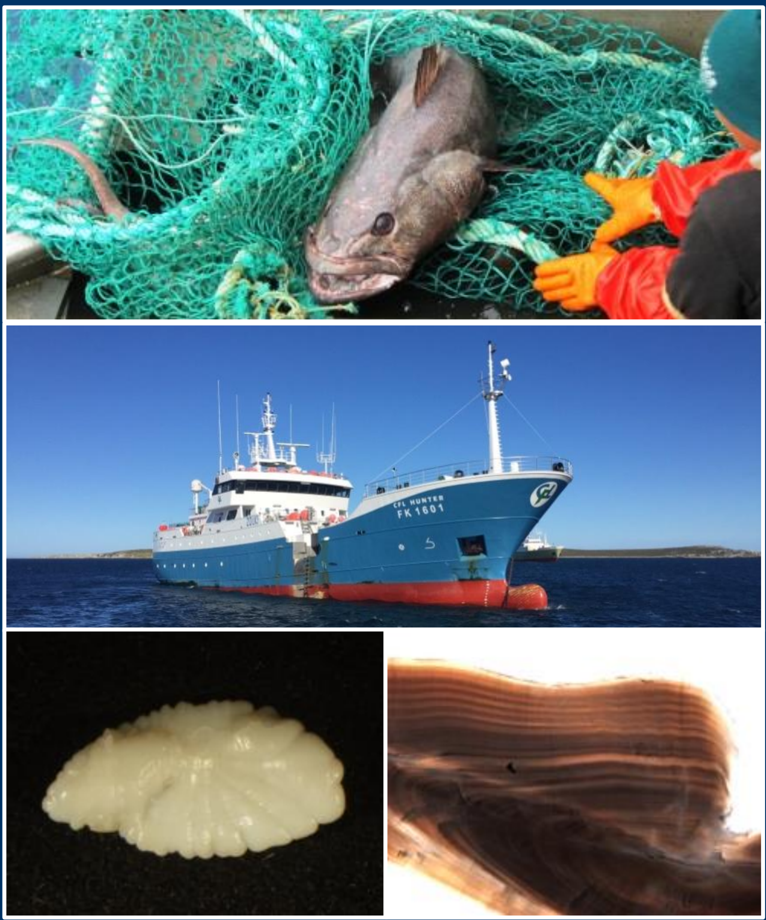
Stock assessment of Patagonian toothfish (*Dissostichus eleginoides*) in the Falkland Islands to 2022



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Summary

1. This report provides an updated Bayesian age-structured stock assessment of Patagonian toothfish *Dissostichus eleginoides* in Falkland Islands waters, using data up to the end of 2022. A new feature of this assessment is the inclusion of toothfish tag-release and tag-recapture data into the model.
2. The initial spawning stock biomass SSB_0 was estimated at 24,429 tonnes, and the current spawning stock biomass SSB_{2022} at 11,416 tonnes, both lower than in the previous assessment (by 8.9 and 4.9%, respectively).
3. The ratio of current spawning stock biomass to initial spawning stock biomass (SSB_{2022}/SSB_0) was estimated at 0.467, slightly lower than in the previous assessment (by 3.5%). According to established harvest control rules (HCR), SSB_{2022}/SSB_0 ratio places the stock in the *expansion range*.
4. Projections from the current model indicated that SSB/SSB_0 ratio will drop to the *target range* by 2026 and remain within the *target range* until the end of the 35-year projection period.
5. Based on HCR, the recommendation is to maintain toothfish annual TAC in the longline fishery at its current level of 1,040 tonnes.

1. Introduction

Patagonian toothfish (*Dissostichus eleginoides*; hereafter toothfish) is a long-lived slow-growing species found on the shelves and slopes of South America and around the sub-Antarctic islands of the Southern Ocean. In Falkland Islands waters, toothfish spawn on the slopes of Burdwood Bank at ca. 1000 m depth, with a minor spawning peak in May and a prominent peak in July-August (Laptikhovsky *et al.* 2006). The eggs, larvae, and small juveniles (<10 cm TL) develop and grow in epipelagic layers of the Falkland Current, with early juveniles of 10-12 cm TL (<1 year old; Lee 2017) occurring on the Patagonian shelf at depths ~100 m (Arkhipkin and Laptikhovsky 2010). Immature toothfish remain on the shelf for 3-4 years and then undertake a characteristic ontogenetic migration into deeper waters where adults reside and spawn (Arkhipkin and Laptikhovsky 2010).

The Falkland Islands toothfish longline fishery began in 1992 as an exploratory fishery and became an established fishery in 1994 (Laptikhovsky and Brickle 2005). Fishing was traditionally conducted using the 'Spanish' longlining system (with a few vessels using the *Mustad Autoline* system in the early years) until the 'umbrella' system was introduced in 2007 to reduce the loss of hooked toothfish to depredation. The umbrella system consists of hooks set in clusters, with an umbrella of buoyant netting attached above each cluster. The umbrella floats above the cluster whilst the gear is on the seabed but folds over the cluster and hooked fish during hauling, preventing depredation (Brown *et al.* 2010). Following initial trials in 2007, all vessels operating in the Falkland Islands longline fishery adopted the umbrella system in 2008.

Besides targeted longline fishery, toothfish are also taken as bycatch in the shelf-based (<400 m depth) finfish and calamari trawl fisheries. In the finfish fishery, toothfish is a commercially valuable bycatch; in the calamari fishery, toothfish are typically discarded due to the small size of the specimens (20-30 cm TL). These fisheries exploit different parts of the toothfish population in distinct areas: longlining occurs on the deep-water slope, 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 presents an updated Bayesian age-structured stock assessment for toothfish in Falkland Islands waters, using data up to the end of 2022.

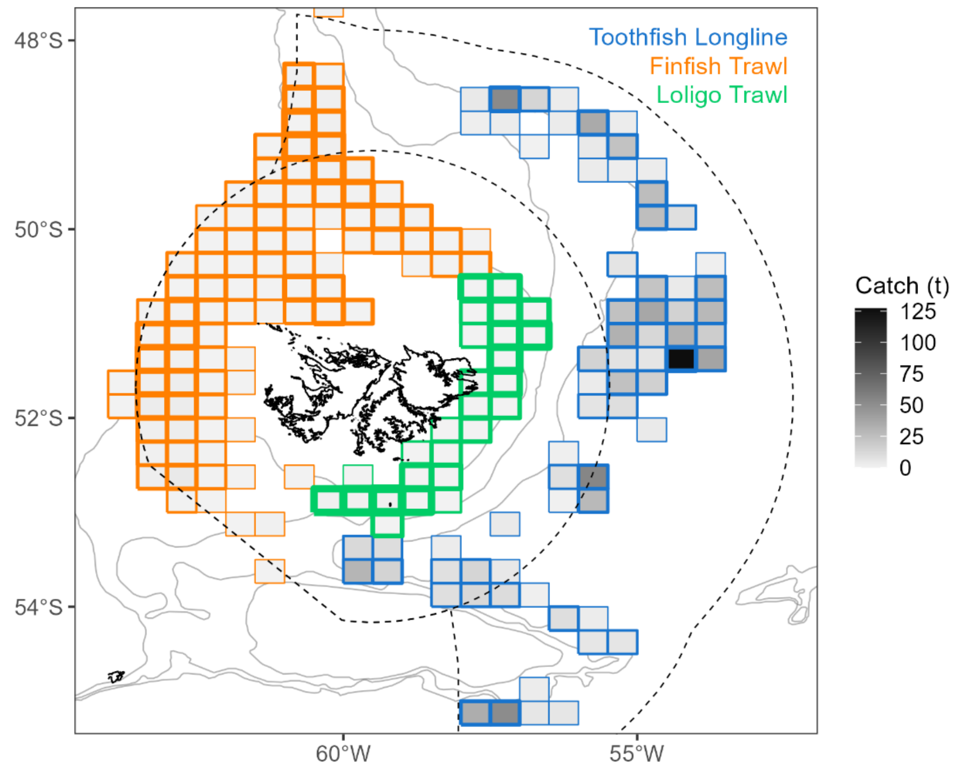

Figure 1. Distribution of toothfish catch and effort by fisheries in 2022. The thickness of grid lines is proportional to vessel-days. Grey-scale is proportional to toothfish catch biomass.

1.1. Stock structure and assumptions

A long-term research programme investigating the stock structure of toothfish on the Patagonian Shelf revealed complex patterns (Lee 2023). Research results indicate high levels of spatial-temporal variability in the extent of connectivity during the early life-history phases of egg and larval dispersal. Recruitment on the Falkland Islands Shelf arises from both Burdwood Bank and southern Chile spawning areas. It is proposed that high recruitment pulses are dominated by input from the Burdwood Bank spawning contingent. These pulses show strong spatial-temporal variability. Stable recruitment at lower levels occur from the southern Chile spawning contingent where they are retained on the western shelf of the Falkland Islands. Further, evidence of connectivity across the region through the active migration of adults appears to occur on a relatively small scale. Current results demonstrate that the stock structure arising from the retention of mixed contingents across the Falklands Shelf remains discrete (within the Falkland Islands Conservation Zone) until the adult life-history stages. Therefore, considering the currently available information, for this assessment we assumed that there is one discrete toothfish stock present in Falkland Islands waters.

2. Methods

The stock assessment was undertaken using a statistical catch-at-age model implemented in CASAL (Bull et al. 2012) version v2.30-2016-05-01 rev. 5470. The model assumes a single homogenous area, but the spatial heterogeneity of the population is represented by three geographically distinct commercial fisheries (longline, calamari trawl, and finfish trawl) and their gear-specific selectivity and fish availability (i.e. fleets-as-areas approach). The longline fishery is further split into two distinct fisheries according to gear type (Spanish-system and umbrella-system longline) to accommodate for different catchability between the two.

2.1. Model updates

Compared to the previous assessment (Skeljo *et al.* 2022), this assessment was updated with (a) catch and effort data for the umbrella-system longline fishery in 2022, (b) catch data for the finfish and calamari trawl fisheries in 2022, (c) biological data (size-structure and maturity) for all fisheries and research surveys in 2022 and (d) additional age readings for 2015-2019. Besides the regular data updates, a new feature of this assessment is the inclusion of toothfish tag-release and tag-recapture data into the model.

2.2. Data

Several datasets were used to inform the assessment, either as observations or input parameters. Observations appear in the objective function and are used to fit the model; input parameters are estimated outside the model and then treated as fixed parameters within the model (Table 1).

Table 1. Data used in the assessment model.

Data type	Data	Time series
Observations	**CPUE**	
	Spanish-system longline	1996-2007
	umbrella-system longline	2007-2022
	Catch-at-age	
	Spanish-system longline	1992, 1994-2009
	umbrella-system longline	2007-2022
	finfish trawl	1988-1992, 1997-1999, 2002-2022
	calamari trawl	2008-2022
	groundfish survey	2015-2022
	calamari pre-season survey	2015-2022
	Tag recaptures	
	umbrella-system longline	2017-2022
	Fish scanned for tags	
	umbrella-system longline	2017-2022
Input parameters	**Removals**	
	Spanish-system longline	1992-2010, 2013
	umbrella-system longline	2007-2022
	finfish trawl	1987-2022
	calamari trawl	1987-2022
	groundfish survey	2015-2022
	calamari pre-season survey	2015-2022
	Length-weight relationship	
	all fisheries combined	1989-2022
	Von Bertalanffy growth	
	all fisheries combined	2014-2021
	Maturity	
	all fisheries combined	1988-2022
	Tag releases	
	umbrella-system longline	2016-2021

Catch-per-unit-effort (CPUE)

Although CPUE data were available for all four fisheries, only the longline CPUE was used as a relative abundance index in the model; a decision motivated by the inconsistency of the toothfish CPUE in trawl fisheries, where its bycatch may change due to factors other than stock abundance (e.g. fisheries switching targets or areas). Longline CPUE data were divided between the Spanish- and umbrella systems to allow for different catchability between the two, as has previously been observed (Brown *et al.* 2010). During the transition from the Spanish- to the umbrella system (2007-2009), both 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 in an umbrella system started low and gradually increased to ~50%, followed by a rapid switch to fully adopting the umbrella system (however, timing differed between vessels). Because our analysis uses data aggregated by day, daily catch reports with both types of lines set by the same vessel had to be resolved; we assigned daily catch reports with >90% of hooks set in an umbrella system to the corresponding fishery and excluded 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 classify them.

For the Spanish-system longline fishery, 95 daily catch reports from remote areas (outside the region 47° W - 70° W and 40° S - 57° S) were excluded from the analysis. These records belong to the early years of the fishery (1998-2002) when presumably more exploratory fishing took place. Regarding the umbrella-system longline fishery, only catch reports belonging to Falkland Islands vessels were used. Since the onset of the umbrella system, the fishery was dominated by a Falkland Islands vessel (*CFL Gambler*, replaced by *CFL Hunter* in 2017), occasionally assisted by up to two chartered Chilean vessels. None of the chartered vessels participated in the Falkland Islands fishery in more than two years since 2007, resulting in inconsistent CPUE data. 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) that were not in place for the Falkland Islands vessel. The above led to the conclusion that only the Falkland Islands vessels CPUE should be used as an index of abundance. With a similar goal, data from dedicated 'tagging trips' and longlines set at depths <600 m were excluded from the analysis; tagging trips because part of the actual catch was unreported (released fish), and shallow-water sets because these were experimental fishing to collect brood stock for the toothfish rearing facility (commercial longlining is prohibited at depths <600 m).

CPUE was calculated from the selected catch reports, for each fishing day, as reported toothfish catch in kg-per-hook (Spanish system) or kg-per-umbrella (umbrella system). Finally, CPUE was standardised using a generalised linear mixed modelling approach (GLMM), providing a time series of CPUE values as relative abundance indices (Appendix 1). The observation error of the CPUE indices was accounted for in the assessment model via the coefficient of variation (CV) estimates obtained directly from a GLMM standardisation. To account for any additional variance on top of observation error, that may arise from the differences between model simplifications and real-world variation, a process error CV = 0.1 was assumed. The CPUE indices were assumed log-normally distributed about the model-predicted vulnerable biomass via a catchability parameter.

Catch-at-age (CAA)

Age readings used for the assessment were restricted to otoliths sampled in 2014-2021; these were processed and aged at FIFD and were considered the most reliable toothfish age estimates available (Lee 2014, 2015, 2016, 2017, 2018, 2019, 2020). Since the last assessment (Skeljo *et al*. 2022), an additional 261 age readings belonging to otoliths collected in 2015-2019 became available; otoliths collected in 2022 are yet to be processed. In total, 2,914 toothfish age estimates belonging to longline fisheries and 3,120 age estimates belonging to trawl fisheries were used to construct two corresponding age-length keys: $ALK_{LONGLINE}$ and ALK_{TRAWL}. Next, length frequencies of 166,324 toothfish randomly sampled in 1988-2022 were split between the corresponding fisheries or surveys. Age was assigned to each fish by the conditional probability of the appropriate age-length key, i.e. $ALK_{LONGLINE}$ for the longline fisheries, and ALK_{TRAWL} for the calamari fishery, finfish fishery, and surveys. Ages ≥31 years were assigned to a plus group. Then, fish counts-at-age were catch-raised per haul (i.e. multiplied by the catch/sample weight ratio in each observed haul), and aggregated per year and fishery and converted to proportions-at-age. For calamari trawl fishery, CAA data up to 2008 were considered unrepresentative and thus excluded from the analysis (Skeljo and Winter 2020, 2021). The ageing error was accounted for by deriving an error misclassification matrix from a normal distribution with CV = 0.1. The CAA data were assumed independently multinomially distributed about the model-predicted CAA.

A consideration in integrated models is to ensure that the observations are given appropriate weights in the objective function. The CAA data were weighted via effective sample sizes, 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 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:

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

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 re-run with the adjusted effective sample sizes (Table 3).

<u>*Removals*</u>

Removals accounted for the reported catches in the Falkland Islands waters, IUU catches in the Falkland Islands waters and catches lost to undetected whale depredation.

Catch reports from all available years for the four fisheries and two research surveys were used, starting in 1987. Trawl catch reports without licence information were considered calamari trawls if the dominant species in the catch was *Doryteuthis gahi*, and finfish trawls otherwise.

No information on IUU fishing within the Falkland Islands waters was available; therefore, we utilized the data for the Antarctic region (Table 2, Agnew *et al*. 2009), which gives estimates of IUU catches as a proportion of reported catches in 1980-2003. For later years, we took grey-literature estimations (e.g. CCAMLR Secretariat 2010) that IUU fishing decreased significantly in the southern oceans and assumed IUU catches equal to 5% of the reported catches in the Falkland Islands waters.

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 accounted for in the catch reports. 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 signs of whale depredation. Models included parameters longline position, fishing depth, year, month, number 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 the stock assessment, and by projecting the depredation ratios of the models rather than the models themselves, which improved the avoidance of outlier extrapolations.

Adding the reported catches, assumed IUU catches, and estimated whale depredation resulted in total removals used in the assessment model. The removals are treated as input parameters in CASAL and were assumed known without error.

Length-weight relationship

The length-weight relationship was calculated as $W = aL^b$ based on the length and weight measurements of 39,920 toothfish sampled randomly by the observers from commercial catches in 1989-2022. Length-weight parameters are given in Table 4.

Von Bertalanffy growth

The length-at-age relationship was described by the von Bertalanffy growth model $L = L_{inf}\left(1 - e^{-k(age-t_o)}\right)$ based on age estimates and length measurements of the combined 2,914 + 3,120 = 6,034 toothfish sampled randomly by observers from commercial catches in 2014-2021. Von Bertalanffy growth model parameters are given in Table 4.

Natural mortality

Natural mortality (M) was assumed to be 0.165 (Payne *et al.* 2005) and time- and age-invariant. A fixed M value has been introduced in the previous assessment (Skeljo *et al.* 2022) to prevent M from gaining exaggerated values unlikely for long-lived species like toothfish. Even though M would ideally be estimated within the model, there is likely not enough information in the available observations to do so reliably; all toothfish assessments done in CCAMLR waters assume a fixed M value (Earl and Readdy 2021a, 2021b; Ziegler 2021; Massiot-Granier *et al.* 2021a, 2021b; Grüss *et al.* 2021). Model sensitivity to different assumed M values was explored by Skeljo *et al.* (2022).

