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NAFO Division 4T Yellowtail Flounder (*Myzopsetta ferruginea*): Stock Assessment to 2025 and Rebuilding Plan Scientific Requirements

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Canada

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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TABLE OF CONTENTS

ABSTRACT

Yellowtail Flounder (*Myzopsetta ferruginea*) in Northwest Atlantic Fisheries Organization (NAFO) Division 4T has experienced long-term declines in adult biomass driven primarily by elevated, time-varying natural mortality (M). The stock has been below its biomass limit reference point (LRP) since 1997 and within the Critical Zone of the Precautionary Approach (PA) Framework. Under the amended *Fisheries Act* and Fish Stocks Provisions, Fisheries and Oceans Canada (DFO) is legally required to develop a rebuilding plan for this prescribed major stock. This document updates stock status to 2025 using a new state-space model with fishery-independent and fishery-dependent data. The model and inputs are used to provide scientific guidance for developing the rebuilding plan.

Spawning stock biomass (SSB) has been composed almost entirely of young fish for more than a decade. Biomass of age 5+ fish, used to define reference points as a proxy for commercial-sized individuals (25+ cm), has declined by approximately 90% since the 1980s and was estimated at 32% of the LRP in 2025. Fishing mortality (F) has been consistently low relative to M remaining below 0.05 for ages 4–10 throughout the time series and reaching zero after the fishery closure in 2023. In contrast, M of older, large fish has been persistently high frequently exceeding 1.0 for several older age groups. These temporal patterns in M are consistent with ecosystem-wide increases in size-dependent mortality observed in multiple southern Gulf of St. Lawrence fish stocks which have produced severe truncation of the age and size structure and declines in age- and length-at-maturity.

Given strong non-stationarity in growth, maturity, and mortality, traditional equilibrium-based biological reference points could not be estimated. A new LRP was derived from a proxy for B_{MSY} based on the mean biomass of age 5+ during the productive 1985-1989 period, yielding an LRP of 25,800 tonnes (t) and a corresponding candidate upper stock reference (USR) of 51,600 t. Under this framework, the stock has remained in the Critical Zone for nearly three decades with a 100% probability of being below the LRP in 2025. A removal reference point was derived as the mean F from the years of high biomass (F=0.005), though localized management would be required if a fishery resumed due to the concentration of fishing around the Magdalen Islands and bycatch of other critical stocks.

For analyses, the rebuilding target was defined as achieving at least a 75% probability of being at or above the LRP for five consecutive years. Population projections showed that under current productivity and even in the absence of fishing (F=0), the stock is unlikely to rebuild to the target. Only one scenario, combining historically high recruitment with historically low M, resulted in the stock approaching the LRP over a 15-year projection horizon, and even then with substantial uncertainty.

Additional measurable objectives for the rebuilding plan include:

1. restoring age and size structure by increasing the proportion of age 7+, and
2. incorporating interactions with Winter Flounder (*Pseudopleuronectes americanus*), which is frequently caught as bycatch in the Yellowtail Flounder fishery and also has a rebuilding plan.

Progress toward rebuilding should be assessed during the 5-year stock assessment cycle, with an interim indicator update at the halfway point. The interim indicator will assess stock status based on the multispecies bottom trawl research vessel survey and a scaled LRP.

INTRODUCTION

Yellowtail Flounder (*Myzopsetta ferruginea;* formerly *Limanda ferruginea*; Vinnikov et al. 2018) is a right-eyed flounder found in the northwest Atlantic Ocean, ranging from Chesapeake Bay, USA, to the southern Labrador Shelf, Canada (Scott and Scott 1988). In the southern Gulf of St. Lawrence (sGSL), Yellowtail Flounder (hereafter Yellowtail) is primarily distributed in shallow, nearshore waters, where it historically was harvested in localized fisheries. In NAFO Division 4T (Figure 1), the directed commercial Yellowtail fishery was primarily for bait.

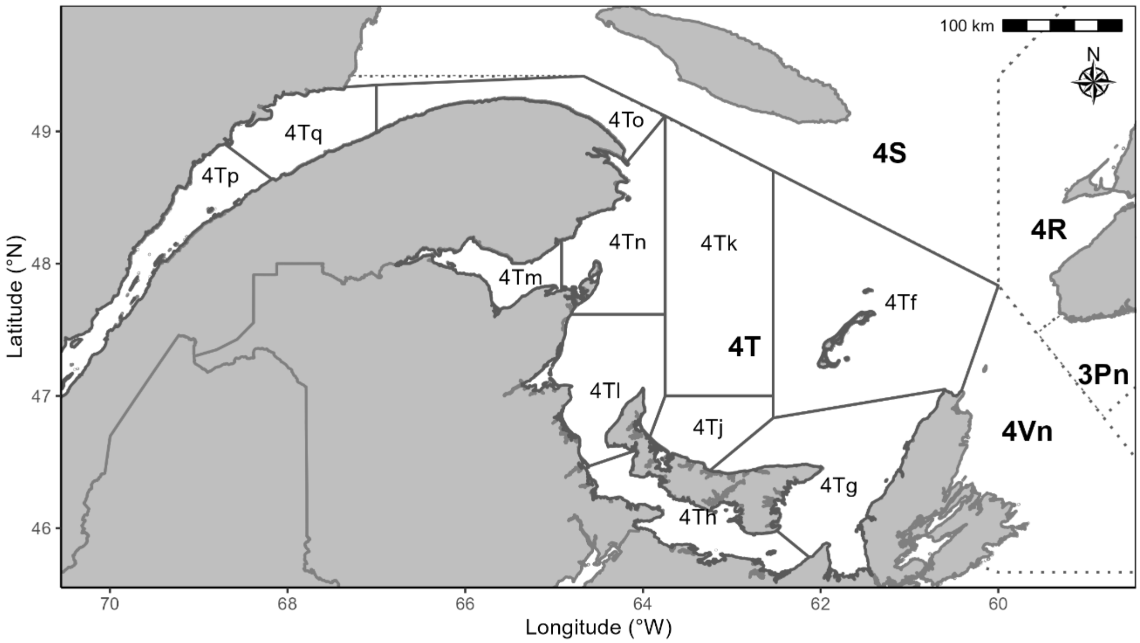

Figure 1: NAFO Divisions (bold font, dotted lines) and NAFO 4T subdivisions (regular font, solid lines) in the southern Gulf of St. Lawrence.

In the most recent Yellowtail stock assessment (Rolland et al. 2022), the Yellowtail biomass was 39% of the biomass limit reference point (LRP) in 2020, and the stock has been below the LRP since 2009. The most recent interim stock status indicator update (DFO 2025) showed that the stock remained below the LRP, in the Critical Zone of the Precautionary Approach Framework (PA), until 2024. Under the Fish Stocks Provisions s.6.2 in the amended *Fisheries Act* (2019) and amended Fishery General Regulations, a rebuilding plan for a prescribed major fish stock must be developed for stocks in the Critical Zone.

This process will provide an updated Yellowtail stock assessment and scientific guidance required for the rebuilding plan. The specific objectives of this document are to:

1. update commercial fishery landings, survey indices, stock assessment model inputs and results to 2025,
2. provide advice on the selection of a rebuilding target,
3. calculate and evaluate the likelihood of achieving the rebuilding target in a specified timeline under various productivity and fishery management scenarios,
4. propose additional measurable objectives,
5. identify indicators for tracking rebuilding progress, and

6. provide guidance on the frequency of the periodic review of the rebuilding plan.

ECOSYSTEM INFORMATION

The biomass of most large-bodied demersal fishes in the sGSL collapsed in the 1980s and early 1990s due to overexploitation, after which fishing effort declined to very low levels (Benoît & Swain 2008; Morin & LeBlanc 2012). Despite this reduction in fishing mortality, these populations have not recovered and, in some cases, have continued to decline. Evidence indicates that the lack of recovery reflects substantial increases in natural mortality among large individuals across the demersal fish community. The changes in size structure occur at the community level, suggesting that the size-dependent mortality patterns estimated for several large-bodied demersal species represent a broader, ecosystem-wide phenomenon (Swain & Benoît 2015).

A major driver of this elevated natural mortality is predation by grey seal (*Halichoerus grypus*), which has been linked to widespread mortality increases among adult groundfish (Benoît & Swain 2008; Swain & Benoît 2015; Neuenhoff et al. 2019; Rossi et al. 2021) and pelagic fish stocks (Benoît & Rail 2016; Turcotte et al. 2021; van Beveren et al. 2024). Grey seal predation risk has also been shown to drive spatial distribution shifts of many sGSL groundfish, with individuals moving into deeper, lower-risk areas as predation pressure increased in traditional habitats (Swain et al. 2015).

Grey seals in the Northwest Atlantic constitute a single population and are assessed as such (Hammill et al. 2023). While there is a herd that breeds in the Gulf of St. Lawrence (GSL) in early winter, grey seals are highly mobile and readily move between the GSL, Scotian Shelf, and Gulf of Maine throughout the year (Breed et al. 2006). The Canadian east coast grey seal population has been rapidly increasing since the 1960s, from 7,800 individuals in 1960 to 360,300 in 2021 (Hammil et al. 2023). While still increasing, a slowing in the rate of increase of the northwest Atlantic grey seal population has been observed in recent years, likely due to density-dependent factors.

FISHERIES

COMMERCIAL FISHERY

Historically, gear mesh size and fishing season were the key management measures on sGSL flatfish stocks, until quota restrictions came into effect in the late 1990s (Clay et al. 1984; Rolland et al. 2022). Following the closure of the 4TVn Atlantic cod fishery in 1993, several management measures were implemented to protect the cod stock and other groundfish species. For Yellowtail, these included a minimum size limit of 25 cm, 10% at-sea observer coverage, 100% dockside monitoring of landings, and limits on bycatch of cod (5% for fixed gear and 10% for mobile gear). Landings spiked in 1997 when a Japanese food market was developed and the fishery was closed in the fall when landings surpassed 800 t. This led to the implementation of a total allowable catch (TAC). The TAC was around 300 t from 1998 to 2015, 225 t from 2016 to 2020, 150 t in 2021 and 2022, and the fishery was closed in 2023 to 2025.

Yellowtail landings in 2021 and 2022 were 148 and 137 t, respectively (Table 1; Figure 2). These landings were slightly higher than the 2016 to 2020 average of 114.5 t. In 2023, only two Yellowtail bycatch events occurred; 10 kg of Yellowtail was caught in September in a 4Tf trawl set, and 192 kg were caught in a 4Tq gillnet set in April. No landings or bycatch were reported from 2024 to 2025. Landings data from 1960 to 1984 are considered unreliable because the species of flatfish caught was often unspecified. Moreover, the landings of all flatfishes were considered to have been underestimated in this early time period (Morin et al. 1998).

In 1991, logbooks became a mandatory licence condition for mobile gear harvesters. Prior to that, harvesters tended not to report the location of capture, including the NAFO subdivision (4T-unspecified; Figure 3). Until the mid-1990s, Yellowtail were caught near the Magdalen Islands (4Tf), off northern Prince Edward Island (PEI) and in Chaleur Bay (4Tj and 4Tm). After 1995, the fishery operated almost exclusively nearshore in the Magdalen Islands. Even within this core fishing area surrounding the Magdalen Islands, catch distribution maps show that the fishery became increasingly inshore over time (Figure 4).

In the 1980s and early 1990s, most Yellowtail landings occurred from August to November, and were predominately taken as bycatch in fisheries directed at American Plaice (*Hippoglossoides platessoides*) and Winter Flounder (*Pseudopleuronectes americanus*) (Figure 3). Since the mid-1990s, however, Yellowtail have increasingly been reported as the targeted species, and most catches occurred earlier, in May to June. This shift coincided with the concentration of the fishery towards the Magdalen Islands. The fishery, which also captures Winter Flounder and Windowpane Flounder (*Scophthalmus aquosus)*, primarily supplies bait for the spring lobster fishery given the limited availability of spring spawning herring in the region.

Trawls and seines are the most common gear types for fishing Yellowtail, but the proportion of landings by gear type has varied considerably through the years. Although the seine fleet was the primary contributor to landings in earlier years, trawlers have accounted for 70 to 80% of landings since 2011 (Figure 3).

BAIT FISHERY

From 2001 to 2012, an exploratory flatfish bait fishery was authorized to supply the Magdalen Islands lobster fishery. This fishery consisted mainly of small lobster boats fishing inshore with otter trawls targeting local stocks of Yellowtail, Winter Flounder and Windowpane Flounder. The bait fishery increased from 5 t of Yellowtail in 2001 to peak at 72 t in 2010 and 62 t in 2011 (DFO 2010). Between 2012 and 2022, the commercial fishery supplied bait in the Magdalen Islands.

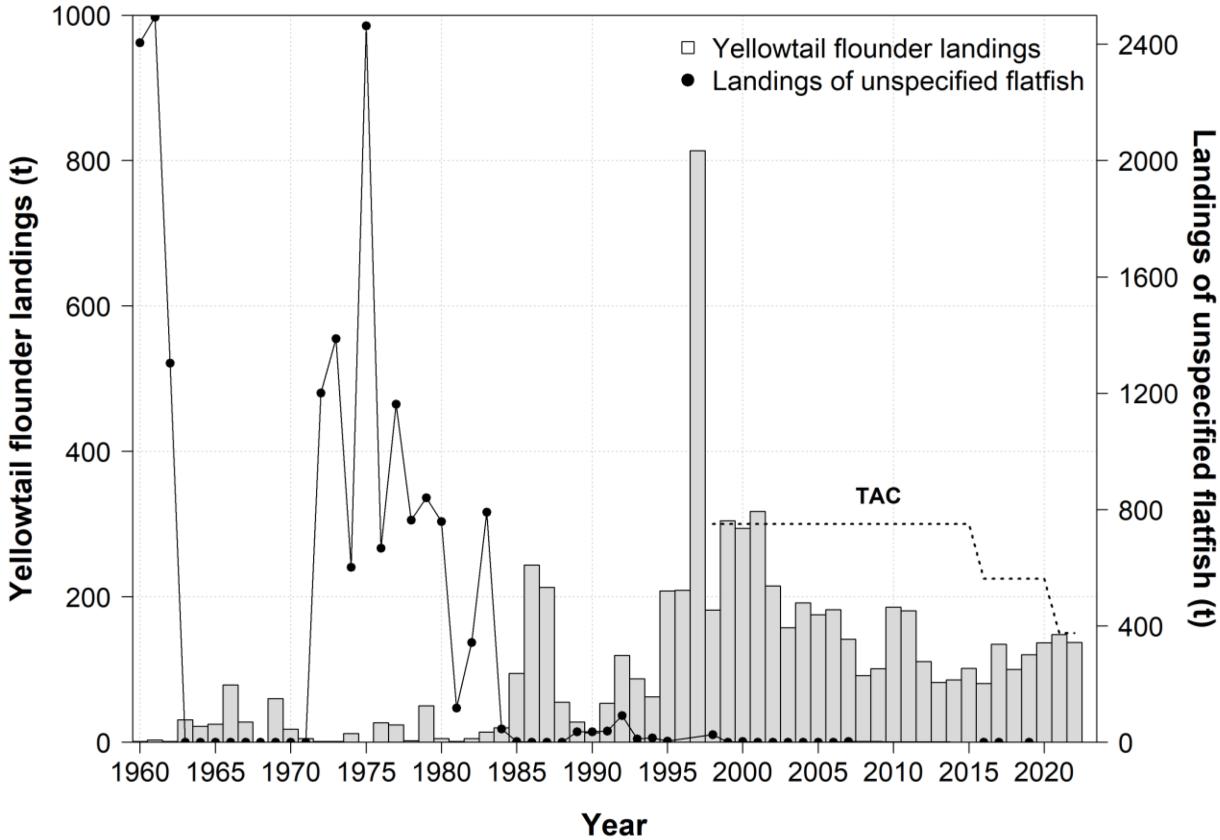

Figure 2: Yellowtail Flounder landings (t) in NAFO Division 4T from 1960 to 2022 (bars). The solid black circles and line correspond to landings of unspecified flatfish. The dotted line indicates the annual Total Allowable Catch (TAC). The fishery was closed after the 2022 season.

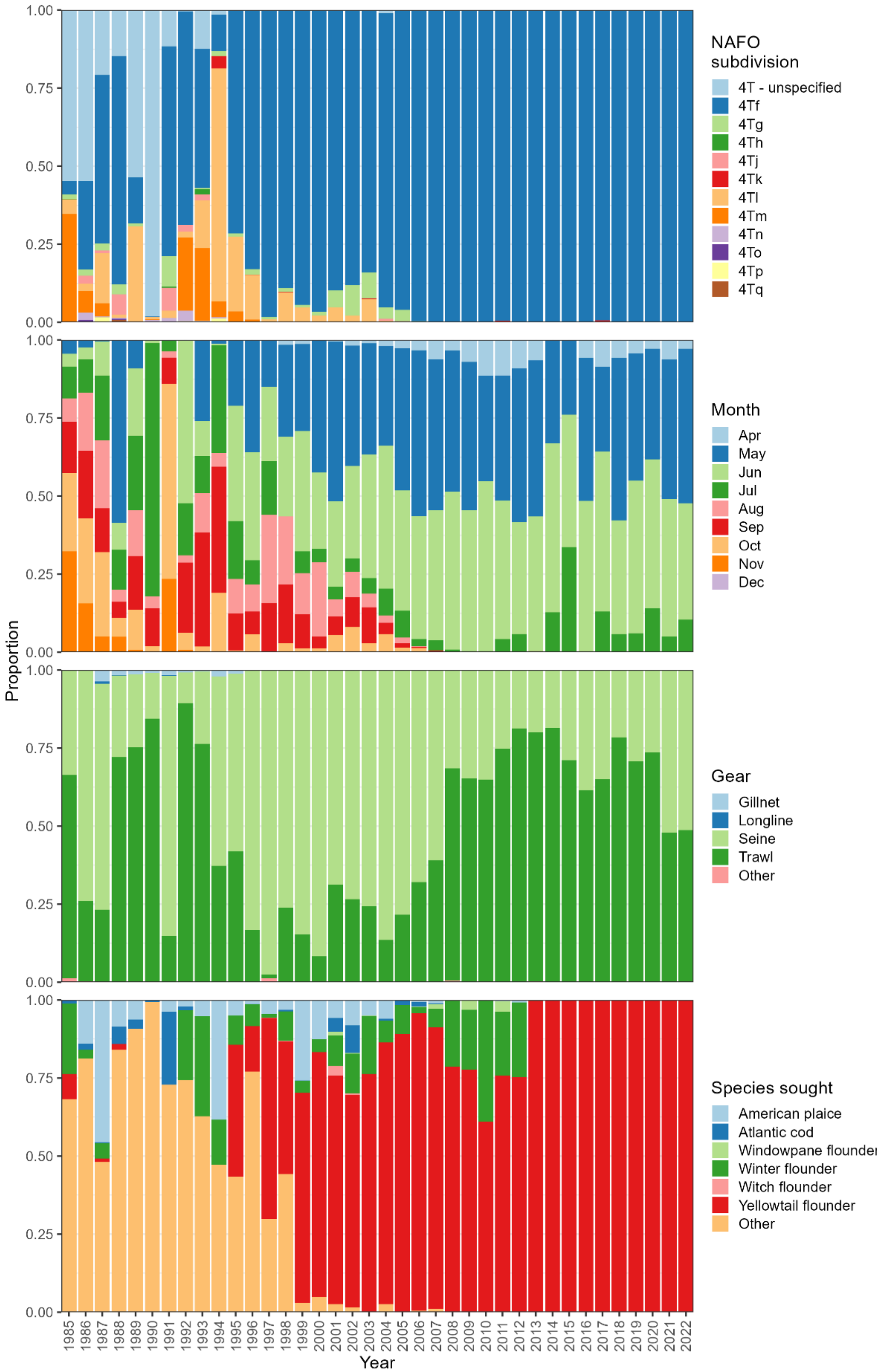

Figure 3: Proportions of Yellowtail Flounder landings from 1985 to 2022 by NAFO 4T subdivision (top panel), fishing month (second panel), type of fishing gear (third panel) and target fishery (bottom panel).

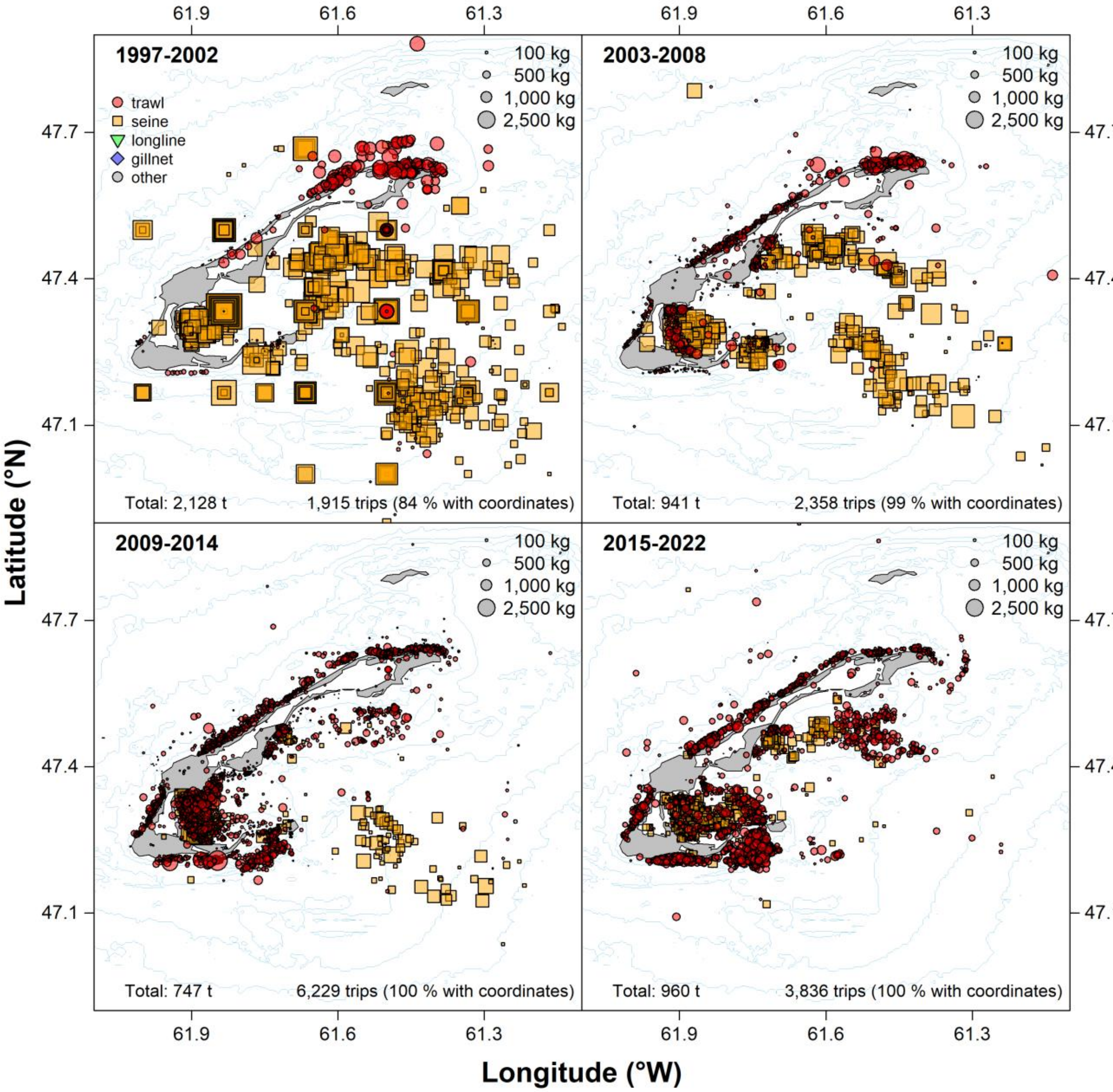

Figure 4: Spatial distribution of catches of Yellowtail Flounder by year and fishing gear type from logbooks.

FISHERY-INDEPENDENT DATA

A multispecies bottom-trawl research vessel survey (RV) of the sGSL has been conducted each September since 1971 (for details see Hurlbut and Clay 1990 and Chadwick et al. 2007). This survey uses a stratified random design, with stratification based on depth and geographic region (Figure 5). Three inshore strata (401-403) were added to the survey in 1984, but they are not included in calculation of indices to preserve the full time series. A number of vessel and gear changes have occurred over the timespan of the survey, with *CCGS Teleost* using a Western IIA trawl being the reference survey platform for analyses. Conversion factors were applied to the RV survey data so that all captures were adjusted to a standard 1.75 nautical miles daytime tow on *CCGS Teleost* using a Western IIA trawl. The conversion factors used are those estimated by Benoît and Swain 2003 and Benoît and Yin 2023.

Each Yellowtail catch was weighed and a subsample of the catch (up to 200 fish) was measured for length, maturity, sex and individual weight. Length-stratified otolith samples were collected for fish ageing, though none were collected from 1987-1999.

Given the concentration of the fishery around the Magdalen Islands (strata 428, 434-436; Figure 6), abundance and biomass indices were generated both at the sGSL and at the Magalen islands area scale.

The spatial distribution of standardized Yellowtail catches (kg per tow) from the September survey was interpolated using inverse distance weighting (IDW). In the sGSL, Yellowtail are distributed along coastal to mid-shore ecosystems (Figure 7). They are distributed in and around the west, northern and sometimes eastern part of PEI, around the Shediac Valley, on the eastern part of the Acadian Peninsula, in St. George's Bay and around the Magdalen Islands.

