

2020 Alternative Stock Assessments Report

Patagonian toothfish (*Dissostichus eleginoides*)



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Summary

1. This report provides an updated alternative stock assessment of Patagonian toothfish in Falkland Islands waters, using data through year 2020. Assessment was done using a Bayesian surplus production model framework JABBA (Winker et al. 2018).
2. Overall, the model showed a negative stock status trend in ~1995-2006, which reverted to a levelled trend following the introduction of TAC system in 2006. The estimated current biomass B_{2020} is 7.7% below B_{MSY} , and the current fishing mortality F_{2020} is 1.2% below F_{MSY} . Annual catches have not surpassed the estimated maximum sustainable yield (MSY) of 1,474 tonnes per year since 2018, indicating sustainable exploitation of the stock.
3. JABBA produced substantially lower absolute and relative biomass estimates compared to the CASAL assessment, but only moderately lower MSY. Even though JABBA estimates were lower, the MSY still suggested that the stock was exploited sustainably in the most recent years (2018-2020).
4. The age-structured model in CASAL remains a preferred approach to toothfish stock assessment in Falkland Islands as it integrates a wider range of data, i.e. takes explicitly into account somatic growth, reproduction, lagged effects of recruitment, natural mortality and multiple fishery-specific selectivities.
5. Findings of this report provide insight into the usefulness of JABBA data-poor approach for other species /stocks managed by FIFD, where data limitations prevent the use of integrated age-structured assessment.

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 in 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 spawning peak in May, and a major peak in July to August (Laptikhovskiy et al. 2006). The juveniles (<10 cm TL) are found on the Patagonian shelf at depths <100 m (Arkhipkin and Laptikhovskiy 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 (Laptikhovskiy et al. 2008).

The Falkland Islands toothfish longline fishery began in 1992 as an exploratory fishery and became an established fishery in 1994 (Laptikhovskiy 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 being 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 finfish fishery toothfish is a commercially valuable bycatch, while in calamari fishery it is usually discarded, due to the small size of the specimens (20-40 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).

Management of the Falkland Islands toothfish fishery has relied on an integrated age-structured stock assessment implemented in CASAL software (Bull et al. 2012) since 2012. Although CASAL has become the standard approach to toothfish stock assessment in the Southern Ocean (Hillary et al. 2006, Dunn and Hanchet 2010, Ziegler and Welsford 2015, Dunn 2019), examining alternative models can provide a useful comparison (Cadrin and Dickey-Collas 2015), and either increase our confidence in CASAL outputs, or prompt re-evaluation of its assumptions. The current report provides an updated stock assessment of toothfish in Falkland Islands waters using a Bayesian surplus production model framework JABBA (Winker et al. 2018), using data through year 2020. The JABBA outputs are compared with the results of integrated age-structured toothfish stock assessment done in CASAL for year 2020 (Skeljo and Winter 2021).

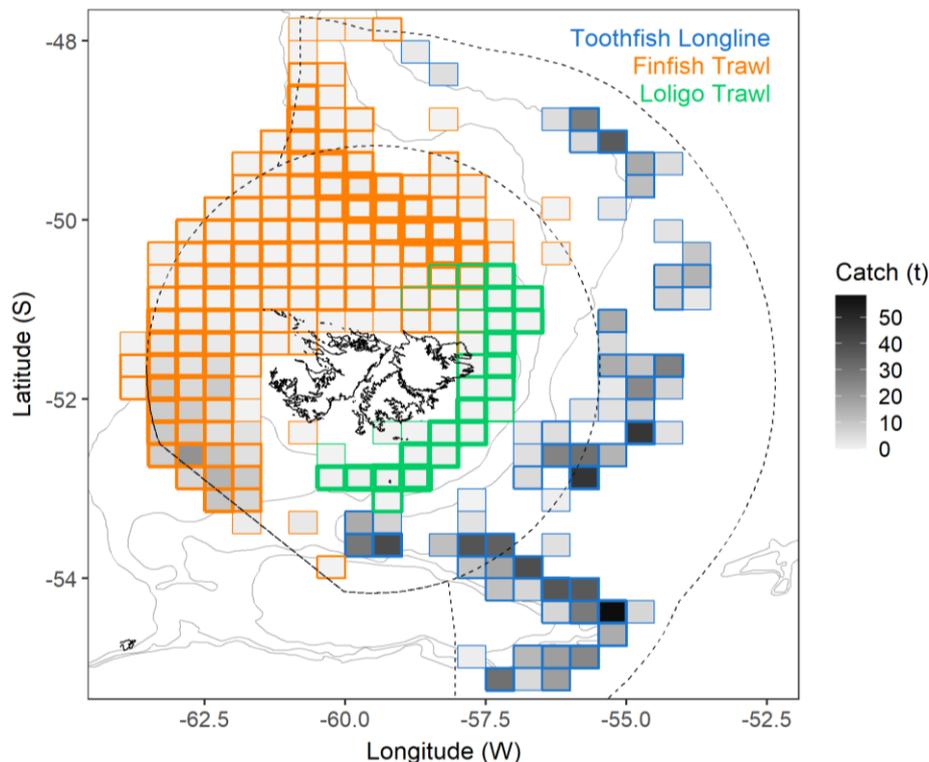


Figure 1. Spatial distribution of toothfish catch and effort by fisheries in 2020. Thickness of grid lines is proportional to the number of vessel days; greyscale is proportional to the toothfish catch biomass in tonnes.

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 a 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). However, toothfish population structure across the Patagonian Shelf is less certain, and it is not yet clear whether there are several separate self-sustaining populations or one large meta-population (Parker 2015). The existence of separate spawning populations south of Diego Ramirez Islands in Chilean waters and the eastern Burdwood Banks in Falkland Islands waters has been proposed (Laptikhovsky et al. 2006, Arana 2009); with otolith microchemistry analysis indicating that larvae settling on the Falkland Shelf originate from a combination of these two spatially distinct areas (Ashford et al. 2012). Early 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.

In order to build on these early studies and to get a better understanding of the toothfish stock structure within Patagonian Shelf (and especially Falkland Islands waters) a range of methodologies were employed by FIFD, most notably: otolith shape analysis, life-history aspects, otolith microchemistry analysis, genetic studies and the re-establishment of a large-scale tag-recapture program using conventional and satellite tags (Randhawa and Lee 2016).

