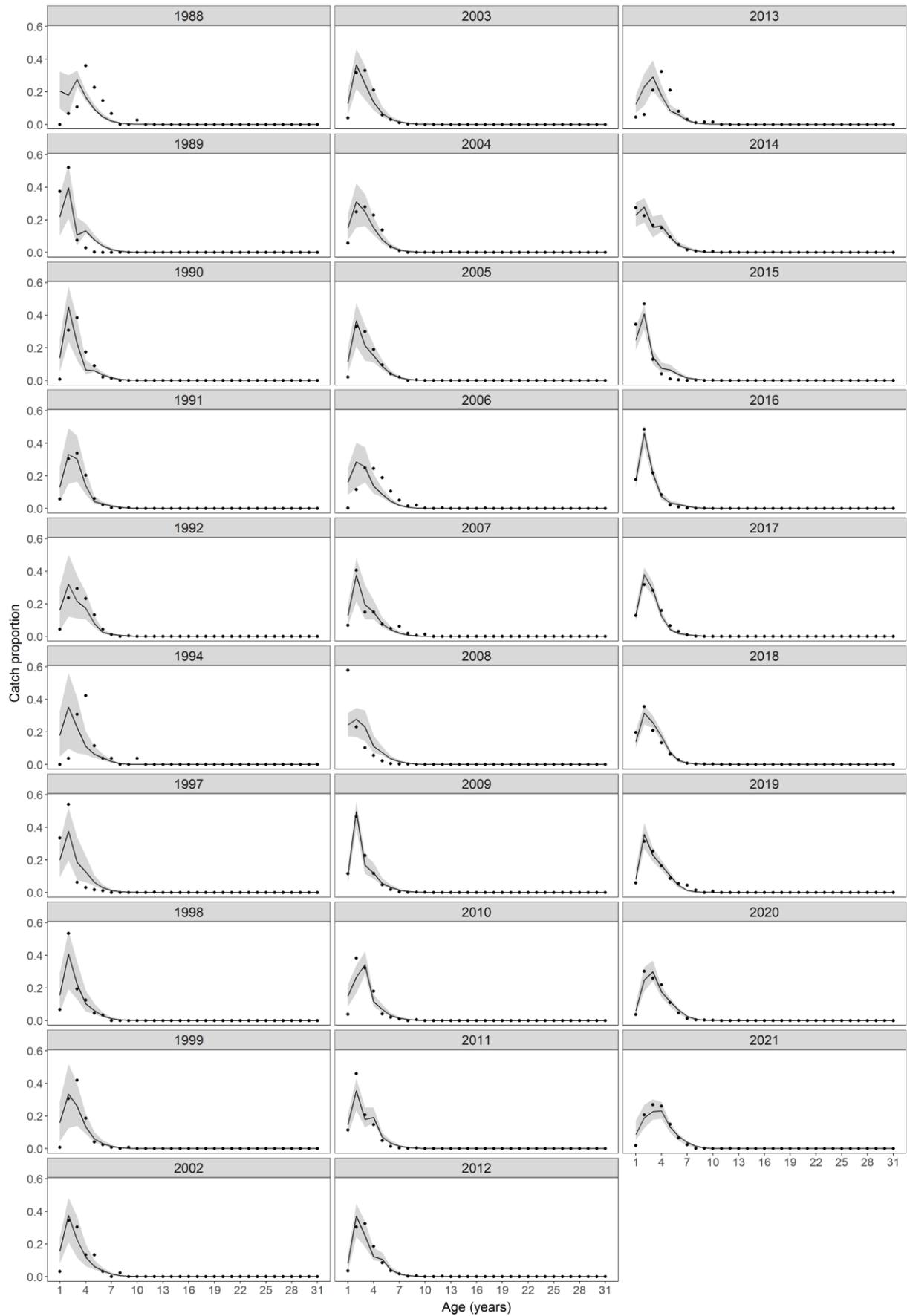
Figure A.7. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the finfish trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.
