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Sammanfattning

Analytiska modeller för att bedöma statusen hos fiskade bestånd används rutinmässigt inom fiskeriförvaltningen. Sådana modeller kan uppskatta lekbiomassa, rekrytering, fångstnivåer och referenspunkter för hållbart nyttjande av bestånd som fiskas kommersiellt.

Datakraven för analytiska modeller varierar beroende på modellen men omfattar i allmänhet beståndets ålderssammansättning, könsmognad, rekrytering och förhållandet mellan lekbiomassa och rekrytering, naturlig dödlighet, fiskeridödlighet, tillväxtparametrar, fångst och ansträngning inom fisket, fiskerioberoende beståndsindex, redskapsselektivitet och längd och viktsamband. Detaljerad kunskap om fiskets fångstsammansättning (längd, ålder och selektivitet) samt tidsserier för sådana data är en särskilt begränsande faktor för den allmänna tillämpningen av analytiska modeller för nationella bestånd.

Här redovisar vi utvecklingen av en beståndsmodell för gös i Hjälmaren. Gösbeståndet i Hjälmaren har stor betydelse för fisket och därför har också en hel av de data som modellen kräver samlats in under årens lopp i olika övervakningsprogram. Ingående data uppvisar i viss mån motstridiga trender, vilket innebär att resultaten från modellen bör tolkas försiktigt.

Enligt modellen är gösbeståndet i Hjälmaren för närvarande på tillbakagång, med minskande lekbiomassa och vissa tecken på försvagad åldersstruktur. Beståndet anses dock ha återhämtningspotential, vilket stöds av förekomsten av enstaka starkare årsklasser som snart kommer att gå in i fisket. Förvaltningsmål med tillhörande biologiska referenspunkter har ännu inte förankrats av förvaltningen. I ett test med rekommenderade referenspunkter för andra bestånd ($F_{40\%}$, $B_{40\%}$; Mace, 1994), indikerar modellresultaten att beståndet är över dessa gränser, trots den minskande lekbiomassan.

Även om modellen kan ge en värdefull baslinje för framtida förvaltning av gös i Hjälmaren är det motiverat att använda resultaten med stor försiktighet på grund av osäkerhet och databegränsningar. Prioriterade områden för att förbättra modellen i framtiden är att kvantifiera nya CPUE index från fisket, fångsterna i fritidsfisket samt att utöka den pågående övervakningen av gösbeståndet. Tills mer och bättre data har samlats in, och modellen ger mer tillförlitliga resultat, rekommenderar vi att bedömningar av Hjälmarens gösbestånd fortsatt bedöms enligt standarden i Fiskbarometern.

Summary

Analytical stock assessment models are routinely used in fisheries management. These models can estimate spawning stock biomass, recruitment, catch levels, and biological reference points for the sustainable use of commercially exploited fish stocks.

The data requirements for analytical models vary depending on the model type, but generally include information on age composition, maturity, recruitment and the relationship between spawning biomass and recruitment, natural mortality, fishing mortality, growth parameters, catch and effort, fishery-independent indices, gear selectivity, and length–weight relationships. A particularly limiting factor for applying these models to national stocks is the lack of detailed knowledge on catch composition (length, age, selectivity) and the absence of long time series of such data.

In this report, we present the development of an analytical assessment model for pikeperch in Lake Hjälmaren. The stock is of high importance to both commercial and recreational fisheries, and therefore many of the necessary data components have been collected over the years through various

monitoring and research programs. Some input datasets show conflicting trends, which means the results should be interpreted with caution.

According to the model, the pikeperch stock in Hjälmaren is currently in decline, with decreasing spawning stock biomass and some signs of a weakened age structure. Nevertheless, the stock appears to have recovery potential, supported by the presence of a few stronger year classes that will soon enter the fishery. Management goals with associated biological reference points have not yet been established by managers. A test using recommended reference points from other stocks ($F_{40\%}$, $B_{40\%}$; Mace, 1994), suggest that the stock is currently above $B_{40\%}$, despite the recent decline in spawning stock biomass.

Although the model can provide a valuable baseline for future management of pikeperch in Lake Hjälmaren, it is important to apply the results cautiously due to the data limitations and uncertainty. Key priorities to improve the model include quantifying recent CPUE index for the commercial fishery, adding recreational catches and expanding biological monitoring of the stock. Until more and better data have been collected and the model produces more reliable estimates, we recommend that assessments of the Hjälmaren pikeperch stock continue to follow the current standard applied in Fiskbarometern (SLUs current system for assessing the status of commercial fish stocks).

Contents

1. Background

Pikeperch (*Sander lucioperca*) is the most ecologically and economically important fish species in Lake Hjälmaren, supporting both commercial and recreational fisheries. Currently, assessments of the biological status of the stock relies on an indicator framework based on mortality, biomass and size structure (Larsson, et al. 2024). Data availability suggests that analytical model frameworks may be possible to apply.

This report presents the development of a full analytical stock assessment of the pikeperch population in Lake Hjälmaren using the Stock Synthesis (SS3) modelling framework. The assessment draws on multiple data sources, including commercial catch and effort data, fishery independent surveys and biological sampling (age and length composition) to estimate key stock parameters such as spawning stock biomass (SSB), fishing mortality (F), and recruitment (R).

In addition to presenting the results of the reference model, we explore the sensitivity of the assessment to alternative model configurations, retrospective patterns, and uncertainties in key data sources. We also implement age-based indicators (ABI) as a complementary approach for evaluating long-term changes in stock structure and resilience (Griffiths, et.al. 2024).

The aim of this work is to provide a science-based foundation for future management advice and to support the development of a formalized management framework for pikeperch in Lake Hjälmaren. While the assessment model offers valuable insights, its outputs should be interpreted with an understanding of the data limitations and uncertainties highlighted throughout the report.

1.1 Stock definition

Geographical boundaries

The home range of pikeperch in large lakes and coastal areas of the Baltic Sea typically varies between 10 and 100 km (Saulamo and Neuman, 2002; Saulamo and Thoresen, 2005; Andersson et al. 2015; Östman et al. 2017). Mark-recapture studies have shown that pikeperch of all ages and especially younger individuals, tend to inhabit shallower areas during the summer, which is the growing season, they then migrate to deeper waters in the autumn (Nyberg et al. 1996; Andersson et al. 2015).

Pikeperch also exhibit homing behaviour. Individuals tagged at spawning grounds have been observed returning to the same locations to spawn in subsequent years (Jepsen et al. 1999; Lappalainen et al. 2003; Saulamo and Thoresson, 2005). Transplantation experiments further support this, showing that pikeperch can navigate back to their site of origin (Keskinen et al., 2005; Lehtonen and Toivonen 1987).

In a mark–recapture study conducted in Lake Hjälmaren (Nyberg et al., 1996), the displacement of individuals was generally limited to 1–5 km, with larger fish tending to move greater distances. Fish tagged in the shallow western basin migrated further, likely in search of deeper waters and alternative prey resources during autumn. In contrast, in the eastern basin, where shallow and deep habitats co-occur, fish movements were relatively limited.

Genetic structure

A study on pikeperch genetics from different parts of Lake Hjälmaren found no significant genetic differentiation (Dannewitz et al. 2010). This suggests that the pikeperch in Hjälmaren constitute a single, genetically homogeneous population, and can therefore be managed as a single stock.

1.2 Impact factors

Commercial and recreational fisheries

In Lake Hjälmaren, the primary target species for commercial fisheries are pikeperch and signal crayfish (*Pacifastacus leniusculus*; Jordbruksverket, 2020). Pikeperch are harvested using fixed trap nets (spring to autumn) and gillnets (year-round, including under-ice fishing). Trap nets typically begin capturing pikeperch at around two years of age, with peak catchability observed at five to six years. Mesh size in both the leader arms and cod end of these nets varies depending on gear licensing, particularly for fishers permitted to target eel, where smaller mesh sizes are mandated, increasing the likelihood of bycatch of juvenile pikeperch. Undersized fish and incidental catches are generally released, and post-release survival of pikeperch is reportedly high (Nyberg et al., 1996). Gillnets targeting pikeperch employ mesh sizes of at least 120 mm. This is the legal minimum mesh size, which was raised from 100 mm to 120 mm at the same time as the legal minimum landing size of pikeperch was raised from 40 to 45 cm.

Recreational fishing in Lake Hjälmaren is open-access for angling with handheld gear and does not require catch reporting. Other gear types, such as gillnets, pots, and trolling, are permitted in private waters and to some extent in common waters without effort restrictions. Technical regulations, including minimum mesh sizes, also apply to these areas, and recreational catches may legally be sold. However, quantifying recreational fishing effort and harvest remains challenging due to the absence of a licensing or mandatory reporting system.

The primary source of recreational fishery data is the annual national survey conducted by Statistics Sweden (SCB) on behalf of the Swedish Agency for Marine and Water Management (SwAM). This survey provides point estimates with 95% confidence intervals for key variables such as total, kept, and released catch, and release rates.

Based on the annual survey, the four most frequently targeted species in the recreational fishery (including sport fishing, net fishing and other gears on private waters) in Lake Hjälmaren are perch and pike (each targeted by approximately 40% of surveyed recreational fishers), followed by pikeperch and signal crayfish, each targeted by roughly 20% (based on a sample size of 31 respondents; Sundblad et al., 2025).

The average annual recreational catch for 2019–2023 (95% confidence interval) was estimated at 35 tons (± 26 tons) for perch and 61 tons (± 50 tons) for pike. Catch estimates for crayfish were very uncertain, with an estimated kept catch of 215 tons (± 197 tons). For pikeperch, the mean annual catch (2021 – 2023) was estimated at 110 tons (74 – 220 tons), of which 24 (6.8 – 41 tons) were released. This yields an estimate of 86 tons kept, but with large uncertainty. Considering the uncertainty, these figures suggest that recreational fishing mortality for pikeperch may be comparable to licensed fishing and is likely underrepresented in current assessments. It is unknown to what extent recreational catch is sold.

2. Data

An overview of the datasets included in the model is shown in Figure 1. Additional data on individual length and weight, not shown in the figure, were used to estimate length-weight relationship.

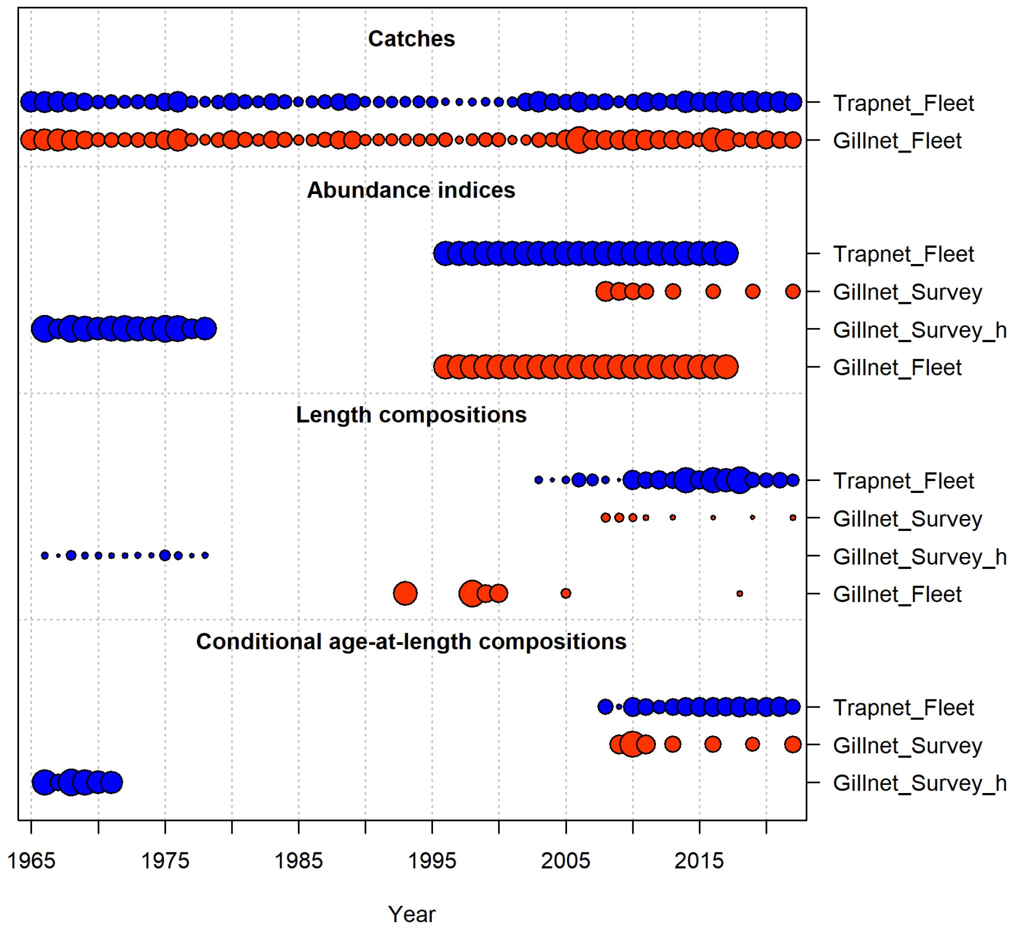

Figure 1. Summary of input data included in the model. Circles are proportional to total annual catch for catches; to precision for abundance indices; and to total sample size for length and conditional age-at-length (age-length key, ALK) compositions. Note that since the circles are scaled relative to maximum within each type, scaling should not be compared between series.

2.1 Commercial and recreational fisheries

2.1.1 Landings

Fisheries landings of pikeperch come from two types of gears, gillnets and trap nets, represented as two separate fleets (Figure 27). However, prior to 1996, total landings were reported without accounting for gear type, therefore the mean ratio between landings by gear in 1996-2021 was applied to separate total landings by gear before 1996. A minimum landing size regulation was implemented for the stock, which was 40 cm prior to 2001 and 45 cm from 2001 onwards. Landings in the reference case model only include commercial fisheries and not recreational. A model with different levels of recreational catches was used in the sensitivity analysis.

2.1.2 Discards

Both trap net and gillnet fisheries incidentally capture pikeperch below the minimum landing size. However, gillnets are generally more size-selective, resulting in proportionally lower discard rates. In contrast, trap nets tend to produce higher volumes of bycatch. Despite this, post-release survival in trap net fisheries is believed to be relatively high, whereas discard mortality is likely to be substantially greater for gillnet-caught individuals (Nyberg et al. 1996).