Results obtained from otolith shape analyses revealed high site fidelity of adult fish across localised regions of the Patagonian Shelf, including southern Chile, the Burdwood Bank, and the continental slope to the north-east of the Falkland Islands (Lee et al. 2018). However, the extent to which these groups functioned as discrete stocks remained unclear. Recent work was therefore 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. Results highlighted further research objectives aimed at (1) the identification of the extent of any potential adult migratory behaviour from non-spawning to spawning areas amongst the adult component of the population, and (2) the extent that temporally variable discrete spatial groups in the shelf-based population arise from a single or multiple spawning areas.

The primary aim of the tag-recapture program that was re-established in June 2016 was to improve our understanding of the movement patterns of toothfish within the region; and to quantify the exchange taking place between adults on the northern and eastern slope, and the spawning grounds on the Burdwood Bank. While the initial medium term (3-years) aim to tag 3000 fish was achieved, the program was extended for a further 4-year time period (Lee and Skeljo, 2020). The most recent tagging survey took place in January 2021, with ~700 toothfish tagged (Skeljo and Pearman 2021). Preliminary analyses based on results over the first 5-years of data are to be undertaken during 2021-2022 in order to improve our understanding of objective 1 defined above.

In order to meet objective 2, otolith microstructures and associated trace elemental composition are being analysed for newly settled age 0+ toothfish from three regions of abundance between 2014 and 2017. Otoliths extracted from progressive cohorts of age 1+ (2015-2018) and 2+ (2016-2019) year old toothfish were also sampled from four key regions of abundance that overlap with the recruitment areas and processed for elemental analyses in the same way (Lee, FIFD, in progress). The results of this study should provide us with an improved understanding of the population structure of Patagonian toothfish across the shelf regions around the Falkland Islands.

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

2.1. Data

Three datasets were used as information for the JABBA stock assessment model: total annual removals by combined fisheries (1987-2020) and catch-per-unit-of-effort (CPUE) time series for Spanish- (1996-2007) and umbrella-system (2007-2020) longline fisheries.

CPUE

Although CPUE data were available for all four fisheries (Spanish longline, umbrella longline, finfish trawl, calamari trawl), only longline CPUE were 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 the vessels). Since data aggregated by day were used in the current 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 assessment (Skeljo and Winter 2020). 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 the 'tagging trips' and from the longline sets at depths <600 m were removed from the analysis. Tagging trips were removed 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 longlining is prohibited at depths <600 m, and the corresponding sets were experimental fishing aiming to collect the 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 model (GLMM), providing a time series of CPUE values (with the associated standard errors) which were assumed relative abundance indices ([Appendix 1](#)).

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, and (c) catches lost to undetected whale depredation.

All reported toothfish catches taken in longline (targeted catch) and trawl fisheries (bycatch) were used, going back to 1987.

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 fishing as a percentage of reported catch in 1980-2003. For years since 2003, we took grey-literature estimates (e.g. CCAMLR 2010) that IUU fishing in the southern oceans has decreased significantly and

assumed IUU to be 5% of the reported catch. The same IUU data were used in the previous assessment (Skeljo and Winter 2020).

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.

2.2. JABBA model setup

JABBA is a Bayesian state-space surplus production model framework, based on the generalized Pella-Tomlinson surplus production function (Pella and Tomlinson 1969) of the form:

$$SP_t = \frac{r}{m-1} B_t \left(1 - \left(\frac{B_t}{K} \right)^{m-1} \right),$$

where r is the intrinsic rate of population growth at time t , K is the carrying capacity, B is stock biomass at time t , and m is a shape parameter that determines at which B/K ratio maximum surplus production is attained (hereafter B_{MSY}/K). The Pella-Tomlinson function reduces to the Schaefer function if the shape parameter $m = 2$, and to the Fox function if m approaches 1. In the current model surplus production was assumed maximized at $B_{MSY}/K = 0.4$, the default setting for the Pella-Tomlinson function and the same value used in the last year's assessment. This ratio was converted into Pella-Tomlinson shape parameter $m = 1.188$, according to the equation:

$$\frac{B_{MSY}}{K} = m \left(\frac{1}{1-m} \right).$$

JABBA estimates fisheries reference points, relative stock biomass and exploitation from the catch and abundance indices time series and the priors for the intrinsic rate of population increase r , the carrying capacity K , and the relative biomass B/K at the start of the available catch time series. It can also estimate process variance σ_{proc}^2 , and additional observation variance for the abundance indices time series σ_{est}^2 . In JABBA, the total observation variance σ_{obs}^2 is separated into three components that are additive in their squared form (Francis et al. 2003), with the total observation variance for abundance index i and year y given by:

$$\sigma_{obs,y,i}^2 = \hat{\sigma}_{SE,y,i}^2 + \sigma_{fix}^2 + \sigma_{est,i}^2$$

where $\hat{\sigma}_{SE}$ are standard error estimates associated with the abundance indices and derived externally from the CPUE standardization model, σ_{fix}^2 is a fixed input variance, and σ_{est}^2 is a model estimable variance. In the current assessment, $\hat{\sigma}_{SE}$ for each annual abundance index were provided to the model, and σ_{fix} was set to 0.2, a commonly used value suggested by Francis et al. (2003). Adding a fixed observation error σ_{fix} to externally estimated standard errors for abundance indices $\hat{\sigma}_{SE}$ is common practice to account for additional sampling errors associated with abundance indices

(Maunder and Piner 2017), such as those caused by year-to-year variation in catchability (Francis et al. 2003).

Priors used in the model are provided in Table 1. Key priors (r , K and B_{1987}/K) are stock-specific and were defined based on the expert knowledge of the stock status (K and B_{1987}/K) or estimated from the species life-history parameters (r). Priors for variances (σ_{proc}^2 , σ_{est}^2) and catchability coefficients ($q_{Spanish}$, $q_{umbrella}$) were set to the default JABBA settings. Once the priors were defined, the model was executed in R environment (R Core Team 2020) using the most recent version of package JABBA (R package version 2.1.6. <https://github.com/jabbamodel/JABBA/>; Winker et al. 2021). The Bayesian posterior distributions of all quantities of interest are estimated by means of a Markov Chains Monte Carlo (MCMC) simulation. Two MCMC chains with 30,000 iterations each were used, with a burn-in of 5,000 for each chain and a thinning rate of five iterations. MCMC chains were investigated for evidence of non-convergence using trace plots and convergence tests of Geweke (1992) and Heidelberger & Welch (1983) as implemented in the *coda* R package (Plummer et al. 2006).

To evaluate model goodness-of-fit, the residual patterns were inspected visually, and the root mean square error (RMSE) was calculated; a relatively small RMSE (≤ 0.3) indicates a reasonably precise model fit to relative abundance indices (Winker et al. 2018). A full JABBA model description, including formulation and state-space implementation, prior specification options and diagnostic tools is available in Winker et al. (2018).