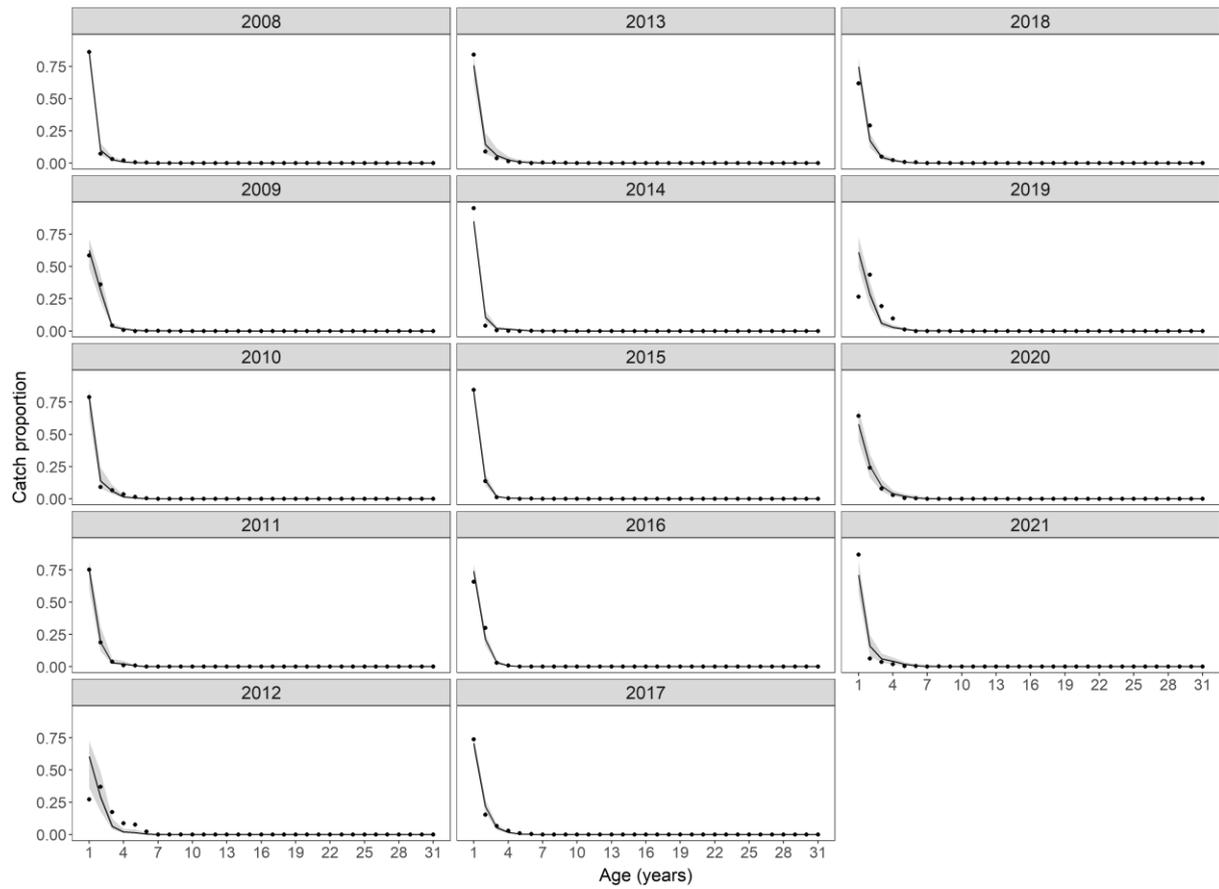
Figure A.8. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the calamari trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

Figure A.9. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the groundfish survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.
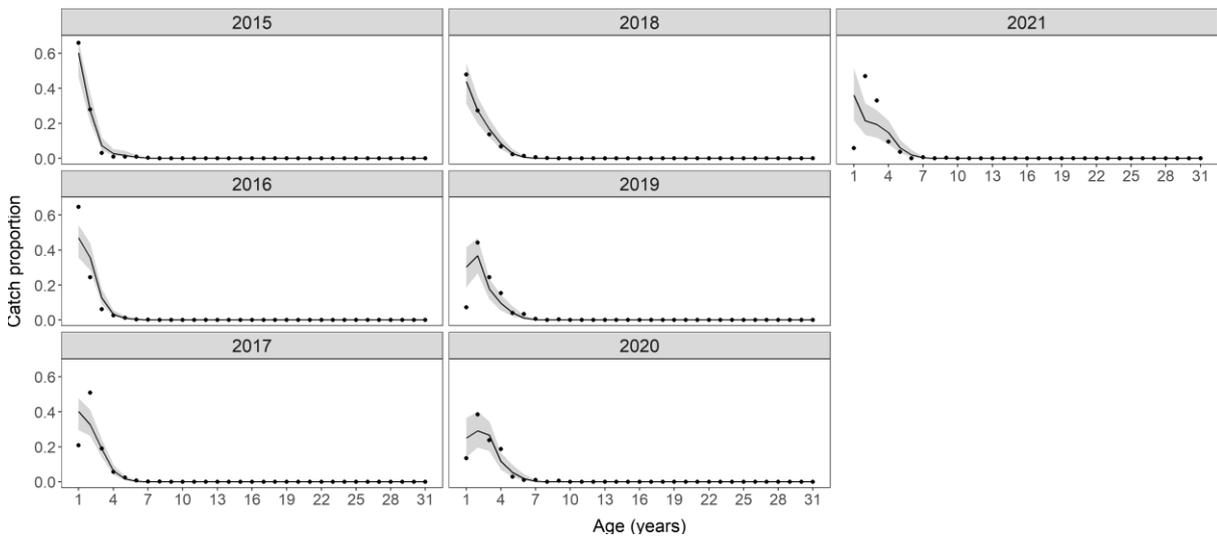


Figure A.10. MPD model fits (solid line) to the observed toothfish catch-proportion-at-age data for the calamari pre-recruitment survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.