Maturity

A maturity-at-age vector was based on the maturity stage data estimated by observers for the 166,324 toothfish sampled randomly from commercial catches during 1988-2022. 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 maturity data strongly indicated some older fish were erroneously assigned 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 maturity data were corrected accordingly (Farrugia and Winter, 2018). Finally, instead of the more typical logistic function, the maturity ogive was fitted using GAM resulting in a maturity-at-age vector with the proportion of mature fish in each age class from 1 to 31+ (plus group). The maturity-at-age vector is given in Table 4.

Tag releases and recaptures

The tagging programme for the Falkland Islands toothfish commenced in 2016, aiming to improve our understanding of toothfish movement patterns within the region. The initial goal of tagging 3000 fish was achieved during four tagging research surveys onboard the longliner in 2016-2018. In addition to surveys, observers have been tasked to tag an average of 25 toothfish per week during their trips onboard the longliner. However, the tagging programme has largely been reliant on dedicated research surveys; in their absence, the number of tagged toothfish declined considerably in 2019-2020. In response, a 4-year extension of the tagging programme has been recommended (Lee and Skeljo 2020) and followed up by renewed tagging efforts since 2021 (Skeljo and Pearman 2021, Nicholls and Raczyński 2023) with a goal of tagging ~1000 longline-caught fish annually, i.e. one fish per tonne of TAC. Since 2016, 5546 toothfish have been tagged and released, and 396 have been recaptured (Table 2).

Table 2. Numbers of longline tag releases and recaptures. Numbers with grey shading were used in the assessment model (i.e. within-year recaptures were excluded). Numbers in brackets are additional recaptures from outside of the FCZ that were used to calculate the tag emigration rate, but not to fit the assessment model.

Releases		**Recaptures**							
Year	**Number**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**Total**
2016	437	14	15	3	7	8	6	8	61
2017	685	-	6	7 (3)	10 (1)	12 (2)	12 (1)	15 (1)	62 (8)
2018	2188	-	-	4 (1)	24 (2)	33 (7)	88 (1)	44 (4)	193 (15)
2019	127	-	-	-	1 (1)	4	6	1	12 (1)
2020	171	-	-	-	-	0	3	1	4
2021	866	-	-	-	-	-	13	22	35
2022	1072	-	-	-	-	-	-	5	5
Total	5546	14	21	14 (4)	42 (4)	57 (9)	128 (2)	96 (5)	372 (24)

The spatial distribution of toothfish tag releases, recaptures, and longline fishing effort is given in Figure 2; it shows that (a) the spatial extent of longline fishing effort remained consistent between 2016-2022, (b) tag releases were well spread across the fishing area in high-release years (2017, 2018, 2021) compared to a more localised spread in low-release years (2019, 2020), and (c) spatial overlap between fish releases and their subsequent recaptures was high.

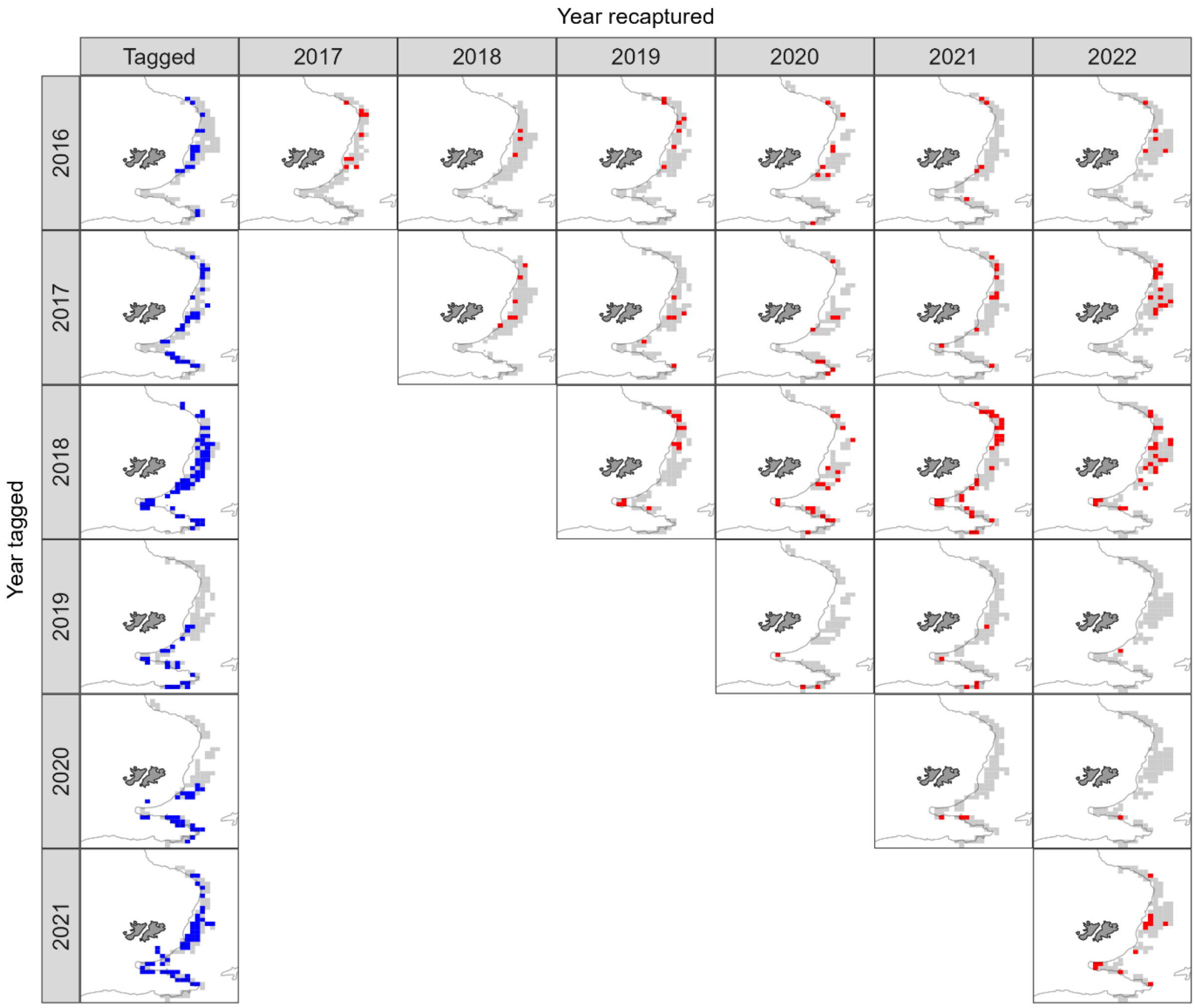

Figure 2. Spatial distributions of toothfish tag releases in FCZ in 2016-2021 (blue), and distributions of recaptures (red) in subsequent years (2017-2022); grey shading denotes the distribution of longline fishing effort in each year. Cells correspond to locations were at least one tagged fish was released (blue) or recapture (red), or where at least one longline was hauled (grey).

The current assessment used the data on longline-caught toothfish tag releases in 2016-2021 and their subsequent recaptures by longline in 2017-2022 (Table 2). Within-year recaptures were excluded from the model to restrict recapture data to tagged fish that had sufficient time to mix with the untagged population, at least at the local population level. The CASAL modelling framework allowed the specification of several tag-related parameters, i.e. initial tag-release mortality, tag-related growth loss, tag detection rate, and tag shedding rate. These parameters were obtained from the literature or estimated outside of the stock assessment model; then used as fixed input values in the assessment.

Initial tag-release mortality was assumed to be 10% (Agnew *et al.* 2006), effectively reducing the number of tagged fish in the population at the time of tagging. Tagging was assumed to result in a retardation of growth in individual toothfish equivalent to half a year of zero growth immediately after tagging (Parker *et al.* 2013). The tag detection rate in the longline fishery was assumed to be 100% due to the high observer coverage (~50%), an excellent record of cooperation with a sole vessel operating in the fishery since 2017, and the fact that each fish is handled individually by several crew-members while un-hooked, processed and packaged.

All tagged toothfish in 2016-2022 have been double-tagged, with a large and a small dart tag. By 2022, 34 double-tagged fish have been recaptured with only large tags and 16 with only small tags remaining, allowing the estimation of tag-shedding rates. Tag shedding rates were calculated following the approach outlined by Dunn *et al.* (2011) and Ziegler (2017) but adapted here to accommodate non-identical tags. The instantaneous shedding rate of large tags was λ_1 = 0.0173 and of small tags λ_2 = 0.0360. Since the CASAL algorithm requires a tag shedding rate for a single-tagged fish only, tag shedding rates for double-tagged fish had to be approximated by a single-tag shedding rate, estimated as λ_S = 0.0027.

Emigration of tagged fish out of the assessed area can violate the assumptions of tag recapture models used in the assessment and result in biomass and stock status being over-estimated. We estimated the emigration rate of toothfish from Falkland Islands waters based on the tag recaptures that occurred inside and outside of the FCZ, following the approach of Burch *et al.* (2017). For the recaptures outside the FCZ, we relied on the reporting by fishing fleets operating in these regions, specifically through collaborative work with Chile and the High Seas Korean fleet. The approach of Burch *et al.* (2017) requires the knowledge of annual harvest rates (i.e. catches and vulnerable biomass) both inside and outside of the FCZ; given the lack of reliable information on overall harvest rates outside of the FCZ, we assumed them equal to inside the FCZ. Tag reporting was assumed to be 50% outside the FCZ, compared to 100% within the FCZ. The tag emigration rate was estimated at λ_M = 0.0478; the tag shedding parameter supplied to CASAL (λ_S) was increased by the value of the emigration rate, providing a simple yet effective approach to correct for the effects of emigration (Burch *et al.* 2017). Methods used to estimate tag shedding and emigration rates will be detailed in a separate report.

The numbers-at-length of toothfish tagged each year, by 10 cm length bin, were used as fixed input in the assessment model. The numbers-at-length of toothfish recaptured each year for each release year, by 10 cm length bin, were used as observations to fit the model. Besides releases and recaptures, the model required the numbers scanned at length (i.e. numbers of fish caught and inspected for a possible tag) in each recapture year; as detection was assumed 100% this was derived by disaggregating the total annual catch in numbers into 10 cm length bins, using the random length-frequency samples from the fishery observer records. For each recapture year, the recaptures-at-length from each release year *t* were fitted, in 10 cm length bins, using a robust binomial log-likelihood:

$$-LL = \sum_i \left[\log(n_i!) - \log\big((n_i - m_i)!\big) - \log(m_i!) + m_i \log\left(Z\left(\frac{M_i}{N_i}, r\right)\right) + (n_i - m_i)\log\left(Z\left(1 - \frac{M_i}{N_i}, r\right)\right)\right]$$

where n_i is the number of scanned fish in length bin *i*, m_i is the number of recaptured fish from release year *t* in length bin *i*, N_i is the expected number of fish in length bin *i* in the available population (tagged and untagged), M_i is the expected number of fish in length class *i* in the available population that have the tag from release year *t*, and *Z*(*x,r*) is a robustification function defined as:

$$Z(x,r) = \begin{cases} x \ \ where \ \ x \geq r \\ r/(2 - x/r) \ \ otherwise \end{cases}$$

where *r* is a non-negative robustification constant; a default value of *r* = 1e-11 was used in the assessment.

The default assumption in the assessment is that the numbers of recaptures are independent between release years and between length bins, and follow a robust binomial distribution. There is evidence that some (probably most) tag-recapture datasets are over-dispersed; that is, they are more variable than would be expected from the above assumptions and thus should be down-weighted (Francis 2016). CASAL provides parameter ϕ as an informal means of allowing for over-dispersion, with the log-likelihood of tag-recapture data being divided by ϕ. The default is ϕ = 1; setting ϕ > 1 implies over-dispersion and down-weights the data (Francis 2016). In the current assessment, the procedure was to use the default ϕ = 1 in the initial model run and use the initial MPD model fit to calculate over-dispersion ϕ_j for each tagging event *j* from its *i* recapture events as:

$$\Phi_j = var\left(\frac{O_{ij} - E_{ij}}{\sqrt{E_{ij}}}\right)$$

where O_{ij} was the observed number of recaptures and E_{ij} was the estimated number of recaptures. Over-dispersion terms for each of the release years were then combined by taking the geometric mean, and the model was re-run with the log-likelihood of tag-recapture data modified by dividing with ϕ (Table 3).

As both CAA and tag-recapture data had to be reweighted, the reweighting was applied first to the CAA data, then to the tag-recapture data (i.e. in successive MPD model runs).

Table 3. Weighting factors (w) used to adjust effective sample sizes of catch-at-age data, and the over-dispersion parameter (ϕ) used to weight tag-recapture data.

w_{LLH}	w_{LLU}	w_{FIN}	w_{LOL}	w_{RFIN}	w_{RLOL}	ϕ
0.018	0.033	0.016	0.018	0.026	0.014	2.843

Table 4. Input parameters assumed in the assessment model.

Component	Parameter	Value				
Length-weight	a ($t \cdot cm^{-1}$)	5.94e-9				
	b	3.123				
Von Bertalanffy growth	L_{inf} (cm)	172.80				
	k (y^{-1})	0.067				
	t_0 (y)	-2.310				
	CV	0.152				
Natural mortality	M (y^{-1})	0.165				
Maturity *(proportion mature at age)*	Age 1	0	Age 12	0.401	Age 23	0.565
	Age 2	0.008	Age 13	0.425	Age 24	0.578
	Age 3	0.040	Age 14	0.446	Age 25	0.592
	Age 4	0.077	Age 15	0.466	Age 26	0.607
	Age 5	0.120	Age 16	0.483	Age 27	0.621
	Age 6	0.167	Age 17	0.497	Age 28	0.637
	Age 7	0.216	Age 18	0.510	Age 29	0.654
	Age 8	0.264	Age 19	0.520	Age 30	0.671
	Age 9	0.306	Age 20	0.530	Age 31+	0.690
	Age 10	0.343	Age 21	0.541		
	Age 11	0.374	Age 22	0.552		
Steepness	h	0.75				
Future recruitment variability	σ_R	0.6				
Ageing error	CV	0.1				
Initial tag-release mortality		0.1				
Tag shedding rate	λ_S (y^{-1})	0.0027				
Tag emigration rate (added to the tag shedding rate in the model)	λ_M (y^{-1})	0.0478				
Tag detection rate		1				
Tag-related no-growth period	(y)	0.5				

2.3. Model setup

Population dynamics

Toothfish population dynamics were described by a Bayesian age-structured model, with age classes from 1 to 31+ years, the last being a plus group. It is a single-sex, single-area, multi-fishery model with an annual cycle divided into three time-steps. Recruitment, fishing mortality from all concurrent fisheries, tag releases and recaptures, 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 (R0) with estimated year class strength (YCS) and a stock-recruitment relationship. Stock recruitment was assumed to follow a Beverton-Holt relationship:

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

where *R* is the recruitment, *SSB* is the spawning stock biomass, SSB_0 is the pre-exploitation equilibrium spawning stock biomass, and *h* is the steepness parameter, defined as the fraction of recruitment from the unfished population when the spawning stock biomass declines to 20% of its unfished level. Recruitment was fixed rather than estimated, as suggested for example by He *et al.* (2006) and Kenchington (2014) and the steepness parameter was assumed to be *h* = 0.75 (Myers *et al.* 1999, Punt *et al.* 2005, Dunn *et al.* 2006).

The initial year in the model was 1987, the first year of recorded data by the FIFD, and the model was run up to 2022. Projections from the model extended for another 35 years, up to 2057. Conditions in the initial year were assumed to be an equilibrium age structure at an unexploited equilibrium biomass (i.e., a constant recruitment assumption).

Within the model, each year's tagging event was included as an additional partition, i.e. the model kept account of the numbers of fish tagged in each year separately. The numbers of fish in the tagged component were modified by initial tag-related mortality (as a proportion) followed by a subsequent ongoing annual tag loss (at a constant rate). The population processes (natural mortality, fishing mortality, ageing, etc.) were then applied collectively over the tagged and untagged components of the model. The proportions-at-age of tagged fish were determined within the model by multiplying the observed proportions-at-length of tagged fish by the proportions of fish at age by length assumed by the model for the overall population (Mormede *et al.* 2014).

<u>*Estimation method*</u>

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

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

<u>*Estimated parameters*</u>

Parameters estimated by the model, their priors, starting values and bounds are given in Table 5. Annual year class strengths (YCS) were estimated for 1986-2021, with the *Haist parameterisation* used to make the YCS parameters average to 1 over 1986-2016 (for the Haist method description, see Bull *et al.* 2012). The catchability coefficient (q) was estimated separately for the two longline fisheries. Uniform-log priors were considered appropriate for SSB_0 (Mormede *et al.* 2014, Ziegler and Welsford 2015) and q (Hillary *et al.* 2006, Ziegler and Welsford 2015), lognormal for YCS (Candy and Constable 2008, Ziegler and Welsford 2015), and uniform for selectivity parameters (Dunn and Hanchet 2010, Mormede *et al.* 2014).

Table 5. Number of parameters (N), priors, start values and bounds for free parameters estimated in the assessment model.

Estimated parameter		N	Prior	Start value	Lower bound	Upper bound
SSB_0		1	uniform-log	40,000	10,000	100,000
YCS		35	lognormal	1	0.001	20
Selectivity $_{LLH}$	a_{50}	1	uniform	10	1	50
	a_{to95}	1	uniform	5	0.05	50
Selectivity $_{LLU}$	a_{50}	1	uniform	10	1	50
	a_{to95}	1	uniform	5	0.05	50
Selectivity $_{FIN}$	a_1	1	uniform	2	1	50
	S_L	1	uniform	1	0.05	50
	S_R	1	uniform	2	0.05	500
Selectivity $_{LOL}$		8	uniform	0.5	0	1
Selectivity $_{RFIN}$		6	uniform	0.5	0	1
Selectivity $_{RLOL}$		6	uniform	0.5	0	1
q_{LLH}		1	uniform-log	1e-5	1e-9	0.1
q_{LLU}		1	uniform-log	1e-5	1e-9	0.1

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

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

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

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

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

The *allvalues* ogive is the most flexible parameterisation, defined by one selectivity parameter for each age class. Since negligible numbers of toothfish older than eight years occur in the calamari trawl fishery, selectivity parameters were estimated for ages 1-8 and fixed to zero otherwise, reducing the number of estimable parameters. The same approach was used for research surveys but for ages 1-6. The empirical *allvalues* ogive was used for calamari trawl fishery and research surveys because standard selectivity curves, such as logistic or double-normal, could not fit the observed CAA data well, with the highest proportions in the catch corresponding to the lowest age classes (descending ogive).

