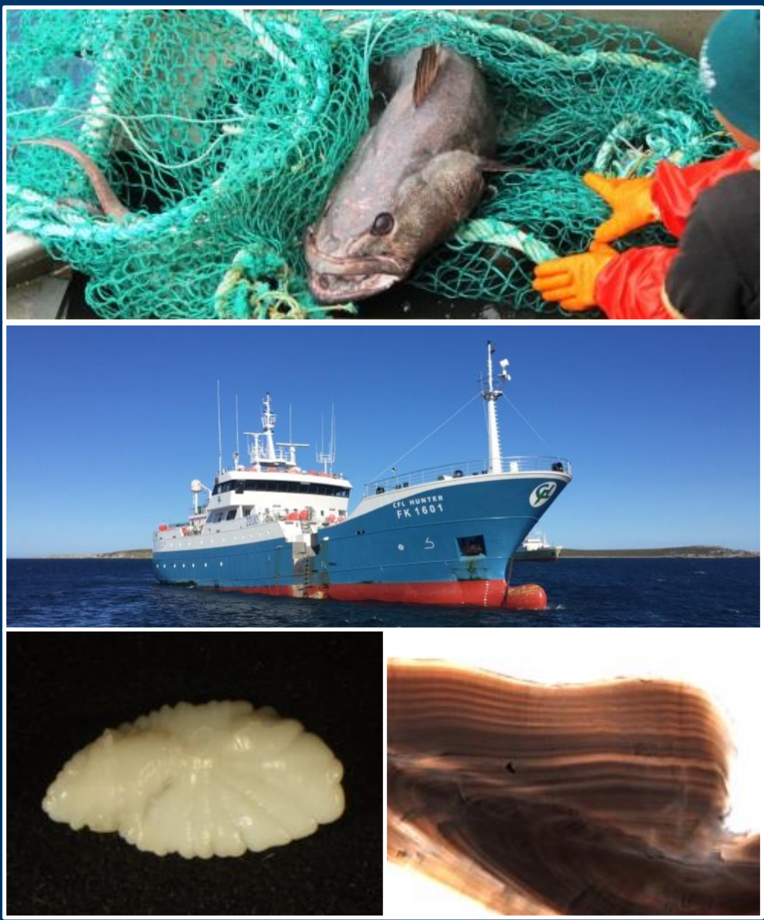
Stock assessment of Patagonian toothfish (*Dissostichus eleginoides*) in the Falkland Islands to 2024



Skeljo F

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

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SA – 2024 – TOO

Participating Scientific Staff

Frane Skeljo (PhD, Senior Stock Assessment Scientist)

Comments provided by: Andreas Winter (PhD, Head of Science)

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Director of Natural Resources

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Summary

1. This report provides an updated integrated stock assessment of Patagonian toothfish *Dissostichus eleginoides* in Falkland Islands waters, using data up to the end of 2024.
2. The initial spawning stock biomass SSB_0 was estimated at 24,813 tonnes, and the current spawning stock biomass SSB_{2024} at 12,361 tonnes.
3. The ratio of current and initial spawning stock biomass (SSB_{2024}/SSB_0) was estimated at 0.498. According to the established harvest control rules (HCR), the SSB_{2024}/SSB_0 ratio places the stock in the *expansion range*.
4. Projections from the current model indicated that the SSB/SSB_0 ratio will drop and remain in the *target range* from 2032 to 2040.
5. Based on HCR, the recommendation is to maintain the 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 ~1,000 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 of ~100 m. 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). Fishery used 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 the 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 by 2008.

Besides targeted longline fishery, toothfish are 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 integrated stock assessment for toothfish in Falkland Islands waters, using data up to the end of 2024.

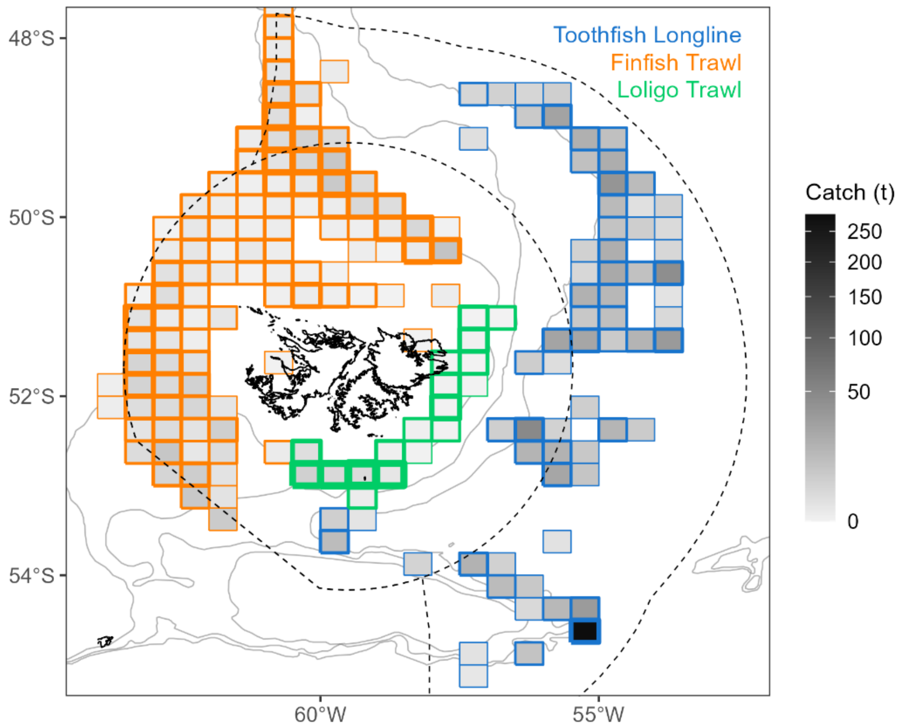

Figure 1. Distribution of toothfish catch and effort per fishery in 2024. The thickness of grid lines is proportional to vessel days. Grey-scale is proportional to toothfish catch biomass.

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 to the Falkland Islands Shelf arises from 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 occurs 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 an integrated age-structured 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 and Winter 2024), this assessment was routinely updated with (a) catch and effort data for the umbrella-system longline fishery in 2024, (b) catch data for the finfish and calamari trawl fisheries in 2024, (c) biological data (size-structure and maturity) for all fisheries and research surveys in 2024, (d) additional age readings for 2022 and 2023, and e) tag release data in 2023 and tag recapture data in 2024.

2.2. Data

Several datasets were used to inform the assessment, either as observations or input parameters (Figure 2). CPUE, catch-at-age and tag-recaptures are observations, i.e. appear in the objective function and are used to fit the model. Parameters of the length-at-age relationship, length-weight relationship and maturity curve are estimated outside the model and then treated as fixed parameters within the model.

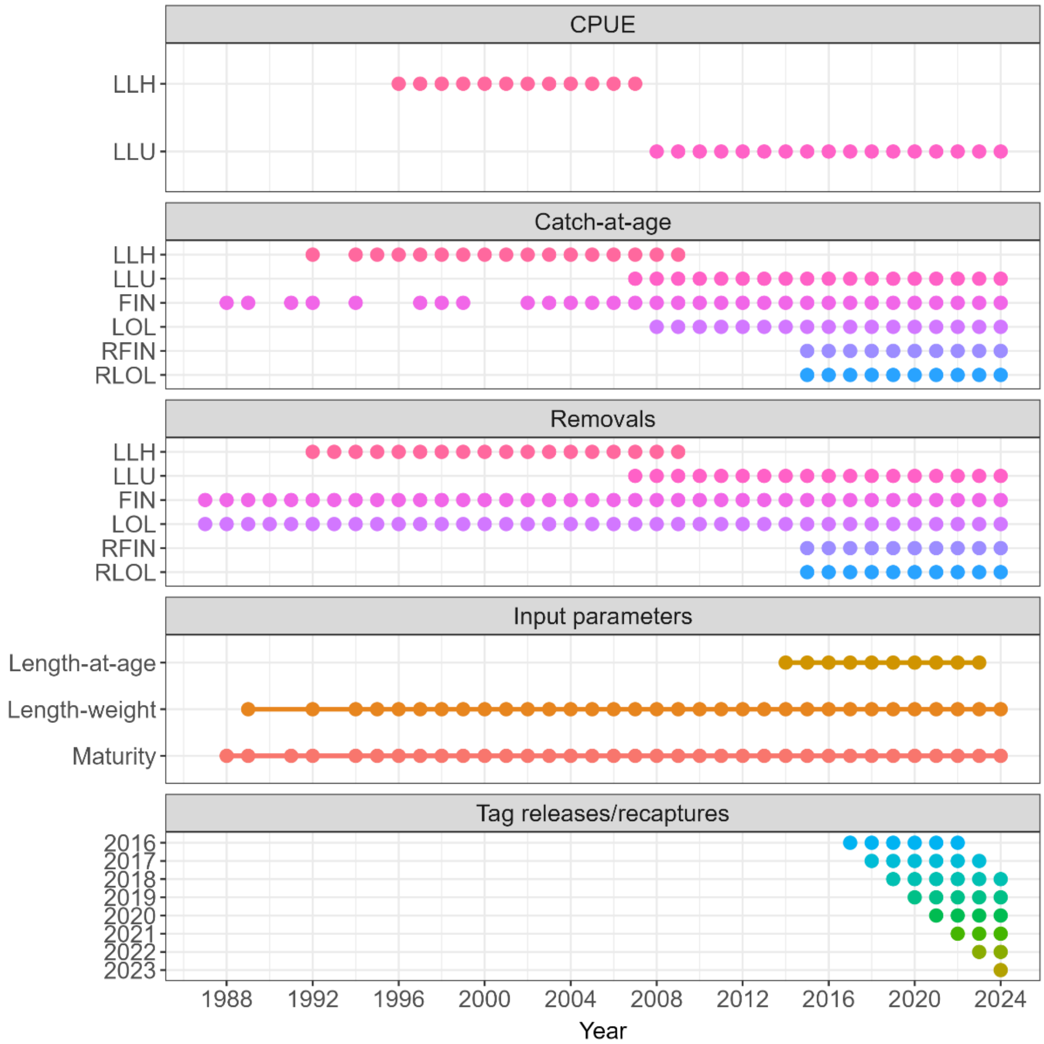

Figure 2. Data used to inform the assessment model. LLH – Spanish-system longline, LLU – umbrella-system longline, FIN – finfish trawl, LOL – calamari trawl, RFIN – groundfish survey, RLOL – calamari pre-season survey. Data used to calculate input parameters were pooled across years. Tag release years are on the y-axis, and tag recapture years are on the x-axis.

Catch-per-unit-effort (CPUE)

Although CPUE data were available for all four fisheries, only longline CPUE was used as a relative abundance index in the model; a decision motivated by the inconsistency of toothfish CPUE in trawl fisheries, where toothfish 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 was previously observed (Brown *et al.* 2010). The transition from the Spanish to the umbrella system occurred in 2007-2008.

For the Spanish-system longline fishery, the catch and effort data were available only aggregated per day (the fishery predates the use of electronic logbooks). 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.

For the umbrella-system longline fishery, the catch and effort data were available per line (electronic logbook data). 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 has participated in the Falkland Islands fishery for more than two years since 2007, resulting in inconsistent CPUE data. Moreover, at least one of the chartered vessels had restrictions imposed on its fishing practice (e.g. a limit on the number of fishing days in the 'best' fishing grounds) that were not in place for the Falkland Islands vessel. Because of the above, only the Falkland Islands' vessels' CPUE was 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 cleaned data as reported toothfish catch in kg-per-hook per day (Spanish system) or kg-per-umbrella per line (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 = 0.05 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 2014-2023; these were processed and aged at the Falkland Islands Fisheries Department (FIFD) and were considered the most reliable toothfish age estimates available (Lee 2014, 2015, 2016, 2017, 2018, 2019, 2020; Lee and Le Luherne 2023; Le Luherne 2025a, 2025b). Since the last assessment (Skeljo and Winter 2024), an additional 477 age readings belonging to otoliths collected in 2022 and 480 readings belonging to otoliths collected in 2023 became available; otoliths collected in 2024 are yet to be processed. In total, 3,726 toothfish age estimates belonging to longline fisheries and 3,490 age estimates belonging to trawl fisheries were used to construct two corresponding age-length keys: $ALK_{LONGLINE}$ and ALK_{TRAWL}. Next, the length-frequency distribution of 174,363 toothfish randomly sampled in 1988-2024 was split between the corresponding fisheries or surveys. Age was assigned to each fish by the conditional probability of the appropriate age-length key: $ALK_{LONGLINE}$ for longline fisheries and ALK_{TRAWL} for calamari fishery, finfish fishery, and trawl 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), aggregated per year and fishery, and converted to proportions-at-age. CAA data for the calamari trawl fishery up to 2008 were considered unrepresentative and thus excluded from the analysis (Skeljo and Winter 2021). Ageing error was applied to the observed CAA by assuming that true ages are normally distributed around read ages with CV = 0.1, and accounted for in the model via a misclassification matrix. The CAA data were assumed to be 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 given by the number of unique hauls sampled. 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 according to *Method TA1.8* (Francis 2011):

$$w_f = \frac{1}{var\left[(\bar{O}_{fy} - \bar{E}_{fy})/\sqrt{(v_{fy}/N_{fy})}\right]}$$

Where w_f is the weighting factor for fishery (or survey) f, $\bar{O}_{fy}$ and $\bar{E}_{fy}$ are calculated from the observed and expected proportions-at-age for fishery f in year y, v_{fy} is the variance of the expected age distribution for fishery f in year y, and N_{fy} is the effective sample size for fishery f in year y defined in stage 1. The model was then re-run with the adjusted effective sample sizes (Table 2).

Removals

Removals accounted for the reported catches in Falkland Islands waters, IUU catches in 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 Falkland Islands waters was available; therefore, we utilised the data for the Antarctic region (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 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 (Figure 3). The removals are treated as input parameters in CASAL and are assumed to be known without error.

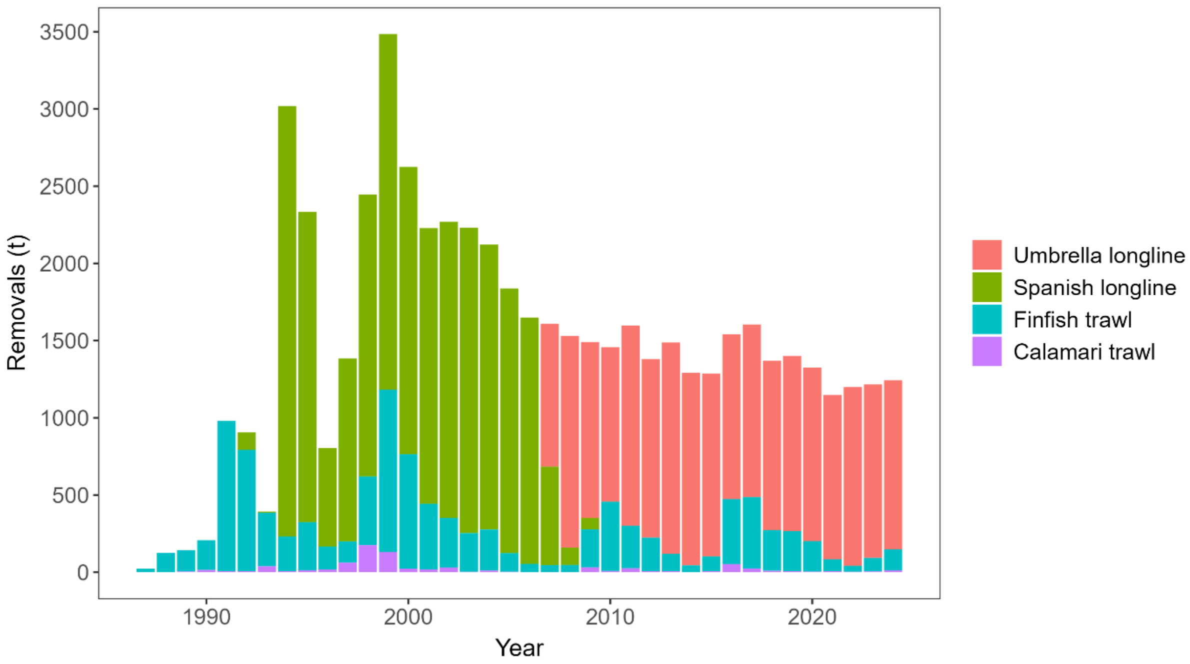

Figure 3. Toothfish removals in FCZ from 1987 to present, per fishery. Removals in trawl fisheries equal the reported catches. Removals in longline fisheries equal the sum of reported catches, assumed IUU catches and estimated depredation.