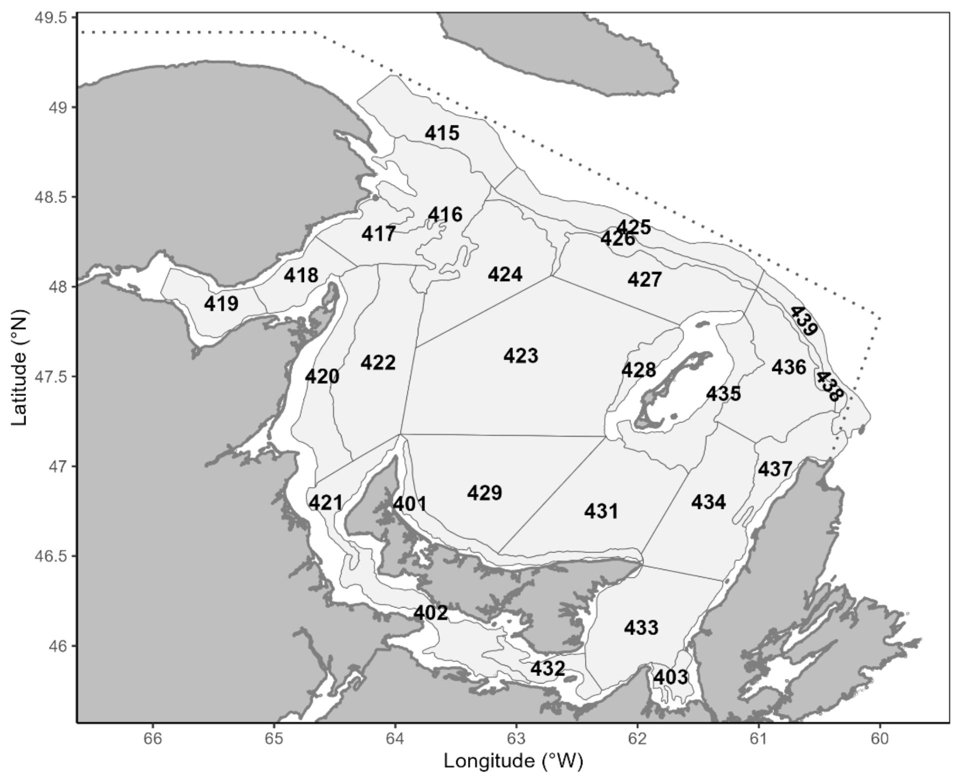

Figure 5: Stratification scheme used for the southern Gulf of St. Lawrence multispecies bottom-trawl research vessel survey. The dotted line indicates the boundary of NAFO Division 4T.

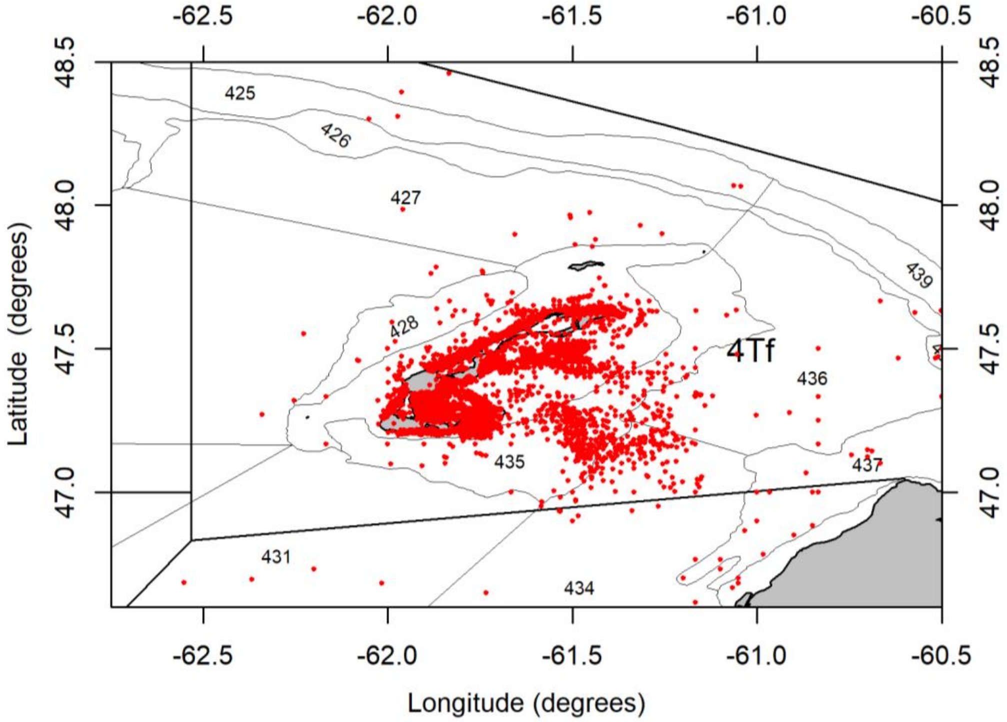

Figure 6: Strata of the southern Gulf of St. Lawrence multispecies bottom-trawl research vessel survey that overlap with the Magdalen Island Yellowtail Flounder fishery (red points).

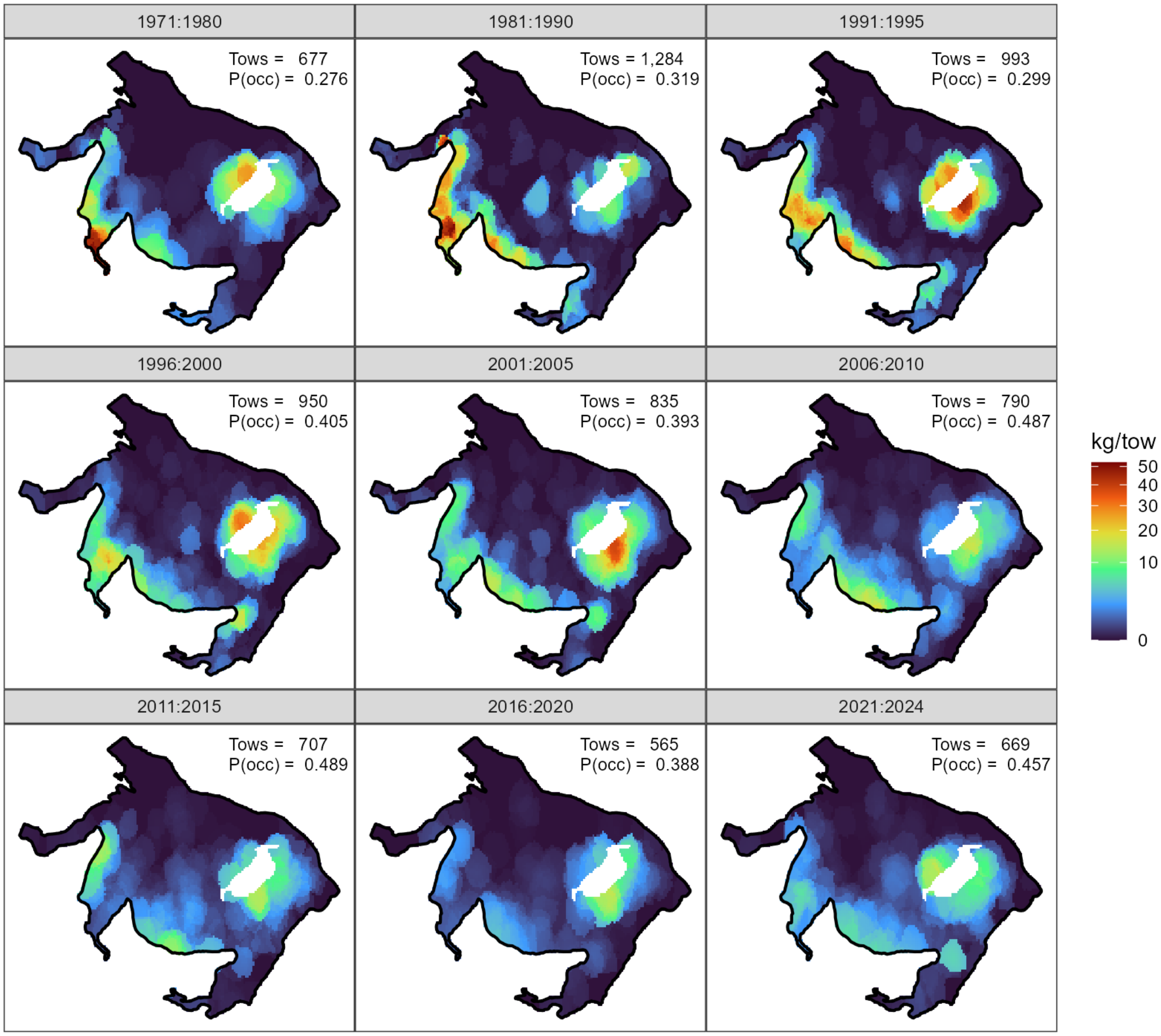

Figure 7: Spatial distribution of September bottom trawl survey catches (in kg per standard tow) of Yellowtail Flounder, 1971 to 2024.

CATCH AT LENGTH

Yellowtail abundance (number per tow) spanning the full sGSL increased from 1971 to 1980, reaching a peak of approximately 30 fish per tow, then decreased through 1984 (Figure 7). From 1985 to 2025, abundance remained relatively stable at approximately 20 fish per tow. Overall, fish <25 cm increased from 1985 to 2014, dropped in 2015, but subsequently increased to levels comparable to the late 2000s. In contrast, the abundance and biomass of fish 25+ cm averaged around 10.5 fish per tow and 2.25 kg per tow prior to 2001 when it began decreasing to reach a sustained low level from 2010 to 2025 (2 fish per tow and.0.3 kg per tow. Biomass at all lengths showed a long term decreasing trend after the 1980s, from about 5 kg per tow to 1.5 kg per tow in recent years.

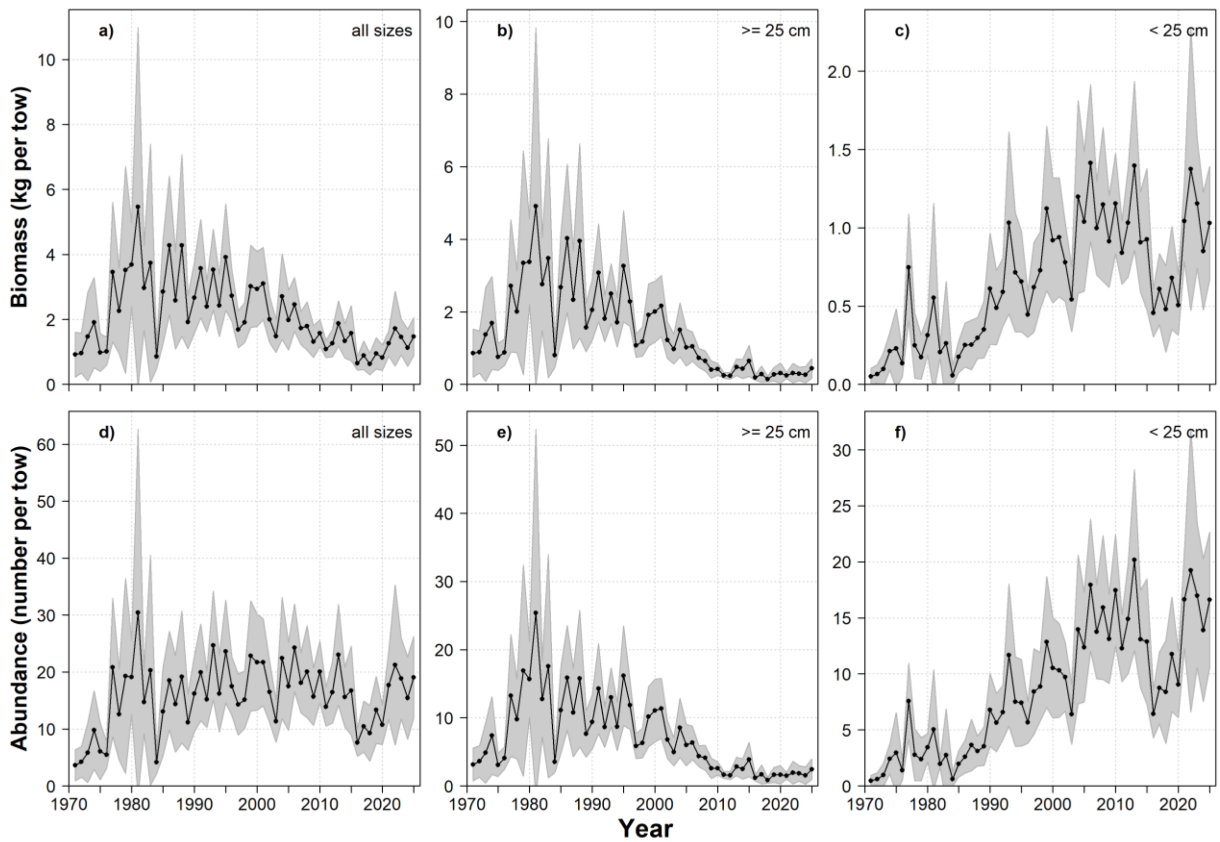

Figure 8: Biomass (kg per tow, top panels) and abundance (number per tow, bottom panels) indices of Yellowtail Flounder from the southern Gulf of St. Lawrence (sGSL) multispecies bottom-trawl research vessel survey at all lengths (left), for fish 25+ cm (middle) and for fish <25 cm (right). Points and lines are the stratified means and the shaded area represents the 95% confidence intervals.

Trends in the biomass and abundance for the Magdalen Islands were broadly similar to those for the entire sGSL across all size classes (Figure 9), however Magdalen Islands catches were generally much greater and displayed more variability compared to the full sGSL. Catches of fish 25+ cm in the Magdalen Islands ranged from 19 fish per tow and 3.9 kg per tow prior to 2001 to reach lows of 6 fish per tow and 1.0 kg/tow since 2020. The decline of fish 25+ cm occurred more rapidly in the Magdalen Islands over the course of roughly 6 years rather than the more gradual decrease over 10-15 years observed across the sGSL. The collapse of these large fish also began earlier in the time series in the sGSL (mid-1990s compared to early-2000s in the Magdalen Islands). Both areas, however, show a roughly 90% collapse in the biomass of the 25+ cm fish when calculated from the 5 highest and lowest consecutive year periods. Due to the greater overall abundance in the Magdalen Islands there continues to be a reduced but sustained abundance.

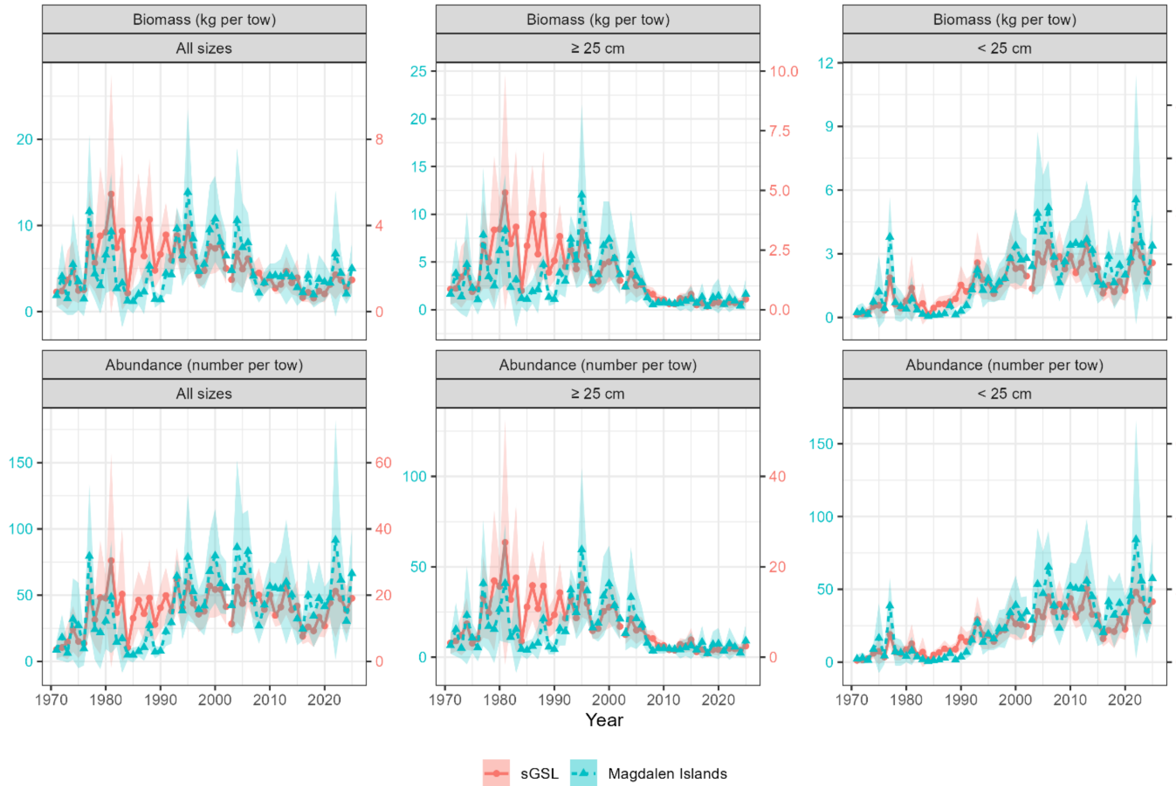

Figure 9: Biomass (kg per tow, top panels) and abundance (number per tow, bottom panels) indices of Yellowtail Flounder from the southern Gulf of St. Lawrence (sGSL) multispecies bottom-trawl research vessel survey at all lengths (left), for fish 25+ cm (middle) and for fish <25 cm (right). Points and lines are the stratified means and the shaded area represents the 95% confidence intervals. Pink circles and solid lines represent the entire sGSL (strata 415 to 439) while blue triangles and dashed lines are specific to the Magdalen Islands (strata 428 and 434 to 436). Numbers to the right of each panel indicate sGSL values, while numbers to the left of each panel indicate Magdalen Islands values.

The stratified mean numbers at length showed a decreasing trend in Yellowtail size in the RV survey for both the Magdalen Islands and the entire sGSL (Figure 10). The decline in size of Yellowtail started in the early 1990s when the modal length was around 28 cm, and gradually continued until 2010 where the modal length was slightly above 20 cm and the abundance of fish 25+ cm was low. The modal length and the abundance of fish 25+ cm in the catch remained low after 2010.

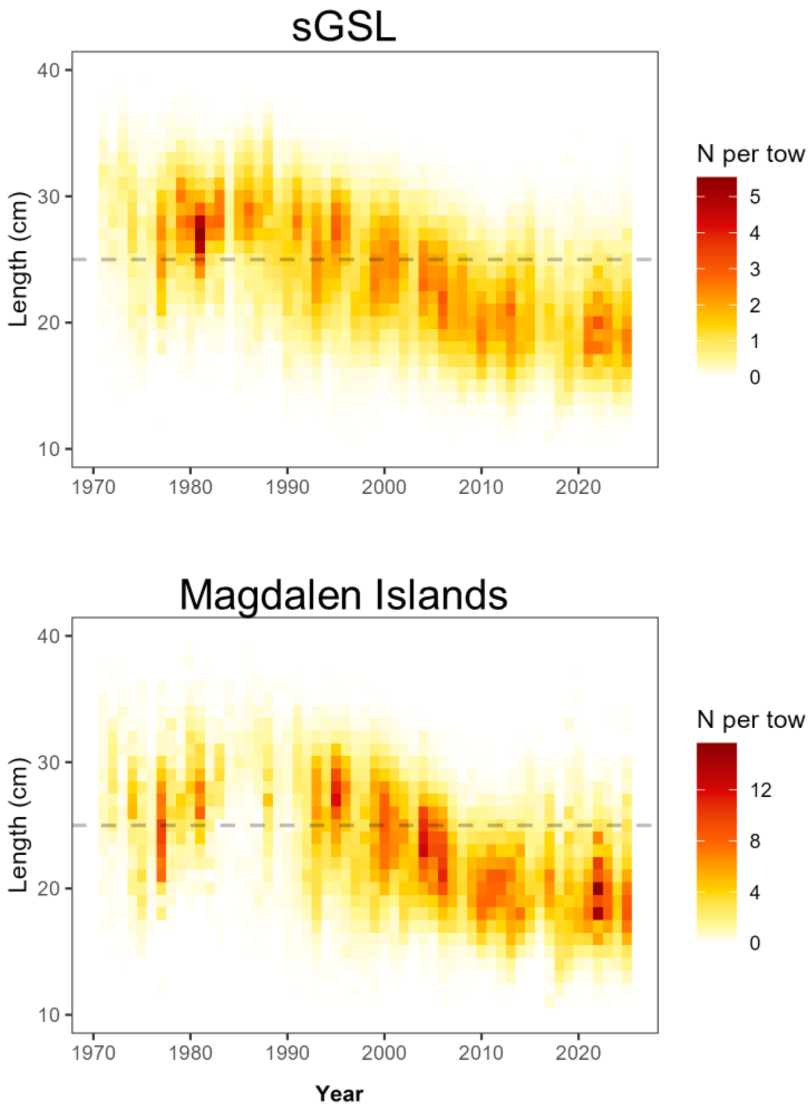

Figure 10: Stratified mean numbers at length (cm) of Yellowtail flounder per tow southern Gulf of St. Lawrence (sGSL) multispecies bottom-trawl research vessel survey for the entire sGSL (strata 415 to 439; top panel) and the Magdalen Islands (strata 428 and 434 to 436).

AGEING AND GROWTH

Age-length keys (ALKs) were developed using RV survey and commercial fisheries biological data for Yellowtail collected between 1978 and 2025 (Appendix A). ALKs were first estimated separately by year and source (survey or fishery) and ager to ensure that age-length relationships were internally consistent within each stratum, following Ogle 2016.

Total length was discretized into 1 cm length bins. Frequencies of age by length category were tabulated and converted to row-wise proportions, yielding estimates of the conditional probability of age given length. For each unaged fish, an age was randomly assigned based on the multinomial probability distribution defined by the corresponding length bin in the ALK. To produce smoothed ALKs, a multinomial logistic regression model was fitted to the aged data using the multinom() function from the nnet package (Gerritsen et al. 2006; Ogle 2016). Age was modelled as a categorical response variable with length category as the predictor.

Predicted age probabilities were generated at 1 cm length intervals across the observed length range.

As in previous assessments, substantial discrepancies were observed among ALKs produced by different agers despite development of a robust protocol and a reference collection for Yellowtail. Previous assessments used ages derived from a single ager, but from the post-1999 period and for only 5 years (Rolland et al. 2022). Here, the sample size was increased where otoliths were available and adequately preserved for age reading. Two agers performed additional age reading, yet within-ager still exhibited the greatest internal consistency. Consequently, one ager was again selected. Due to the low sample size for the 1970s, 1980s and 1990s, decadal ALKs were constructed. Otoliths collected in the 1970s and 1980s were challenging to read, thus, otoliths for this time period were combined and used for the 1980s decade. Given otoliths were not collected from the RV survey in 1987-1999, commercial samples were used to supplement the ALKs for this time period.

CATCH AT AGE

Annual mean weight-at-age vectors were calculated using the decadal ALKs. Missing weight-at-age values were predicted using a Gompertz growth model fit to age and weight data over the whole time series. The Gompertz growth function was as follows:

$$W_t = W_0\left[e^{G(1-e^{-kt})}\right]$$

where W_t is weight and W_0, G and k are model parameters.

Weight-at-age declined across ages 4+ from the late 1980s into the mid-2000s, then stabilized at lower levels through the early 2010s (Figure 11). Growth at ages 6+ increased for a few years in the 2010s. Growth increased in recent years for fish ages 6+. The sample size at old ages decreased with time as older individuals were less caught by the survey, due to their low abundance.

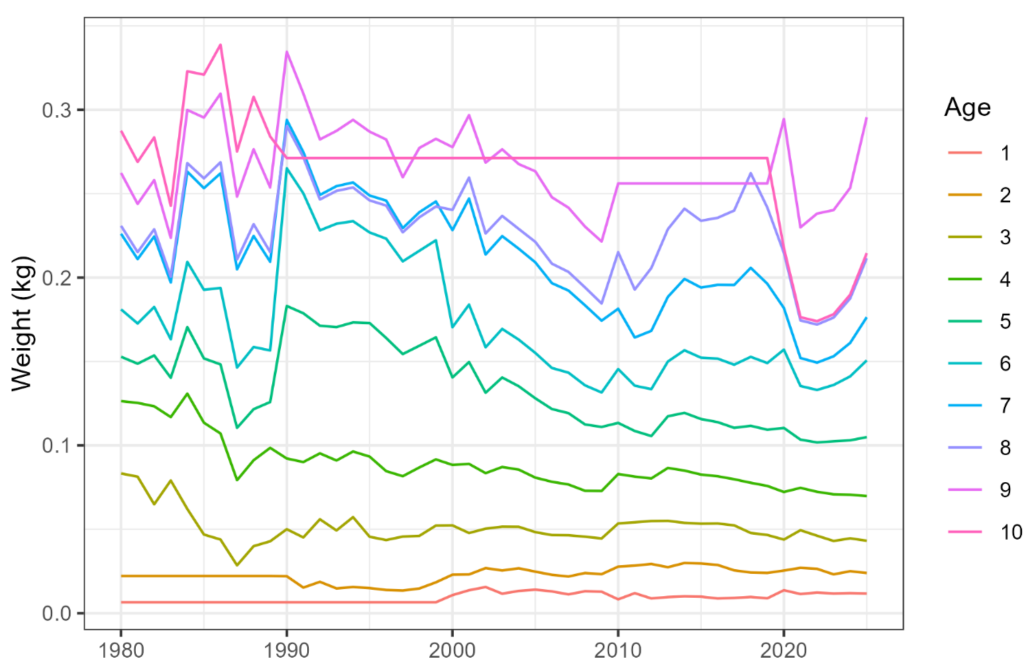

Figure 11: Annual mean weight-at-age of Yellowtail Flounder caught in the southern Gulf of St. Lawrence research vessel (RV) survey between 1980 and 2025.