Table 1. Parameter priors used in JABBA model run, with a brief description of the selection criteria.

Parameter	Prior	Description
r	log-normal; $\mu = 0.143$, $sd = 0.105$	Estimated at species level (<i>Dissostichus eleginoides</i>) using <i>FishLife2.0</i> R package (Thorson 2019)
K	log-normal; $\mu = 60,000$, $cv = 1$	Used in the previous assessment (Skeljo and Winter 2020); Defined to be roughly comparable with the prior used in CASAL
B_{1987}/K	log-normal; $\mu = 1$, $cv = 0.1$	Used in the previous assessment (Skeljo and Winter 2020); Stock is assumed to have been nearly unexploited at this time, based on the very low reported catches
σ_{proc}^2	inverse-gamma (4, 0.01) *	Model default
σ_{est}^2	inverse-gamma (0.001, 0.001) *	Model default
$q_{Spanish}$, $q_{umbrella}$	uniform (1e-30, 1e3) **	Model default

* Inverse-gamma distribution is defined by two scaling parameters; ** uniform distribution is defined by range

3. Results

3.1. Model diagnostics

The model diagnostics plots are given in [Appendix 2](#). The MCMC convergence tests of Geweke (1992) and Heidelberger & Welch (1983) were passed by all estimated parameters. Adequate convergence of the MCMC chains was also corroborated by visual inspection of trace plots, which showed good mixing in general (Figure A.3).

The model fit to the standardized CPUE data was very good for both Spanish- and umbrella-system fisheries (Figure A.4). No patterns were observed in the residuals (Figure A.5), and the goodness-of-fit statistic indicated good model fit (RMSE = 13.4%).

The comparison of posterior distributions and prior densities of key estimated parameters is given in Figure A.6. Posterior distribution of K was much narrower compared to its prior, and the

resulting small PPVR (posterior to prior variances ratio) indicates that the input data were very informative about K . The opposite is true of r , with both PPMR (posterior to prior means ratio) and PPVR close to 1, which suggests that the posterior was largely informed by the prior rather than the input data.

3.2. Model estimates

The key output parameters and stock status estimated by JABBA are summarised in Table 2. The carrying capacity was estimated as $K = 30,306$ t, and the estimated biomass declined from $0.946K$ (28,437 t) in 1987 to $0.369K$ (11,106 t) in 2020. The absolute biomass B and the relative biomass B/K and B/B_{MSY} trends showed a sharp decline in 1995-2001, followed by a moderate decline in 2001-2006, and reverting to a levelled trend afterwards. This was related to the sharp increases in relative fishing mortality F/F_{MSY} in 1994 and more prominently in 1996-1999, followed by a period of sustained high F/F_{MSY} in 2000-2005. Since 2006, F/F_{MSY} has been slightly to moderately high and on a fluctuating, slowly decreasing trend; in three most recent years it has been very close to the sustainable level baseline ($F/F_{MSY} = 1$) (Figure 2).

Relationship between B/B_{MSY} and F/F_{MSY} is illustrated using the Kobe plot (Figure 3), showing that the overfishing ($F/F_{MSY} > 1$) in 1999-2005 (orange area on the plot) quickly reduced the biomass to a level slightly below B_{MSY} . Since 2006, F/F_{MSY} fluctuated closer to the sustainable baseline, and the biomass consequently remained roughly the same until 2020. The fishing mortality decline in 2006 coincided with the introduction of TAC system to the toothfish longline fishery. The estimated current biomass B_{2020} is 7.7% below B_{MSY} , and the current fishing mortality F_{2020} is 1.2% below F_{MSY} . Taking into account the uncertainty of this estimate (grey credibility intervals on the Kobe plot), there is 38.1% probability that the toothfish stock was not overfished ($B > B_{MSY}$) and not experiencing overfishing ($F < F_{MSY}$) in 2020 (green area on the Kobe plot). If only the fishing mortality is considered, as this is something that can be regulated, the cumulative probability of stock not being subjected to overfishing in 2020 was 51.5% (green and yellow areas on the Kobe plot).

According to the Pella-Tomlinson surplus production function, biomass that would produce maximum surplus production (i.e. maximum sustainable yield, MSY) was estimated at $B_{MSY} = 12,122$ t, with the corresponding $MSY = 1,474$ t. In 1998-2005 catches were above the median MSY confidence intervals, leading to a decline in toothfish biomass (Figure 4). Since 2006 catches have been within the 95% confidence intervals of the MSY in each year, and for the last three years, below the median MSY. Biomass is expected to increase if catch were maintained at current levels, considering that surplus production is larger than recent catches.

Table 2. Summary of parameters and stock status estimates.

Parameter	median	95% CI
r	0.144	0.118 - 0.176
K	30,306 t	24,833 - 46,045 t
B_{1987}	28,437 t	22,339 - 44,581 t
B_{2020}	11,106 t	5,821 - 29,889 t
B_{1987}/K	0.946	0.809 - 1.031
B_{2020}/K	0.369	0.212 - 0.675
MSY	1,474 t	1,248 - 2,189 t
B_{MSY}	12,122 t	9,933 - 18,417 t
F_{MSY}	0.121	0.099 - 0.148
B_{2020}/B_{MSY}	0.923	0.531 - 1.687
F_{2020}/F_{MSY}	0.988	0.367 - 1.825