Length-weight relationship

The length-weight relationship was calculated as $W = aL^b$ based on the length and weight measurements of 45,004 toothfish sampled randomly by the observers from commercial catches in 1989-2024. Length-weight parameters are given in Table 3.

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 7,216 toothfish sampled randomly by observers from commercial catches in 2014-2023. Von Bertalanffy growth model parameters are given in Table 3.

Natural mortality

Natural mortality (M) was assumed to be 0.165 (Payne *et al.* 2005) and time- and age-invariant. Even though M would ideally be estimated within the model, there is 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 2023; Readdy and Earl 2023; Masere and Ziegler 2023; Massiot-Granier *et al.* 2023a, 2023b; Mormede *et al.* 2023). 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 174,363 toothfish sampled randomly from commercial catches during 1988-2024. 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, which are considered immature. Analysis of the maturity data strongly indicated that some older fish were erroneously assigned immature (stage 2) when observed. To address this inaccuracy, a generalised 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 3.

<u>*Tag releases and recaptures*</u>

The tagging programme for Falkland Islands toothfish commenced in 2016, aiming to improve understanding of toothfish movement patterns within the region. The initial goal of tagging 3,000 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 on board the longliner. However, the tagging programme largely relied 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 was recommended (Lee and Skeljo 2020) and followed up by renewed tagging efforts since 2021 (Skeljo and Pearman 2021, Nicholls and Raczyński 2023, Le Luherne and Peruzzo 2023, Le Luherne and Desmet 2025) with a goal of tagging ~1,000 longline-caught fish annually, i.e. one fish per tonne of TAC. Since 2016, 8,012 toothfish have been tagged and released within FCZ, and 552 of these have been recaptured (Table 1). A further 536 toothfish were tagged on the High Seas (North Scotia Ridge) in 2023 to improve our understanding of toothfish stock connectivity within the region (Raczynski *et al.* 2024). Under the current assumption of a discrete toothfish stock in Falkland Islands waters (Lee 2023), the recaptures of fish tagged on the High Seas were not used in the assessment.

Table 1. Numbers of tags released within FCZ and recaptured within and outside (in brackets) FCZ. Grey shading denotes the numbers used in the assessment (i.e. within-year recaptures and recaptures after six years at liberty were excluded). Recaptures outside FCZ were used to calculate the tag emigration rate, but not to fit the assessment model. Tags deployed on the High Seas and their subsequent recaptures were excluded from the assessment and the table.

Releases		**Recaptures**									
Year	**Number**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**2023**	**2024**	**Total**
2016	437	14	15	3	7	8	6	8	1	4 (1)	66 (1)
2017	685	-	6	7 (3)	10 (1)	12 (2)	12 (1)	15 (1)	7	3	72 (8)
2018	2189	-	-	4 (1)	24 (2)	32 (7)	89 (1)	44 (6)	12 (6)	22 (1)	227 (24)
2019	127	-	-	-	1 (1)	4	6	1	4	1	17 (1)
2020	171	-	-	-	-	0	3 (2)	1	3	0	7 (2)
2021	866	-	-	-	-	-	13	22	7 (2)	12 (1)	54 (3)
2022	1072	-	-	-	-	-	-	5	14 (1)	8 (3)	27 (4)
2023	1146*	-	-	-	-	-	-	-	4	31 (2)	35 (2)
2024	1319									2	2
Total	8012*	14	21	14 (4)	42 (4)	56 (9)	129 (4)	96 (7)	52 (9)	83 (8)	507 (45)

* An additional 536 toothfish were tagged on the High Seas in 2023.

The spatial distribution of toothfish tag releases, recaptures, and longline fishing effort is given in Figure 4; it shows that (a) the spatial extent of longline fishing effort remained consistent between 2016 and 2024, (b) tag releases were well spread across the fishing area in high-release years (2017, 2018, 2021, 2022, 2023) compared to a more localised spread in low-release years (2019, 2020), and (c) spatial overlap between tag releases and their subsequent recaptures was high.

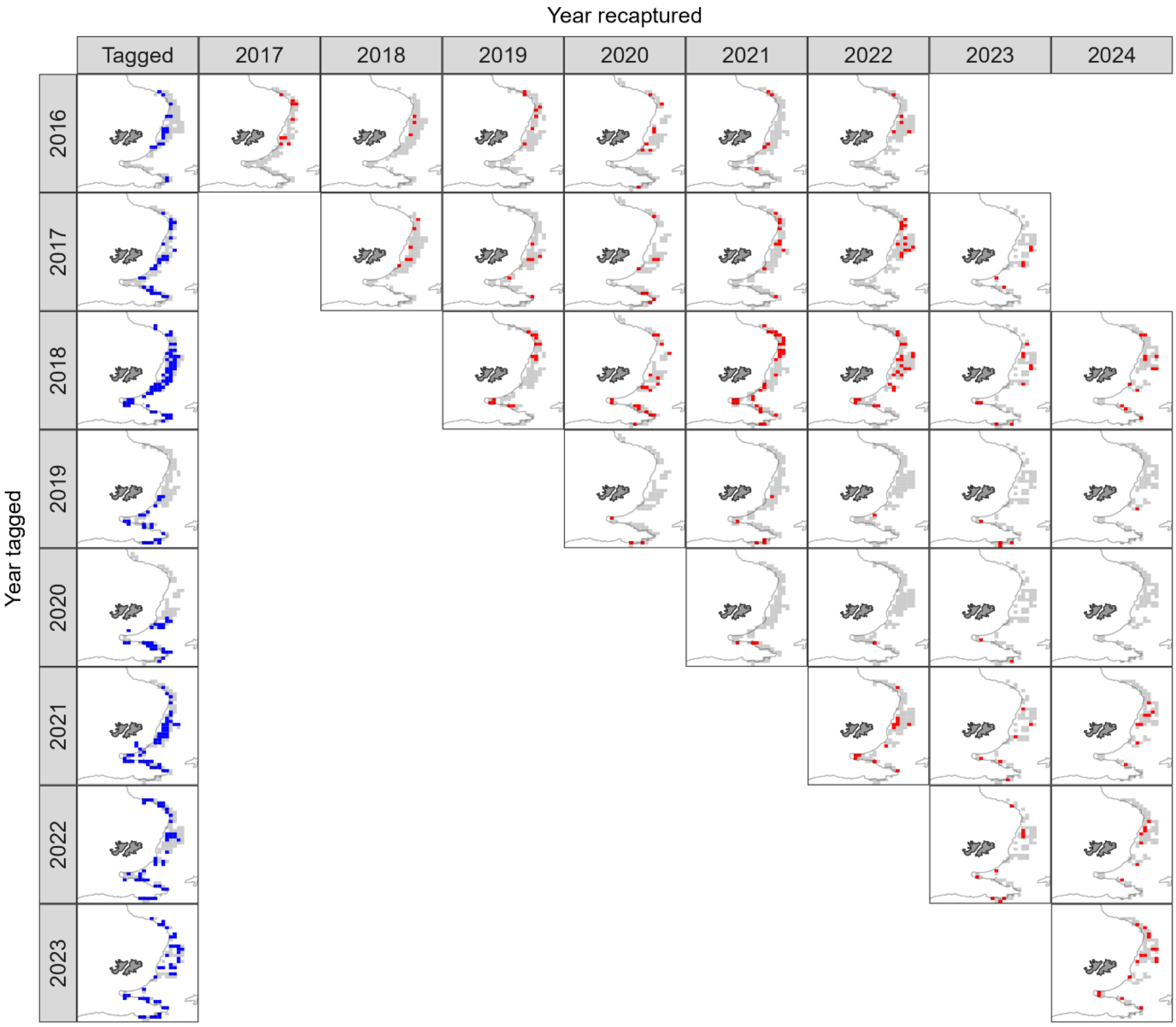

Figure 4. Spatial distributions of toothfish tag releases in FCZ in 2016-2023 (blue) and recaptures in FCZ in 2017-2024 (red); grey shading denotes the distribution of longline fishing effort for each year. Recaptures after six years at liberty were excluded from the model and the plot. Cells correspond to locations where at least one tagged fish was released (blue) or recaptured (red) or where at least one longline was hauled (grey).

The current assessment used the data on longline-caught toothfish tag releases within FCZ in 2016-2023 and their subsequent recaptures by longline in 2017-2024 (Table 1). 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 a 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 and 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 and processed.

The first 6,500 tagged toothfish (2016-2023) were double-tagged, with a large and a small dart tag. By 2024, 51 double-tagged fish were recaptured with only large tags and 20 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 estimated at λ_1 = 0.0167 and of small tags λ_2 = 0.0340. 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 = 0.0021. However, the approximation proved adequate only when the available recapture data were restricted to fish recaptured after ≤6 years at liberty (for more details, see Dunn *et al.* 2011). Since our tagging programme is recent, the only recaptures after >6 years at liberty occurred in 2023 for fish tagged in 2016 and in 2024 for fish tagged in 2016 and 2017 (8 recaptures inside, and 1 outside FCZ; see Table 1); these data were excluded from the model to allow for the above-mentioned approximation and do not appear in corresponding plots throughout the report.

Emigration of tagged fish from the assessed area can violate the assumptions of tag recapture models used in the assessment and result in overestimated biomass and stock status. The emigration rate of toothfish from Falkland Islands waters was estimated based on the numbers of recaptures inside and outside the FCZ of the fish tagged within 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 the FCZ; given the lack of reliable information on overall harvest rates outside the FCZ, they were assumed 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 = 0.0438; 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).

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; they are more variable than 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 all 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 by ϕ (Table 2).

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

Table 2. Weighting factors (w), used to adjust the effective sample sizes of catch-at-age data, and the over-dispersion parameter (ϕ), used to weight the tag-recapture data.

w_{LLH}	w_{LLU}	w_{FIN}	w_{LOL}	w_{RFIN}	w_{RLOL}	ϕ
0.829	0.666	0.183	0.102	0.269	0.530	2.038

Table 3. Input parameters in the assessment model.

Component	Parameter	Value				
Length-weight	a ($t \cdot cm^{-1}$)	5.63e-9				
	b	3.135				
Von Bertalanffy growth	L_{inf} (cm)	178.28				
	k (y^{-1})	0.061				
	t_0 (y)	-2.505				
	CV	0.155				
Natural mortality	M (y^{-1})	0.165				
Maturity *(proportion mature at age)*	Age 1	0	Age 12	0.388	Age 23	0.562
	Age 2	0.003	Age 13	0.413	Age 24	0.579
	Age 3	0.034	Age 14	0.436	Age 25	0.599
	Age 4	0.069	Age 15	0.455	Age 26	0.620
	Age 5	0.110	Age 16	0.473	Age 27	0.643
	Age 6	0.156	Age 17	0.487	Age 28	0.666
	Age 7	0.202	Age 18	0.500	Age 29	0.689
	Age 8	0.248	Age 19	0.512	Age 30	0.712
	Age 9	0.289	Age 20	0.523	Age 31+	0.734
	Age 10	0.327	Age 21	0.535		
	Age 11	0.359	Age 22	0.548		
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})	0.0021				
Tag emigration rate (added to the tag shedding rate in the model)	λ_M (y^{-1})	0.0438				
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 (Table 4). 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 step 2; and ageing in step 3. Since both fishing and natural mortality occur in time step 1, the process was to apply half the time step's natural mortality, then fishing mortality instantaneously, then the remaining half of the time step's natural mortality.

Table 4. Annual cycle of the current toothfish stock assessment model with three timesteps, and the processes in order of occurrence within each timestep.

Time-step	Period (approx.)	Sequence of processes	Observations
1	January - June	Recruitment	
		0.25 M	
		Fishing (removals by all fisheries and surveys instantaneously)	CPUE, LF (CAA), tag recaptures
		0.25 M	
		0.5 tag loss	
2	July - December	0.25 M	
		Spawning (SSB calculation)	
		0.25 M	
		0.5 tag loss	
3	Year-end (instant)	Age increment	

M – natural mortality, SSB – spawning stock biomass, CPUE – catch per unit of effort, LF – catch length frequencies, CAA – catch at age.

Recruitment to the population was calculated by multiplying average recruitment (R0) by 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 an unfished population when the spawning stock biomass declines to 20% of its unfished level. Steepness 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 2024. Projections from the model were extended for another 35 years, up to 2059. 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).

Estimation method

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

The posterior distribution of the parameters in a Bayesian analysis was estimated using the Monte-Carlo Markov Chain (MCMC) method. The starting point of each chain was the corresponding MPD estimate, the first 100,000 iterations were dismissed (burn-in), and every 1,000th value was taken from the following 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, the single-chain stationarity test of Geweke (1992) and the stationarity and half-width tests of Heidelberger and Welch (1983).

Estimated parameters

Parameters estimated by the model, their priors, starting values and bounds are given in Table 5. Annual year class strengths (YCS) were estimated for 1986-2023, with the *Haist parameterisation* used to make the YCS parameters average to 1 over 1986-2018 (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		38	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 $_{LOL}$	a_1	1	uniform	2	1	50
	S_L	1	uniform	1	0.05	50
	S_R	1	uniform	2	0.05	500
Selectivity $_{RFIN}$	a_1	1	uniform	2	1	50
	S_L	1	uniform	1	0.05	50
	S_R	1	uniform	2	0.05	500
Selectivity $_{RLOL}$	a_1	1	uniform	2	1	50
	S_L	1	uniform	1	0.05	50
	S_R	1	uniform	2	0.05	500
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). Logistic selectivity ogive was used for longline fisheries, and double-normal for trawl fisheries 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

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

Penalties

Besides observations and priors, the final components of the objective function are penalties. The model included a catch limit penalty and a vector average 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. Penalty multipliers were set to 100 for catch limits and 20 for the YCS vector average penalty (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 1,000 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 (2022-2024) and future recruitments (2025-2059) 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 stabilise. CASAL searches over mortality rates *F* to find F_{MSY}, the value that maximises 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 in Appendix 2. The model fitted standardised CPUE data relatively well, i.e. it captured the declining overall trend for the Spanish system longline fishery and a more levelled trend for the umbrella-system longline fishery (Figure A.3). Corresponding trends in normalised residuals for both longline fisheries are 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, except for longline fisheries where the model consistently overestimated the proportion of 1-3-year-old fish

(Figure A.11). The model fit to the observed mean toothfish age was good in all cases except the Spanish system longline fishery (Figure A.12).

The model fit to tagging data was generally good; however, notable discrepancies occurred 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 or (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 profiling was done by fixing SSB_0 over a range of plausible values (20,000 - 60,000 t) and estimating the remaining parameters. CPUE and tag-recapture data of fish tagged in 2016, 2019, and 2024 favoured lower biomass estimates. CAA data for the Spanish system longline, calamari trawl fishery and the two surveys, and tag-recapture data of fish tagged in 2020 and 2022, found that higher biomass estimates are more likely. CAA data for the umbrella system longline and finfish trawl fishery, and tag-recapture data of fish tagged in 2017, 2018, and 2021 were at, or close to, the MPD biomass estimate (Figures A.16, A.17).