Annual values of trawlable abundance at age in the RV survey catch were calculated using decadal ALKs. Using annual length-weight relationships, the total weight of the sampled fish was calculated (sample weight). The sampling ratio was calculated as the sample weight divided by the catch weight. The length frequencies were scaled to the catch by multiplying them by the sampling ratio. The ALKs were applied to the annual scaled length frequency to obtain the trawlable numbers at age. The annual stratified mean trawlable abundance was then used to obtain annual stratified mean proportions at age of trawlable abundance. Annual age-aggregated biomass indices were calculated by doing the product of numbers at age and weight-at-age and summing over ages. Annual catch CVs were calculated by year across fishing sets.

Numbers at age showed strong interannual variability (Figure 12). Older fish (ages 7+) were common in the 1980s RV survey but were rarely caught and in low numbers after the 1980s. Fish of ages 3 to 5 consistently dominated the catches since 1990. There was little cohort signals in the age composition. Younger fish (ages 1) appeared regularly but at much lower abundances throughout the series, while ages 2 increased in the catch after 2010 and age 3 increased in the catch after 1990.

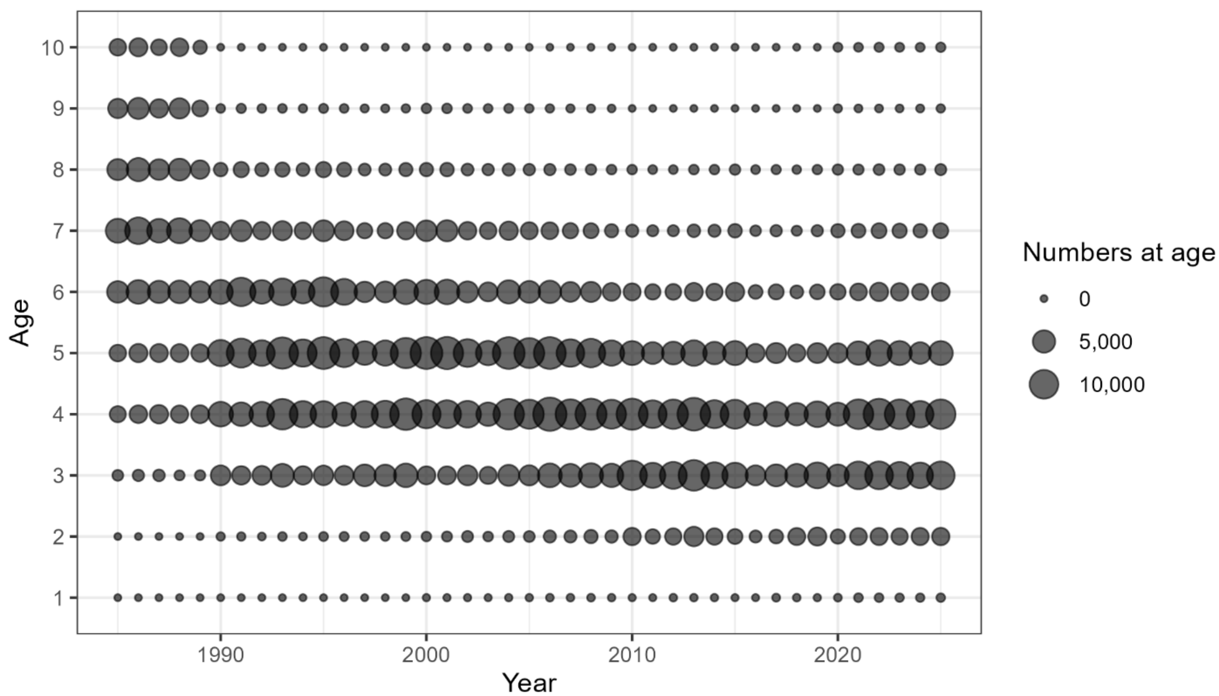

Figure 12: Numbers at age of Yellowtail Flounder caught in the southern Gulf of St. Lawrence research vessel (RV) survey between 1985 and 2025.

MATURITY OGIVES

Annual trends of length-at-maturity were calculated over the RV survey time series. Sampled fish were classified into various maturity stages as per Rolland et al. (2022). There was a strong decrease in the length-at-maturity for both sexes over the study period (Figure 13). The length at which 50% of fish were mature averaged 23.6 cm and 22.2 cm respectively for females and males during the first 5 years (1971-1975), but decreased to 14.4 cm and 13.5 cm for the last 5 years (2021-2025). Age-at-maturity ogives were estimated by decade, due to gaps in the data. Annual ogives were also calculated using annual maturity and annual ages derived from lengths using the decadal age length keys, with similar results. As with length-at-maturity, the age at which 50% of fish were considered mature decreased over time, from 3.7 in the 1970s to 2.1 in the 2020s (Figure 14). There was no difference in age-at-maturity between sexes.

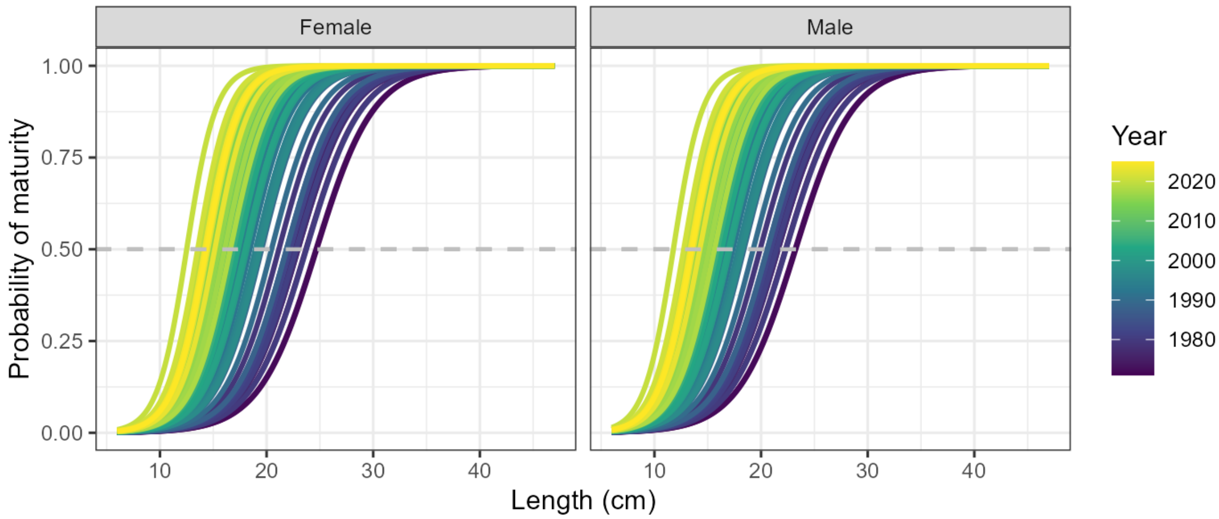

Figure 13: Probability of maturity by length (cm), sex, and year for Yellowtail Flounder, estimated from September bottom trawl survey biological data, 1971 to 2025. Note that year 1985 was omitted from the figure.

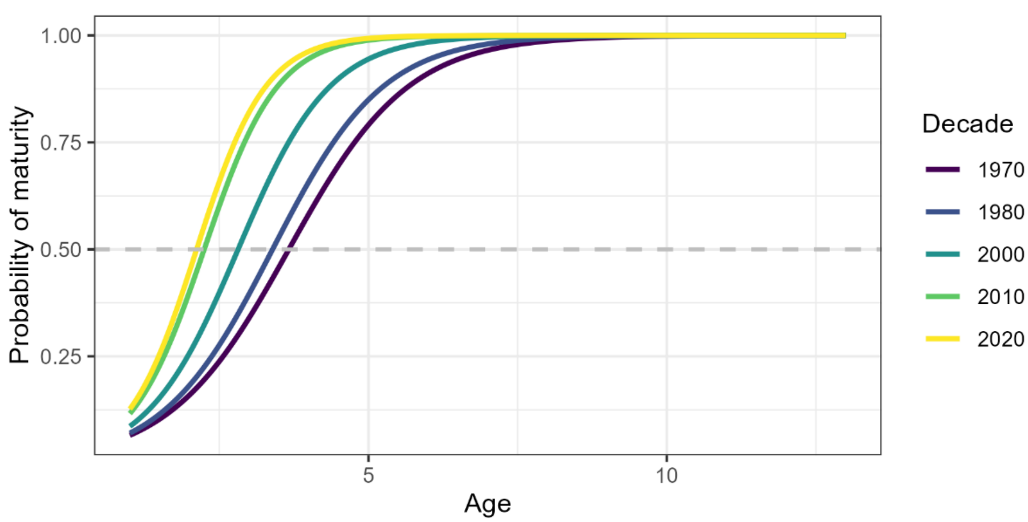

Figure 14: Probability of maturity by age and decade for Yellowtail Flounder, estimated from September bottom trawl survey biological data, 1971 to 2025.

TOTAL MORTALITY

Estimates of the instantaneous rate of total mortality (Z) were derived from the catch rates at age in the RV survey. Estimates were obtained using catch curves as in Sinclair (2001), where Z is the slope of the descending limb of the catch curve. The mean Z was estimated in moving windows of 7 years with the log catch rate at ages 4 to 10 as the dependent variable, age as the covariate and year-class included as a factor (to control for variation in year-class strength).

Z increased continuously between the late 1980s and 2009, peaking at values greater than 1.0 (Figure 15). Z remained high but decreased slightly between 2016 and 2019. Those dynamics included changes in fishing mortality (F), which increased gradually between 1990 and 2000, but remained relatively stable after 2000 (see population model results). Moreover, estimated F was of a much lower magnitude. Hence, changes in Z are assumed to be largely driven by changes in natural mortality (M). These trends in estimated Z from the DFO survey and the relative magnitude of Z in periods of low F (around 2010) were highly similar to Georges Bank (NAFO 5Z) Yellowtail (DFO 2019).

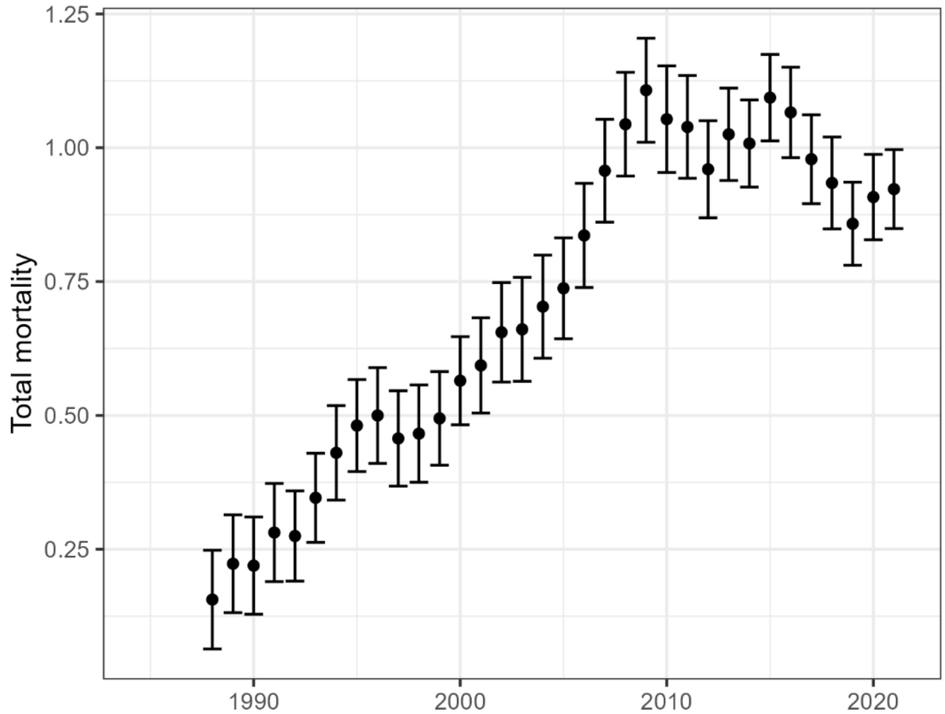

Figure 15: Total mortality estimated from the RV survey catch curves of Yellowtail Flounder.

FISHERY-DEPENDENT DATA

CATCH AT LENGTH

Annual catch at length estimates for Yellowtail in NAFO 4T were generated from pooled port and at-sea observer samples from the commercial fishery gathered from 1985 to 2022. Prior to calculating the catch at length, individual commercial length-frequency samples were first scaled to the level of the total fishing trip catch. For at-sea observer samples, samples were first scaled to the level of each individual fishing activity, summed, and then scaled to the level of the total catch for the fishing trip. Sample-weights were calculated from length-frequency values and length-weight regression parameters estimated from the RV survey from each corresponding year. These conversions were sex-specific when sex was recorded in the samples. Samples were separated by fishing gear type, namely trawl and seine which represent the majority of landings over the 1985 to 2022 period. Samples for all months and all geographic locations within the sGSL were pooled to generate annual estimates. Samples were lacking in some years, thus the relative length-distribution was assumed to be an average of adjacent years. For 1997, an average of 1996 and 1998 was used and for 1999, 2000, and 2001 averages of 1998, 2002, and 2003 were used. The fishery proportions at age in years where length frequencies were averages of adjacent years were not used in the state-space stock assessment model.

Total catch at length estimates were obtained by scaling available samples to the level of sGSL landings by fishing gear type, then summing over gear types.

The length distributions between trawlers and seiners were generally similar prior to 2010 (Figure 16), with the exception of 2002 (possibly be due to the small sample size). A large increase in catch of smaller fish occurred between 2009 and 2012, more importantly in trawl catches. The proportion of fish <25 cm in the catch was approximately 80% in 2010, 2011 and 2012, then decreased to 35 to 40% in 2013 to 2022 (Figure 17). Trawlers generally landed more small fish than seiners in all years post-1998. With few exceptions, catch at length distributions were all unimodal, with a modal shift from 31 cm in the late 1980s to 28-29 cm in the late 1990s, to a mode of 25 cm in the late 2000s.

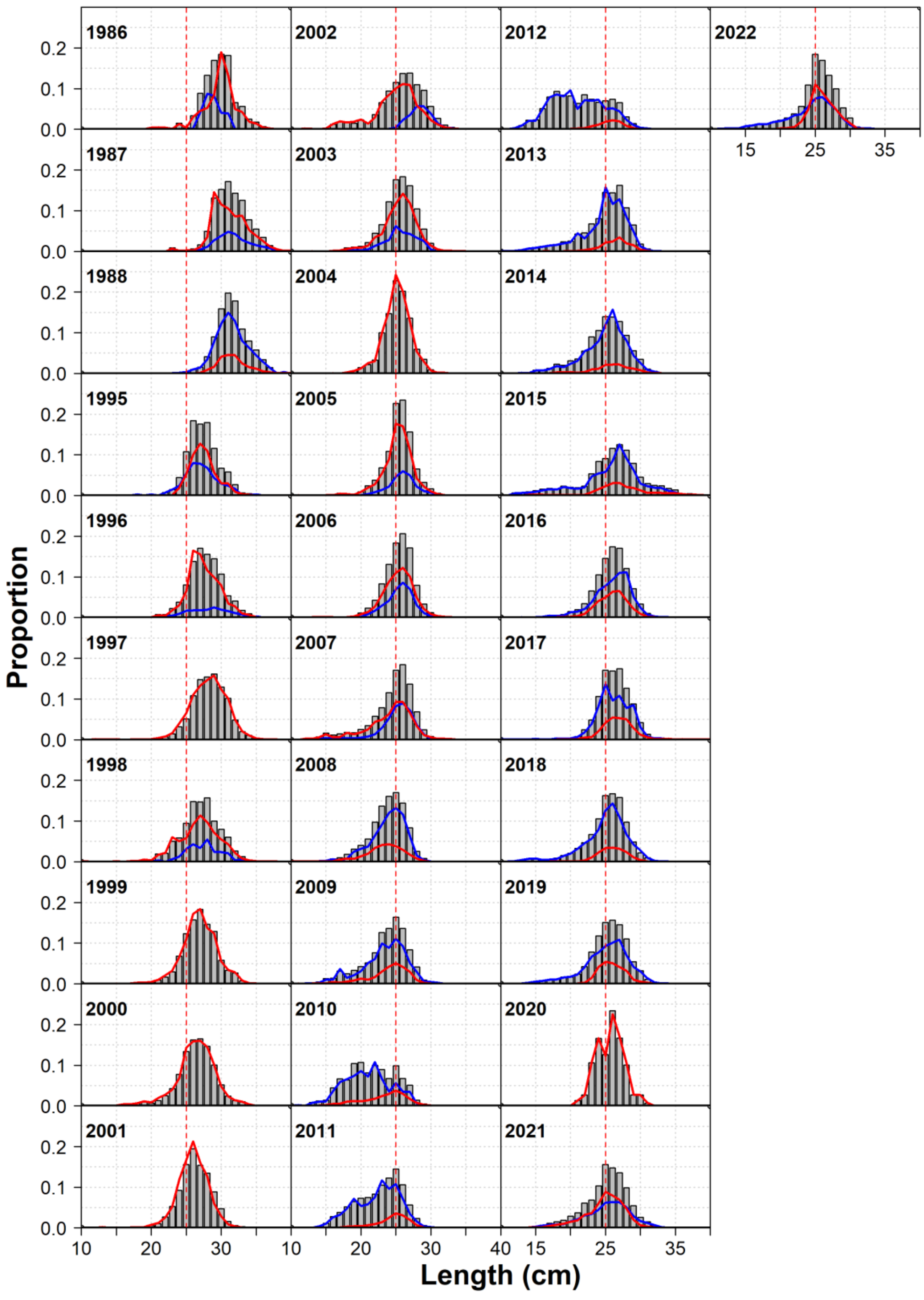

Figure 16: Proportions-at-length of Yellowtail Flounder catches based on commercial and observer samples. The vertical red dashed lines correspond to the 25 cm legal size limit. Overlaid solid lines indicate portions of the total proportions represented by trawler (blue) and seiner (red) catches. Note that for certain years, no trawl samples were available (e.g. 1999 to 2001).

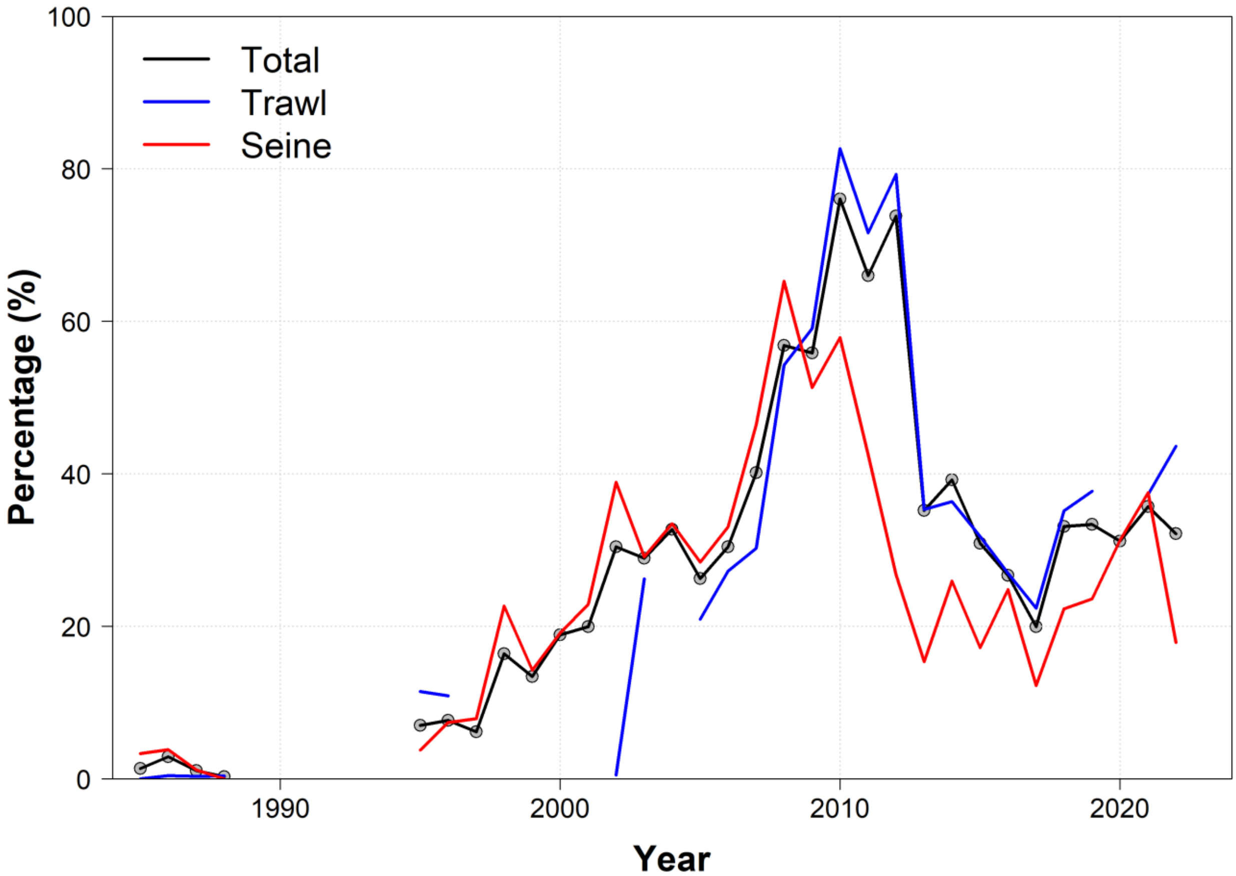

Figure 17: Annual percentages of Yellowtail Flounder catches consisting of fish under 25 cm, based on catch at length estimates from trawl catches (blue line), seine catches (red line), and for gears combined (black line and points).

CATCH AT AGE

The age composition of fishery catches was broadly similar between the trawl and seine fisheries, with both gears dominated by ages 4 to 7 (Figure 18). In the trawl fishery, ages 4 and 5 contributed the largest proportions in most years, with older fish (ages 8+), appearing predominately in the earlier part of the time series. Seine catches were largely composed of ages 4 to 6, but proportions of younger fish (ages 3-4) tended to be slightly higher and older ages (ages 8+) appeared less frequently and at lower levels than in trawl catches.

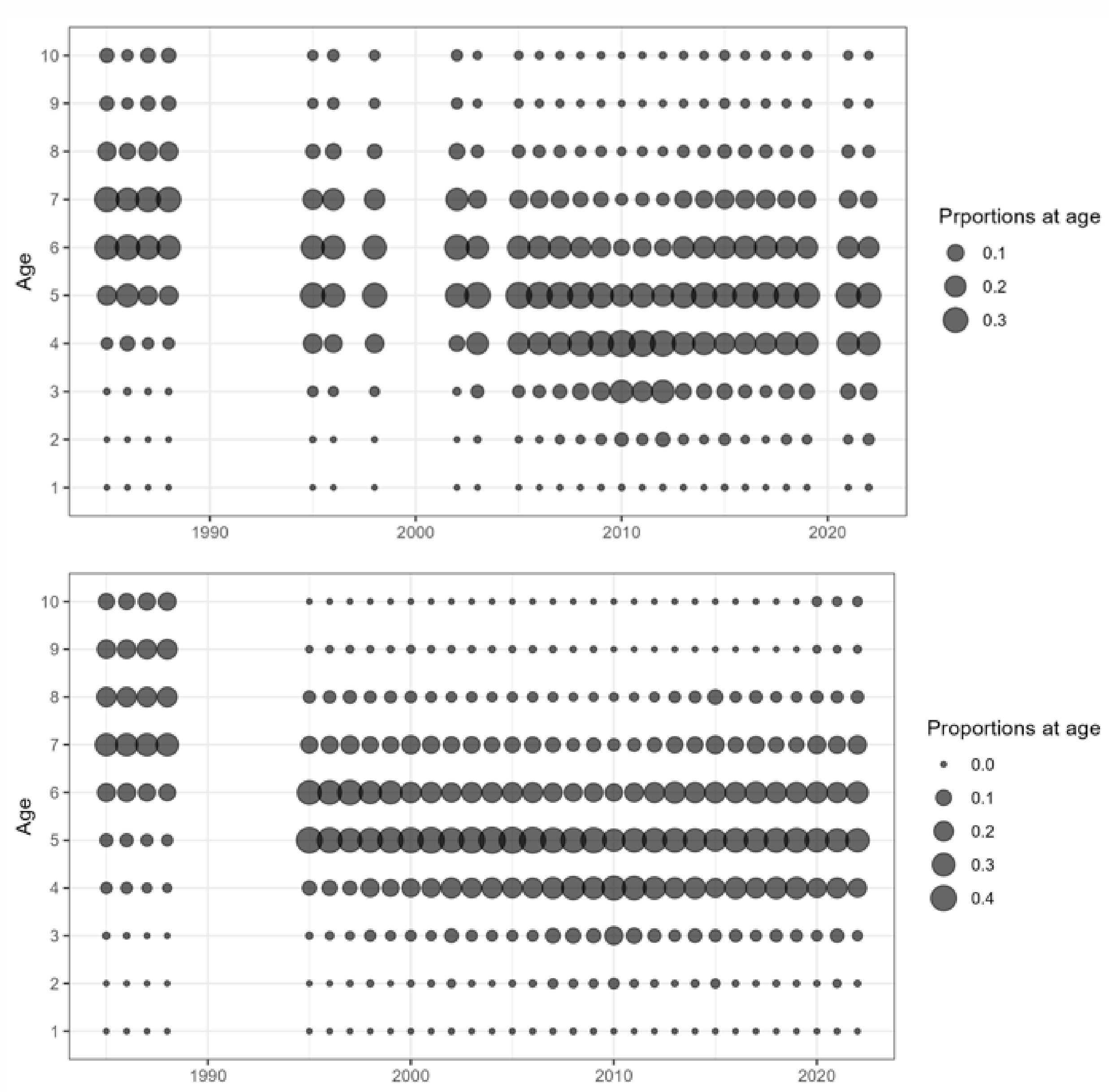

Figure 18: Proportions at age of Yellowtail Flounder caught in the trawl fishery (top panel) and seine fishery (bottom panel).

