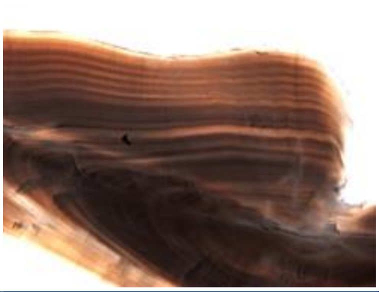
2019 Stock Assessment Report

Patagonian toothfish
(*Dissostichus eleginoides*)



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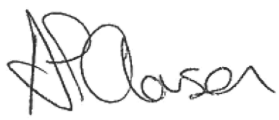

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Summary

1. This report provides an updated Bayesian age-structured stock assessment of *Dissostichus eleginoides* in Falkland Islands waters, using data through year 2019. Several changes were introduced in the 2019 model, regarding both data treatment and model assumptions. In addition, sensitivity of the model outputs to alternative modelling assumptions was investigated.
2. Current spawning stock biomass was estimated at 10,637 tonnes and the ratio of current spawning stock biomass to initial spawning stock biomass (SSB_{2019}/SSB_0) at 0.440. The estimated SSB_{2019}/SSB_0 was lower than SSB_{2018}/SSB_0 in 2019, but this was anticipated by the previous assessment's projection.
3. Projection from the current model showed that the spawning stock biomass will continue to decrease until 2023, reaching a minimum SSB/SSB_0 ratio of 0.424, before increasing back above the upper target reference point of 0.45 by 2029. Maximum sustainable yield (MSY) was estimated at 1,890 tonnes, almost the same as in 2019.
4. The recommendation for the toothfish longline fishery is to maintain the total allowable catch (TAC) at 1,040 tonnes, same as the previous year. The recommendation is based on the estimated SSB_{2019}/SSB_0 ratio being within the harvest control rules *target range* (0.40 - 0.45), and not projected to fall below the *trigger reference point* (0.40).

1. Introduction

Patagonian toothfish (*Dissostichus eleginoides*) is a large notothenioid fish found on the southern 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 abundance peak in May, and a major peak in 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, and when juveniles attain 10-12 cm TL (<1 year old; Lee 2017), they start to migrate towards the Patagonian shelf and are found at depths <100 m (Arkhipkin & 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 (Laptikhovsky et al. 2008).

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 around the fish that has been caught, 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 fishery in the Falkland Islands.

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 (20-30 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 year 2019.

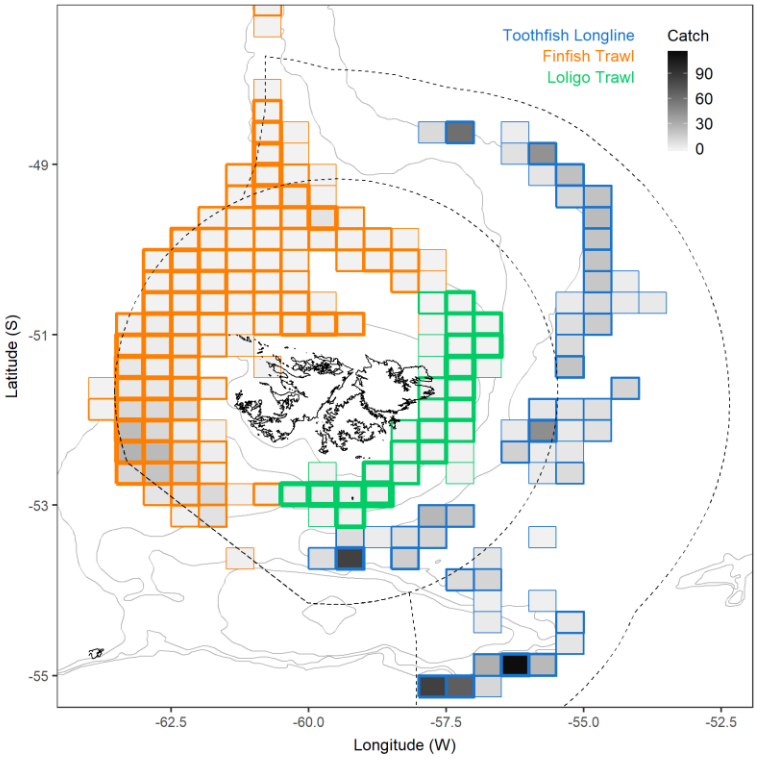

Figure 1. Spatial distribution of toothfish catch and effort by fisheries in 2019. Thickness of grid lines is proportional to the number of vessel days; greyscale is proportional to the toothfish catch biomass (tonnes).

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).

In order 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, otolith microchemistry analysis and analysis of toothfish migrations using conventional and satellite tags (Farrugia 2018). Shape analysis revealed significant differences in sagittal otoliths mean shape between different regions on the Patagonian Shelf, but further evidence is required to identify to what extent this reflects localised stock delineation (Lee et al. 2018). Otoliths microchemistry analysis suggested that larvae settling on the Falkland Shelf originate from two spatially distinct areas, presumably eastern Burdwood Banks in Falkland Islands waters, and south of Diego Ramirez Islands in Chilean waters (Ashford et al. 2012, Randhawa et al. 2020 *in review*). The existence of separate spawning populations off southern Chile and south of the Falkland Islands on the Burdwood Bank has already been proposed by several authors (Laptikhovsky et al. 2006, Arana 2009, Ashford et al. 2012). Finally, the tagging work done in Falkland Islands waters showed a very high level of 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.

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. However, the uncertainty of this assumption has to be acknowledged, and should be periodically reviewed to reflect the best available information.

2. Methods

The assessment of *D. eleginoides* was done in CASAL (Bull et al. 2012, Dichmont et al. 2016), a generalised fish stock assessment software capable of integrating a variety of different types of input data in parameter estimation. The assessment was based on the four-fishery model (Spanish-system longline, umbrella-system longline, finfish trawl and calamari trawl). Information from these fisheries cover varying time periods and/or areas, and give us an insight into the variety of issues that need to be addressed in toothfish stock assessment.

2.1. Model changes

The current assessment incorporates new data collected in 2019, including (a) catch and effort data for the umbrella-system longline fishery, (b) catch data for the finfish and calamari trawl fisheries, (c) age 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. These are listed here for reference, and explained in more detail further in the text:

Catch-per-unit-effort (CPUE) data for the umbrella-system longline fishery

- Only CPUE data pertaining to Falkland Islands flagged vessels were used in the analysis;
- Tagging trips, and longline sets at depths <600 m, were removed from the analysis;
- Hooks-per-umbrella data were corrected (as they had been assigned an erroneous sequence), and were subsequently found to be a significant explanatory variable in the generalised linear model (GLM) for CPUE standardization;
- Longline soak time was standardized per line length, prior to being considered as explanatory variable in the GLM for CPUE standardization.

CPUE data for the Spanish-system longline fishery

- Outliers were thoroughly inspected (number of hooks, soak time, CPUE) and when necessary removed from the analysis;
- Longline soak time was standardized per hook, prior to being considered as explanatory variable in the GLM for CPUE standardization.

Removals (IUU fishing)

- IUU catches were corrected, as some of the values used in the previous assessment erroneously included reported catches as IUU (Agnew 2000).

Age data (combined fisheries)

- Only age readings from otoliths collected and aged in 2015-2019 were used to construct the age-length key.

Selectivity model for the calamari trawl fishery

- The double-normal selectivity ogive was replaced with the CASAL *allvalues* ogive, where a single selectivity parameter is estimated for each age class.

New model output included in the report

- CPUE model fit residual plot (per fishery);
- Catch-at-age model fit residual bubble plot (per fishery);
- Selectivity-at-age plot (per fishery);
- Year class strength (YCS) time series plot;
- Observed vs. model fitted mean catch-at-age plot (per fishery);
- Sensitivity analyses plots.

2.2. Data

Several datasets were used as information in the assessment, either as *observations* or *input parameters* (Table 1). Observations are data which 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, all pertaining to the commercial fisheries. On the other hand, input parameters are 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. Datasets used for the stock assessment

Data type	Data	Time series
Observations	**CPUE**	
	Spanish-system longline	1996-2007, 2013
	umbrella-system longline	2007-2019
	Catch-at-age	
	Spanish-system longline	1992, 1994-2007, 2013
	umbrella-system longline	2007-2019
	finfish trawl	1988-1989, 1991-1994, 1997-1999, 2002-2019
	calamari trawl	1989-1995, 1998-1999, 2002-2019
Input parameters	**Removals**	
	Spanish-system longline	1992-2007, 2013
	umbrella-system longline	2007-2019
	finfish trawl	1987-2019
	calamari trawl	1989-2019
	Length-weight	
	all fisheries combined	1989-2019
	Length-at-age	
	all fisheries combined	2015-2019
	Maturity-at-age	
	all fisheries combined	1988-2019

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 gears /techniques (Brown et al. 2010).

For the umbrella-system longline, additional data selection had to be performed in order to avoid introducing bias in the CPUE estimates. The most substantial decision, compared to the previous year's assessment, was to use only the CPUE data from Falkland Islands flagged vessels. The reason is that the fishing was predominantly done by a single Falkland Islands vessel since the onset of the umbrella-system (*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 not regular commercial fishing (most likely they were experimental fishing, with the aim of collecting brood stock for the toothfish rearing facility).

The CPUE data selected for inclusion in the analysis were prepared for modelling in three steps. First, unstandardized CPUE values were calculated for each longline set as the reported toothfish catch in kg per 1,000 hooks. Second, these were multiplied by the whale depredation rates. Estimation of whale depredation is described in more detail in the *Removals* section of the report, but essentially, toothfish catch depredated from the longline before being hauled on board is also accounted for when calculating CPUE. Since this 'true' catch equals reported + depredated catch, resulting CPUE values will on average be higher than the ones calculated solely from the reported catch. Third, CPUE was standardised using a generalised linear model (GLM), providing a time series of CPUE values which were assumed to be 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 GLM. 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 $cv = 0.2$ was added. The CPUE indices were assumed to be log-normally distributed about the model-predicted vulnerable biomass, via a catchability parameter.

<u>*Catch-at-age distribution*</u>

The catch-at-age distribution was treated separately for each of the four fisheries. The longline catch-at-age data had to be treated separately to match the longline CPUE data (this is a model requirement), while the trawl data were split between finfish and calamari fisheries due to their differences in legal net mesh size and fishing grounds, leading to a distinct catch-at-age distributions.

Toothfish ageing data used in the stock assessment was restricted to the otoliths sampled in 2015-2019. All the otoliths from this period 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). The age estimates of 0 years were excluded from the analysis as they were too few (only 56 overall and none in the last two years) to be reliably distributed, and the remaining 3,952 toothfish age estimates were used to construct a single age-length key in R package FSA (Ogle et al. 2019). Next, 166,537 toothfish length measurements (sampled randomly by the observers from commercial catches in 1988-2019) were split between the four corresponding fisheries, and age was assigned to each fish by conditional probability of the age-length key. Ages ≥31 years were assigned to a plus group. Finally, catch-at-age 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 error misclassification matrix from a normal distribution with $cv = 0.1$. The catch-at-age data were assumed independently multinomially distributed about the model-predicted catch-at-age.

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 catch-at-age 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 catch-at-age 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, and (c) catches lost to undetected whale depredation.

Catch reports from all available years for the four fisheries 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. Trawl catch reports under experimental license were considered calamari trawls if >50% of the catch was calamari, or if the report was within 7 days of a report by the same vessel that did catch >50% calamari. Otherwise, experimental-license trawls 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 from *Table 2* in Agnew et al. (2009), which give estimates of IUU fishing by region as a percentage of reported catch in 1980-2003, and data from *Table 8* in CCAMLR (2010), which give estimates of IUU fishing in CAMLR Convention Area as a percentage of reported catch in 1988-2009. Since these data don't cover the whole assessed period, some extrapolations were made regarding the level of IUU fishing in later years, mostly based on the assumption of IUU decline post-2000. In our assessment we used the data for the *Antarctic region* from Agnew et al. (2009), as it pertains specifically to toothfish, and was used in the previous year's model. It is important to note that previous assessments also used data reported by Agnew (2000) to describe IUU fishing in Falkland Islands, but closer inspection revealed that the assumed IUU catches were actually the reported Falkland Islands catches. These data were excluded from the current assessment, leading to a different assumed IUU fishing for the period 1992-1997. As a part of model sensitivity analyses, two alternative levels of IUU fishing were tested; one was predominantly based on the data for the South-west Atlantic from Agnew et al. (2009) and the other on the data for CAMLR Convention Area from CCAMLR (2010). The SW Atlantic data were supplemented with CCAMLR data for 2004-2009, and from 2010 onwards IUU was set to a constant value in both datasets, calculated as the average of the last three years (2007-2009) from the CCAMLR data. The complete list of IUU values used in the main and alternative analyses is given in Appendix 2 (Table A.2).