Penalties

Besides observations and priors, the 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. A 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. A vector average penalty was used to encourage YCS to average 1. An ogive smoothing penalty was used to encourage *allvalues* selectivity ogives to be like a fourth-degree polynomial, i.e. smooth in appearance. Penalty multipliers were set to 100 for catch limits and 20 for the YCS vector average and ogive smoothing penalties (for details on penalty calculations, see Bull *et al.* 2012).

Projections

Projections were carried out by running the model for 35 years into the future, using randomised recruitments and hypothetical catches. A total of 1000 simulations were run, each using the same (MPD) estimate of model parameters and the same hypothetical future catches (i.e. simulations differed only in terms of the randomised recruitments). The most recent (2021-2022) and future recruitments (2023-2057) were assumed log-normally distributed with standard deviation σ_R = 0.6. Future catch-split between fisheries was assumed from catch history and the current longline catch quota: umbrella-system longline = 1,040 t, finfish trawl = 300 t, and calamari trawl = 30 t.

Deterministic yield calculations

Deterministic MSY is the maximum constant annual catch (using the specified catch split) that can be sustained under deterministic recruitment (i.e. YCS = 1). The corresponding mortality rate is F_{MSY}, and the corresponding SSB is B_{MSY}. Simulations are run for different values of mortality F, starting from an unfished equilibrium state and running until the total annual catch C_F and spawning stock biomass SSB_F stabilize. CASAL searches over mortality rates F to find F_{MSY}, the value that maximizes C_F. Then MSY and B_{MSY} are C_F and SSB_F, respectively. The calculations are based on a single set of model parameters (i.e. MPD).

3. Results

3.1. Model fits

Diagnostics plots of model fit to observed data are provided in Appendix 2. The model fitted standardised CPUE data of the umbrella-system longline fishery reasonably well, with 95% CI of the observations and estimates overlapping in all years but one. The fit to the Spanish-system data was slightly worse, underestimating observed CPUE in the early years and overestimating in the later years of the fishery; however, the model adequately captured the declining overall trend (Figure A.3). Corresponding trends in normalised residuals for both longline fisheries are shown in Figure A.4.

The model fit to catch-at-age data was good for all four fisheries and both research surveys (Figures A.5 - A.10). The corresponding residual bubble plots show no clear patterns, with the possible exception of longline fisheries, where the model tends to overestimate the proportion of 8-9 years old fish (Figure A.11). The model fit to the observed mean toothfish age was good in all cases except the Spanish-system longline fishery (Figure A.12).

The model fit to tagging data was generally good; however, notable discrepancies were found between the observed and expected recaptures of fish tagged in 2018 (the model overestimated the number of recaptures in 2019 and underestimated in 2021). That may be the result of (a) a comparatively large number of fish tagged in 2018, almost equal to all other years combined, and (b) variation in the spatial overlap of tagging and fishing effort, in combination with low movement rates of fish (Figures A.13 - A.15).

Likelihood profiles were carried out by fixing SSB_0 over a range of plausible values (15,000 - 60,000 t) while the remaining parameters were estimated. Only the Spanish-system CPUE data and recaptures of fish tagged in 2019 favoured lower biomass estimates. The umbrella-system CPUE data, recaptures of fish tagged in 2020, and CAA data for trawl fisheries, Spanish-system longline and calamari pre-season survey all found higher biomass estimates more likely. The CAA data for the umbrella-system longline and groundfish survey and the recaptures of fish tagged in 2016, 2017, 2018 and 2021 favoured MPD biomass estimate (Figures A.16, A.17).

MCMC trace plots showed no evident lack of convergence in most of the estimated parameters, with the notable exception of selectivity parameters for calamari trawl fishery and both surveys (all associated with the *allvalues* descending selectivity ogive) (Figure A.18). The convergence test of Geweke (1992) and the Heidelberger and Welch (1983) stationarity and half-width tests suggested failure to converge for some of the mentioned selectivity parameters. Autocorrelations in the MCMC samples for mentioned selectivity parameters were high, indicating slow mixing in MCMC chains (Figure A.19).

Contributions to the objective function of each dataset, prior and penalty, are provided in Appendix 3.

3.2. Model estimates

MPD estimates (with MCMC 95% credible intervals) of initial spawning stock biomass (SSB_0), current spawning stock biomass (SSB_{2022}) and current spawning stock biomass relative to SSB_0 (SSB_{2022}/SSB_0) are given in Table 6. Estimates of SSB_0 and $SSB_{current}$ were approximately halfway between the estimates of the previous two assessments (8.9% and 4.9% lower than the last one); the estimate of $SSB_{current}/SSB_0$ was marginally lower than in the previous two assessments (3.5% lower than the last one). MCMC posterior distributions of SSB_0 and SSB_{2022}/SSB_0 (Figure 4) were noticeably narrower than in the previous assessment; likelihood profiles suggest this is likely due to tag-recapture data being highly informative on the SSB_0. The estimated historical SSB trend up to 2022 is shown in Figure 3. Deterministic MSY was estimated at 1,653 t (4.3% lower than in the previous assessment).

Table 6. MPD estimates (and MCMC 95% credible intervals) of SSB_0, SSB_{2022} and SSB_{2022}/SSB_0.

SSB_0	SSB_{2022}	SSB_{2022}/SSB_0
24,429 (22,567 - 27,042)	11,416 (9,691 - 14,289)	0.467 (0.428 - 0.529)

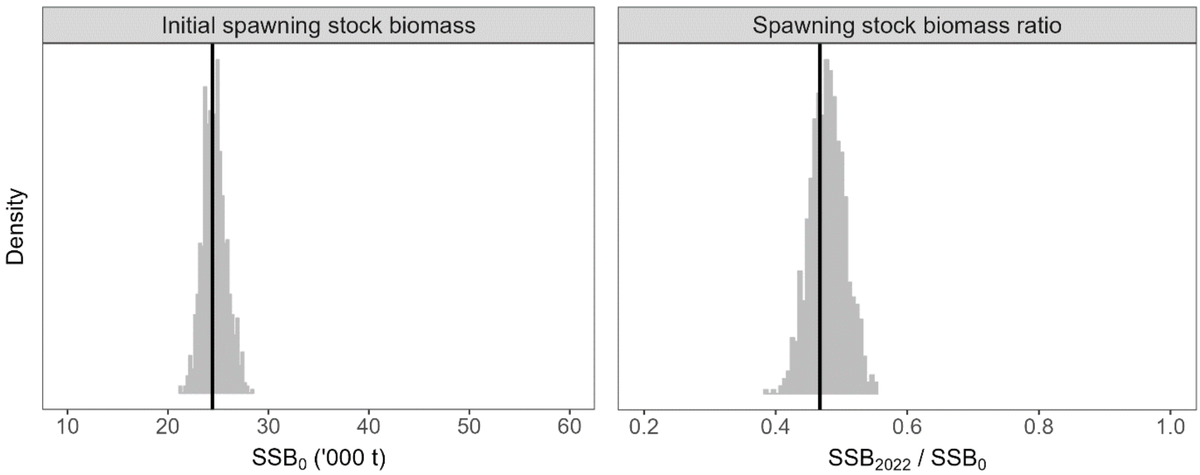

Figure 3. MCMC samples from posterior distribution of SSB_0 and SSB_{2022}/SSB_0. Vertical black lines denote MPD point estimates.

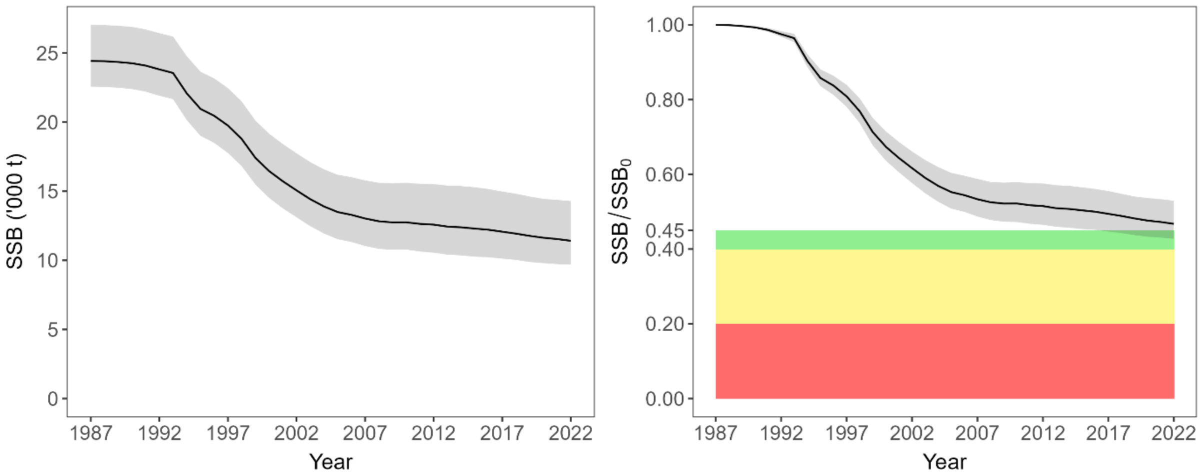

Figure 4. MPD estimate of the historical SSB trajectory; absolute on the left and relative to SSB_0 on the right (black line). Shaded areas denote MCMC 95% credible intervals of the model fit. Harvest control rule ranges are colour coded for reference: *target range* in green (SSB/SSB_0 = 0.45-0.40), *trigger range* in yellow (SSB/SSB_0 = 0.40-0.20) and *closure range* in red ($SSB/SSB_0 < 0.20$).

Estimated selectivity ogives appeared reasonable, in line with our knowledge of toothfish ontogenetic migrations and fishery interactions with the stock (Figure 5). The calamari trawl fishery catches the youngest fish; a combination of fishing in shallow waters (=only young fish is available) and using a small mesh size (=low gear selectivity), resulting in a descending right limb selectivity ogive. For the finfish trawl fishery (assumed dome-shaped) the model estimated maximum retention for 2-year-old fish and decreasing retention of younger (=escapes due to gear selectivity or not yet available at finfish trawl grounds) and older fish (=unavailable at trawling depths, i.e. moves to deeper waters). The two longline fisheries had almost identical selectivity curves (assumed logistic), catching predominantly older fish (=available in deeper waters). Survey selectivities were roughly comparable to commercial trawl fisheries, as surveys employ similar gear and cover similar grounds. However, survey selectivity parameters experienced convergence issues, resulting in wide MCMC 95% credible intervals.

Year class strength estimates for the most recent years (2015-2021) corresponded well to the recruitment estimates independent of the model (Lee *et al.* 2021, Skeljo 2023), with recruitment peaks in 2015 and 2017, followed by 4+ years of low recruitment (Figure 6). No independent survey data was available for the earlier years, making it difficult to confirm the YCS trend estimated in the model.

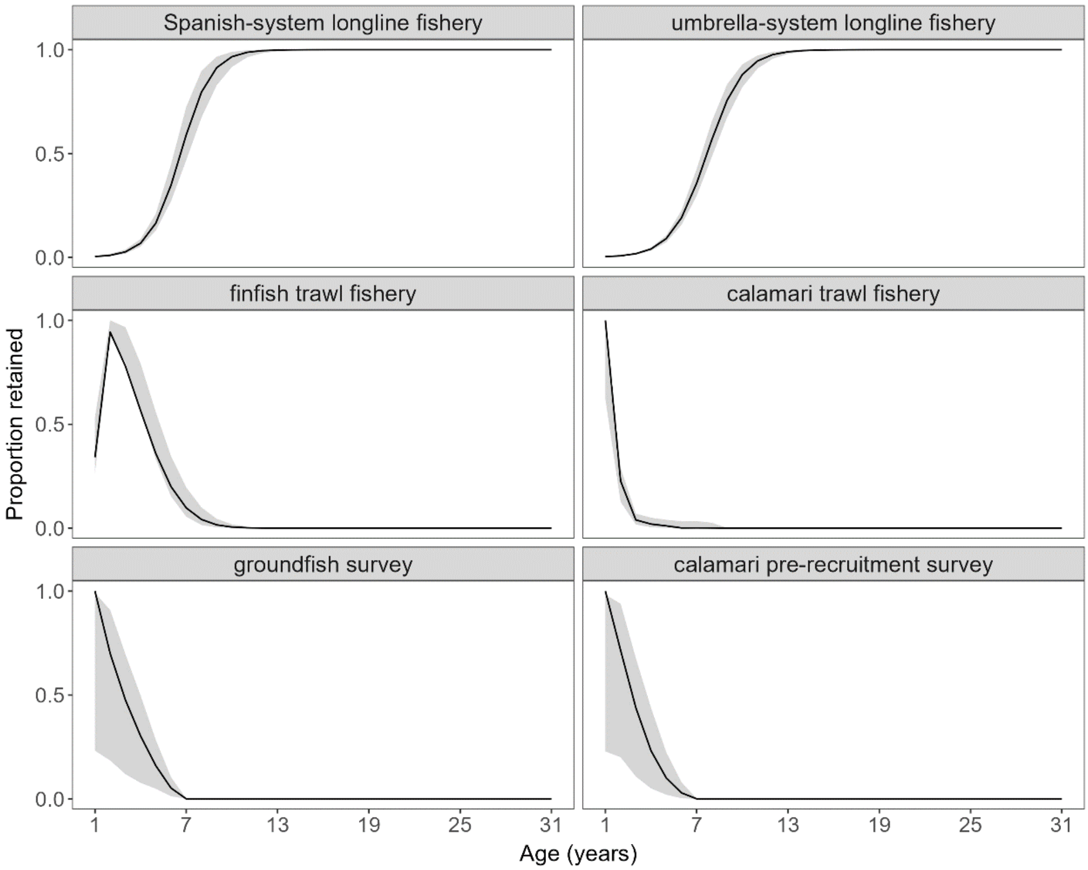

Figure 5. MPD estimate of selectivity ogives for four fisheries and two surveys (black lines). Shaded areas denote MCMC 95% credible intervals of the model fit.

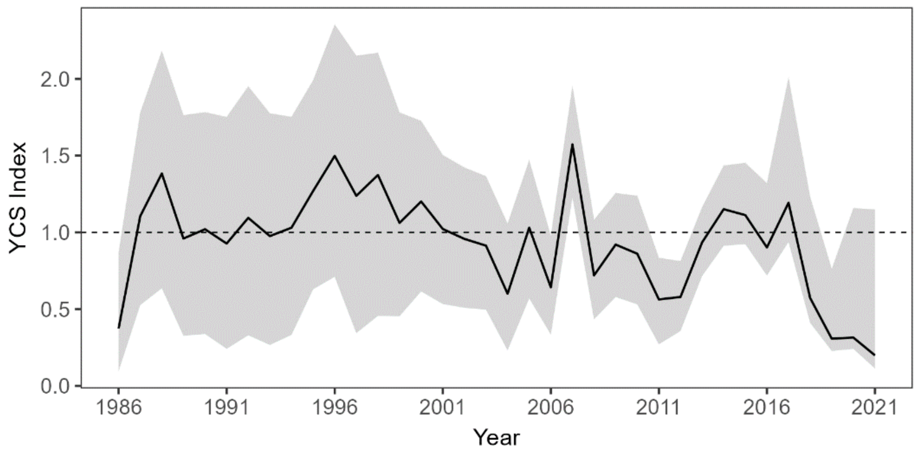

Figure 6. MPD estimate of year-class strengths in 1986-2021 (solid black line). Shaded area denotes MCMC 95% credible intervals of the model fit.

3.3. Model projections

Projections of SSB/SSB_0 under the assumption of random recruitments in 2021-2057 and constant annual catches in 2023-2057 are shown in Figure 7. The median SSB/SSB_0 is currently within the HCR *expansion range* but is projected to drop to the *target range* by 2026. Throughout the projection period, the median SSB/SSB_0 reaches a minimum of 0.43 before increasing to 0.45 by the end of the 35 years. The probability of the SSB/SSB_0 ratio falling below existing management thresholds, corresponding to the upper bounds of HCR ranges, was calculated for each year of the projection period as the proportion of the 1000 simulations below the respective threshold (Figure 8). The highest probability of falling below 0.45, 0.40 and 0.20 threshold during the projection period was estimated at ~64%, ~37% and 2.2%, respectively.

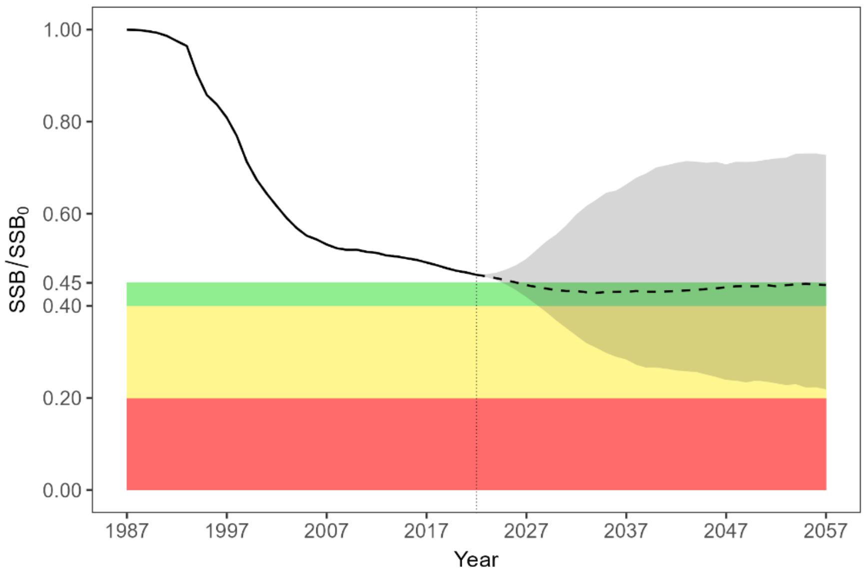

Figure 7. Future SSB/SSB_0 projections from the model MPD estimate. Shown are MPD estimate (solid black line), median of the projections (dashed black line), and 95% confidence intervals of the projections (shaded area). Harvest control rule ranges are colour coded for reference: *target range* in green (SSB/SSB_0 = 0.45-0.40), *trigger range* in yellow (SSB/SSB_0 = 0.40-0.20) and *closure range* in red ($SSB/SSB_0 < 0.20$).