2.2 Abundance indices

2.2.1 Commercial fisheries

A biomass index, expressed as catch per unit effort (CPUE in kg per 1000 m net per night for gillnets and kg per night for trap nets), was estimated based on fisheries-dependent data for the period 1995–2017. CPUE values were calculated separately for each gear type to account for differences in fishing method and efficiency. Data originates from official statistics and the monthly log-books provided by the fishers to the responsible agency. Previously the Swedish Board of Fisheries was responsible for catch and effort statistics. Since 2011, SWaM has overseen the statistics, and SLU Aqua continued calculating an effort index for several years. However, due to difficulties with how effort was reported, SLU could no longer maintain the time series, and the index ended in 2017.

2.2.2 Fisheries independent surveys

Fisheries independent CPUE was calculated from standardized monitoring programs conducted across Sweden's four largest lakes. Due to differences in gear design and sampling protocols, the survey data are divided into two periods: historical (1966-1978) and recent (2008-2022).

The recent gillnet survey index (denoted gillnet_survey) is based on monitoring data collected using the standard multi-mesh gillnet "Bkust9+2" (55 m in length and 1.8 m in depth), which includes the following mesh sizes: 30-15-38-10-48-12-24-60-19-6.25-8 mm. The recent survey site has also changed over time (Figure 2 and Table 1). One area was surveyed between 2008-2011 and another area in 2013, 2016, 2019 and 2022. Both surveys were carried out in the main basin of Storhjälmaren, although the earlier (2008–2011) sampling area was generally shallower and more affected by signal crayfish presence. Surveys are done in August with 32 nets set overnight (Sandström et al. 2016, Axenrot et al. 2024).

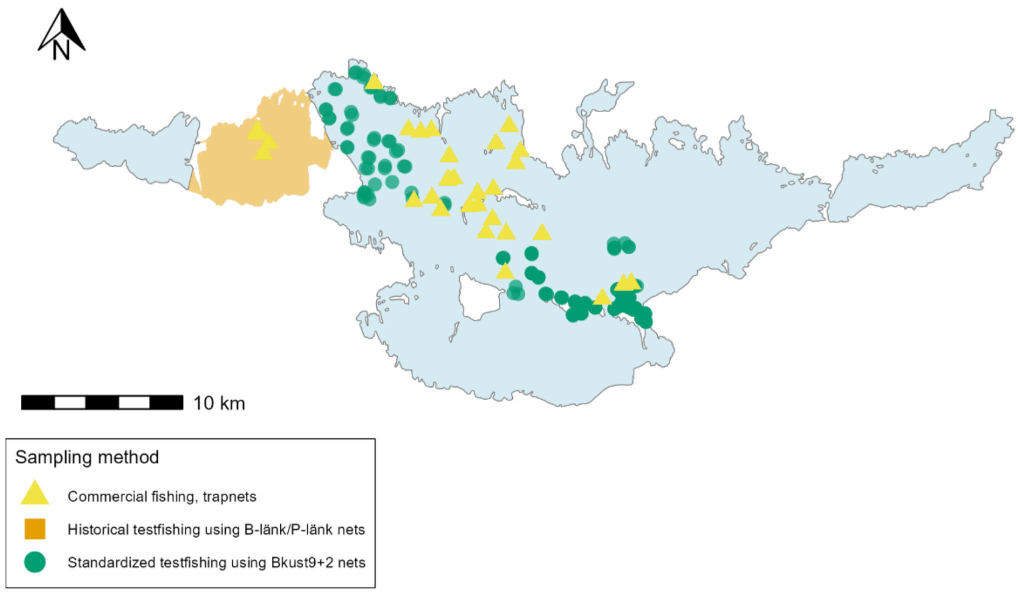

Figure 2. Map of sampling locations in Hjälmaren. Since coordinates for historical survey locations are unavailable, orange polygon depicts their approximate area

Table 1. Scientific gillnet survey description

Year	Area	Net name	Sample size	Sample weight	Number of_nets
1955	Mellanfjärden	Blänk	33	9750	36
1956	Mellanfjärden	Blänk	68	28550	30
1957	Mellanfjärden	Blänk	31	16750	30
1958	Mellanfjärden	Blänk	60	31950	50
1959	Mellanfjärden	Blänk	68	29350	54
1960	Mellanfjärden	Blänk	44	23650	60
1962	Mellanfjärden	Blänk	44	16350	60

Year	Area	Net name	Sample size	Sample weight	Number of_nets
1963	Mellanfjärden	Blänk	99	48635	60
1964	Mellanfjärden	Blänk	109	50750	60
1965	Mellanfjärden	Blänk	137	70200	90
1966	Mellanfjärden	Blänk	253	136700	90
1967	Mellanfjärden	Blänk	129	90350	90
1968	Mellanfjärden	Blänk	185	107850	87
1969	Mellanfjärden	Blänk	123	54900	60
1970	Mellanfjärden	Blänk	96	46900	60
1971	Mellanfjärden	Blänk	92	41700	60
1972	Mellanfjärden	Blänk	88	39050	60
1973	Mellanfjärden	Blänk	82	47250	60
1974	Mellanfjärden	Blänk	93	56800	60
1975	Mellanfjärden	Blänk	159	70950	60
1976	Mellanfjärden	Blänk	113	49150	60
1977	Mellanfjärden	Blänk	46	21430	60
1978	Mellanfjärden	Blänk	75	41300	60
2008	Storhjälmaren	Bkust9	166	30637	39
2009	Storhjälmaren NV	Bkust9+2	143	19812	25
2010	Storhjälmaren NV	Bkust9+2	125	12285	24
2011	Storhjälmaren NV	Bkust9+2	77	8661	24
2013	Storhjälmaren SO	Bkust9+2	61	13155	32
2016	Storhjälmaren SO	Bkust9+2	50	9791	32
2019	Storhjälmaren SO	Bkust9+2	38	8138	32
2022	Storhjälmaren SO	Bkust9+2	57	11899.1	32

The historical gillnet survey index (denoted *gillnet_survey_h*) is based on data from surveys conducted in the 1960s using a different net type, “Blänk” (Svärdsson and Molin, 1966; 1968; 1973; 1981). These nets were 30 m long and 1.5 m deep, with various mesh sizes (9, 11, 13, 16, 18 and 20 varv/aln, which is equal to 67, 54, 46, 37, 33 and 30 mm bar mesh) linked together. The historical survey was performed at a third site, Mellanfjärden, a shallow basin located northwest from the main central basin (shown in orange in Figure 2). Surveys were conducted in early to mid-September with one net for ten nights in a row, per year. CPUE was estimated as the average biomass per day.

The stock assessment model assumes that the surveys are all representative of the same population. The differences among the survey sites and gears have initially been examined in a previous version of the model, where we attempted to develop a joint stock index. In the model described here however, we used “raw” (not standardised) index data without accounting for these differences. The index was estimated as total weight of pikeperch caught per net/night/year.

2.3 Biological data

2.3.1 Pre-model analysis

Stock assessment models require several life-history parameters, which were estimated from biological sampling from both fisheries and scientific survey. For growth parameters the von Bertalanffy function: $length = L_\infty \cdot (1 - e^{-k \cdot (age - t_0)})$, where L_∞ is asymptotic length, k is growth coefficient and t_0 is age, when length is 0 (von Bertalanffy, 1938) was fitted to the length at age data, separately for beginning of time series (scientific survey from 1966-1971) and more recent period (from both trap net fisheries and scientific survey 2008-2022). Data from the earlier time series were however not sufficient to get significant estimates of parameters, thus estimates from the recent period (Figure 3) were used for the entire time series (see Figure 4 for how those estimates fitted data from beginning of time-series).

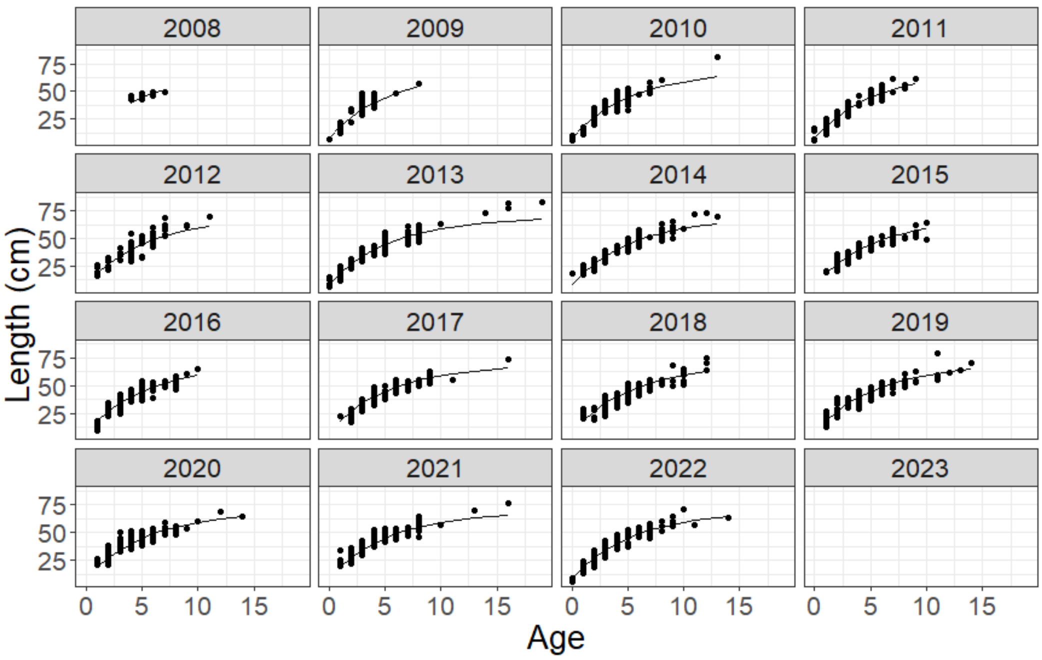

Figure 3. Estimation of L. Hjälmaren pikeperch growth. The line shows estimated growth obtained by fitting a von Bertalanffy growth function to the length at age data from 2008-2022 (dots).

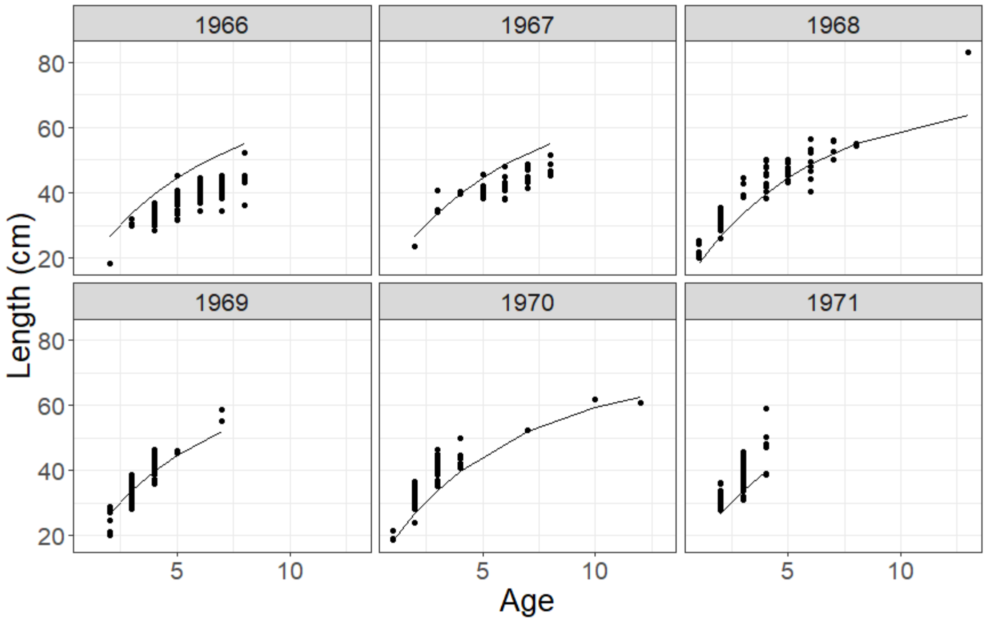

Figure 4. Applying estimates of L. Hjälmaren pikeperch growth. Length at age data from 1966-1971(dots) were insufficient for growth estimation and thus growth estimates (line) from 2008-2022 were used also for those early years.

For length- weight relationships, a non-linear function was fitted to the weight at length data pooled over the years (from both trap net fisheries and scientific survey): $weight = \alpha \cdot length^{\beta}$ (Figure 5, Table 2).

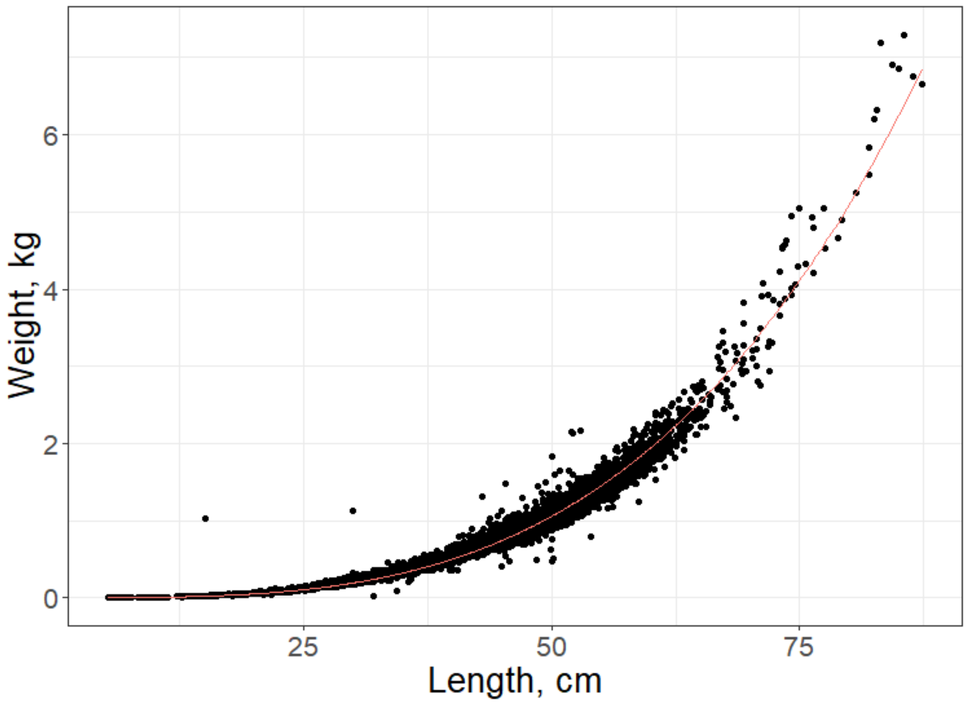

Figure 5. Estimation of weight-length relationship (line) from data (dots).