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 accounted for in the reports. In order to address this, Winter and Pompert (2016) modelled whale depredation in Falkland Islands waters by comparing the toothfish

CPUE with and without whales present on observed longline sets, using a generalised additive model (GAM). This allowed us to extrapolate the toothfish catch lost to whale depredation for all commercial longline sets, based on the fishing month, longline position, fishing depth, number of hooks set and soak-time. As the GAM is probabilistic, some longline sets obtained model-fit depredation rates <1, implying less toothfish catch in the absence of whale depredation. That outcome is obviously artefactual, and to make the estimates more precautionary, depredation rates for individual longline sets were therefore adjusted upwards by dividing them with the 5th percentile of their own distribution; a value of approximately 0.87 for the Spanish- and 0.96 for the umbrella-system longline.

In order to combine the above-mentioned catch components into total removals by fishery, first the IUU catches were added to the reported catches in each year, and then the undetected whale depredation rate was applied. Effectively, this assumes that reported and IUU catches experience the same average rate of whale depredation. Total removals were used in the assessment model run, and since removals are treated as input parameters and not observations in CASAL, they were assumed known without error. The removals partitioned into reported, IUU and depredated catch are summarized in Appendix 2 (Table A.3).

Length-weight relationship

The length-weight relationship was calculated as $W = aL^b$, based on the length and weight measurements of 33,205 toothfish sampled randomly by the observers from commercial catches in 1989-2019. 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 Appendix 2 (Table A.4).

Length-at-age relationship

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 3,952 toothfish sampled randomly by observers from commercial catches in 2015-2019. Parameters L_{inf}, k and t_0 are summarized in Appendix 2 (Table A.4).

Maturity-at-age vector

Maturity-at-age vector was based on the maturity stage data estimated by the observers for 152,755 toothfish, sampled randomly from commercial catches in 1988-2019. 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 (2019). Finally, instead of the more typical logistic function, 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 Appendix 2 (Table A.5, Figure A.4).

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, four-fishery 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 described as a Beverton-Holt relationship, with a *steepness* parameter set to the commonly used reference value h = 0.75 (Brandão and Butterworth 2009, Dunn and Hanchet 2010, Mormede et al. 2011, 2013, 2014). Steepness is 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).

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 2019. Projections from the model extended for another 35 years, up to 2054. Conditions in the initial year were assumed to be equilibrium age structure at an unexploited equilibrium biomass.

Estimation method

Model parameters were estimated by minimising an objective function, which is the sum of the negative log-likelihoods from the observations, negative-log Bayesian priors, and penalties that constrain the parameterisations. The estimated parameter values presented in the report are so-called MPD (maximum posterior density) point estimates (Bull et al. 2012).

To estimate the joint posterior distribution of the parameters in a Bayesian analysis, 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 100th value was taken. The resulting 10,000 values represent a systematic sample from the Bayesian posterior distribution for the parameter of interest. In the report these samples were shown either in the form of histogram with superimposed MPD estimate or summarised in the form of 2.5th and 97.5th percentiles.

Estimated parameters

The parameters estimated by the model, their priors, starting values and bounds are given in Table 2, and detailed further in the text.

Table 2. Number (N), priors, start values and bounds for the 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*		32	lognormal	1	0.001	20
M		1	uniform	0.13	0.05	0.75
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
q_{LLH}		1	uniform-log	-	1×10^{-9}	0.1
q_{LLU}		1	uniform-log	-	1×10^{-9}	0.1

LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, *YCS consists of 32 parameters (for years 1986-2018) and values in the table apply to each of them, **Selectivity$_{LOL}$ consists of 8 parameters (for fish ages 1-8) and values in the table apply to each of them

SSB_0 is the estimated virgin 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. 2011, 2013, 2014). Year class strength multipliers (YCS) were estimated for the period 1986-2018 (32 parameters, one for each year), using the *Haist parameterisation* to make the YCS parameters average to 1 over the period 1986-2014 (for the Haist method description see Bull et al. 2012). For YCS, informative (lognormal) prior with μ = 1 and *cv* = 1.1 was used (Constable et al. 2006a, 2006b). Natural mortality (M) was assumed to be constant across all age classes, and the start value of 0.13 year^{-1} was set (Dunn and Hanchet 2010, Mormede et al. 2011, 2013, 2014). 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. Three types of selectivity ogives were used: logistic for longline fisheries, double-normal for finfish trawl fishery and CASAL *allvalues* for calamari trawl fishery. 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

$$\begin{aligned} f(x) &= 2^{-[(x-a_1)/S_L]^2}, \quad (x \le a_1) \\ &= 2^{-[(x-a_1)/S_R]^2}, \quad (x > a_1). \end{aligned}$$

The *allvalues* ogive is defined by one selectivity parameter for each age class, meaning that for our catch-at-age data we would have 31 parameters. Since negligible number of toothfish older than 8 years was 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 empirical *allvalues* ogive was used for calamari trawl fishery because standard selectivity curves, such as the double-normal used previously (Farrugia and Winter 2019) could not fit well the catch-at-age patterns observed in the data, with the highest proportions in the catch corresponding to the lowest ages (descending ogive).

Selectivities were assumed to remain constant throughout the modelled period. For all selectivity parameters uninformative priors were used (Dunn and Hanchet 2010, Mormede et al. 2011, 2013, 2014). It is important to note that what we term 'selectivity' is actually 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 constrain the ogive to a 4th degree polynomial. Penalty multipliers were set to 100 for catch limits and 20 for YCS vector average and ogive smoothing (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 2015-2054 was assumed to be log-normally distributed with *sd* = 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%).

2.4. Sensitivity analyses

Several sensitivity analyses were carried out in order to better understand the potential implications of modelling decisions on the results of the stock assessment. Using the 2019 model presented in this report as the base-case, we explored the effects of:

a) *Alternative values of fixed input parameters*. We tested four alternative values of Beverton-Holt *steepness* (0.65, 0.70, 0.80 and 0.85).
b) *Alternative model structure.* We tested the effects of changing the selectivity ogive type for calamari trawl fishery, modifying the assumptions about YCS, and assuming different levels of IUU fishing.
c) *Alternative relative weighting of the observations*. We compared the relative information coming from the CPUE and catch-at-age data, to check does it lead to the similar model estimates. This was achieved by assigning different weights to the catch-at-age data and then rerunning the model; in total 6 different scenarios were tested, with catch-at-age data being either up-weighted (by multiplying the sample sizes by a factor of 2, 5 or 10) or down-weighted (by multiplying the sample sizes by a factor of 0.5, 0.2 or 0.1).
d) *Alternative future catches.* We compared the effects of different potential future catches (taken by umbrella-system logline, finfish trawl and calamari trawl fisheries) on the projected SSB.

The results of sensitivity analyses were calculated as MPD point estimates, except for the alternative future catch projections, which were calculated as medians from 5000 MCMC runs.

3. Results

Model fits
Diagnostics plots of the model fits to the different observation datasets are provided in Appendix 3. The model fit to the standardized CPUE data for the umbrella-system longline was moderately good, with estimated values falling within observation 95% CI in all analysed years. However, fit to the older Spanish-system data was rather poor, with the model underestimating CPUE for the first five years of the fishery and overestimating it for the remaining seven years (Figure A.5). Corresponding trends in residuals for both longline fisheries are shown in Figure A.6.

The model fit to the catch proportion-at-age data was generally good for all four fisheries, except for calamari trawl fishery in the earlier years, i.e. prior to 2007 (Figures A.7, A.8, A.9 and A.10). The corresponding residual bubble plots show no clear pattern for longline and finfish trawl fisheries, but for calamari fishery in earlier years, model tends to overestimate the proportion of 1 year old fish and underestimate the proportions of 2-4 year old fish (Figure A.11). The comparison of observed vs. fitted mean age of caught toothfish confirmed the same trend, with good fit in all cases except for calamari trawl fishery prior to 2008 (Figure A.12). The overall poor model fit to the pre-

2007/08 calamari trawl fishery could be due to different gear selectivity, or different fish availability (more specifically – different availability of certain toothfish age classes in the fishing area) in this period. However, another possibility is that the bias was introduced in the data due to the different sampling protocols in early and late period, i.e. different level of attention given to accounting for the juvenile toothfish (which can be difficult to distinguish from certain other species in the juvenile stage). In general, much fewer toothfish were sampled during the earlier years of calamari fishery, with more fish sampled in 2008 alone then in the previous 15 years combined. At this point it is hard to discern which explanation is more plausible, the different fishery selectivity or different sampling protocol, but further exploration should be undertaken in time for the next assessment.

Model estimates

The key output parameters estimated by the stock assessment model are summarised in Table 3, and detailed further in the text.

Table 3. Key output parameters estimated by the model.

Parameter	MPD value	MCMC 95% CI
SSB_0	24,199 t	21,316 - 92,249 t
SSB_{2019}	10,637 t	7,877 - 78,504 t
SSB_{2019}/SSB_0	0.440	0.364 - 0.875
MSY	1,890 t	1,665 - 7,205 t
M	0.186 y^{-1}	0.171 - 0.241 y^{-1}

The initial spawning stock biomass (SSB_0) and the current spawning stock biomass (SSB_{2019}) estimates were somewhat higher than in the previous year's assessment ($SSB_{0\ \text{in}\ 2018}$ = 22,669 t; SSB_{2018} = 10,596). However, the ratio of current spawning stock biomass to initial spawning stock biomass (SSB_{2019}/SSB_0) estimate was lower than the last year's (SSB_{2018}/SSB_0 = 0.467). This decline was anticipated by the previous assessment, and although minor, it places the SSB_{2019}/SSB_0 ratio below the 0.45 threshold, i.e. under a different *harvest control rule* (HCR). According to HCR, the stock was in the *expansion range* in 2018, and currently it is estimated to be in the *target 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.

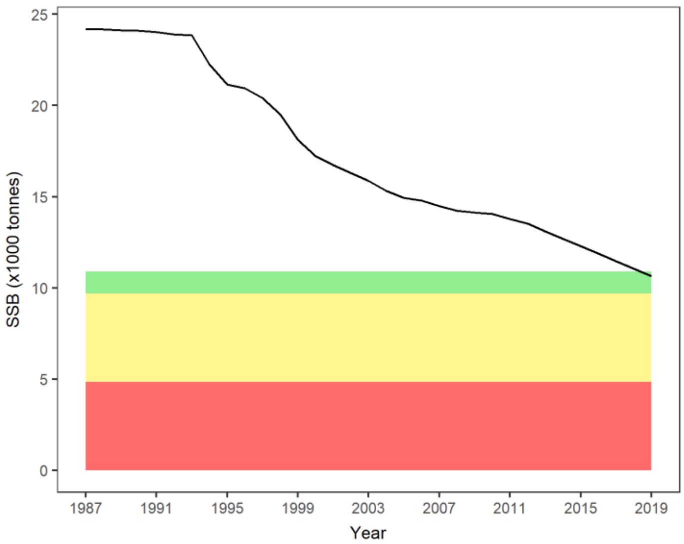

Figure 2. MPD estimates of historical spawning stock biomass (SSB). Harvest control rule (HCR) ranges are colour coded for reference: *target range* in green (SSB/SSB_0 = 0.45-0.40), *trigger range* in yellow (SSB/SSB_0 = 0.40-0.20) and *closure range* in red (SSB/SSB_0 < 0.20).

Maximum sustainable yield (MSY), estimated under the assumption of a constant future catch partition, was almost the same as in 2018 (MSY_{2018} = 1,899 t). Deducting from the MSY 300 t for finfish trawl and 30 t for calamari trawl fishery leaves 1,560 t, well above the current longline toothfish TAC (1,040 t).

MCMC posterior distributions of estimated parameters displayed positive skewness with narrow lower bounds and wide upper bounds (Figure 3). The three closely related parameters of SSB_0, SSB_{2019}, and MSY were not strongly constrained by the model to an upper limit, but at the same time are naturally lower-bounded at zero as biomass cannot be negative, resulting in asymmetrical 95% confidence intervals.

