



Effects of a fishery closure on the European eel stock on the Swedish west coast

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ABSTRACT

Fishing bans are implemented as management measures aimed at rebuilding depleted stocks, albeit with varying levels of success. In 2012, the commercial fishery for the European eel stock on the Swedish west coast was closed to increase the spawning migration from this local stock. However, despite being regarded as one of the most important management measures implemented in Swedish eel management, an assessment of the effects of this fishing closure on the eel stock has not previously been conducted. Here, in an integrated analysis, we use historical catch data in conjunction with standardized fisheries independent fyke-net data from three sites along the Swedish west coast to build a novel stock assessment model to quantify the effects of the fishery closure. The model is length-based, and escapement rates—that is, the rate at which eels escape from the system to migrate to and spawn in the Sargasso Sea—are estimated as an integral part of the analysis. Our results suggest that total escapement has increased fourfold compared to historical levels (compared to 1988–2011) and that the stock has reached its potential maximum escapement under current low recruitment conditions. Hence, the fishing closure seems to have had its intended effect of rebuilding a depleted stock, though it should be acknowledged that post fishing closure estimates of eel escapement are based on survey data from only three local sites. Nevertheless, our study constitutes a rare example of where the effects of management actions targeting European eel have been quantified.

Introduction

Fishery closures are widely recognized as a key conservation measure for mitigating overexploitation and promoting the recovery of depleted fish stocks (Kenchington et al., 2018). By reducing or eliminating fishing mortality, closures allow populations to rebuild, restore age and size structures, and improve recruitment success (Halpern et al., 2010). Such measures are particularly effective for species that are severely overfished or face additional environmental pressures. While fishery closures can be challenging to implement due to potential socio-economic impacts, especially for fishermen dependent on the fishery, they are increasingly being used as part of ecosystem-based management strategies aimed at achieving long-term sustainability (Gaines et al., 2010; Halpern et al., 2010), albeit with varying level of success (e.g. Brodie et al., 2010; Schrank, 2005; Shackell et al., 2021).

The panmictic European eel (*Anguilla anguilla*) has experienced a

dramatic decline over the past several decades, with recruitment levels falling by up to 95 % since the 1970s throughout its distribution range which spans from northern Norway and Iceland in the north to northern Africa in the south, and from the Atlantic shores of Europe and Africa in the west to the Mediterranean and Black Sea region in the east (Moriarty and Dekker, 1997; Dekker, 2004; ICES, 2024). The European eel undergoes a complex life cycle, where leaf-shaped Leptocephalus larvae hatch in the Sargasso Sea, then drift via currents to the coastal regions of the European and African continents where they metamorphose into eel-shaped transparent glass eels and colonize coastal and inlands waters. There, glass eels develop into pigmented yellow eels, grow large, and eventually develop into silver eels, at which state they initiate their spawning migration back towards the Sargasso Sea.

Due to its large distribution range and complex life-history, the decline of the European eel population has been attributed to a combination of factors, including overfishing, habitat loss, barriers to

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migration, pollution, and changing oceanic conditions (Dekker, 2004; Drouineau et al., 2018; Righton et al., 2025). As a result, the European eel has been classified as "Critically Endangered" since 2008 by the International Union for Conservation of Nature (IUCN) (Pike et al., 2020), prompting urgent management and conservation efforts across its distribution range.

The European Union introduced a recovery plan for the European eel in 2007 (Council Regulation (EC) No 1100/2007) with the requirement for member states to develop Eel management plans (EMPs); which Sweden completed in 2008 and adopted in 2009 (Jo2008/3901). Sweden implemented several conservation measures, restricting the commercial fisheries and licences, including the closure of the commercial eel fishery on the Swedish west coast in 2012. In addition, recreational fishing targeting eel was banned in 2007. The closure of the commercial eel fishery in 2012 was intended to reduce fishing mortality and contribute to the recovery of the European eel stock. As Sweden is managed as one single eel management unit in its EMP (Jo2008/3901), the closure of the west coast fishery is considered to counterbalance under-protection in other areas such as inland waters (Andersson et al., 2019).

Although the closure of the commercial eel fishery on the Swedish west coast was implemented over a decade ago, the potential effects of the closure on the eel population have not been quantified. Whilst there is some indication of increasing catch-per-unit-effort (CPUE) in standardized fisheries independent fyke-net surveys for large eels in the years directly after the ban (Andersson et al., 2019), changes in vital population statistics such as stock size, size structure and most importantly escapement (i.e. the biomass of eels initiating their migration towards the Sargasso Sea to spawn), are still unquantified. Assessing and understanding the effects of management measures, such as fisheries closures, is essential to determining their effectiveness and informing future management and conservation strategies.

To address this gap, this study aims to evaluate the effect of the 2012 eel fishery closure on the Swedish west coast. In an integrated stock assessment analysis (Maunder and Punt, 2013), using historical recreational and commercial catch statistics and data from a fisheries independent monitoring program with fyke-nets, a length-based model (Haddon, 2011; Zhang and Cadigan, 2022) is employed to analyse post-closure changes in population size, size structure and escapement. Previous studies assessing the eel stock on the Swedish west coast have done so using fisheries dependent data prior to the closure (Dekker, 2012; Jo2008/3901; Svedäng, 1999) or made preliminary analyses based on survey indices from fisheries independent data only a few years post-closure (Andersson et al., 2019). Hence, no study has so far evaluated the effects of the eel fishery closure at the stock level on the Swedish west coast. More specifically, it has not been assessed to what extent the fishery-closure has affected the local stock size, size structure and escapement of eels from the Swedish west coast.

By focusing on the biological impact of the fishery closure, this study provides the first evaluation of the potential recovery of the European eel stock on the Swedish west coast. Our findings suggest that escapement has increased approximately fourfold as compared to mean escapement levels (in 1988–2011) prior to the fishery closure in 2012, and that the stock is now in a state where escapement is at its maximum level under the current record low recruitment conditions. Overall, these findings contribute to the broader understanding regarding the efficiency of fishery closures for critically endangered species such as the European eel and support the development of evidence-based practices in fisheries management of exploited fish species.

Methods

In order to estimate changes in stock size, size structure and escapement of the European Eel stock on the Swedish west coast over time, we combined three primary data sources in an integrated stock assessment analysis (Maunder and Punt, 2013). The data sources consist

of standardized fisheries-independent fyke-net surveys from three sites along the Swedish west coast (Fig. 1, Fig. S1), commercial catch statistics from the period prior to the fishing closure in 2012 and recreational catch statistics prior to 2007. Although commercial catch statistics are also available prior to 1988 when the survey started, the analysis was restricted to the period where the fisheries-independent survey data were available. The stock assessment model was fitted in a Bayesian framework and implemented in the statistical software STAN (Gabry et al., 2024; Stan Development Team, 2024). Four independent Markov chains were run and each chain included a burn-in period of 1000 iterations, followed by 1000 iterations kept for posterior analysis. Initial values for the Markov chains were drawn from the prior distributions (Table 1). All code and data for reproducing the analyses are available in an open repository (<https://zenodo.org/records/17084120>).

Data

Survey data

Standardised fisheries-independent fyke-net surveys (HaV, 2015, 2016) are conducted annually in August by the Department of Aquatic Resources at the Swedish University of Agricultural Sciences (SLU Aqua). In these surveys, a predefined number of double-winged fyke-nets are soaked in the morning and lifted the following morning during a number of consecutive days, with the start date varying depending on weather conditions and other factors. The survey sites Fjällbacka, Ringhals and Stenungsund are presently ongoing time series which started in 1998, 1988, and 2002 respectively (Fig. 1, Fig. S1), and were originally selected to represent the fish community on shallow bottoms in relatively undisturbed reference areas.

The sampling program has been subject to revisions over the years. Prior to 2021, fyke-nets were set daily in a predefined number of randomly selected stations within each survey site (annual number of fyke-net days: Fjällbacka – 96 (1998–2020); Ringhals – 72 (1988–1997), 54 (1998–2024); Stenungsund – 90 (2002–2020)). From 2021 onwards, sampling in Fjällbacka and Stenungsund is conducted following a depth stratified sampling program (HaV, 2016) and consequently total effort has decreased (annual number of fyke-net days: Fjällbacka – 70; Stenungsund – 20).