CATCH-PER-UNIT-EFFORT

The previous assessment presented catch-per-unit-effort (CPUE) indices for both the seine and trawl Yellowtail fisheries for the period 1985–2020 (Rolland et al. 2022). For the present assessment, the CPUE time series was updated to 2022. Because CPUE cannot be estimated when the fishery is closed, CPUE trends were compared with RV survey catch rates to assess whether both indicators reflected similar changes in Yellowtail biomass in the Magdalen Islands area. This comparison was particularly important because commercial-sized fish (≥25 cm) exhibited a more rapid decline in the fished area around the Magdalen Islands than across the sGSL as a whole.

Scaled CPUE values for each gear and scaled biomass catch rates from the Magdalen Islands RV survey strata were compared. There was substantial overlap and similarity in the trends in catch for both CPUEs and the RV survey around the Magdalen Islands (Figure 19). CPUE of the trawl fishery are nearly identical to the Magdalen Islands RV survey strata catch rates, whereas the CPUE of the seine fishery is shifted slightly to the right. Given the significant overlap, this suggests that the Magdalen Islands RV survey strata is likely or reliable source of additional information to monitor the Yellowtail dynamics in this local area, even if the fishery occurred more inshore than the RV survey coverage.

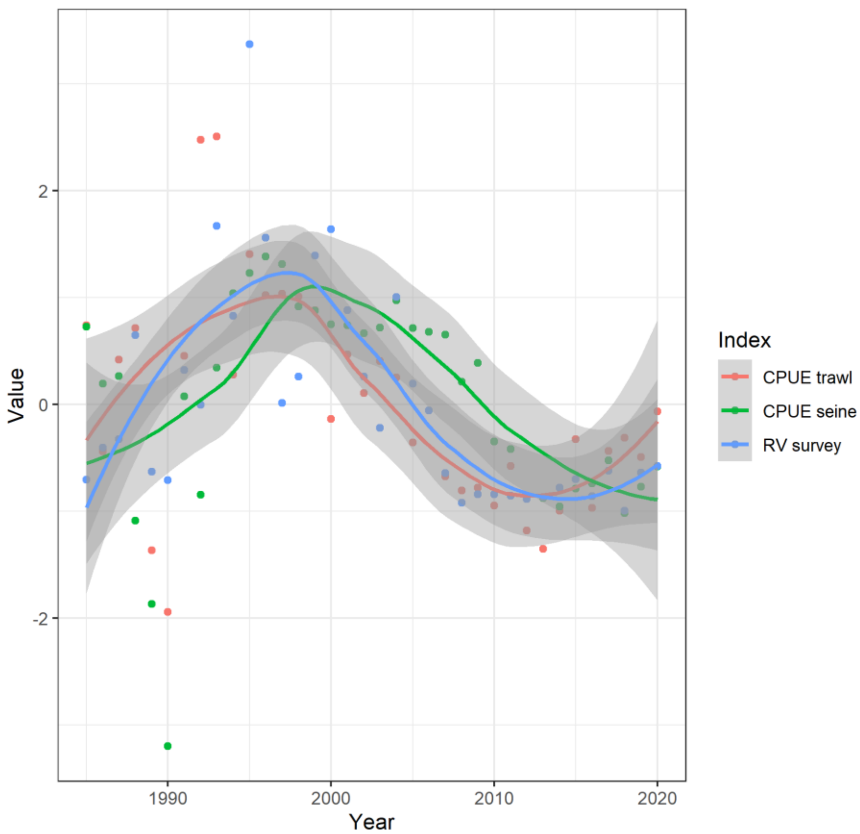

Figure 19: Scaled catch rates of Yellowtail Flounder from the commercial CPUE by fishing gear (trawl and seine) and the southern Gulf of St. Lawrence multispecies bottom-trawl research vessel survey.

STOCK ASSESSMENT MODEL

FORMER LENGTH BASED MODEL

Assessing the status of this stock is challenging due to limited data and weak fishery signals. Fishery catch is relatively low throughout most of the modelled period, with the exception of 1997, which makes estimation of population scale difficult. In addition, ageing data are limited. As a result, the model is driven primarily by the increase in the abundance and biomass of Yellowtail <25 cm, concurrent with the decline in the abundance and biomass of individuals ≥25 cm. To account for these dynamics, a length-based age-structured model was developed and applied in the 2015 (Surette and Swain 2016) and 2020 (Rolland et al. 2022) stock assessments. These models were implemented in AD Model Builder (Fournier et al. 2012) and fit using penalized maximum likelihood.

The model used total landings, annual abundance indices from the RV survey by length aggregated groups (<25 cm and ≥25 cm), the proportion of the annual landings by the two length groups (in numbers of fish), the average weight of individuals in each of the length groups for the annual fishery catch and the annual RV survey indices, the annual mean weight-at-age, and the annual vector of proportion mature at age based on length-at-age data for 2000, 2007 and 2015.

Abundance at age in the model was mapped into the two length groups used for the survey indices and catch proportions based on length-at-age data for 2000, 2007 and 2015. Based on these data, fish aged 1-3 years were assigned to the <25 cm length group, those aged 5 years and older were assigned to the ≥25 cm length group, and half the 4 year olds were assigned to each length group.

Time varying M of two age groups (1-3 and 5-8+) was estimated separately for five time blocks (6-year blocks between 1985 and 2008 and a final 7-year block). M of 4-year olds was calculated as the average of M of both age groups in a given year. A prior for M was also used for the first time block (1985 to 1990). Prior means were 0.8 and 0.3 for initial M's for ages 1-3 and 5+, respectively (with standard deviation (SD)=0.05 or 0.025, respectively). In both assessments, M for the older age group was at the upper bound of the parameter in the terminal year block (M=2.0). The population scale was also sensitive to the number of years in time blocks.

In initial trials, RV survey catchability (q) was estimated to be 1.4, with the survey indices at the scale of trawlable abundance. For small flatfish like Yellowtail, such a high value of q would only be possible if sampling locations in the RV survey were strongly biased towards areas where high densities of Yellowtail occurred and these high catch rates were extrapolated to areas where Yellowtail densities are actually low. Given the distributions of Yellowtail and location of RV sampling sites, this is not a plausible hypothesis. Thus, an informative prior for q was used in the model based on estimates of catchability of American Plaice to the RV survey trawls obtained by Harley and Myers (2001). For a flatfish of 30-35 cm in length, their estimate for q was about 0.3. A prior for log(q) with a mean of -1.14 (q=0.32) and SD=0.1 was used in the model in 2015. In 2020, a prior with a mean of -0.3566 on the log scale (q=0.7) and SD=0.15 was used.

The length based model was updated to 2024 with q = 0.3 and SD = 0.25 (see below). Selected results are presented in Appendix B. Briefly, the model estimates of important quantities for the assessment and management of the stock (SSB, F) were consistent with previous assessments. However, as was also the case in previous assessments, the M estimation was problematic, according to two main diagnostics. First, the last two year blocks of M estimates for ages 5-8+ were at the upper bound (M=2), which suggest that M would likely be higher if it was not limited. Second, the M estimation is using penalized likelihoods to constrain the deviations in the random walks. Without these penalties and priors for the mean, M for the age group 1-3 cannot be estimated, suggesting that there is insufficient information to estimate either the mean M or the deviations in M for this age group. While the age composition data, calculation of total mortality, and identified mechanisms strongly suggest that natural mortality of older fish has increased through time, the estimation method in the length based model did not seem appropriate, particularly as there are alternative methods currently available. Accordingly, the length based model was not recommended for this assessment and a state-space model was developed.

AGE BASED STATE-SPACE MODEL

Models relying on penalized deviations with fixed or iteratively tuned penalties, like the length based stock assessment model, introduce subjectivity in the estimation of parameters. State-space models, such as those implemented in the WHAM R package (Stock and Miller 2021), address this using random effects, estimating standard deviation (SD) components to objectively constrain temporal variation while reducing the total number of parameters. The flexibility of WHAM allows the inclusion of random effects for key processes including natural mortality, numbers-at-age, selectivity, and catchability, while also supporting the inclusion of environmental covariates to link population dynamics to external drivers. Here, a state-space model was implemented in WHAM for Yellowtail.

Methods

Data inputs were

1. annual fishery total catch and proportions in numbers at age by fleet (seine and trawl),
2. annual proportions in numbers at age and age aggregated biomass index from the RV survey,
3. maturity-at-age from the RV survey, and
4. weight-at-age from the RV survey, used for both the fishery and the survey inputs.

A self-weighing Dirichlet distribution treating zero observations as missing (random effects) was used for all age composition.

Numbers at age in the first year were estimated as age specific fixed effects. Recruitment (age 1) was estimated as mean with annual deviations treated as random effects. Process error in numbers at age (2+) transition was not allowed.

Catchability to the RV survey was assigned an initial value of 0.3 and a gaussian prior on the logit transform of the catchability parameter (as in Surette and Swain 2016) with a SD of 0.25.

Sensitivity runs showed that the mean M or annual deviations in M were not estimable for the age group 1-3. Accordingly, M for this age group was modeled as a fixed mean of 0.7, based on the mean M estimates across life-history based methods for these ages (Cope and Hamel 2022; Appendix C). Mean M was estimated for three age groups 4-7, 8-9 and 10+). For these, process error in M was modeled as AR1 random effects over years.

Selectivity to the seine fishery, trawl fishery and RV survey were modeled as logistic selectivity curves without fixing parameters. Selectivity time blocks were as follows: 1985 to 1989 and 1990 to 2007 were shared blocks for the seine and trawl fleets, 2008 to 2025 were distinct blocks for each the trawl and seine fleets. The RV survey selectivity was modeled as three time blocks: 1985 to 1989, 1990 to 2008 and 2009 to 2025.

Model diagnostics were assessed through visual inspections of fit to the data, convergence checks, delta AIC between sensitivity runs, standard error and confidence intervals of parameter estimates, examination of raw and one step ahead residuals, jitter and self-test analyses, retrospective patterns, Mohn’s rho and biological credibility of estimates and derived quantities.

Results

The two-fleet model converged and model parameters were well estimated, with low SE and narrow CIs (Table 2).The maximum gradient was 6.84e-12. The jitter analysis produced negative log-likelihoods, SSB and F estimates that were identical to the base model. The self-simulation study lower 2.5% and upper 97.5% intervals for F covered zero in all years. The parameter estimates were stable across simulations. SSB showed moderate negative bias in years 1988 to 1996 and slight positive bias in years 2017 to 2021. The 2.5% and upper 97.5% intervals covered zero in all other years. Although moderate bias in SSB occurred it was limited to specific periods, was not directional and did not persist across the time series. Bias in SSB was 0 in the terminal year.

The retrospective analysis revealed almost no bias in mean F, with a Mohn's ρ of 0.021. Recruitment retrospective patterns were more important, although acceptable, with a Mohn's ρ of -0.173. SSB estimates displayed little retrospective pattern, with small deviations and a Mohn's ρ of -0.03. Overall, while recruitment showed moderate negative bias, SSB and F estimates were robust, supporting model reliability.

The model fit to age composition data was good for the 2 fleets (Figure 20 and Figure 21) and the RV survey (Figure 22). The trawl fleet age composition had a few years where residuals were larger but overall the residuals were small and without pattern. Model fit to both the trawl and seine gear catch was good. Model fit to the RV survey biomass was generally good over the time series, despite moderate to high annual variation in the survey index (Figure 23). The consecutive decrease and increase in survey biomass in the last decade was well captured by the model. The confidence bands of model fit to aggregated catch and index included zero in all years and showed no systematic trend in model fit over time.

Selectivity at age was logistic for all age composition data. The selectivity at age increased over time for younger fish, in the RV survey (Figure 24) and in both fleets (Figure 25 and Figure 26). The selectivity of younger fish kept increasing in the third time block of the RV survey (2009-2025), whereas it varied little between the second and third time blocks of the fishing fleets.

The uncertainty in SSB estimates was low over the time period (CV< 0.3). SSB gradually declined between 1985 and the late 2010s, then SSB gradually increased until 2024 (Figure 27). However, the SSB is now composed of younger fish, and the estimated increase in SSB was attributed to the increase in biomass of fish ages 1 to 4 (Figure 28). The biomass of Yellowtail age 5+ was slightly above 30,000 t in 1985 but gradually decreased over time (Figure 29). 5+ biomass remained below 5,000 t between 2018 and 2024. 5+ biomass in 2025 was 8,269 t (95% CI: 5,666-10,872 t).

The relationship between SSB and recruitment did not seem to suggest a stock recruit relationship could describe the recruitment dynamics, as the relationship is mostly linear of the range of estimates (Figure 30). Models using Ricker or Beverton-Holt stock recruit models were not able to estimate the parameters for these parametric functions. Interannual recruitment variability was relatively low and changes in recruitment occurred over relatively long periods of time. The number of recruits increased between 1985 and 2005 before rapidly decreasing until 2014. Recruitment gradually increased again until 2022 and remained at intermediate levels since.

The uncertainty in F estimates by fleet were low over the time series (CV<0.4). F was low at all ages over the time series and average F for ages 4 to 10 remained below 0.05. F was lowest in early years (1985 to 1995) and generally increased until the late 2010s. F decreased after 2020 and was zero after 2022. Periods of relatively high F coincided with relatively high landings (1995-2012) or low 4+ abundance.

The mean natural mortality by age group were estimated at 0.7 for the age group 4 to 7, at 1.3 for the age group 8-9 and 1.1 for the 10+. The AR1 process parameters showed good precision for σ with SEs and confidence intervals indicating acceptable identification. The ρ parameter estimate had larger SE and CIs (Table 2). M generally increased, with variation, between 1987 and 2008. M then generally decreased between 2009 and 2019. Between 2020 and 2025, M increased from 0.52 to 0.72 for the age group 4-7, from 0.96 to 1.34 for the age group 8-9 and from 0.82 to 1.15 for the 10+. Overall, M estimates were high for all age groups, as supported by the severe age truncation in age composition data, and as seen in previous stock assessments.

A coarse approximation of the scale of the Yellowtail consumption by grey seal in the sGSL can be performed using available data and estimates. Benoît and Rail (2016) estimated the consumption of all prey by grey seals in the sGSL ecosystem to have increased from <10,000 t per year in 1960 to over 120,000 t in 2014. Yellowtail contribution in weight to the reconstructed grey seal diets in the GSL varied between 3.1% (1994 to 2003; Hammill et al. 2007) and 0.52% (2022 to 2023; Irani et al 2025). Assuming 100,000 t of prey, this rendered between 520 and 3,100 t of consumed Yellowtail in a given year. Assuming 150,000 t of prey, this renders between 780 and 4,650 t of Yellowtail consumed in a given year. Comparatively, the mean estimated M removals of Yellowtail ages 4+ between 1994 and 2023 was 12,949 t annually.

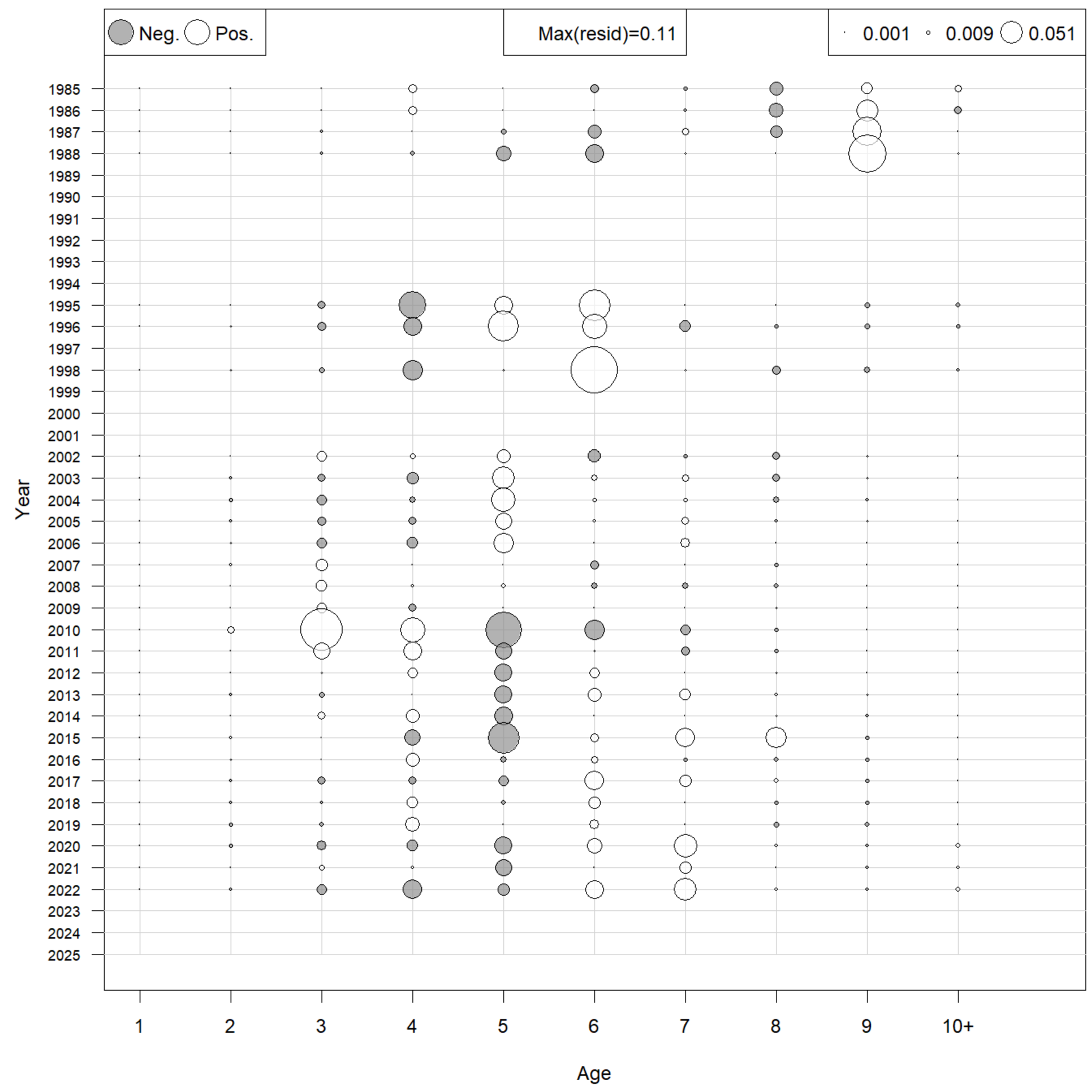

Figure 20: Seine fleet age composition residuals (observed-predicted) between 1985 and 2025. White circles are positive residuals and grey circles are negative residuals. Circle size is proportional to the residual value.

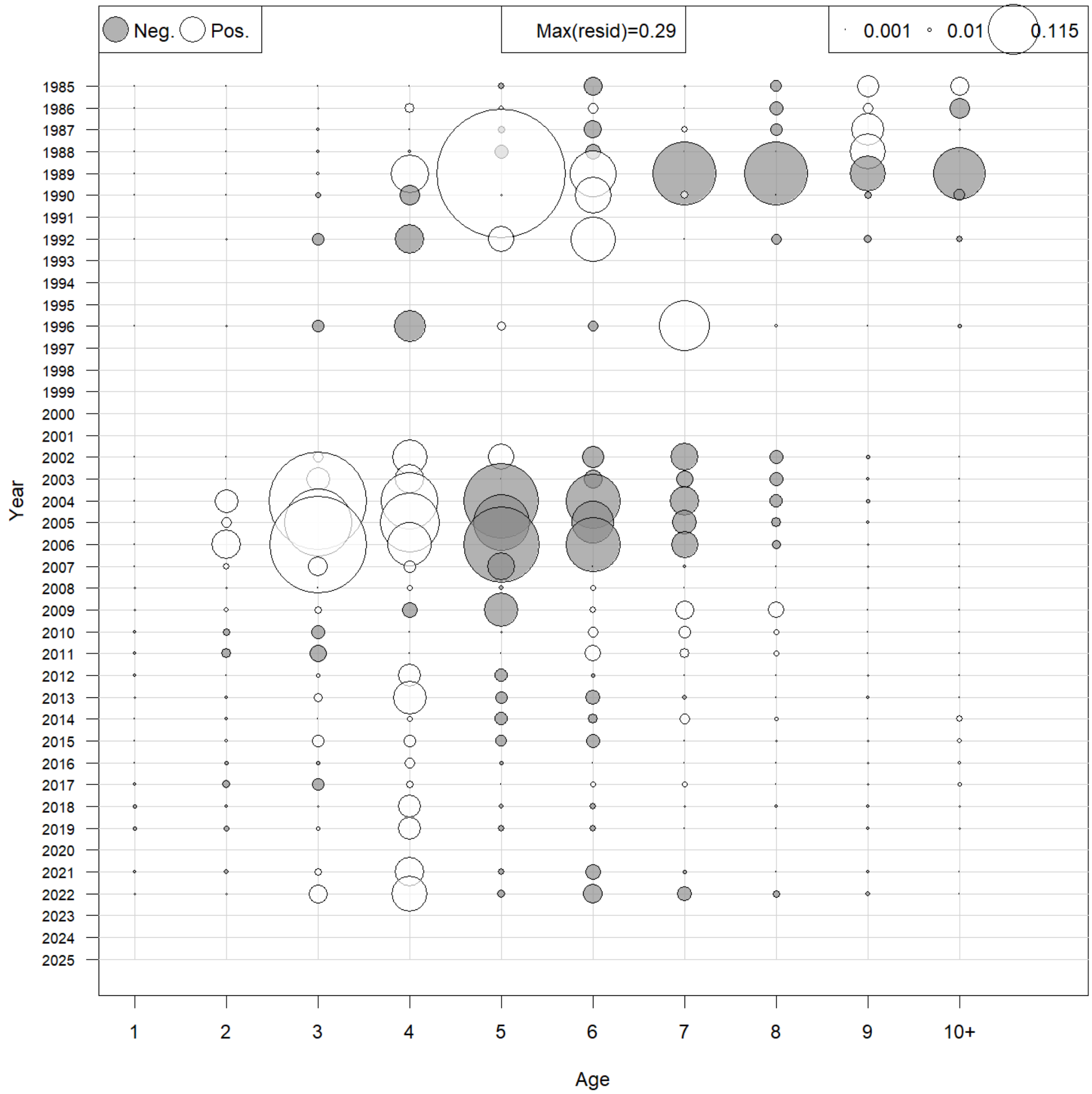

Figure 21: Trawl fleet age composition residuals (observed-predicted) between 1985 and 2025. White circles are positive residuals and grey circles are negative residuals. Circle size is proportional to the residual value.

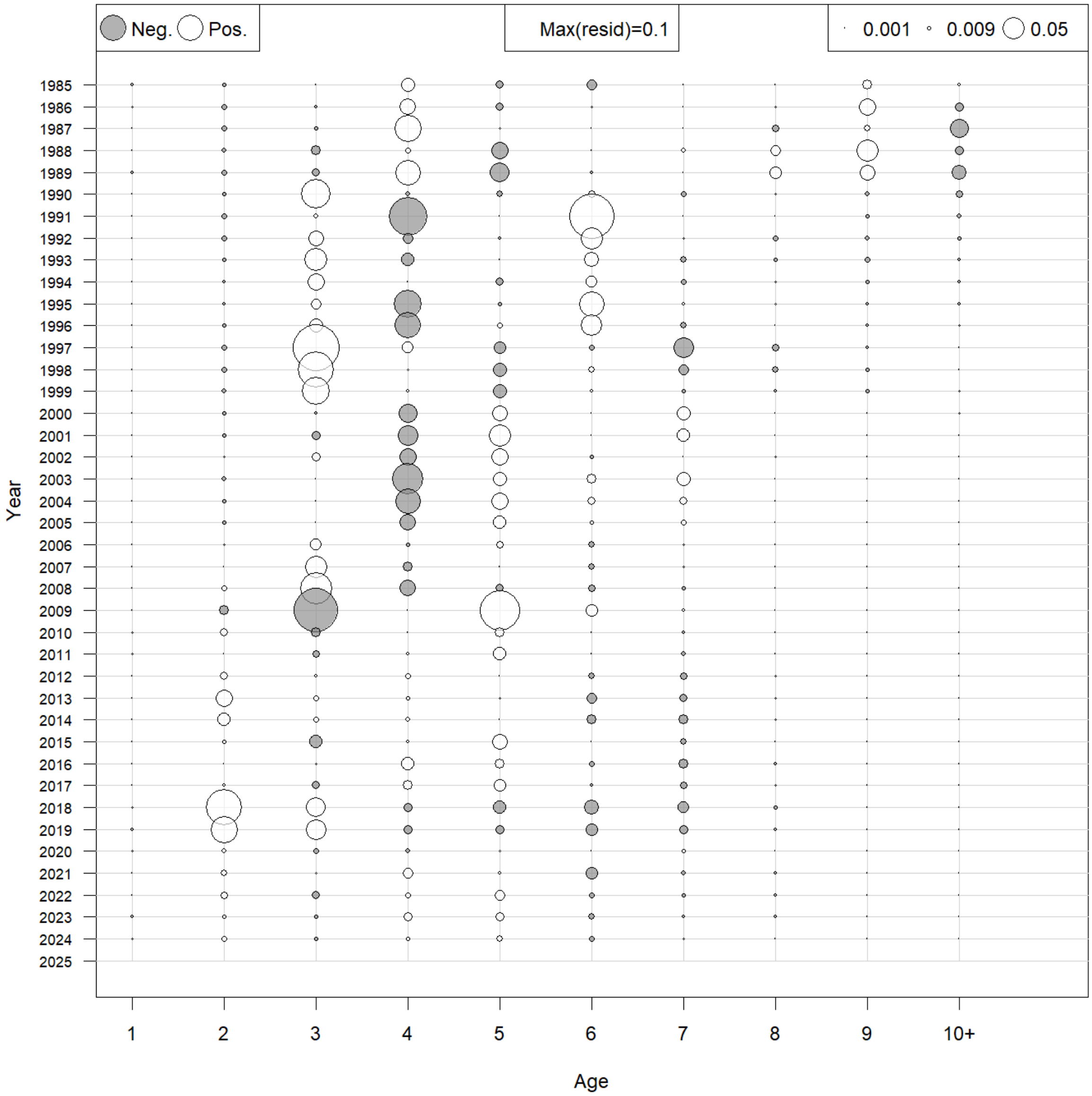

Figure 22: RV survey age composition residuals (observed-predicted) between 1985 and 2025. White circles are positive residuals and grey circles are negative residuals. Circle size is proportional to the residual value.