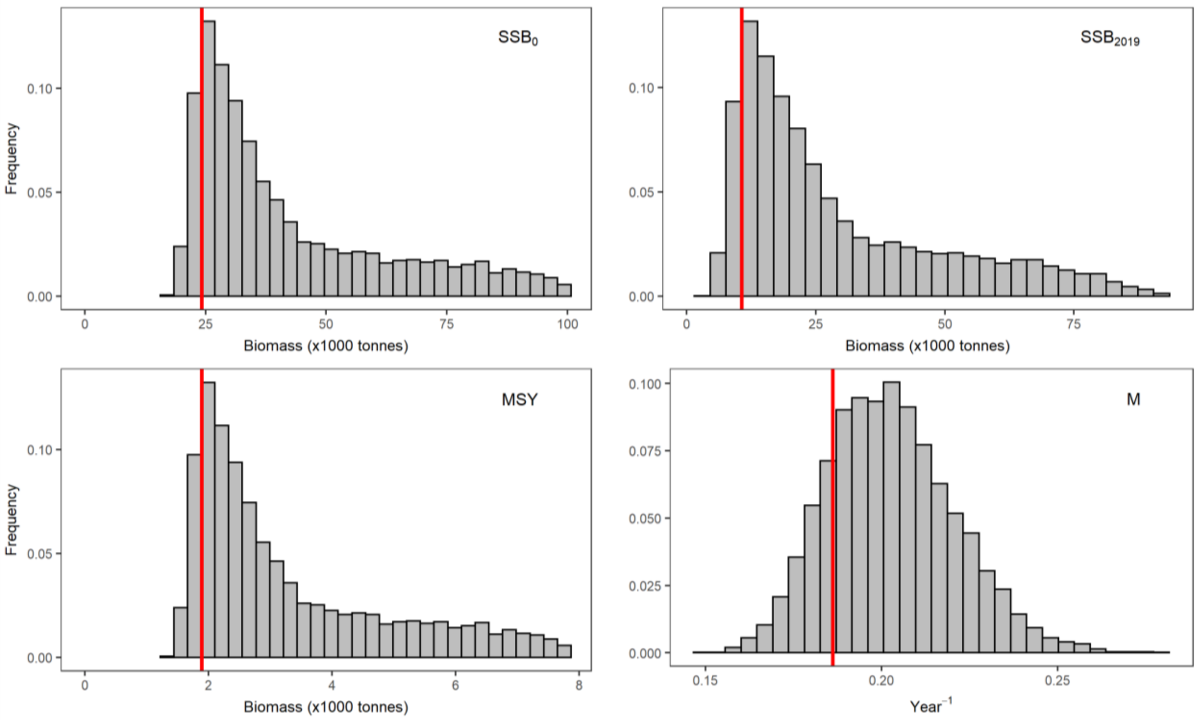

Figure 3. MCMC samples from the posterior distribution of the key estimated parameters, with MPD point estimates added as a reference (red lines).

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.

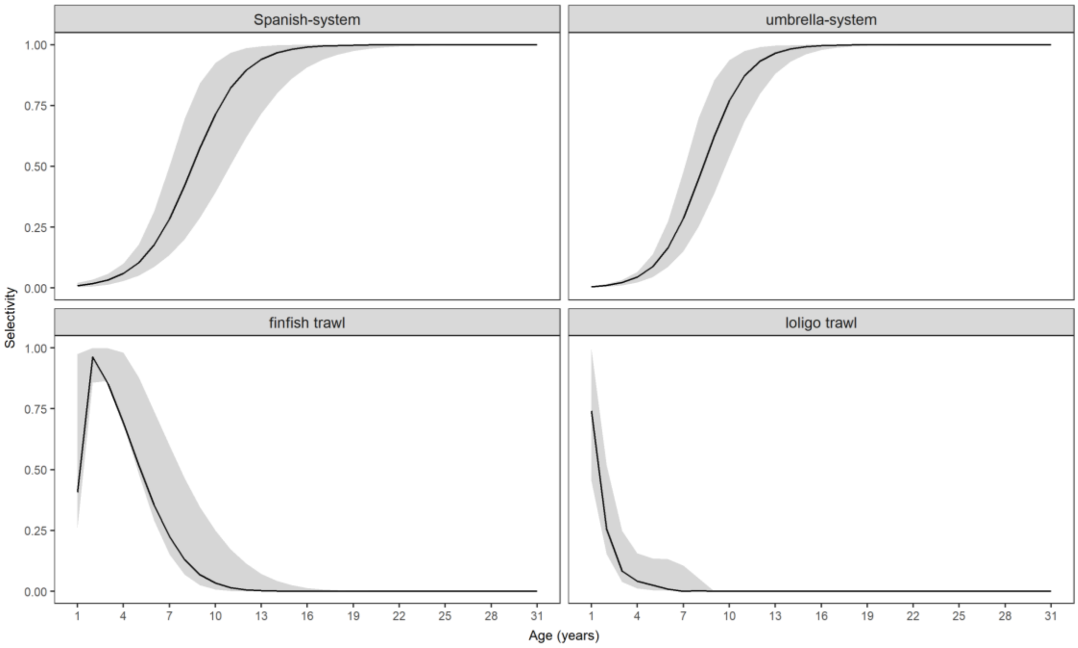

Figure 4. MPD estimates of selectivity ogives for four fisheries (line), with 95% confidence intervals obtained from the MCMC sample (shaded area).

The estimated year class strength (YCS) showed high variability, with an overall downward trend in the assessed period (Figure 5). However, it is questionable whether this information comes from the observations (CPUE and catch-at-age data), or indicates an over-parameterised model, with the YCS deviations essentially fitting data that possess no information on recruitment (Candy and Constable 2008). This was further explored in the sensitivity analyses and elaborated in the discussion section.

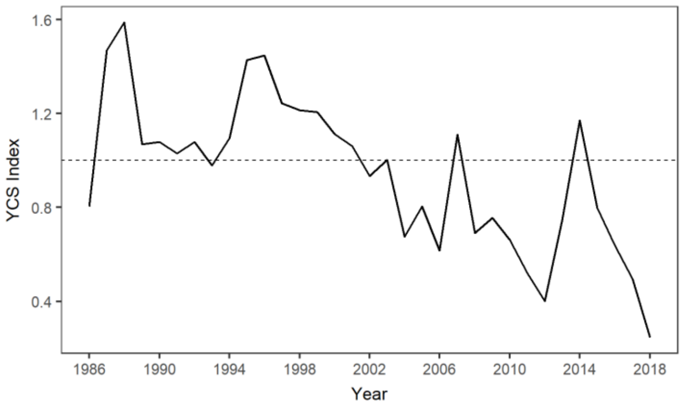

Figure 5. MPD estimates of year-class strength (YCS).

Model projections

The future trend of SSB/SSB_0 was projected based on 5000 MCMC runs, with random lognormal recruitment from 2015-2054 and constant annual catches from 2020-2054 (umbrella-system longline 1,040 t, finfish trawl 300 t, calamari trawl 30 t) (Figure 6). The median SSB/SSB_0 ratio is currently within the HCR *target range*, but is on a declining trend, expected to reach its minimum point of 0.424 (95% CI: 0.372 - 0.522) in 2023. However, this minimum point is still within the target range, and the median SSB/SSB_0 ratio is expected to slowly start increasing afterwards, reaching 0.45 by 2029.

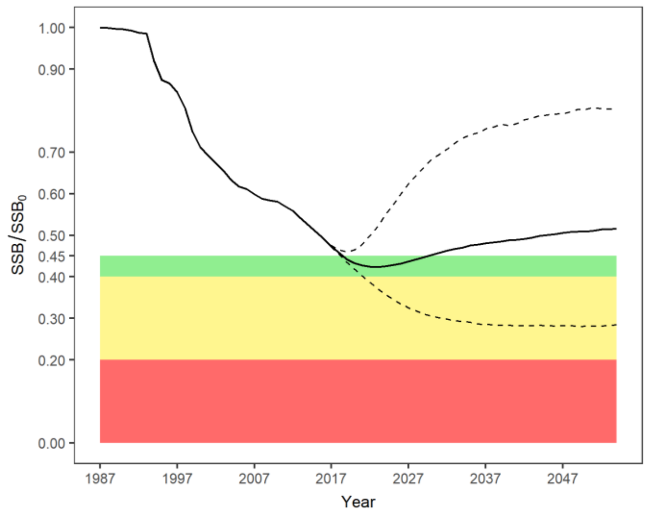

Figure 6. Projected SSB/SSB$_0$ trend based on 5000 MCMC runs, assuming random lognormal recruitment from 2015-2054 and constant annual catches from 2020-2054. Solid line is the median and broken lines are 95% confidence intervals of SSB/SSB$_0$. Harvest control rule (HCR) ranges are colour coded for 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).

The probability of SSB/SSB$_0$ falling below management thresholds, corresponding to the upper bounds of HCR ranges, is shown in Figure 7. SSB/SSB$_0$ is below 0.45 management threshold at the moment and probability is high that it will remain that way in the near future, although with a clear declining trend. Probability of SSB/SSB$_0$ falling below the 0.40 threshold during the projection period is dome-shaped, with a peak in 2025-2027 before declining (Figure 7). This probability is highest for 2026 (29.4%), and by 2054 it declines to 17.8%. Finally, probability of SSB/SSB$_0$ falling below 0.20 management threshold is extremely low, <0.5% during the whole projected period (2020-2054). The seemingly contradictory result that the probability of stock being below 0.45 is decreasing at the same time that the probability of falling below 0.40 is steeply increasing (2021-2024), has to do with the SSB/SSB$_0$ projection confidence intervals - they are asymmetrical around the median trend, and their spread is widening as the projections move further ahead (Figure 6). This is especially pronounced during the first projected years, and leads to a peculiar result where the median SSB/SSB$_0$ is increasing, but its CI intervals are getting wider, so the estimated probability of it falling below 0.40% is increasing as well.

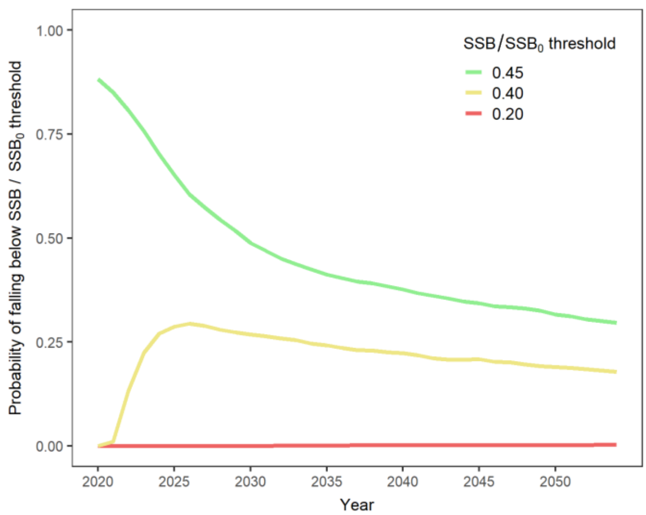

Figure 7. Probability of stock falling below SSB/SSB$_0$ management thresholds, based on 5000 MCMC projections.

Sensitivity analyses
The results of different sensitivity trials are summarised in Table 4, with the current model included as a base-case for the reference. Only the key estimated parameters are given here (SSB_0, SSB_{2019} and SSB_0/SSB_{2019}), as they form the basis of harvest control rules and have a direct effect on the management decisions.

Table 4. MPD estimates of key assessment parameters (SSB_0, SSB_{2019} and SSB_{2019}/SSB_0) obtained from different sensitivity trials. All biomass estimates are given in tonnes.

Model run	SSB_0	SSB_{2019}	SSB_{2019}/SSB_0
Base-case	24,199	10,637	0.440
Alternative fixed input parameters			
h = 0.65	24,448	10,590	0.433
h = 0.70	24,211	10,567	0.436
h = 0.80	24,094	10,671	0.443
h = 0.85	23,978	10,643	0.444
Alternative model structure			
Selectivity $_{LOL}$ = double-normal	23,928	10,370	0.433
YCS = 1 (constant)	25,934	14,679	0.566
YCS = lognormal prior (cv = 0.6)	25,957	13,082	0.504
YCS = uniform prior	23,538	9,623	0.409
YCS = Haist (1986-2018)	24,451	11,395	0.466
IUU = SW Atlantic	25,502	10,685	0.419
IUU = CCAMLR Area	25,500	10,527	0.413
Alternative weighting of catch-at-age data			
Catch-at-age w = 0.1	22,417	8,148	0.363
Catch-at-age w = 0.2	22,335	8,585	0.384
Catch-at-age w = 0.5	23,049	9,571	0.415
Catch-at-age w = 2	25,585	11,807	0.461
Catch-at-age w = 5	27,040	12,797	0.473
Catch-at-age w = 10	27,223	12,653	0.465

Changing the Beverton-Holt steepness parameter (h) had only a small effect on the estimated parameters; when h increased, the estimated SSB_{2019} and SSB_{2019}/SSB_0 increased as well, but only slightly. This is not surprising, since the lowest estimated SSB_{2019}/SSB_0 ratio was 0.43, and at this level the Beverton-Holt relationship has relatively little effect on the annual recruitment estimates. Therefore, the sensitivity of the model to changes in steepness is low, at least at the current stock level.

Changing the selectivity ogive used to describe the calamari trawl fishery had a negligible effect on the model results, with double-normal ogive resulting in only slightly less optimistic estimates than the *allvalues* ogive used in the base-case assessment (Figure 8).