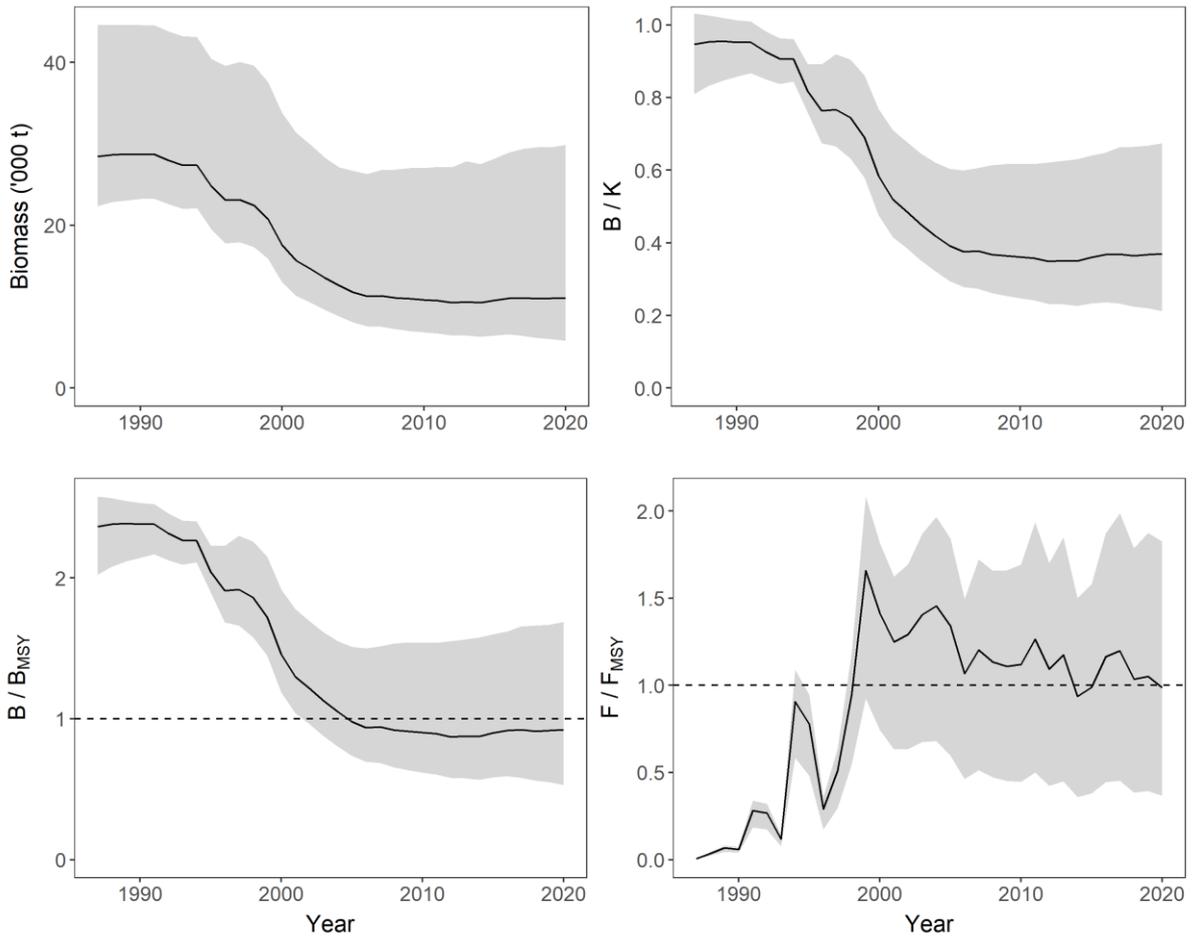


Figure 2. Estimated trends in absolute biomass (top left), biomass relative to K (top right), biomass relative to B_{MSY} (bottom left) and fishing mortality relative to F_{MSY} (bottom right). Solid black lines are medians and shaded areas denote 95% confidence intervals.

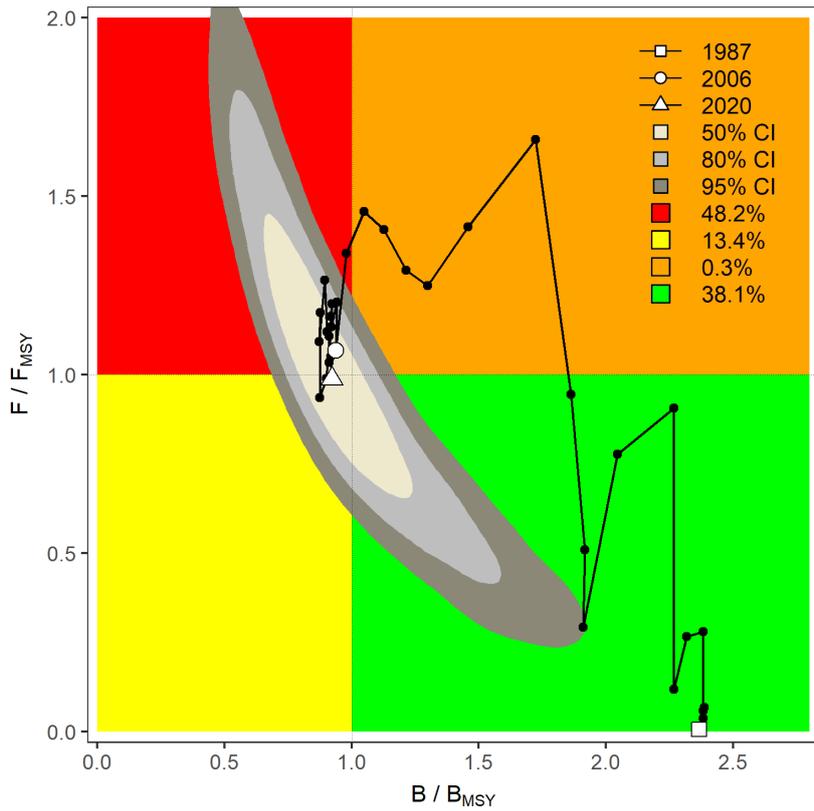


Figure 3. Kobe phase plot showing estimated trajectory of B/B_{MSY} and F/F_{MSY} for the toothfish stock in 1987-2020. Grey shaded areas denote the 50%, 80%, and 95% credibility intervals for the last assessment year. The probability of the last year estimate falling within each quadrant is indicated in the figure legend.

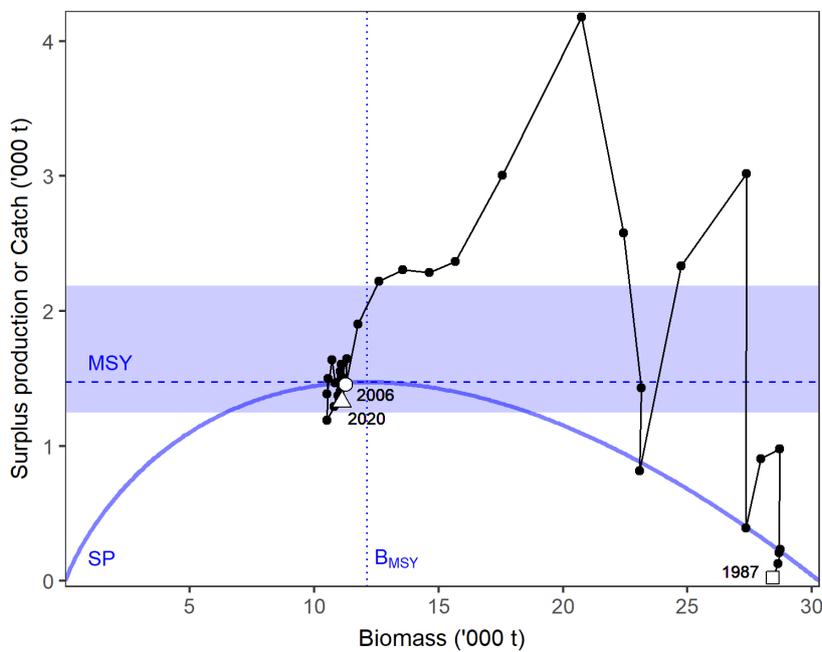


Figure 4. Surplus-production phase plot showing Pella-Tomlinson curve SP (solid blue line) and catch/biomass trajectory for the toothfish stock in 1987-2020 (black line). Catches on the SP curve would maintain the biomass, catches above the curve will shrink future biomass, and catches below the curve allow future biomass to increase. Year 2006 (white dot) marks the introduction of the TAC system to the longline fishery. Estimated MSY (dashed blue line) and B_{MSY} (dotted blue line) are added for reference. Blue shaded area denotes 95% confidence intervals of the MSY .