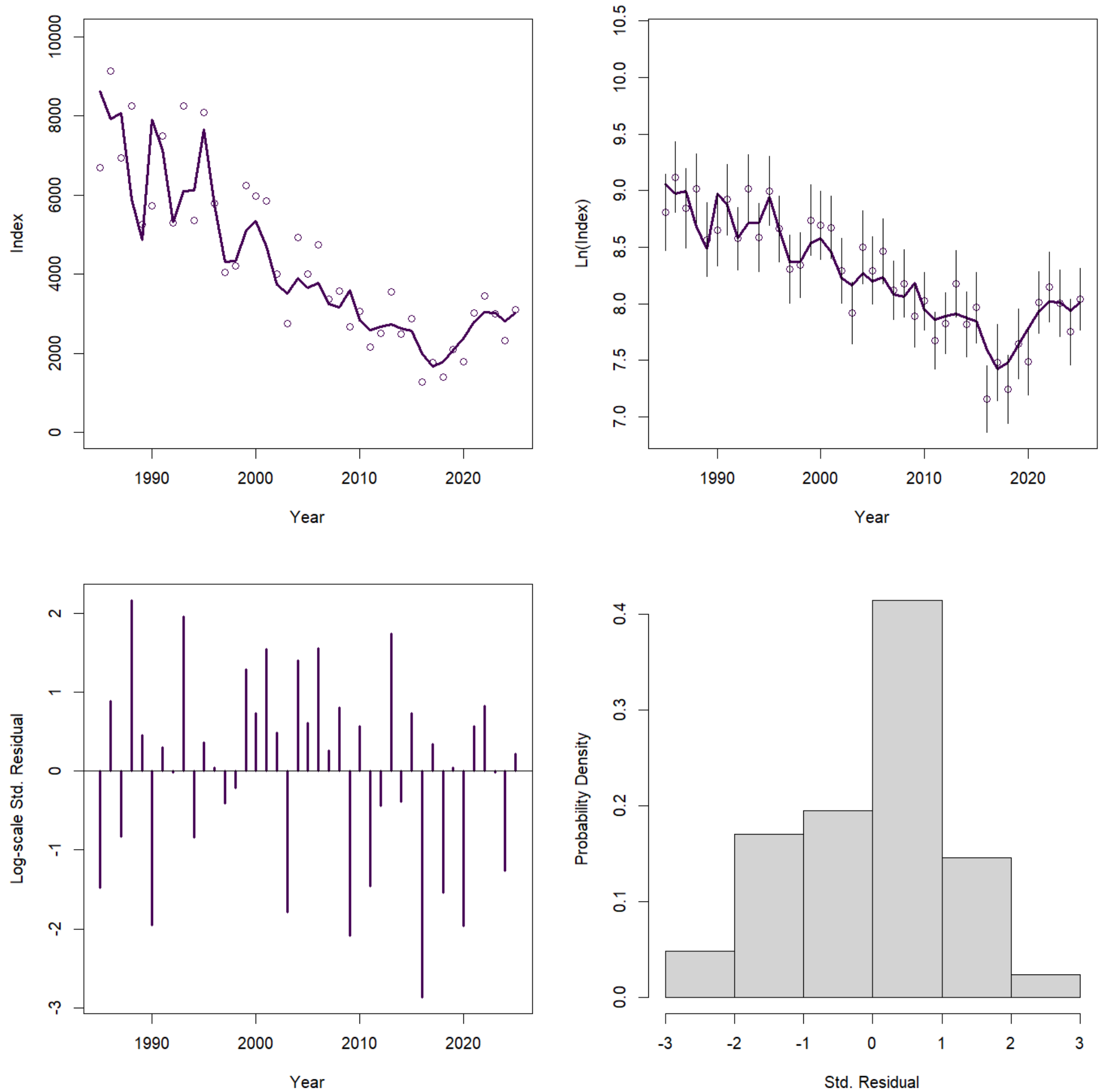

Figure 23: Model fit to the RV survey biomass index (top left), and to the log of the index (top right), standardized residuals by year (bottom left) probability density of standardized residuals (bottom right).

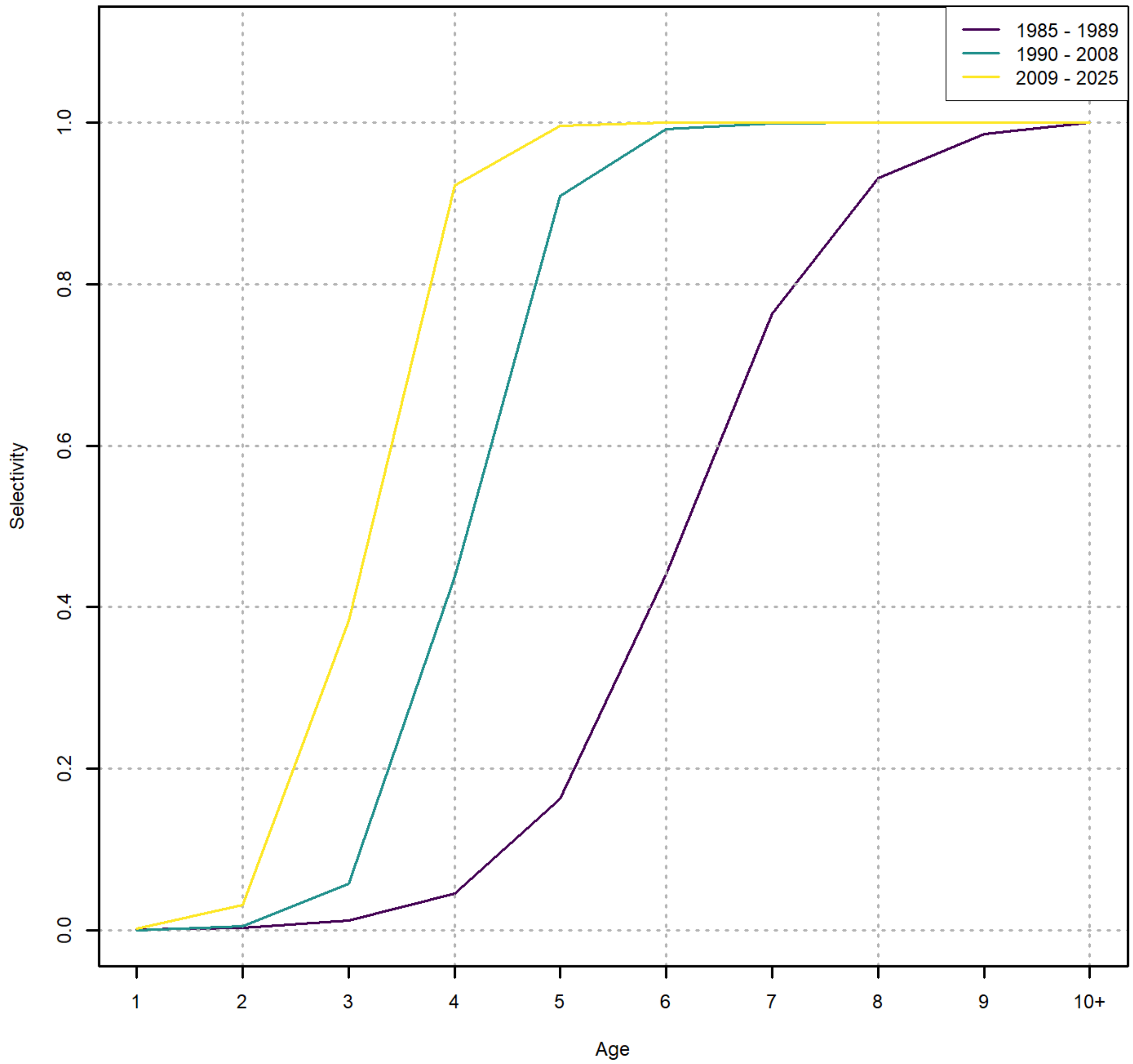

Figure 24: Selectivity at age for three time blocks (colored lines) in the RV survey.

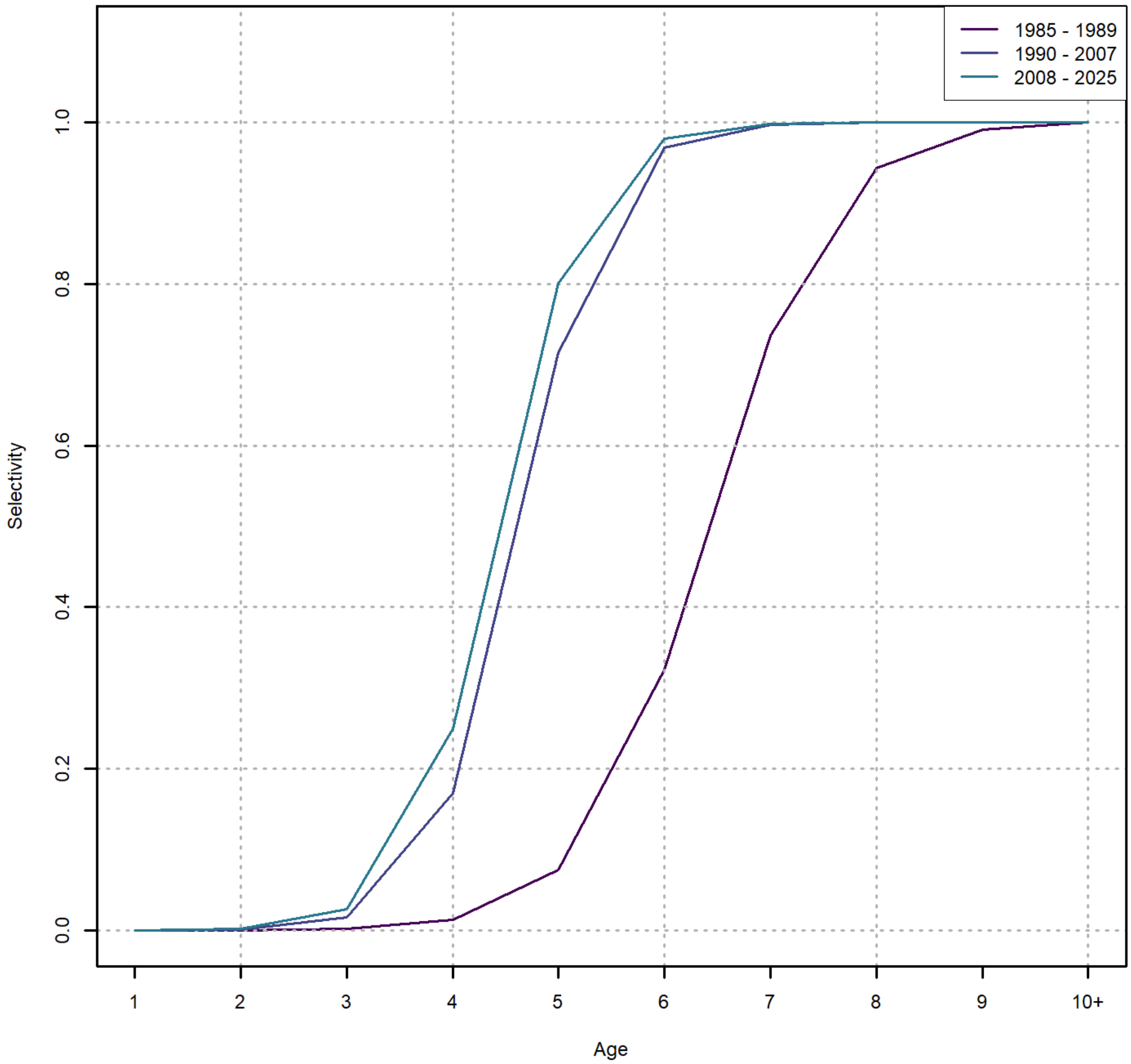

Figure 25: Selectivity at age for three time blocks (colored lines) in the Seine fleet fishery.

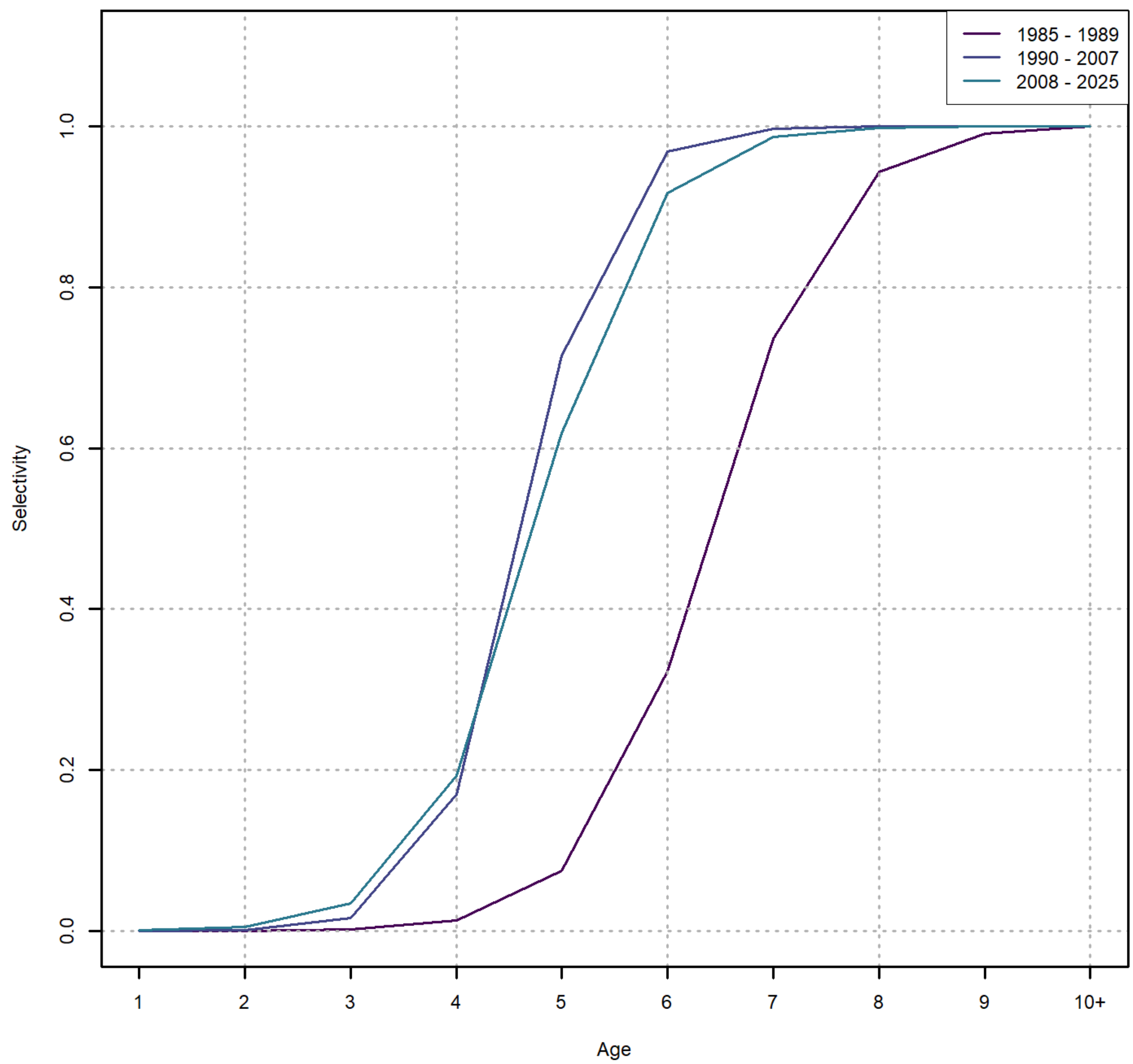

Figure 26: Selectivity at age for three time blocks (colored lines) in the Trawl fleet fishery.

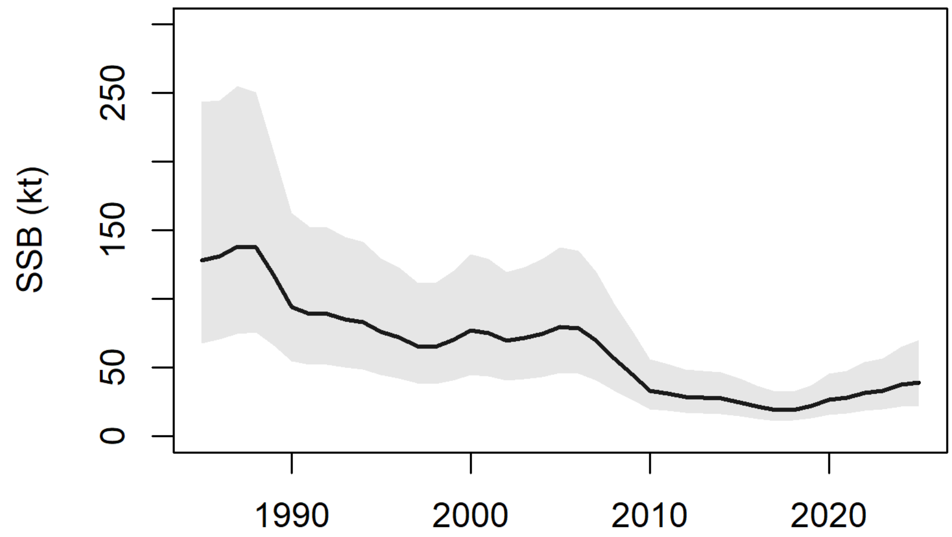

Figure 27: Yellowtail flounder spawning stock biomass (SSB; kt) between 1985 and 2025.

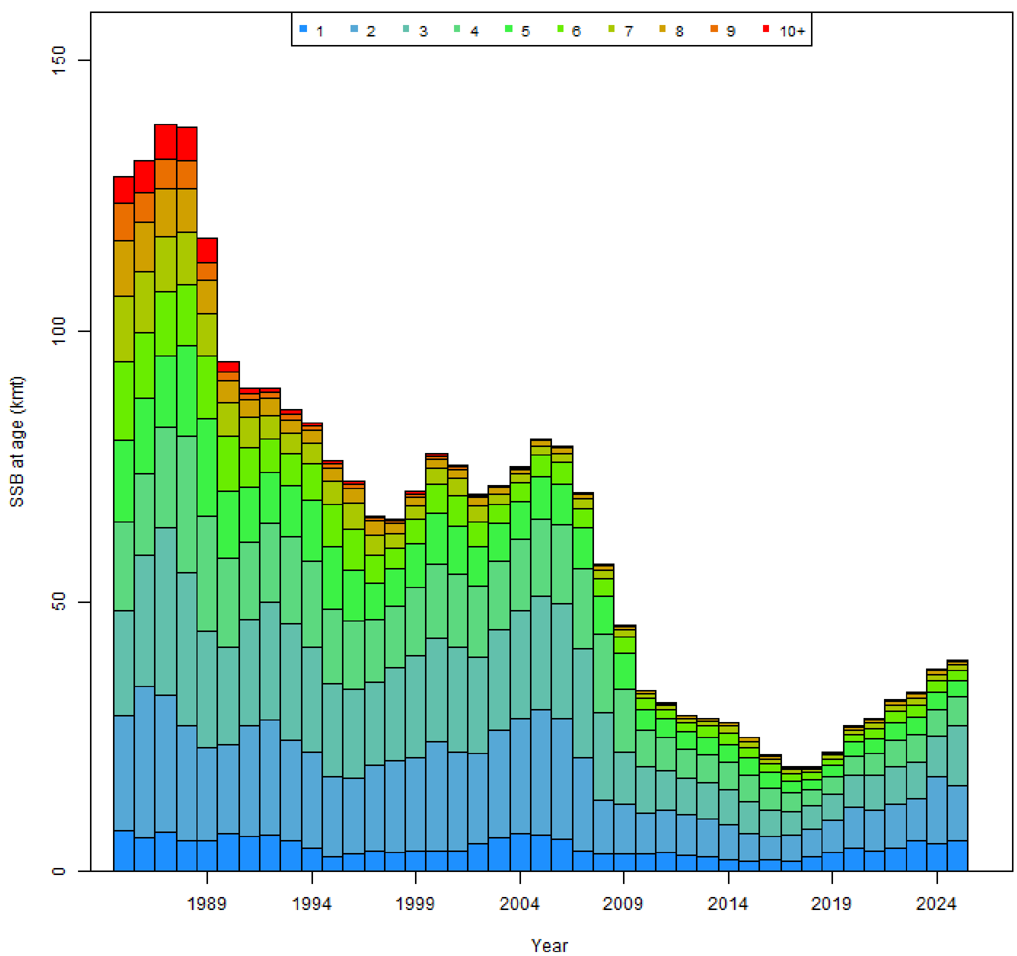

Figure 28: Yellowtail Flounder spawning stock biomass (SSB; kt) at age (colors) between 1985 and 2025.

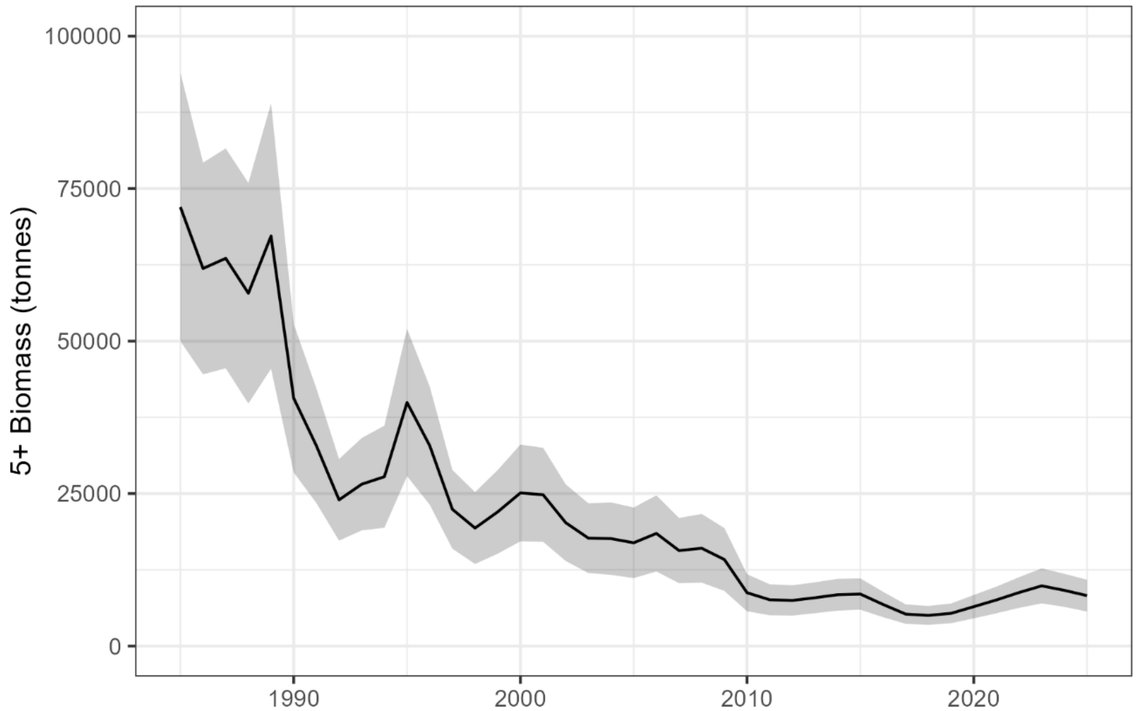

Figure 29: 5+ biomass (tonnes) of Yellowtail Flounder between 1985 and 2025. Solid line is the mean and shading is the 95% confidence interval.