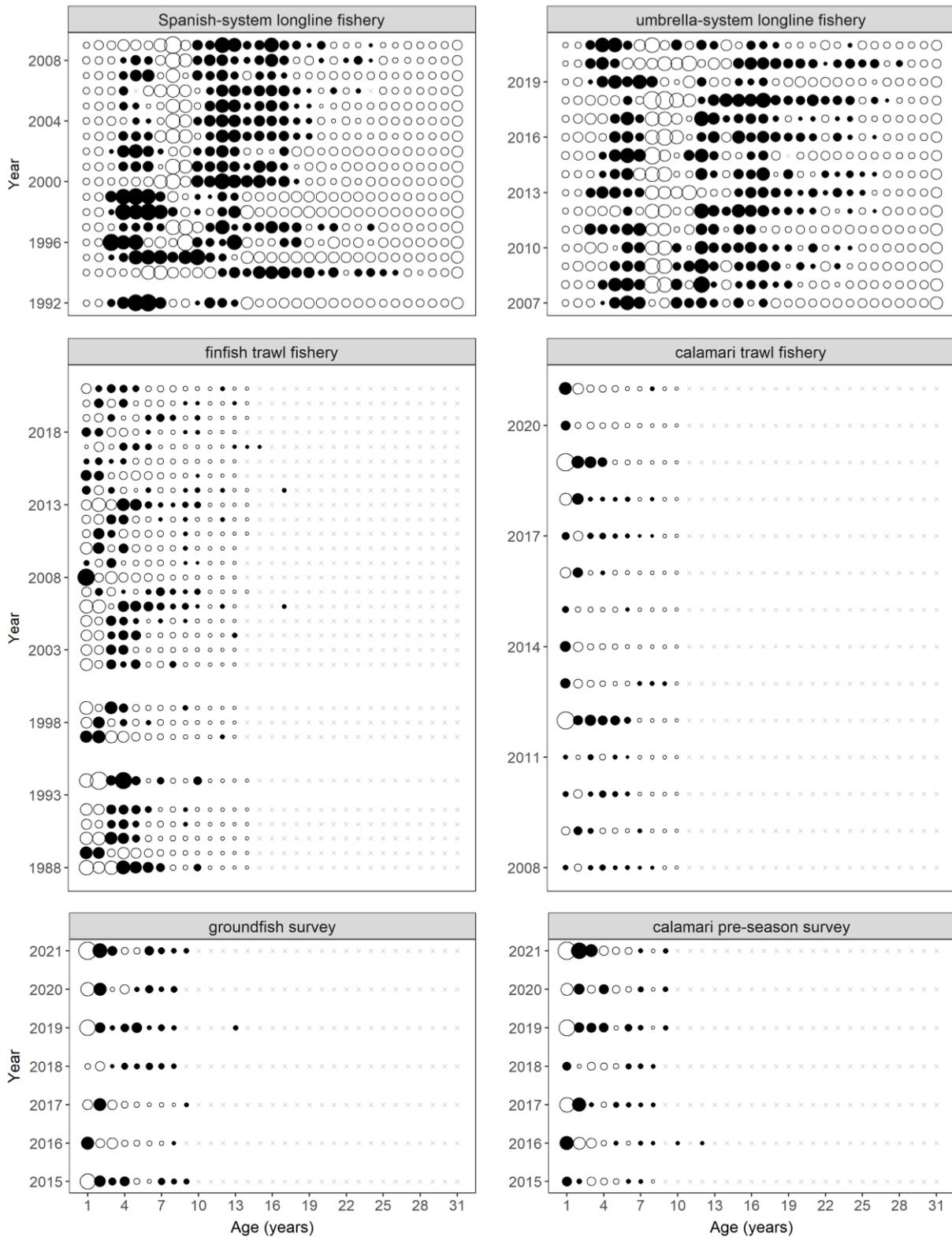
Figure A.11. Residuals from the model fit to observed catch-at-age for four fisheries and two research surveys. Bubble size is relative to the absolute residual value; positive residuals shown as full circles, negative as empty circles.
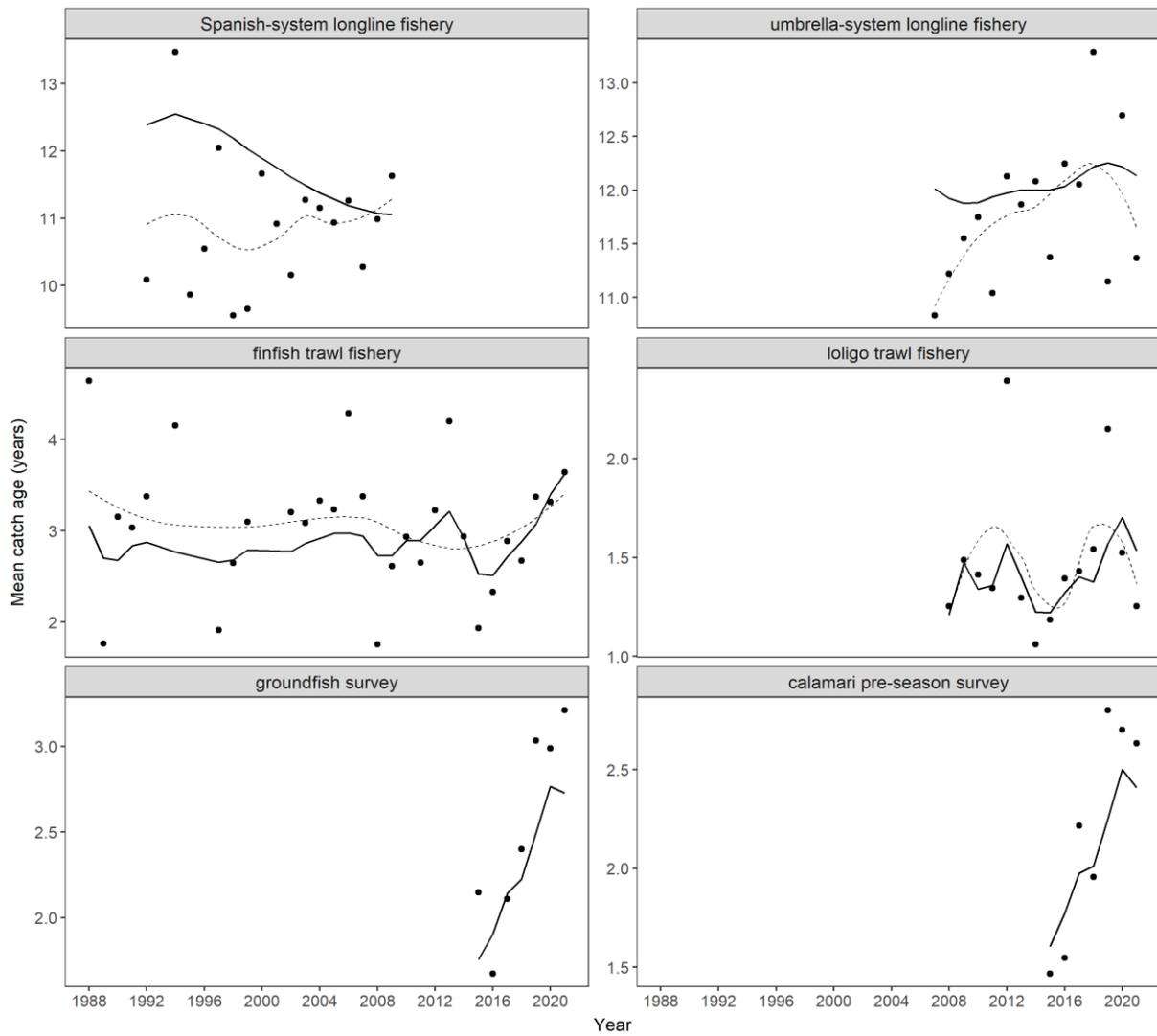
Figure A.12. Model fits (solid lines) to the observed toothfish mean catch-at-age data for four fisheries and two research surveys (black dots); dashed lines denote loess smoothers for the observations (span = 0.75); smoothers were omitted for the survey datasets due to the low number of data points.