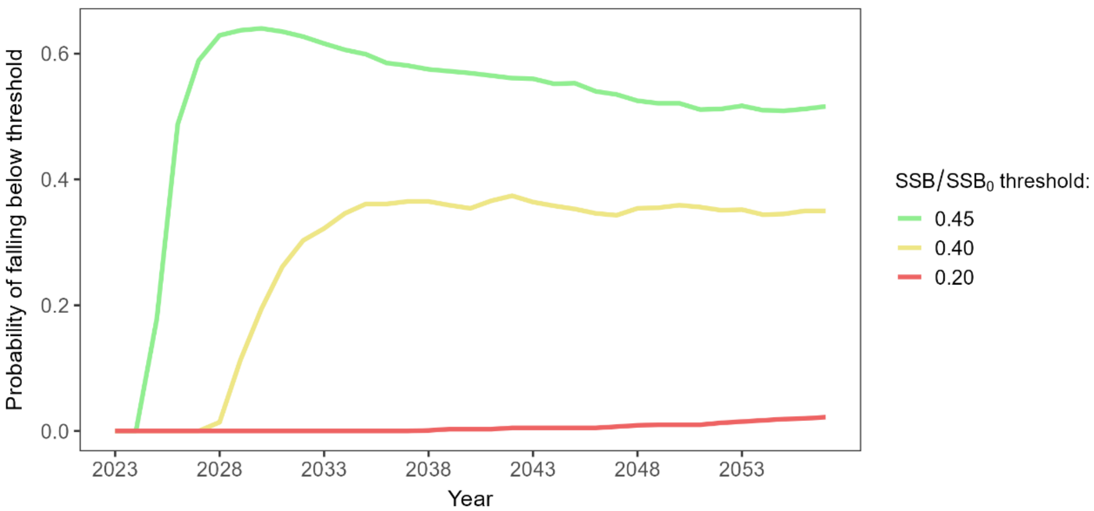

Figure 8. Probability of stock falling below designated SSB/SSB_0 management thresholds; based on projections.

3.4. Sensitivity analyses

Sensitivity analyses were carried out to better understand the influence of tag-recapture data and assumptions on the model outcomes. The results of sensitivity trials are summarised in Table 7, with the current model (base-case) included for reference.

In the first set of trials, we evaluated model sensitivity to different weights assigned to the tag-recapture data. Tag-recapture data are weighted via the tag dispersion parameter (ϕ); we tried both up-weighting (ϕ = 1) and down-weighting (ϕ = 5-100) tag-recapture data compared to the base-case scenario. Down-weighting beyond ϕ = 100 was tested but had a negligible further impact on model outcome and is not presented. Altering the weighting of the tag-recapture data had a minor effect on MPD model outcomes; up-weighting resulted in a slightly lower, and down-weighting in slightly higher estimates of SSB_0, SSB_{2022}, and SSB_{2022}/SSB_0 (Figure 9, top). Also, up-weighting resulted in a narrower SSB_0 likelihood profile, strongly constrained to the upper limit and suggesting that higher biomass estimates were less likely; conversely, progressive down-weighting resulted in progressively wider SSB_0 likelihood profiles, less constrained to the upper limit (Figure 9, bottom).

In the second set of trials, we evaluated model sensitivity to different assumed annual emigration rates (λ_M). Altering the values of the emigration rate over a plausible range had a small to moderate effect on MPD model outcomes. As expected, higher emigration rates lead to lower estimates of SSB_0, SSB_{2022}, and SSB_{2022}/SSB_0, and vice-versa (Figure 10). At the extreme ends of the analysed range, increasing the emigration rate from 0 to 10% resulted in an 8.3% decrease in SSB_0, 17.5% in SSB_{2022}, and 9.9% in SSB_0/SSB_{2022}.

Table 7. MPD estimates of SSB_0 and key derived quantities obtained from different sensitivity trials. SSB_0 and SSB_{2022} estimates are given in tonnes.

Model run	SSB_0	SSB_{2022}	SSB_{2022}/SSB_0
Alternative tag weighting			
ϕ = 1	24,290	11,251	0.463
ϕ = 2.843 (base-case)	24,429	11,416	0.467
ϕ = 5	24,561	11,560	0.471
ϕ = 10	24,742	11,754	0.475
ϕ = 20	24,848	11,890	0.478
ϕ = 50	25,060	12,087	0.482
ϕ = 100	25,146	12,178	0.484
Alternative emigration rate			
λ_M = 0.00	25,583	12,623	0.493
λ_M = 0.02	25,059	12,076	0.482
λ_M = 0.04	24,590	11,587	0.471
λ_M = 0.0478 (base-case)	24,429	11,416	0.467
λ_M = 0.06	24,175	11,148	0.461
λ_M = 0.08	23,808	10,765	0.452
λ_M = 0.10	23,468	10,409	0.444

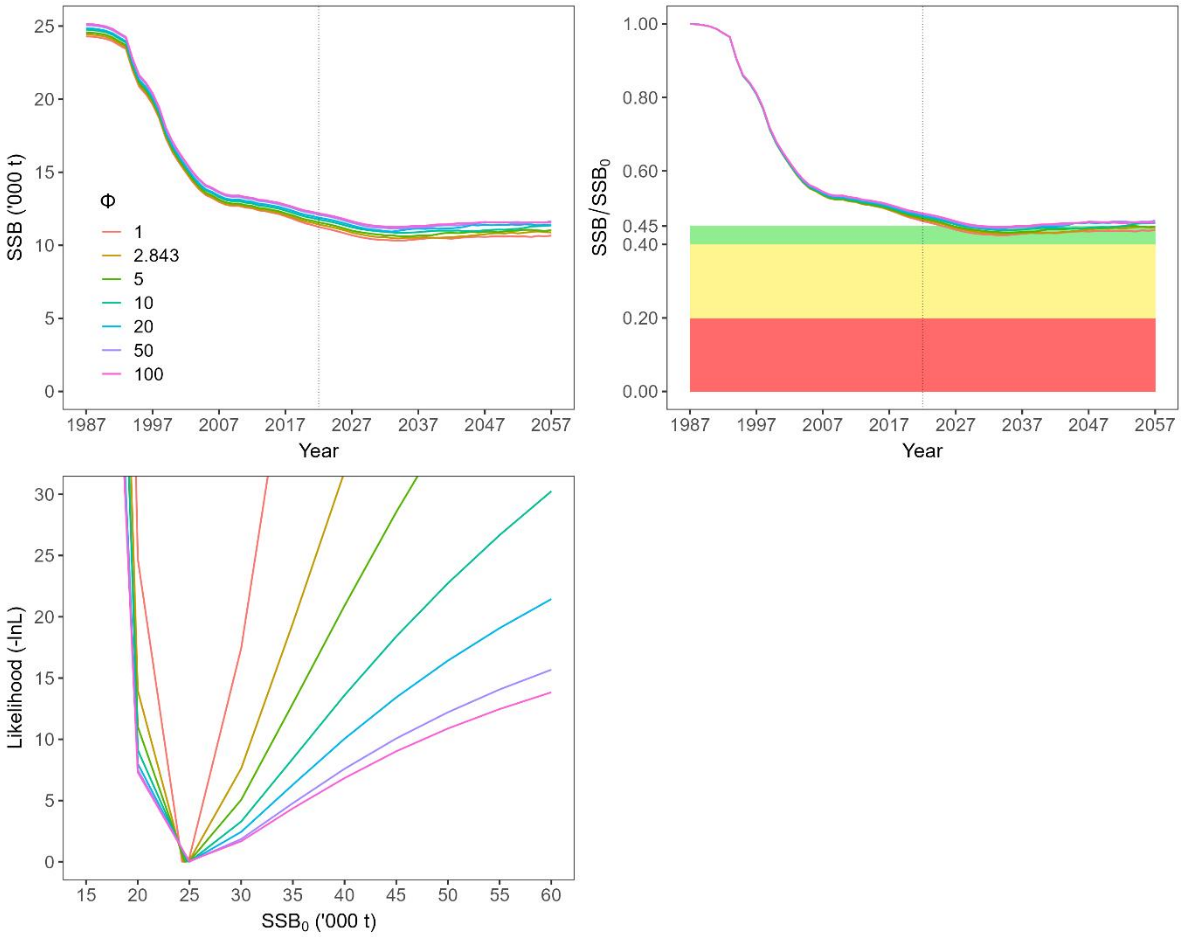

Figure 9. MPD estimates and future projected trajectories of SSB (top-left) and SSB/SSB$_0$ (top-right), and SSB$_0$ likelihood profiles (bottom), for sensitivity trials with alternative tag dispersion parameter (ϕ).

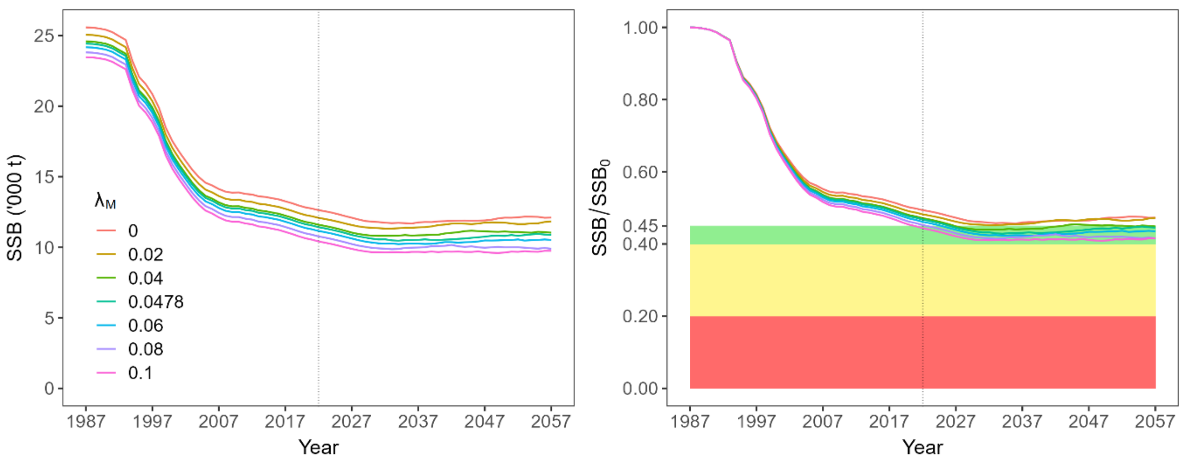

Figure 10. MPD estimates and future projected trajectories of SSB (left) and SSB/SSB$_0$ (right), for sensitivity trials with alternative emigration rates (λ_M).

4. Discussion

This report presents an updated age-structured stock assessment for toothfish in the Falkland Islands waters, using data up to the end of 2022. Besides the regular updates of the catch and effort data, biological data, and age readings, a new feature of this assessment was the inclusion of the toothfish tag-release and tag-recapture data into the model. Tag-recapture data were used as an index of absolute abundance, thus reducing model reliance on commercial CPUE data. Even though widely used as an index of relative abundance in stock assessment, shortcomings of commercial CPUE data are well known (Harley *et al.* 2001, Maunder and Punt 2004, Ye and Dennis 2009, Thorson *et al.* 2017, Maunder *et al.* 2006, 2020).

Tag-recapture data are routinely used in toothfish stock assessments. Indeed, all toothfish stocks assessments in CCAMLR waters, i.e. South Georgia, South Sandwich Islands, Heard and McDonald Islands, Kerguelen, Crozet Island, and Ross Sea, use extensive long-term tag-recapture datasets (Earl and Readdy 2021a, 2021b; Ziegler 2021; Massiot-Granier *et al.* 2021a, 2021b; Grüss *et al.* 2021). In comparison, the Falkland Islands toothfish tagging programme is still in development; it commenced in 2016, but the tagging effort varied widely over time. The initial goal of tagging 3000 fish was achieved in 2016-2018, followed by a considerable decrease in the tagging effort in 2019-2020. The programme has been expanded and extended as a long-term protocol in 2021, aiming to tag ~1000 fish annually (roughly one fish per tonne of TAC) to support the stock assessment. The current assessment was informed by ~4500 releases and ~330 recaptures (once within-year and out-of-zone recaptures were excluded). These numbers are still low, but future assessments will benefit from the extended tagging protocol; updating the model with the new tag-recapture data should become a routine task now that R and CASAL codes have been established.

Compared to the previous year, the current assessment resulted in a somewhat lower estimate of SSB_0, $SSB_{current}$, and $SSB_{current}/SSB_0$. Running multiple models with different data combinations indicated that the change was primarily driven by the tag-recapture data, with little effect from the CAA and CPUE data updates for 2022. The same was indicated by the first set of the sensitivity trials; severely down-weighting tag-recapture data resulted in an estimate comparable to the previous year (only slightly lower SSB_0 and $SSB_{current}$, and identical $SSB_{current}/SSB_0$). Even though the inclusion of tag-recapture data led to a less optimistic MPD estimate in the base-case model, the change from the previous assessment was relatively minor, considering this was a completely new and independent set of observations. A more notable change occurred in the shape of posterior distributions of SSB_0 and $SSB_{current}/SSB_0$; both were more strongly constrained to the upper limit, resulting in narrower and more symmetrical credible intervals. Examination of likelihood profiles suggested this was due to tag-recapture data being highly informative on the SSB_0; recaptures of fish tagged in all years but 2020 found high biomass estimates were less likely.

A feature of tag-recapture data was that it allowed us to account for toothfish emigration from the Falkland Islands waters (Lee *et al.* 2022). The emigration rate was estimated external to the assessment model from the numbers of tag recaptures inside and outside of the FCZ and then used as a fixed input in the assessment following the approach of Burch *et al.* (2017). When estimating the emigration rate, we had to make assumptions on annual harvest rates and the tag reporting rate outside of FCZ, as no such information was available at the time of analysis. Assumptions were precautionary but should be revisited if additional data become available. Toothfish emigrate from FCZ to Chilean waters (confirmed recaptures), High Seas (confirmed recaptures), and presumably Argentine waters (anecdotal evidence of three recaptures, unconfirmed); obtaining a reliable pooled estimate of annual harvest rates and tag reporting rate for these three areas might prove difficult. Given the uncertainty associated with the estimated emigration rate, its influence on the model outcomes has been tested in the second set of sensitivity trials. Assuming no emigration takes place resulted in an almost identical estimate of SSB_0, $SSB_{current}$, and $SSB_{current}/SSB_0$ to the previous year, while increasing it led to a progressively less optimistic model outcome. However, even under the assumption of a 10% emigration rate, the stock was estimated to be in the HCR *target range* and projected to remain in the *target range* in future. It is of note that only 24 toothfish have been recaptured outside of FCZ so far; the extended tagging protocol is expected to lead to an increased number of recaptures, thus improving our understanding of toothfish emigration.

Projections from the current assessment were somewhat less optimistic than in the previous assessment, with SSB/SSB_0 now projected to decrease from the *expansion range* to the *target range* in the near future. Partially, the decrease was due to the lower MPD model estimate of $SSB_{current}/SSB_0$, which was the starting point for projections. However, the main driver of the projected decrease was the cumulative effect of four consecutive years of weak toothfish recruitment; the model estimated below-average YCS in 2018-2021, in line with recruitment estimates external to the model (Lee *et al.* 2021). Running alternative projections starting from 2019, 2020, 2021, or 2022, i.e. following one, two, three, or four consecutive years of weak recruitment, showed that extended weak-recruitment periods result in an increased drop in projected SSB/SSB_0 within the next 5-10 years. This emphasizes the need to monitor juvenile toothfish abundance during research surveys and to protect high recruitment cohorts while on the shelf, possibly via spatiotemporal management of trawl fisheries (Skeljo 2023).

5. Management advice

Management advice is based on the harvest control rules (HCR) established for the Falkland Islands toothfish longline fishery (Farrugia and Winter 2018, 2019) (Appendix 4). The estimated SSB_{2022}/SSB_0 ratio of 0.467 was above the *upper target reference point* (0.45), i.e. in the *expansion range*; projections from the current model indicated that SSB/SSB_0 ratio will drop to the *target range* by 2026 and remain within the *target range* throughout the projection period. 2022 was the third consecutive year with $SSB_{current}/SSB_0$ ratio within the *expansion range*; however, since SSB/SSB_0 projections under the current TAC showed a decrease below 0.45 within ten years, no alteration of TAC was anticipated by HCR at this point.

The recommendation is to maintain toothfish annual TAC in the longline fishery at its current level of 1,040 tonnes.

6. References

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Appendix 1. CPUE standardization

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CPUE was standardized using Generalized Linear Mixed Models (GLMMs; Pinheiro and Bates 2000); GLMMs were fitted using package glmmTMB (Brooks *et al.* 2017, Magnusson *et al.* 2017) implemented in R version 4.1.3 (R Core Team, 2022). Before modelling, data were explored following the protocol described by Zuur *et al.* (2010); explanatory variables were inspected for outliers and collinearity using visual assessments, Pearson correlation coefficients (>0.5) and variance inflation factors (>3). Continuous explanatory variables were scaled by subtracting the mean and dividing by the standard deviation. Daily catch reports with zero toothfish catch were presumed to represent erroneous entries or broken sets and excluded from the analysis (1.3% of daily catch reports in the Spanish system and none in the umbrella system fishery).

The response variable was defined as daily toothfish CPUE expressed in kg-per-hook (Spanish system) or kg-per-umbrella (umbrella system) and modelled using Gamma distribution with a log link function. The explanatory variables considered in the model as either fixed or random effects are given in table A.1.