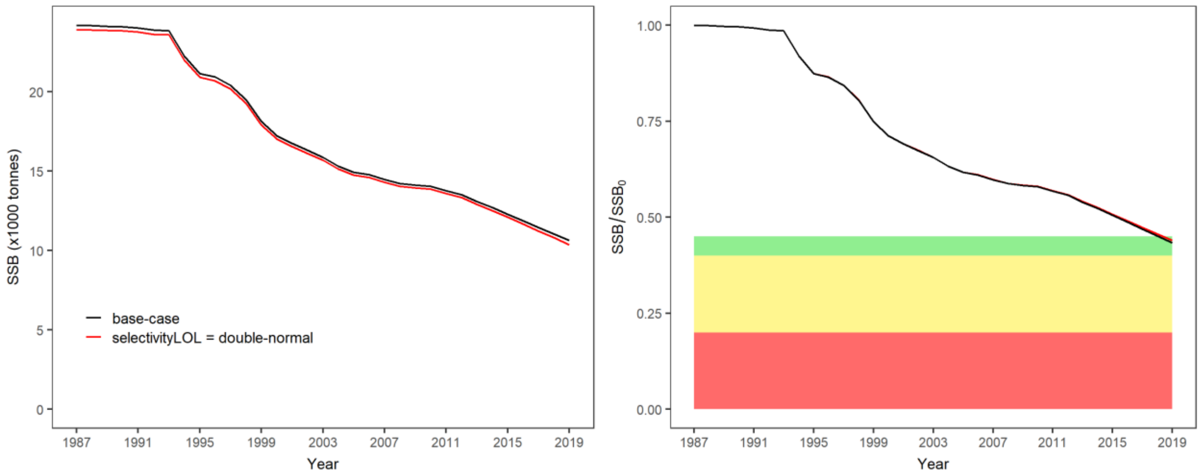

Figure 8. Estimated SSB and SSB/SSB_0 trends for the base-case model (black line) and the alternative model in which the selectivity of calamari trawl fishery was described by double-normal ogive

As mentioned previously, the YCS estimates can be considered questionable because it is unclear whether they represent information available in the data, or are simply adapted by the model to fit data which have no information on recruitment (Candy and Constable 2008). Since independent data on YCS are rarely available (e.g. from pre-recruitment scientific surveys) and the use of model estimates cannot be avoided, different authors approach this problem by making specific assumptions and treating the YCS data in different ways (Candy and Constable 2008, Hillary et al 2006, Mormede et al 2014). Here we are not trying to evaluate the merits of different approaches, but merely to explore the extent of their influence on the model results. Four alternative models were run, each with a specific assumption regarding the YCS, borrowed from available reports and papers on toothfish stock assessment (Table 5). The results showed that different assumptions about YCS can have significant impact on the outcome of the assessment (Figure 9). Setting the YCS to 1 (i.e. assuming that recruitment was constant throughout the assessed period) resulted in the most extreme model outcome and a unique historical SSB trend, but the remaining models were more similar. However, even without the YCS = 1 scenario, the remaining models would lead to different HCR-s, meaning they would suggest different management actions. On the negative side, this is a cause for concern, as varying assumptions about YCS can lead to different conclusions regarding stock status. On the positive side, our base-case model produced a precautionary result compared to the alternatives, except for the model using a uniform YCS prior.

Table 5. Description of the alternative models' assumptions regarding the YCS. Changes from the base-model are highlighted in bold.

Model run	YCS description
base-case	lognormal prior, μ=1, cv=1.1, Haist parameterisation over 1986-2014
YCS = lognormal prior (cv = 0.6)	lognormal prior, μ=1, **cv=0.6**, Haist parameterisation over 1986-2014
YCS = Haist (1986-2018)	lognormal prior, μ=1, cv=1.1, Haist parameterisation over **1986-2018**
YCS = uniform prior	**uniform prior**, Haist parameterisation over 1986-2014
YCS = 1 (constant)	**YCS is effectively ignored**, i.e. the recruitment doesn't vary over years

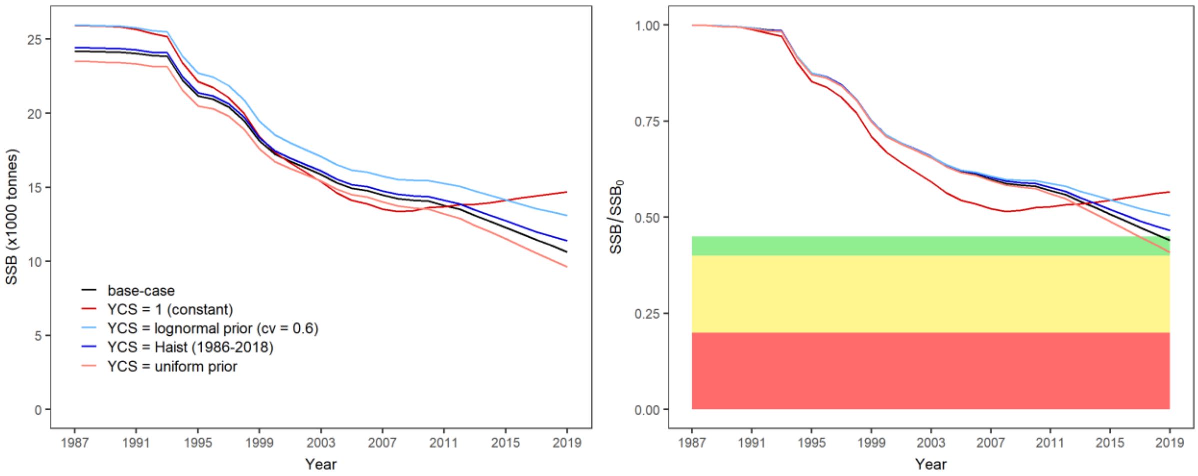

Figure 9. Estimated SSB and SSB/SSB$_0$ trends for the base-case model (black line) and four alternative models with different YCS assumptions. It is important to note that in the *YCS = 1* scenario we used a fixed rate of YCS, whereas in the others YCS was fitted by the model based on the given priors.

IUU toothfish catches were introduced in the model as a part of the total removals. Since no reliable data on the IUU fishing were found for the Falkland Islands waters, estimates belonging to other regions or periods were used, and should be regarded with caution. In order to test the effect of IUU catch estimates on the stock assessment outcome, base-case model was compared to two alternative models with higher assumed levels of IUU fishing (detailed description of these datasets is available in the methods section). The results showed that the higher assumed IUU catches lead to less optimistic assessment outcomes, although the impact was comparatively low-to-moderate, especially having in mind large difference in IUU level between the base-case and alternative models (Figure 10). It is not clear what can be done regarding this issue, as it is difficult to favour one scenario over the others. Perhaps the best course of action would be a more extensive review of the available literature, possibly resulting in a composite IUU estimate to be used in the future assessments.

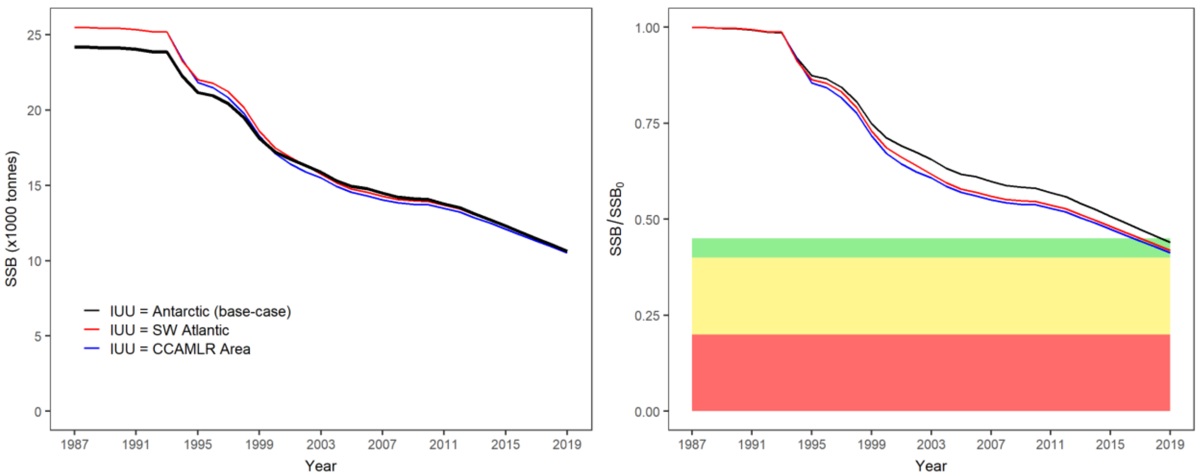

Figure 10. Estimated SSB and SSB/SSB$_0$ trends for the base-case model (black line) and two alternative models with different assumed levels of IUU fishing.

In the integrated stock assessment, models are fitted to multiple datasets that may contradict each other, and potentially lead to different outcomes. The main datasets in our assessment are CPUE and catch-at-age observations, and the sensitivity analyses were performed in order to detect is there a tension between the two. The procedure was to assign different weights to catch-at-age data, either up-weighting or down-weighting it relative to the CPUE data, run the model, and compare the results. If both datasets contain the same information about the stock (i.e. they are not biased in respect to each other), changing their relative weights shouldn't affect the model outcome. Six alternative models were run, three of them down-weighting the catch-at-age data by multiplying the sample sizes in turn by 0.1, 0.2 and 0.5, and three up-weighting it by multiplying the sample sizes in turn by 2, 5 and 10. The re-weighting had a major impact on the outcome of the assessments, confirming that conflict exists between the CPUE and catch-at-age data (Figure 11). Up-weighting the catch-at-age data resulted in the higher estimates of both SSB and SSB/SSB_0, and down-weighting it had the opposite effect, resulting in more pessimistic outcomes. This can be interpreted as catch-at-age data pulling the estimates upwards, and CPUE data pulling them downwards. Our base-case model is a compromise between the two, and although this is a reasonable solution, further effort should be made to remove or reduce the data conflict. This will most likely require extensive analytical work, as many alternative model structures and parameterisations will have to be tested, together with thorough data inspection, especially regarding CPUE and its standardization.

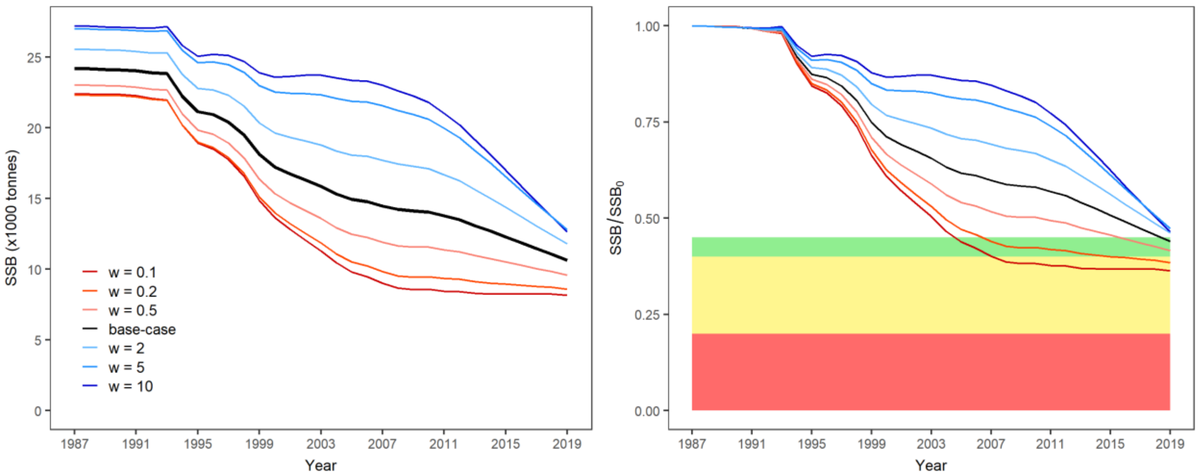

Figure 11. Estimated SSB and SSB/SSB_0 trends for the base-case model (black line) and six alternative models with different weights assigned to catch-at-age data.

The final set of the sensitivity analyses is a special case, as it deals only with the model projections, from 2020 onwards. In the base-case model, the projections of SSB and SSB/SSB_0 were estimated assuming the constant future toothfish catches of 1040 / 300 / 30 tonnes in longline / finfish trawl / calamari trawl fisheries respectively. Here we explored the effect of change in the catch levels on the future stock status. For finfish trawl fishery the future toothfish catches ranging from 100 to 500 t were tested (Figure 12a), and for calamari trawl fishery from 10 to 50 t (Figure 12b), as this roughly corresponds to reported historical catches. Comparison of the two revealed that, although the calamari fishery catches younger /smaller specimens, high overall catches taken in the finfish fishery could have much larger impact on the future stock status. Therefore, close monitoring of the toothfish catches in finfish fishery should be continued, and the existing '1.5% toothfish bycatch limit' should be kept. Increase of toothfish catches in finfish trawl fishery above ~470 t reverses the expected positive future trend of SSB/SSB_0. For the toothfish longline fishery, we tested the effect of increasing the removals from the fishery (note that this can be a combination of commercial catch and different levels of whale depredation), ranging from 800 to 1500 t (Figure 12c). As expected, increasing the future catches resulted in less optimistic projections for all alternative models, and vice versa. However, it is interesting to notice that 200 t increase in longline catches had almost the same effect on the future stock status as did 100 t increase in finfish trawl catches. This is a reasonable result, as toothfish caught in trawl fishery are smaller, and catching 100 t of small specimens would remove much higher number of fish from the stock than the longline catch of comparable size.