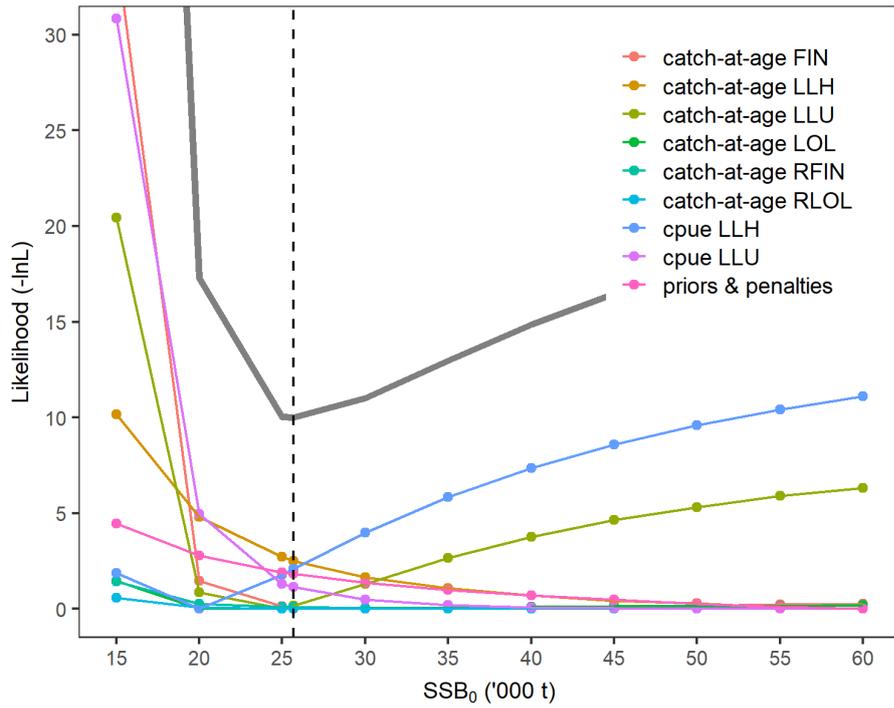
Figure A.13. Likelihood profiles (-2 log-likelihood) across a range of SSB$_0$ values. Values for individual data sets were rescaled to have a minimum of zero, while the total objective function was rescaled to a minimum of 10 for easier visualisation (solid grey line). The dashed vertical line denotes MPD estimate for SSB$_0$. LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-recruitment survey.
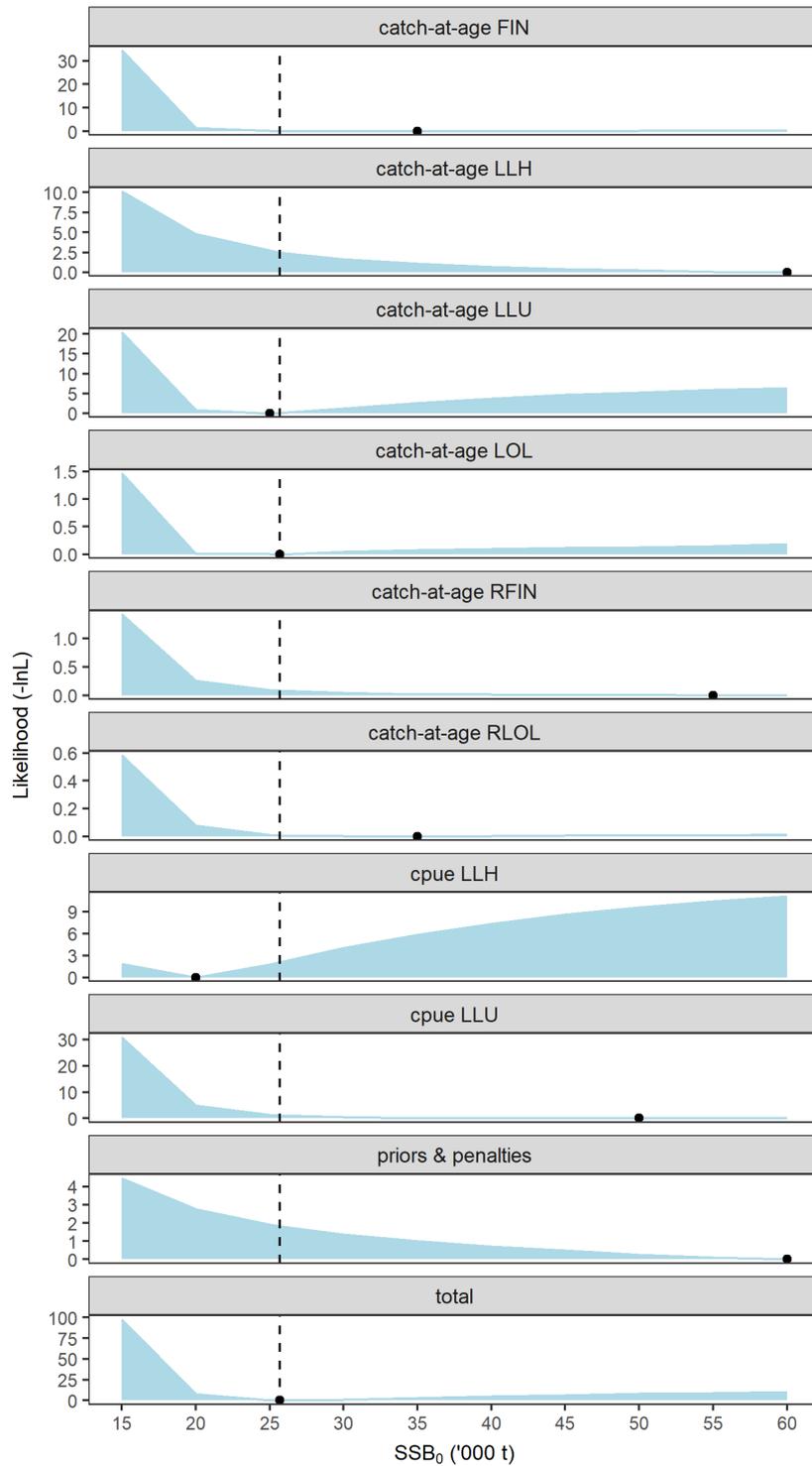