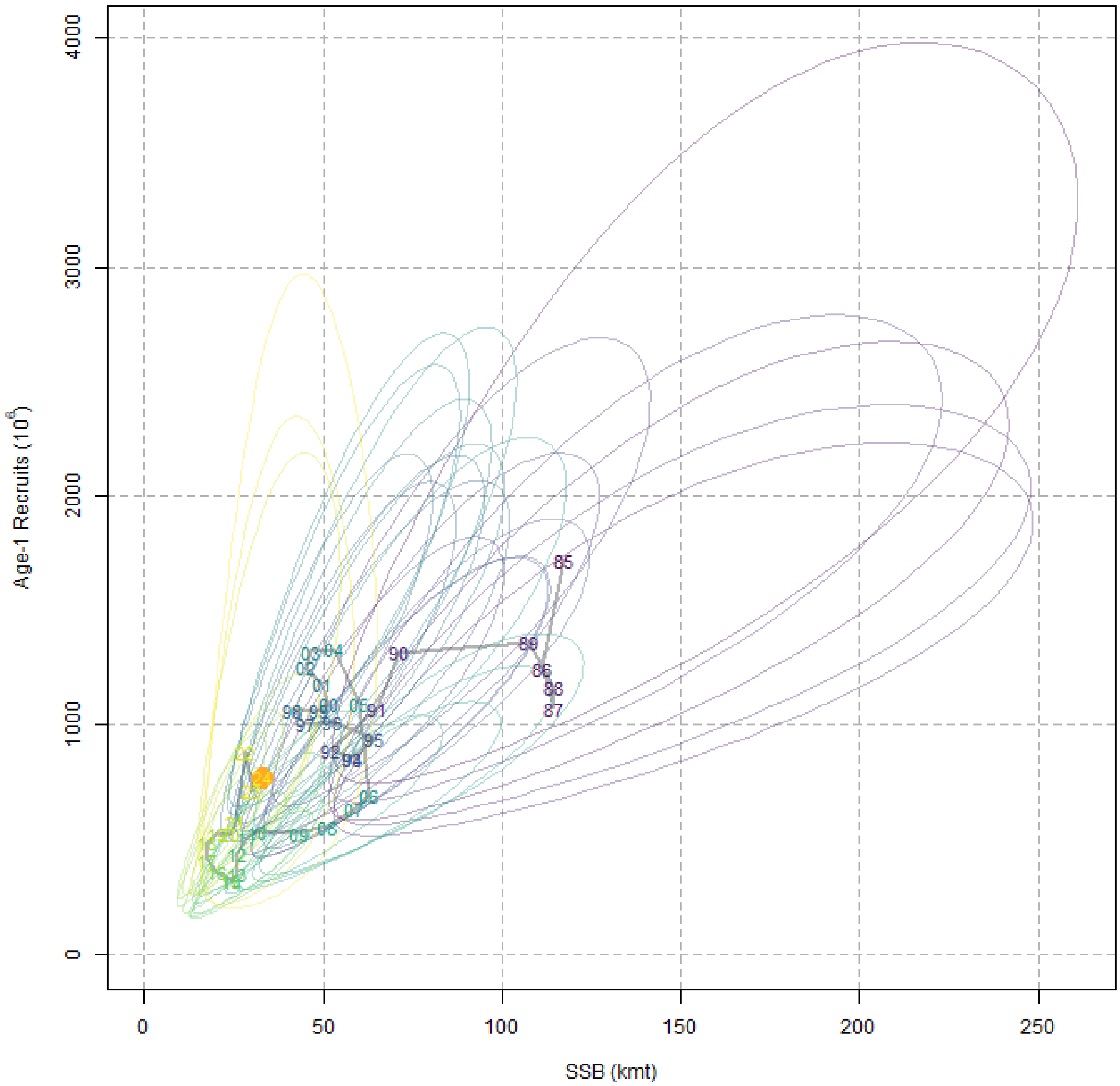

Figure 30: Recruits (number of age 1 fish; hundred thousands) in function of spawning stock biomass (SSB, kt) between 1985 and 2025 (colored labels and circles). Points are stock recruit pairs, the solid line tracks years and circles indicate uncertainty in estimates.

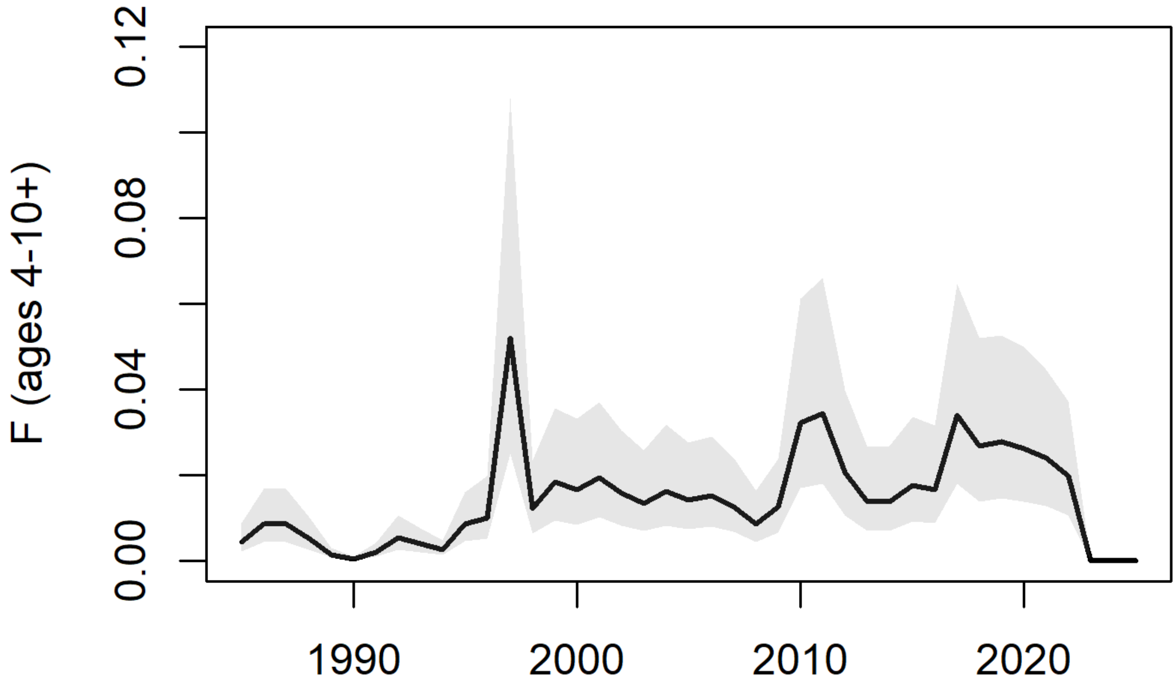

Figure 31: Average fishing mortality (F) for Yellowtail Flounder ages 4 to 10+ between 1985 and 2025. Solid line is the mean and shading is the 95% confidence interval.

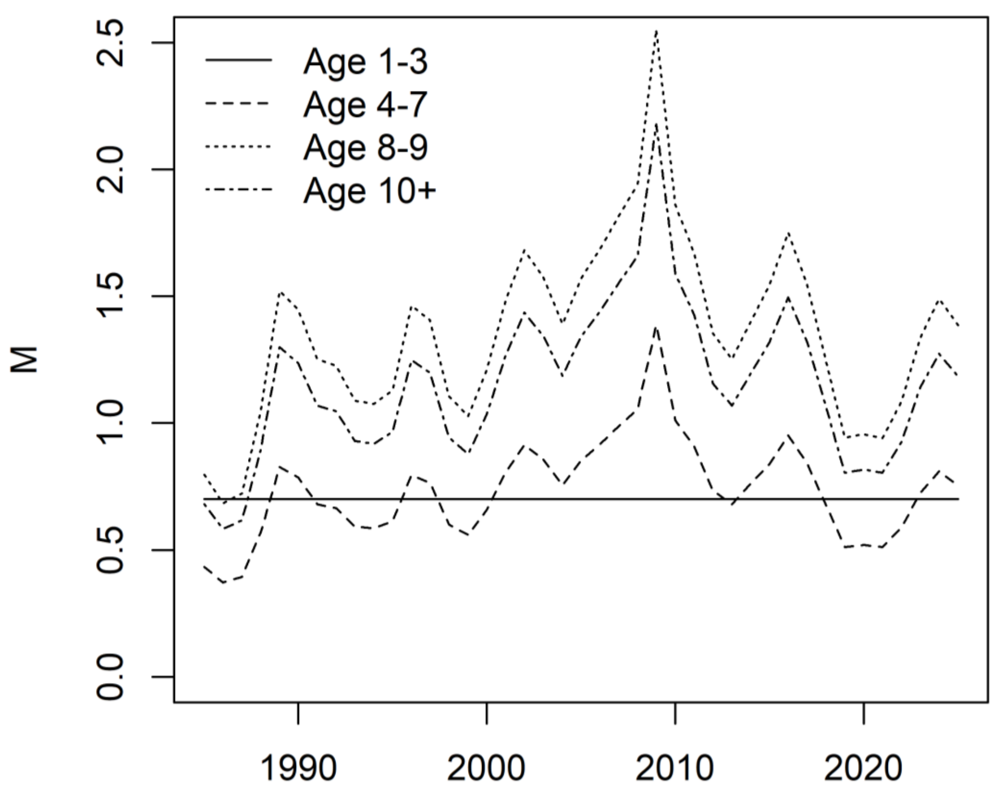

Figure 32: Yellowtail Flounder natural mortality (M) by age group (line types) between 1985 and 2025.

Discussion

Available evidence indicates that the dynamics of Yellowtail in the sGSL have been strongly influenced by substantial temporal changes in natural mortality, particularly size-dependent increases in mortality affecting older, larger individuals. Assuming static mortality of younger fish, the increased natural mortality of older fish provides a mechanistic explanation for the observed increase in the abundance of young fish and the concurrent decline in older individuals, and are consistent with broader changes in productivity observed throughout the sGSL marine fish community (Benoît and Swain 2008; Swain and Benoît 2015).

Fishing mortality has been estimated to be low relative to natural mortality at the scale of the sGSL stock, suggesting that natural mortality is the dominant driver of population dynamics. However, fishing effort is geographically concentrated around the Magdalen Islands, and localized impacts of fishing mortality may not be detectable in models representing the entire sGSL population. Nonetheless, widespread trends in length and age composition across the sGSL suggest that elevated natural mortality, rather than fishing mortality alone, is the primary factor shaping stock status at the broader spatial scale.

This decline in older fish abundance has been accompanied by a contraction in size structure and a marked reduction in length and age at maturity, with length at 50% maturity declining from approximately 23 cm in the early 1970s to approximately 14 cm in recent years. Earlier maturation is consistent with life-history theory predicting adaptive responses to increased mortality of older individuals (Law 1979; Reznick and Ghalambor 2006). However, maturation at smaller sizes may entail increased reproductive costs and reduced adult survival (Hutchings 1994; Roff 2002), and may alter maternal effects influencing offspring fitness and recruitment dynamics (Hutchings 1991; Reznick et al. 1996; Heath et al. 1999; Berkeley et al. 2004; Green 2008).

Although theory predicts that reductions in maternal size and age structure may decrease recruitment success, this was not evident in model-estimated recruitment. The stock-recruit relationship was largely linear, with no indication of reduced recruitment rates at lower stock sizes, despite the spawning population being dominated by younger individuals. In recent years, SSB was estimated to increase, primarily driven by higher abundance of younger fish (ages 2-4) rather than older individuals (ages 5+).

Changes in Yellowtail population structure appear consistent with broader ecosystem-level patterns observed in the sGSL. Increases in natural mortality of large individuals have been documented across the demersal fish community, suggesting a community-wide shift toward stronger size-dependent mortality (Benoît and Swain 2008; Swain and Benoît 2015). Increased abundance of smaller fish has been attributed to reduced predation following earlier declines in large demersal predators, possibly combined with reduced incidental mortality due to decreased fishing effort (Benoît and Swain 2008; Swain et al. 2013). Predation by grey seals is widely considered a major contributor to increased natural mortality in the sGSL ecosystem (Hammill et al. 2014; Neuenhoff et al. 2019; Swain et al. 2019a, 2019b). As grey seal are predominantly piscivorous and reside in the GSL year-round, this increase in predator biomass coincides with declines in several fish populations known to be their prey and has raised concerns regarding their role in preventing recovery of commercial fish stocks (Benoît and Swain 2008; Benoît and Swain 2011; Hammill et al. 2014; Irani et al. 2025). Alternative explanations for elevated mortality in sGSL groundfish including unreported catch, emigration, disease, contaminants, environmental stress, or life-history change, generally lack strong empirical support relative to the predation hypothesis (Swain 2011; Swain and Benoît 2015).

The level of M removals were within a reasonable range when compared to expectations from seal diet data. The estimated M removals were not intended to directly quantify Yellowtail consumption by grey seals; rather, they provide a baseline for comparing the estimated quantities of Yellowtail removed by M. The uncertainty in population scale further complicates comparisons between expected removals inferred from diet and those attributed to M, as the population scale itself, and consequently the M removals, can vary depending on the used prior for q.

Selecting an appropriate model configuration is critical because misattributing the source of process error can bias key management quantities (Szuwalski et al. 2018). For instance, incorrectly assigning deviations in natural mortality to other processes—or vice versa—has been shown to introduce substantial bias into management metrics (Li et al. 2024). Given evidence for temporal changes in M and the consistency of results across regional models, process error was preferentially attributed to M.

Improvements over the previous model include the integration of age composition data rather than relying solely on size-aggregated catch and size-aggregated survey biomass, allowing the model to better inform cohort dynamics and estimate underlying processes. The age length keys now include information from the earlier period of the time series, allowing to refine the resolution of growth and maturity dynamics. Using a state-space framework allowed to reduce the dependence on priors and penalized likelihoods. Young fish M is now fixed to a value based on life-history traits. M for older fish now varies annually rather than by broad blocks, and the estimates no longer consistently hit parameter bounds as they did in the earlier model.

STOCK STATUS

The biomass LRP represents the upper bound of stock states that should be avoided in order to prevent serious harm to the stock and is the boundary between the Critical and Cautious zones of DFO’s PA Policy (DFO 2009). In the context of fisheries, serious harm is an undesirable state that may be irreversible or only slowly reversible over the long-term (DFO 2023). It may be directly or indirectly due to fishing, other human-induced impacts, or other natural causes, and occurs at states before extirpation is a concern. These states can be associated with impaired productivity or reproductive capacity, resulting from changes to biological processes such as recruitment, growth, maturation and survival, and may lead to a loss of resilience, defined as an impaired ability to rebuild, exceed replacement or to recover from perturbation. These states can be associated with an elevated risk of depensation or Allee effect (i.e., negative density dependence, in which the intrinsic rate of increase for a stock decreases, rather than increases, as abundance declines) and are states where population dynamics are generally poorly understood.

Traditional fisheries reference points are based on the assumption of a system in equilibrium. However, the incorporation of time-varying productivity is becoming increasingly common among fish stocks (e. g., Thorson et al. 2015; O’Leary et al. 2020), especially in the face of a changing climate (Szuwalski et al. 2023). The choice of the productivity time period used to derive reference points can have important effects on the perceived stock status and associated harvest rates (Bessell-Browne et al. 2024; Hebert et al. 2025). In NAFO 4TVn Atlantic Cod, dynamic LRPs based on time-varying productivity parameters collapsed to zero after 1990, illustrating the risk of using moving baselines when productivity declines persistently (Turcotte et al. 2026). Static LRPs could be considered to be more consistent with the DFO definition of the LRP to avoid serious harm in cases of declining productivity, simply because they define a goal to preserve higher levels of biomass, below which stock dynamics become increasingly uncertain (Hebert et al. 2025). When management objectives require the population to remain above an absolute biological threshold, a static LRP is the more appropriate choice (Bessell-Browne et al. 2024).

Estimating reference points for Yellowtail poses several challenges relating to time varying productivity as the stock is not at equilibrium over the current assessment time frame. Non-stationarity in growth, maturity, and natural mortality all contribute to the stock's lack of equilibrium over time. Growth generally declined over the time series, as indicated by estimates of both L50 and A50. A declining trend in the size and age at maturation is often a symptom of high mortality. When mortality is high, early maturation is favored because it increases the probability of surviving to reproduction (Swain 2011; Forestier et al. 2020). Model estimates of natural mortality of Yellowtail age 4+ increased gradually between 1990 and 2010 and remained high afterwards. This increase in M resulted in a heavy truncation of the age composition where most of the population is composed of fish ages 1 to 4 and 5+ fish are almost absent from the population. However, recruitment does not seem to be impaired as the number of recruits did not decrease as SSB decreased over time.

Yellowtail is assessed and managed using the biomass of commercial size fish, 25+ cm. Reference points were historically based on a proxy for B_{MSY}, either derived from the RV survey biomass (Surette and Swain 2016) or the length based stock assessment model (Rolland et al. 2022). Here, to be consistent with previous assessments and with the basis for resource management, biomass reference points were evaluated using 5+ biomass, which is the modal age of a 25 cm fish in the contemporary ALK. Consequently, the basis for establishing reference points and management quantities should be equivalent between this and previous assessments.

PRODUCTION ANALYSIS FOR SERIOUS HARM

Prolonged low production and low biomass states may indicate serious harm to a stock's ability to rebuild to target levels (Kronlund et al. 2018). The persistence of low production and low biomass states was assessed by examining the relationship between surplus production and 5+ biomass. Surplus production was calculated as:

$$P_t = C_t + B_{t+1} - B_t$$

Where t indexes year, P is the stock surplus production, C is the fishery catch and B is the stock biomass for ages 5+.

Time series were inspected for values indicating simultaneous sustained low production and low biomass that could be indicative of serious harm. As in Forrest et al. (2023), the definition of "low" was somewhat subjective, with low production states identified as sustained periods of production near or below zero and low biomass states as prolonged biomass levels well below historical values.

While biomass of ages 5+ generally declined, Yellowtail surplus production varied considerably between 1985 and 1999, alternating between brief periods of positive and negative production (Figure 32). From 2000 to 2016, biomass continued to decrease and production remained consistently low, fluctuating around zero. Both production and biomass increased slightly after 2016. Overall, the period since 2000 has been characterized by persistently low production and low biomass, potentially indicating serious harm to the stock productivity.

Under typical density-dependent dynamics, the per capita rate of population growth is expected to increase as population size declines, due to reduced intraspecific competition at lower densities (Nicholson 1933). However, when population size falls below a critical threshold, this relationship can reverse and per capita growth declines instead, which is named an Allee effect (Courchamp et al. 1999). The relationship between 5+ biomass and the production rate (P/B) did not show evidence of an Allee effect. The linear regression was negative (β=-0.000002539) but not statistically significant (p=0.0964), indicating compensatory dynamics over the range of biomass (Figure 33).

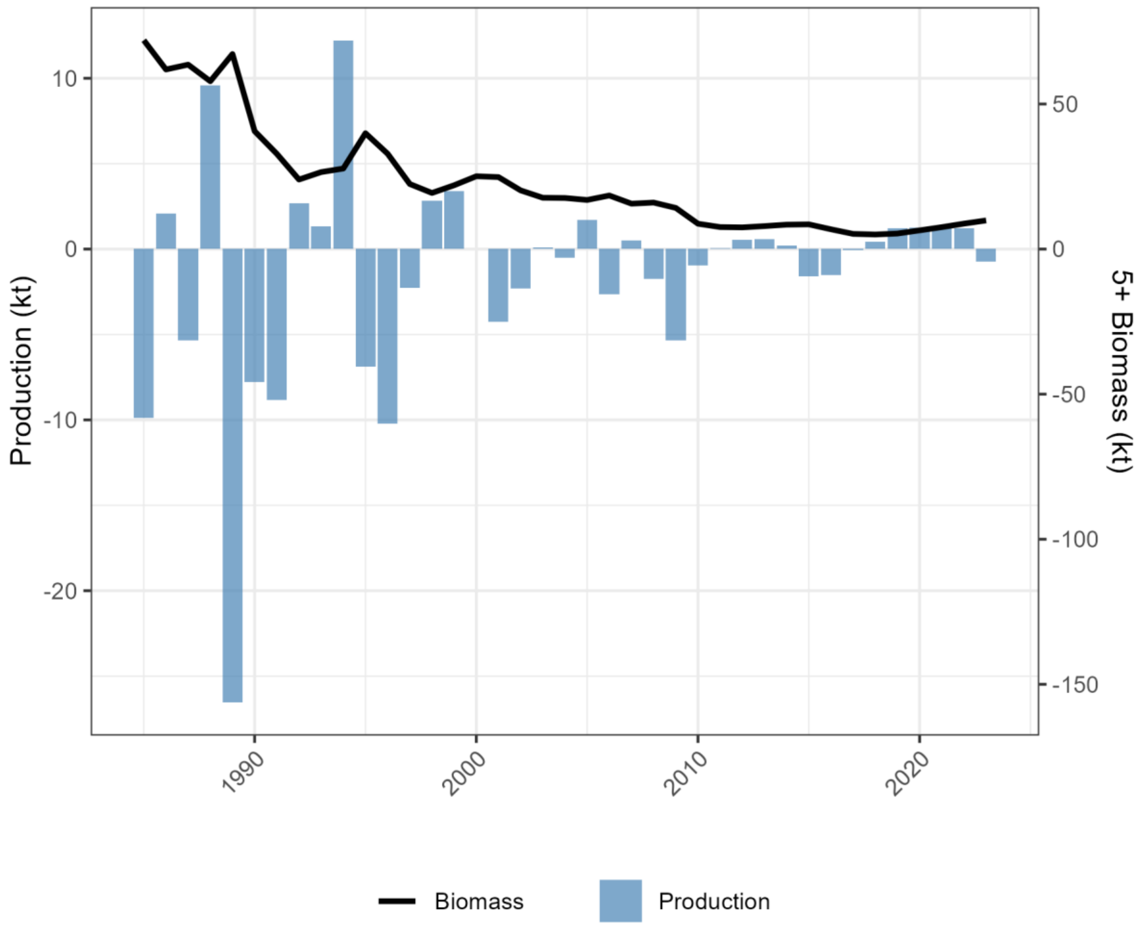

Figure 33: Annual estimates of 5+ biomass (black line, kt) and surplus production (blue bars, kt) between 1985 and 2024.

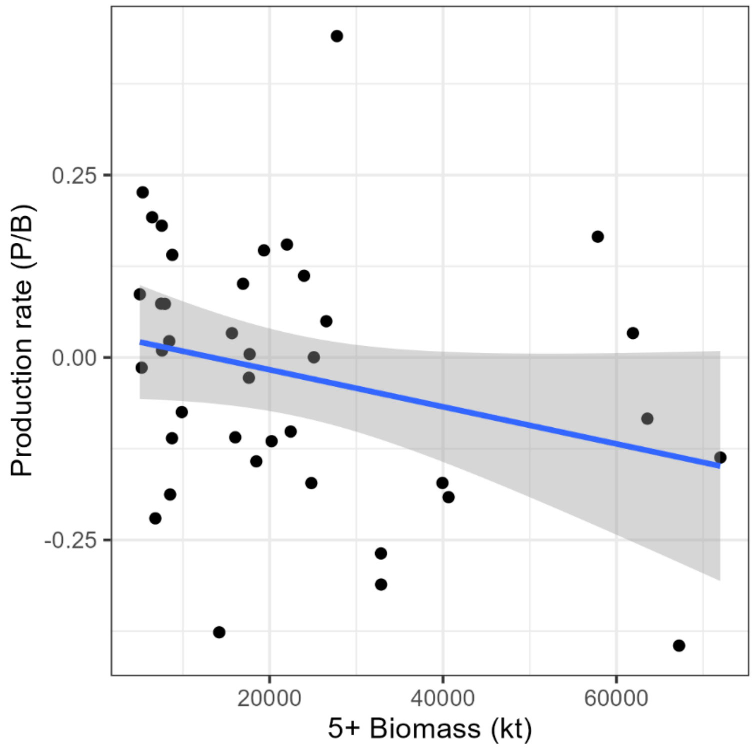

Figure 34: Production rate (surplus production per unit of biomass) in function of 5+ biomass between 1985 and 2024, blue line and grey shading are linear regression.

BIOMASS REFERENCE POINTS

Viable options for defining reference points for this stock are limited. Estimates of biomass that produce maximum sustainable yield (B_{MSY}) or virgin biomass (B_0) are typically used to derive LRPs from stock assessment models. However, they require stationarity in demographic productivity parameters. The absence of equilibrium in growth, maturity and natural mortality precludes the use these methods for Yellowtail. Furthermore, there is no stock recruit relationship to estimate MSY based reference points or derive biomass at 50% Rmax, and the use of 5+ biomass precludes the usage of a fraction of B_0.

In the absence of an estimate of B_{MSY} from an explicit model, the PA framework provides guidance to identify reference points and harvest rules (DFO 2009). The provisional estimate of B_{MSY} could be taken as follows (select the first feasible option): (1) the biomass corresponding to the biomass per recruit at $F_{0.1}$ multiplied by the average number of recruits; or (2) the average biomass over a productive period; or (3) the biomass corresponding to 50% of the maximum historical biomass.

(1) $F_{0.1}$ methods tend to allow for higher F and when M is higher than 0.2 (Turcotte and McDermid 2024), and can lead to deriving limit reference points below the Allee threshold (Turcotte et al. 2026). When evidence that M increased over time for a stock is strong, a method suggesting higher F thresholds as M increases should not be used to derive biomass or fishing reference points (Legault and Palmer 2015).This proxy for B_{MSY} was not supported for Yellowtail.

(2) Productive periods were defined as periods of 5 to 10 years during which stock production (as calculated above) and 5+ biomass were simultaneously high. The proxy for B_{MSY} was

calculated as the mean 5+ biomass in the identified years. The candidate LRP derived from this proxy was calculated as 40% of its value.

Biomass was highest at the beginning of the time series. Production varied substantially over the first decades, with years or series of years alternating between positive and negative production. However years of high production occurred while biomass was high between 1985 to 1989. Years of high production occurred after 1985 but biomass was either declining or already low. The B_{MSY} proxy was the mean 5+ biomass in years 1985 to 1989, and the associated candidate LRP, $0.4B_{MSY}$, was estimated at 25,800 t of 5+ biomass. A candidate USR can also be derived using the proxy for B_{MSY}; defined as $0.8B_{MSY}$ and estimated at 51,600 t of 5+ biomass (Figure 34).

Using these reference points, the 5+ biomass has been under the LRP since 1997. In 2025, 5+ biomass was 8,269 t (95% CI: 5,666 – 10,872), which is 32% of the LRP, and the probability that 5+ biomass was under the LRP was 100%.