The data in this study use the results from the 0–6 m depth strata, which has consistently been fished throughout the whole sampling period at each survey site. The number of fish in the catch, species and individual lengths are a few of the recorded variables (Andersson et al., 2019). Furthermore, number of eels caught is recorded per fyke net day (24 h), allowing for a CPUE time-series which can be used as a proxy for eel abundance (Fig. S1). As the length of eel caught in the fyke-nets has historically been measured with different precisions (1 and 2.5 cm), all individuals were lumped into 5 cm length-classes, letting the model represent eel dynamics per 5 cm class. Moreover, as the fyke-nets do not capture small eels representatively, a catch curve analysis was made and the peak CPUE in this catch curve was used as the minimum size-class represented by the model. This threshold, eels above 40 cm, corresponds well to the size limits historically being imposed on the Swedish west coast eel fishery (See *Commercial and recreational fishery catch statistics*).

Commercial and recreational fishery catch statistics

The Swedish west coast stretches from the Norwegian border to Öresund, i.e. 320 km coastline in Skagerrak and Kattegat (Fig. 1). Along this open coast there has historically been a fishery for yellow eels. In the late 1800s, rising international demand for eel led to a surge in eel fishing in Sweden, creating an opportunity to establish a commercial fyke-net eel fishery, targeting yellow eel on the Swedish west coast (Magnusson and Dekker, 2021). Prior to this, eels were primarily caught using baited pots and traps or longlines during the summer, and spears in the winter (Haneson and Rencke, 1923). Despite eel fishery landings beginning to decline across Europe in the 1960s (Dekker, 2003), those

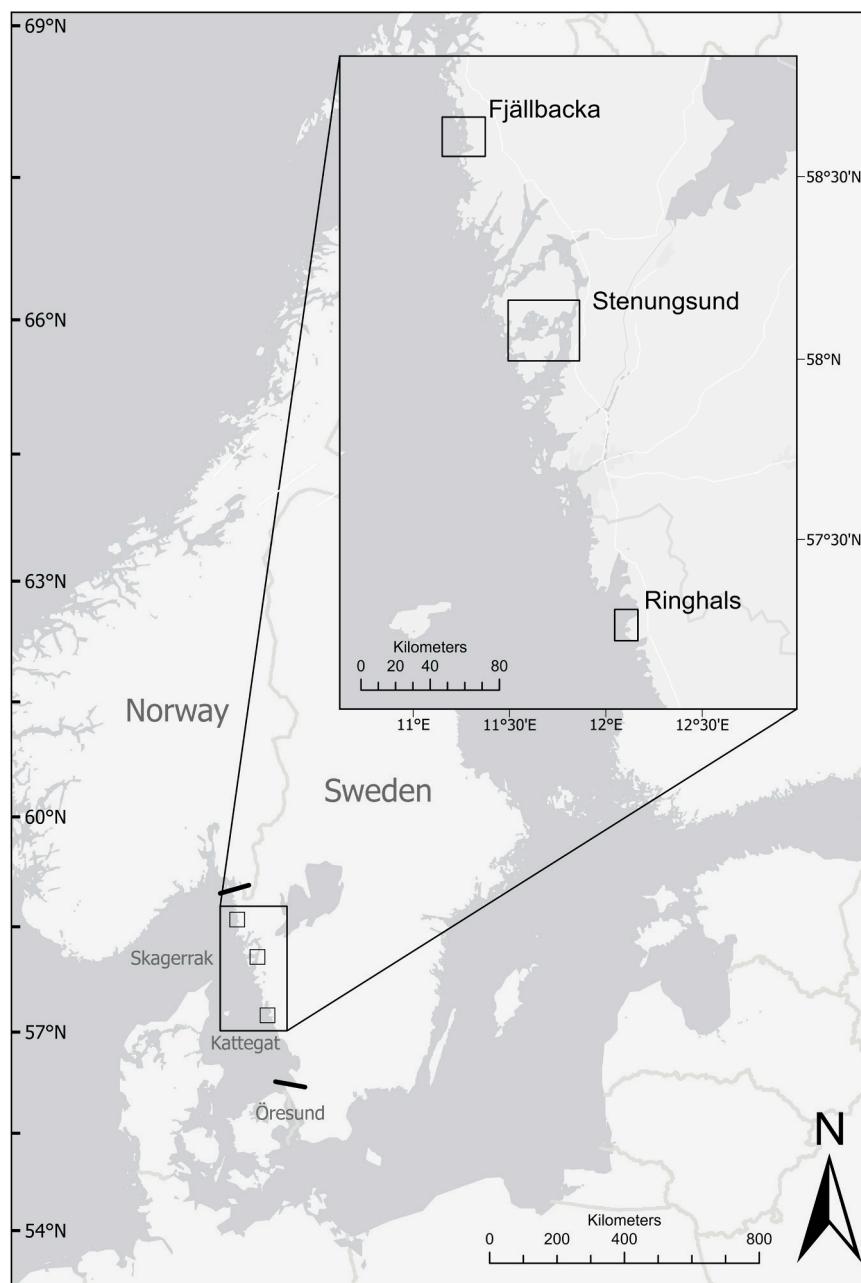


Fig. 1. Map showing locations of the standardized fisheries-independent fyke-net surveys along the Swedish west coast. Black thick lines show the northern and southern limits of the historical commercial eel fishery along the Swedish west coast.

on the Swedish west coast did not show a decline until the mid-1990s (Magnusson and Dekker, 2021). In fact, reported eel catches on the Swedish west coast gradually increased throughout most of the 20th century, peaking between 1980 and 2000 (Magnusson and Dekker, 2021). Most of the eel was exported, as local demand for yellow eel on the Swedish west coast remained low compared to in other countries. The eel fishery has always been a small-scale fishery, normally operated by single individuals (Andersson et al., 2019), and managed with minimal legislation. Yet, a minimum size limit has been established; 35 cm since 1907, 37 cm since the mid-1950s, and 40 cm from 2007. There is little documentation on the recreational fishery for eel on the Swedish west coast (Jo2008/3901), but it is likely that the recreational catch increased with the availability of cheap fyke-nets (van Gemert et al., 2024). From the data available recreational eel catches were close to commercial landings, just shy of 200 tonnes per year (<https://www.scb.se/>). The recreational fishery was closed in 2007, and the commercial

fishery was closed in 2012.

Model description

As the European eel is notoriously difficult to age, especially older individuals (Durif et al., 2020; ICES, 2009; Svedäng et al., 1998), population dynamics were assessed using a length-based model instead of an age-based model. The length-based model estimates growth and hence does not require age data (e.g. Haddon, 2011; Zhang and Cadigan, 2022). The population dynamics were assessed using predefined length-classes (5 cm length-classes) and abundances per length-class from the standardized fisheries-independent fyke-net surveys, total commercial catches and total recreational catches were used as independent data sources in the model. The model assumes that fyke-net time series are related to the “true” stock size through a simple multiplicative factor, a.k.a. a catchability coefficient, which is estimated in

Table 1

Prior distributions for all parameters in the model and references to equations in which they appear.

Param (θ)	PDF	Equations
$\begin{bmatrix} k \\ l_\infty \end{bmatrix}$	$MVN\left(\begin{bmatrix} 0.07 \\ 86.08 \end{bmatrix}, \begin{bmatrix} 0.0004 & -0.17 \\ -0.17 & 77.79 \end{bmatrix}\right)$	4, S1
CV_{inc}	Lognormal($\ln(0.3)$, 0.1)	5, S2
$F_{t=1988}^{com}$	Lognormal($\ln(0.18)$, 0.5)	7, S3
F^{rec}	Lognormal($\ln(0.13)$, 0.5)	S3
$\sigma_{lnF^{com}}$	Lognormal($\ln(0.3)$, 0.5)	8, S4
z_i	$N(0, 1)$	10
$\ln(\bar{r})$	$N(\ln(4 \cdot 10^6), 1)$	15, 19, S5
CV_r	Lognormal($\ln(0.2)$, 0.5)	17, S5
ϕ	Gamma(1, 1)	25, S6
$\sigma_{\beta_{at}}$	Half- $N(0, 1)$	S7
$\sigma_{\beta_{al}}$	Half- $N(0, 1)$	S7
β_a	$N(0, 1)$	27, S8
q	Lognormal($\ln(10^{-7})$, 2)	27, S9
CV_c	Lognormal($\ln(0.2)$, 0.2)	23, S10
$N_{t=1988}[l]$	Lognormal($\ln(N^*[l])$, 1)	18

the model and effectively scales the fyke-net data (number of eels caught per length-class and fyke-net day) to the scale of the whole system (total number of eels on the entire Swedish west coast). The scale of the system, that is, the stock size on the Swedish west coast, is further informed by historical catch statistics through the Baranov catch equation (Baranov, 1918).