Figure A.14. Likelihood profiles across a range of SSB$_0$ values. Negative log likelihood values rescaled to have minimum of zero for each dataset. The dashed vertical lines denote MPD estimate for SSB0; dots denote the SSB0 values with the minimum negative log likelihood value for each dataset. LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-recruitment survey.
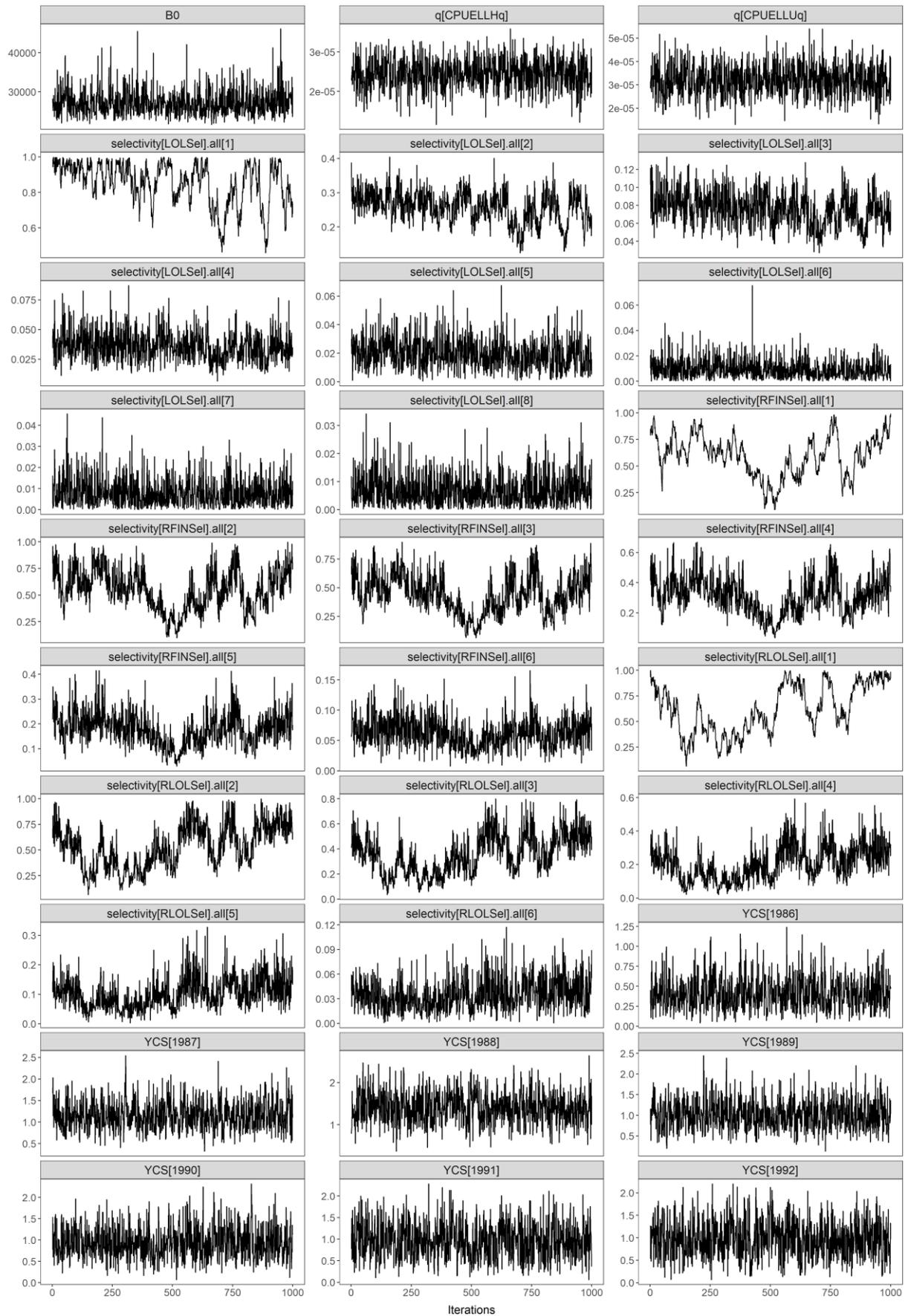
Figure A.15. MCMC posterior trace plots for the estimated parameters; note that the selectivity parameters for LLH, LLU and FIN have been omitted to keep the number of plots manageable (figure 1 of 2).