Table A.1. Explanatory variables considered in the CPUE standardization.

Explanatory variables		Variable type	Effect
Spanish-system	umbrella-system		
Year*	Year*	Categorical	Fixed
Month*	Month*	Categorical	Random
Area*	Area*	Categorical	Random
Depth	Depth	Continuous	Fixed
Soak-time	Soak-time*	Continuous	Fixed
Vessel*	-	Categorical	Random
-	Hooks-per-umbrella	Categorical	Fixed

* Variables included in the final model.

The *Year* effect is the quantity of interest and had to be included in the final model; 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 the R package MuMIn (Barton 2009). The *Month* and *Area* variables were treated as random effects, attempting to capture temporal or spatial dependency in CPUE; the spatial resolution was 1° Lon x 1° Lat. The *Depth* variable is the average fishing depth, and *Soak-time* is the sum of soak times of the lines belonging to a single response CPUE value (usually multiple lines were set by a given vessel on a given day). The *Vessel* variable was treated as a random effect in the Spanish-system standardization to account for dependence in CPUE values belonging to the same vessel due to e.g. vessel fishing power and skipper/crew skills and behaviour. The *Vessel* variable was excluded from the umbrella-system CPUE standardization because only two vessels appeared in the model (too few to treat it as a random effect) and never fished concurrently in a year, making the *Vessel* and *Year* effect indistinguishable. The umbrella system had an additional variable, the number of *Hooks-per-umbrella*, which progressively decreased from 10 hooks initially to 8 hooks in December 2007, to 7 hooks in March 2014, and to 6 hooks in June 2016).

The final GLMM fitted to the Spanish-system data included *Year* as a fixed effect and *Month*, *Region* and *Vessel* as random effects; the model explained 19.9% of the overall variation in CPUE. Standardized and unstandardized CPUE time series had similar declining trends (Figure A.1). The final GLMM fitted to the umbrella-system data included *Year* and *Soak-time* as fixed effects and *Month* and *Region* as random effects; the model explained 18.7% of the overall variation in CPUE. Standardized and unstandardized CPUE time series were similar and lacked a clear trend (Figure A.2).

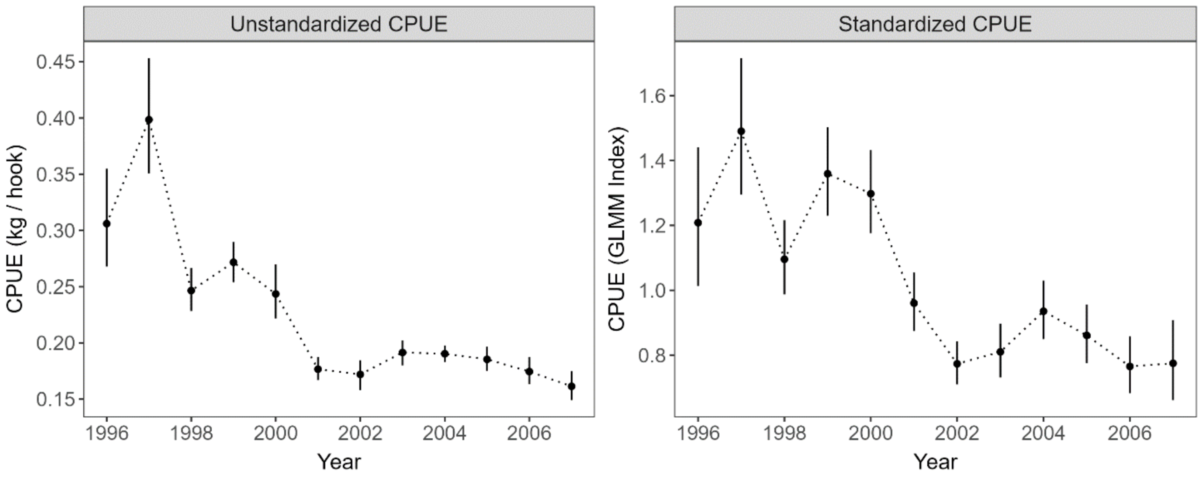

Figure A.1. Spanish-system longline fishery unstandardized and standardized CPUE time series; black vertical lines denote 95% confidence intervals.

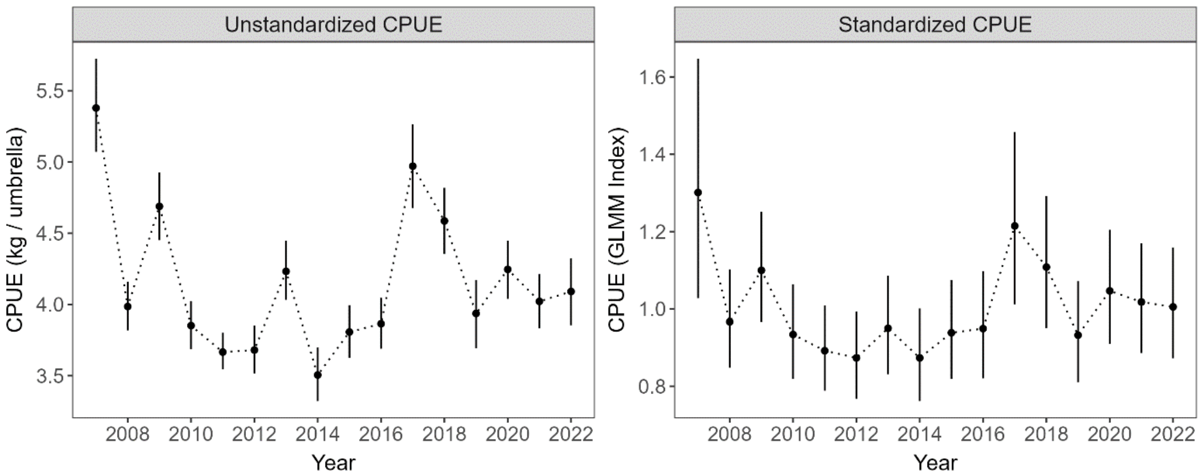

Figure A.2. Umbrella-system longline fishery unstandardized and standardized CPUE time series; black vertical lines denote 95% confidence intervals.

Appendix 2. Diagnostics plots

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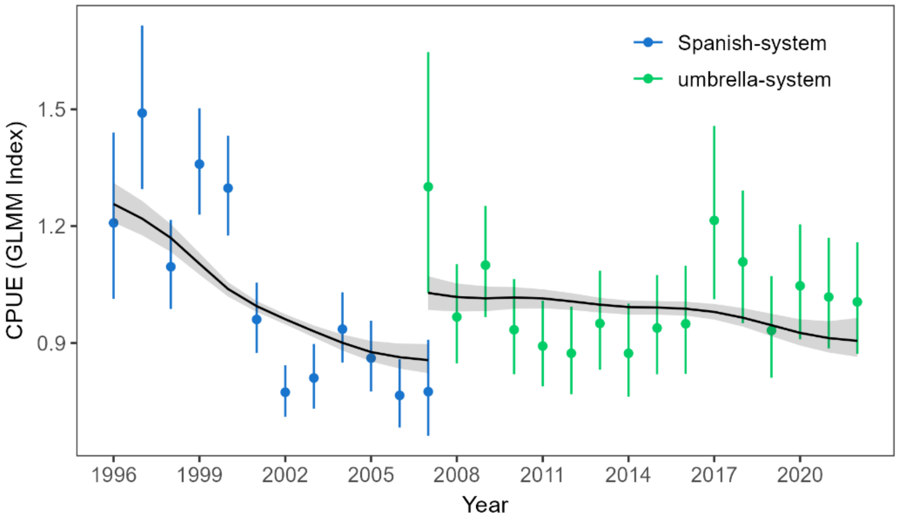


Figure A.3. MPD model fit (black line) to standardised CPUE indices for Spanish-system (blue dots) and umbrella-system longline (green dots); Vertical blue and green lines denote 95% confidence intervals of standardised CPUE indices; shaded areas denote MCMC 95% credible intervals of model fit.

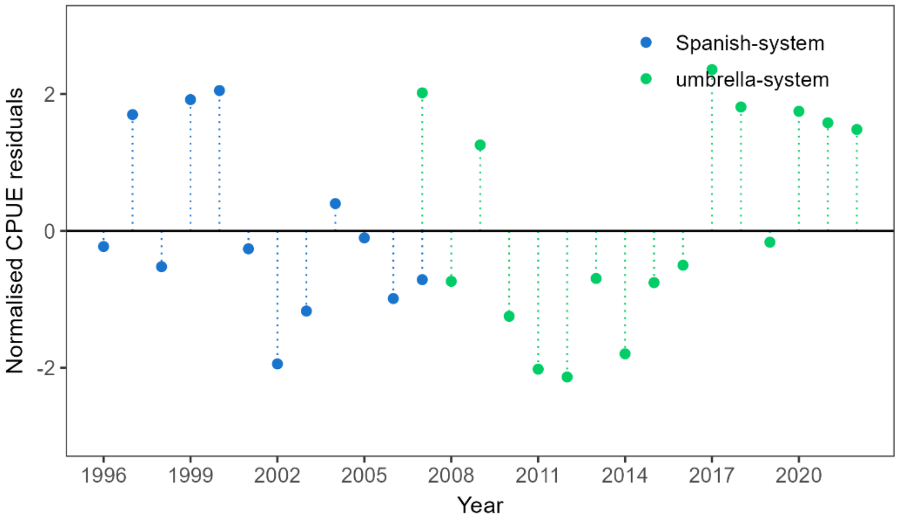


Figure A.4. Normalised residuals from model fit to standardized CPUE time series; for Spanish-system (blue) and umbrella-system longline (green).

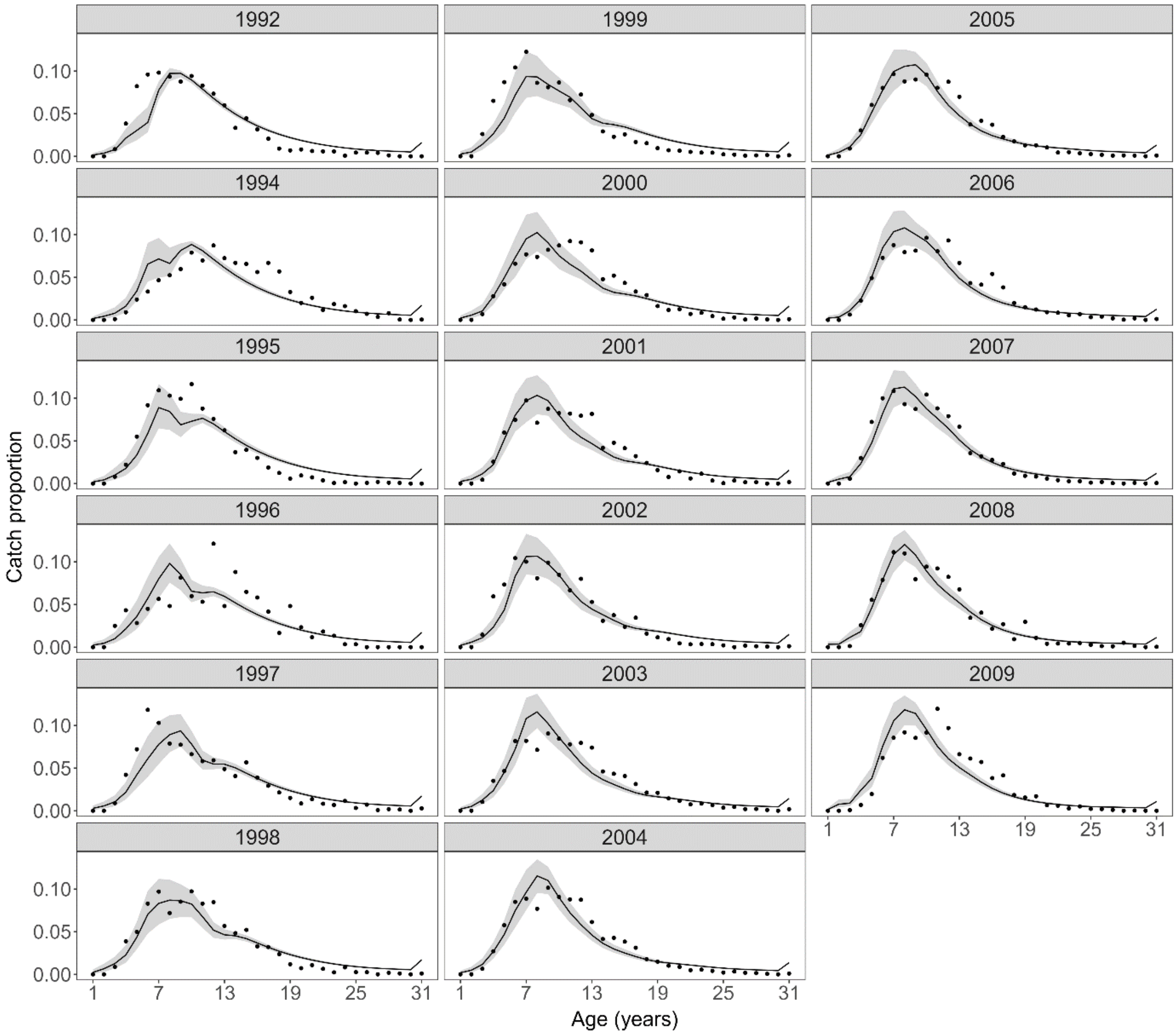


Figure A.5. MPD model fit (solid lines) to observed catch-at-age for Spanish-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

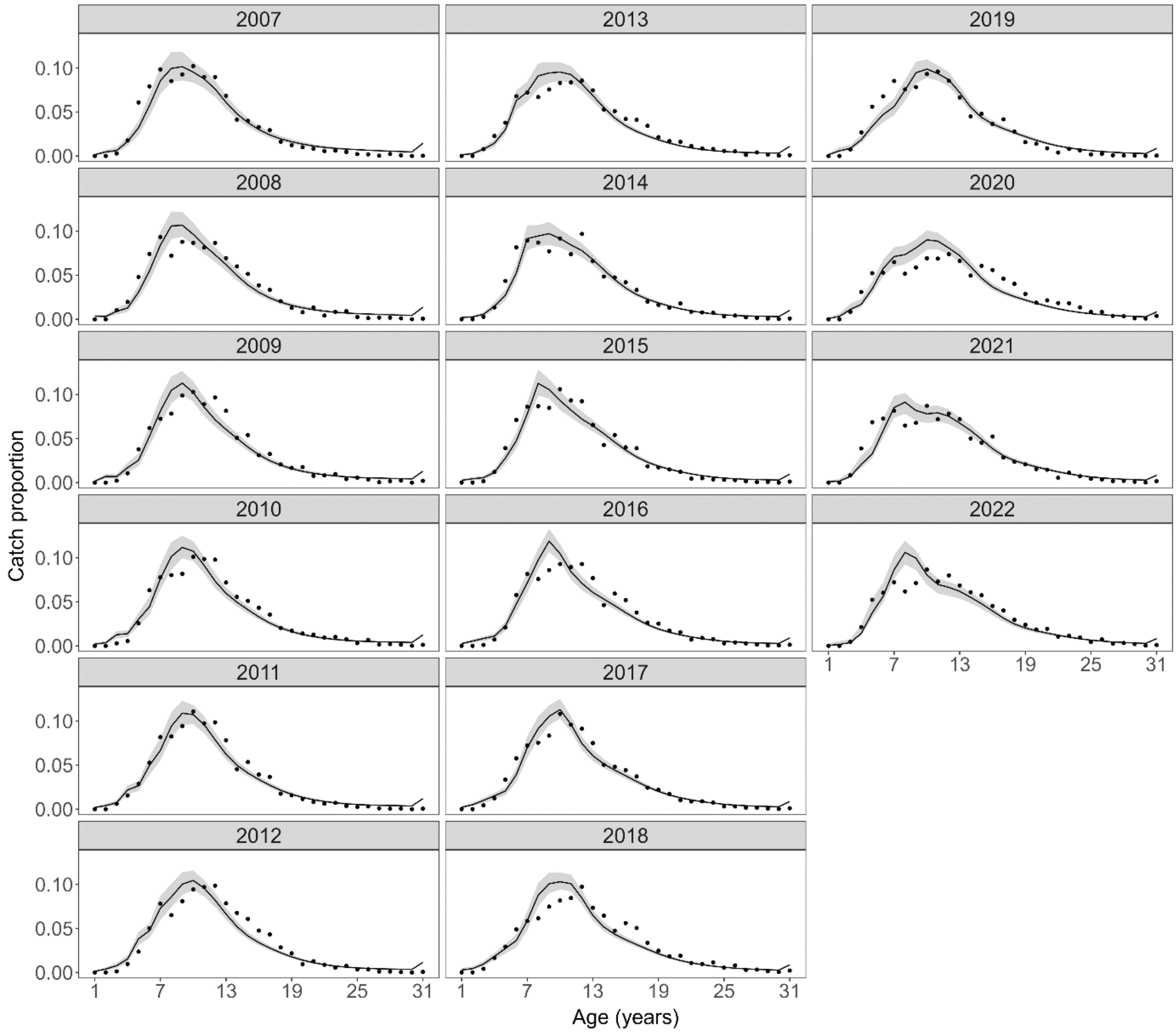


Figure A.6. MPD model fit (solid lines) to observed catch-at-age for umbrella-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