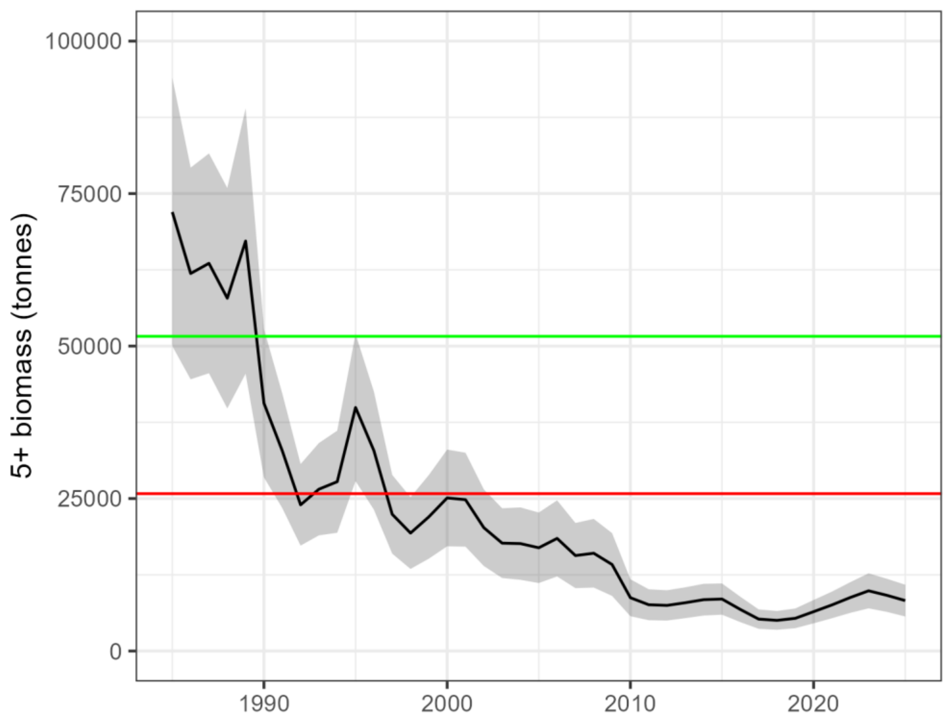

Figure 35: Yellowtail flounder 5+ biomass (tonnes) between 1985 and 2025, red line is the biomass limit reference point, green line is the upper stock refence point, black line is mean biomass and grey shading is 95% confidence intervals.

REMOVAL REFERENCE POINT

In the absence of an estimate of F_{MSY} from an explicit model, a proxy for F_{MSY} can be used as a removal reference point (DFO 2009). The proxy for F_{MSY} was the average mean F over a period of high biomass and stable F, defined as years 1985 to 1990. The average F for ages 4 to 10+ was 0.005. In 2025, F was 0 and 100% probability of being below the removal reference.

This very low removal reference is the result of the stock assessment being at the 4T scale, while the removals are very localized to the Magdalen Islands area. Consequently, if a fishery was to resume, localized population dynamics and removal reference should be used. A surplus production model using the RV survey strata for the Magdalen islands region and the local catch time series should be enough data input to derive a model useful for the management of this fishery.

REBUILDING TARGET AND TIMELINE

REBUILDING TARGET

For a prescribed major fish stock subject to the Fish Stocks Provisions, the legal obligation of a rebuilding plan under section 6.2 only applies when the stock is at or below its LRP. However, to increase the likelihood that a stock will not decline back to or below its LRP and to be consistent with the PA Policy intent to grow depleted stocks to healthier levels, a rebuilding plan will remain in effect until the stock reaches its rebuilding target. Once the stock reaches its rebuilding target, the rebuilding plan will come to an end and the stock will be subject to the Integrated Fisheries Management Plan (IFMP) or other management plan. DFO guidelines on rebuilding plans state that the rebuilding target must be set at a level above the LRP so that there is a very low to low likelihood of the stock being below its LRP (<5-25% probability).

To be consistent with other rebuilding plans in the sGSL and in line with the guidance, a rebuilding target could be defined as the biomass where there is at least a 75% probability that the stock is at or above the LRP. This does not represent a fixed number, as the Yellowtail stock is assessed using a stock assessment model and the associated levels of uncertainty in estimates such that the numerical value of the target is model-dependent and is subject to change with every assessment as years of data are added and/or as model changes are implemented. The objective of being above the LRP, particularly with the uncertainty in biomass estimates for this stock being relatively small, means that there is theoretically a higher probability of the stock falling below the LRP than if the rebuilding target was set closer to the USR or the TRP for example. The science guidelines to support development of rebuilding plans for Canadian fish stocks states that the rebuilding target should be set far enough above the LRP so that there is a low probability of falling below the LRP in the short to medium term (DFO 2021). If this rebuilding target is used, it could be made more robust by incorporating a requirement that the stock be at or above the LRP for 5 consecutive years, and that population projections should show a positive trajectory under harvest. Five years was selected since a rebuilding timeline (see below) could not be calculated or used to inform the choice of the number of years of growth that would minimize the probability of the stock falling below the LRP in the short to medium term. The number of years has consequently been set to the multi-year assessment cycle for this stock. This is also the frequency of review of the rebuilding plan (see below).

REBUILDING TIMELINE

A rebuilding plan also requires that the timeline to rebuild be identified in order to track rebuilding progress with respect to the objectives and management measures. The international standard and the approach recommended by DFO (2021) is to estimate the time to reach the rebuilding target in the absence of all fishing (T_{min}). Given the low productivity of the stock and a lack of response to reduced fishing pressure, rebuilding of the stock is not expected to occur until stock productivity improves (Figure 35). Consequently, T_{min} cannot be calculated and the timeline is set to the generation time as per the rebuilding plan guidance (DFO 2021). Generation time is identified as the same as in Georges Bank (NAFO 5Z), 12-16 years, given both stocks similarly mature between ages 2 and 3.

To estimate the minimum time for the stock to reach the rebuilding target (i.e., at or above the LRP with a 75% probability) in the absence of fishing, various productivity scenarios based on recruitment and natural mortality were used to investigate the levels necessary for the stock to rebuild to the rebuilding target using the projection function of the assessment model.

Projections were performed using the project_wham function of the WHAM R package. The base model was projected forward for 15 years using , F=0 and the maturity and weight-at-age in years 2020 to 2024.

Projections were performed using two recruitment scenarios: (1) average recruitment deviations from years 2020 to 2024 to reflect current recruitment levels, and (2) the average recruitment deviations from a period of higher recruitment; years 1985 to 1990. Projections were also performed using the average M by age group in various series of years. The first scenario used the average M by age group from years 2020 to 2024 to reflect current M levels. Three other scenarios used average M by age group in years 1985-1990, 2000-2005 and 2010-2015. Natural mortality scenarios were performed across the two recruitment scenarios, for a total of 6 population forecast.

Only one scenario, in which stock productivity increased through both higher recruitment and lower natural mortality, resulted in the stock reaching the LRP (Figures 35 to 37). In this projection, the probability of biomass exceeding the LRP after 15 years was 70%, which does not meet the rebuilding target. However, uncertainty in projected biomass is substantially greater than the uncertainty associated with biomass estimates during the assessment period. If a similar level of uncertainty were applied to biomass projections over the 15-year period, the rebuilding target would be met. The recruitment scenarios considered are also uncertain, as they assume mean recruitment levels rather than incorporating stock-recruit relationships; if recruitment were to increase beyond these assumed levels in response to biomass increases, the stock could rebuild more rapidly. In addition, potential changes in growth and maturity were not accounted for, which may also influence rebuilding outcomes. Overall, this scenario is highly uncertain and should be interpreted cautiously, serving primarily to illustrate the conditions required for the stock to potentially meet the rebuilding target rather than as a prediction of future stock status.

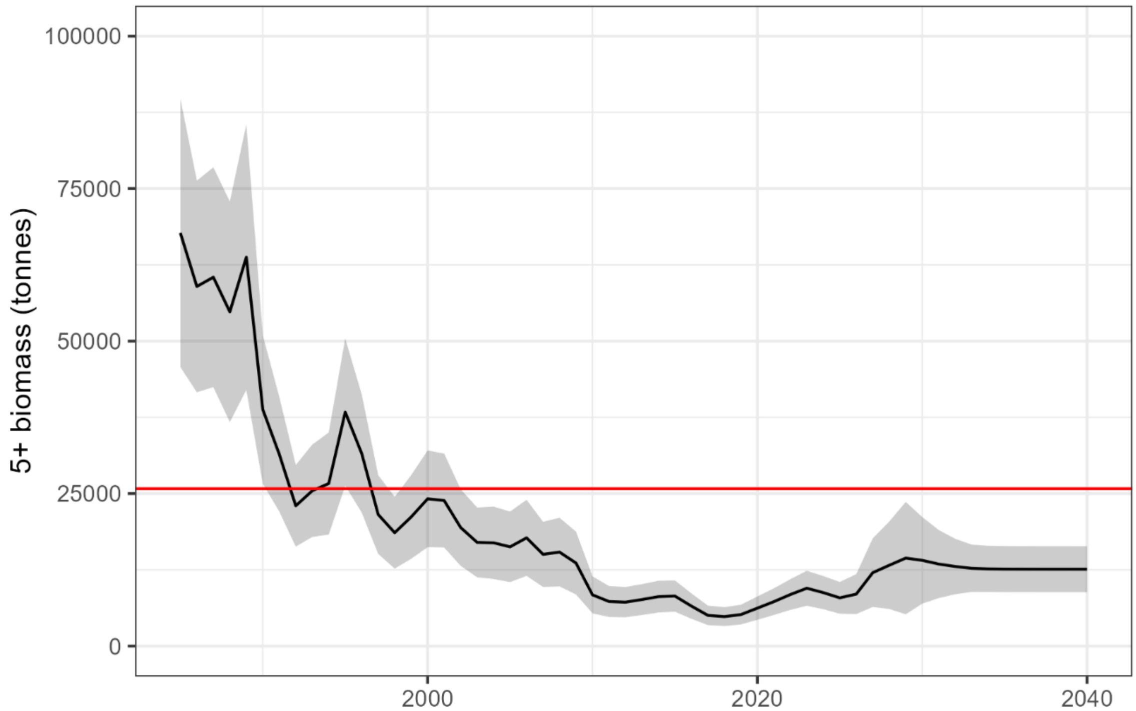

Figure 36: Projected 5+ biomass in current recruitment and M conditions, solid black line is the mean, shading is the 95% confidence intervals and the solid red line is the biomass limit reference point.

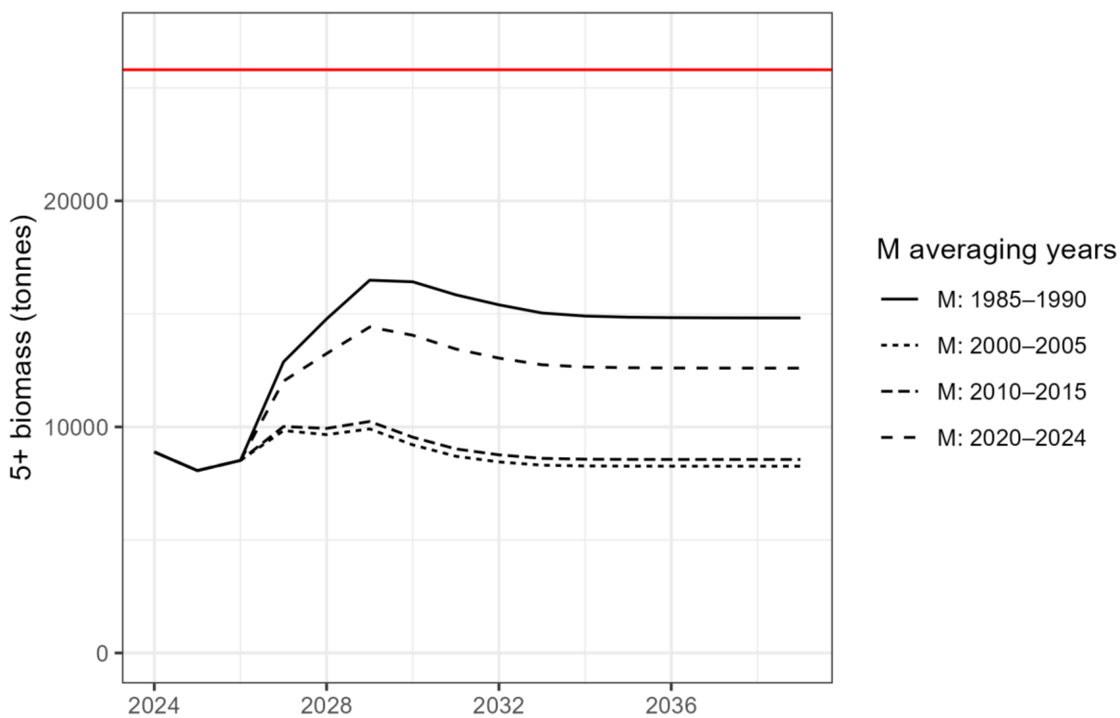

Figure 37:Projected 5+ biomass under historical M scenarios and current recruitment, solid black line is the mean and the solid red line is the biomass limit reference point.

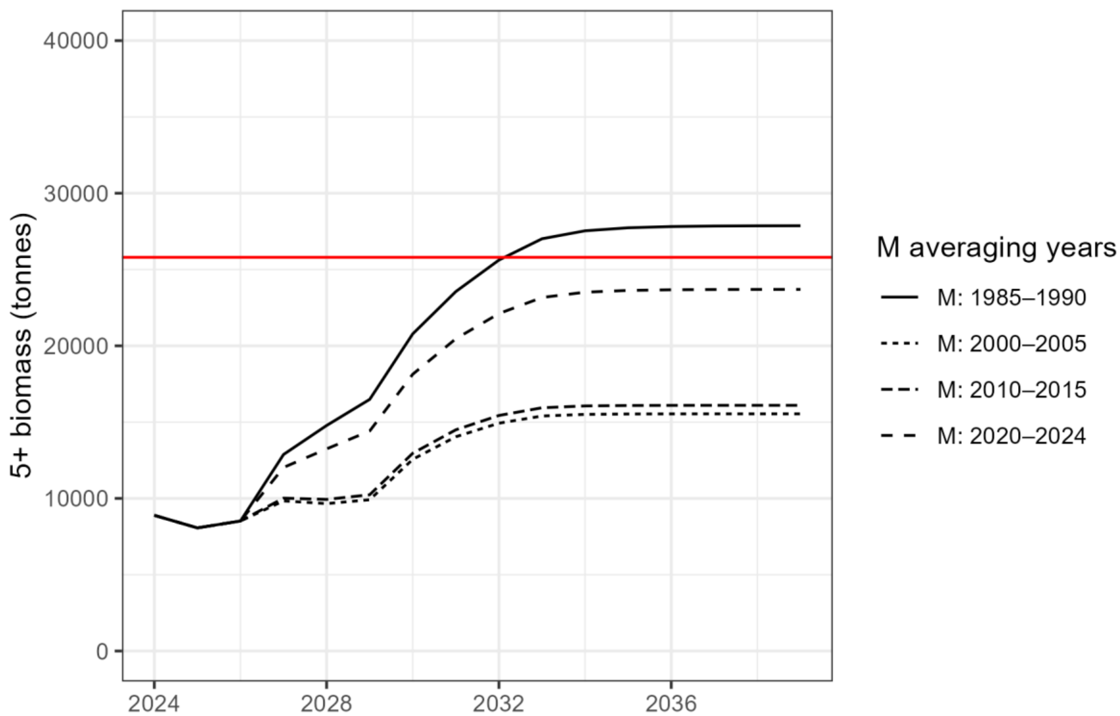

Figure 38: Projected 5+ biomass under historical M scenarios and high recruitment, solid black line is the mean and the solid red line is the biomass limit reference point.

ADDITIONAL MEASURABLE OBJECTIVES

Rebuilding objectives may include other metrics beyond biomass-based measures (DFO 2021).

AGE AND SIZE STRUCTURE

Truncation of the age and size structure of Yellowtail has occurred throughout the time series. The RV survey catch at age shows that the abundance of older fish (ages 7+) were consistently observed in the 1980s and made up 60% of the catch, but were far less common after 1990 (Figure 12). Since 2020, these older fish represent only 0.05% of the catch. There is also a significant loss of large, commercially harvestable fish (25+ cm ; Figure 10). The loss of older and larger individuals can impact reproductive success (discussed above).

A rebuilding plan objective should include an increase in the proportion of Yellowtail aged 7+ to averages observed in the 1980s (60%).

BYCATCH CONSIDERATIONS

Winter Flounder and Windowpane Flounder have been consistently caught in the Yellowtail fishery since 1999, and have collectively exceeded the weight of Yellowtail in this fishery since 2007 (Figure 39). As with Yellowtail, Winter Flounder in the sGSL have been primarily fished to supply bait, and much of the fishery is also concentrated around the Magdalen Islands (DFO 2017). Winter Flounder has been below its LRP since 2006 (DFO 2017). There is no targeted fishery for Windowpane Flounder in the sGSL, and the species is not managed by DFO. Data from the RV survey suggest that Windowpane are primarily concentrated to the south of PEI, with smaller concentrations around the Magdalen Islands (unpublished).

Given the close association between Yellowtail and Winter Flounder, an additional measurable objective could be to consider the two species as one unit when defining the rebuilding target for Yellowtail. In this case, both species would need to be above their LRP in order for either to be considered above the Critical Zone of the PA Framework.

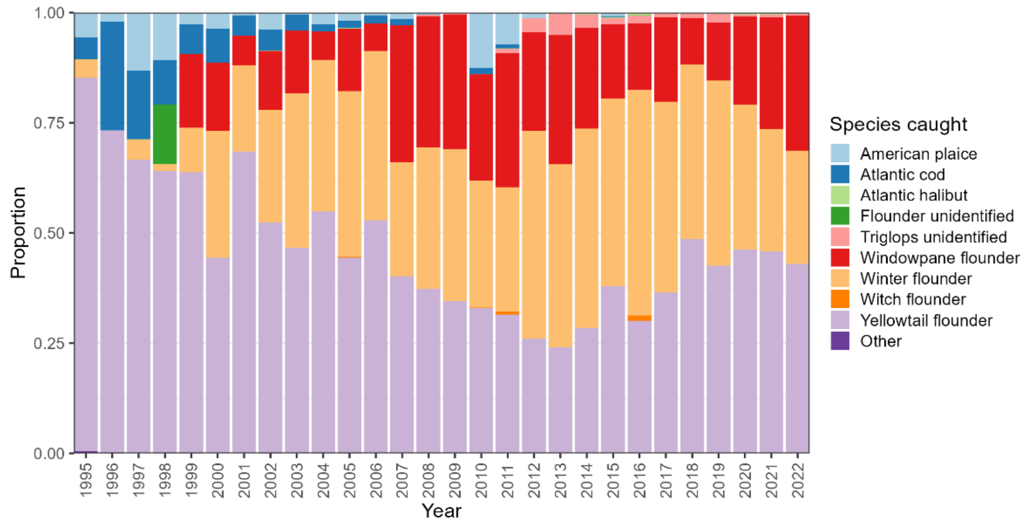

Figure 39: Proportion of total weight caught by species in the Yellowtail Flounder fishery from 1995 to 2022.

HABITAT

Section 2(1) of the *Fisheries Act*, defines fish habitat as “water frequented by fish and any other areas on which fish depend directly or indirectly to carry out their life processes, including spawning grounds and nursery, rearing, food supply and migration areas”. For some stocks, the availability and quality of habitat may be important for rebuilding the stock when tightly linked to stock declines or rebuilding potential. With respect to this definition, habitat loss or degradation is unlikely to have contributed to the stock decline or play a role in preventing the recovery of the stock. sGSL Yellowtail. Overall, the sGSL has experienced a trend towards warmer waters, shorter duration of ice season, and lower ice volume (Galbraith et al. 2025), however there is no indication that this warming has altered the potential habitat for Yellowtail.

HOW TO TRACK REBUILDING PROGRESS

Rebuilding progress will be tracked using the sGSL Yellowtail stock assessment model and monitoring of productivity parameters (natural mortality, recruitment, and growth). Projections and decision tables will be provided to monitor the progress towards attaining objectives of the rebuilding plan. Rebuilding plan progress should be tracked as part of the multi-year assessment cycle which occurs every 5 years. An interim update occurs at the halfway point between full stock assessments. The interim update will consist of an updated indicator of stock status from the RV survey for fish of 25+ cm, with the 1997 biomass as the re-scaled LRP (to correspond with the year the stock crossed the LRP in the stock assessment model; Figure 39).

Figure 40: RV survey catch rates (solid lines) and 3-year moving average (dashed line) of large Yellowtail flounder (25+ cm; kg per tow) and LRP-proxy (red line).

A full stock assessment would be triggered during the interim update if the stock indicator is above the LRP-proxy. Regardless of when a new stock assessment is to be initiated, at least 6-12 months lead time is required before the new stock assessment is initiated to allow for analyses of other indicators and biological sampling that will be needed for the interpretation of the population trajectory.

FREQUENCY OF REBUILDING PLAN UPDATE

The periodic review of the rebuilding plan should be set to the same 5-year multi-year assessment cycle used for the stock assessment of sGSL Yellowtail with an interim update at the halfway point. The interim update consist of an updated indicator of stock status from the RV survey for fish of 25+ cm. Objectives of the rebuilding plan should also be revised and models should be updated if stock productivity or external factors influencing the stock dynamics change.

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TABLES

Table 1: Annual recorded landings in tonnes (t) of Yellowtail flounder and unspecified flatfish in NAFO Div. 4T, 1960 to 2025. Data from 1960 to 1995 are from NAFO files. Data from 1996 to 2025 are from fishery landings and logbooks from the DFO Statistics Branch (ZIFF files).

Year	Yellowtail	Flatfish spp.	Year	Yellowtail	Flatfish spp.
1960	2	2405	1993	88	11
1961	7	2493	1994	62	16
1962	2	1304	1995	208	5
1963	51	0	1996	209	0
1964	39	0	1997	813	0
1965	51	0	1998	182	26
1966	125	0	1999	305	0
1967	55	0	2000	294	3
1968	6	0	2001	317	0
1969	243	0	2002	215	0
1970	44	0	2003	158	0
1971	5	0	2004	192	1
1972	3	1201	2005	175	1
1973	1	1388	2006	182	0
1974	21	602	2007	142	3
1975	0	2464	2008	92	0
1976	29	668	2009	101	0
1977	25	1163	2010	186	0
1978	3	764	2011	181	0
1979	52	841	2012	111	0
1980	41	759	2013	82	0
1981	10	118	2014	86	0
1982	6	344	2015	102	0
1983	26	792	2016	81	0
1984	82	46	2017	135	0
1985	215	3	2018	100	0
1986	396	0	2019	120	0
1987	404	0	2020	136	0
1988	198	0	2021	148	0
1989	43	36	2022	137	0
1990	15	37	2023	0.2	0
1991	54	39	2024	0	0
1992	119	93	2025	0	0

Table 2: Parameter estimates, standard errors, and confidence intervals for the state-space stock assessment model

Parameter	Estimate	Std. Error	95% CI lower	95% CI upper
Mean Recruitment	769799.50	210429.60	450501.50	1315404.00
NAA σ (age 1)	0.48	0.07	0.36	0.64
RV fully selected q	0.30	0.08	0.18	0.49
Seine, Trawl a50 (Block 1)	6.42	0.18	6.06	6.77
Seine, Trawl 1/slope (increasing) (Block 1)	0.57	0.03	0.51	0.63
Seine, Trawl a_50 (Block 2)	4.64	0.06	4.53	4.75
Seine, Trawl 1/slope (increasing) (Block 2)	0.40	0.01	0.38	0.42
Seine a50 (Block 3)	4.44	0.06	4.32	4.57
Seine 1/slope (increasing) (Block 3)	0.40	0.01	0.38	0.42
Trawl a50 (Block 4)	4.74	0.12	4.51	4.98
Trawl 1/slope (increasing) (Block 4)	0.52	0.02	0.48	0.56
RV a50 (Block 5)	6.18	0.22	5.74	6.59
RV 1/slope (increasing) (Block 5)	0.72	0.05	0.63	0.82
RV a50 (Block 6)	4.10	0.06	3.99	4.22
RV 1/slope (increasing) (Block 6)	0.39	0.01	0.38	0.42
RV a50 (Block 7)	3.16	0.05	3.07	3.25
RV 1/slope (increasing) (Block 7)	0.34	0.01	0.32	0.36
Seine age comp, Dirichlet: dispersion (Φ)	95.97	10.83	76.94	119.71
Trawl age comp, Dirichlet: dispersion (Φ)	41.78	3.82	34.92	49.99
RV age comp, Dirichlet: dispersion (Φ)	112.62	9.97	94.67	133.96
Mean log(M) intercept for age 4-7	-0.37	0.11	-0.58	-0.15
Mean log(M) intercept for age 8-9	0.249	0.11	0.033	0.465
Mean log(M) intercept for age 10+	0.096	0.199	-0.294	0.486
M RE σ	0.24	0.053	0.156	0.37
M RE AR1 ρ (year)	0.624	0.187	0.13	0.87

APPENDIX A: AGE-LENGTH KEYS

Otoliths were collected for ageing from the RV survey from 1970 to 1986 and from 2000 to present. Otoliths collected in from 1970 to 1986, were stored in glycerin, whereas contemporary samples were cleaned and stored dried. Commercial samples were collected from the period 1996 to 2006 and again from 2018 to 2022. Strong disparities between agers was described in Surette and Swain (2016) and Rolland et al. (2022). Significant work has been conducted to try to reduce the discrepancies among agers including development of a reference collection, enhance training, annotation of otoliths, increased production of ages, and paired ageing among agers. Despite this, inconsistencies remains. Furthermore, re-ageing of otoliths stored in glycerin are difficult due to degradation of the otoliths, and sometimes reduced or enhanced clarity of rings and checks. A subset of samples from the early time period have been sent for bomb radiocarbon analysis, however the results were not available at the time of this report.

The number of valid aged otoliths by year, source, and ager are shown in Table A1.

Table A1: Number of valid otoliths of Yellowtail Flounder aged per year from the southern Gulf of St. Lawrence multispecies bottom-trawl research vessel (RV) survey and the commercial fishery aged by 4 agers (SABS, GLF1, GLF2, and GLF3).