Population process

The population dynamics is length-based, following a standard difference equation:

$$\mathbf{N}_t = \mathbf{GS}_{t-1}\mathbf{N}_{t-1} + \mathbf{R}_t \quad (1)$$

where \mathbf{N}_t is a $L \times 1$ vector of abundances per length-class at time step t , \mathbf{G} is a $L \times L$ growth transition matrix with elements p_{ij} defining the probability of moving from length-class j to length-class i in one time step, \mathbf{S}_t is a $L \times L$ diagonal matrix with diagonal elements, $S_{i,i,t} = e^{-Z_{i,t}}$, representing the survival of eels in length-class i in time step t , \mathbf{R}_t is a $L \times 1$ vector with recruitment in time step t and L represents the number of length-classes.

Growth transitions. Elements, p_{ij} , of the growth transition matrix, \mathbf{G} , define the probability of moving from length-class j to length-class i in one time step (See also Haddon, 2011; Zhang and Cadigan, 2022),

$$p_{ij} = \int_{l_{i-1}}^{l_i} \int_{l_{j-1}}^{l_j} f_\Delta(y-u) f_u(u) du dy \quad (2)$$

where $f_\Delta()$ is the probability density function (pdf) of growth increments $y-u$ over one time step and $f_u()$ is the pdf of initial length u before growth. For simplicity, we assume that all individuals in length-class j have the same initial length before growth; the midpoint of a length-class. In this case, the equation simplifies to,

$$p_{ij} = \int_{l_{i-1}}^{l_i} f_\Delta(y - \bar{l}_j) dy \quad (3)$$

where \bar{l}_j is the midpoint of length-class j ($\bar{l}_j = (l_j + l_{j-1})/2$).

Many different models may be used to model growth increments, $f_\Delta()$. Here, we use the Fabens (1965) version of the von Bertalanffy growth curve, which is a commonly applied growth model, and assume a normal distribution for variation in growth increments. That is,

$$\mu_x = (l_\infty - \bar{l}_j)(1 - e^{-k}) \quad (4)$$

and

$$(y - x) | x \sim N(\mu_x, \sigma_x) \quad (5)$$

where μ_x is the mean growth increment of an eel of length x (or \bar{l}_j), l_∞ is the average maximum (asymptotic) length (cm) of the von Bertalanffy growth curve, k is the maximum growth rate (cm yr^{-1}), y is the length to which an eel of length x may grow during one year and $\sigma_x = \mu_x \times CV_{inc}$, where CV_{inc} is the coefficient of variation of growth increments.

Survival. Survival, that is, the probability that an eel remains on the coast from one time step to the next is derived from multiple mortality components. For a given length-class and year, total mortality ($Z_{l,t}$) is given by:

$$\begin{cases} Z_{l,t} = M_l + E_l + F_t^{com} + F^{rec} & \text{if } t < 2007 \\ Z_{l,t} = M_l + E_l + F_t^{com} & \text{if } 2007 \leq t < 2012 \\ Z_{l,t} = M_l + E_l & \text{if } t \geq 2012 \end{cases} \quad (6)$$

where M_l is the natural mortality rate of length-class l , E_l is the escapement rate, that is, the rate at which eels in length-class l escape from the coast to migrate to and spawn in the Sargasso Sea ($E_l = 0$ for eels < 55 cm), F_t^{com} is the commercial fishing mortality rate (yr^{-1}) year t , assumed to be the same for all length-classes, and F^{rec} is a length-class and time invariant recreational fishing mortality rate (yr^{-1}). The different time periods $t < 2007$, $2007 \leq t < 2012$ and $t \geq 2012$ reflect the period where both recreational and commercial fishing was allowed, the period where only commercial fishing was allowed, and the period for the fishing ban, respectively.

Due to the presumed inertia in the commercial fishery, fishing mortality was modelled as a random walk process, and as fishing mortality can only take on positive values the random walk process was modelled on the ln-scale, following:

$$\ln(F_t^{com}) = \ln(F_{t-1}^{com}) + \epsilon_{lnF_t^{com}} \quad (7)$$

where $\epsilon_{lnF_t^{com}}$ is a normally distributed variable:

$$\epsilon_{lnF_t^{com}} \sim N(0, \sigma_{lnF^{com}}) \quad (8)$$

with a standard deviation of $\sigma_{lnF^{com}}$.

Recreational fishing mortality was assumed constant across years as it is likely that the effort of the recreational fishery targeting eel has historically been relatively constant.

It was explicitly assumed that escapement rate increases with eel length as the silverying process is expected to be size-dependent (van Gemert et al., 2024). Escapement rates, E , were therefore modelled as an ordered vector with escapement rates increasing over length-classes, such that:

$$E_1 < E_2 < \dots < E_n \quad (9)$$

This vector was obtained by first treating random variable z_i as distributed from a standard normal distribution:

$$z_i \sim N(0, 1) \quad i = 1, \dots, n_E + 1 \quad (10)$$

where n_E is the number of length-classes from which eels are assumed to escape from the system (eels > 55 cm). The vector of random normal variables was thereafter transformed using a softmax transformation:

$$z_i^{prob} = \frac{e^{z_i}}{\sum_{j=1}^{n_E+1} e^{z_j}}, \quad i = 1, \dots, n_E + 1 \quad (11)$$

which effectively converts any vector to the probability scale, ensuring:

$$\sum_{i=1}^{n_E+1} z_i^{prob} = 1 \quad (12)$$

Cumulative sums of these probabilities were then used to derive the

following length-class specific escapement probabilities:

$$P_{l-l_{E_{\min}}-1+i}^{\text{esc}} = \sum_{i=1}^{n_E} z_i^{\text{prob}} \quad (13)$$

where the probability, P_l^{esc} , represents the probability that an eel in length-class l escapes the system in one year. Here, $l_{E_{\min}}$, is the index of the smallest length-class from which eels are assumed to escape and n_E is the number of length-classes from which eels are assumed to escape from the system.

The difference in the upper limits in the summations in Eqs. (12) and (13) arises because, by definition, all eels in the largest length-class will not escape the system in one year. Hence, the probability that an eel in the largest length-class escape the system must be lower than 1.

Using escapement probabilities, P_l^{esc} , instantaneous escapement rate was calculated as:

$$E_l = -\ln(1 - P_l^{\text{esc}}) \quad (14)$$

Recruitment. In line with the general European decline in glass eel recruitment (ICES, 2024), recruitment of young eels to the Swedish west coast has also declined (van Gemert et al., 2024). However, this decline seems not to be reflected in the standing stock of yellow eels (Svedäng, 1999; Fig. S1), potentially due to density dependent processes occurring when young yellow eels colonize coastal and inland waters, moderating variation in glass eel recruitment (Acou et al., 2011; Bevacqua et al., 2011; Harrison et al., 2014). Thus, there are no clear reasons as to why recruitment of yellow eels (R_r) should be modelled in a specific way. For simplicity, we therefore assume that eels only recruit to the first length-class (l_{40-45}) considered in the model and that recruitment follows a lognormal process, that is:

$$r_t = e^{\ln(\bar{r}) + \epsilon_{lnr,t}}, t = 1989, \dots, T \quad (15)$$

where \bar{r} is expected recruitment over the years considered in the analysis and $\epsilon_{lnr,t}$ are normally distributed process errors:

$$\epsilon_{lnr,t} \sim N(0, \sigma_{lnr}) \quad (16)$$

where σ_{lnr} is the standard deviation of the recruitment process on the ln-scale, which is parameterized, for convenience, using a CV of recruitment on the arithmetic scale (Methot and Wetzel, 2013):

$$\sigma_{lnr} = \sqrt{\ln(1 + CV_r^2)} \quad (17)$$

Initial state. The initial states, that is, the length-class specific population sizes in 1988, were derived from the model equilibrium, with some process errors included to allow for some deviations from this equilibrium. Specifically, the initial states were derived from:

$$N_{t=1988}[l] \sim \text{Lognormal}(\ln(N^*[l]), 1) \quad (18)$$

where $N_{t=1988}[l]$ is the initial state and $N^*[l]$ is the model equilibrium for length-class l .

The equilibrium abundances were derived from:

$$N^* = (\mathbf{I} - \mathbf{G} \mathbf{S}_{t=1988}^{\text{com}} F^{\text{rec}})^{-1} \mathbf{R}_{r=e^{\ln(\bar{r}) + \sigma_{lnr}/2}} \quad (19)$$

where \mathbf{I} is the identity matrix, \mathbf{G} is the growth transition matrix, $\mathbf{S}_{t=1988}^{\text{com}} F^{\text{rec}}$ is the survival probability when commercial fishing mortality rate is assumed at its initial value and recreational fishing is affecting the stock, and $\mathbf{R}_{r=e^{\ln(\bar{r}) + \sigma_{lnr}/2}}$ is a vector with mean recruitment on the arithmetic scale as its first element and zero elsewhere.