To estimate a maturity ogive (proportion of mature fish at length) various maturity stages were divided into immature and mature and pooled over the years (from both trap net fisheries and scientific survey). Thereafter the proportion of mature individuals at each length was estimated and non-linear function was fitted and weighted by the number of fish at length (Figure 6): $proportion\ mature = \frac{1}{1+e^{(-\alpha\cdot(length-l50))}}$, where l_{50} is length at which 50% of population is mature.

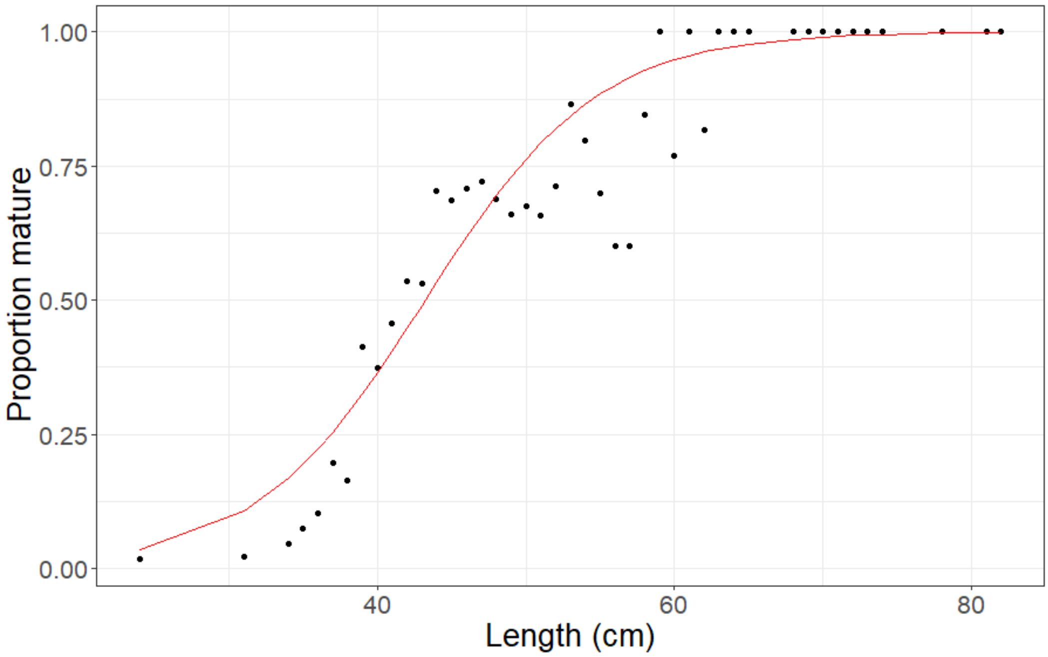

Figure 6. Estimation of maturity ogive (line) from proportion of mature individuals at length (dots).

2.3.2 Data used within model

Size structure

Annual length distribution data used in the model are based on samples collected from both fisheries-independent monitoring surveys (historical and recent) and from the commercial fishery (commercial catch sampling).

Age-length key (ALK)

To construct an age-length key, the number of fish of a specific length at each age group were estimated per year separately for each fishery - trap nets, historical survey (1966-1971) and recent survey (2009-2022). There was no data from fisheries using gillnets.

2.4 Data processing

Size structure

To reflect differences in discard survival between gear types, length distribution data were processed accordingly. For trap net catches, individuals below the minimum landing size were excluded from the dataset, based on the assumption of high post-release survival (38.6-74.4% recapture rates of fish 20-40 cm length; Nyberg et al., 1996). For example, almost 50% of the fish recaptured once and released were recaptured a second time. Also, the proportion of fish recovered increased with the number of times of recapture (from 38.6% at first recapture, to 48.3% for the second recapture, 65.7% a third time and 74.4% recaptured a fourth time). The highest number of recaptures was one individual who was recaptured 40 times (Nyberg et al., 1996). Unfortunately, however, the study by Nyberg et al. (1996) did not estimate survival rates, due to an unknown degree of mortality caused by the recapture in trap-nets and the handling before release, and therefore an explicit survival rate is unknown and could not be used in the assessment.

All undersized pikeperch captured by gillnets were retained in the length distribution data, as discard mortality for this gear type is assumed to be near 100%.

All individuals sampled during fisheries-independent surveys, regardless of size, were included in the length distribution used for analysis.

3. Stock assessment model

3.1 General description

Model selection

The model chosen for stock assessment of pikeperch in Lake Hjälmaren was the Stock Synthesis (SS3) model (Methot and Wetzel 2013). Stock Synthesis is an age- and size– structured stock assessment model that simulates fish growth, maturation, mortality and other biological processes (population sub-model) and estimates values of parameters describing those and other processes (for example fisheries selectivity) from data sources (observation sub-model). The model also estimates goodness of fit to the data in order to get best fitting parameters (statistical sub-model), and potentially projects and quantifies whether management objectives are fulfilled (forecast sub-model).

3.2 Model settings

The assessment model for pikeperch is a single-area, annual, age-based model where the population is comprised of 14+ age-classes (with age 14 representing a plus group) with sexes combined (male and females are modelled together). The model starts in 1966 and the initial population in Stock synthesis models is assumed to be in an unexploited state, for that the initial catch was assumed to be the average catch in 1966-1968. Fishing mortality was modelled using the hybrid F method (Methot and Wetzel 2013). Option 5 was selected for the F report basis; this option represents fishing mortality requested by the ICES framework (i.e. a simple unweighted average F over fully selected age classes (ages 4–6), denoted F_{bar}).

Samples sizes, CVs, data weighting

Data contributions to the model's log-likelihood function were weighted via their associated measurement variance, represented by coefficients of variation (CVs). The CV thus represents how much flexibility the model has to deviate from the data, which can be used for weighting purposes. For the commercial fleet, the CV for catches was set at 0.05, while a higher CV of 0.1 was applied to initial catch years to reflect greater uncertainty. The annual sample size for length distributions and age-length-key (ALK) was the number of fish sampled.

For survey biomass indices (historical and recent), CVs were estimated from the standard error (SE) of each year index and log-transformed as recommended in the Stock Synthesis manual ($CV = \sqrt{\ln{(1 + (SE)^2})}$; Methot and Wetzel 2013).

Spawning stock biomass and recruitment

Spawning stock biomass (SSB) was estimated at the beginning of each year. Recruitment was modelled as a single pulse event occurring at the start of each year following a Beverton–Holt (BH) stock–recruitment relationship (SRR). Variation in recruitment was estimated as deviations from the SRR, using option 2 as advised by SS best practice guides. Recruitment deviates were estimated for 2008 to 2022 (15 annual deviations), since there was a gap in the ALK from 1972 to 2007. Recruitment deviates were assumed to have a standard deviation (σR) of 0.5 and steepness (h) of 0.93 (estimated from life history parameters from Fishlife database; Thorson 2020). Recruitment bias adjustment was estimated in the preliminary model runs (Methot and Taylor, 2011).

Growth, weight and maturity

Key life-history parameters were estimated from biological sampling conducted from 2008 to 2022, prior to SS3 model fitting (Chapter 2.3.1, Table 2). Although SS3 allows for the re-estimation of growth parameters within the model using the ALK, these were kept fixed. This decision was made because both Linf and k were already incorporated in the estimation of natural mortality (described in the next section), which would otherwise risk model overparameterization.

Table 2. Growth, length-weight and maturity parameters used in the model. Linf is the asymptotic length, k is the growth parameter, Lmin is length at minimum age (0.5 years), alpha and beta are coefficients for the weight-length relationship, L50 is the length (cm) where 50% of the populations is assumed to be mature and slope is the estimate for the maturity relationship.

Parameter	Value
Linf (cm)	69.9
k	0.177
Lmin (cm)	13.5
alpha (length-weight)	2.254 x10^-6
beta (length-weight)	3.34
L50 (cm, maturity)	43.22
slope (maturity)	0.173

Natural mortality

The model incorporates age-specific natural mortality (M), which is assumed to remain constant over time (Figure 7, Table 3). M was estimated based on the Chen and Watanabe method (Chen and Watanabe, 1989), using the website "barefootecologist.com.au/shiny_m.html". This method estimates M as a declining function of age, based on parameters from the von Bertalanffy growth model. To limit model complexity and reduce the number of estimated parameters, M was specified at four age breakpoints: 0.5, 1.5, 5, and 15 years. For all other ages, M was linearly interpolated between these breakpoints. Additional estimates of M were evaluated as alternative model formulations (described in Chapter 3.3).

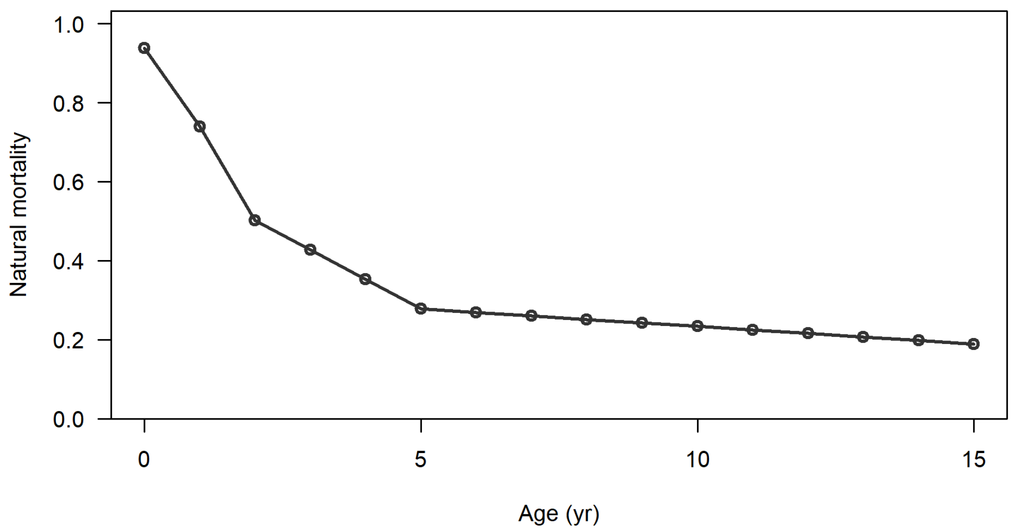

Figure 7. Age-specific natural mortality used in the stock assessment model.

Selectivity

Selectivity for both the licensed fisheries and the monitoring surveys was modelled as a function of fish length. The change in minimum landing size regulation for pikeperch (from 40 cm to 45 cm in 2001) was incorporated into the model through two temporal blocks: 1966–2000 and 2001–2022. For the trap net fleet, the peak of the selectivity curve was fixed at 40 cm for the first time block and at 45 cm for the second, in accordance with the regulatory shift. Other parameters of the trap net selectivity curve were estimated across the entire model period (1966–2022), since we only have length distribution data from trap net fisheries starting from 2003. In contrast, gillnet length distribution data were available both before and after the regulatory change, allowing all selectivity parameters for the gillnet fleet to be estimated independently for each time block. The full selectivity configuration and corresponding parameter estimates are presented in Table 3.

Additional model assumptions include the relationship between spawning biomass and reproductive output. Specifically, fecundity was assumed to be proportional to spawning biomass, which is considered reasonable in this case, as there is no well-established relationship between relative fecundity (i.e., egg production per gram of female) and body length in pikeperch (Lappalainen et al., 2003).

Table of parameters and starting values

Table 3. Settings of the pikeperch model. The table includes the number of estimated parameters, the initial values (from which the numerical optimization is started), the intervals allowed for the parameters, the value estimated by the model. Parameters in bold are fixed and not estimated by the model.

Parameter	Initial value	Bounds (low, high)	Estimated value
Natural mortality (ages: 0.5, 1.5, 5, 15)	**0.938, 0.54, 0.278, 0.189**		
Recruitment			
$Ln(R_0)$	9	(1, 30)	8.12
Steepness (h)	**0.93**		
Recruitment variability (σ_R)	**0.5**		
Ln (Recruitment deviation): 2008 - 2022			
Recruitment autocorrelation	**0**		
Initial catches	Mean of catches in 1966-1968		
Initial F trap net fleet	0.009	(0.001, 1)	0.09
Initial F gillnet fleet	0.009	(0.001, 1)	0.092
	Length selectivity		
Trap net fleet			
peak (1966-2000)	**40**		
peak (2001-2022)	**45**		
top_logit	-15	(-15, 50)	-5.52
ascend_se	-4.36	(-20, 50)	-4.43
descend_se	20	(-10, 20)	16.2
Gillnet fleet 1966-2000			
Peak	56.02	(4, 74.5)	42.97
top_logit	-15	(-15, 50)	17.5
ascend_se	4.297	(-20, 15)	2.14

Parameter	Initial value	Bounds (low, high)	Estimated value
descend_se	25	(-10, 25)	7.5
Gillnet fleet 2001-2022			
Peak	56.02	(4, 74.5)	51.7
top_logit	-15	(-15, 50)	17.5
ascend_se	4.297	(-20, 15)	3.35
descend_se	25	(-10, 25)	7.51
Historical gillnet survey 1966-1978			
peak	15.86	(4, 60)	39.28
top_logit	-15	(-15, 50)	-9.61
ascend_se	3.85	(-15, 8)	4.5
descend_se	20	(-15, 20)	6.44
Recent gillnet survey 2008-2022			
peak	15.86	(4, 60)	19.34
top_logit	-15	(-15, 50)	17.5
ascend_se	3.85	(-15, 8)	4.56
descend_se	20	(-15, 20)	2.5
		Catchability	
Extra variability added to input standard deviation	**0.1**		

3.3 Model diagnostics

The estimated selectivity curves appear biologically reasonable. Both the historical and recent survey gears were estimated to be more effective at sampling smaller individuals than the commercial fishing gears (Figure 8, Figure 9). The model also suggests that the recent survey gear captures small pikeperch more effectively than the historical gear, which is expected given the mesh-sizes in the different gears.