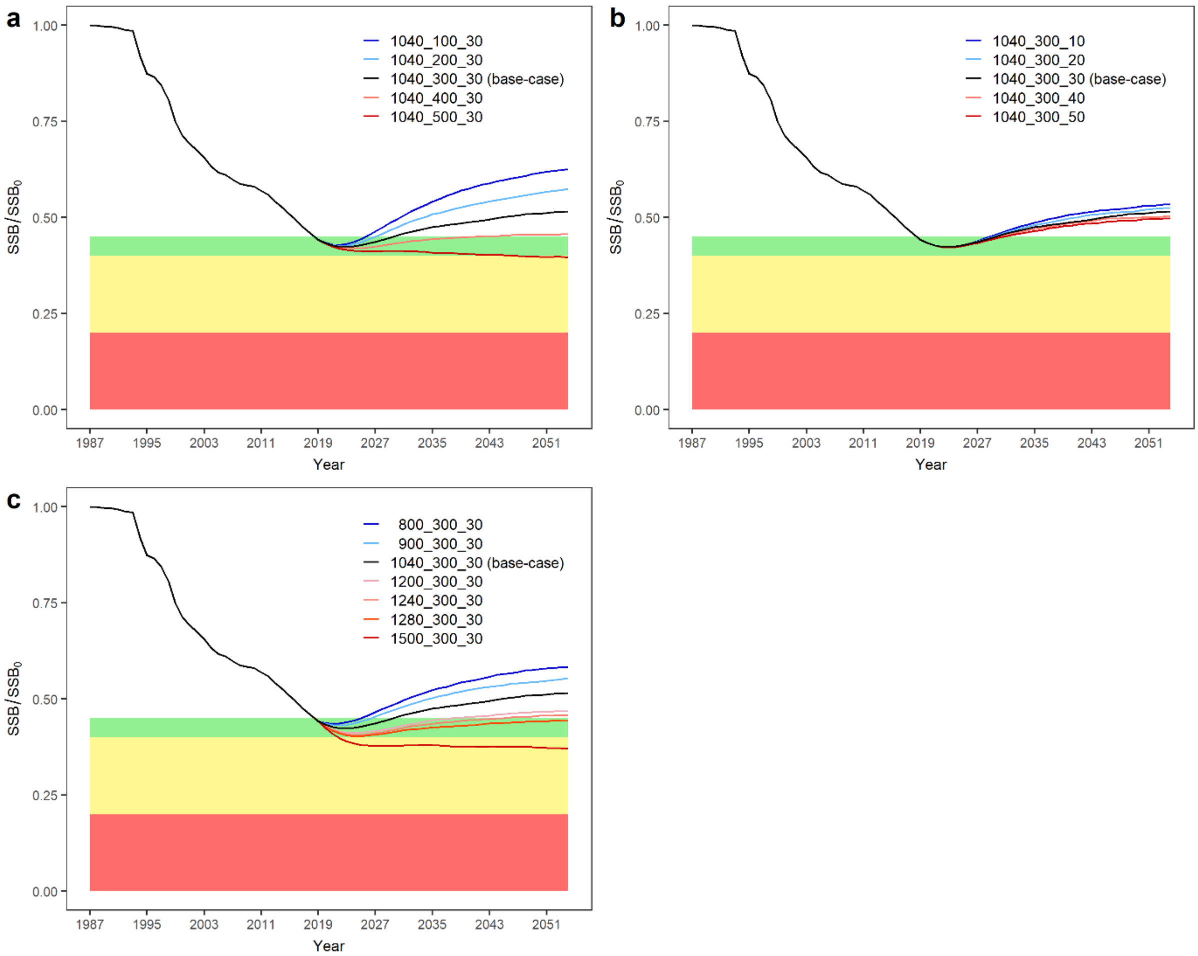

Figure 12. Estimated SSB and SSB/SSB_0 trends for the base-case model (black line) and: (a) four alternative models with different finfish trawl catches, (b) four alternative models with different calamari trawl catches and (c) six alternative models with different longline catches. Future toothfish catch split between fisheries is shown as longline / finfish trawl / calamari trawl catch in tonnes.

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 the observers during commercial trips. Compared to the 2018 assessment, this assessment incorporates (a) new observations and ageing data for 2019, (b) revised longline CPUE time series (including both raw data and GLM standardized indices), (c) change in the selectivity curve used for the calamari trawl fishery, and (d) change in the input ageing dataset. This updated assessment resulted in a slightly lower estimate of SSB_{2019}/SSB_0 than that obtained last year, but the decline was minor, as anticipated by the projections from the previous assessment. According to the projections from the current model, this declining trend will reach its low in 2023, but the stock is still expected to remain within the HCR target range.

Model fits to the data were adequate, except for fits to the Spanish-system longline CPUE data, and calamari trawl fishery catch-at-age data in early years (pre-2007). CPUE data are of particular importance in the assessment because there are no fisheries independent surveys, and these are the only data which provide an index of stock abundance. The poor fit to the Spanish-longline CPUE data could be due to the current assessment model parameterization (e.g. conflict between different datasets), but it is also possible that the standardization of CPUE time series for the early years of fishery needs to be adjusted (e.g. accounting for some vessels using the *Mustad Autoline* system instead of Spanish-system longline, including additional explanatory variables in the GLM, etc.). Regarding calamari trawl fishery catch-at-age data, two distinct trends were noticed: prior to 2007 catch-at-age distributions were mostly dome-shaped, dominated by 2- or 3-year old toothfish, and from 2007 onwards they were right-hand descending, dominated by 1-year old specimens. Since both catch-at-age trends belong to the same fishery with a single selectivity ogive, the model couldn't produce a good fit to both, resulting in a poor fit to pre-2007 data. The common solution to this issue is splitting the dataset into two 'sub-fisheries', with individual selectivity ogives to achieve a better model fit (Ziegler and Welsford 2015). Based on visual inspection of our current model fits, promising approach for the future assessment might be to split calamari fishery into pre-2007 fishery described by the double-normal selectivity ogive, and post-2007) fishery described by the *allvalues* ogive.

In order to test the robustness of the assessment model, a number of alternative modelling assumptions were investigated. Changing the Beverton-Holt steepness parameter, selectivity ogive for calamari fishery, or level of IUU fishing had relatively low impact on the assessment outcome. However, different assumptions regarding YCS had more substantial effects, and this issue will require further analytical work. The difficulty of estimating YCS in toothfish stock assessment was reported by several authors (Candy and Constable 2008, Hillary et al 2006, Mormede et al 2014), concluding that over-parameterisation of the model is commonly observed along with seemingly spurious YCS estimates. The YCS issue was also brought up in the external review of Falkland Islands toothfish assessment and identified as a challenging problem that would mostly likely take a year or two to resolve (Bergh 2018). One option that might be worth exploring is inclusion of the additional fishery-independent data on toothfish recruitment into the model (i.e. from groundfish and calamari pre-season surveys), although the merits of this approach are inconclusive (Candy and Constable 2008, Mormede et al 2014). Spatio-temporal distribution and abundance of juvenile toothfish in Falkland Islands waters are currently being analysed at FIFD (based on the survey data from 2009-2019), and could potentially prove useful in informing the stock assessment about recruitment and YCS.

The sensitivity analyses giving different relative statistical weights to CPUE and catch-at-age data revealed that there is a 'tension' in the model, i.e. these two datasets (or specific components thereof) are giving conflicting information about the stock status. This will need to be explored in more detail, perhaps by omitting the observations from the model one by one (there are 4 catch-at-age and 2 CPUE components) and analysing how this affects the assessment outcome. Once the understanding of the relative merits of different dataset components is improved, alternative model structures that would reduce the tensions between the data (or ideally remove them altogether) should be explored.

5. Management advice

Current management advice is based on a set of harvest control rules (HCR) established to manage the Falkland Islands toothfish longline fishery (Farrugia and Winter 2018, 2019) (Appendix 4). Estimated SSB_{2019}/SSB_0 ratio of 0.440 is below *upper target reference point* (0.45) and above *trigger reference point* (0.40), i.e. in the *target range*, as defined by HCR. Although the model projects a further decrease in SSB_{2019}/SSB_0 ratio in the next few years, it is nevertheless expected to stay above 0.40, in the target range. While the stock is at this level, the total allowed catch (TAC) should not be increased, but further conservation measures are not required either.

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 insights from the current assessment, as well as recommendations from the external review of Falkland Islands toothfish stock assessment (Bergh 2018), several points for future consideration and model refinement were identified:

CPUE standardization

- Explore the option of explanatory variable interactions in GLM;
- Explore the option of increasing the resolution of spatial data (e. g. using grid squares in addition to /instead of regions) to better capture the spatial heterogeneity of the four fisheries;
- Explore the option of binning continuous variables (depth, soak time), to decouple these variables from distributional assumptions;
- Explore the option of modelling CPUE using GLMM and/or GAM.

Observations

- Explore the tension that exists between different observations, namely between CPUE and catch-at-age datasets. This can be achieved by systematically omitting some of the data components and rerunning the analysis, or by trying different combinations of weighting the datasets (two CPUE components and four catch-at-age components).

Selectivity

- Explore the option of modelling the calamari trawl fishery as two separate fisheries, pre-2007 and post-2007, with different selectivities.

Recruitment

- Explore whether the data from the groundfish and calamari pre-season surveys could be helpful in estimating toothfish YCS.

Maturity

- Review the maturity-at-age estimates and underlying data; if needed, collect new samples and attempt to model maturity ogive (logistic and/or GAM) based on these data alone.

7. References

Agnew DJ. 2000. The illegal and unregulated fishery for toothfish in the Southern Ocean, and the CCAMLR catch documentation scheme. Marine Policy 24, 361-374.

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.

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.

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 life style. 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.

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.

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.

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.

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.

Dichmont CM, Deng RA, Punt AE, Brodziak J, Chang Y-J, Cope JM, Ianelli JN, Legault CM, Methot Jr. RD, Porch CE, Prager MH, Shertzer KW. 2016. A review of stock assessment packages in the United States. Fisheries Research 183, 447-460.

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.

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.

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.

Farrugia TJ. 2018. Stock discrimination research overview for Patagonian toothfish in the Falkland Islands. Fisheries Department, Directorate of Natural Resources, Falkland Islands Government, 13 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.

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

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.

Laptikhovsky VV, Arkhipkin AI, Brickle P. 2008. Life history, fishery and stock conservation of the Patagonian toothfish, around the Falkland Islands. American Fisheries Society Symposium 49, 1357-1363.

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, 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. 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.

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.

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

Maunder MN, Starr PJ. 2003. Fitting fisheries models to standardised CPUE abundance indices. Fisheries Research 63, 43-50.

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.

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.

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.

Ogle DH, Wheeler P, Dinno A. 2019. FSA: Fisheries Stock Analysis. R package version 0.8.26, https://github.com/droglenc/FSA.

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

Randhawa HS, Lee B, Brickle P, Reid MR, Arkhipkin AI. 2020. Oceanographic cues determine the recruitment demography of toothfish *Dissostichus eleginoides* on the Patagonian Shelf: evidence from otoliths microchemistry. *In review*.

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.

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.

Ziegler P, Welsford D. 2015. An integrated stock assessment for the Heard Island and the McDonald Islands Patagonian toothfish (*Dissostichus eleginoides*) fishery in Division 58.5.2. Document WG-FSA-15/52. CCAMLR.

Appendix 1. CPUE standardization

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Spanish- and umbrella-system longline CPUE was standardized using generalized linear model (GLM), with a log link function and normally distributed error (Maunder and Starr 2003, Maunder and Punt 2004). Individual longline haul CPUE values (expressed as toothfish catch in kg per 1000 hooks) were the response variable, and the explanatory variables considered in the model are given in table A.1.

Table A.1. Explanatory variables considered in the CPUE standardization GLM, 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
-	Umbrella-spacing	Categorical

* Variables which were found statistically significant and included in the final model.

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 at which longline is set (in meters). *Soak-time* was calculated in hours-per-hook for Spanish-system longline, and hours-per-line for the umbrella-system. *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 two additional variables: *Umbrella-spacing* (which was changed from 40 m between umbrellas to 22 m between umbrellas after November 2014) and 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).

Year effect is the quantity of interest so it must be a part of the final CPUE model, and the remaining explanatory variables were added to the *Year* by forward stepwise selection, and included in the final model only if they improved R^2 by at least 0.5%.

Fitting GLM to the Spanish-system data showed that the explanatory variables *Year*, *Month*, *Region*, *Soak-time* and *Vessel* are statistically significant, although the model explained only 17.1% of the overall variation in CPUE. Standardized and unstandardized CPUE time series showed similar trends, with high values in the first 4-5 years of fishery, followed by the lower, but relatively steady values in the later years (Figure A.1).