Likelihood

The total composite log likelihood is composed of three parts, each part connecting individual data sources to the population dynamic

model (Eq. 1):

$$LL_{\text{tot}} = LL_{C^{\text{com}}} + LL_{C^{\text{rec}}} + LL_{\text{Survey}} \quad (20)$$

where LL_{tot} is the total log likelihood, $LL_{C^{\text{com}}}$ is the log likelihood for the commercial catch data, $LL_{C^{\text{rec}}}$ is the log likelihood for the recreational catch data and LL_{Survey} is the log likelihood for the survey data.

Likelihood commercial catch. Commercial catch was modelled following the Baranov catch equation (Baranov, 1918), assuming no size selectivity in the gear:

$$C_{l,t}^{\text{com}} = \frac{F_t^{\text{com}} w_l N_t[l] (1 - e^{-Z_{l,t}})}{Z_{l,t}}, \text{ for } t < 2012 \quad (21)$$

where $C_{l,t}^{\text{com}}$ is the commercial catch (tons) of eels in length-class l during year t and w_l is the average weight (in tons) of one individual in length-class l . w_l was derived from length-weight data from eels collected from the fishery independent fyke-net surveys ($n = 5660$; See Fig. S2).

Estimates of total annual commercial catch were thereafter obtained by summing length-class catches, that is:

$$E[C_t^{\text{com}}] = \sum_{l=1}^L C_{l,t}^{\text{com}} \quad (22)$$

where l is an index of the length-class being considered and L is the total number of length-classes in the model.

Modelled catches were coupled to data assuming lognormal observation errors, that is:

$$C_t^{\text{com}} \sim \text{Lognormal}(\ln(E[C_t^{\text{com}}]), \sigma_{\ln C_t^{\text{com}}}) \quad (23)$$

where $\sigma_{\ln C_t^{\text{com}}}$ was parameterized in terms of a coefficient of variation on the arithmetic scale (Eq. 17; Eq. S10; Methot and Wetzel, 2013).

Likelihood recreational catch. Recreational catch was modelled in the same way as the commercial catch, but recreational fishing mortality was, in contrast to commercial fishing mortality, assumed to be constant across years. Historical data on recreational catch, for years 1994 and 1999, were retrieved from Statistics Sweden (<https://www.scb.se/>). Modelled catches were coupled to data assuming lognormal observation errors, that is:

$$C_t^{\text{rec}} \sim \text{Lognormal}(\ln(E[C_t^{\text{rec}}]), \sigma_{\ln C_t^{\text{rec}}}) \text{ where } t \in \{1994, 1999\} \quad (24)$$

where C_t^{rec} is the mean catch observed for years 1994 and 1999 (200 and 174 tons, respectively), $E[C_t^{\text{rec}}]$ is the expected (modelled) recreational catch for year t , and $\sigma_{\ln C_t^{\text{rec}}}$ is the standard deviation of recreational catches on the ln-scale for years 1994 and 1999. For convenience, the ln-scale standard deviations were parameterized in terms of a coefficient of variation on the arithmetic scale (Eq. 17), where CVs were set to 2.07 and 2.41 based on highly uncertain recreational fishery data for years 1994 and 1999 (<https://www.scb.se/>), respectively.

Likelihood survey. The population dynamic model was coupled to count data ($n = 39258$) from the standardized fisheries independent fyke-net surveys using a negative binomial likelihood parameterized in terms of its ln-mean and a dispersion parameter (Stan Function Reference, 2024):

$$y_{l,t,a,i} \sim \text{NegBinLog}(\ln(\lambda_{l,t,a}), \phi) \quad (25)$$

where $\lambda_{l,t,a}$ is the expected number of eels in length-class l caught at time t in survey site a , and ϕ is a dispersion parameter. $y_{l,t,a,i}$ is the actual data, that is, the number of eels in length-class l caught at time t in survey site a , where i is an index of the observation.

The specific parameterization of the negative binomial distribution used in this model infers an assumed variance of observations according

to:

$$\text{Var}[y_{l,t,a}] = \lambda_{l,t,a} + \frac{\lambda_{l,t,a}^2}{\phi} \quad (26)$$

where the dispersion parameter, ϕ , scales how variance is related to the expected value of the process.

Survey observations were coupled to the population dynamic process (Eq. 1) through the following equation, allowing for site specific deviations from the general population dynamic trajectory:

$$\ln(\lambda_{l,t,a}) = \ln(q) + \ln(\mathbf{N}_t[l]) + \beta_a + \beta_{a,t} + \beta_{a,l} \quad (27)$$

where q is a scaling parameter, sometimes called a catchability coefficient, which when multiplied by the hidden state $\mathbf{N}_t[l]$ converts the hidden state for length-class l and time step t to the scale of the survey data. Linear contrasts between the population trajectory on the ln survey scale ($\ln(q\mathbf{N}_t[l]) = \ln(q) + \ln(\mathbf{N}_t[l])$) and site specific effects not explained by the hidden state were modelled through the parameters β_a , $\beta_{a,t}$ and $\beta_{a,l}$. These parameters explain general mean differences in number of eels caught per fyke-net between sites a and the hidden state, β_a , general differences in number of eels caught at time t and site a and the hidden state, $\beta_{a,t}$, and general difference in number of eels of length l caught in site a and the number of individuals of length l in the hidden state, $\beta_{a,l}$.

Priors

All prior distributions are defined in Table 1, and the rationale behind their choice is provided in Supplementary text 1.

Predictions

Vital population statistics follow naturally from the posterior distributions of the parameters related to the population process described in detail above (Population processes). However, one of the most essential population statistics to keep track of from an eel management and conservation perspective is escapement, which is a derived quantity.

Annual escapement from each length-class along the Swedish west coast, $\hat{B}_{l,t}$, was calculated using the Baranov catch equation (Baranov, 1918):

$$\hat{B}_{l,t} = \frac{E_l w_l \mathbf{N}_t[l] (1 - e^{-Z_{l,t}})}{Z_{l,t}} \quad (28)$$

where E_l is the escapement rate for length-class l , w_l is the mean weight (tons) of one individual in length class l obtained from a length-weight relationship for eels on the Swedish west coast (Fig. S2), $\mathbf{N}_t[l]$ is the standing stock of individuals in length-class l year t , and $Z_{l,t}$ is the total mortality of eels in length-class l year t .

Total escapement was derived by summing length-class specific escapement estimates, that is:

$$\hat{B}_t = \sum_{l=1}^{l_{\max}} \hat{B}_{l,t} \quad (29)$$

B_{best} , (van Gemert et al., 2024), that is, the maximum potential escapement given current recruitment levels, was derived by estimating escapement under equilibrium conditions, assuming mean recruitment and no fishing mortality:

$$B_{best,l} = \frac{E_l w_l \mathbf{N}^*[l] (1 - e^{-(M_l + E_l)})}{M_l + E_l} \quad (30)$$

$$B_{best} = \sum_{l=1}^{l_{\max}} B_{best,l} \quad (31)$$

where equilibrium abundances,

$$\mathbf{N}^* = (\mathbf{I} - \mathbf{G} \mathbf{S}_{F=0})^{-1} \mathbf{R}_{r=e^{\ln(\bar{r}) + \sigma_{lnr}/2}} \quad (32)$$

are defined as in Eq. (19) except that commercial and recreational fishing mortality is here set to zero.

Sensitivity analysis

As natural mortality, M_l , is uncertain and known to strongly affect model output in stock assessment models (Clark, 1999; Lee et al., 2011), and the recreational catch statistics are also very uncertain, we performed a sensitivity analysis comparing escapement outputs for various model inputs, varying M_l and recreational fishing mortality. With this regard, three different values of natural mortality were used, and recreational fishing mortality was either included or excluded in the models, creating a total of six different models. The natural mortality rate parameter, M_l , was based on: (i) Svedäng (1999) who used the total instantaneous mortality (Z) in a reference area on the Swedish west coast with no fishing, for years 1994–1997, as an estimate of natural mortality ($M_l = 0.23 \text{ yr}^{-1}$); (ii) van Gemert et al. (2024) who used a relatively low value of natural mortality for eels in Swedish inland waters ($M_l = 0.1 \text{ yr}^{-1}$); and (iii) Bevacqua et al. (2011) who built a natural mortality model for the European eel assuming that natural mortality depends on size, density and temperature. Here, we assume an average eel density and a mean water temperature of 10°C as input to this model, creating a length-class specific natural mortality vector ($\mathbf{M} = [0.073, 0.062, 0.053, 0.046, 0.041, 0.036]$), with natural mortality rates being lower than the other two natural mortality rates described above (i.e. i and ii).