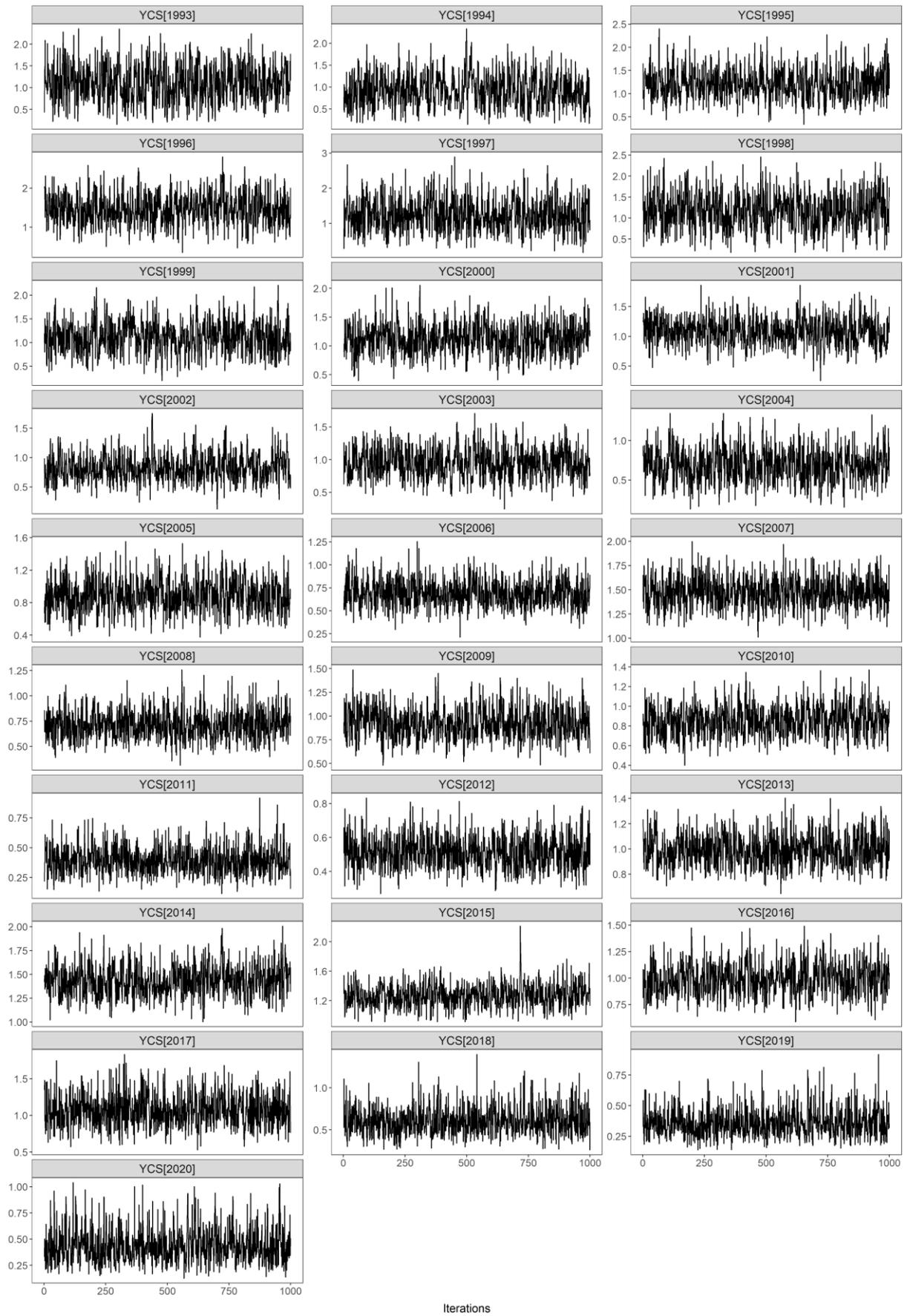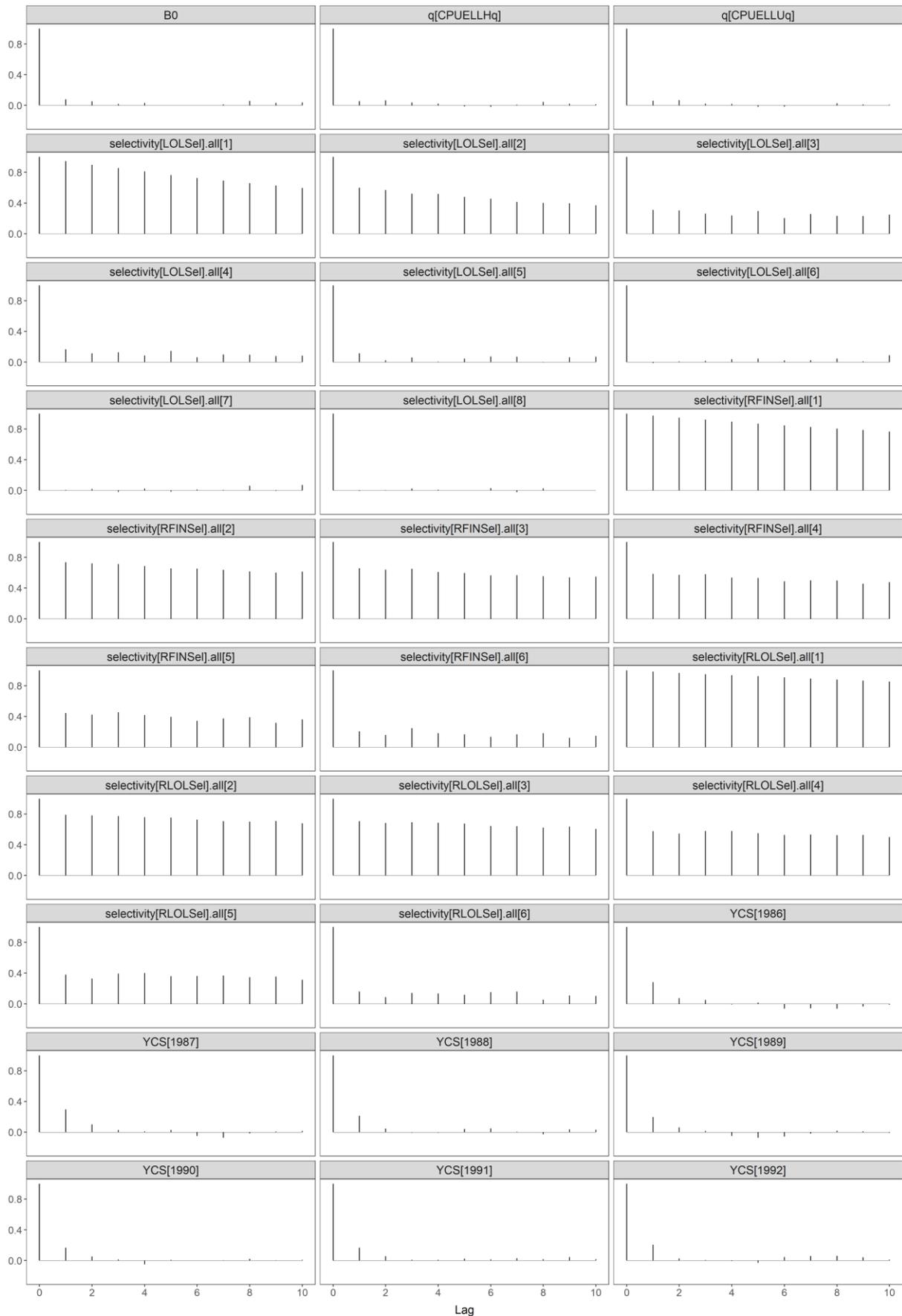
Figure A.15. Continued (figure 2 of 2).

Figure A.16. MCMC autocorrelation lag plots for the estimated parameters; note that