The trap net fleet is assumed to exercise active selectivity, whereby fishers can release individuals below the legal minimum landing size. A threshold-type selectivity for the trap net fleet was therefore implemented in the model, without incorporating discard mortality. It is therefore assumed that all individuals below the minimum landing size are not caught, which mimics a discard survival rate of 100%. This is likely an overestimate compared to observed recapture rates (38.6-74.4%; Nyberg et al., 1996), but provides a practical solution as survival-length relationship is lacking.

In contrast, the gillnet fleet lacks the ability to actively release undersized fish and is therefore more constrained by passive gear selectivity. The gillnet selectivity curve is estimated to be right-shifted, targeting larger fish relative to the surveys. The similarity in the shape of the selectivity curves between gillnets and survey gears (illustrated by parallel blue and red lines in Figure 8) supports this interpretation, while still reflecting the legal and operational differences between the fleets.

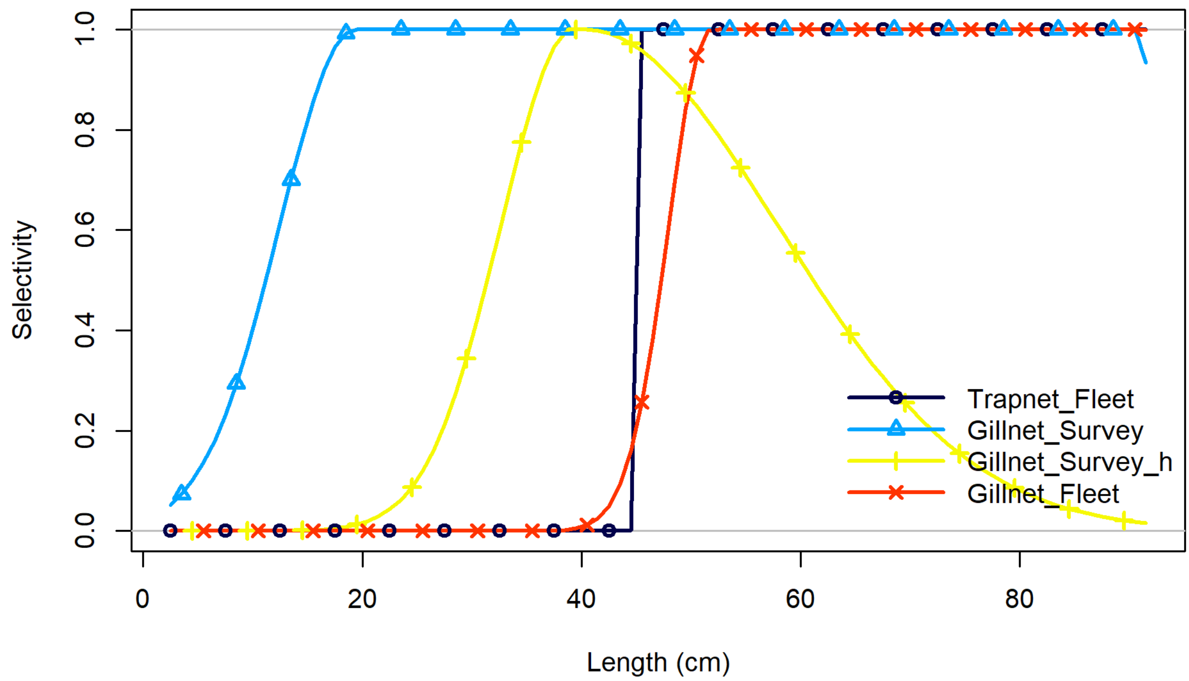

Figure 8. Length-based selectivity in 2022 of commercial fishery using trap nets and gillnets, historical survey (Gillnet_survey_h) and recent survey (Gillnet survey).

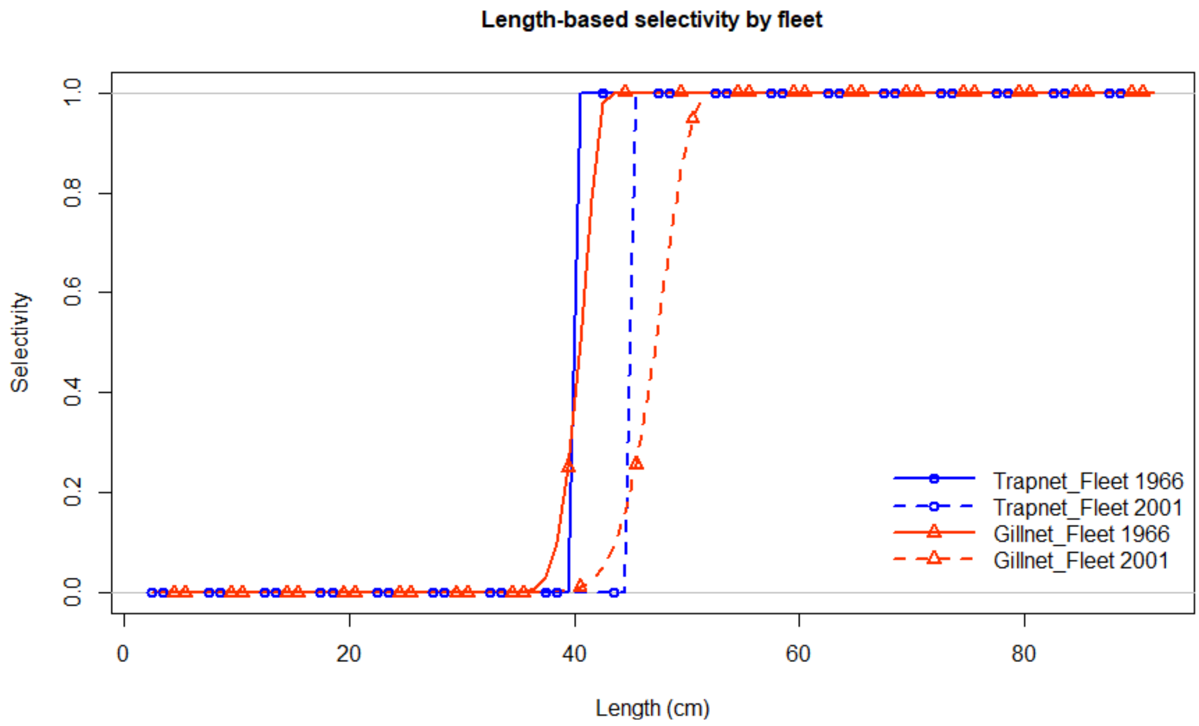

Figure 9. Length-based selectivity of commercial fishery using trap nets and gillnets in 1966-2000 (1966) and 2001-2022 (2001).

Model fit

The modelled length distributions showed good fit to the observed data for the trap net fleet, gillnet fleet, and the historical gillnet survey (Figure 10). In contrast, the fit to the recent gillnet survey was comparatively poorer. This discrepancy likely arises from the complex nature of multi-mesh gillnets, which tend to produce multi-modal length distributions due to varying mesh sizes. Especially the 6.25- and 8-mm mesh panels, which capture varyingly sized young-of-the-year fish depending on that year's growth. Since selectivity was not estimated at the level of individual mesh sizes in this model, capturing the full complexity of the recent survey gear's selectivity presents a challenge. This limitation is expected and highlights a common issue in fitting selectivity functions to data from composite gear types.

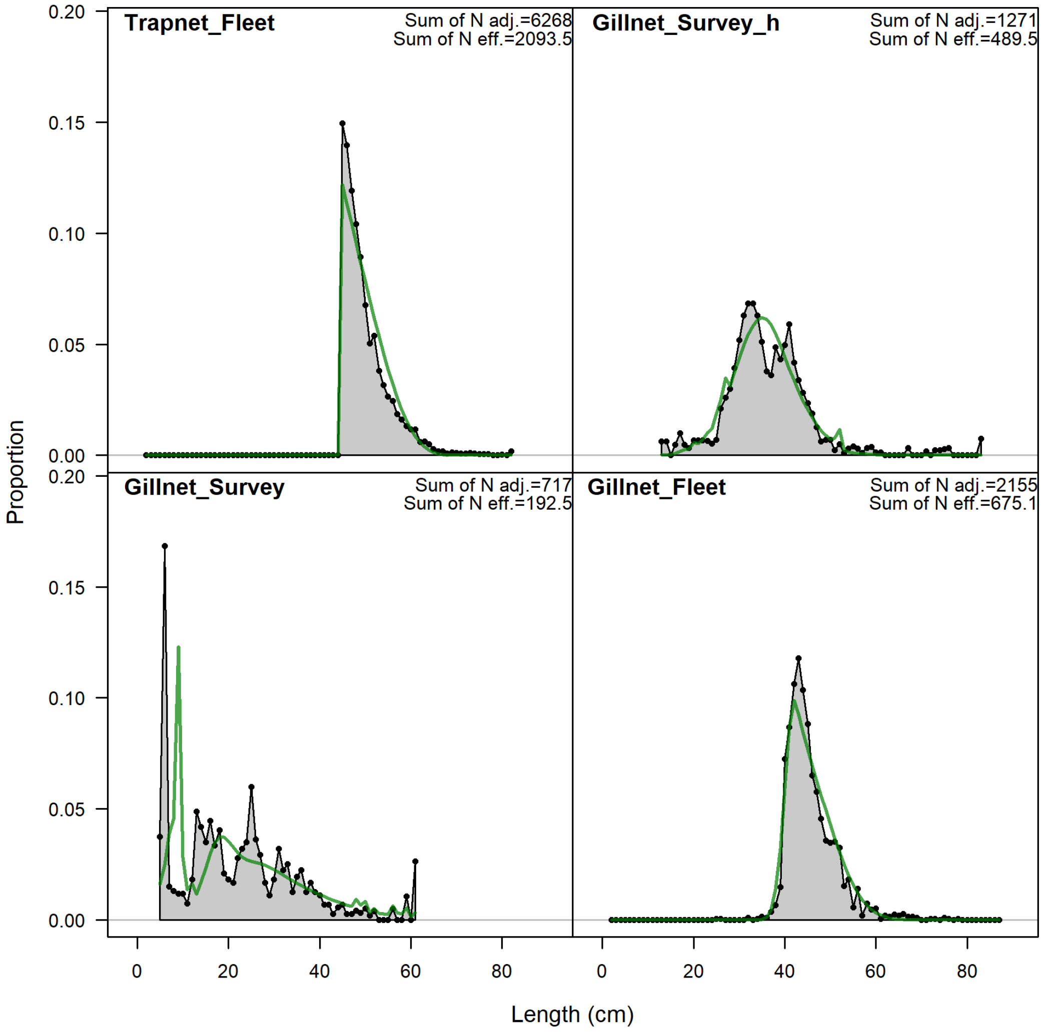

Figure 10. Model fit (line) to observed length distribution data (grey polygon), pooled across years.

Even though the model estimates for the recent survey indices are within the estimated confidence intervals, the model does not follow the trend well (Figure 12). For historical survey and fisheries-dependent indices the model picks the trend, but with a time-lag (Figures 11, 13, 14)., The model achieved a better fit for the trap net fleet compared to the gillnet fleet (Figure 13, Figure 14). This may reflect differences in gear-specific selectivity, reporting consistency, or sampling precision across fleets.

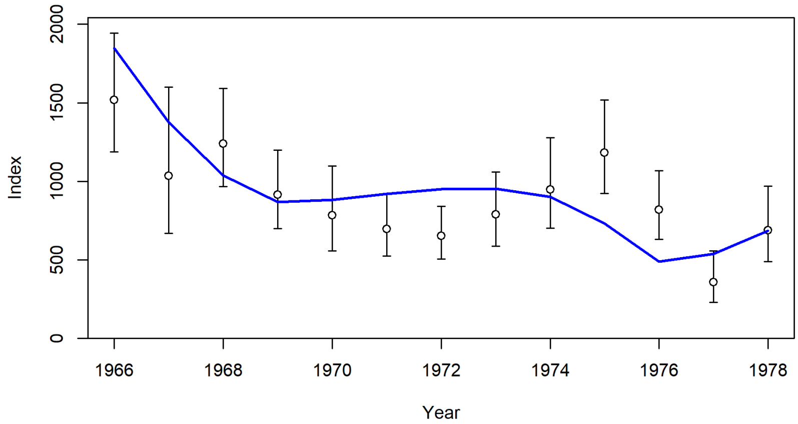

Figure 11. Model fit (line) to biomass index (g fish caught per net/night/year) from historical gillnet survey (whiskers indicate CV).

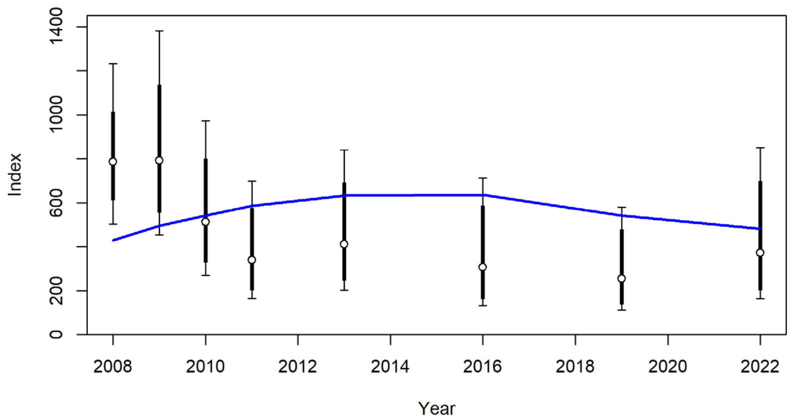

Figure 12. Model fit (line) to biomass index (g fish caught per net/night/year) from recent gillnet survey (whiskers indicate CV).