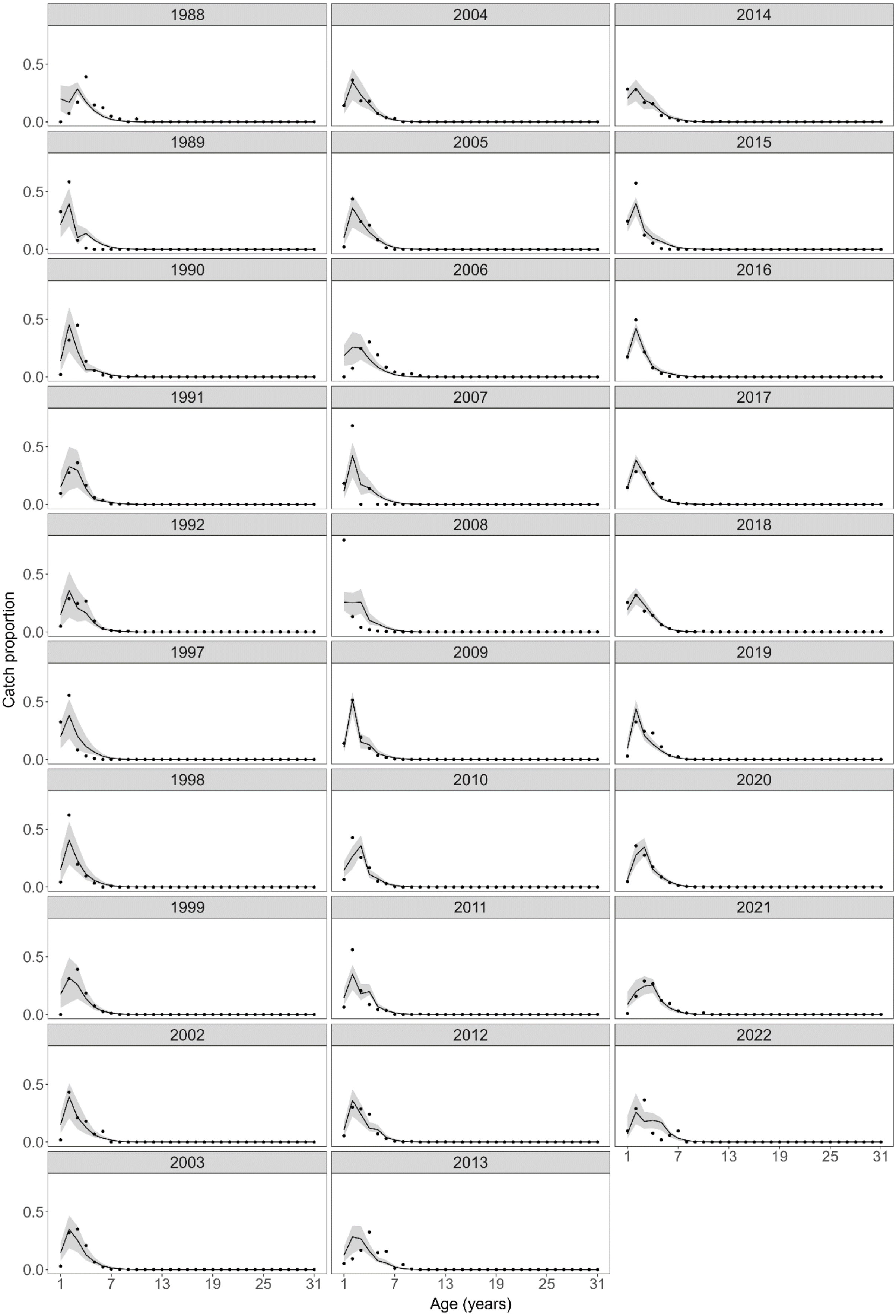


Figure A.7. MPD model fit (solid lines) to observed catch-at-age for finfish trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

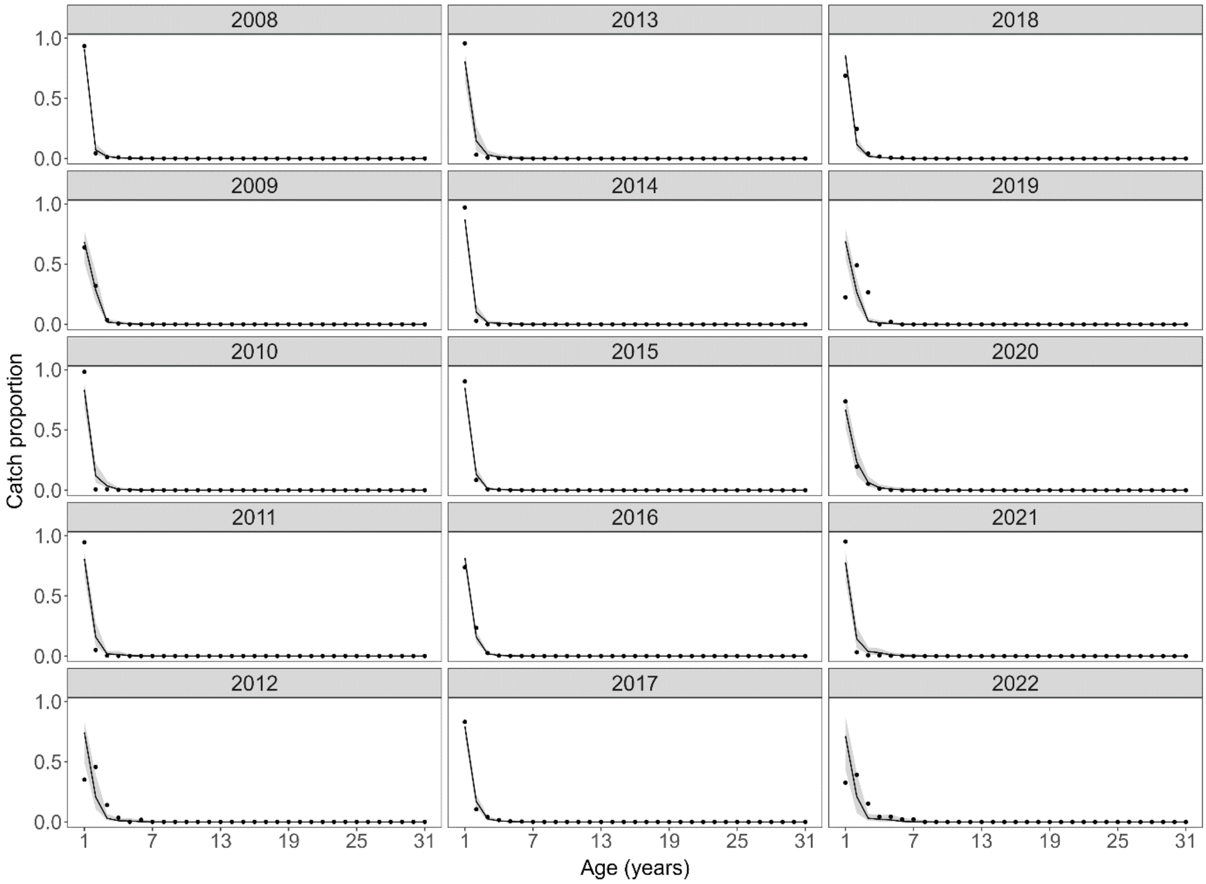


Figure A.8. MPD model fit (solid lines) to observed catch-at-age for calamari trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

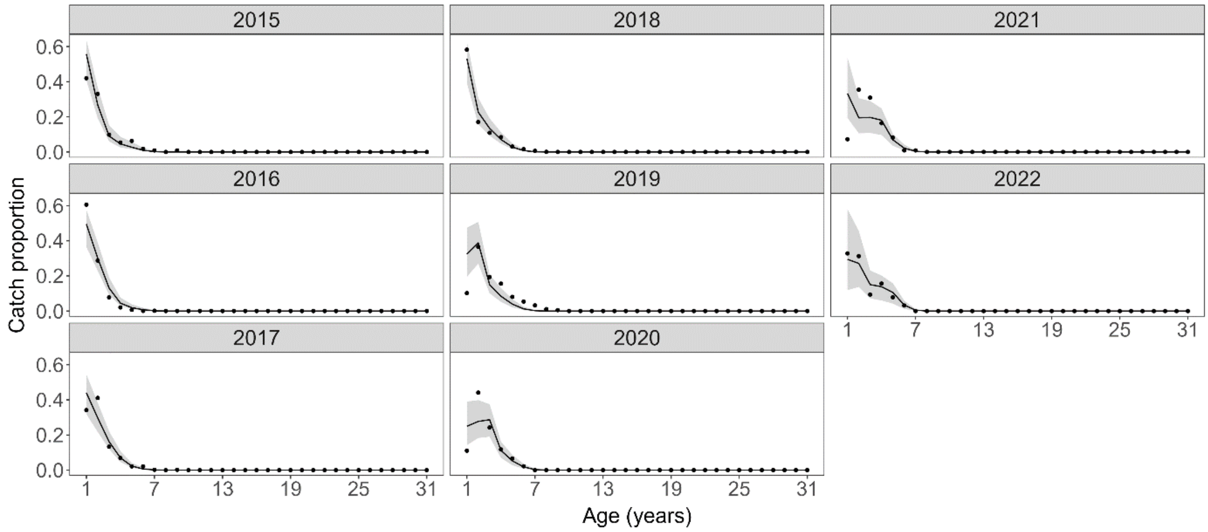


Figure A.9. MPD model fit (solid lines) to observed catch-at-age for groundfish survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.

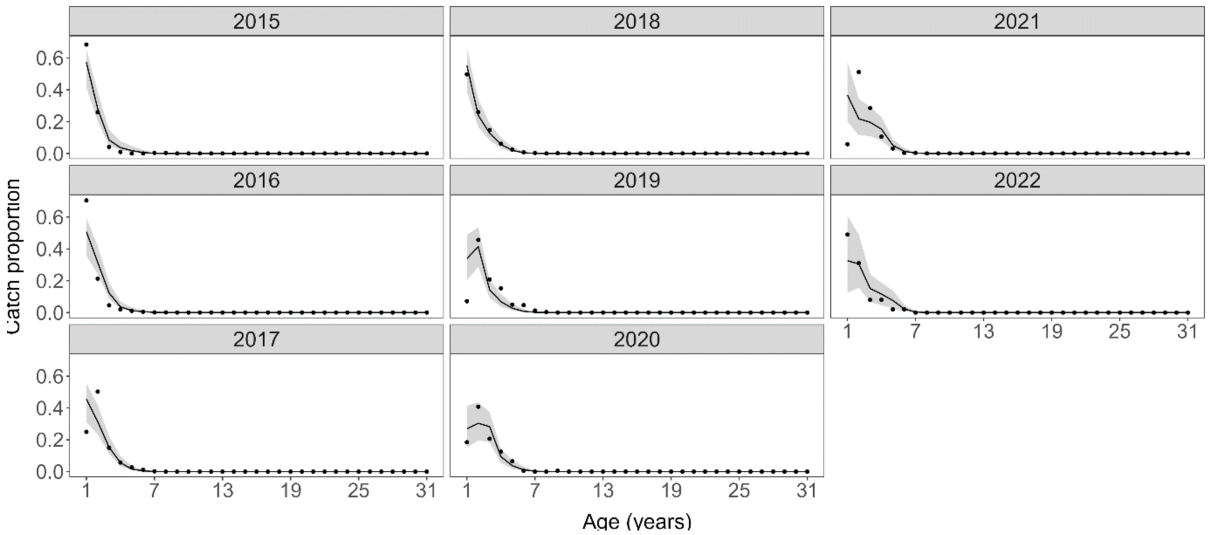


Figure A.10. MPD model fit (solid lines) to observed catch-at-age for calamari pre-season survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.

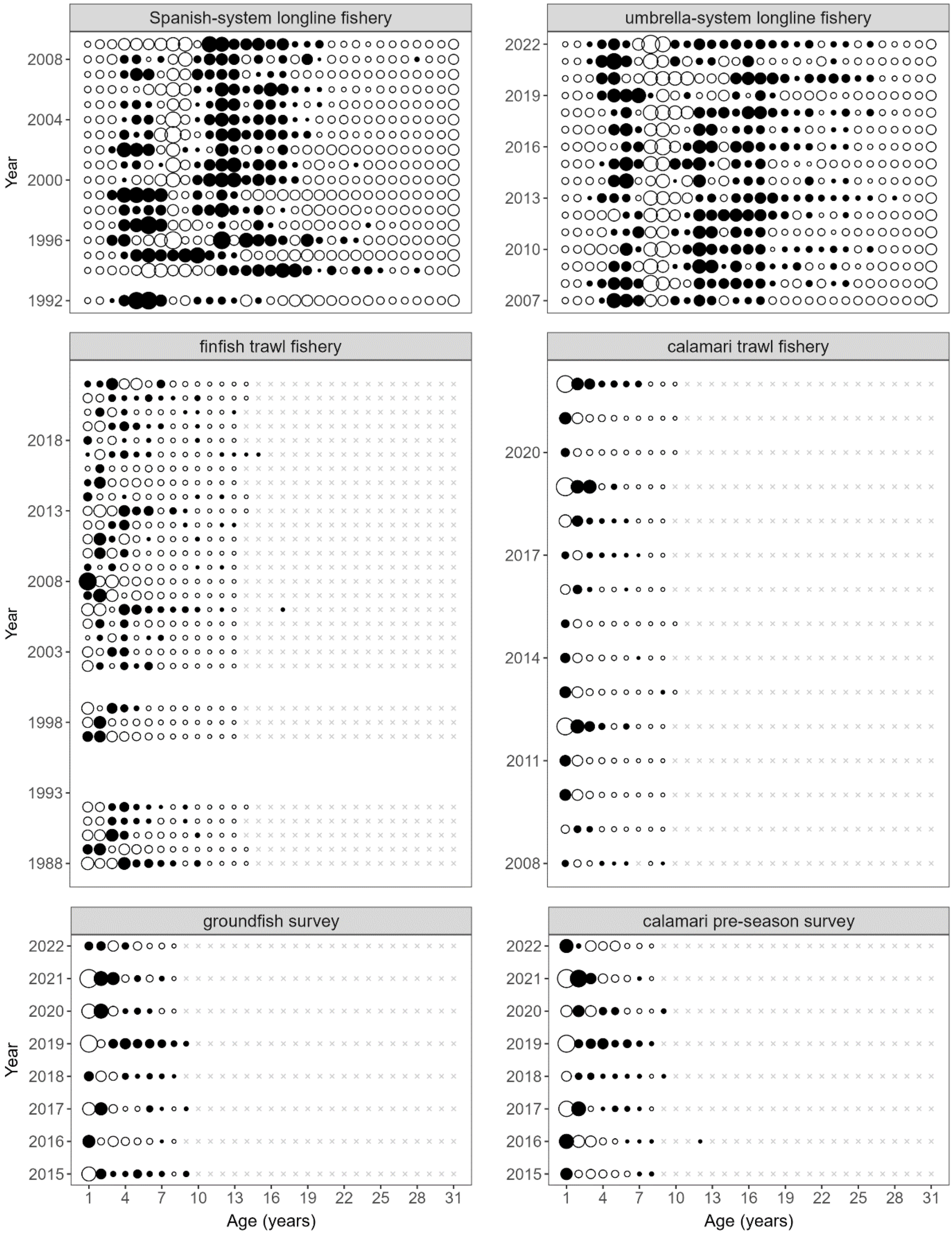


Figure A.11. Residuals from the model fit to observed catch-at-age for four fisheries and two research surveys. Bubble size is relative to the absolute residual value; positive residuals are denoted by full circles, and negative by empty circles.