the selectivity parameters for LLH, LLU and FIN have been omitted to keep the number of plots manageable (figure 1 of 2).
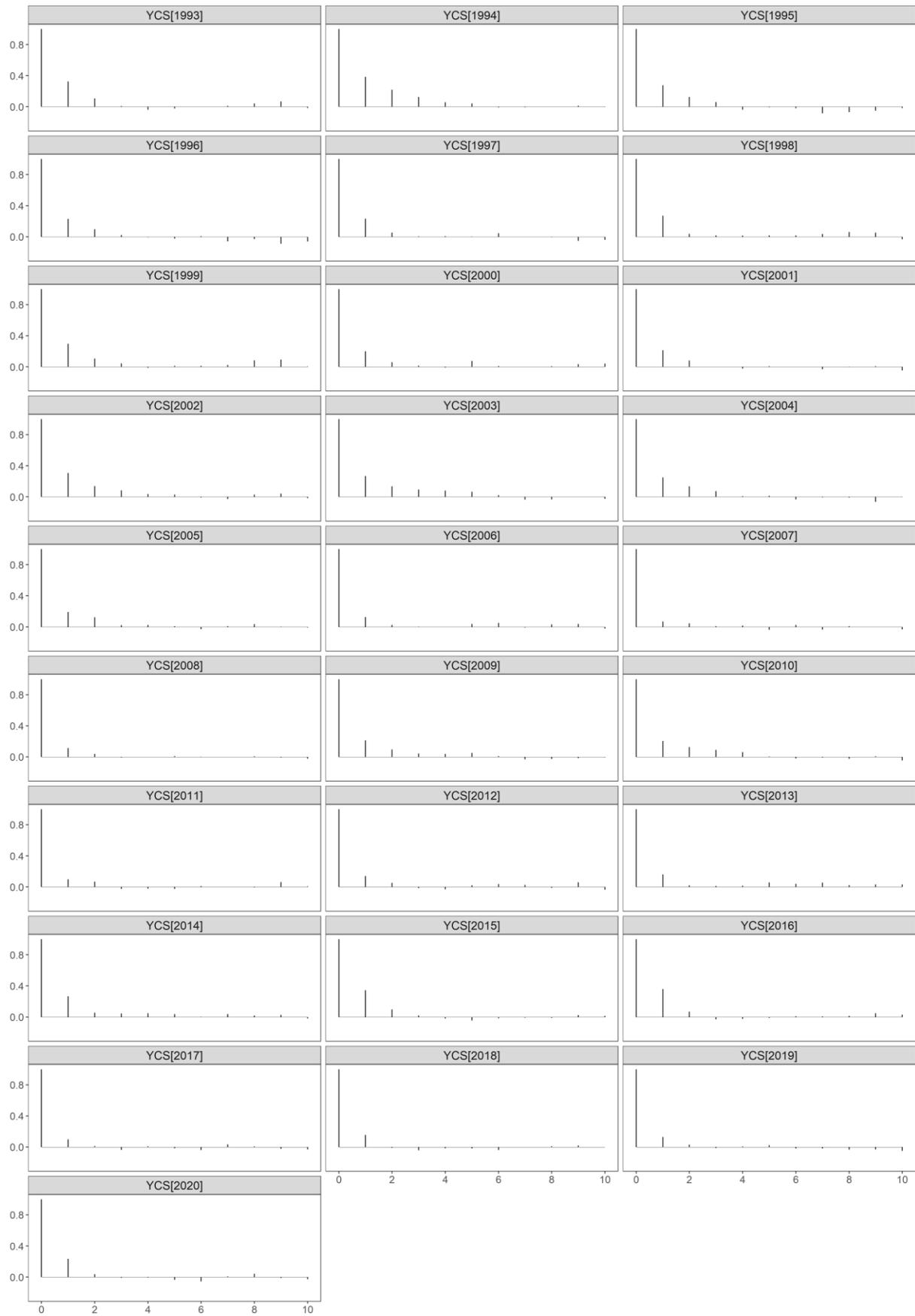
Figure A.16. Continued (figure 2 of 2).