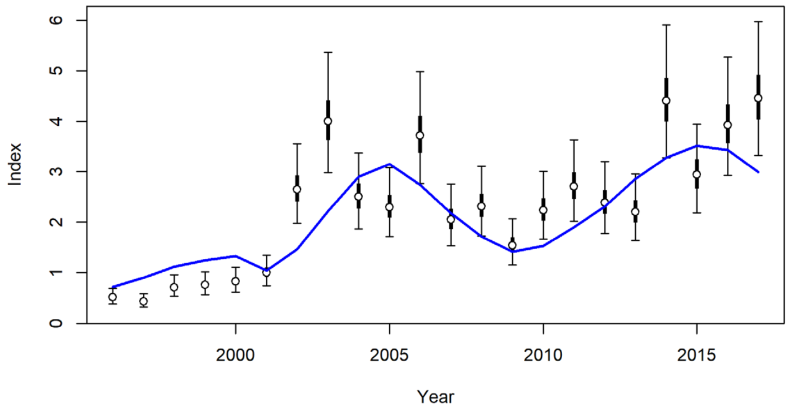

Figure 13. Model fit (line) to CPUE index (kg fish caught per net/night/year) from the trap net fleet (whiskers indicate CV).

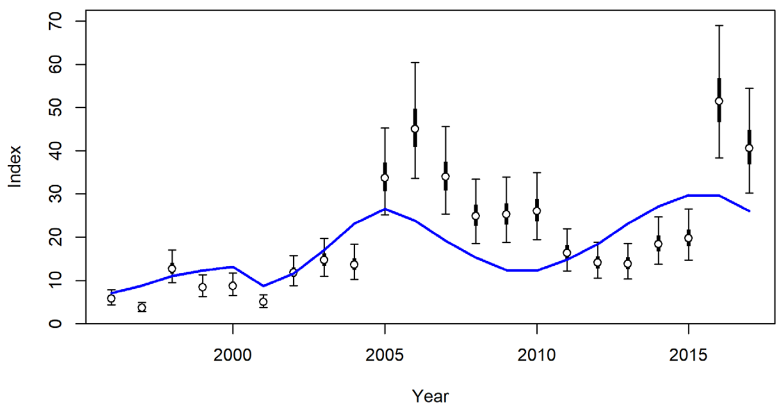

Figure 14. Model fit (line) to CPUE index (kg fish caught per 1000 m of net/night/year) from the gillnet fleet (whiskers indicate CV).

Retrospective analysis

Retrospective analysis is a diagnostic tool used to evaluate the stability and reliability of parameter estimates and biological reference points, and to detect potential systematic bias in model outputs (Hurtado-Ferro et al. 2015; Szuwalski et al. 2018). It involves fitting the stock assessment model to the complete dataset, followed by sequential refits in which data from the most recent year are removed one year at a time.

For this assessment, the retrospective analysis was conducted by sequentially omitting the final five years of data. The results indicated no evidence of model instability (Figure 15).

Furthermore, Mohn's rho, a commonly applied metric for quantifying retrospective patterns (Hurtado-Ferro et al. 2015), was calculated for spawning stock biomass (SSB), fishing mortality (F) and recruitment (R). As a rule of thumb, long-lived species have been proposed to have rho-values within a range of -0.15 - 0.20, while the range for short-lived species can be wider -0.22 - 0.30 (Hurtado-Ferro et al. 2015). Values outside those ranges may indicate concerns about the retrospective patterns. The estimated Mohn's rho for SSB was 0.21 and for F and recruitment was -0.18, all of which are slightly outside the limits for long-lived species, indicating some retrospective patterns with SSB, recruitment and F estimates. Specifically they illustrate that the model has a tendency to overestimate SSB and underestimate F and recruitment, both of which should be considered when interpreting the assessment results.

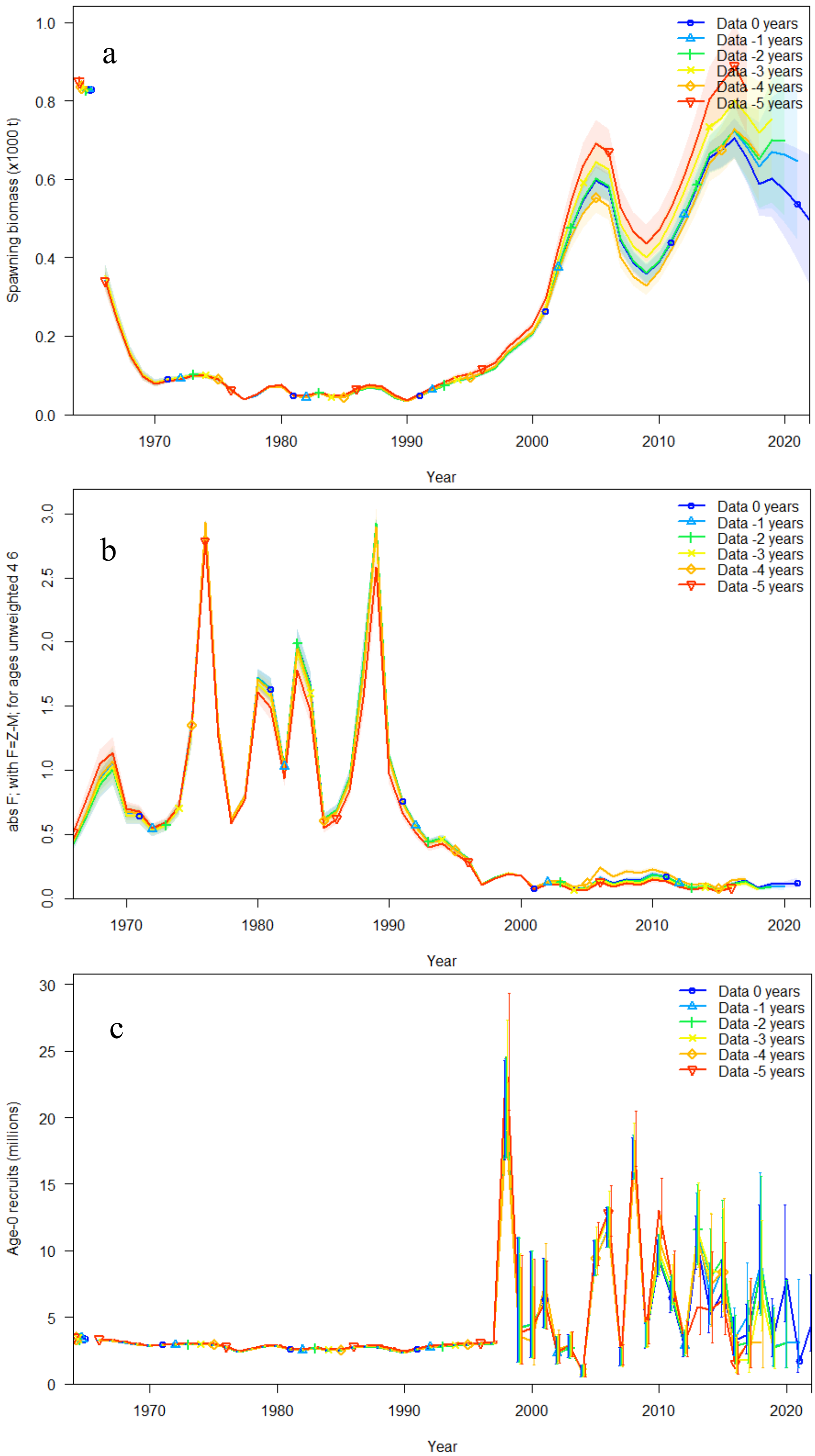

Figure 15 Retrospective analysis for spawning stock biomass (panel a), fisheries mortality (F4-6, panel b), and recruitment (panel c).

Impact of datasets on model

We investigated effects of individual datasets on the fit of other components, by removing one dataset at a time (Figures 16, 17, 18, 19, 20). The results indicate that the greatest impact on model fit (Figure 16, Figure 17) and stock estimates (Figures 18, 19, 20) was caused by the exclusion of length distribution data from the fisheries, particularly from the trap net and gillnet fleets (denoted as *-ldist_trapnet* and *-ldist_gillnet* in Figures 16-20). These datasets appear to be particularly informative for estimating selectivity and the size structure in the population.

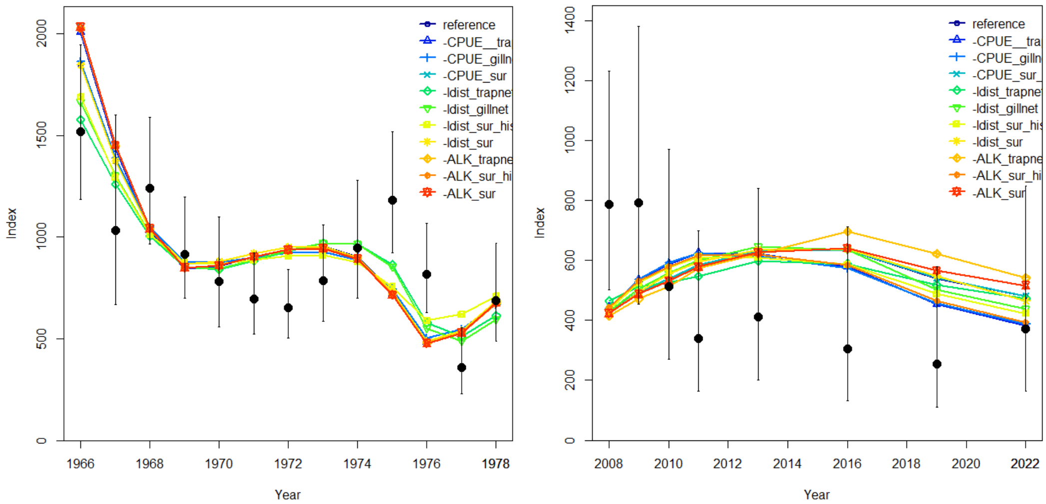

Figure 16. Effect of individual datasets on the fit of indices from historical (left) and recent (right) gillnet survey.

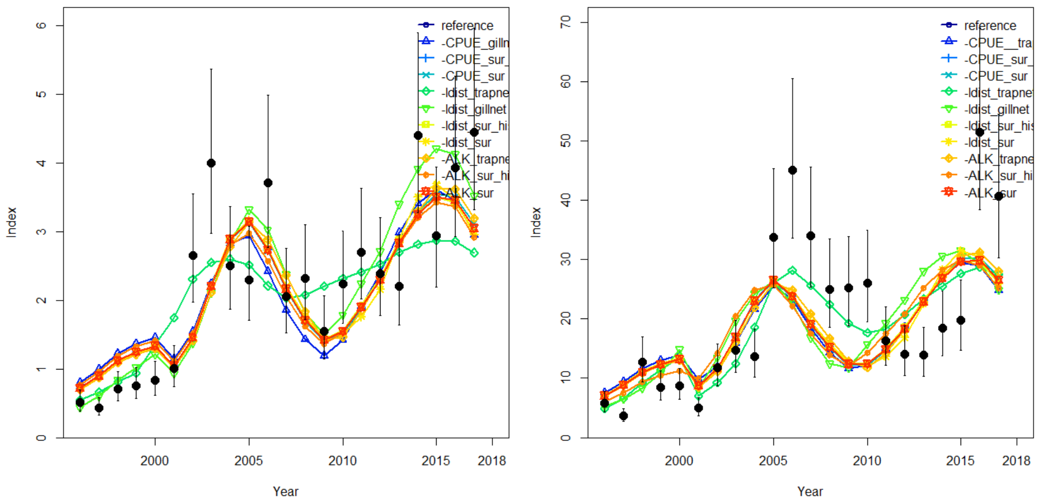

Figure 17. Effect of individual datasets on the estimation of spawning stock biomass.

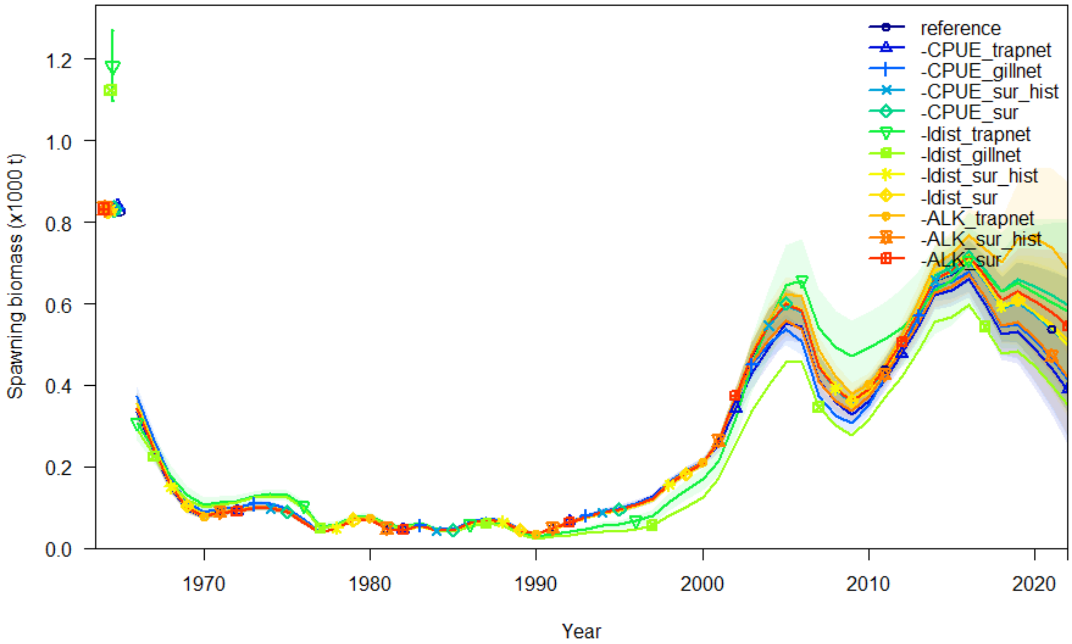

Figure 18. Effect of individual datasets on the estimation of spawning stock biomass.