An additional sensitivity analysis tested the sensitivity of escapement estimates and parameter estimates to changes in the priors of two of the parameters in the model. Specifically, prior means (on the arithmetic scale) were reduced and increased by 50 % for both the mean recruitment (\bar{r}) and the von Bertalanffy growth parameter k . These two parameters are likely two of the most important parameters affecting eel escapement as they directly affect the number of recruits to the stock (\bar{r}) and the rate at which eels grow into length-classes at which eels leave the system (k).

Results

General model convergence and parameter estimates

No divergent transitions and no major issues with the Hamiltonian Markov Chain (HMC) simulations were found (Stan Development Team, 2024). \hat{r} values indicate that the HMCs had mixed well (All $\hat{r} < 1.02$, Fig. S3). In general, parameter estimates were plausible (Tab. S1), with posterior estimates often having higher precision than priors, indicating that the model had learned from input data.

Model validation

Model fit was assessed using broad scale as well as more detailed posterior predictive checks. For more broad scale patterns, the model gave a good fit to observation data, predicting the number of eels caught per fyke-net day (Fig. S4-6) to satisfaction.

Zooming in on a somewhat more detailed level, model predictions of CPUE per length-class for the fyke-net survey sites are still plausible (Fig. 2), especially given that there are three data sources used as input data, which increases the potential for data conflicts. However, some discrepancies between model output and data clearly exist. For example, the model gives a relatively bad fit to fyke-net data in the survey site Fjällbacka (Fig. 2a), a system standing out in comparison to the other survey sites (Fig. 2b-c) as there are very few large eels being caught in this site (eels above 55 cm) and the dynamics of large eels appears to be very erratic (Fig. 2a).

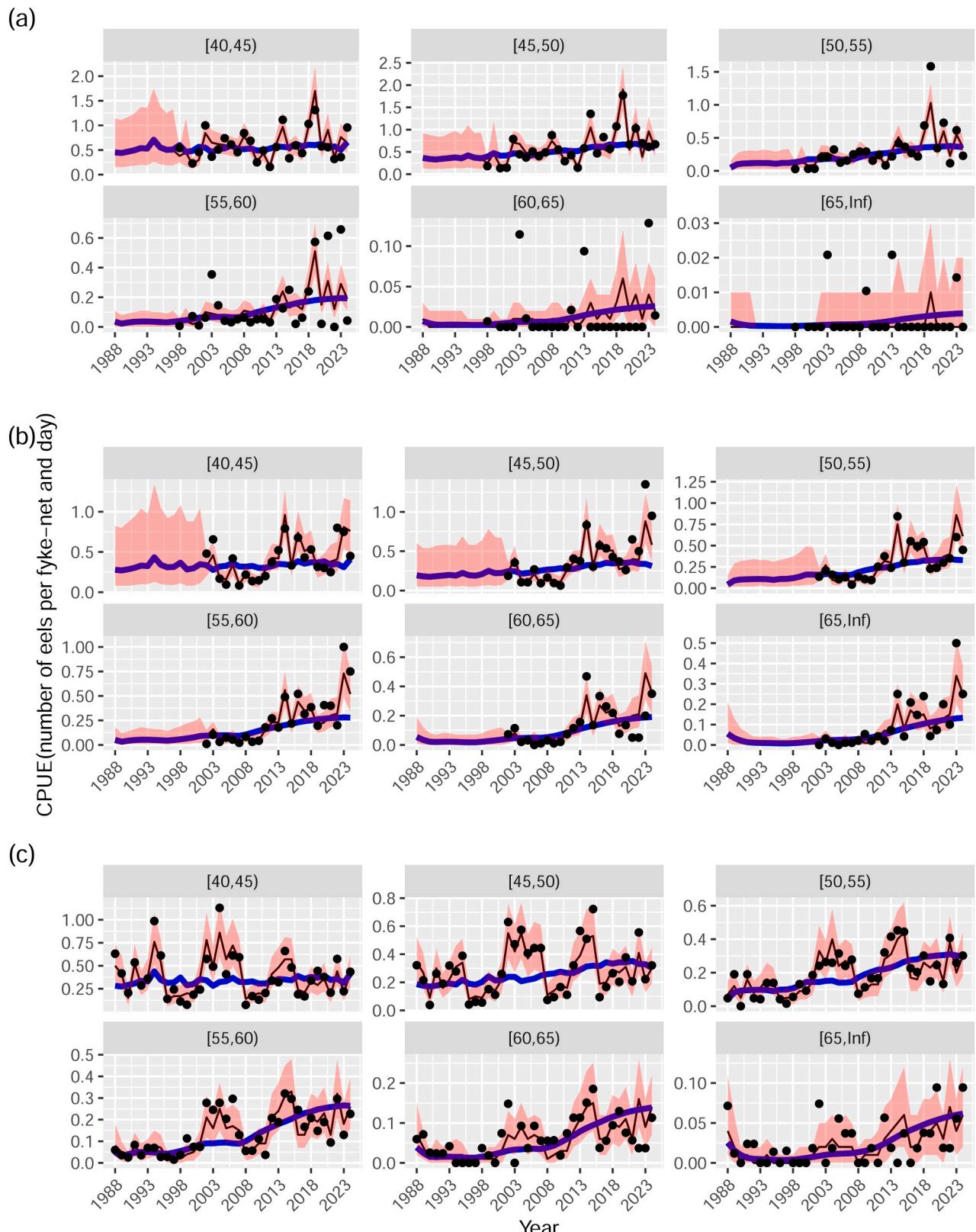


Fig. 2. Predictions and observations of catch per unit effort in the three survey sites **a**) Fjällbacka, **b**) Stenungsund and **c**) Ringhals. Black dots show observations, that is, the total number of eels in a specific length-class caught in the survey a given year, divided by the total number of fyke-net days. Black lines and pink areas show medians and 90 % credibility bounds of the posterior predictive distributions of the same quantity (Eq. 25). For comparison, blue thick lines show the population trajectory ($N_t[l]$) rescaled to the catch-per-unit effort scale for each survey site (i.e. number of eels caught per length-class, day and fyke-net [$\vartheta_{l,t,a} = qN_t[l]e^{\beta_a + \beta_{at}}$]). Subpanels show the development of different length-classes. The limits, indicated by the symbols “[” and “]” in the subpanel labels, denote eels greater than or equal to and less than the numbers displayed, respectively.

The model gives relatively good estimates of commercial catches, although it underestimates catch in years with large commercial catches (Fig. 3). For recreational catches, posterior medians are somewhat lower than data although predictions are highly uncertain, reflecting the large uncertainty in recreational catch statistics used as input in the model (Fig. S7).

Apart from giving reasonable predictions of input data (Figs. 2–3), the model also appears to be robust against changes in prior means of mean recruitment (\bar{r}) with posterior estimates being only modestly affected by these changes (Tab. S2). However, escapement estimates are sensitive to changes in the prior mean of the growth parameter k (Tab. S2).

Predictions

As with any age- or length-based stock assessment model, stock size or any other population statistic depends on the input to the stock, that is, recruitment (Fig. 4). Here, we assume that recruitment is a stationary process, and that recruitment varies randomly over time. In the model, mean recruitment is estimated at about 4.8 [3.7, 6.9] (median [5 % Credibility limit, 95 % credibility limit]) million eels yr^{-1} , with the highest recruitment occurring in year 1994, estimated at about 7.1 [5.0, 11.0] million eels (Fig. 4b).

Although recruitment to the fishery shows no emergent pattern over time (Fig. 4b), the model clearly suggests that the abundance of eels in all length-classes above the recruitment length-class (l_{40-45}) have increased over time (Fig. 4a), a pattern which is specifically pronounced for large eels (>45 cm). For length-classes above 55 cm, which is the assumed size threshold above which eels can develop into silver eel and leave the system, the model suggests that there has been a sharp increase in standing stock from the record low years around the mid 90's. Moreover, the model suggests a total standing stock of eels on the Swedish west coast at approximately 21.1 [15.8, 31.8] million individuals in 2024, a value that should be set in contrast to the beginning of the time series (in 1988) where total eel abundance is estimated at 10.0 [7.5, 14.5] million eels.