Fitting GLM to umbrella-system data showed that the explanatory variables *Year*, *Month*, *Region*, *Soak-time* and *Hooks-per-umbrella* are statistically significant, and the model explained 28.5% of the overall variation in CPUE. Comparison of the umbrella-system standardized and unstandardized annual CPUE indexes is shown in Figure A.2. The most prominent feature of the unstandardized data is steep increase in CPUE in 2017, followed by the decline during the next two years, but still with significantly higher values than in the earlier years of fishery. This corresponds to the entry of the new vessel into the fishery, i.e. *CFL Hunter* replaced the *CFL Gambler* from the beginning of 2017 (as mentioned before, only the data belonging to these two vessels were used in the analysis). Second trend is less obvious as it is partially masked by the mentioned 'new vessel' feature, but broadly speaking, there was an increase in unstandardized CPUE from 2014 to 2019. This can be explained by the decrease in the number of hooks-per-umbrella, introduced voluntary by the fisherman over time. Since hooks are set in clusters, reducing their number from 8 to 7 to 6 didn't affect the catches per umbrella much, but it was perceived as the reduced effort (calculated as the total number of hooks per longline set) and lead to an increase in unstandardized CPUE. However, the number of hooks-per-umbrella was significant explanatory variable in GLM, and in the standardized CPUE time series this increasing trend was removed. It is worth pointing out that the option of using the umbrellas instead of hooks as the unit of effort was explored as well, but the results were almost exactly the same as when using hooks and having hooks-per-umbrella as a significant explanatory variable in GLM.

The distribution of the residuals from the GLM fit to Spanish- and umbrella-system data was consistent with the assumption of normality (Figure A.3).

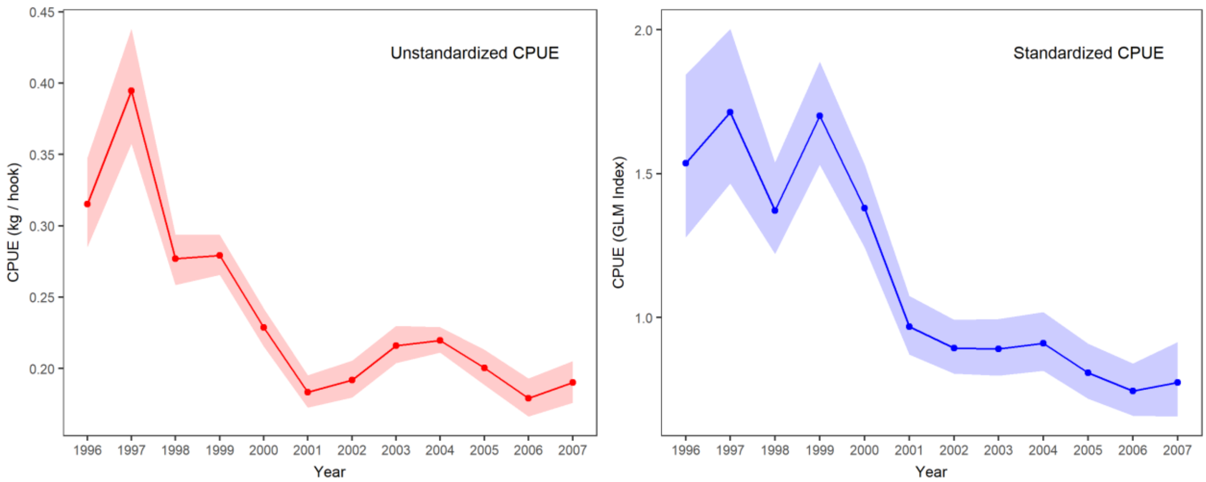

Figure A.1. Spanish-system longline CPUE time series: unstandardized CPUE expressed as toothfish catch in kg per hook (left), and standardized CPUE indices from the GLM (right); shaded areas correspond to 95% confidence intervals.

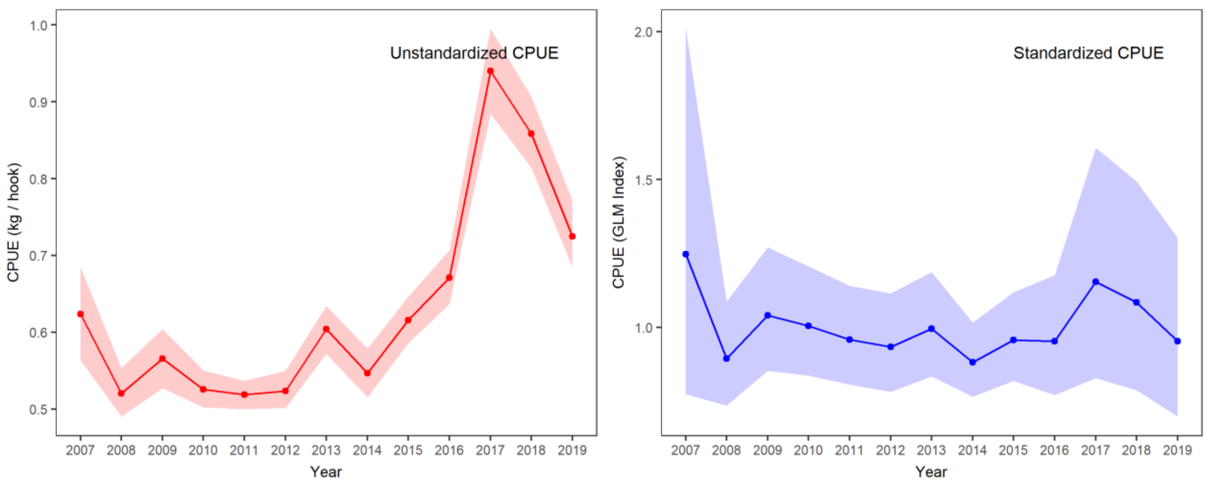

Figure A.2. Umbrella-system longline CPUE time series: unstandardized CPUE expressed as toothfish catch in kg per hook (left), and standardized CPUE indices from the GLM (right); shaded areas correspond to 95% confidence intervals.

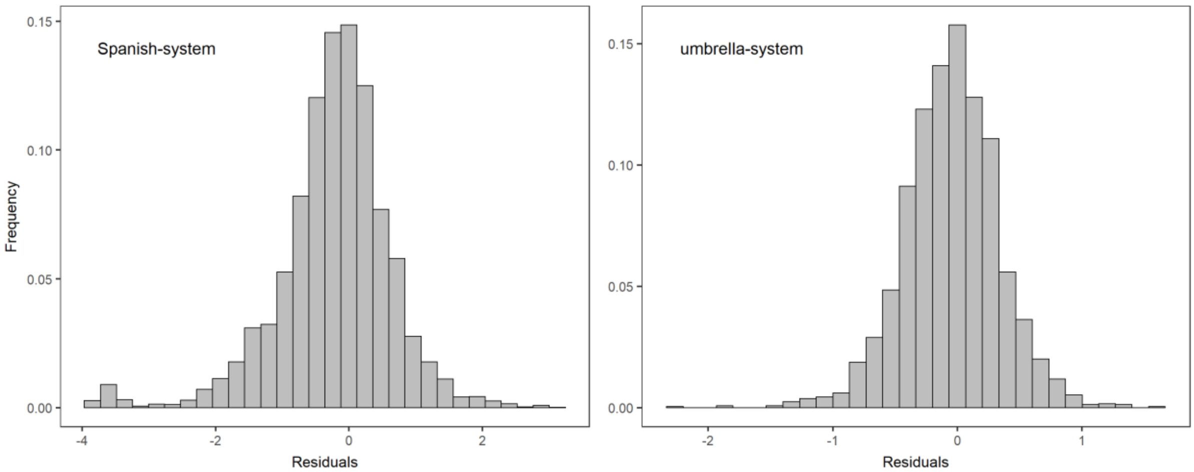

Figure A.3. Density histograms of residuals from the generalized linear model (GLM) fitted to the Spanish- and umbrella-system longline CPUE data.

Appendix 2. Input parameters

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Table A.2. Different scenarios of potential toothfish IUU catches in Falkland Islands waters. Values in bold are copied from the corresponding publications. Values in italics are extrapolations based on the overall trend of post-2000 IUU decline.

Year	IUU as a % of removals		
	Antarctic region (Agnew 2009)	SW Atlantic region (Agnew 2009)	CCAMLR Area (CCAMLR 2010)
1987	**0**	**18**	**0**
1988	**0**	**18**	**2**
1989	**0**	**18**	**5**
1990	**2**	**24**	**27**
1991	**2**	**24**	**21**
1992	**2**	**24**	**38**
1993	**2**	**24**	**44**
1994	**2**	**24**	**16**
1995	**15**	**34**	**64**
1996	**15**	**34**	**72**
1997	**15**	**34**	**54**
1998	**15**	**34**	**33**
1999	**15**	**34**	**30**
2000	**7**	**32**	**39**
2001	**7**	**32**	**44**
2002	**7**	**32**	**29**
2003	**7**	**32**	**12**
2004	*5*	*14*	**14**
2005	*5*	*17*	**17**
2006	*5*	*22*	**22**
2007	*5*	*10*	**10**
2008	*5*	*6*	**6**
2009	*5*	*12*	**12**
2010	*5*	*9*	*9*
2011	*5*	*9*	*9*
2012	*5*	*9*	*9*
2013	*5*	*9*	*9*
2014	*5*	*9*	*9*
2015	*5*	*9*	*9*
2016	*5*	*9*	*9*
2017	*5*	*9*	*9*
2018	*5*	*9*	*9*
2019	*5*	*9*	*9*

Table A.3. Reported, estimated IUU and estimated depredated longline catches. IUU catches are based on the Antarctic region data (Table A.2)

Year	Catch (tonnes)		
	Reported	Estimated IUU	Estimated depredated
1992	111.5	2.2	20.6
1993	7.7	0.2	1.7
1994	2733.2	54.7	393.4
1995	1745.5	261.8	325.5
1996	512.7	76.9	100.0
1997	998.1	149.7	208.7
1998	1700.4	255.1	291.2
1999	2405.0	360.8	422.9
2000	1976.0	138.3	301.2
2001	1444.7	101.1	250.6
2002	1472.4	103.1	211.8
2003	1517.6	106.2	315.2
2004	1807.9	90.4	347.1
2005	1614.6	80.7	298.0
2006	1303.9	65.2	227.9
2007	1550.5	77.5	267.5
2008	1469.4	73.5	201.3
2009	1159.0	58.0	110.6
2010	942.9	47.1	96.6
2011	1225.6	61.3	155.8
2012	1085.1	54.3	161.8
2013	1303.4	65.2	198.9
2014	1221.4	61.1	134.2
2015	1123.2	56.2	166.6
2016	1022.9	51.1	158.7
2017	1031.6	51.6	130.2
2018	981.7	49.1	140.3
2019	1047.6	52.4	124.9

Table A.4. Length-weight and length-at-age input parameters.

Relationship	Parameter	Value
Length-weight	a	6.150×10^{-9}
	b	3.115
Length-at-age (von Bertalanffy)	L_{inf}	171.770
	k	0.065
	t_0	-2.606
	cv	0.145

Table A.5. Maturity-at-age input parameters.

Age	Proportion mature	Age	Proportion mature	Age	Proportion mature
1	0.000	12	0.479	23	0.673
2	0.015	13	0.510	24	0.688
3	0.055	14	0.538	25	0.706
4	0.101	15	0.562	26	0.728
5	0.153	16	0.582	27	0.750
6	0.209	17	0.599	28	0.769
7	0.266	18	0.614	29	0.784
8	0.318	19	0.628	30	0.793
9	0.365	20	0.640	31	0.800
10	0.407	21	0.651		
11	0.444	22	0.661		

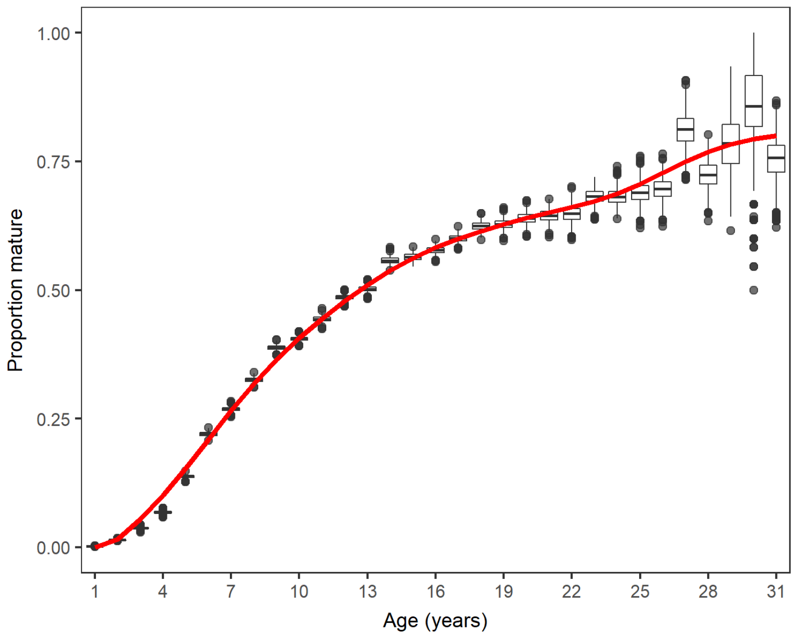

Figure A.4. Maturity-at-age ogive, fitted by GAM to the corrected maturity data.