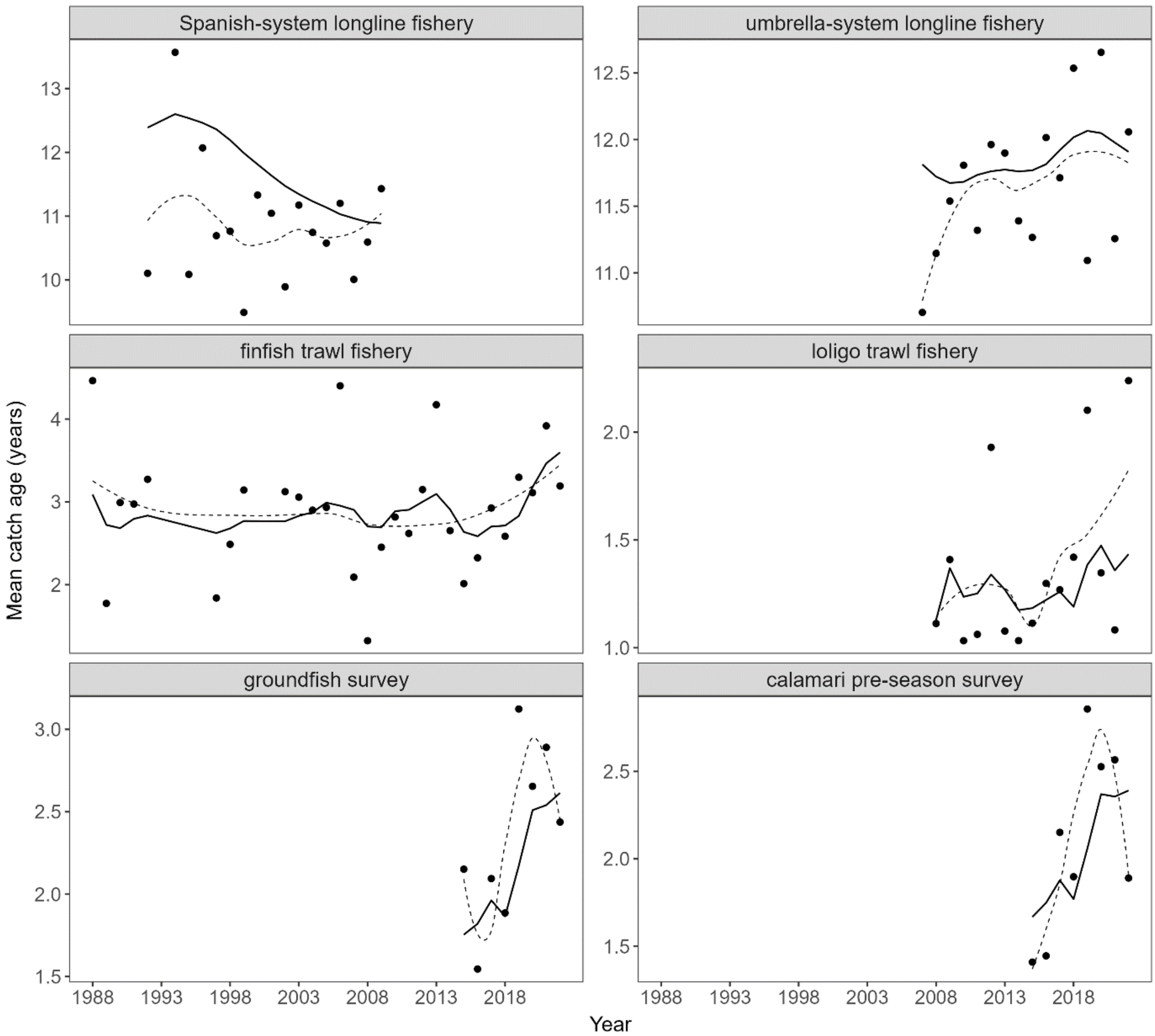
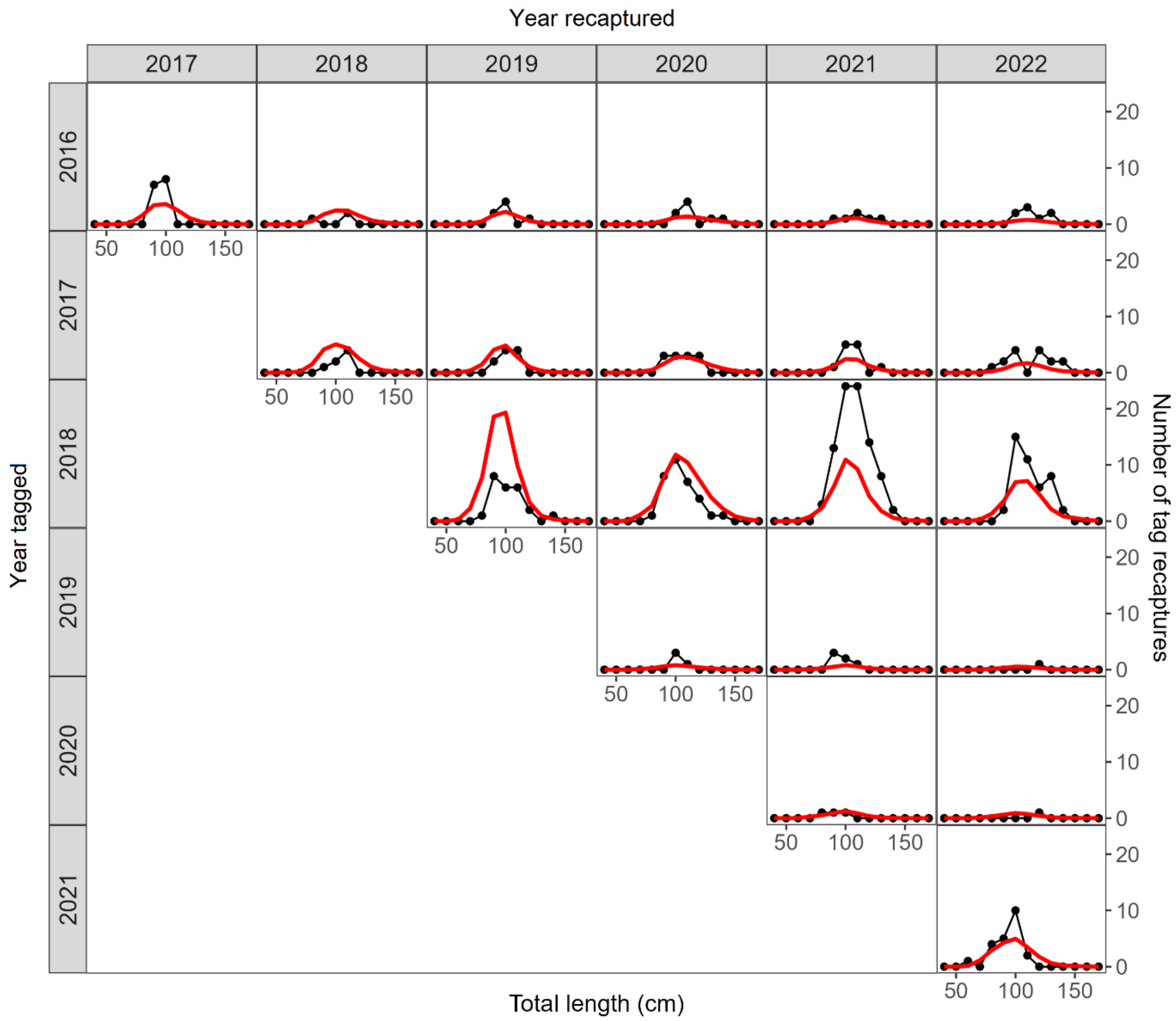
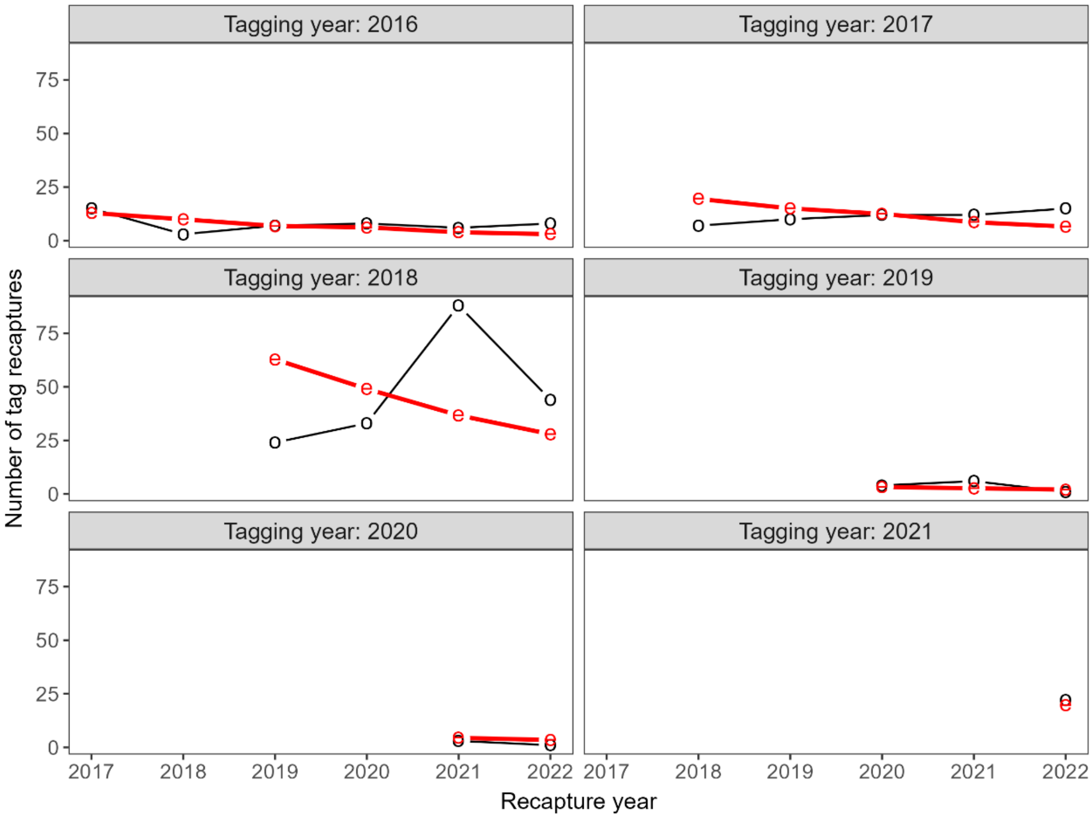


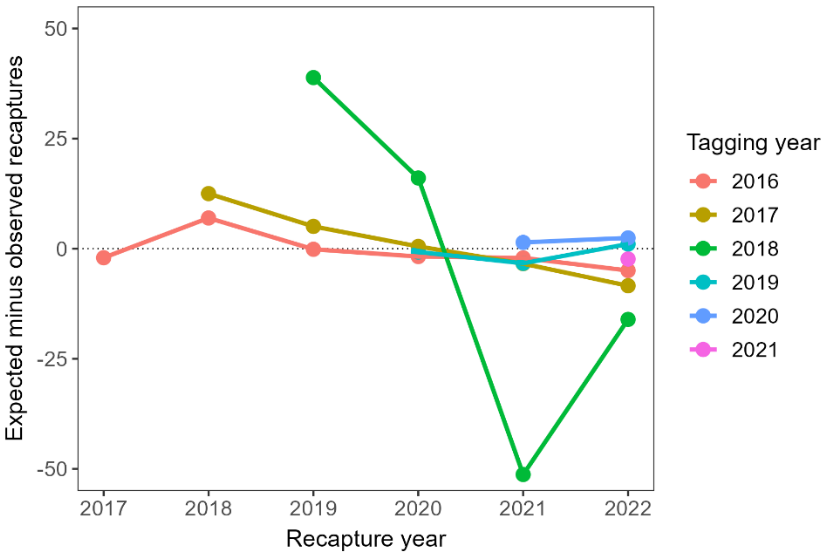
Figure A.12. Model fit (solid lines) to observed mean age-at-capture for four fisheries and two research surveys (black dots); dashed lines denote loess smoother for observations (span = 0.75).



A.13. Model fit (red lines) to observed tag recapture numbers by 10 cm length bins (black dots); for tag releases in 2016-2021 and tag recaptures in 2017-2022.



A.14. Model fit (red lines) to observed tag recapture numbers (black dots); for tag releases in 2016-2021 and tag recaptures in 2017-2022.



A.15. Differences between fitted and observed tag recapture numbers; for tag releases in 2016-2021 and tag recaptures in 2017-2022.

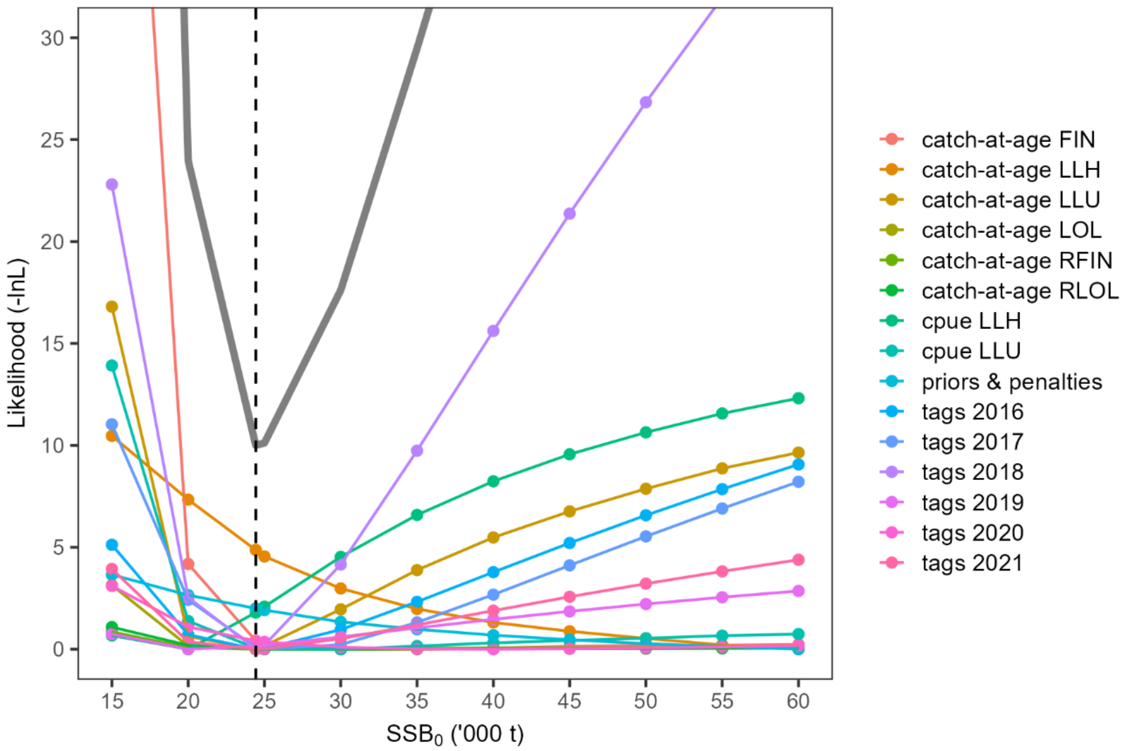


Figure A.16. Likelihood profiles for SSB_0. Negative log-likelihood values for individual datasets were rescaled to a minimum of zero, while the total objective function was rescaled to a minimum of 10 for easier visualisation (solid grey line). The dashed vertical line denotes the MPD estimate of SSB_0. LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-season survey.

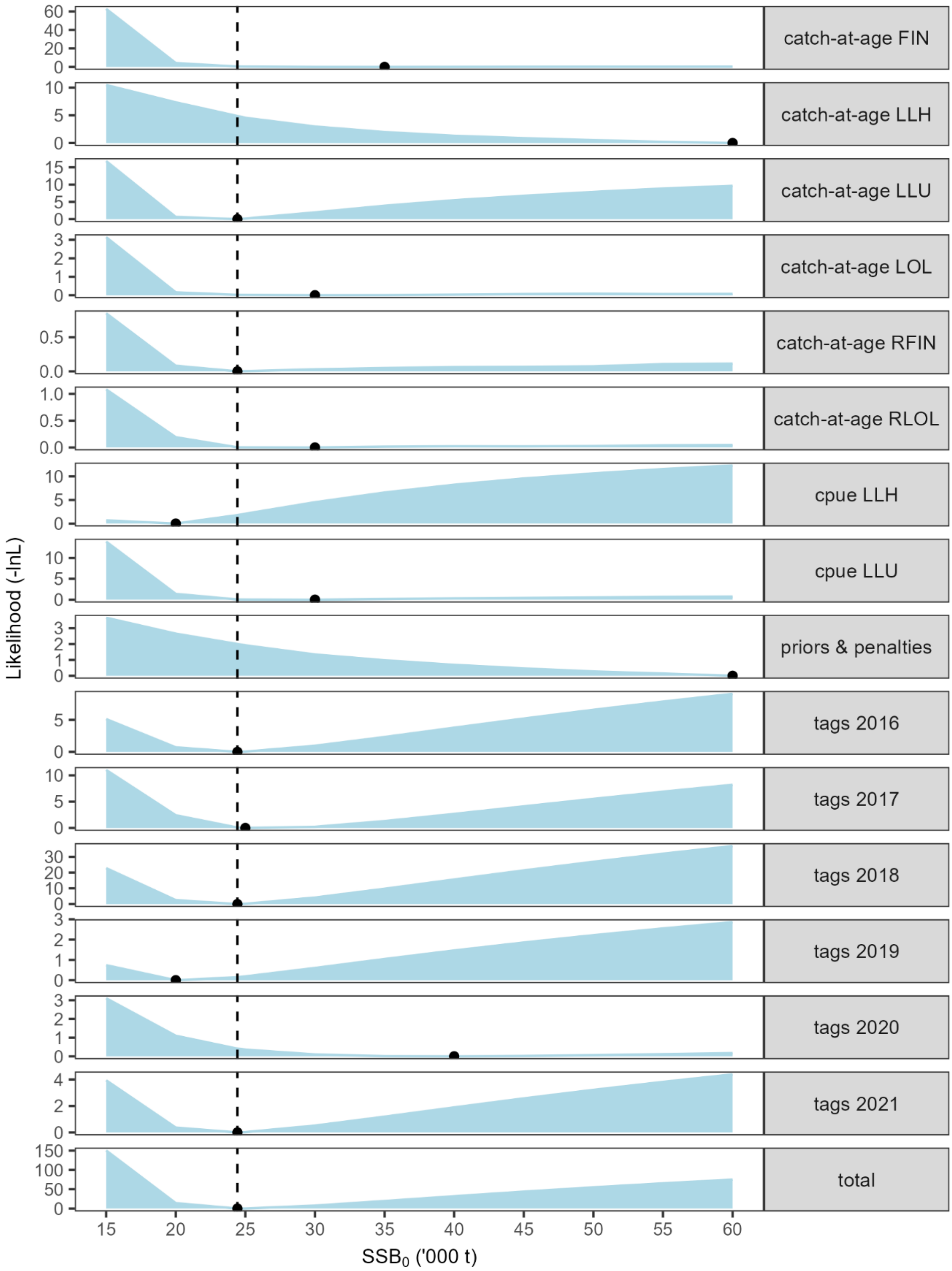


Figure A.17. Likelihood profiles for SSB_0. Negative log-likelihood values for individual datasets and the total objective function were rescaled to a minimum of zero. The dashed vertical line denotes the MPD estimate of SSB_0; dots denote SSB_0 values with the minimum negative log-likelihood value for each dataset. LLH - Spanish-system longline, LLU - umbrella-system longline, FIN - finfish trawl, LOL - calamari trawl, RFIN - groundfish survey, RLOL - calamari pre-season survey.