Escapement from the system depends on the standing stock of large

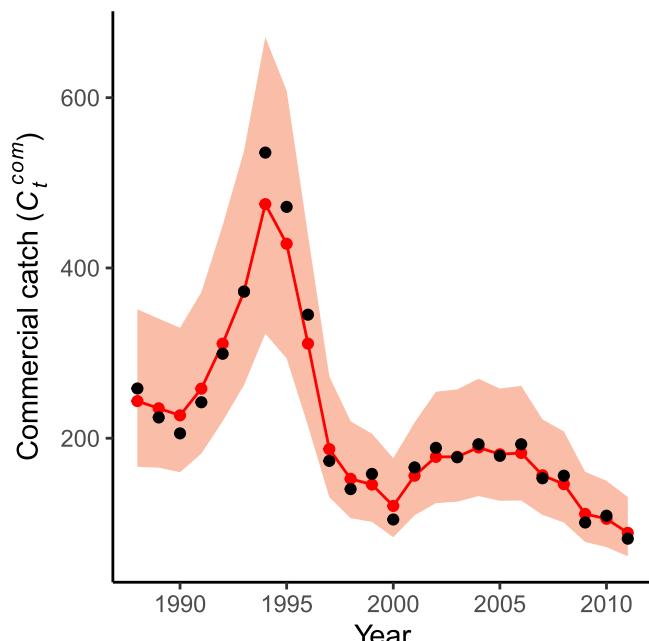


Fig. 3. Predictions and observations of total commercial landing (tons) of the European eel stock on the Swedish west coast. Black dots illustrate observed landings, the red line with dots illustrates the median of the posterior distribution and the pink area displays 90 % credibility bounds.

eels (eels > 55 cm), natural mortality, fishing mortality and the rate at which large eels (eels > 55 cm) escape from the system (Table 2). In the model, escapement rate is assumed to increase with eel length and estimated escapement rates, E_t , correspond to an annual escapement of approximately 23 % [3 %, 44 %] of the eels in length-class l_{55-60} , 38 % [10 %, 59 %] of the eels in length-class l_{60-65} and 56 % [16 %, 90 %] of the eels in length-class $l_{65-\infty}$ (Table 2).

As recruitment and fishing mortality are the only time varying factors directly affecting the standing stock and escapement of eels in the model, it is natural to follow how fishing mortality and total escapement relate over time (Fig. 5). The model suggests that fishing mortality peaked around 1994 with a total fishing mortality ($F = F_{t=1994}^{\text{com}} + F^{\text{rec}}$; 0.40 [0.29, 0.53]) corresponding to an approximate 32.9 % [25.2 %, 41.3 %] annual removal from the stock. According to the model, the high fishing mortality in the 90's also corresponds to a record low total escapement. Following a decline in commercial fishing mortality at the end of the 90's, escapement increased and stabilized on a somewhat higher level (~ 80 –90 tons). Later, after the implementation of the recreational fishing ban in 2007 and a decrease in commercial fishing mortality, total escapement increased again, eventually reaching the maximum potential escapement under the current level of mean recruitment ($B_{\text{best}}=286$ [153, 436]; "main model" in Table 3) around year 2021, i.e., approximately a decade after the fishing closure. Hence, the model suggests that, following the closure of recreational (closed in 2007) and commercial fisheries (closed in 2012), the stock has now recovered, and escapement is approximately four times higher (4.32 [3.51, 5.55]) than the mean escapement level during the years where fishing was allowed (year 1988–2011) (Table 3).

It is important to note that estimates of total escapement are strongly affected by model assumptions. Therefore, six different models were set up to test assumptions on natural mortality and recreational fishing mortality, and indeed, by varying natural mortality and the recreational fishing mortality assumption, escapement estimates are approximately four times higher for the model with highest escapement, B_{best} , (model 3; 811 [610, 1192]) as compared to the model with the lowest escapement estimate (model 4; 182 [97, 245]) (Table 3). Estimates of escapement, B_{best} , are sensitive to changes in natural mortality and increase with decreasing natural mortality. Inclusion of recreational fishing mortality in the model also increases escapement, B_{best} , compared with the case where recreational fishing mortality is not included in the model (Table 3). Nevertheless, although the estimated level of total escapement depends strongly on model input, the general time varying pattern in total escapement is robust to model choice (Table 3). That is, all models tested suggest that the Swedish west coast eel stock has recovered under the current level of mean recruitment.

Discussion

Since the complete closure of the eel fisheries on the Swedish west coast (recreational in 2007 and commercial in 2012), we find that the stock size has increased and that the escapement of eels initiating their spawning migration has increased. Hence, the fishing closure seems to have had its intended effect, to increase escapement of eel from the Swedish west coast. These findings are in line with the general recognition of fishery closures being a key conservation measure for mitigating overexploitation and promoting the recovery of depleted fish stocks, thus allowing exploited populations to rebuild and their age- and size-structures to be restored (Halpern et al., 2010; Kenchington et al., 2018). Although both recoveries and non-recoveries following fishery closures have been reported for various fish species' stocks (e.g. Brodie et al., 2010; Schrank, 2005; Shackell et al., 2021), evidence from eel fishery closures are scarce (Cutts et al., 2024), as are general evaluations of eel management implementations (ICES, 2025). Nevertheless, one example exists, the closure of the recreational and commercial eel fishery in Ireland in 2009. Three years after the fishery closure, silver eel production, recapture rates of silver eels and average length of female

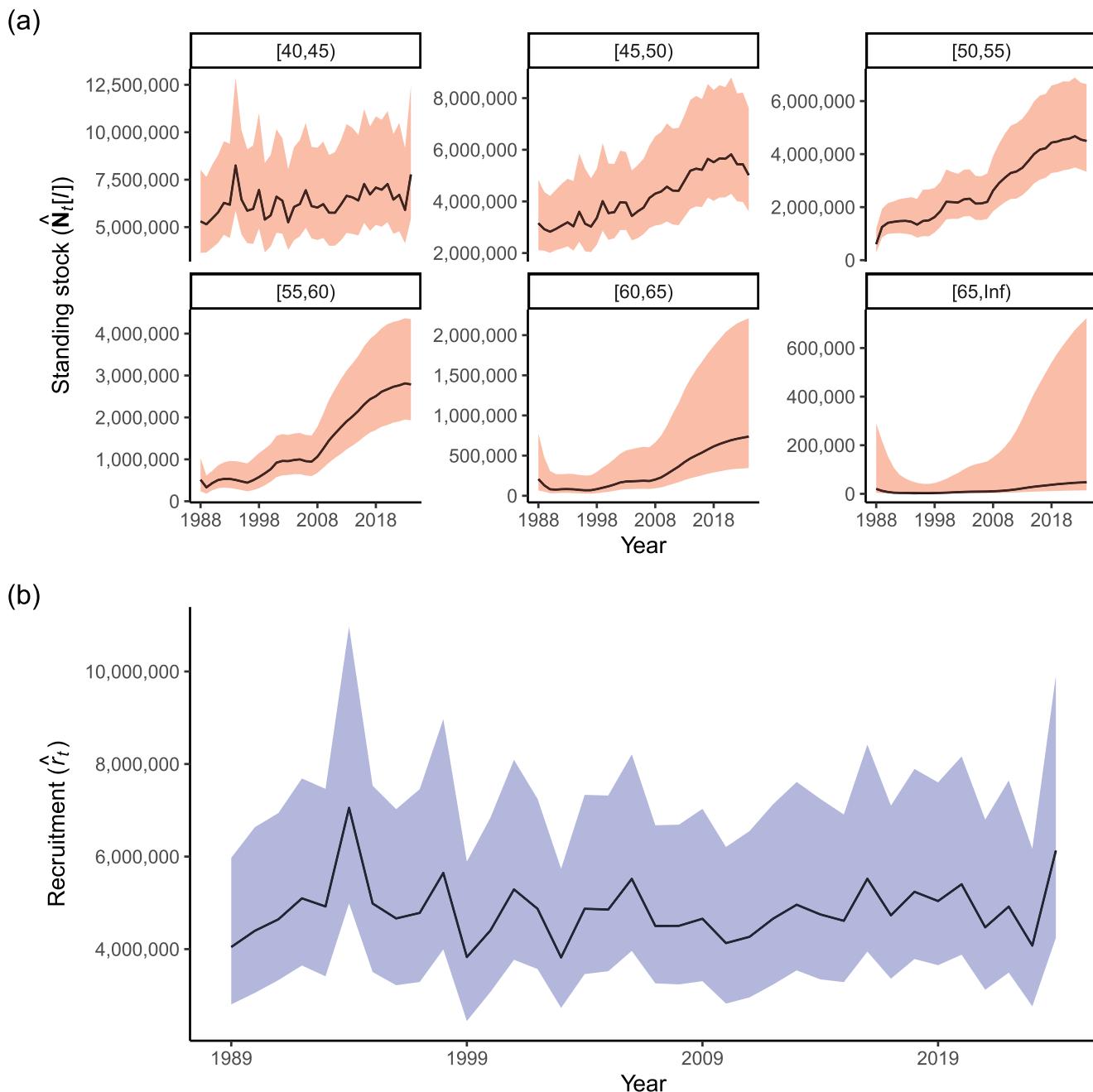


Fig. 4. Estimates of the standing stock ($\hat{N}_t[l]$) and recruitment (\hat{r}_t) of the European eel on the Swedish west coast. **a)** Model estimates of the development of the number of eels per length-class on the Swedish west coast ($\hat{N}_t[l]$). The black lines show the median and red areas show 90 % credibility bounds of the posterior distribution. Subpanels show the development of different length-classes. The limits, indicated by the symbols “[” and “]” in the subpanel labels, denote eels greater than or equal to and less than the numbers displayed, respectively. **b)** Estimates of the number of eels recruiting (\hat{r}_t) to the smallest length-class considered in the model (l_{40-45}). The black line illustrates the median and blue areas 90 % credibility bounds of the posterior distribution.