Appendix 3. Diagnostics plots

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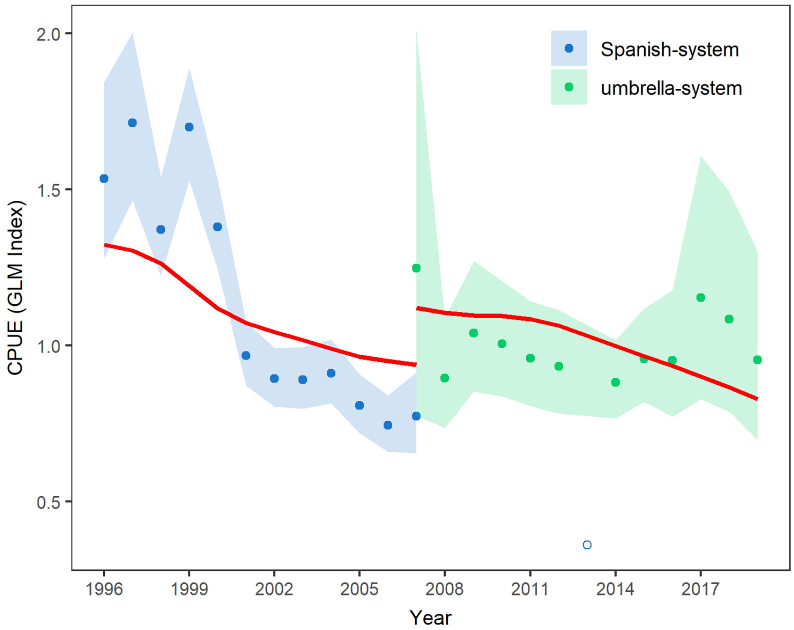

Figure A.5. Model fit (red line) to the standardised CPUE time series for Spanish-system (blue dots) and umbrella-system longline (green dots); shaded areas correspond to 95% confidence intervals of the standardized CPUE indices.

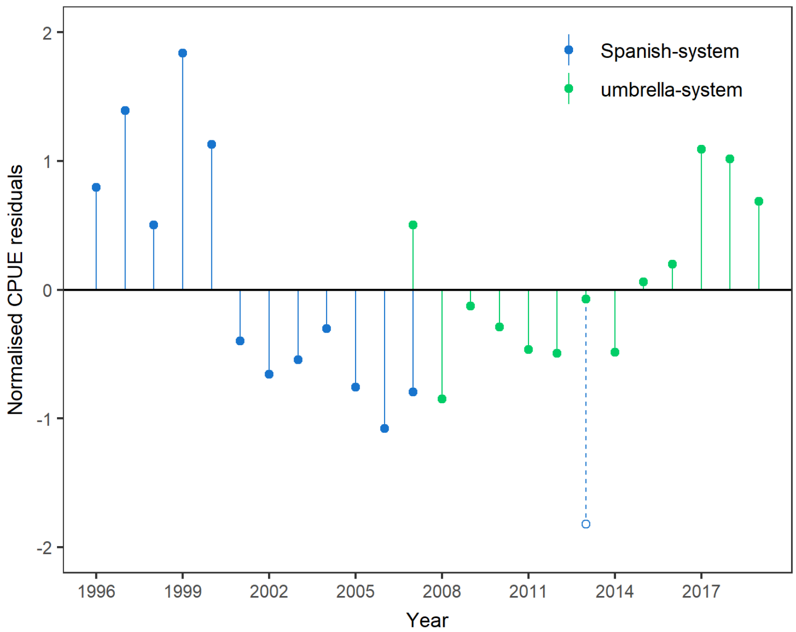

Figure A.6. Normalised residuals from the model fit to standardized CPUE time series; for Spanish-system (blue) and umbrella-system longline (green).

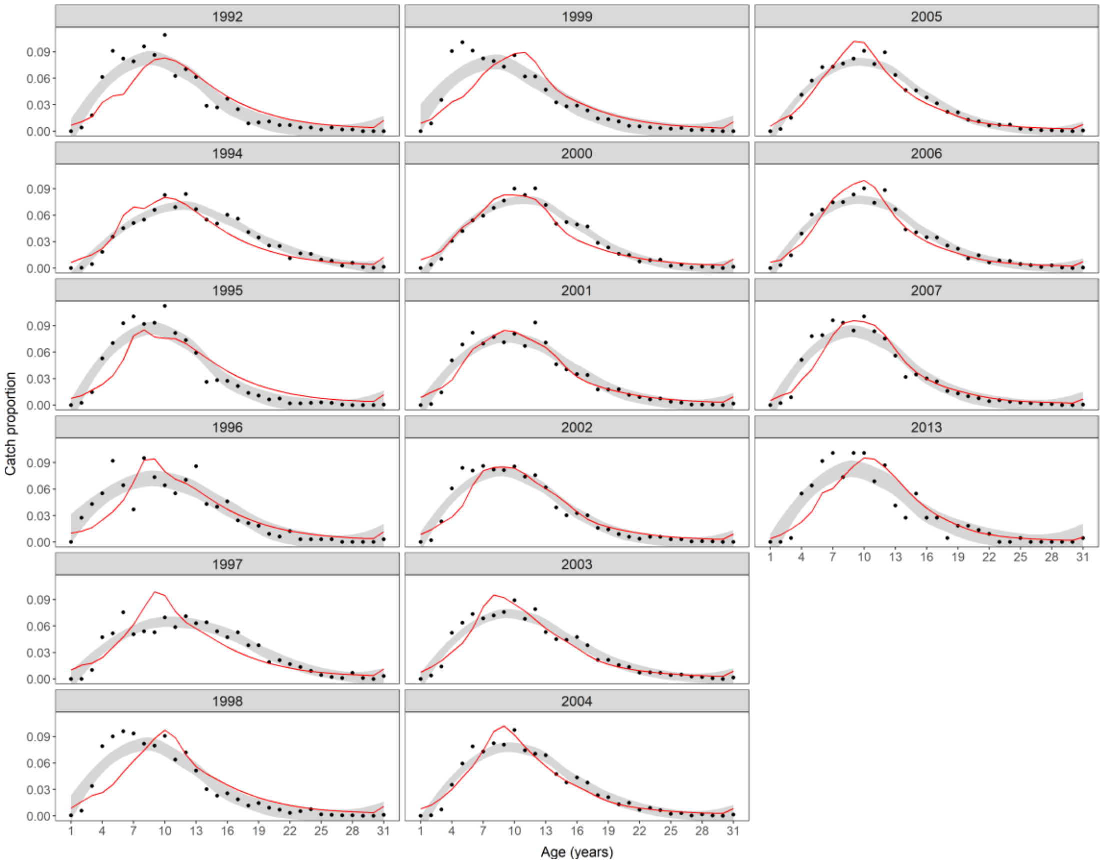

Figure A.7. Model fits (red line) to the observed toothfish catch-proportion-at-age data for the Spanish-system longline fishery (black dots); shaded areas correspond to the loess smoother 95% confidence intervals (span = 0.75).

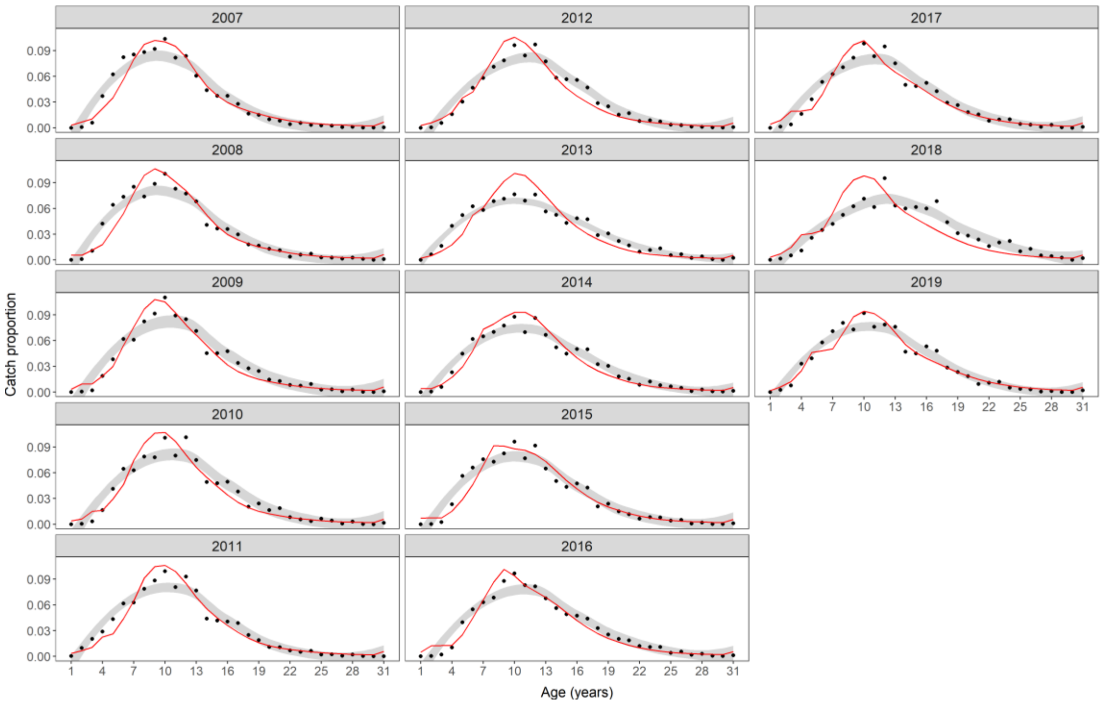

Figure A.8. Model fits (red line) to the observed toothfish catch-proportion-at-age data for umbrella-system longline fishery (black dots); shaded areas correspond to the loess smoother 95% confidence intervals (span = 0.75).

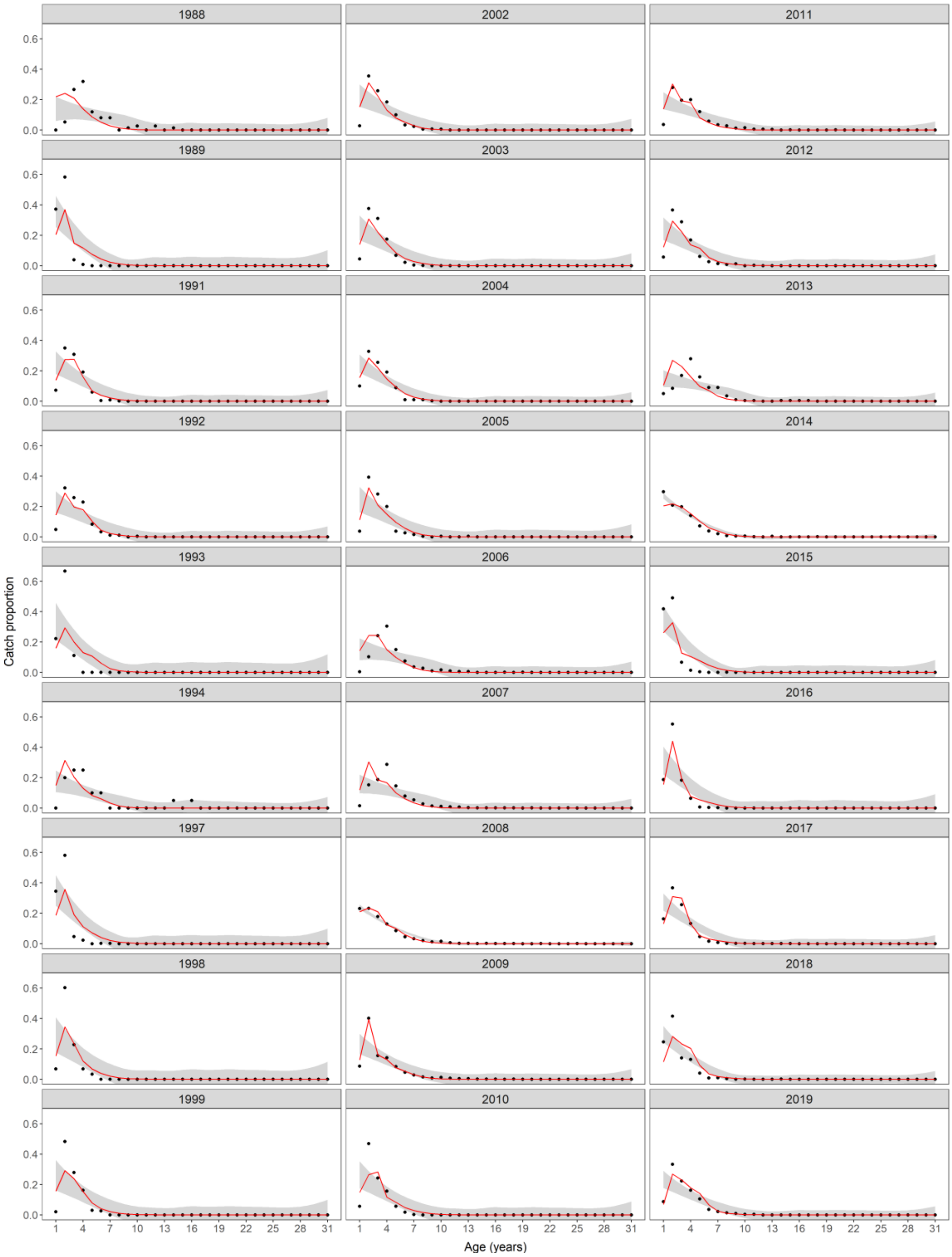

Figure A.9. Model fits (red line) to the observed toothfish catch-proportion-at-age data for finfish trawl fishery (black dots); shaded areas correspond to the loess smoother 95% confidence intervals (span = 0.75).