MCMC trace plots showed no evident lack of convergence in estimated parameters, except for the left-hand limb shape parameter of double-normal selectivity curve for the calamari trawl fishery and both surveys (Figure A.18). The stationarity test of Geweke (1992) and the Heidelberger and Welch (1983) stationarity and half-width tests suggested failure to converge for the mentioned selectivity parameters (Table A.2) and autocorrelations in their MCMC samples 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 (Table A.3).

3.2. Model estimates

MPD estimates (with MCMC 95% credible intervals) of initial spawning stock biomass (SSB_0), current spawning stock biomass (SSB_{2024}) and current spawning stock biomass relative to SSB_0 (SSB_{2024}/SSB_0) are in Table 6. The estimates of absolute SSB were slightly lower (2-4% across the time series), and of relative SSB higher (0-3% across the time series) compared to the 2024 assessment. MCMC posterior distributions of SSB_0 and SSB_{2024}/SSB_0 are in Figure 5; since the inclusion of tag-recapture data into the model in the 2023 assessment, these posterior distributions were noticeably narrower and only slightly asymmetrical compared to the earlier years. 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 2024 is in Figure 6. Deterministic MSY was estimated at 1,720 t, similar to the 2024 assessment (1,699 t).

Table 6. MPD estimates (and MCMC 95% credible intervals) of SSB_0, SSB_{2024} and SSB_{2024}/SSB_0.

SSB_0	SSB_{2024}	SSB_{2024}/SSB_0
24,813 (23,191 - 27,219)	12,361 (10,841 - 14,823)	0.498 (0.467 - 0.549)

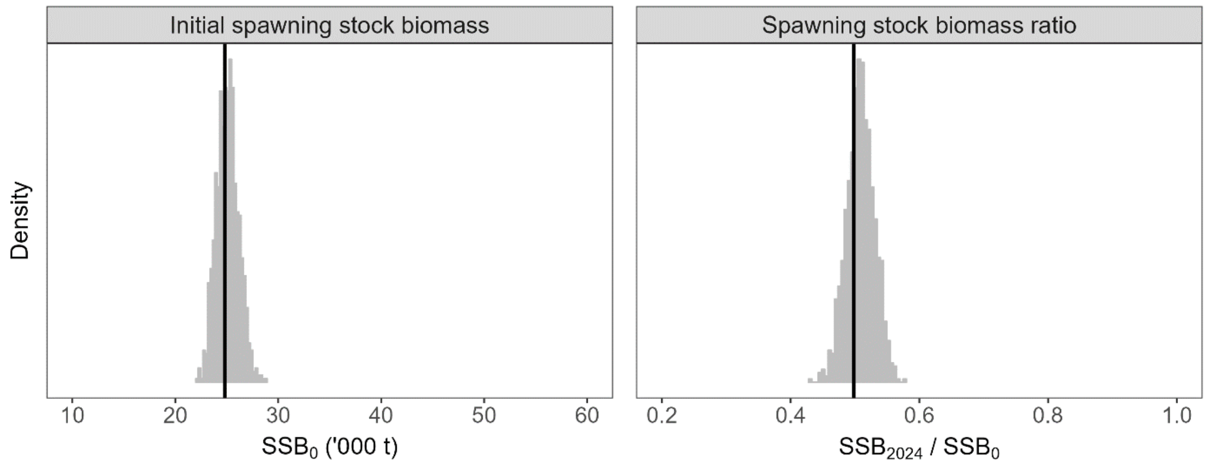

Figure 5. MCMC samples from the posterior distribution of SSB_0 and SSB_{2024}/SSB_0. Vertical black lines denote MPD point estimates.

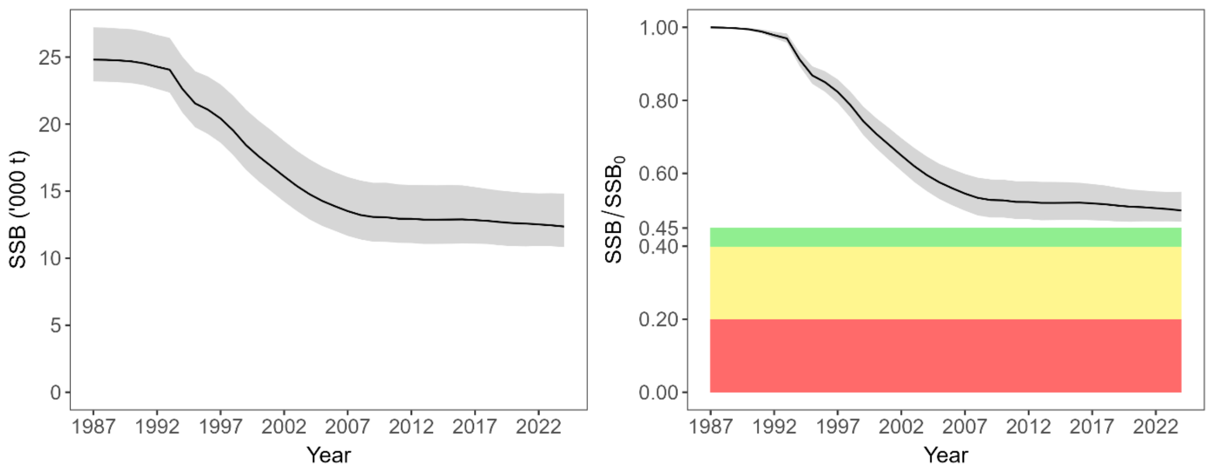

Figure 6. MPD estimate of the historical SSB trajectory; absolute on the left and relative to SSB_0 on the right (black line). Shaded areas denote MCMC 95% credible intervals of the model fit. Harvest control rule ranges are colour coded for reference: *target range* in green (SSB/SSB_0 = 0.45-0.40), *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 7). The calamari trawl fishery catches the youngest fish; a combination of fishing in shallow waters (=only young fish are available) and using a small mesh size (=low gear selectivity), resulting in a selectivity described by the right limb of the assumed double-normal selectivity ogive. For the finfish trawl fishery (assumed double-normal), 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). Trawl survey selectivities were roughly comparable to commercial trawl fisheries, as surveys employ similar gear and cover similar grounds.

The estimated year class strength for 2015-2020 aligned well with model-independent recruitment estimates (Lee *et al.* 2021), indicating high recruitment in 2015-2017 and low in 2018-

2020 (Figure 8). No model-independent recruitment estimates were available for the earlier or later years, making it challenging to confirm the YCS trend estimated in the model.

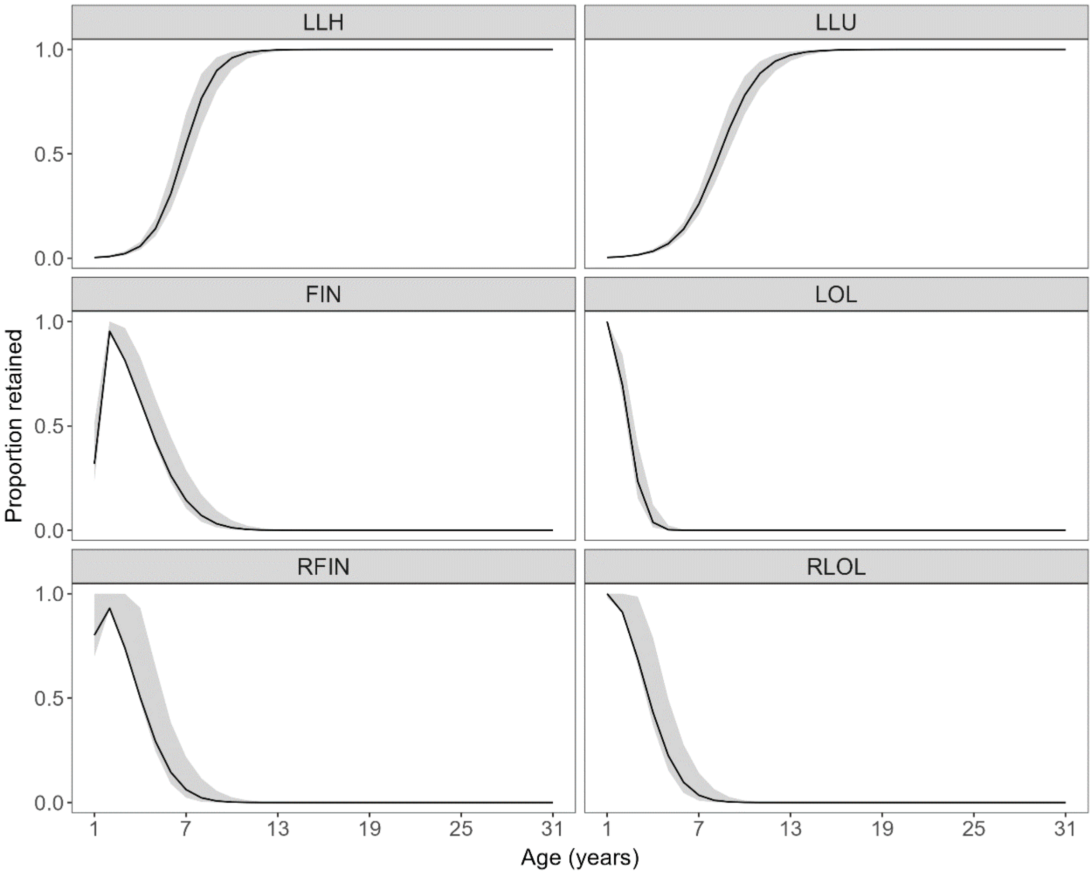

Figure 7. MPD estimate of selectivity ogives for four fisheries and two surveys (black lines). Shaded areas denote MCMC 95% credible intervals of the model fit.

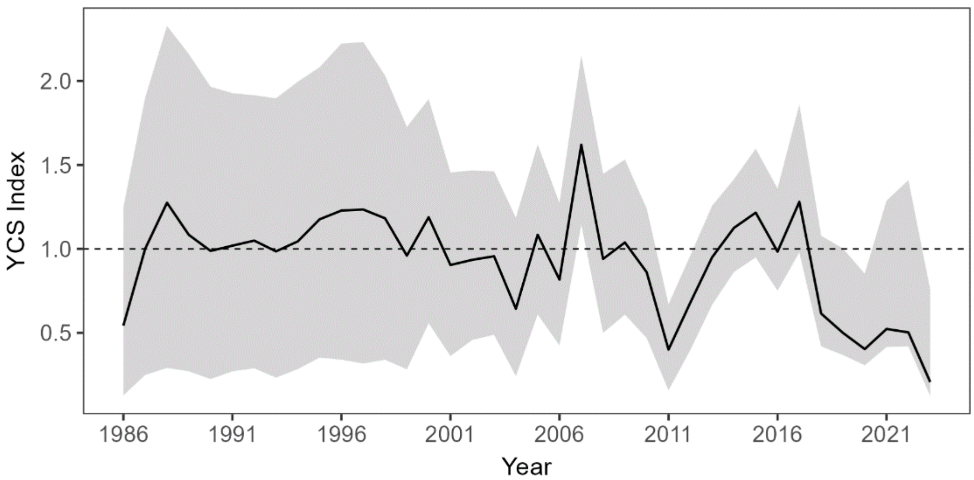

Figure 8. MPD estimate of year-class strengths in 1986-2023 (solid black line). The shaded area denotes MCMC 95% credible intervals of the model fit.

3.3. Model projections

Projections of SSB/SSB_0 under the assumption of random recruitments in 2022-2059 and constant annual catches in 2025-2059 are in Figure 9. The median SSB/SSB_0 is currently within the HCR *expansion range* but projected to drop to the *target range* by 2032. Throughout the projection period, the median SSB/SSB_0 reaches a minimum of 0.44 in 2036 before increasing to 0.45 by 2040. 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 10). The highest probability of falling below 0.45, 0.40 and 0.20 thresholds during the projection period was estimated at 55%, 32% and 1%, respectively.

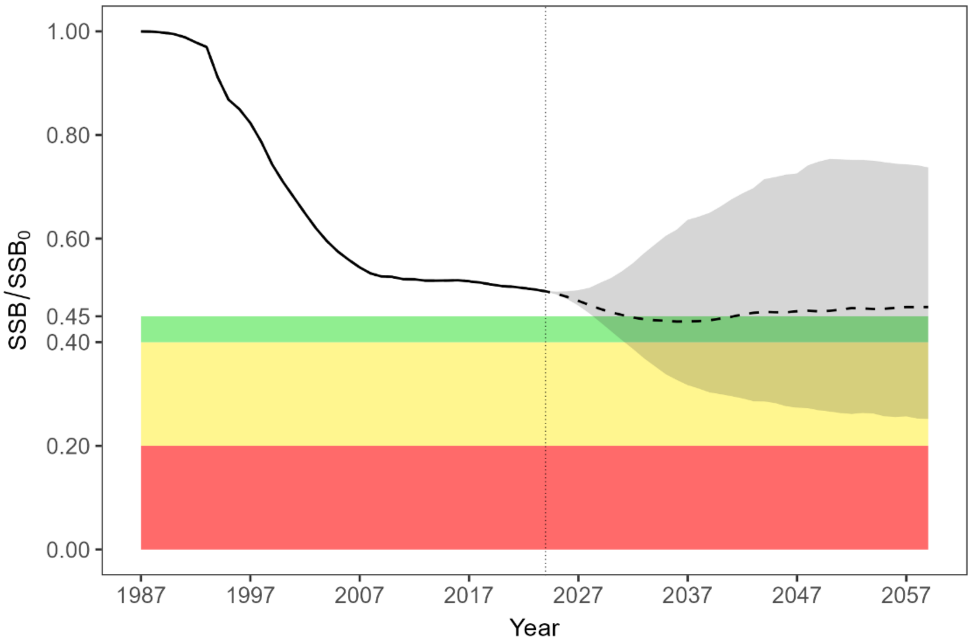

Figure 9. Future SSB/SSB_0 projections from the model MPD estimate. Shown are the MPD estimate (solid black line), median of the projections (dashed black line), and 95% confidence intervals of the projections (shaded area). Harvest control rule ranges are colour coded for reference: *target range* in 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).

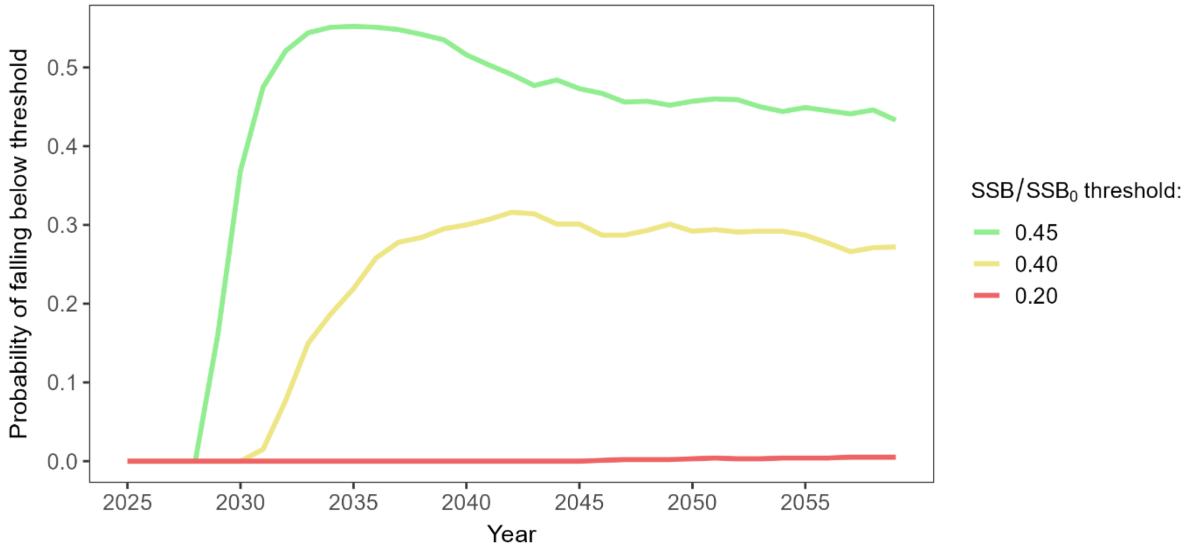

Figure 10. Probability of stock falling below designated SSB/SSB$_0$ management thresholds, based on projections.