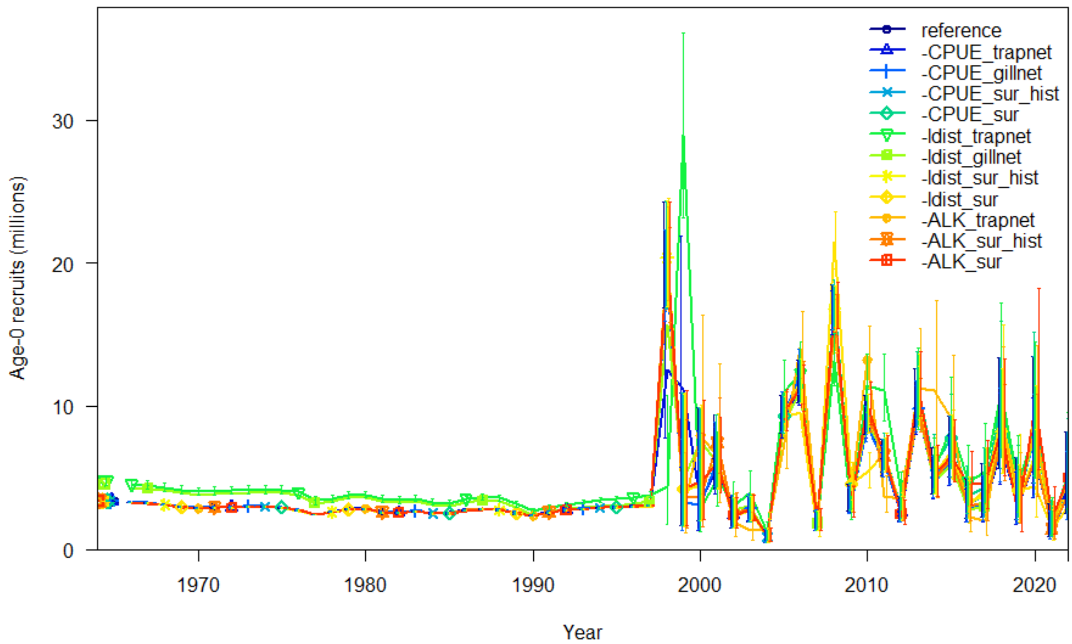

Figure 19. Effect of individual datasets on the estimation of recruitment.

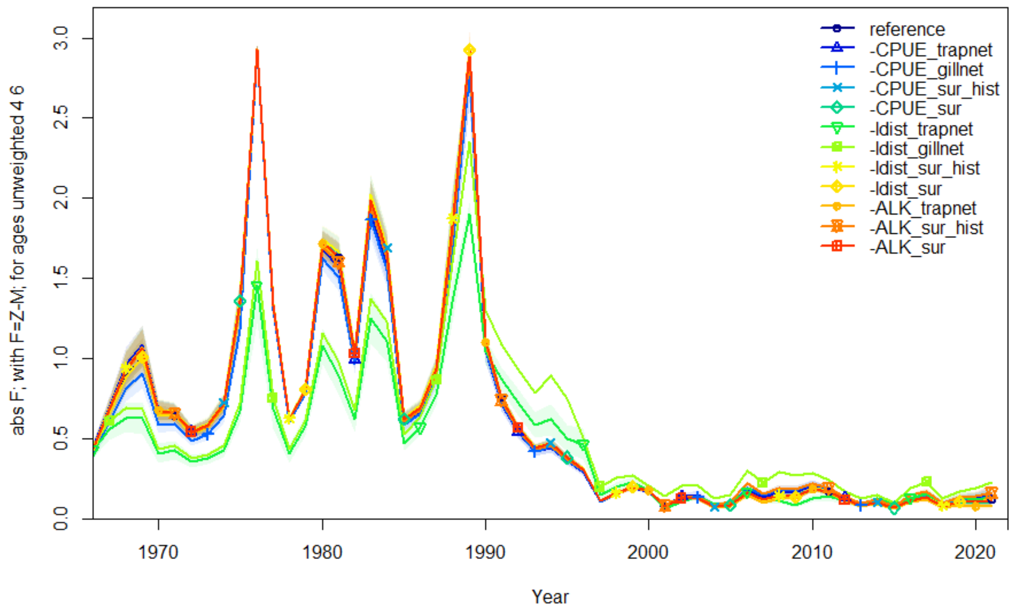

Figure 20. Effect of individual datasets on the estimation of fishing mortality at ages 4-6 (Fbar).

Assessment of alternative models

As alternative model formulations we tested assumptions on natural mortality and inclusion of recreational catches.

Assumptions on natural mortality are one of central questions in the stock assessment models. Methods of estimating it vary from assuming a fixed age invariant rate of 0.2 (i.e. the classic M assumption), to estimating it with different methods from life-history parameters and mark-recapture experiments.

In the reference model (Chapter 3.2) we used the Chen and Watanabe method (Chen and Watanabe, 1989) to estimate natural mortality from life-history parameters. As alternatives, we also tested estimates from methods of Gislason et al. (2010), Charnov et al (2013), Lorenzen (2022), the average of the four methods, and classic 0.2 (Figures 21, 22, 23, 24).

Mortality estimated using the Chen and Watanabe method was the lowest among the four methods for young pikeperch, but slightly higher for old pikeperch when compared to estimates of the Gislason method (Figure 21). Mortality estimated using the Lorenzen method was lower than both the Charnov and Gislason methods for young fish, but the highest for old fish compared to the other methods.

These differences in estimated natural mortality for young and old fish have impacts on the model. The increase in mortality of old fish using the Lorenzen method produced the highest SSB (Figure 22, Figure 23). This is because the catches remain the same, and thus in a scenario of higher M, there must have been more fish present in the population to provide the same level of catch. In comparison, the Charnov and Gislason methods, which estimate a higher mortality for young fish, yield model outputs with higher recruitment (Figure 24). Assuming a natural mortality of 0.2, yielded lower SSB compared to the reference model.

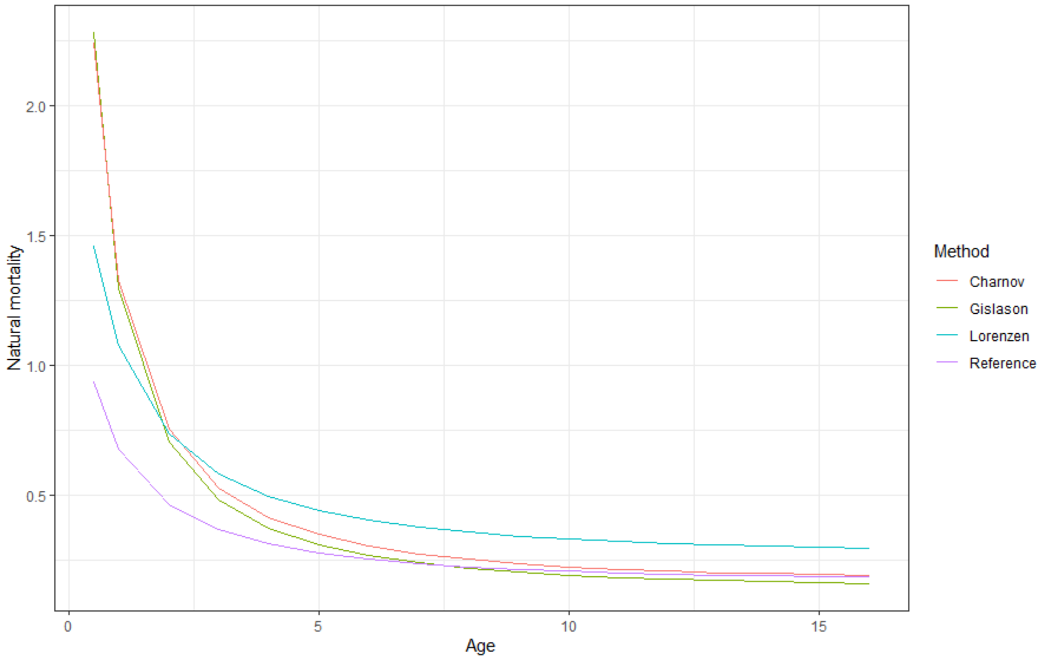

Figure 21. Pikeperch natural mortality at age estimated using different methods. Reference refers to the Chen and Watanabe method.

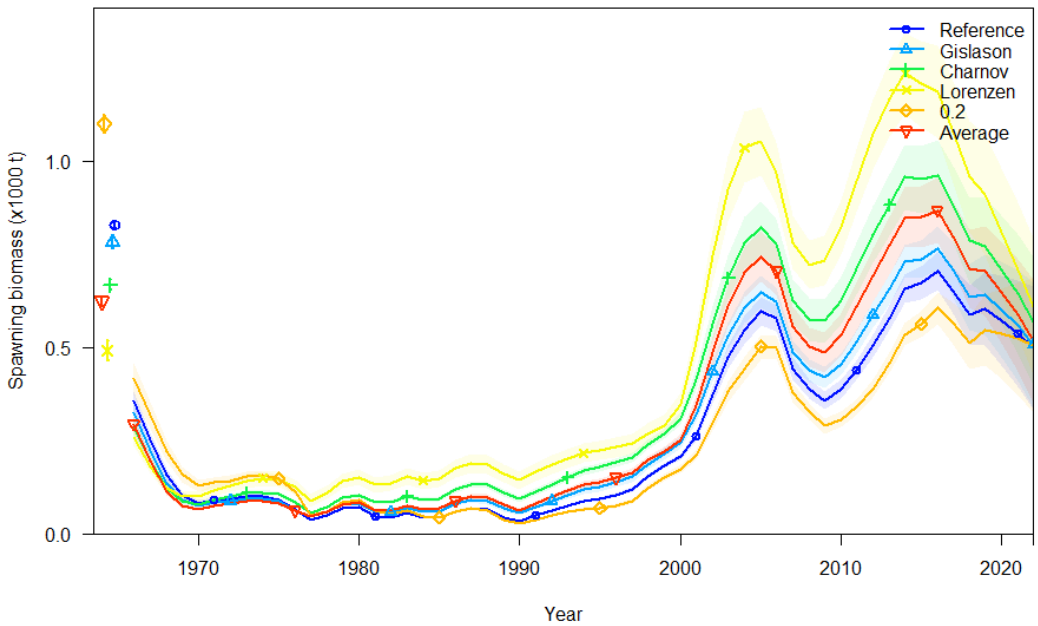

Figure 22. Effect of different methods for natural mortality on the estimation of spawning stock biomass.

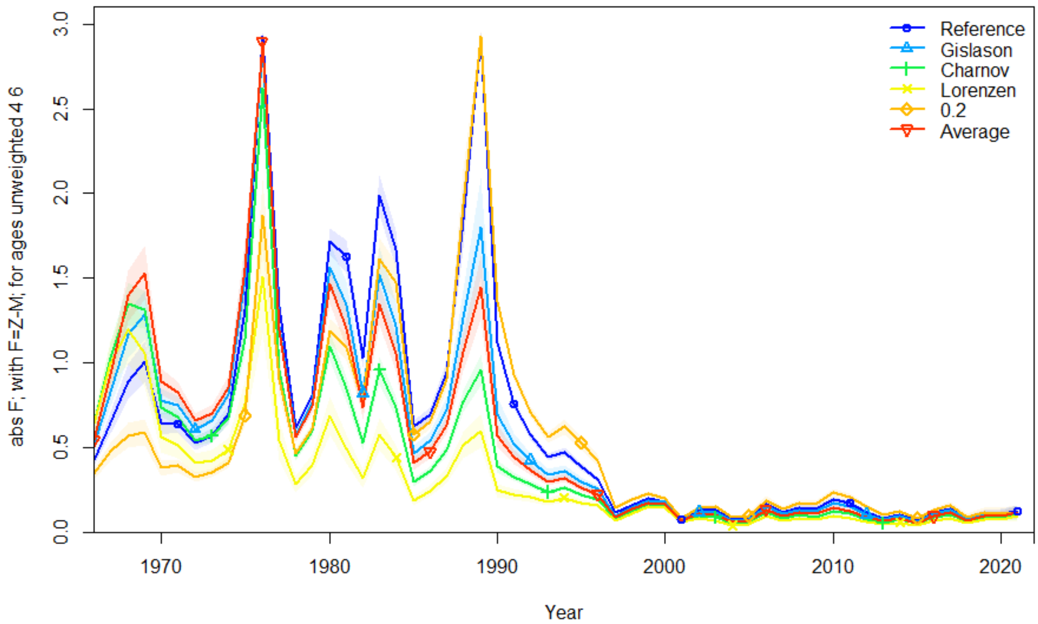

Figure 23. Effect of different methods for natural mortality on the estimation of fishing mortality

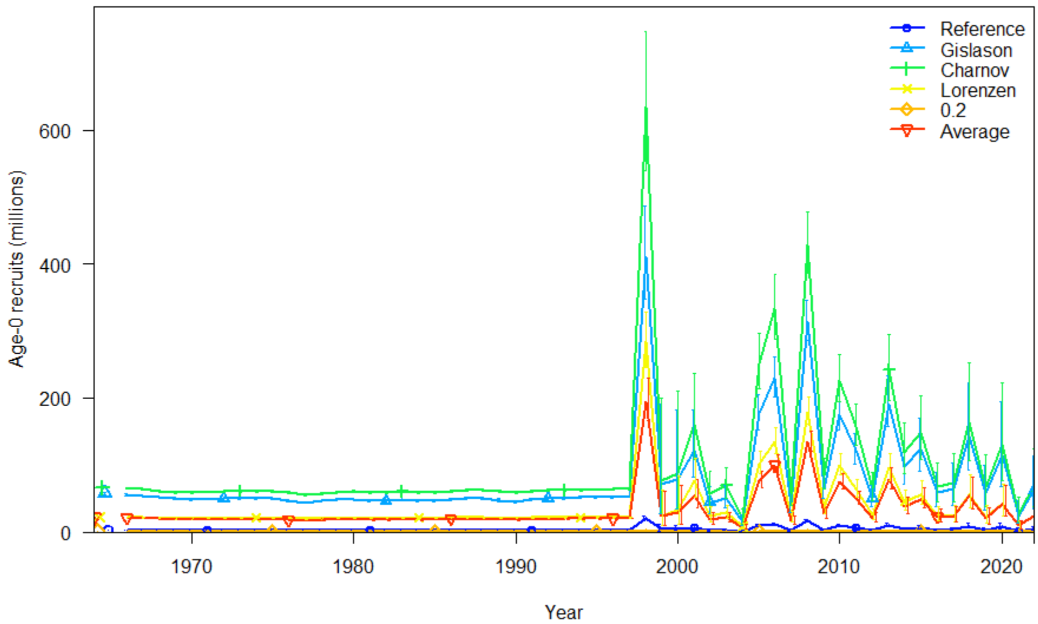

Figure 24. Effect of different methods for natural mortality on the estimation of recruitment

Recreational fisheries are potentially important for the pikeperch stock development, although catches are highly uncertain. To investigate the potential impact of the recreational fisheries, we tested three levels of catches based on average annual catches in 2018-2022, which are approximately similar to total commercial landings but with large uncertainty (Chapter 1.2, Sundblad et al., 2025).