Year	Source	SABS	GLF1	GLF2	GLF3
1972	RV	167	.	.	.
1973	RV	184	20	.	110
1974	RV	286		.	.
1975	RV	195	19	.	.
1976	RV	185	20	.	.
1977	RV	271	21	.	.
1978	RV	358	20	.	67
1980	RV	312	20	.	23
1981	RV	235		.	.
1982	RV	279	279	.	.
1983	RV	.	.	.	67
1984	RV	.	191	.	.
1986	RV	143	143	.	43
1996	Commercial	22	.	.	21
1997	Commercial	.	.	.	117
1999	Commercial	.	.	.	145
2000	RV	.	353	.	186
2001	RV	.	197	148	179
2002	RV	.	136	.	.
2003	RV	.	198	134	191
2004	RV	.	.	.	.
2005	RV	.	187	146	170
2006	RV	.	.	.	179
2007	RV	.	346	.	186
2008	RV	.	.	.	182
2009	RV	.	165	133	179
2010	RV	.	.	.	179
2011	RV	.	20	122	195
2012	RV	.	.	.	187
2013	RV	.	150	132	191
2014	RV	.	133	.	183
2015	RV	.	131	.	.
2016	RV	.		.	190
2017	RV	.	148	131	194
2018	RV	.	20	.	173
2018	Commercial	.	.	.	46
2019	RV	.	.	660	.
2019	Commercial	.	.	.	75
2020	RV	.	168	.	190

Year	Source	SABS	GLF1	GLF2	GLF3
2020	Commercial	.	.	.	12
2021	RV	.	20	.	189
2021	Commercial	.	.	.	46
2022	RV	.	15	.	172
2022	RV	.	5	.	88
2022	Commercial	.	.	.	44
2023	RV	.	20	.	188
2024	RV	.	70	.	189

For this report, the decision was made to proceed with ages provided by ager GLF3 given the continued discrepancies between agers and the internal consistency of GLF3. Ager GLF3 aged the reference collection to evaluate and ensure the consistency of age reading over time. A sub-sample of 100 randomly selected Yellowtail otoliths from across years, size, and sex were aged by GLF3 and compared to the reference ages. The results show an overall agreement of 92% (mean coefficient of variation of 0.3) in 2026 (Figure A1). For older Yellowtail, GLF3 tends to underestimate the age.

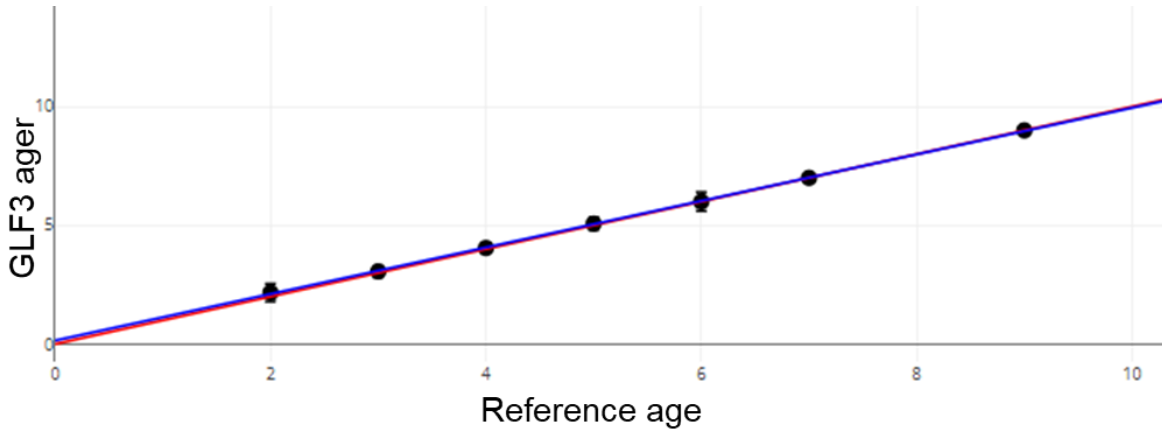

Figure A1: Comparison of ages obtained by GLF3 during the 2026 reference collection test. Points and bars indicate the average age with the coefficient of variation and are represented by the blue line. The red line indicates the expected reference age.

Ages of GLF3 were pooled as described in the Ageing and Growth section and decadal ALKs (Figure A2) were constructed to use in stock assessment and rebuilding plan advice.

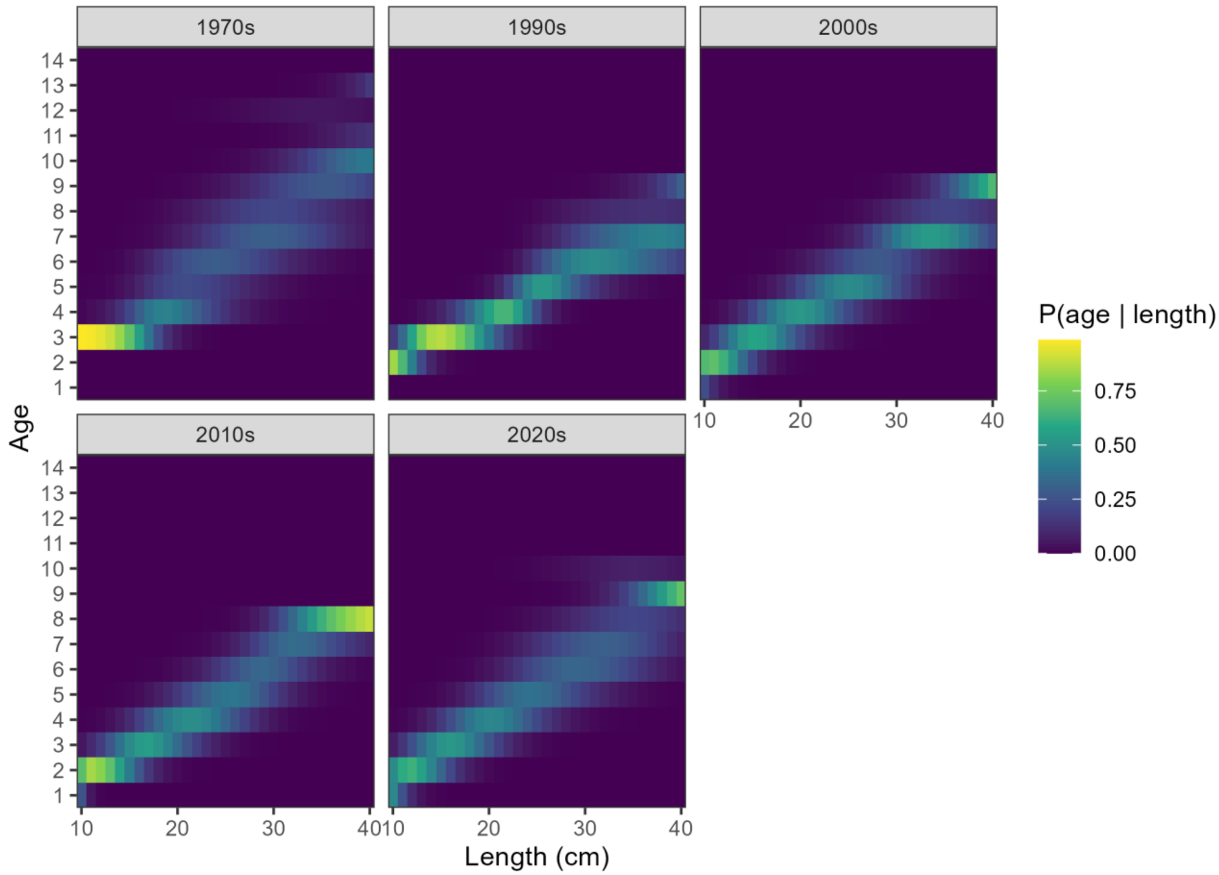

Figure A2: Age length keys by decade for Yellowtail flounder.

APPENDIX B: LENGTH BASED MODEL OUTPUT

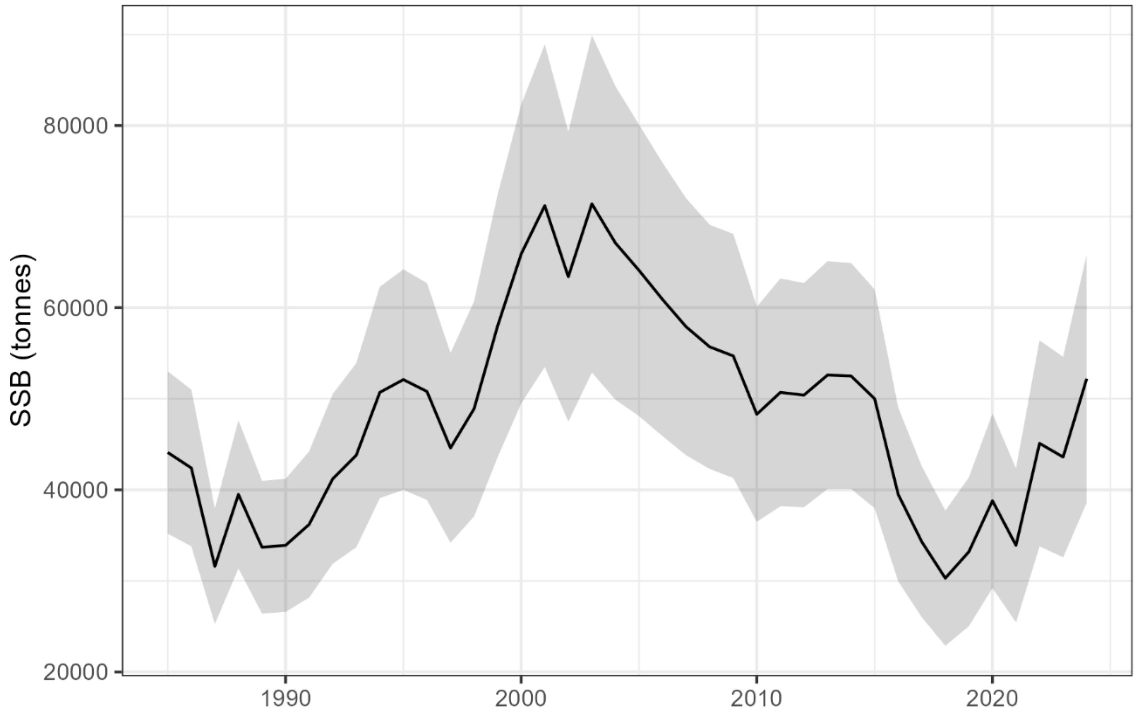

Figure B1: Yellowtail flounder spawning stock biomass (SSB; tonnes), between 1985 and 2024 from the length based stock assessment model.

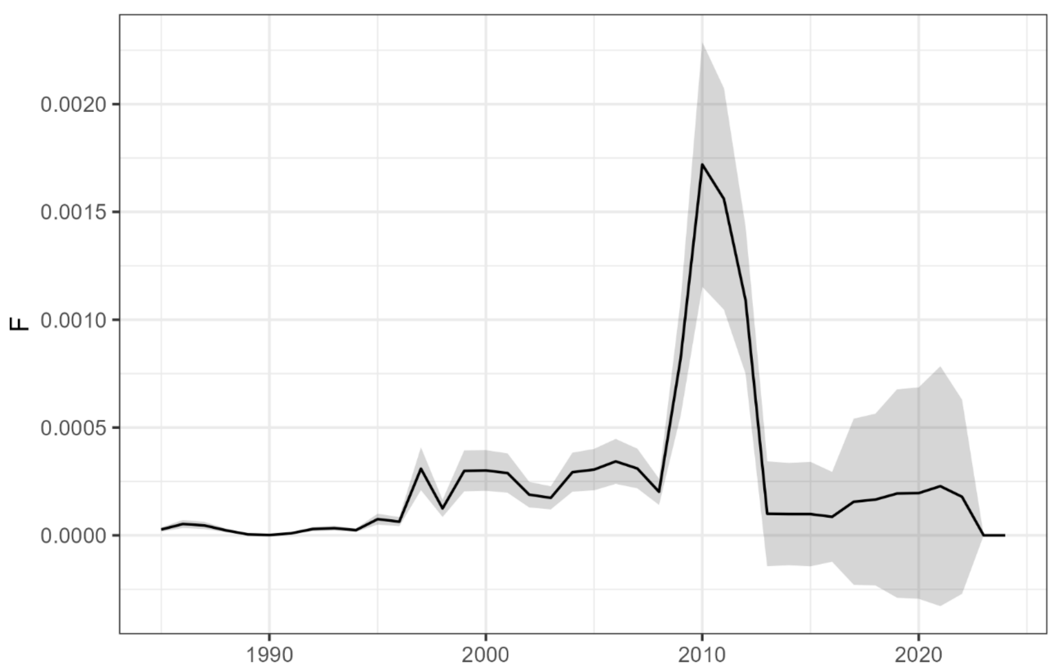

Figure B2: Yellowtail flounder fishing mortality (F), between 1985 and 2024 from the length based stock assessment model.

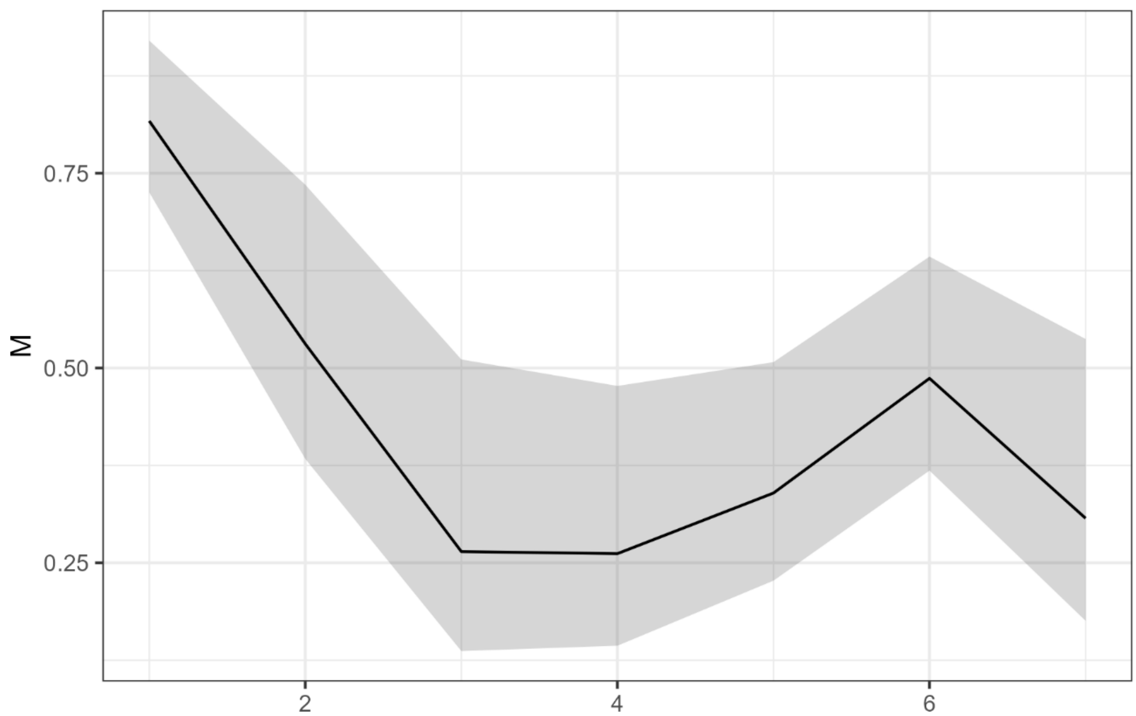

Figure B3: Yellowtail flounder natural mortality for ages 1-3 for year blocks between 1985 and 2024 from the length based stock assessment model.

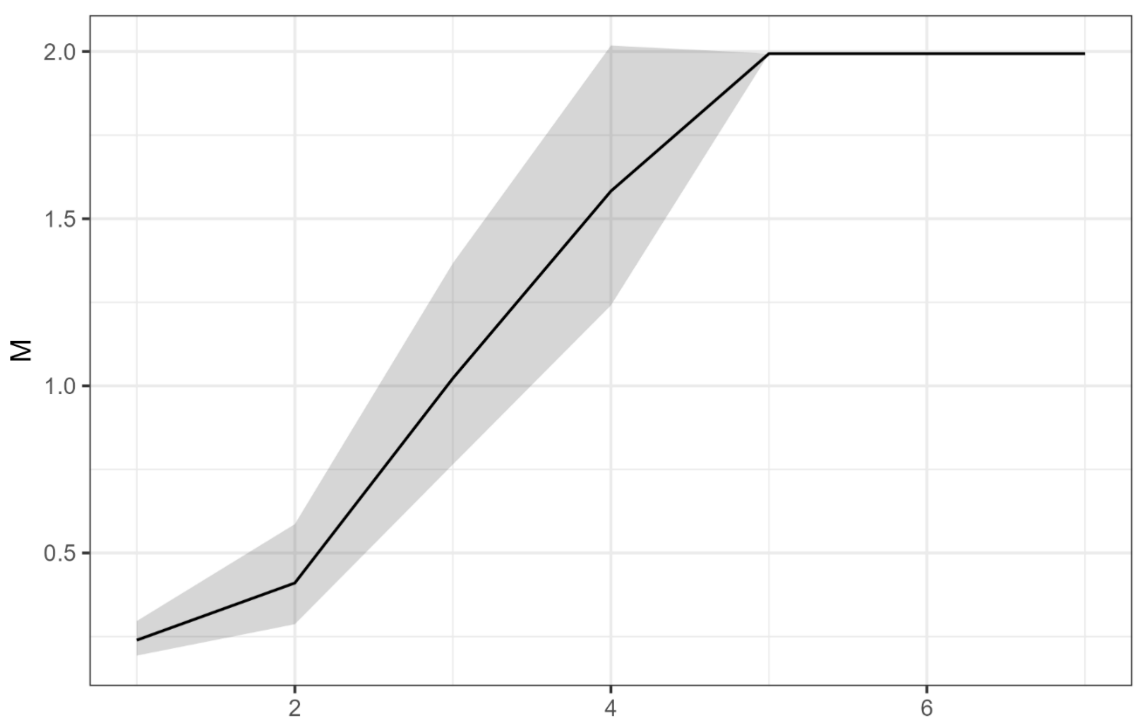

Figure B4: Yellowtail flounder natural mortality for ages 5-10 for year blocks between 1985 and 2024 from the length based stock assessment model.

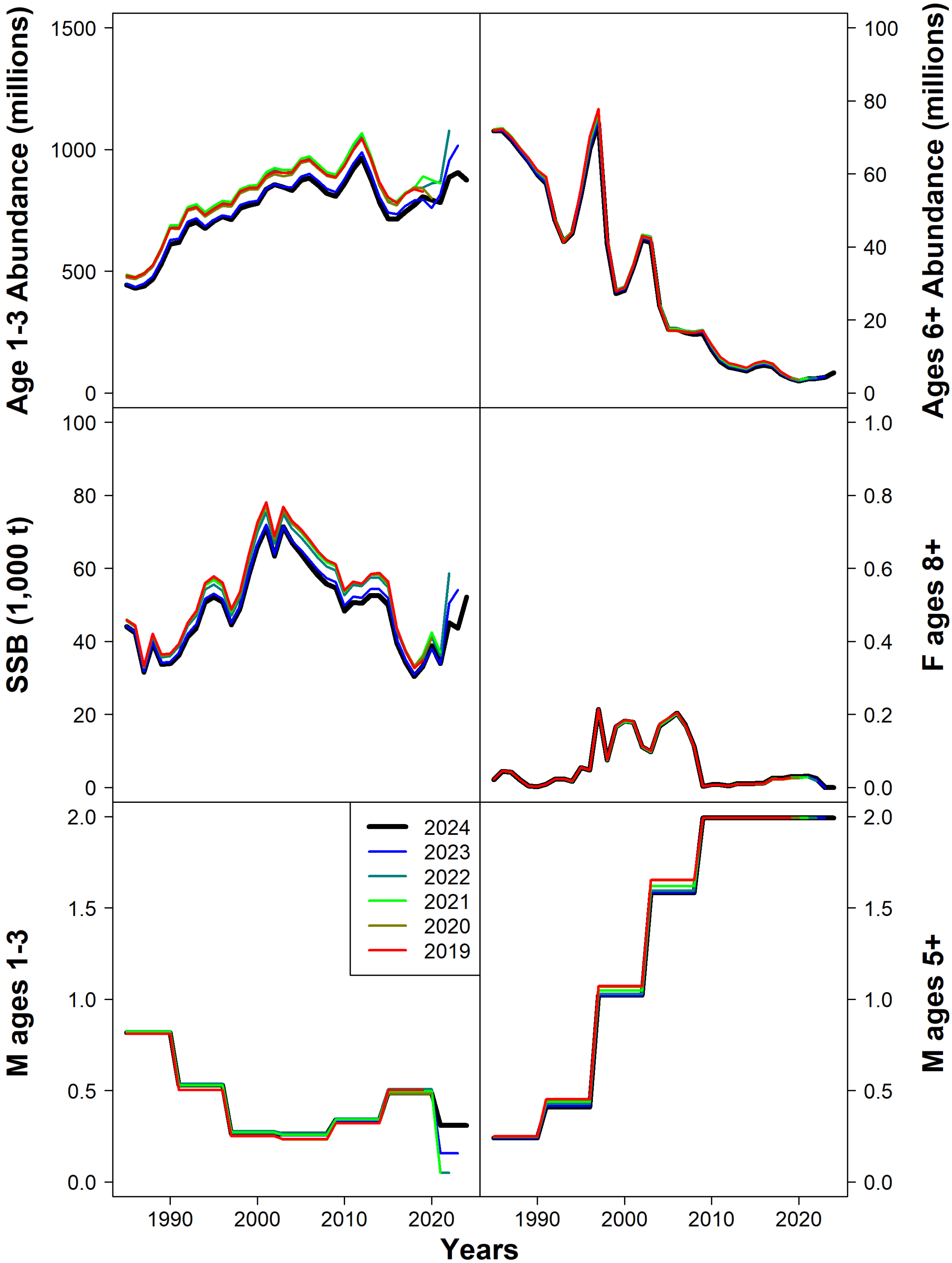

Figure B5: Yellowtail flounder retrospective patterns from the length based stock assessment model.

APPENDIX C: M PRIOR

Time varying M of two age groups (1-3 and 5-8+) was estimated separately for five time blocks (6-year blocks between 1985 and 2008 and a final 7-year block). M of 4-year olds was calculated as the average of M of both age groups in a given year. A prior for M was also used for the first time block (1985 to 1990). Prior means were 0.8 and 0.3 for initial M's for ages 1-3 and 5+, respectively (with standard deviation (SD)=0.05 or 0.025, respectively). In both assessments, M for the older age group was at the upper bound of the parameter in the terminal year block (M=2.0). The population scale was also sensitive to the number of years in time blocks.

Sensitivity runs showed that the mean M or annual deviations in M were estimable for the age group 1-3. Accordingly, M was modeled as a fixed mean of 0.7, based on the mean M estimates across life-history based methods for these ages (Cope and Hamel 2022; Appendix B). Mean M was estimated for three age groups 4-7, 8-9 and 10+). For these, process error in M was modeled as AR1 random effects over years.

To inform the mean M for ages 1 to 3, the natural mortality tool: empirical estimators of Natural Mortality (M) was used (Cope and Hamel 2022). A Von Bertalanfy model was fit to age and length data from all available years using the FSA R package (Ogle 2016). The von Bertalanffy growth model provided a good fit to the length–at–age data and converged after five iterations. Estimated asymptotic length (L_∞) was 41.48 (SE = 0.83), growth coefficient (K) was 0.150 year^{-1} (SE = 0.0068), and theoretical age at zero length (t_0) was −0.80 (SE = 0.075). All parameter estimates were highly significant ($p < 0.001$). Longevity was set to 20 years.

The mean natural mortality at age 1 to 3 from the Gislason, Charnov and CnW methods was 0.7.

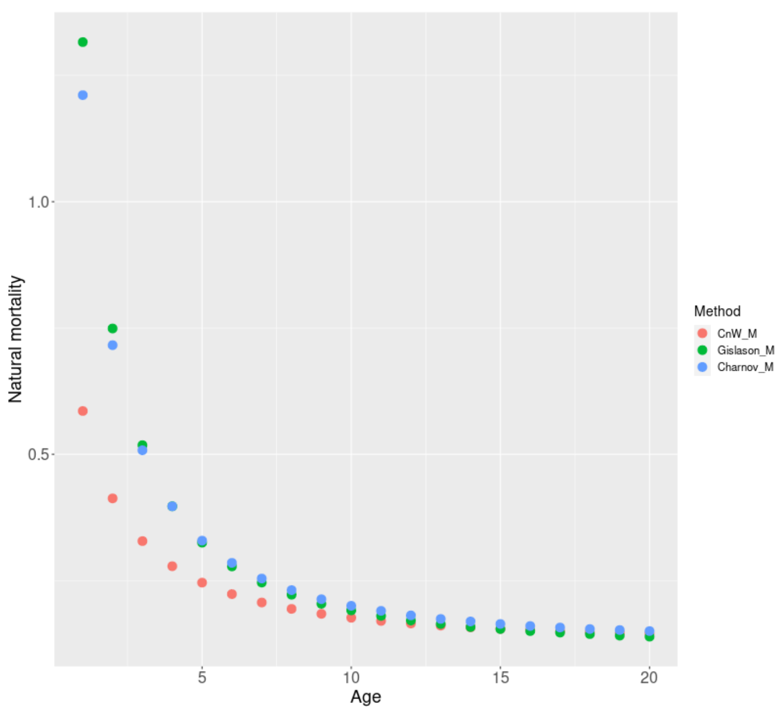

Figure C1: Natural mortality at age estimates by method(colored points).