4. Discussion

This report presents an updated integrated stock assessment for toothfish in Falkland Islands waters, using data up to the end of 2024. It is the third consecutive assessment in which tag-recapture data have been used as an index of absolute abundance, thus reducing model reliance on commercial CPUE data. Although commercial CPUE data are widely used as an index of relative abundance in stock assessments, their shortcomings 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 employed in toothfish stock assessments in CCAMLR waters; extensive long-term tag-recapture datasets are utilised for assessments in South Georgia, South Sandwich Islands, Heard and McDonald Islands, Kerguelen, Crozet Island, and Ross Sea (Earl and Readdy 2023; Readdy and Earl 2023; Masere and Ziegler 2023; Massiot-Granier *et al.* 2023a, 2023b; Mormede *et al.* 2023). In comparison, the Falkland Islands toothfish tagging programme is still relatively new; it began in 2016, although the tagging effort has varied widely between years. The initial goal of tagging 3,000 fish was achieved between 2016 and 2018, followed by a considerable decrease in tagging efforts in 2019-2020. The programme was expanded and formalised as a long-term protocol in 2021, aiming to tag approximately 1,000 fish annually (one fish per tonne of TAC) to support the stock assessment. Since the extension, tagging has predominantly been conducted during dedicated surveys, aiming to cover the entire fishing area. In 2024, the tagging programme underwent further formalisation, with the fishing area delineated based on the distribution of fishing effort over the most recent five years and then divided into six geographically distinct sub-areas. The required number of tags was allocated among sub-areas in proportion to the fishing effort (Le Luherne and Desmet 2025). As of 2024, fish are being tagged with a single (large) tag instead of two (one small, one large); this needs to be accounted for when calculating tag-shedding rates in future assessments. The current stock assessment was informed by approximately 6,700 releases and around 450 recaptures (once within-year and out-of-zone recaptures were excluded). These numbers remain low compared to some toothfish assessments (e.g. for South Georgia, with upwards of 58,000 releases and 7,800 recaptures by the end of 2022); given its significance for the stock assessment, the tagging programme should be continued and established as a permanent requirement.

The estimated model parameters and derived quantities in the current assessment were similar to those in the previous stock assessment (Skeljo and Winter 2024). The current model resulted in a slightly higher estimate of absolute SSB and a lower estimate of relative SSB across the time series; however, differences were never more than a few percent in any year. The SBB ratio in the most recent year was almost the same in the current (SSB_{2024}/SSB_0 = 0.50) and the previous assessment (SSB_{2023}/SSB_0 = 0.49). This was not unexpected, as there were no major changes to the model structure or assumptions compared to the previous year, and the data updates through 2024 didn't suggest notably different trends. The effect of different data updates on model outcomes was still explored by successively excluding CPUE, CAA, and tag recapture data for 2024 and re-running the model. The comparison indicated that 2024 CPUE data favoured slightly more optimistic estimates of the SSB ratio, CAA data had almost negligible effect, and tag recapture data favoured slightly more pessimistic estimates. However, the differences were minor. Changes to input parameters (Von Bertalanffy growth parameters, length-weight coefficients, maturity vector, tag shedding and tag emigration rates) due to data updates were minimal and had a negligible effect on model outcomes.

The future projections of SSB/SSB_0 in the current assessment were similar to those in the previous assessment (Skeljo and Winter 2024), with an anticipated drop from the *expansion range* to the *target range*. The projected drop is a response to a series of weak recruitments estimated by the model (below-average YCS in 2018-2020) and supported by an independent analysis (Lee *et al.* 2021). However, the projection was slightly more optimistic than in the previous assessment, with stock projected to remain below the *expansion range* for a shorter period (2032-2040 compared to 2029-2045 in the previous assessment). This change is partially due to a higher MPD estimate of $SSB_{current}/SSB_0$ in the current model. The MPD estimate was the starting point for the projections, and everything else being the same, it would take the projected SSB/SSB_0 longer to drop below 0.45 when starting from a higher $SSB_{current}/SSB_0$ value. The projected SSB/SSB_0 recovery to the *expansion range* earlier than in the previous assessment is likely due to a slightly higher estimated recruitment strength in 2021 (last year with model-estimated YCS in the current model) compared to 2020 (last year with model-estimated YCS in the previous model). The current projections assumed random YCS for 2022 and 2023 instead of model-estimated values because of limited information on the most recent recruitments available to the model (a common approach; see Masere and Ziegler 2023). For example, if a strong toothfish recruitment occurred in an area not covered by the calamari trawl fishery or surveys (e.g. on the west or southwest Falkland Islands Shelf), it might take a few years before this fish appears in the catches and thus become available to the model. Model estimates of YCS could potentially be improved by introducing the survey biomass index in the analysis, in addition to already used survey CAA data. Preliminary tests were conducted in 2025 and suggested that sufficient data is available. However, before becoming a part of the base-case model, further work is required on the survey density data standardisation, followed by model testing and sensitivity analysis; this work is anticipated in 2026. Given the influence of recent recruitment strengths on model projections, close monitoring of juvenile toothfish abundance during research surveys and commercial calamari fishing seasons needs to be emphasised. Protection of high recruitment age-0 cohorts while on the shelf via spatiotemporal management of calamari trawl fishery has been proposed (Skeljo 2023), and a protocol was established in early 2024. However, no noticeable recruitment of age-0 toothfish into calamari fishery grounds occurred in 2024 or 2025, and the protocol has not been tested in practice yet.

Currently, FIFD is conducting a dedicated project on toothfish reproductive traits in Falkland Islands waters, aiming to revise the current maturity-at-age data and provide insight into the prevalence of skipped spawning in females. However, sampling mature toothfish proved challenging. The only longliner fishing in Falkland Islands waters has been unavailable during the peak spawning time due to an established maintenance schedule (in Spain). In 2025, the vessel remained in the Falkland Islands, and an arrangement was made with CFL (taking into consideration vessel layover and crew change schedule) for survey during the July-October fishing trip. However, the fishing schedule changed, and the targeted fishing trip was cancelled following the announcement of tariffs on all imports to the US (a major toothfish market for CFL). Given that the current maturity-at-age information used by the model needs to be revised sooner rather than later, the recommendation is for the existing samples to be processed and analysed. The analysis of the new samples should be complemented by a revision of the currently used maturity data (comparison to approaches used in other toothfish stocks is advised). Once available, the revised data will be considered an interim solution and introduced to the stock assessment until a more comprehensive sampling and analysis can be done.

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) (Appendix 4). The estimated SSB_{2024}/SSB_0 ratio of 0.498 was above the *upper target reference point* (0.45), i.e. in the *expansion range*; projections from the current model indicated that the SSB/SSB_0 ratio will drop and remain in the *target range* during 2032-2040. The year 2024 was the fifth consecutive year with $SSB_{current}/SSB_0$ estimated to be in 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 the toothfish annual TAC in the longline fishery at its current level of 1,040 tonnes.

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Appendix 1. CPUE standardisation back to text

CPUE was standardised 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. Continuous explanatory variables were scaled by subtracting the mean and dividing by the standard deviation. Reports with zero toothfish catch were presumed to represent erroneous entries or broken sets and were excluded from the analysis (1.2% of days for the Spanish system and 0.01% of lines for the umbrella system fishery).

The response variable was defined as toothfish CPUE expressed in kg-per-hook per day (Spanish system) or kg-per-umbrella per line (umbrella system) and modelled using a 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 standardisation.

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

* 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* variable aims to capture intra-annual trends in effort concentration. The *Area* variable aims to account for spatial heterogeneity in stock density. Area was defined by the 1° Lon x 1° Lat grid squares; these were considered sufficiently small and numerous to accommodate spatial patterns in CPUE, while avoiding overfitting. Only grid squares with at least 10 catch reports across the time series were kept for the analysis, resulting in the dismissal of 20 daily catch reports for the Spanish system, and 9 line catch reports for the umbrella system. The *Depth* variable is the average fishing depth of each line (umbrella system) or the average fishing depth of multiple lines set in a day (Spanish system). The *Soak-time* variable was defined for the umbrella system only, representing the soak time of individual lines. The *Soak-time* variable was not available in a suitable format for the Spanish System, i.e. it was reported as the sum of soak times of multiple lines set on the same day. As the number and size of individual lines is unknown, soaking a 10,000-hook line for 10 hours would be reported as 10-hour soak time, while soaking two 5,000-hook lines for 10 hours would be reported as 20-hour soak time; when the same number of hooks was deployed for the same length of time in both cases. The *Vessel* variable was defined in the Spanish-system standardisation 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 standardisation because only two vessels appeared in the model and never fished concurrently in a year, making the *Vessel* and *Year* effects indistinguishable. Finally, the *Year:Area* interaction was included as a random effect in the CPUE standardisation to account for potentially different temporal trends between areas.

The final GLMM fitted to the Spanish-system data included *Year*, *Month*, *Area*, *Vessel*, and *Year:Area* interaction; the model explained 26.2% of the overall variation in CPUE. The final GLMM fitted to the umbrella-system data included *Year*, *Month*, *Area*, *Depth*, *Soak-time*, and *Year:Area* interaction; the model explained 29.0% of the overall variation in CPUE.

The standardised CPUE index was constructed following the ‘predict-then-aggregate’ method, as described in Hoyle *et al.* (2024). The *Month*, *Vessel*, *Depth*, and *Soak-time* were considered ‘catchability’ covariates, and *Year* and *Area* ‘density’ covariates. The density covariates (and their interaction) were used to predict CPUE that would have occurred in each area each year, dropping the partial effect of catchability covariates by setting them at a reference level. The advantage of treating *Year:Area* interaction as a random effect is that, instead of having to impute a value of the standardised CPUE for the missing combinations of year and area, the posterior mean of the assumed normal distribution of the random effect (estimated within the model) can be used together with the parameter values for the fixed effects of *Year* and *Area* to determine these values (Campbell 2015). Finally, CPUE was aggregated across areas in each year; this is often done by weighting CPUE in each area by the area size. However, in this case, all the areas were of equal size, and the annual CPUE was calculated as an arithmetic mean of all individual areas' CPUEs in a given year. Standardised and unstandardised CPUE time series of the Spanish system and umbrella system are in Figures A.1 and A.2, respectively.

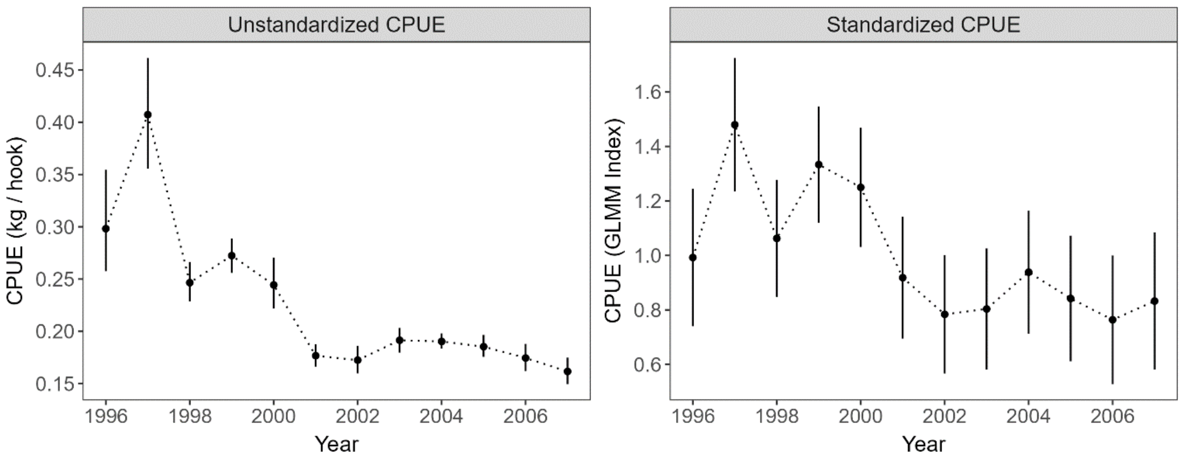

Figure A.1. Spanish-system longline fishery unstandardised and standardised CPUE time series; black vertical lines denote 95% confidence intervals.

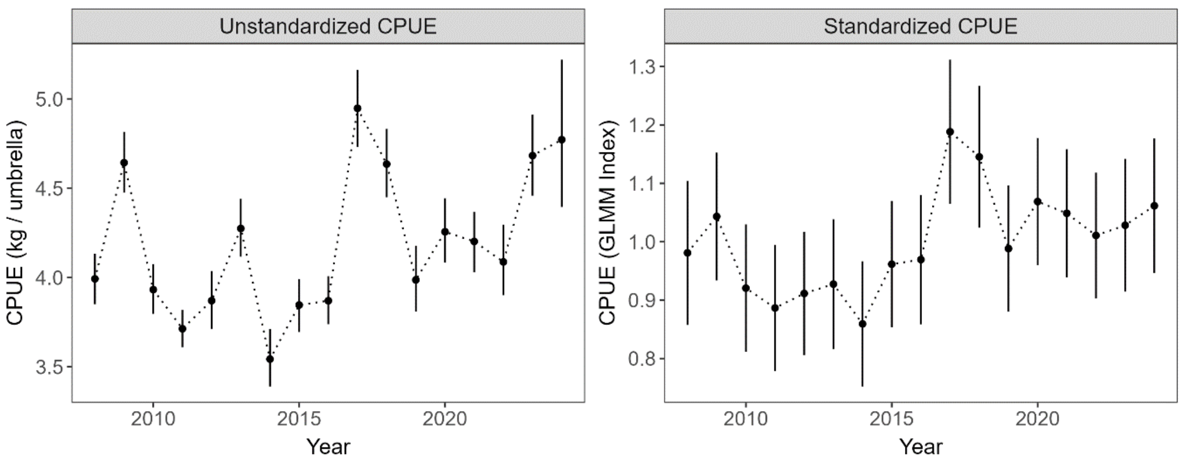

Figure A.2. Umbrella-system longline fishery unstandardised and standardised CPUE time series; black vertical lines denote 95% confidence intervals.