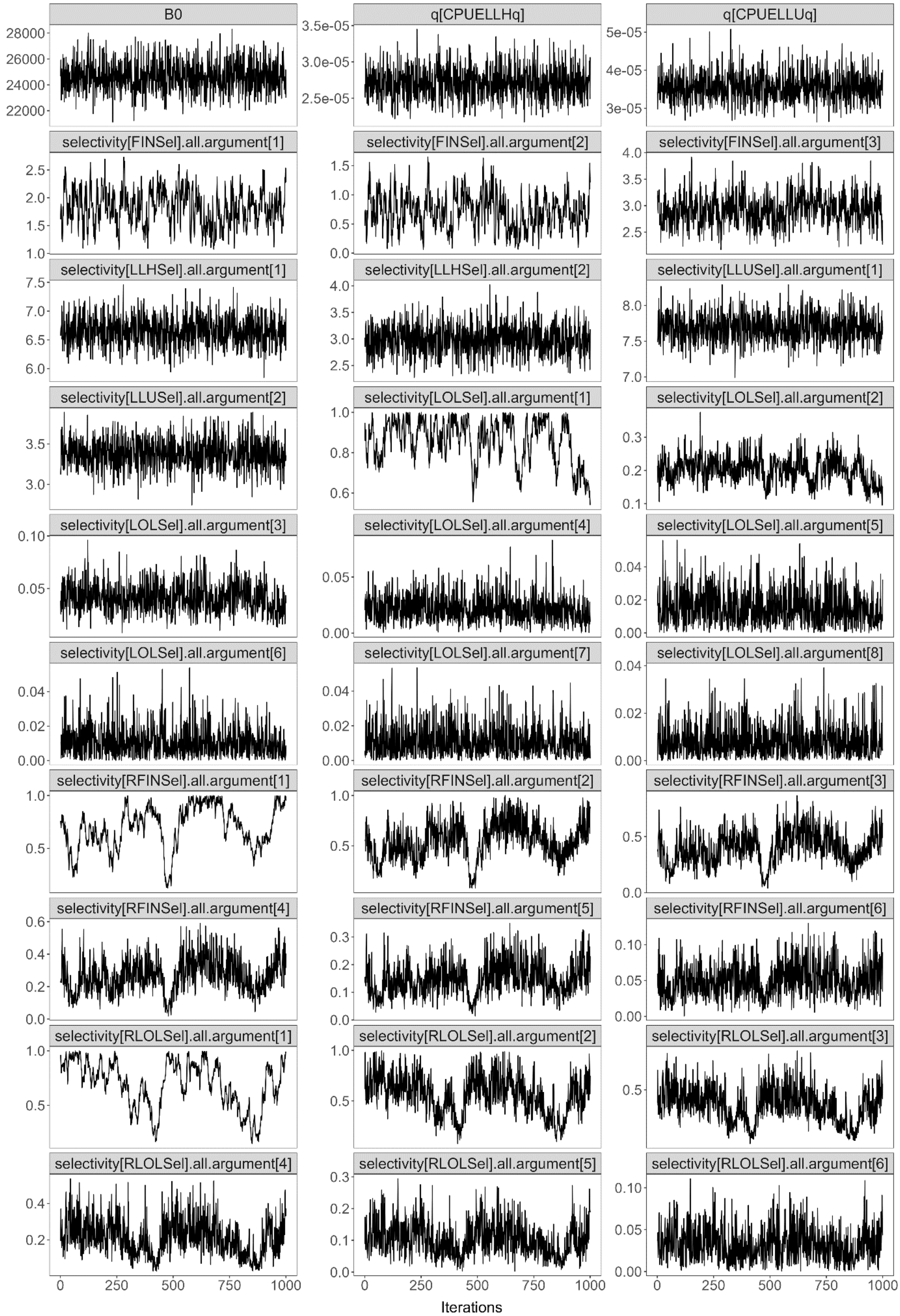


Figure A.18. MCMC posterior trace plots for all estimated parameters (figure 1 of 3).

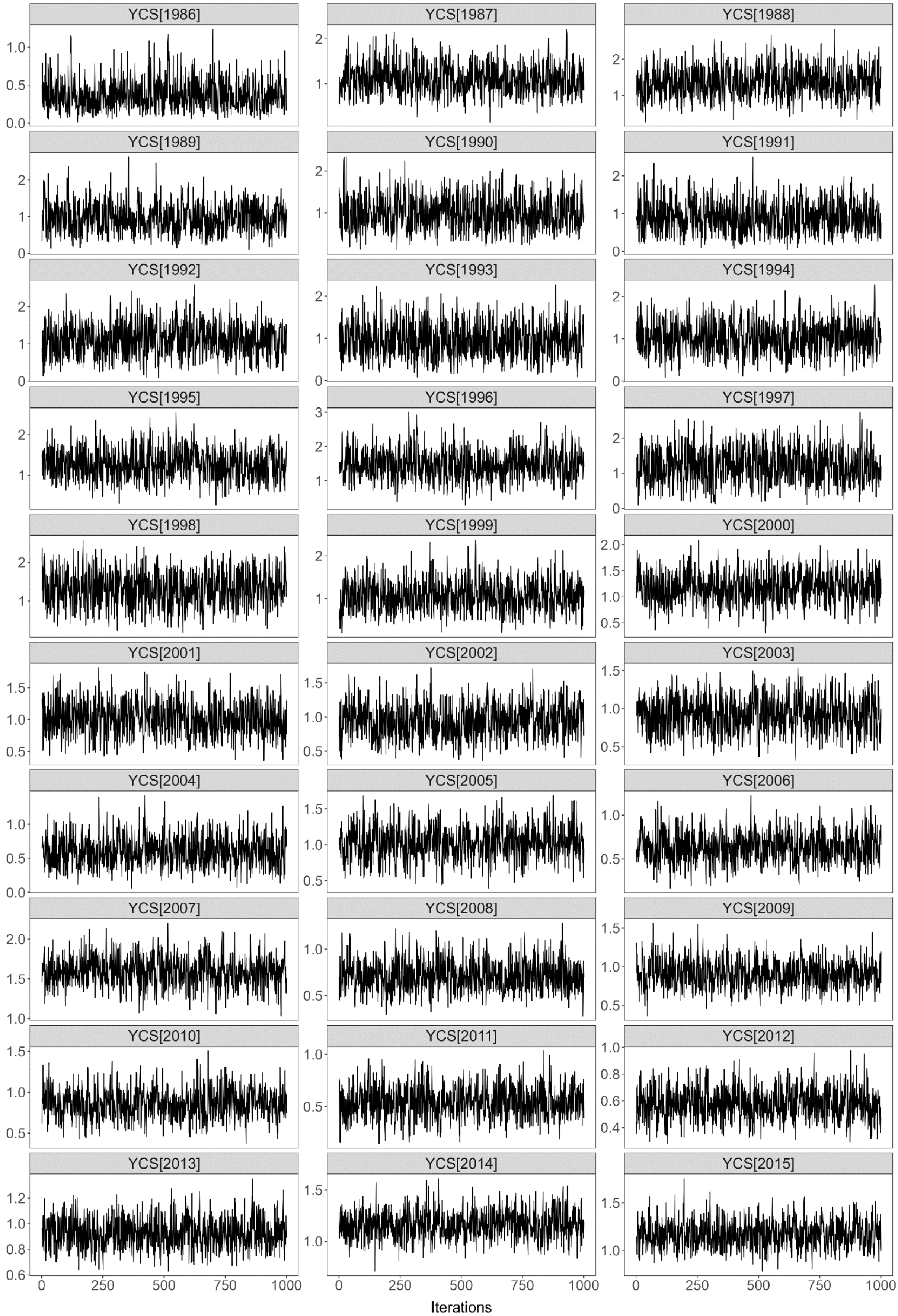


Figure A.18. Continued (figure 2 of 3).

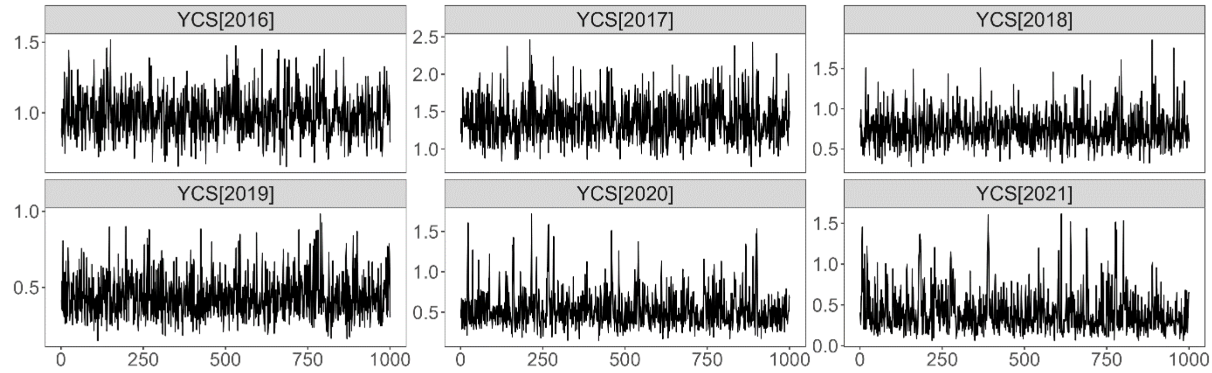


Figure A.18. Continued (figure 3 of 3).

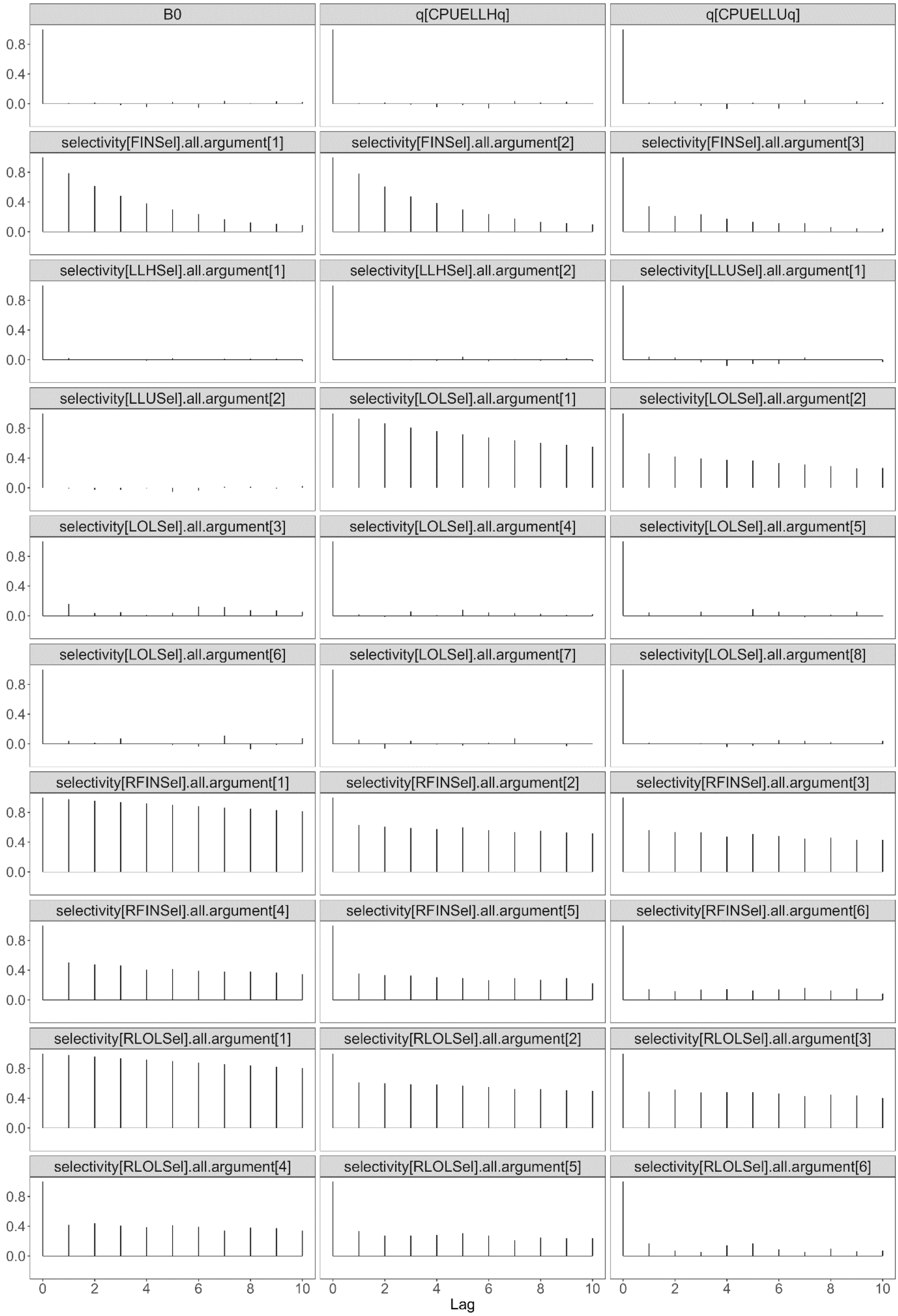


Figure A.19. MCMC autocorrelation lag plots for all estimated parameters (figure 1 of 3).

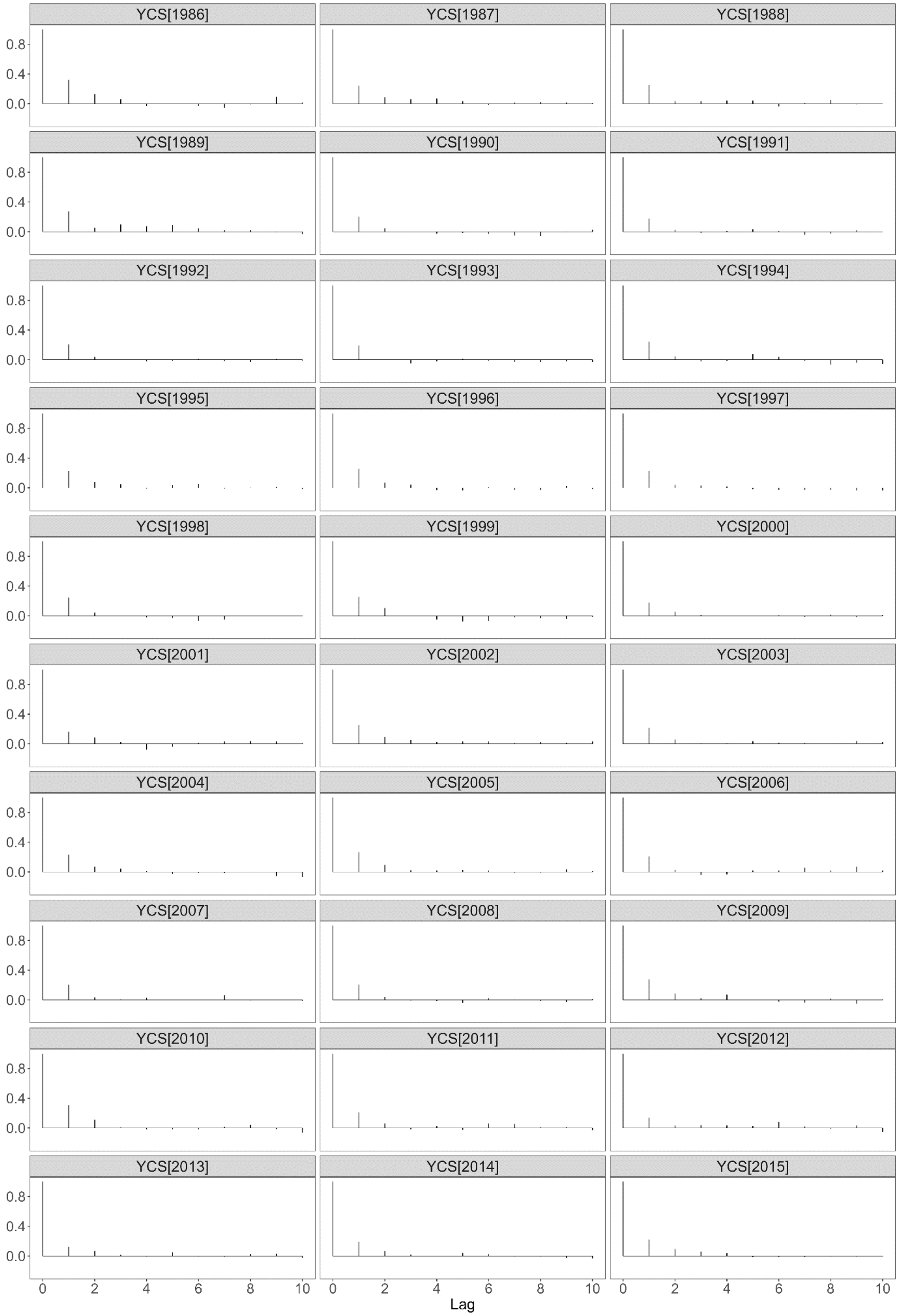


Figure A.19. Continued (figure 2 of 3).

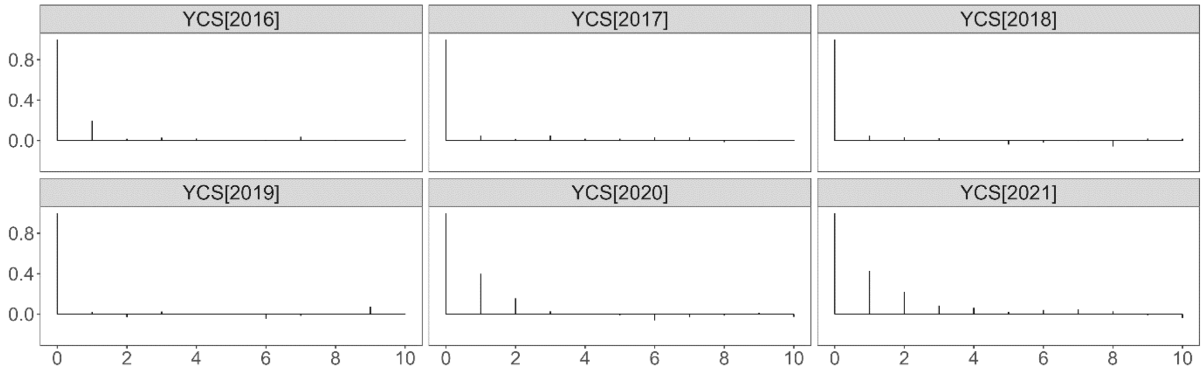


Figure A.19. Continued (figure 3 of 3).

Appendix 3. Objective function contributions

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Table A.2. Contributions to the objective function in the MPD model run.

Objective function components	Values
Observations	
CPUE $_{LLH}$	-16.9
CPUE $_{LLU}$	-23.1
Catch-at-age $_{FIN}$	145.0
Catch-at-age $_{LLH}$	446.6
Catch-at-age $_{LLU}$	696.0
Catch-at-age $_{LOL}$	46.4
Catch-at-age $_{RFIN}$	41.6
Catch-at-age $_{RLOL}$	32.8
Tags 2016	20.0
Tags 2017	21.1
Tags 2018	40.8
Tags 2019	6.2
Tags 2020	3.3
Tags 2021	5.4
Priors	
SSB_0	10.1
YCS	-24.3
q $_{LLH}$	-10.5
q $_{LLU}$	-10.2
All selectivity priors	0.0
Penalties	
YCS $_{MEAN\_1}$	7.0
Selectivity ogive smoother $_{LOL}$	0.6
Selectivity ogive smoother $_{RFIN}$	0.3
Selectivity ogive smoother $_{RLOL}$	0.1
All catch limit penalties	0.0
Total objective function	1438.2

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

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 the industry.

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