eels had not changed considerably in River Shannon as compared to before the closure (MacNamara and McCarthy, 2014). In other words, no clear effects of the fishery closure were observed. However, because eels are long-lived and our model predicts a recovery time of around a decade, studies longer than three years may be required to detect the effects of different eel management measures (Åström and Dekker, 2007). Moreover, additional threats, including up and down stream migration barriers such as hydroelectric dams likely limited the observable direct effect of the Irish fishery closure (MacNamara and McCarthy, 2014). Hence, other anthropogenic impacts such as migration barriers need to be considered when evaluating the effects of fishery

closures of highly migratory fish species such as eel. Yet, for the Swedish west coast, fishing mortality is the only known anthropogenic mortality source on the standing stock of eels (Magnusson and Dekker, 2021), potentially allowing for the recovery of the stock we have observed.

The model framework and estimates presented here, suggesting that the eel stock has recovered on the Swedish west coast, reside on the assumption that eels are stationary, forming a local stock affected by natural processes and local management measures. It is well known that eel can grow large and spend their whole growth life-stage as yellow eels in saline coastal habitats and shift between saline and freshwater systems (e.g. Daverat et al., 2006; Denis et al., 2023; Durif et al., 2023;

Table 2

Escapement rate parameters. This table shows posterior estimates and priors for length-class specific escapement rates, E_l . θ refers to marginal posterior distributions and θ_{pr} refers to prior distributions. Values refer to the median, and inside brackets the 5th and the 95th percentiles, of the posterior and prior distributions, respectively. Prior distributions were acquired through simulations and posterior estimates obtained from the model fit.

Param	Posterior(θ)	Prior(θ_{pr})
E_{55-60}	0.26 [0.03, 0.58]	0.22 [0.03, 0.92]
E_{60-65}	0.48 [0.10, 0.89]	0.70 [0.17, 1.85]
$E_{65-\infty}$	0.83 [0.18, 2.33]	1.58 [0.44, 3.31]

Tsukamoto et al., 1998). Results from otolith micro-chemistry analysis of yellow eels caught at the Swedish west coast further support the assertion that these yellow eels are stationary, although only a few individuals have been analysed from a few sites (Myrenås, 2024; Tzeng et al., 1997). In addition, the long tradition of a developed commercial yellow eel fishery along the Swedish west coast (Magnusson and Dekker, 2021; Svedång, 1999) and continuous catches of differently sized yellow eel in the monitoring program along the coast (See Survey data in

Methods; Figs. 1–2; Fig. S1) indicate that yellow eel utilizes the coast to feed and grow large. However, future studies are needed for assessing the habitat utilization of yellow eels along the Swedish west coast, studies that could either further assess the microchemistry of yellow eel otoliths caught along the coast or use large-scale mark-recapture techniques for eels tagged on the coast. Such studies would confirm the validity of the modelling approach taken here, choosing the stock unit to be the geographically confined area of coastal eel habitats along the Swedish west coast.

Aside from natural recruitment, young eels have also entered the Swedish west coast through the release of glass eels imported from abroad (usually referred to as restocking). Releases peaked between 2011 and 2020, during which an average of 840,000 glass eels per year were released along the coast (van Gemert et al., 2024; <https://aqua.slu.se/al/>). Assuming a length-at-release of 9.7 cm, an average growth of 6.0 cm yr⁻¹ (Myrenås et al., 2024; See also Fig. S8), and a natural mortality of 0.23 yr⁻¹, an average of 260,000 recruits of 40 cm are produced from 840,000 released glass eel after 5 years. Compared to this study's estimate of an annual recruitment of around 4760,000 individuals of 40 cm to the Swedish west coast, the total impact of restocking on the abundance of eel at Swedish west coast would then

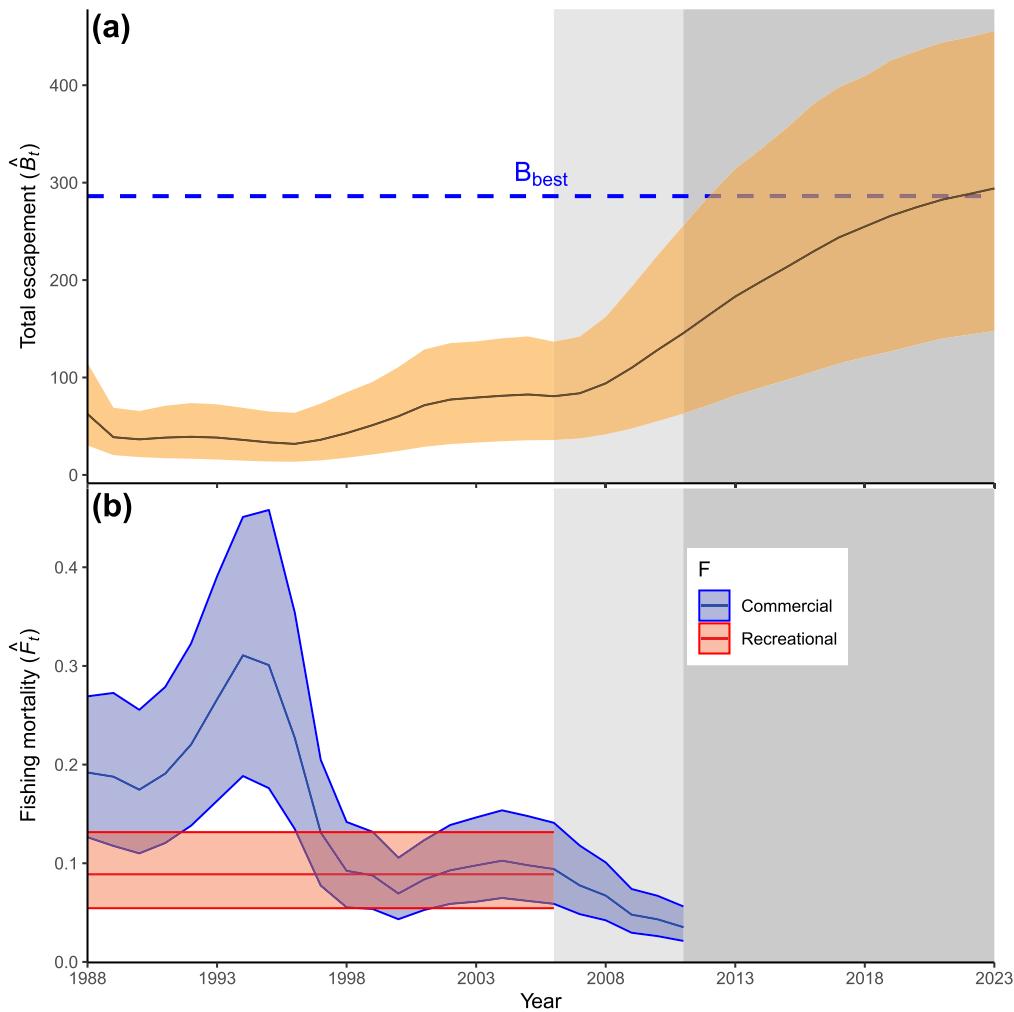


Fig. 5. Effect of the fishery closure on total escapement (in tons) from the Swedish west coast. Subpanels show estimates of a) total escapement (\hat{B}_t) and b) commercial (\hat{F}_t^{com} ; in blue) and recreational (\hat{F}_t^{rec} ; in red) fishing mortality. Background colors highlight different time periods associated with different fishery management regulations: a period where both commercial and recreational fishing was allowed (Year < 2007; white), a period where only commercial fishing was allowed (2007 ≤ Year < 2012; light grey) and a period with a complete fishing closure (Year ≥ 2012; dark grey). Lines display medians of the posterior distribution and colored areas show 90 % credibility bounds. The blue dashed line shows the median of the posterior distribution of \hat{B}_{best} , the maximum number of eels escaping from the system under the current level of mean recruitment, and no anthropogenic mortality.