Appendix 2. Diagnostics plots

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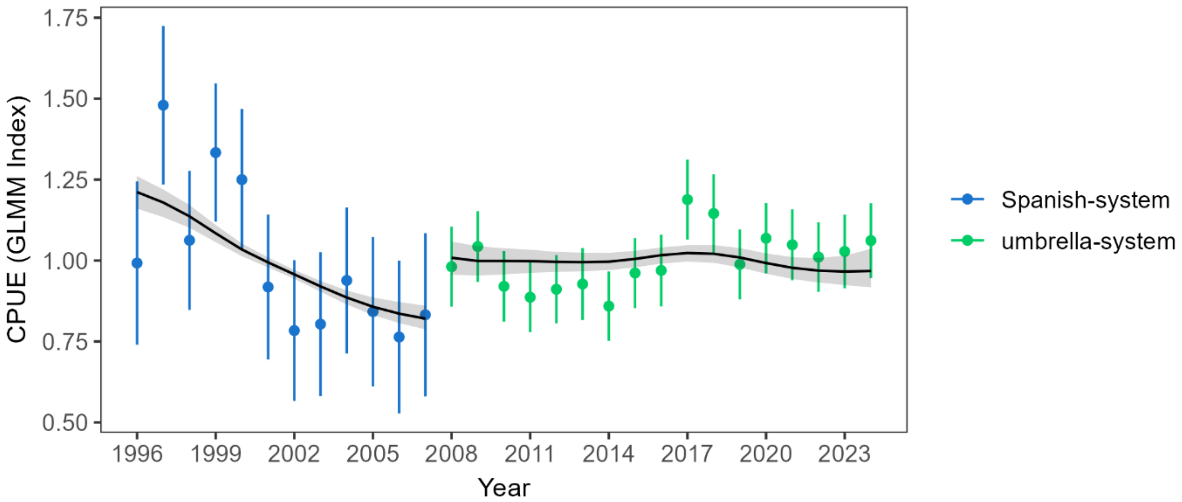


Figure A.3. MPD model fit (black line) to standardised CPUE indices for Spanish-system (blue dots) and umbrella-system longline (green dots); Vertical blue and green lines denote 95% confidence intervals of standardised CPUE indices; shaded areas denote MCMC 95% credible intervals of model fit.

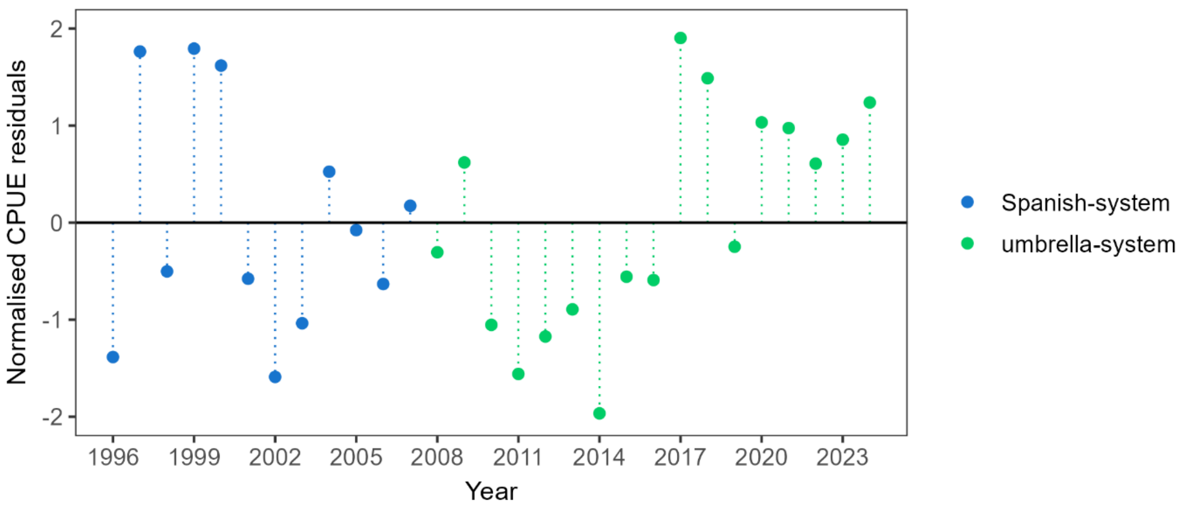


Figure A.4. Normalised residuals from model fit to standardised CPUE time series; for Spanish-system (blue) and umbrella-system longline (green).

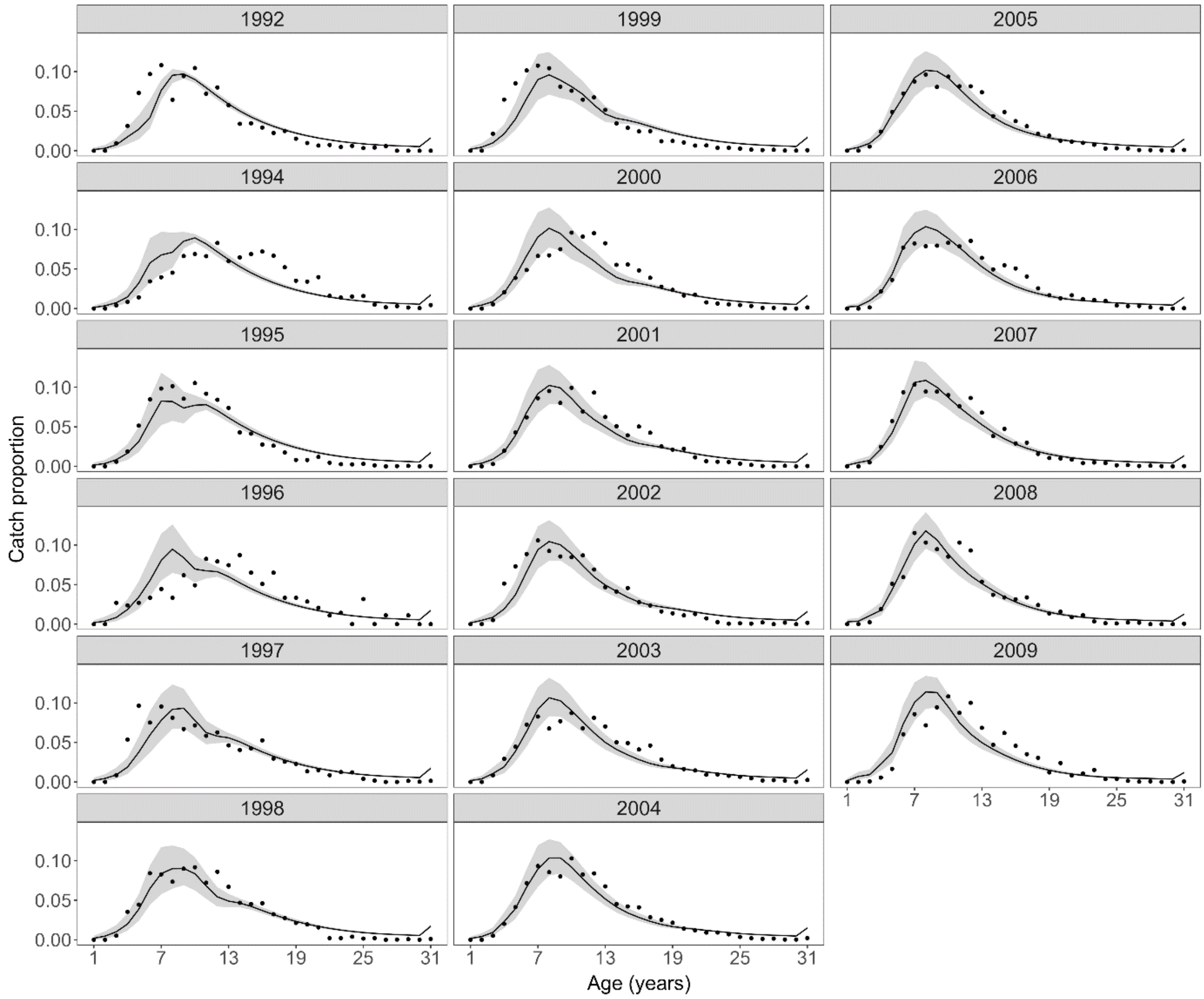


Figure A.5. MPD model fit (solid lines) to observed catch-at-age for Spanish-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

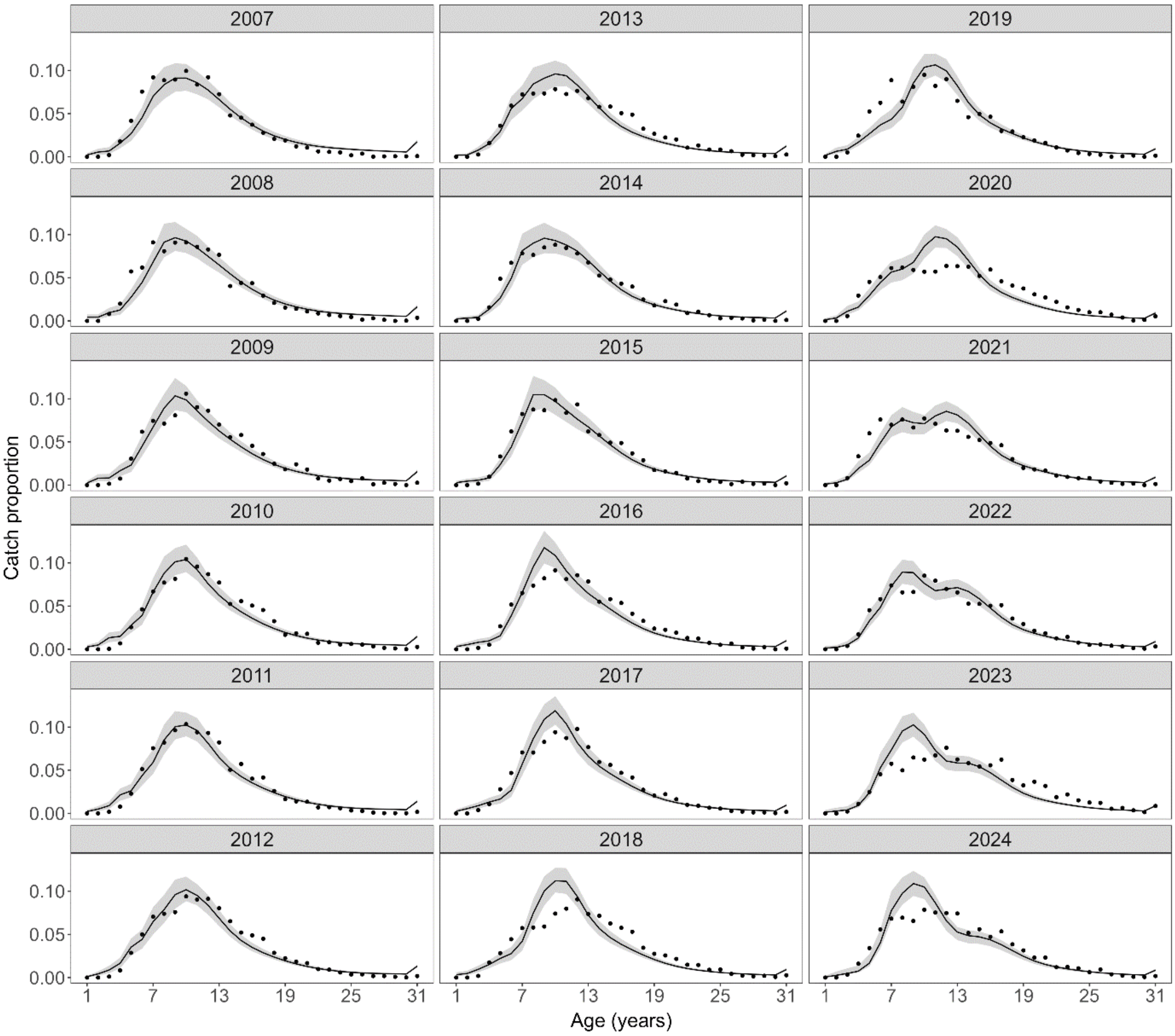


Figure A.6. MPD model fit (solid lines) to observed catch-at-age for umbrella-system longline fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

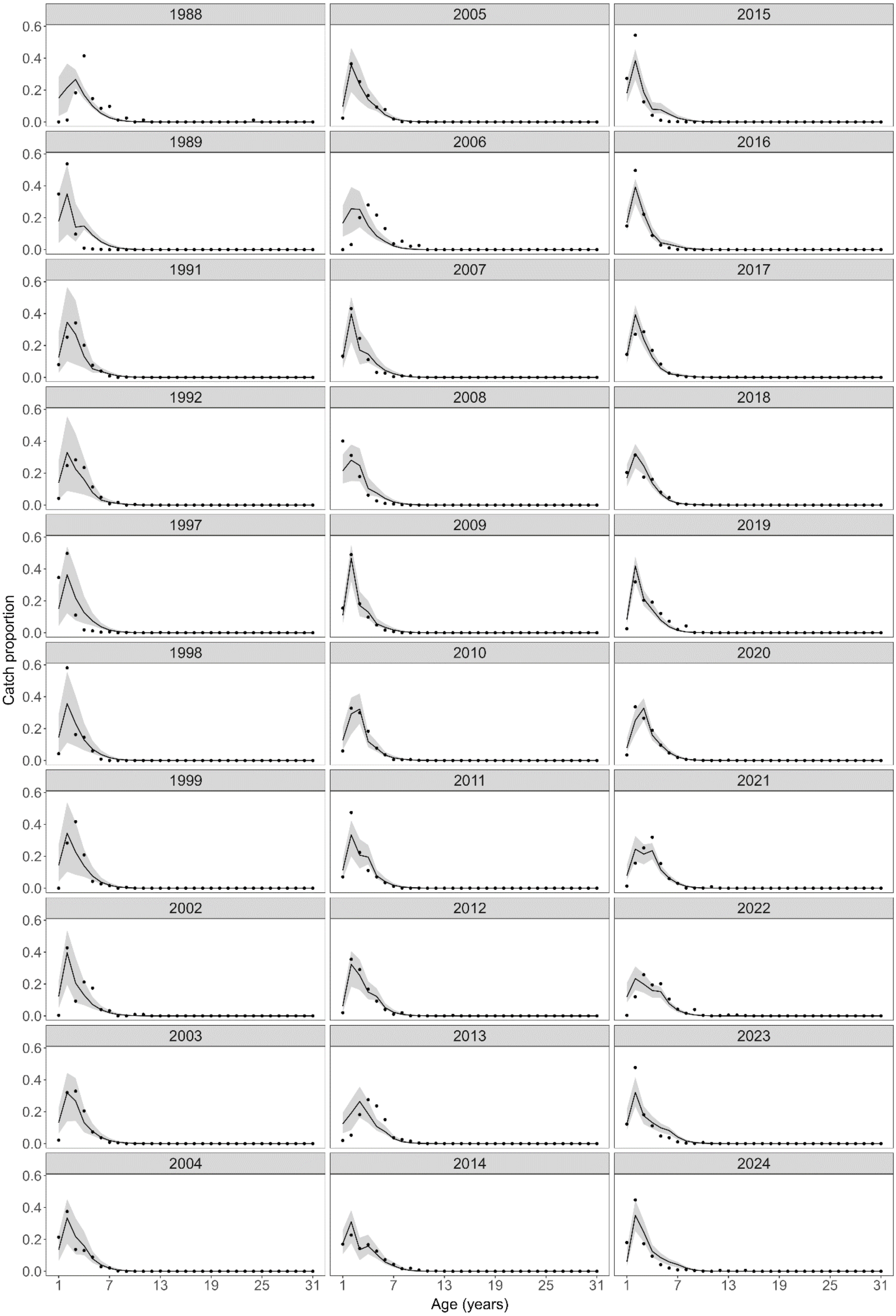


Figure A.7. MPD model fit (solid lines) to observed catch-at-age for finfish trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