3.3. Retrospective analysis

The retrospective analysis was conducted by successively removing one to six final years of data from the 2020 model and rerunning the analysis, in order to evaluate whether there were any strong changes in model results based on data availability. To quantify the bias between the retrospective models, the commonly used formulation of *Mohn's rho* statistic (Mohn 1999) was computed (Hurtado-Ferro et al. 2014). The estimated Mohn's rho for B (-0.12), B/K (-0.10), B/B_{MSY} (-0.10) and F/F_{MSY} (0.17) fell within the acceptable range of -0.15 and 0.20 (Hurtado-Ferro et al. 2014, Carvalho et al. 2017) and consequently indicated that the retrospective pattern was relatively small. Retrospective plots showed more optimistic stock status trajectories when data up to 2018-2020 were included in the analysis, compared to using data up to 2014-2016 (Figure 5). This is likely a reflection of generally higher annual abundance indices recorded since 2017, coinciding with the introduction of the new vessel to the fishery (*CFL Gambler* was replaced by *CFL Hunter*).

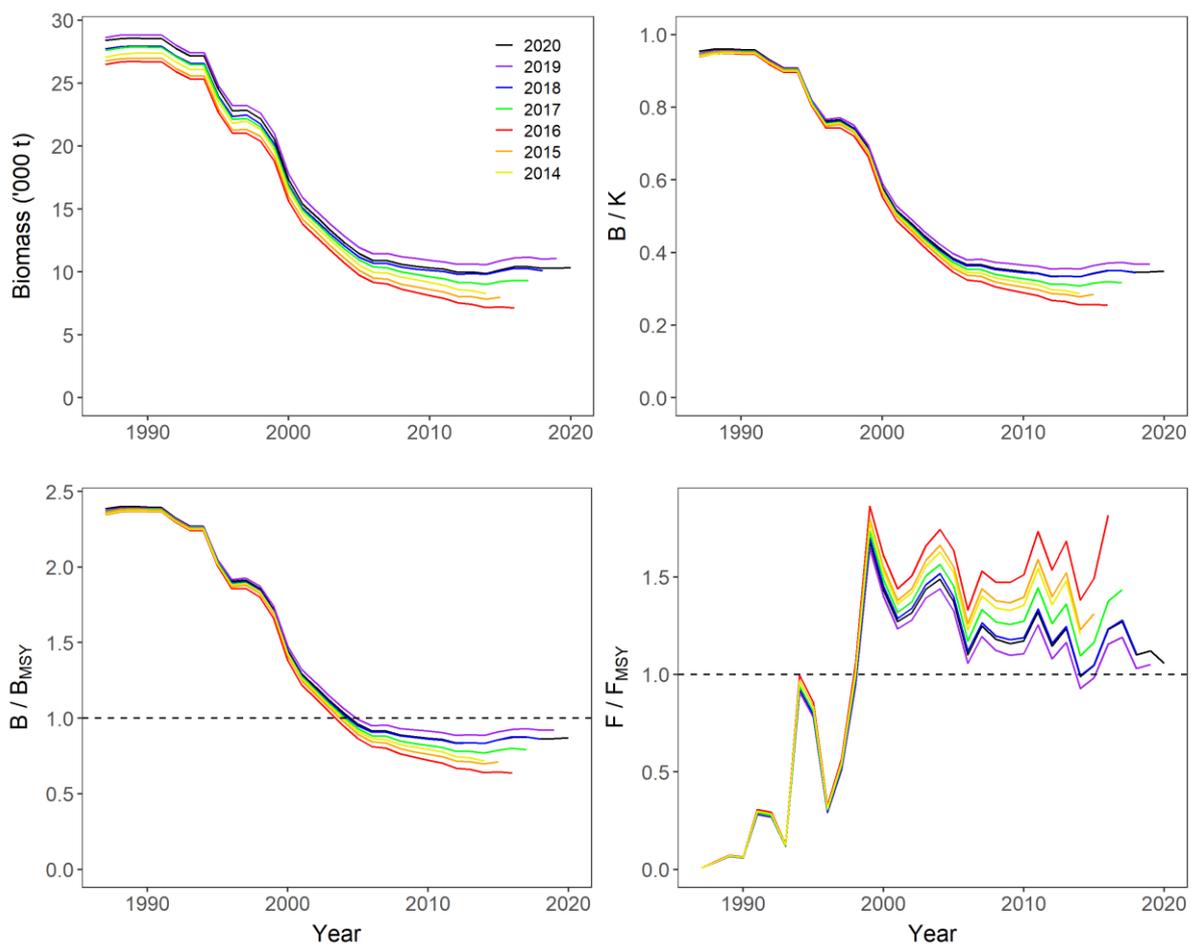


Figure 5. Estimated trends in biomass, B/K , B/B_{MSY} and F/F_{MSY} for the 2020 model (black line) and six retrospective model runs. The numeric label indicates the year up to which individual retrospective model was run (inclusive).

3.4. Comparison of JABBA and CASAL assessment

As CASAL does not provide estimates of K , the comparison with JABBA was approximated by setting K equivalent to B_{1987} for the CASAL model (i.e. the total biomass in 1987 was assumed to be at the carrying capacity). Overall, JABBA estimated substantially lower absolute and relative biomass compared to the CASAL assessment, but only moderately lower MSY (Table 3). Although the absolute biomass estimates were different, the time-series trends were very similar, with the exception of the last ten years when JABBA estimated levelled trend and CASAL estimated slowly decreasing trend in biomass (Figure 6). A simple linear regression between the CASAL time series and the JABBA time series of biomass resulted in a very good fit ($R^2 = 0.97$), and confirmed that the trends are indeed very similar to each other for most of the time series, and diverge only in the later years (~2011-2012).