## Appendix 3. Objective function contributions

Table A.2. Contributions to the objective function in the MPD model run. LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-recruitment survey.

| Objective function components | Values |
|---|---|
| **Observations** | |
| CPUE $_{LLH}$ | -17.9 |
| CPUE $_{LLU}$ | -20.5 |
| Catch-at-age $_{FIN}$ | 161.7 |
| Catch-at-age $_{LLH}$ | 436.9 |
| Catch-at-age $_{LLU}$ | 603.7 |
| Catch-at-age $_{LOL}$ | 67.6 |
| Catch-at-age $_{RFIN}$ | 51.4 |
| Catch-at-age $_{RLOL}$ | 41.2 |
| | |
| **Priors** | |
| $SSB_0$ | 10.2 |
| YCS | -24.1 |
| q $_{LLH}$ | -10.6 |
| q $_{LLU}$ | -10.3 |
| All selectivity priors | 0.0 |
| | |
| **Penalties** | |
| YCS $_{MEAN\_1}$ | 7.2 |
| Selectivity ogive smoother $_{LOL}$ | 0.3 |
| Selectivity ogive smoother $_{RFIN}$ | 0.5 |
| Selectivity ogive smoother $_{RLOL}$ | 0.1 |
| All catch limit penalties | 0.0 |
| | |
| **Total objective function** | 1297.5 |

## Appendix 4. Harvest control rules

Based on the CASAL model output, the following decision matrix of harvest control rules has been established to manage the Falkland Islands toothfish longline fishery (Farrugia and Winter 2018, 2019):

1. *Expansion range:* If the ratio of $SSB_{current}$/$SSB_0$ has remained above the upper target reference point (45%) for 3 consecutive years and the SSB projection with the current TAC shows no decrease below 45% for at least 10 years (one generation) under precautionary assumptions, the Director may authorize an increase in longline TAC to a level that continues to show no projected $SSB_{current}$/$SSB_0$ decrease to below 40% (trigger point) for at least 10 years under precautionary assumptions.

2. *Target range:* If the ratio of $SSB_{current}$/$SSB_0$ is between 40% and 45% (within the target range), current longline TAC is reviewed in relation to stock trends. Current TAC may be maintained if $SSB_{current}$/$SSB_0$ has increased from the previous assessment, or if the SSB ratio projection shows a level status under precautionary assumptions. TAC may not be increased, but it may be decreased if age-structure distributions anticipate weak recruitment.

3. *Trigger point and range:* If the ratio of $SSB_{current}$/$SSB_0$ falls to ≤ 40% (trigger point), longline TAC will be decreased to a level that projects an increasing SSB trend under precautionary assumptions. The magnitude of the proposed TAC reduction will be examined using three methods (adapted from ICES, 2017):

    a. Indexed to the reduction of the MSY estimates:
    $$TAC_{year} = TAC_{year-1} * \left( MSY_{year}/MSY_{year-1} \right)$$
    b. Indexed to the reduction of the SSB estimates:
    $$TAC_{year} = TAC_{year-1} * \left( SSB_{year}/SSB_{year-1} \right)$$
    c. Indexed to the reduction in SSB ratios:
    $$TAC_{year} = TAC_{year-1} * \left( SSB\ ratio_{year}/SSB\ ratio_{year-1} \right)$$

    TACs obtained from all three methods will be projected forward in the stock assessment model and the trends in SSB will be compared. The final method will be chosen based on it returning the SSB ratio to above 40% within 10 years (one generation) of the SSB ratio falling below 40%. If more than one method meets this requirement, the chosen method will also depend on discussions between the Fisheries Department and industry.

4. *Limit reference point:* If the ratio of $SSB_{current}$/$SSB_0$ is ≤ 20%, the longline fishery will be closed pending comprehensive evaluation of conditions required to rebuild the stock. The Director may authorize test fishing to measure biological parameters of the stock, subject to close monitoring by the Fisheries Department.