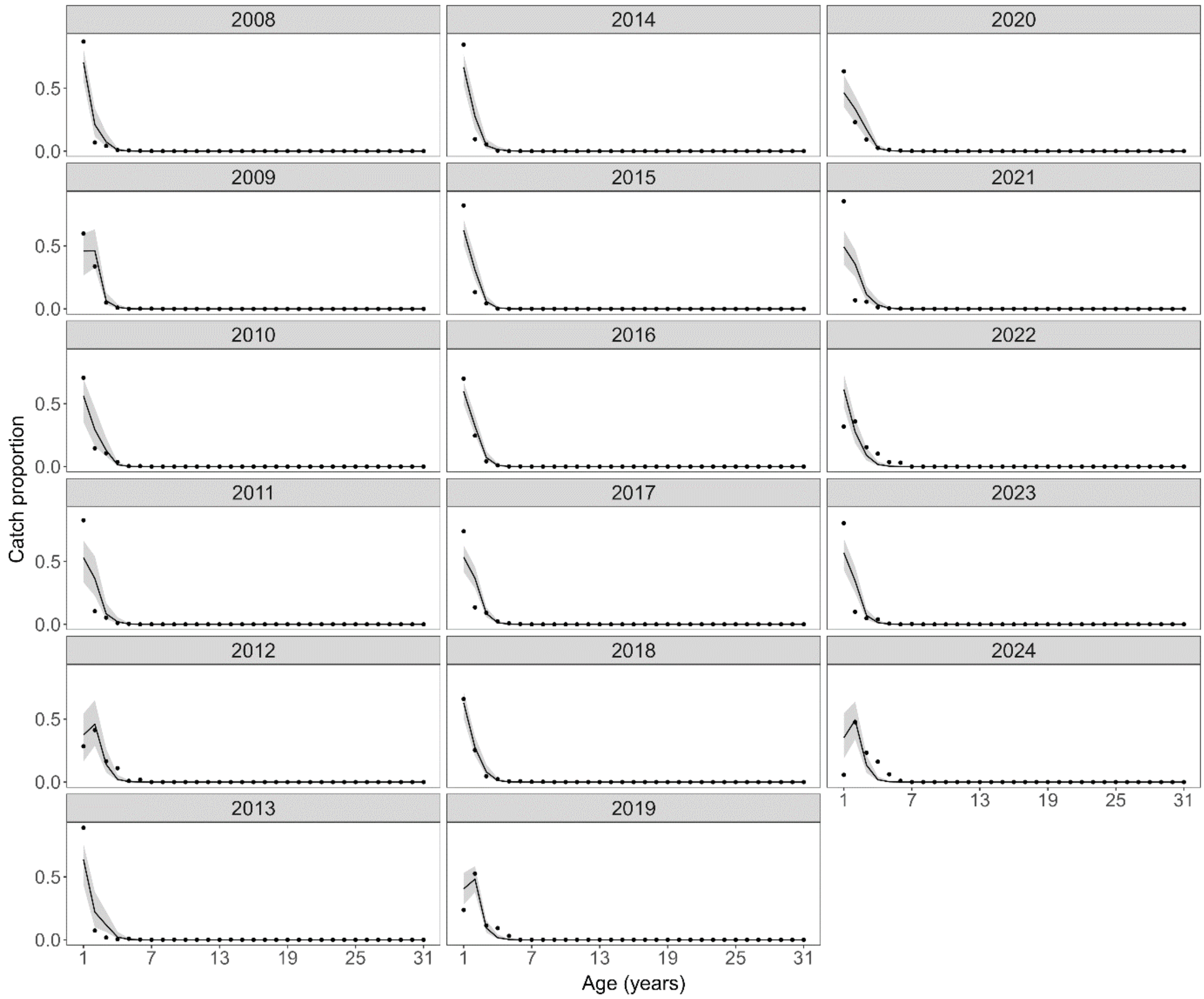


Figure A.8. MPD model fit (solid lines) to observed catch-at-age for calamari trawl fishery (dots); shaded areas denote MCMC 95% credible intervals of the fit.

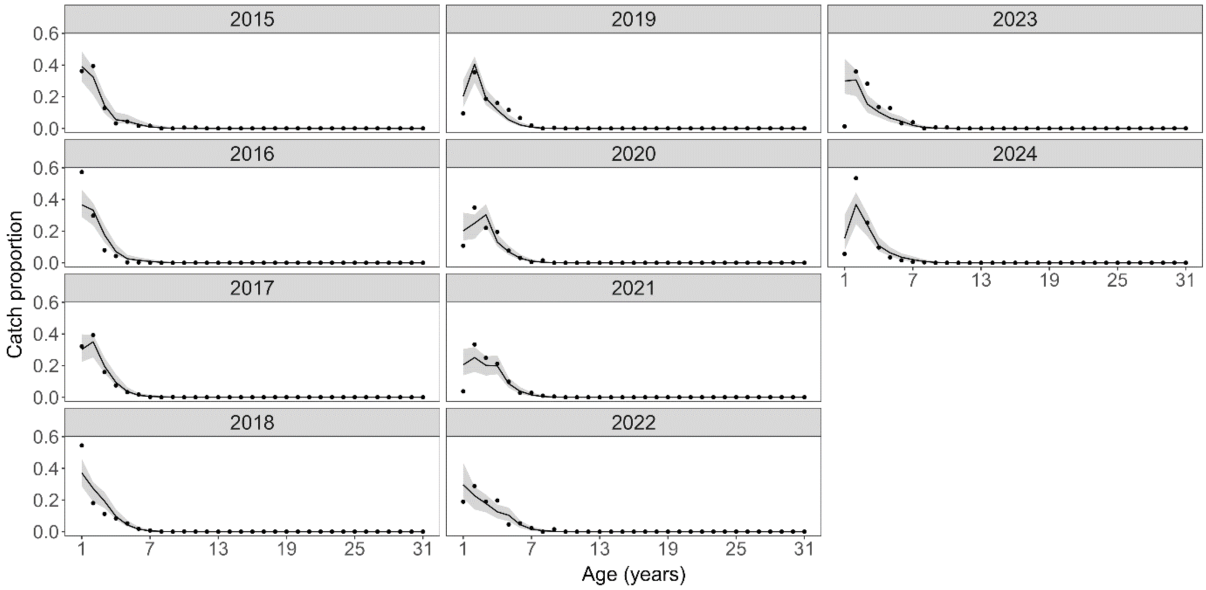


Figure A.9. MPD model fit (solid lines) to observed catch-at-age for groundfish survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.

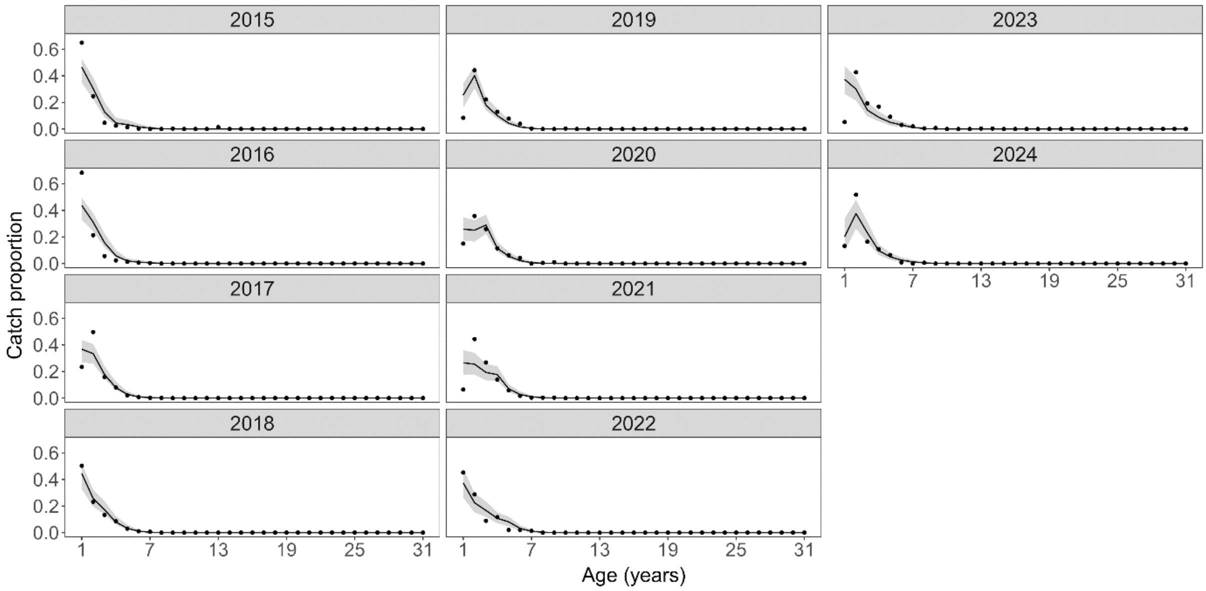


Figure A.10. MPD model fit (solid lines) to observed catch-at-age for calamari pre-season survey (dots); shaded areas denote MCMC 95% credible intervals of the fit.

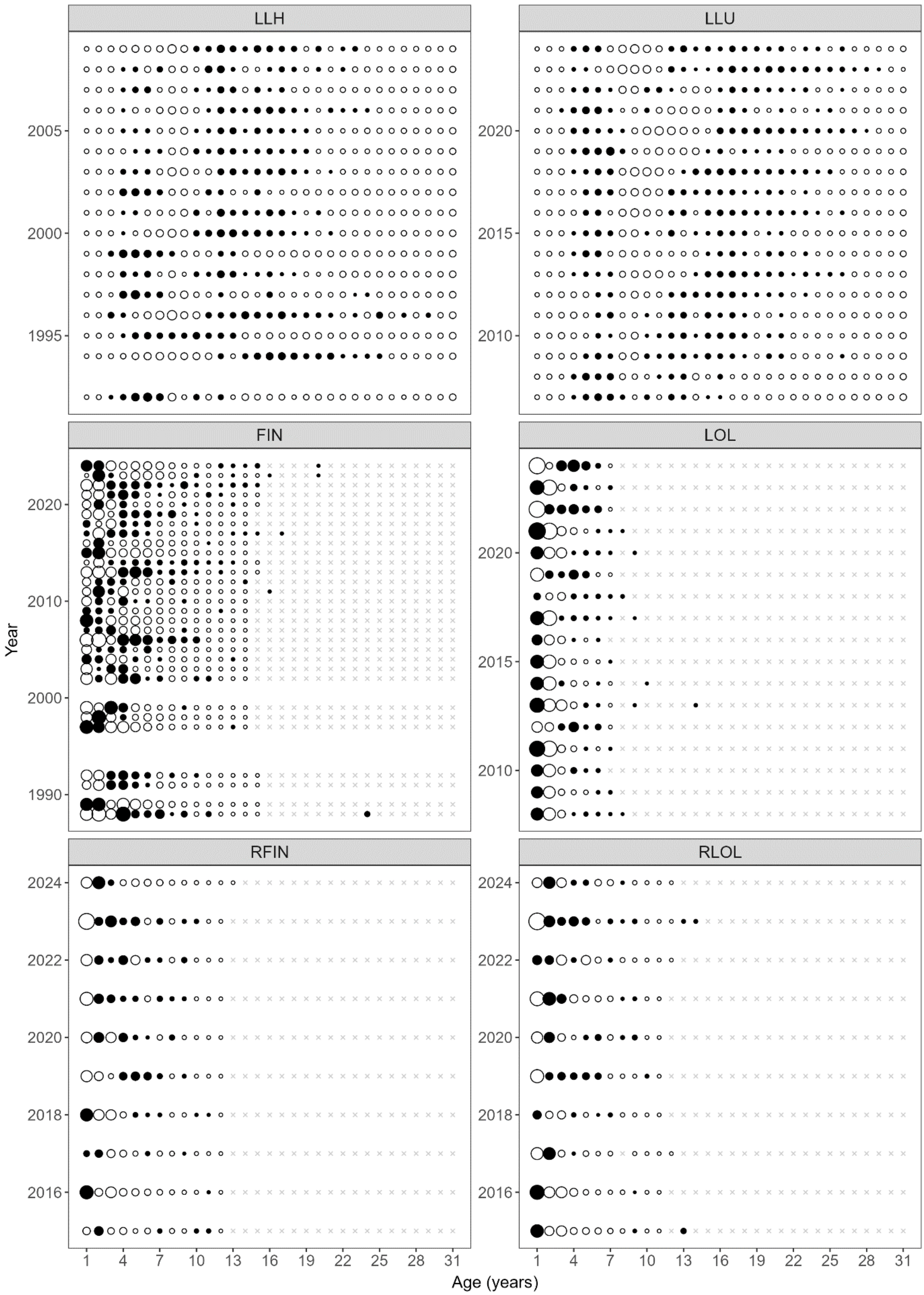


Figure A.11. Residuals from the model fit to observed catch-at-age for four fisheries and two research surveys. Bubble size is relative to the absolute residual value; positive residuals are denoted by full circles, and negative by empty circles.