Table 3. Comparison of the selected output parameters estimated by CASAL and JABBA stock assessment models for the toothfish in Falkland Islands waters (1987-2020). All biomass estimates are given in tonnes.

Parameter	CASAL	JABBA
K	$\sim B_{1987}$	30,306 (24,833 - 46,045)
B_{1987}	60,652 (55,187 - 295,228)	28,437 (22,339 - 44,581)
B_{2020}	31,868 (30,430 - 316,679)	11,106 (5,821 - 29,889)
B_{2020}/K	0.525 *	0.369 (0.212 - 0.675)
MSY	1,850 (1,637 - 7,550)	1,474 (1,248 - 2,189)

* Calculated as B_{2020}/B_{1987}

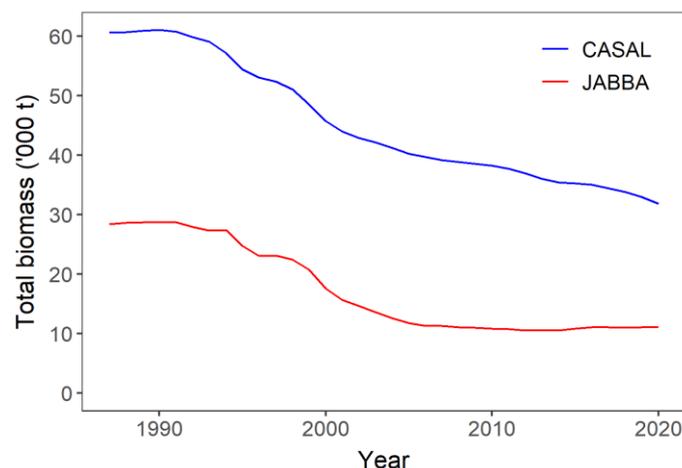


Figure 6. Absolute biomass trends estimated by statistical-catch-at-age model in CASAL, and surplus production model in the JABBA framework.

4. Discussion

In this assessment, JABBA (*Just Another Bayesian Biomass Assessment*) framework was used to fit a generalised Bayesian surplus production model to the catch and CPUE data belonging to the Falkland Islands toothfish stock. Surplus production models (SPMs) are among the least data demanding population models that can produce estimates of MSY and associated fisheries reference points, and despite a number of limitations (Maunder 2003, Punt and Szuwalski 2012), remain an integral tool for data-limited to -moderate stock assessments (Dichmont et al. 2016, Punt et al. 2015). The main limitation of SPMs is that they ignore the stock's size/age structure and therefore fail to account for dynamics in gear selectivity (Wang et al. 2014) and lagged effects of recruitment and mortality (Aalto et al. 2015, Punt and Szuwalski 2012), which can both lead to biased assessment results. However, SPMs have been considerably enhanced by the introduction of Bayesian methods with improved prior formulations, development of frameworks that allow incorporating both observation and process errors, and Bayesian state-space modelling approaches (Winker et al. 2018).

Compared to the 2019 JABBA assessment, this assessment uses revised longline catch and CPUE time series up to 2019, updated catch and CPUE for 2020, and updated prior for the intrinsic population growth rate r . Catch and CPUE revision has already been described (methods section in this report, also Skeljo and Winter 2021); regarding prior of r , in 2019 we defined it in terms of broad species resilience categories proposed by Froese et al. (2017). However, those authors recommend these categories as a starting point but advise the users to carefully consider all available information and then select the most suitable prior of r for the stock in question. Following their advice, in the current assessment prior of r was estimated using R package *FishLife*, release 2.0 (available online at (<https://github.com/James-Thorson/FishLife/releases/tag/2.0.0>)). *FishLife2.0* produces r estimates for selected species and/or higher taxonomic levels based on an integrated analysis of all life history parameters from *FishBase* (www.fishbase.org; Froese and Pauly 2000) and spawning-recruitment relationship data series from *RAM Legacy Database* (www.ramlegacy.org; Ricard et al. 2012). A full description of *FishLife2.0* model is available in Thorson (2019).

The revised and updated data resulted in a similar JABBA model estimates as in previous year, although slightly more optimistic regarding the absolute and relative biomass. On the other hand, estimates of r and MSY were somewhat lower, most likely due to the different prior of r used. Sensitivity analysis conducted as a part of the previous year's assessment (Skeljo and Winter 2020) showed that prior of r is the most influential prior in the analysis, and the current assessment's model diagnostics showed that the estimates of r were strongly influenced by the specified prior. This highlights the need to provide a well-informed species-specific prior, and in that respect R package *FishLife2.0* proved useful.

One of the main reasons for conducting alternative toothfish stock assessment in JABBA is to provide a comparison with the age-structured integrated stock assessment in CASAL. Overall, JABBA produced lower estimates of absolute biomass, relative biomass and MSY , but the model still suggested that the stock was exploited sustainably in the most recent years (2018-2020), with the total fishery removals being lower than the surplus production. The discrepancy between the two models was not surprising, as the type of data used can influence model estimates. In this case, the main difference in data used was the absence of composition data (i.e. catch-at-age) in the JABBA surplus production model, exempting the JABBA model from estimating fishery-specific selectivity patterns and optimizing population trends only on the abundance index. Catch-at-age data can be highly influential on the estimated biomass trajectory and may not be consistent with the overall trend in the abundance index (Wetzel and Punt 2011), which was indeed the case in the CASAL SCAA model used for comparison here (Skeljo and Winter 2021). The surplus production abundance index reflects biomass only and may particularly have caused the JABBA model to underestimate the potential of the juvenile year classes. In general, including multiple types of data such as composition data along with abundance indices allows for more complex population dynamics models as the basis of stock assessments, and has previously been shown to improve estimation performance (Magnusson and Hilborn 2007).

In conclusion, the age-structured model in CASAL should be a preferred approach to toothfish stock assessment in Falkland Islands, as it integrates wider range of data, i.e. takes explicitly into account somatic growth, reproduction, lagged effects of recruitment, natural mortality and multiple fishery-specific selectivities. However, findings of this report provide an insight into the usefulness of JABBA data-poor approach for other species/stocks managed by FIFD, where data limitations prevent the use of integrated CASAL assessment.

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Appendix 1. CPUE standardization

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CPUE data belonging to the commercial Spanish- and umbrella-system longline fisheries are the main source of information on stock abundance available to the current 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 has been slightly modified this year by employing a generalized linear mixed modelling approach (GLMM; Pinheiro and Bates 2000), an extension of the generalized linear modelling approach (GLM) used in the previous 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 were 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 three broad areas: (a) within the Falklands zone and south of 53.5° S (Burdwood Bank spawning area), (b) within the Falklands zone and north of 53.5° S, and (c) outside the Falklands zone. *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*, *Soak-time* and *Vessel*, and the model explained 21.9% 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 13.3% 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 change 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.