The levels were intermediate (120 tons), low (50% of intermediate) and high (150% of intermediate). We assumed that the recreational fisheries started in 2001, after the change of minimum landing size. Starting catch was 0 ton with a linear increase until 2005, when landings from commercial fisheries also increased. Recreational fisheries were divided in two types of gears: handheld and gillnets. We assumed that selectivity mirrors selectivity of trapnet and gillnet fisheries, respectively. The ratio between the handheld gear and gillnet catches were based on how pikeperch catches are divided by gear type in all the large lakes combined; 40% handheld, 36% from nets and 24% released. We disregard post-release mortality and focus on kept catch, yielding a handheld to net ratio of 53:47 (40/(40+36)) (Sundblad et al., 2025). Although this may be slightly biased towards nets, as pikeperch release rates (assumed from primarily handheld gears) are lower in Hjälmaren (0.16 CI 0.06-0.26) compared to the other Swedish large lakes; Mälaren 0.60 (0.48-0.71); Vänern 0.42 (0.17-0.67); Vättern 0.64 (0.59-0.70).

The inclusion of recreational fisheries in the model increased the estimated SSB by almost a factor 2 in the peak years of 2005 and 2016 compared to the reference model (Figure 25). Recruitment differences were largest in 1997-2015 (Figure 27), while fishing mortality differed at the end of the time series (2019-2022, Figure 26).

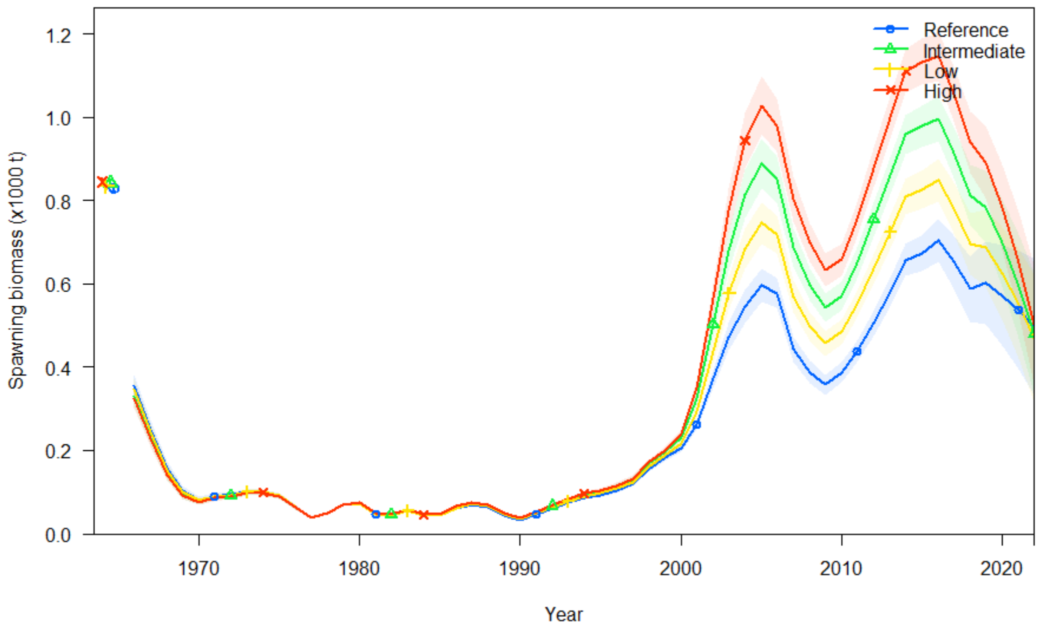

Figure 25. Effect of recreational fisheries on the estimation of spawning stock biomass.

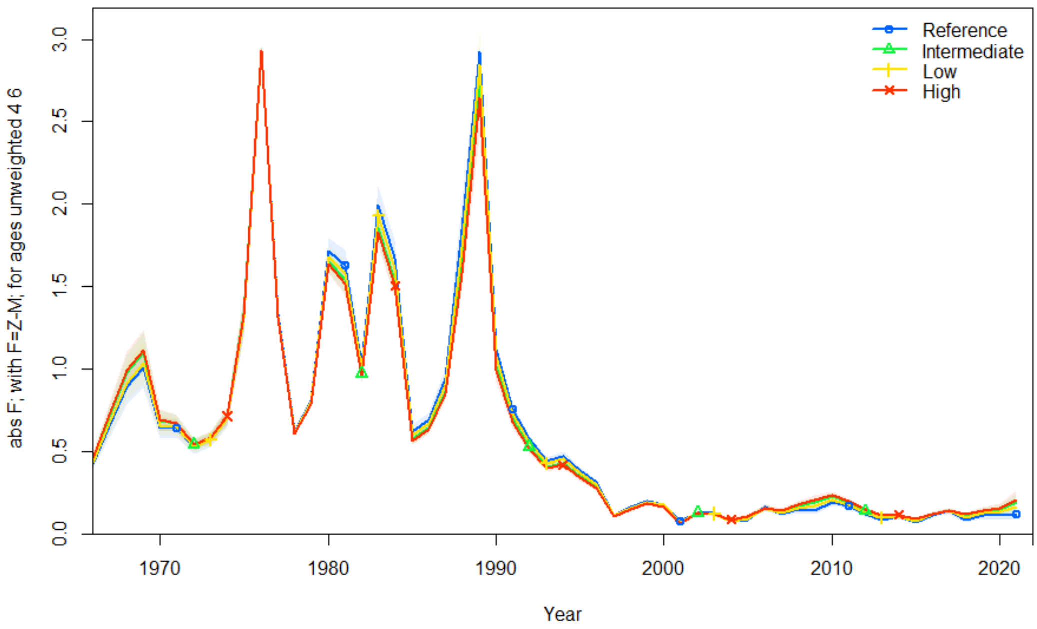

Figure 26. Effect of recreational fisheries on the estimation of fishing mortality.

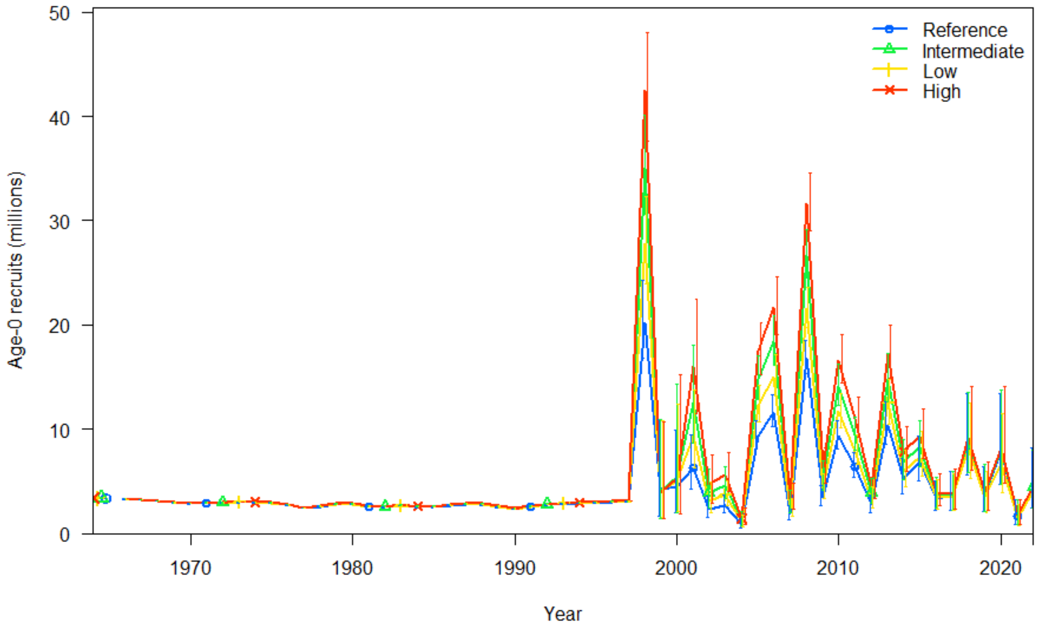

Figure 27. Effect of recreational fisheries on the estimation of recruitment.

3.4 Model results

Development of key parameters (SSB, F, R)

The estimated spawning stock biomass (SSB; Figure 28, Table 4) declined in the beginning of the time series, followed by a period of recovery during the 1990s reaching two peaks in 2005 and 2016, followed by declines in 2006-2009 and 2017-2022, indicating a recent deterioration in stock status.

Fishing mortality (F) had an increasing trend with some fluctuations in the beginning of the time series, but the most pronounced decrease occurred in 1990-2000 and remained low after Recruitment (R) remained relatively stable prior to 1998, a period during which limited age composition data constrained the estimation of annual variability. After 1998, recruitment estimates displayed greater fluctuation, including the appearance of notably strong year classes in 1998 and 2008 (Figure 28). Since 2008, recruitment has a negative trend with several slightly stronger years.

The decline in SSB is about 30% from 2016 to 2022 and continues to decline in forecast. Such a decline in a relatively short period is difficult to understand. It does not appear to be related to an increase in reported landings, nor an increase in F (Figure 28). Further, input data in the form of recent gillnet surveys does not indicate a decline either (Figure 12). One possible explanation is the lower than usual recruitment which is estimated to have occurred during the same time-interval (Figure 28). However, an index of recruitment based on the catch of two-year-old fish in commercial trap nets suggest that recruitment has been weak since 2014, and strong again 2018-2020 (Fiskbarometern, 2024). Taking also discussions with fishers into consideration seems to corroborate the SSB decline suggested by the model. Nevertheless, SSB in 2022 is still higher than in 2007-2012, after which stock was able to reach one of the two recent SSB peaks.

Table 4. Summary of the stock assessment. Total stock biomass (TSB) and spawning stock biomass (SSB) are in tonnes, Fbar (4-6) is average fishing mortality for ages 4-6, recruitment is in thousands of individuals, and landings are in tonnes.

Year	TSB	SSB	F_{bar} (4-6)	Recruitment	Landings
1966	672	355	0.42	3283	237
1967	534	251	0.65	3225	239
1968	410	157	0.89	3115	202
1969	334	103	1.01	2970	160
1970	299	81	0.64	2870	98
1971	311	90	0.64	2914	108
1972	313	94	0.53	2933	96
1973	321	101	0.57	2963	108
1974	318	101	0.70	2962	124
1975	303	91	1.31	2922	172

Year	TSB	SSB	F_{bar} (4-6)	Recruitment	Landings
1976	256	64	2.93	2746	198
1977	204	38	1.33	2419	84
1978	226	49	0.62	2594	61
1979	259	69	0.81	2787	103
1980	258	72	1.72	2811	168
1981	212	47	1.63	2567	112
1982	210	44	1.03	2522	79
1983	230	54	1.99	2653	148
1984	203	43	1.67	2511	110
1985	202	43	0.62	2508	53
1986	237	60	0.69	2716	80
1987	251	69	0.94	2790	112
1988	242	64	1.85	2749	155
1989	208	44	2.93	2519	147
1990	187	34	1.12	2337	66
1991	217	48	0.76	2580	70
1992	239	62	0.57	2735	72
1993	259	75	0.44	2830	68
1994	281	87	0.47	2900	80
1995	295	93	0.38	2929	71
1996	318	104	0.31	2974	66
1997	345	119	0.11	3026	30
1998	403	155	0.16	20221	53
1999	546	182	0.20	4216	73
2000	716	206	0.18	4431	71
2001	879	263	0.07	6332	48
2002	1030	372	0.13	2315	110
2003	1097	473	0.13	2670	165
2004	1092	546	0.07	838	127
2005	1050	596	0.08	9340	165
2006	982	578	0.16	11651	289
2007	871	443	0.12	1901	166
2008	883	388	0.15	16863	162
2009	961	358	0.14	3492	131
2010	1076	388	0.19	9332	181
2011	1174	439	0.17	6447	194
2012	1243	505	0.12	2758	166
2013	1284	579	0.08	10398	148
2014	1339	656	0.10	5239	201
2015	1327	673	0.07	6852	144

Year	TSB	SSB	F_{bar} (4-6)	Recruitment	Landings
2016	1358	705	0.11	3310	251
2017	1267	652	0.14	3679	261
2018	1143	587	0.08	8702	144
2019	1142	603	0.11	3631	196
2020	1093	572	0.11	7921	191
2021	1063	537	0.12	1674	191
2022	1010	498	0.11	4512	158
2023	977	488	0.18	3321	0
2024	914	476	0.18	3319	0
2025	840	451	0.18	3313	0

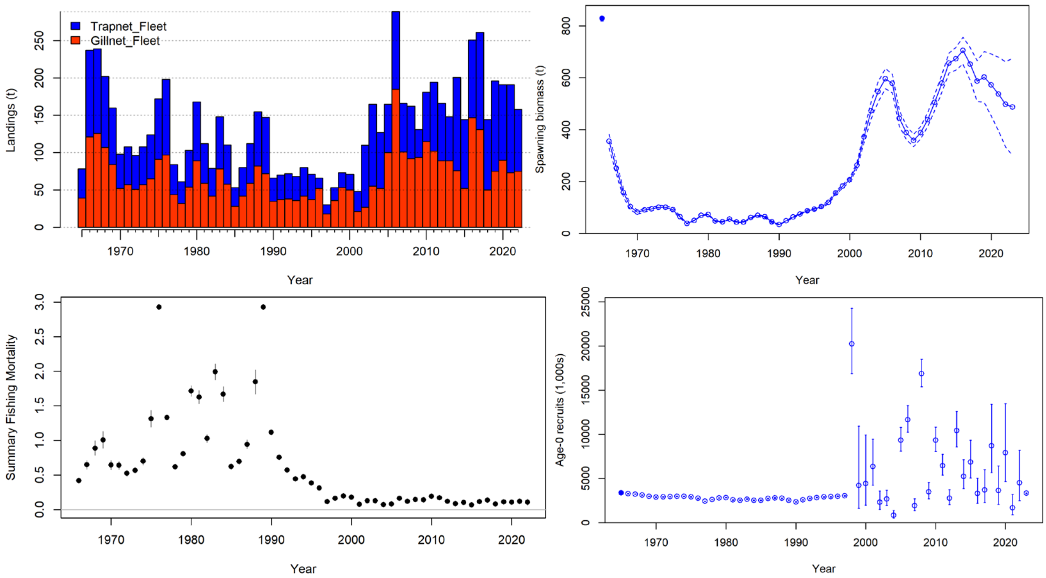

Figure 28. Summary of reported landings (top left) and the output of the assessment model. Spawning stock biomass (SSB, top right), fisheries mortality for ages 4-6 (bottom left) and recruitment (bottom right) are shown with 95% confidence intervals. Landings by fleet and SSB are in tonnes, recruitment is in thousands of individuals.