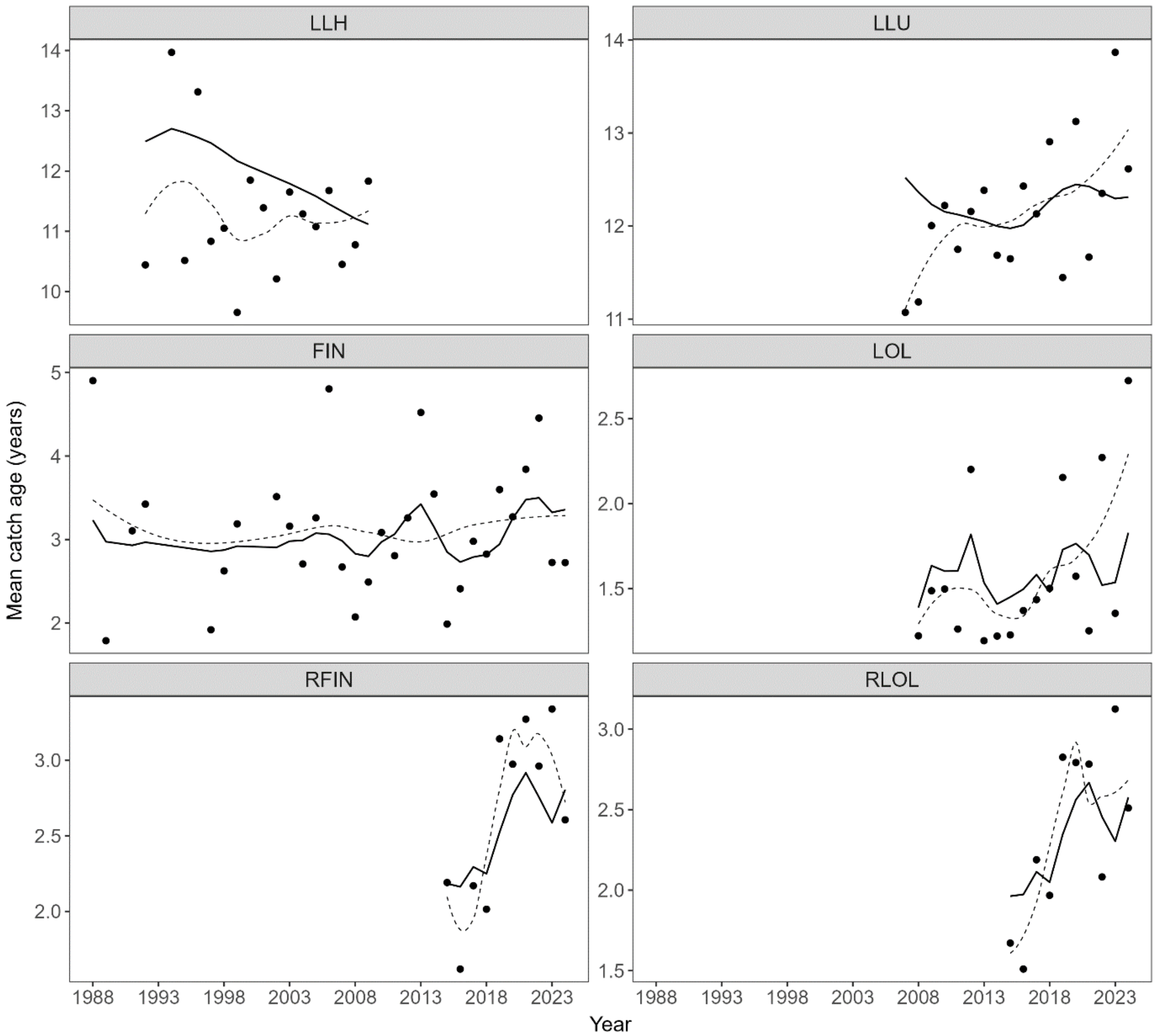
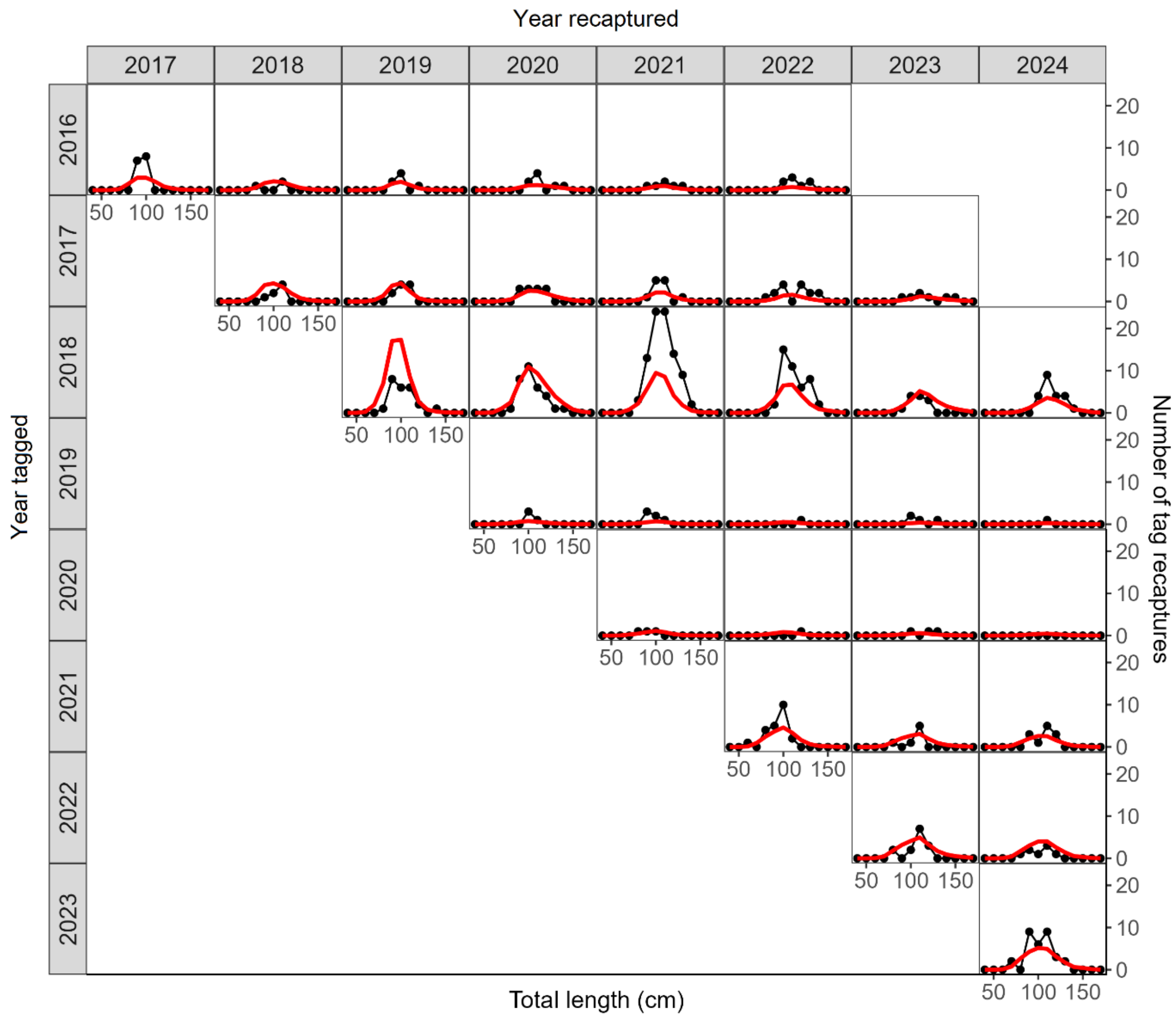
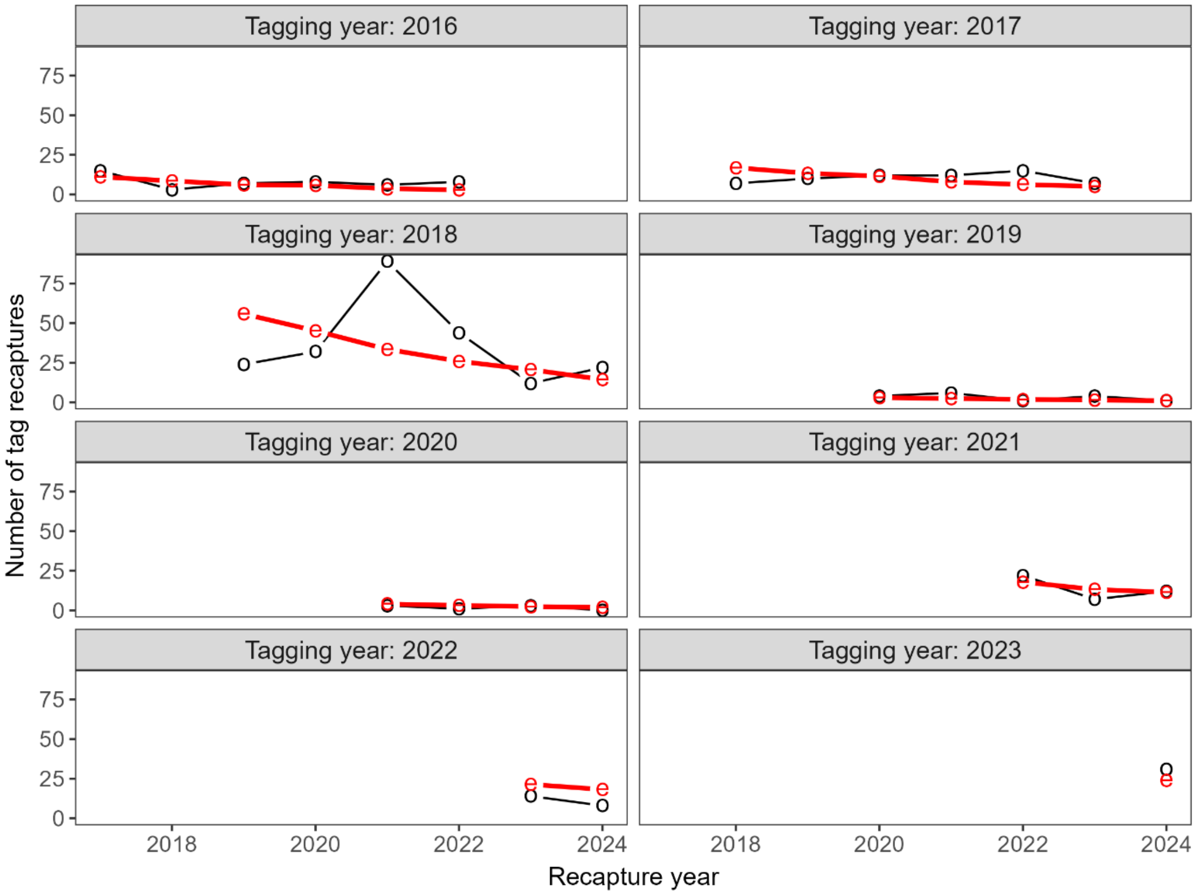


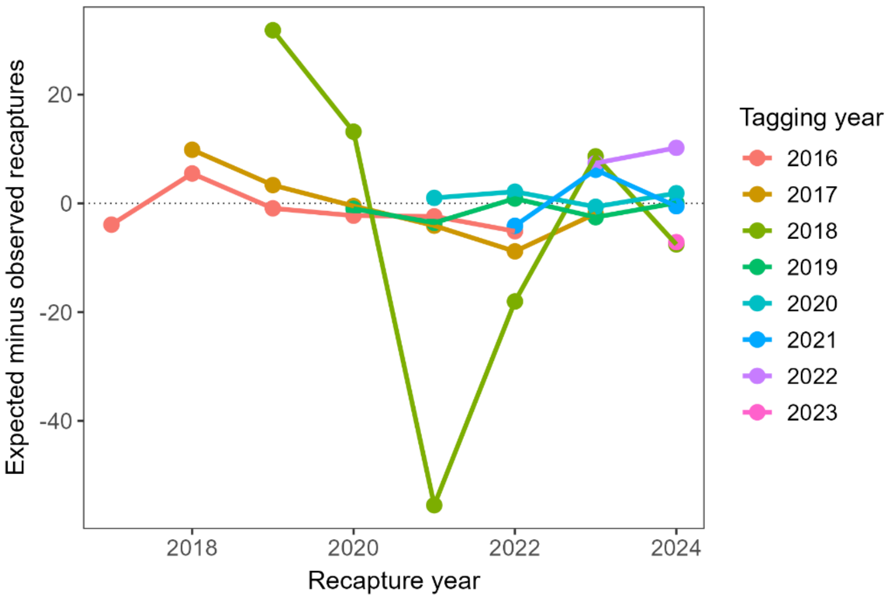
Figure A.12. Model fit (solid lines) to observed mean age-at-capture for four fisheries and two research surveys (black dots); loess smoothers with span = 0.75 (dashed lines) are added to aid visual interpretation of trends in the observed data.



A.13. Model fit (red lines) to observed tag recapture numbers by 10 cm length bins (black dots); for tag releases in 2016-2023 and tag recaptures in 2017-2024. Recaptures after six years at liberty were excluded from the model and the plot.



A.14. Model fit (red lines) to observed tag recapture numbers (black dots); for tag releases in 2016-2023 and tag recaptures in 2017-2024. Recaptures after six years at liberty were excluded from the model and the plot.



A.15. Differences between fitted and observed tag recapture numbers; for tag releases in 2016-2023 and tag recaptures in 2017-2024. Recaptures after six years at liberty were excluded from the model and the plot.

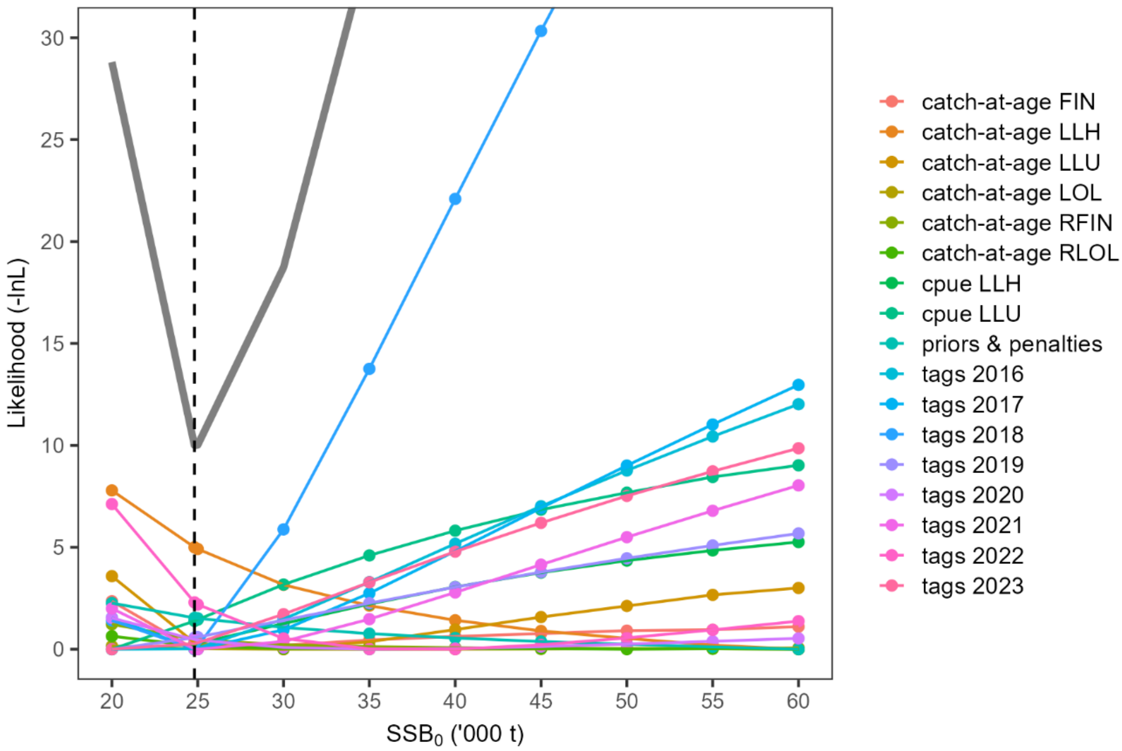


Figure A.16. Likelihood profiles for SSB_0. Negative log-likelihood values for individual datasets were rescaled to a minimum of zero, while the total objective function was rescaled to a minimum of 10 for easier visualisation (solid grey line). The dashed vertical line denotes the MPD estimate of SSB_0.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

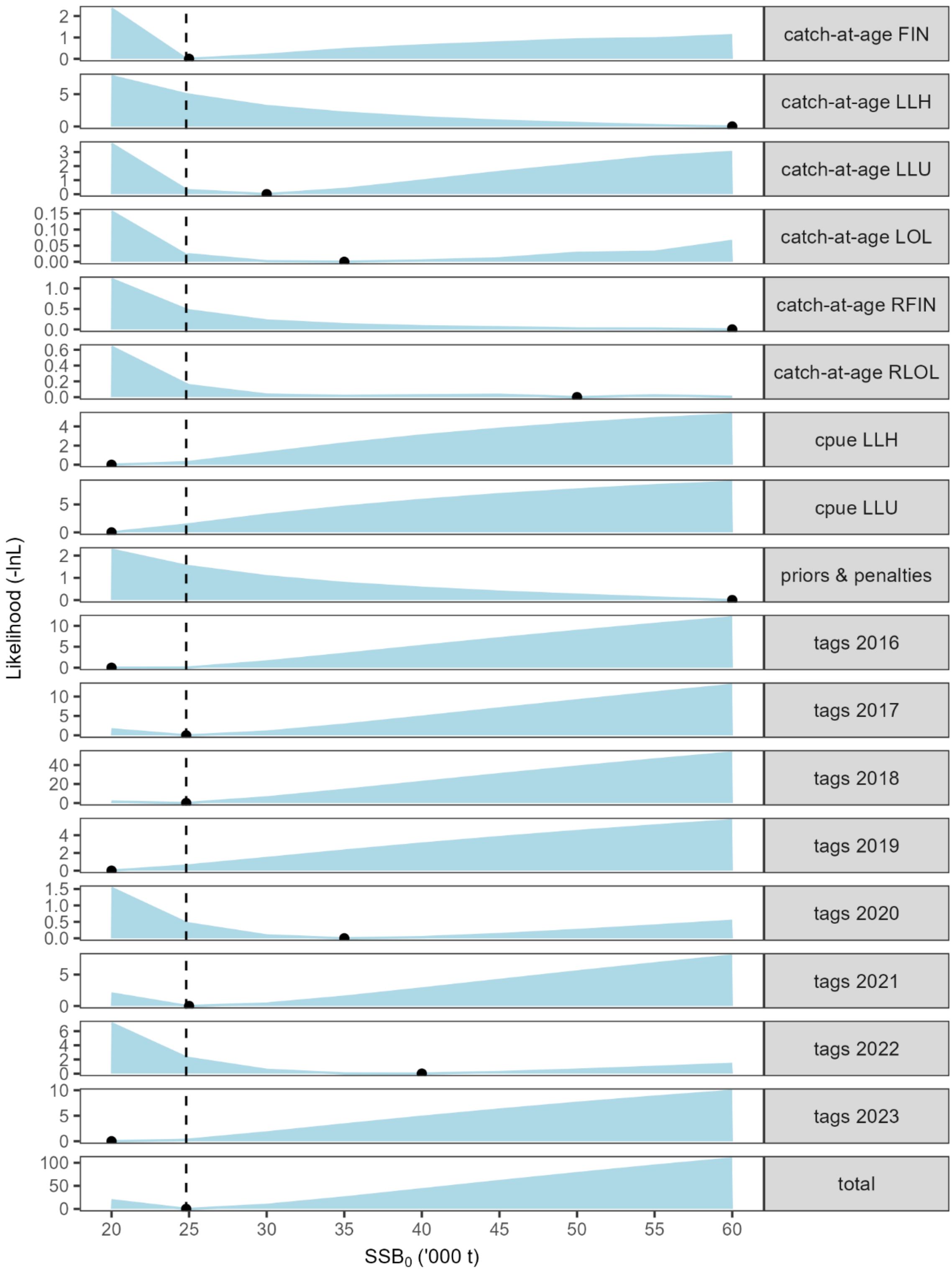


Figure A.17. Likelihood profiles for SSB_0. Negative log-likelihood values for individual datasets and the total objective function were rescaled to a minimum of zero. The dashed vertical line denotes the MPD estimate of SSB_0; dots denote SSB_0 values with the minimum negative log-likelihood value for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