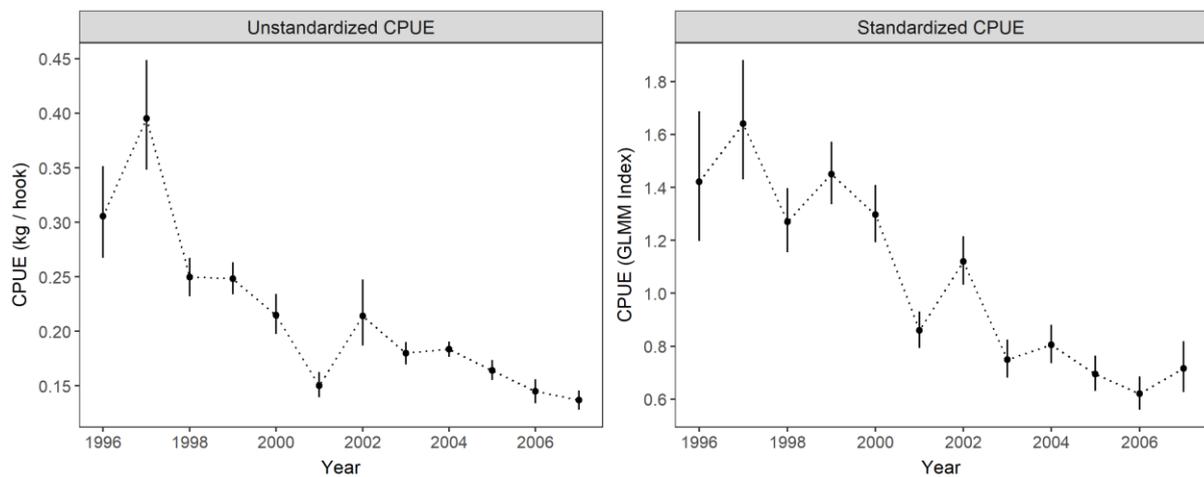


Figure A.1. Spanish-system longline unstandardized and standardized CPUE time series; black vertical lines correspond to 95% confidence intervals.

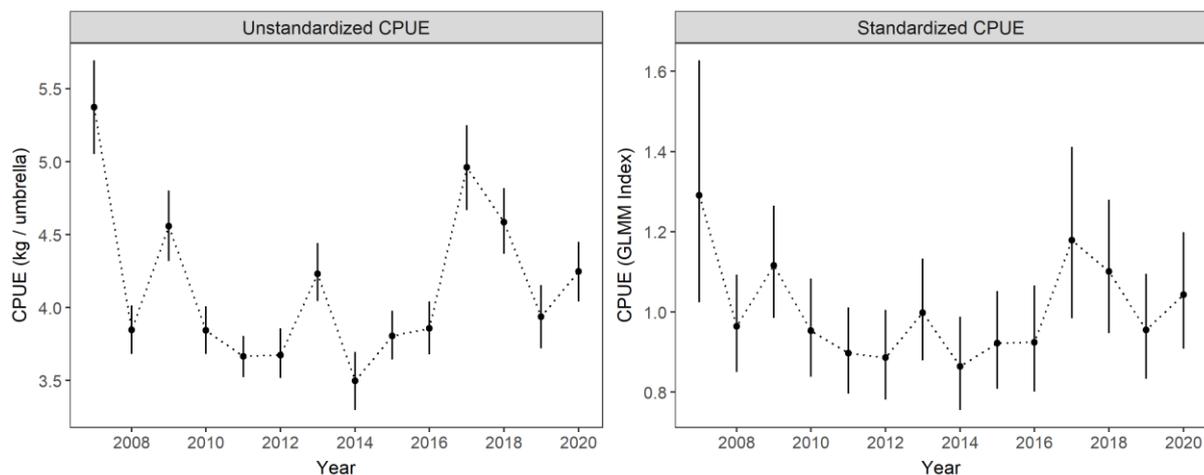


Figure A.2. Umbrella-system longline unstandardized and standardized CPUE time series; black vertical lines correspond to 95% confidence intervals.

Appendix 2. Diagnostics plots

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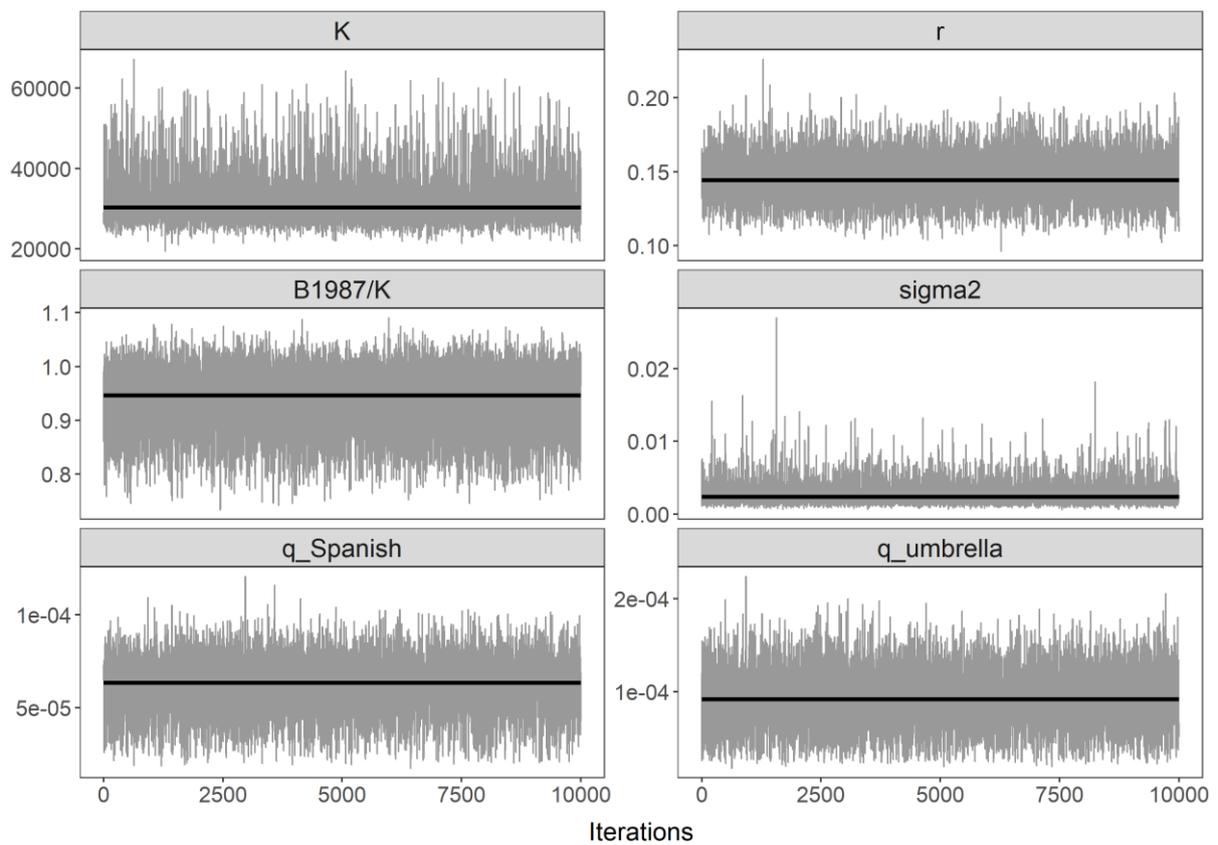