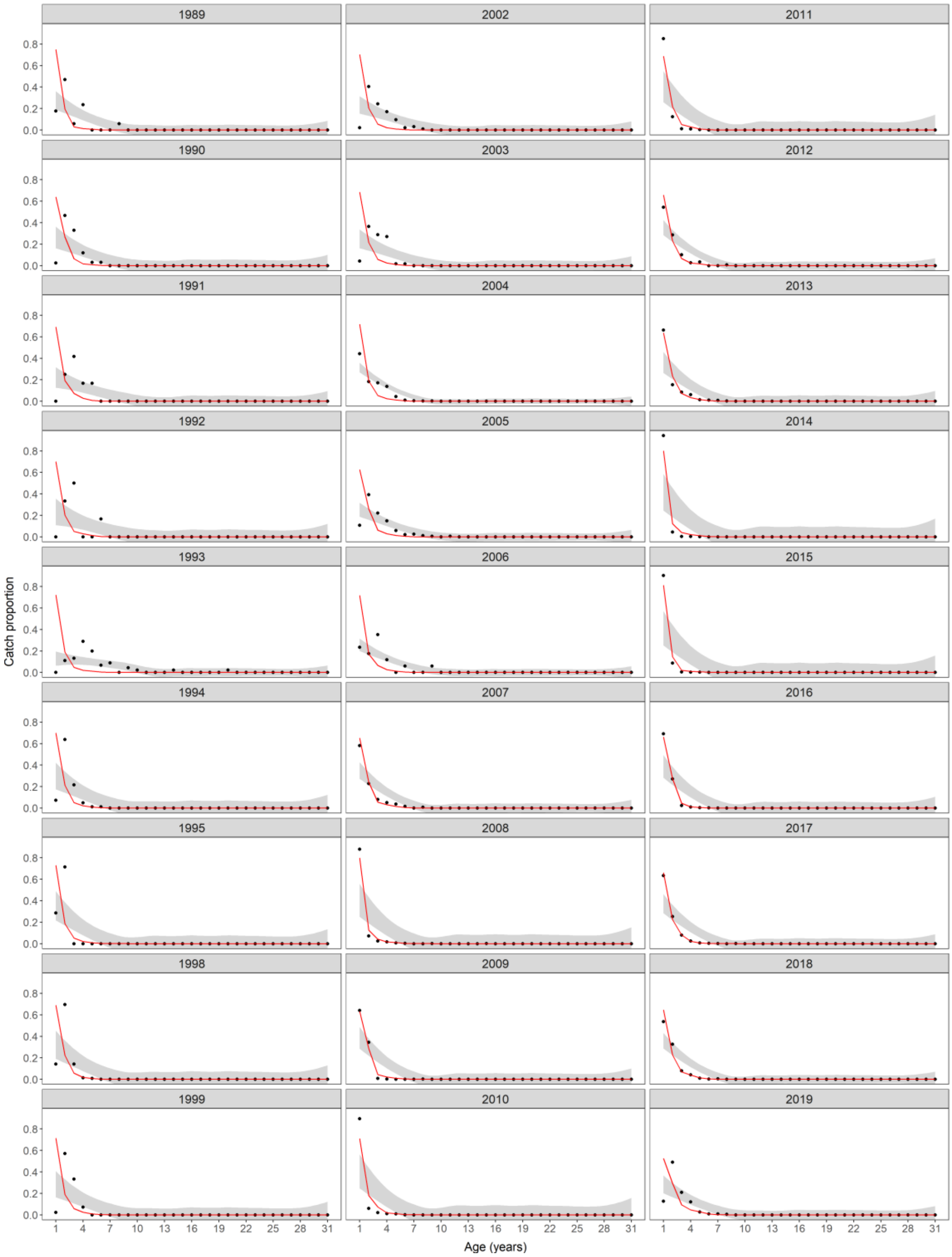

Figure A.9. Model fits (red line) to the observed toothfish catch-proportion-at-age data for calamari trawl fishery (black dots); shaded areas correspond to the loess smoother 95% confidence intervals (span = 0.75).

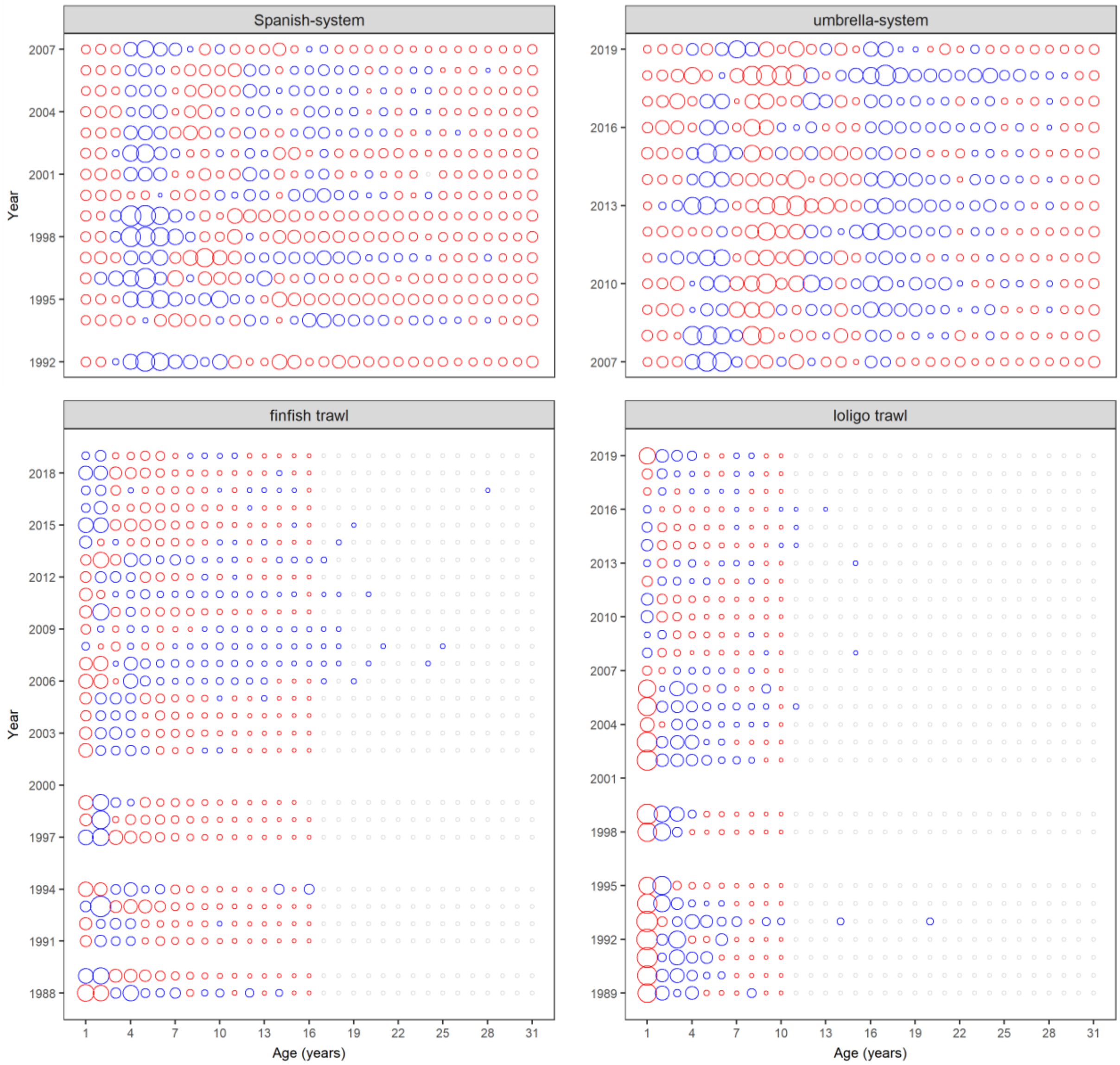

Figure A.11. Residuals from the model fit to observed catch-at-age for four fisheries. Bubble size is relative to the absolute residual value; positive residuals shown in blue, negative in red.

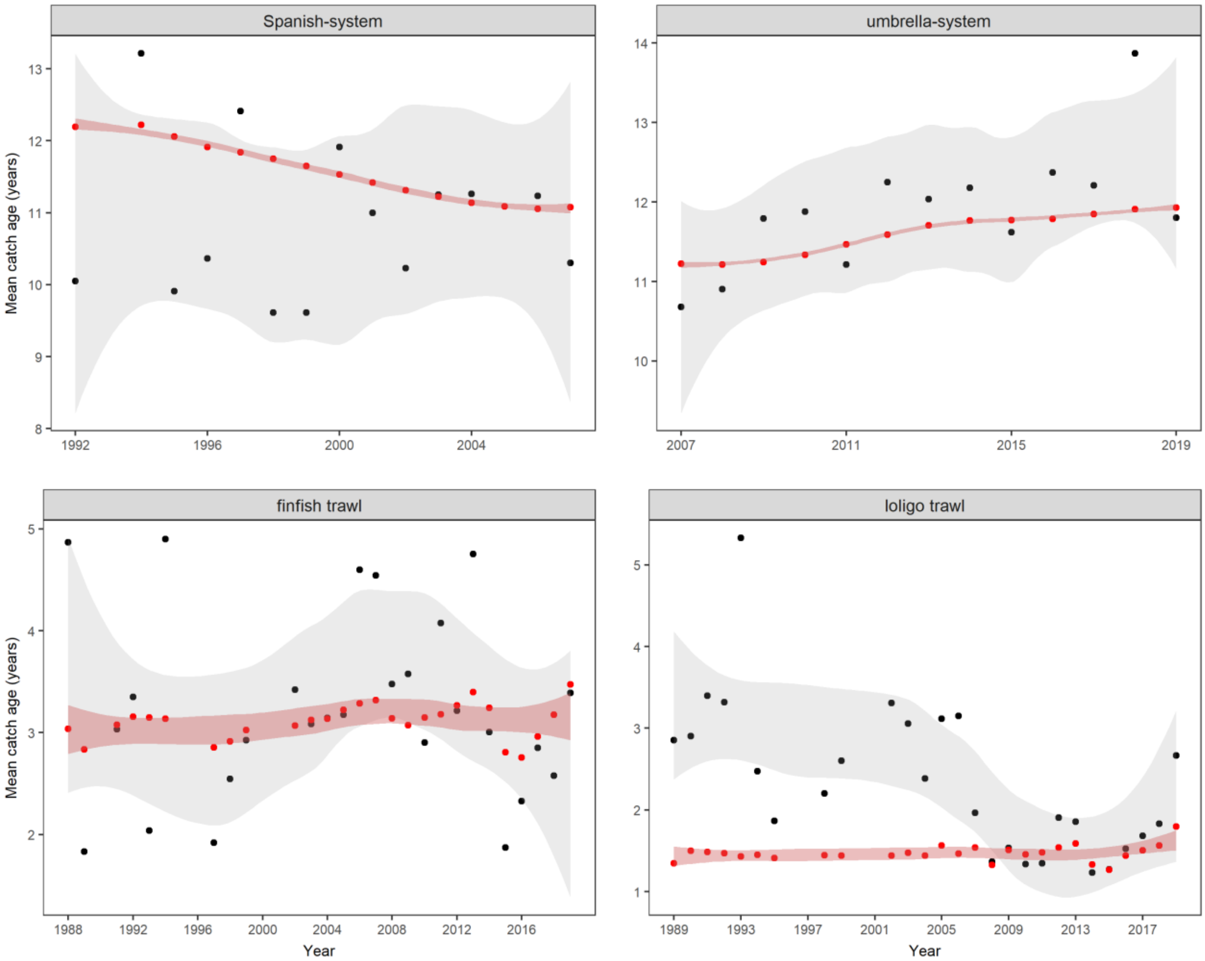

Figure A.12. Model fit (red dots) to the observed toothfish mean catch-at-age data for four fisheries (black dots); shaded areas correspond to the loess smoother 95% confidence intervals (span = 0.75), for model fits (red) and observations (grey).

Appendix 4. Harvest control rules

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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.