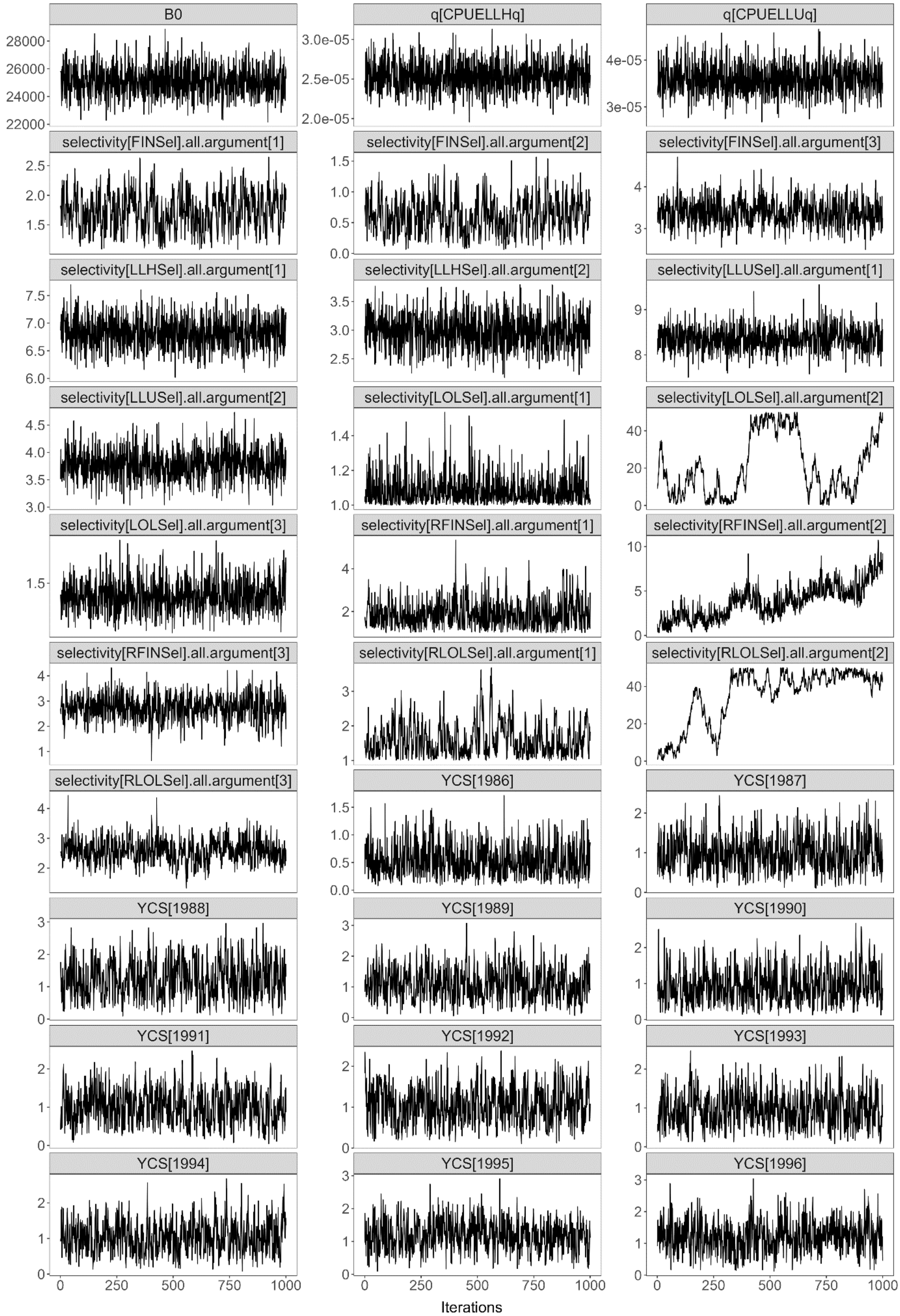


Figure A.18. MCMC posterior trace plots for all estimated parameters (figure 1 of 2).

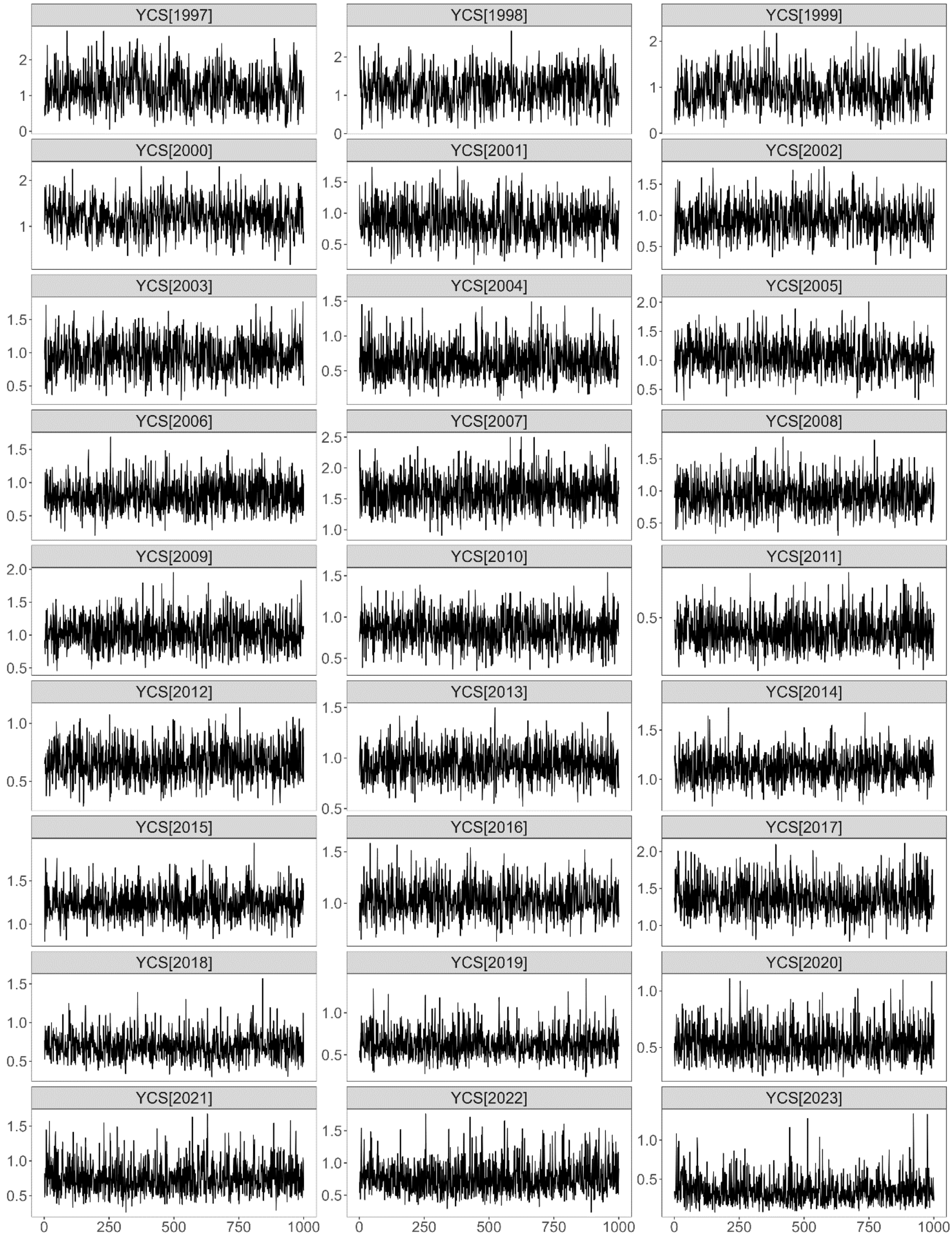


Figure A.18. Continued (figure 2 of 2).

Table A.2. MCMC convergence diagnostic results: Geweke's single-chain stationarity test and Heidelberger and Welch's stationarity and half-width tests.

Parameter	Geweke's stationarity test		Heidelberger and Welch's stationarity test		Heidelberger and Welch's halfwidth test		
	p-value	outcome	p-value	outcome	mean	halfwidth	outcome
B0	0.958	passed	0.982	passed	25088.3	57.64	passed
q[CPUELLHq]	0.337	passed	0.616	passed	0	0	passed
q[CPUELLUq]	0.773	passed	0.489	passed	0	0	passed
selectivity[FINSel].all.argument[1]	0.767	passed	0.379	passed	1.732	0.04	passed
selectivity[FINSel].all.argument[2]	0.69	passed	0.417	passed	0.609	0.035	passed
selectivity[FINSel].all.argument[3]	0.448	passed	0.606	passed	3.379	0.027	passed
selectivity[LLHSel].all.argument[1]	0.093	passed	0.5	passed	6.825	0.014	passed
selectivity[LLHSel].all.argument[2]	0.004	failed	0.117	passed	2.979	0.015	passed
selectivity[LLUSel].all.argument[1]	0.588	passed	0.262	passed	8.356	0.016	passed
selectivity[LLUSel].all.argument[2]	0.701	passed	0.282	passed	3.777	0.017	passed
selectivity[LOLSel].all.argument[1]	0.571	passed	0.096	passed	1.086	0.005	passed
selectivity[LOLSel].all.argument[2]	0.264	passed	0.573	passed	20.933	12.706	failed
selectivity[LOLSel].all.argument[3]	0.515	passed	0.066	passed	1.392	0.009	passed
selectivity[RFINSel].all.argument[1]	0.358	passed	0.114	passed	1.91	0.061	passed
selectivity[RFINSel].all.argument[2]	0	failed	0.017	failed	-	-	-
selectivity[RFINSel].all.argument[3]	0.582	passed	0.563	passed	2.739	0.042	passed
selectivity[RLOLSel].all.argument[1]	0.041	failed	0.781	passed	1.603	0.082	passed
selectivity[RLOLSel].all.argument[2]	0	failed	0.078	passed	43.551	1.861	passed
selectivity[RLOLSel].all.argument[3]	0.053	passed	0.853	passed	2.576	0.047	passed
YCS[1986]	0.799	passed	0.176	passed	0.546	0.026	passed
YCS[1987]	0.897	passed	0.976	passed	0.982	0.042	passed
YCS[1988]	0.583	passed	0.51	passed	1.24	0.063	passed
YCS[1989]	0.895	passed	0.657	passed	1.108	0.057	passed
YCS[1990]	0.966	passed	0.292	passed	0.953	0.045	passed
YCS[1991]	0.416	passed	0.411	passed	1.006	0.049	passed
YCS[1992]	0.149	passed	0.83	passed	1.055	0.044	passed
YCS[1993]	0.268	passed	0.742	passed	0.975	0.045	passed
YCS[1994]	0.91	passed	0.579	passed	1.056	0.046	passed
YCS[1995]	0.954	passed	0.163	passed	1.179	0.043	passed
YCS[1996]	0.96	passed	0.058	passed	1.215	0.053	passed
YCS[1997]	0.816	passed	0.093	passed	1.185	0.05	passed
YCS[1998]	0.594	passed	0.522	passed	1.171	0.04	passed
YCS[1999]	0.095	passed	0.617	passed	0.931	0.035	passed
YCS[2000]	0.201	passed	0.661	passed	1.204	0.031	passed
YCS[2001]	0.346	passed	0.358	passed	0.881	0.021	passed
YCS[2002]	0.053	passed	0.086	passed	0.943	0.019	passed
YCS[2003]	0.638	passed	0.554	passed	0.95	0.019	passed
YCS[2004]	0.643	passed	0.663	passed	0.643	0.016	passed
YCS[2005]	0.073	passed	0.092	passed	1.064	0.018	passed
YCS[2006]	0.346	passed	0.075	passed	0.819	0.016	passed

YCS[2007]	0.526	passed	0.099	passed	1.61	0.014	passed
YCS[2008]	0.713	passed	0.886	passed	0.93	0.013	passed
YCS[2009]	0.425	passed	0.751	passed	1.038	0.015	passed
YCS[2010]	0.128	passed	0.463	passed	0.848	0.01	passed
YCS[2011]	0.74	passed	0.314	passed	0.388	0.008	passed
YCS[2012]	0.751	passed	0.175	passed	0.663	0.009	passed
YCS[2013]	0.151	passed	0.709	passed	0.944	0.01	passed
YCS[2014]	0.238	passed	0.545	passed	1.122	0.011	passed
YCS[2015]	0.812	passed	0.998	passed	1.238	0.011	passed
YCS[2016]	0.68	passed	0.976	passed	1.032	0.011	passed
YCS[2017]	0.007	failed	0.062	passed	1.374	0.016	passed
YCS[2018]	0.263	passed	0.515	passed	0.701	0.012	passed
YCS[2019]	0.081	passed	0.515	passed	0.626	0.01	passed
YCS[2020]	0.038	failed	0.364	passed	0.534	0.008	passed
YCS[2021]	0.236	passed	0.203	passed	0.754	0.014	passed
YCS[2022]	0.271	passed	0.016	failed	-	-	-
YCS[2023]	0.007	failed	0.113	passed	0.355	0.011	passed

Figure A.19. MCMC autocorrelation lag plots for all estimated parameters (figure 1 of 2).

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

Appendix 3. Objective function contributions

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Table A.3. Contributions to the objective function in the MPD model run.

Objective function components	Values
Observations	
CPUE $_{LLH}$	-16.9
CPUE $_{LLU}$	-33.2
Catch-at-age $_{FIN}$	158.5
Catch-at-age $_{LLH}$	419.1
Catch-at-age $_{LLU}$	547.6
Catch-at-age $_{LOL}$	53.8
Catch-at-age $_{RFIN}$	61.5
Catch-at-age $_{RLOL}$	56.1
Tags 2016	28.1
Tags 2017	33.4
Tags 2018	71.5
Tags 2019	13.8
Tags 2020	8.1
Tags 2021	18.6
Tags 2022	12.7
Tags 2023	8.9
Priors	
SSB_0	10.1
YCS	-27.2
q $_{LLH}$	-10.6
q $_{LLU}$	-10.2
All selectivity priors	0.0
Penalties	
YCS $_{MEAN\_1}$	7.5
All catch limit penalties	0.0
Total objective function	1411.3

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):

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 authorise 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 in response to substantial indications of unfavourable conditions for the stock.

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 authorise test fishing to measure the biological parameters of the stock, subject to close monitoring by the Fisheries Department.