Figure A.3. MCMC posterior trace plots for the estimated parameters. Black line denotes the median.

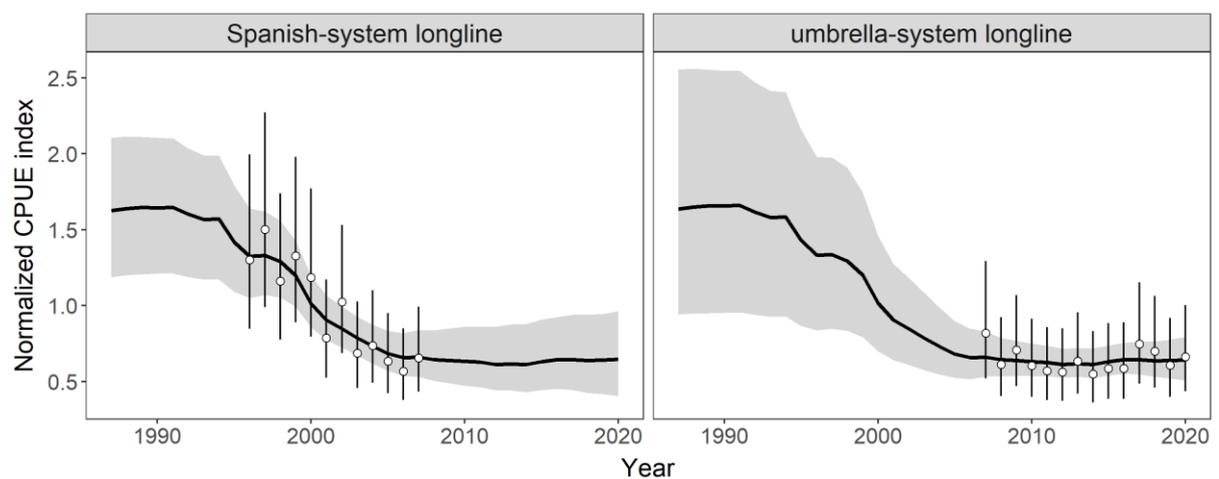


Figure A.4. Model fit (black line) to the normalised CPUE indices (white dots) for Spanish- and umbrella-system longline. Vertical lines denote 95% confidence intervals of the normalised CPUE indices; shaded areas denote 95% credibility intervals of the model fit.

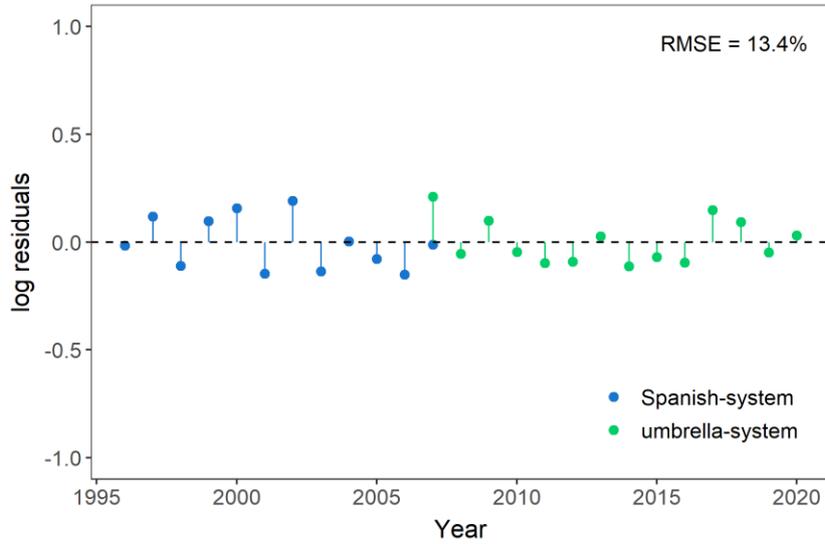


Figure A.5. Residuals from the model fit to the observed CPUE indices; for Spanish-system (blue dots) and umbrella-system longline (green dots). RMSE: root mean square error.

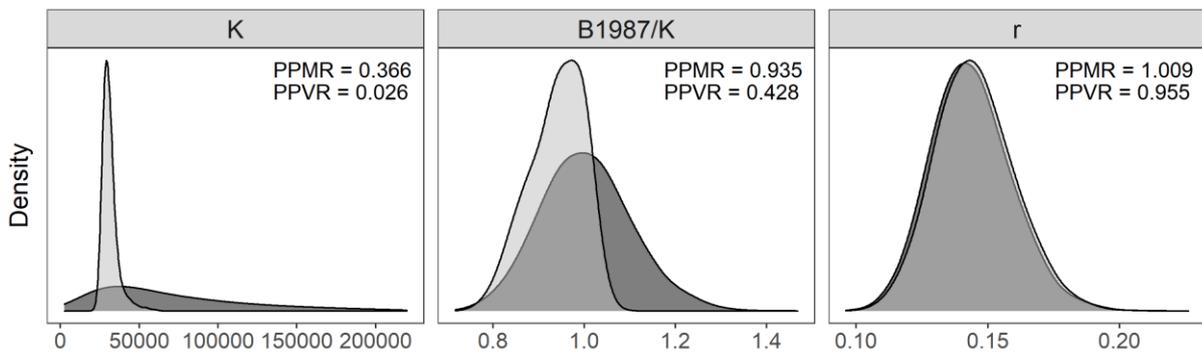


Figure A.6. Prior (dark grey) and posterior distributions (light grey) of key estimated parameters. PPMR: Posterior to Prior Means Ratio; PPVR: Posterior to Prior Variances Ratio.