Age-based indicators

Model estimates (SSB, recruitment, age structure, and F) were used to calculate age-based indicators (ABIs), following the method developed by Griffiths et al. (2024). The ABI framework expands the commonly used reference points of SSB and F that would yield long-term maximum sustainable yield (SSB_{MSY}, F_{MSY}), by considering also the age- and size structure of the stock. The age-based indicator (ABI_{MSY}) yields information on the age structure relative to the equilibrium age structure at F_{MSY}, as well as an age structure indicator (ABI_0) relative to age structure under a scenario of no fishing (F_0). Here, however we used $F_{40\%}$ as that was used as a target reference value in our model (see Chapter 4 for more details) and therefore calculate ABI_{40}.

In 2022 (Figure 29) the modelled age structure was characterized by a lower abundance of older individuals compared to the equilibrium under no fishing, but higher than under F_{40}. The number of individuals at age 2 and 4 in 2022 were higher than expected at equilibrium and are indicative of strong year-classes, i.e., individuals born in 2018 and 2020. ABI_{40} showed a decline during the early 1970s, followed by a steady recovery from 1996 onwards, with values exceeding 1 ($\log(ABI_{40}) > 0$) in most years thereafter (Figure 30). This suggests that the proportion of older fish in the stock has generally been higher than expected under F_{40}, indicating that fisheries mortality has been lower than if fishing at the level to reach 40% of virgin biomass (B_{40}), thus providing a potential buffer against stock depletion. ABI_0, which compares the current age structure to that expected under no fishing, shows a similar trend, but remains low. This reflects the persistent truncation of the age structure due to historical fishing pressure. Overall, the presence of older age classes and recruitment pulses suggest that the stock has retained some resilience and recovery potential. However, the model consistently estimates a recent decline in SSB, coupled with a relative scarcity of older individuals in the most recent years (Figure 29), as discussed above.

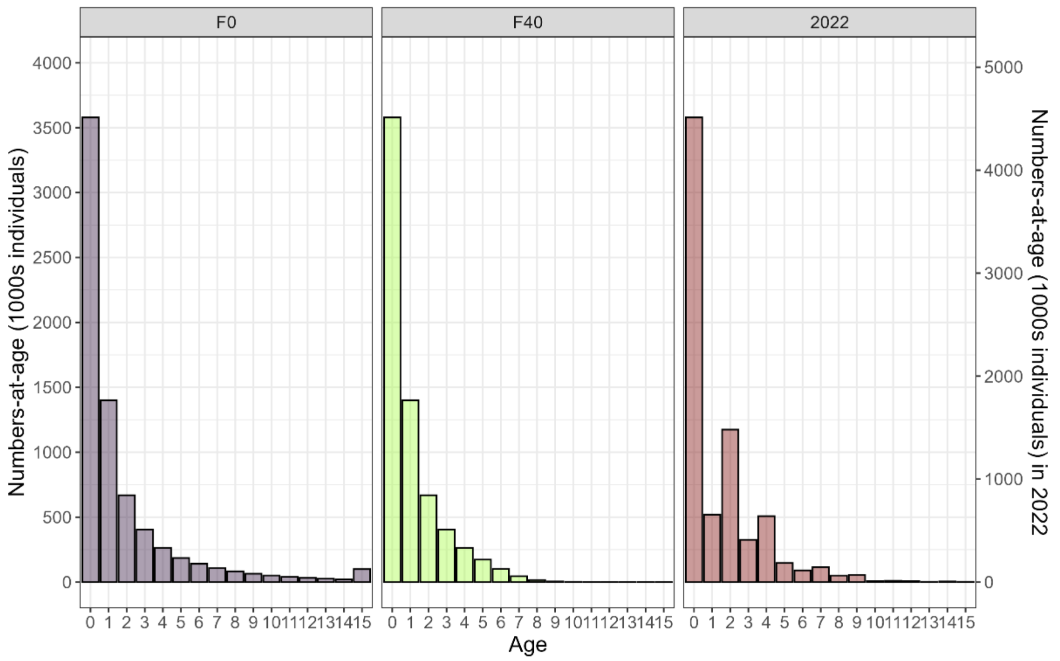

Figure 29. Comparison of population age structure at no fishing (F0), equilibrium under F40, and the estimated age structure in 2022 (note different scale for 2022 for easier comparison of age structures).

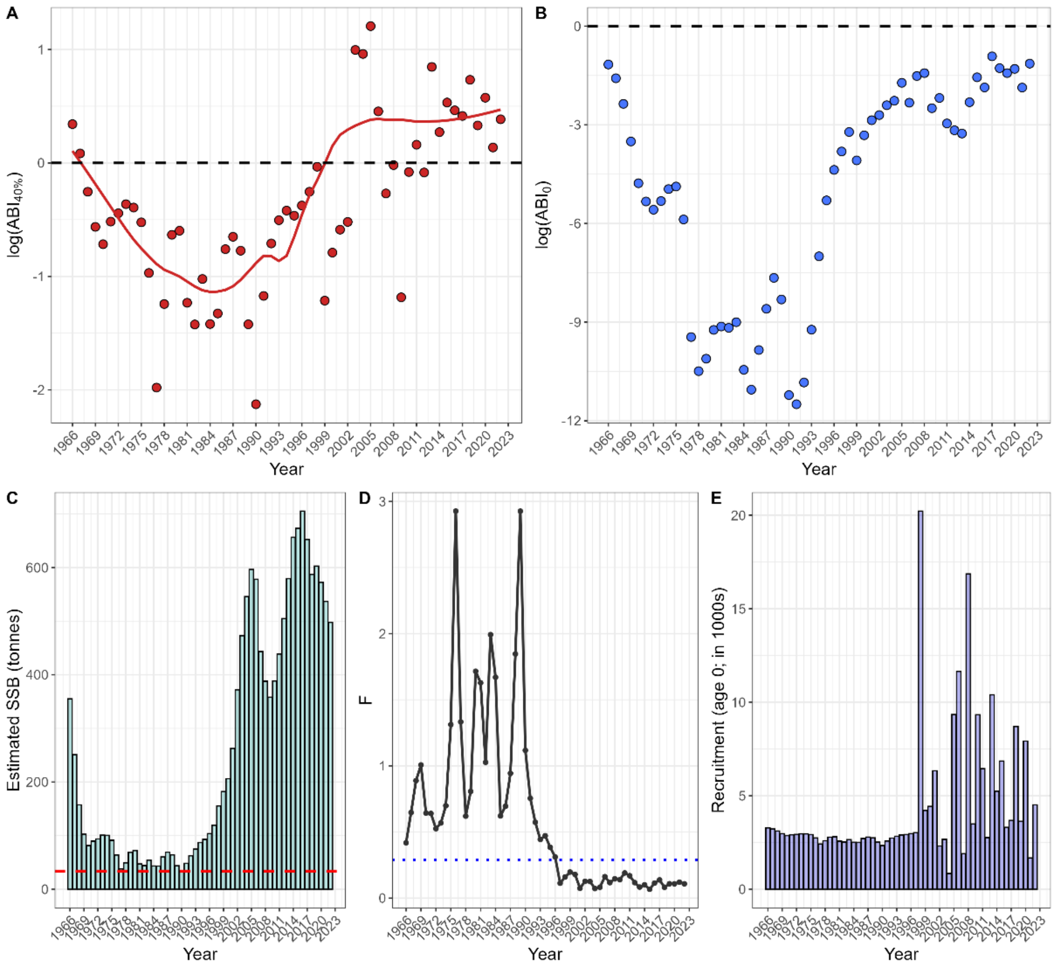

Figure 30. Summary of age structure and stock status for pikeperch in Hjälmaren. Time series of age structure relative to FMSY and age structure relative to F0 are shown in panels A and B respectively. The estimated SSB, F and recruitment are shown in panels C, D and E respectively. The coloured lines in panels C and D represent the following reference points: Blim (red dashed) and F_{MSY} (blue dotted). Note that there is no management goal for this stock, and that no reference points have been decided (Chapter 4).

3.5 Conclusions and recommendations

Developing an assessment model based on limited and sometimes conflicting data sources is a challenging task. Despite this, the overall model fit to length distributions and biomass indices (Figures 10, 11, 12, 13, 14) is generally within the confidence intervals (except for gillnet fleet indices; Figure 14). A key point of concern is the discrepancy between the recent gillnet survey (Figure 12) and the gillnet fleet index (Figure 14) that overlap in time but indicate different temporal trends, thus creating conflicting developments in the raw data. The stock assessment model can therefore not be expected to fit both indices, and we expect a poorer fit to one of the two indices. Both the reference and alternative model formulations provide a better fit to the fleet index than the survey index, which could be an effect of the fleet containing more years of data and therefore carrying more information (from the model's perspective). Consequently, when interpreting trends and drawing conclusions, greater confidence may be placed on patterns supported by more comprehensive datasets, while acknowledging potential uncertainties introduced by survey-fleet discrepancies.

Although model fit is sub-optimal, particularly for certain indices, it consistently indicates a declining, though not drastically, trend in the SSB in recent years (Figure 28), a pattern that is concerning and somewhat difficult to understand and explain.

Additional actions that can be taken to further improve the model:

- A time-series of recreational catches is highly needed, but difficult to obtain in retrospect. Highly uncertain data indicate that recreational catches of pikeperch can be substantial. The possibility to estimate a ratio of commercial:recreational catches could be an option to explore in the future. We explored the effect that inclusion of recreational data would have on the model and found that effect on estimated SSB was substantial. Thus we hope for recreational catch data to be available in future.
- Revisit historical and recent survey abundance indices and assess various modelling techniques to better describe the stock development, e.g. by including not only site and gear, but also temperature and other environmental factors (since catches in survey can be influenced by e.g. temperature (Naddafi et al. 2022)).

Complementary studies that could also improve our understanding of this stock:

- The current catch index from commercial catches ended in 2017. This is because reliable effort data from commercial fisheries has been missing thereafter. With the newly introduced electronic logbooks, we anticipate a development of a much-needed catch index.

- Acoustic telemetry has recently been carried out on pikeperch in Lake Hjälmaren, which could give insights into migration patterns within the lake as well as a measure of total mortality. This would provide input to model assumptions regarding representativity of different sites and indices.
- Further investigations of selectivity in commercial gears, (e.g. comparing 100 mm, 120 mm, and larger mesh sizes) is necessary to validate model assumptions and better inform selectivity parameters.
- A comparison of old and current monitoring sites by comparing the current gear (Bkust9+2) with the old one (Blänk). This would help to calibrate past and present survey indices to achieve better temporal consistency.

4. Biological reference values

There is currently no formal management plan or agreed-upon biological reference points for pikeperch in Lake Hjälmaren, which limits the ability to perform a fully benchmarked stock assessment. We recommend that fisheries managers initiate a structured discussion around management objectives and target reference levels, not only for pikeperch but also for other nationally managed freshwater stocks (Naddafi et al. 2023).

Management objectives should consider a combination of mortality (F), biomass (SSB), and age-/size-based indicators, in line with modern ecosystem-based fisheries management principles.

In the absence of defined reference points, we made the following tentative assumptions: B_{40} was used as the biomass reference point, defined as 40% of unfished (virgin) biomass, with the estimated corresponding $F_{40} = 0.29$ (the value that is very close to the average F_{40} of 0.31 estimated for 8 species by Shertzer et.al. 2024). After 2000, the stock's SSB hasn't declined below these reference points (Figure 31). We emphasize that these values are provisional and should not be used for management until they are discussed and validated within a formal management framework.

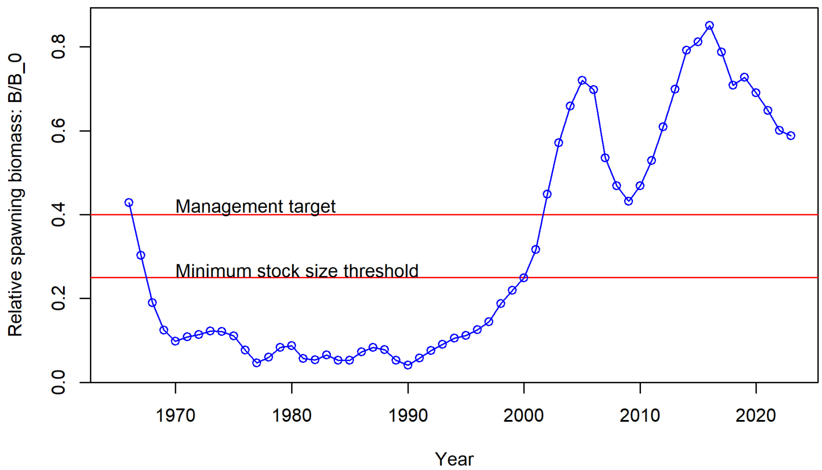

Figure 31. Spawning stock biomass relative to the unfished level: B/B0.

5. Model code

Data, scripts to organise data, the model folder and scripts to pot and analyse model results are available at internal storage: \\storage-dh.slu.se\restricted$\Stora sjoarna\Data\SS3\SS3 gös\SS3 gös_Hjälmaren\From Nataliia\ data and model 2025

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