Table 3

Estimates of total escapement depending on assumptions on natural mortality and recreational fishing mortality (the rationale behind the choice of model inputs is provided in methods section "Sensitivity analysis"). B_{best} , maximum escapement under the current level of mean recruitment, was derived from Eqs. (30–32). $\bar{B}_{1988–2011}$ represents estimates of mean annual escapement for the years when fishing was allowed, i.e. years 1988–2011. $\Delta\bar{B}_{best}$ represents how much escapement could potentially increase, i.e. the recovery potential of the stock, in comparison to the mean escapement for the fishery period (years 1988–2011) ($\Delta\bar{B}_{best} = B_{best}/\bar{B}_{1988–2011}$).

Model	Natural mortality (M_t)	Recreational fishing	B_{best}	$\bar{B}_{1988–2011}$	$\Delta\bar{B}_{best}$
Main model	0.23	Yes	286 [153, 436]	66 [30, 115]	4.32 [3.51, 5.55]
Model 2	0.1	Yes	607 [432, 877]	133 [74, 219]	4.59 [3.63, 6.38]
Model 3	Size based	Yes	811 [610, 1192]	179 [113, 285]	4.53 [3.62, 6.06]
Model 4	0.23	No	182 [97, 245]	43 [20, 67]	4.24 [3.44, 5.48]
Model 5	0.1	No	365 [276, 450]	80 [45, 120]	4.54 [3.53, 6.52]
Model 6	Size based	No	491 [407, 581]	112 [74, 157]	4.38 [3.49, 5.88]

appear to be relatively small ($\sim 5\%$), especially as this number is likely in the upper range of the potential surplus production of eels that would be expected for eels originating from restocking. As in other fish species, young eels experience higher mortality rates than older individuals (Bevacqua et al., 2011; Lorenzen, 2022). Hence, a natural mortality at 0.23 yr^{-1} , which is assumed in this calculation experiment, likely results in an eel recruitment of 40 cm eels that is higher than would be the case if the mortality of young individuals was specifically considered. However, the local impact in the area at which restocking took place could be larger. For instance, Myrenås (2024) found that around 26 % of eels caught in the Stenungsund fyke-net survey originated from restocking over the years 2013–2021, and restocking has been significant in the surrounding of Stenungsund (Myrenås, 2024; van Gemert et al., 2024). Moreover, it also appears as if eel abundances are higher in the Stenungsund survey site as compared to the estimated general population trend (Fig. 2b), suggesting that local restocking events may have led to a local surplus production of yellow eels in the area which is over and above the levels expected from the overall west coast stock development. For the other two survey sites, Fjällbacka and Ringhals, no clear pattern between estimates of local abundances and the overall population trajectory are observed (Fig. 2a & c), suggesting that restocking may mostly have local effects and may not affect the overall recruitment of eels to the Swedish west coast to a large extent.

Here, in a relatively complex analysis, historical catch statistics were used in conjunction with standardized fisheries independent fyke-net data to quantify the potential effects of a fishing closure for eels on the Swedish west coast. The fyke-net CPUE data is notoriously noisy and the extent to which this data informs predictions is hard to assess. However, the general population dynamic trend, showing increasing abundances for the larger length-classes (Fig. 4), appears to be apparent in all fyke-net survey sites (Fig. 2; Fig. S1). A simple variance partitioning of predictions further supports this assertion as model predictions for the fyke-net data in all survey sites are dominated by the population trajectory for the larger length-classes (Tab. S3). Hence, an increase in abundance, following the fishing closure, is not merely an artefact of the model assuming that catches and fishing mortality

determines the response of the stock, as an increased abundance of the largest length-classes is directly observed in all fyke-net survey sites (Fig. 2). Moreover, length-class specific population trajectories (Fig. 4), and implicitly assumed growth parameters (Tab. S1), could not be informed by data if the CPUE per length-class data from fyke-net surveys were not available. Hence, although the survey data is noisy, both the general pattern of increasing abundances of the larger length-classes in the survey data following the fishing closure on the Swedish west coast, and the size distribution within survey sites, are indeed informing the population trajectories (that are moreover very uncertain due to the noisy nature of the fishery independent fyke-net data).

The primary reason for building a length-based model, rather than a more commonly used age-based model, for the European eel stock on the Swedish west coast is that age-reading of the European eel is biased and error-prone (Durif et al., 2020; ICES, 2009; Svedäng et al., 1998). If unaccounted for, age-reading bias would likely lead to biased inference (Bertignac and de Pontual, 2007; Reeves, 2003). Hence, the population dynamics of this difficult-to-age-fish are likely better represented by a length-based rather than an age-based model (Zhang and Cadigan, 2022). In the length-based model, the CPUE per length-class data from fyke-net surveys are the only data source that informs the growth process (Eqs. 4–5), but priors assigned to growth parameters are also influential. Here, priors on the growth parameters k and l_∞ were assigned based on an external analysis in which a von Bertalanffy growth model was fitted to a large length-stratified age-length data set on eels from the Swedish west coast (Fig. S8; Supplementary text 1). As this large age-length data set was not used inside the model, it could be considered underused. In this regard, we therefore propose future studies to develop age-reading bias correction for the European eel and then include this information in a separate likelihood component in the model. Further, as the age-length data is length-stratified, which infers that fish age can be considered random given length, a conditional age at length likelihood component (Methot and Wetzel, 2013; Piner et al., 2016; Perreault et al., 2020) accounting for age-reading bias and age-reading errors, is specifically proposed. With such an extension of the model, the growth process might be better informed, potentially leading to better informed estimates of escapement of the European eel stock from the Swedish west coast.

It has been estimated that approximately 0.2 million eels migrated annually from the Swedish west coast to spawn in the Sargasso Sea during 1994–1997 (Svedäng, 1999). Assuming an average weight of a 65 cm eel, which is the size at which all eels are assumed to migrate from the system in Svedäng (1999), at 0.47 kg (Fig. S2) this estimate corresponds to a total escapement at about 94 tons yr^{-1} . For the same time-period, our model suggests a total escapement at about 34 [14, 67] tons yr^{-1} . As recruitment and fishing mortality estimates are similar in the two studies (estimated recruitment (eels above 40 cm), $\bar{r}_{t=1994-1997}$, at 5.4 [4.2, 7.8] million eels yr^{-1} in this study vs. 5.4 million eels (eels above 37 cm) yr^{-1} in Svedäng 1999, and $F = \bar{F}_{t=1994-1997}^{\text{com}} + F^{\text{rec}}$ at 0.33 [0.25, 0.43] yr^{-1} in this study as compared to $F = 0.31\text{ yr}^{-1}$ in Svedäng 1999) the most important factor causing differences in escapement estimates is likely the different growth models used in the two studies. Svedäng (1999), who estimated a linear growth rate at 4.5 cm yr^{-1} , calculated the time it takes (for an eel from entering the fishery, i.e. eels $> 37\text{ cm}$) to reach a silver eel size of 65 cm at 6.2 years. By comparison, according to the length-based model used here, it would take on average 11.6 years for an eel entering the fishery at length 37 cm to reach the same silver eel size (at the posterior medians and an age at length relationship according to the von Bertalanffy growth function [$\text{Age}_t = t_0 - \frac{1}{k} \ln(1 - \frac{L}{L_\infty})$]). Hence, as the annual survival (the combined effect of fishing mortality and natural mortality) and total recruitment in the two studies are similar, it is likely mainly the different times it takes to reach the silver eel size, and henceforth the different total life-time survivals, that causes the difference in escapement estimates between the two studies. Moreover, it is worth noting that the estimates provided

here are based on a length-based population dynamic model, explicitly accounting for time varying changes in catches, recruitment and stock sizes, whereas Svedäng (1999) used a static approach assuming fishing mortality, recruitment and catches were fixed and hence time invariant. Nevertheless, it is encouraging that estimates from two studies, using vastly different modelling approaches, are not too dissimilar.

Our estimates suggest that the closure of the commercial and recreational eel fisheries on the Swedish west coast has led to the intended effect of increasing the standing stock and silver eel escapement. According to the model, escapement has increased approximately fourfold compared to mean escapement levels prior to the complete closure of the fishery in 2012 and that the eel stock on the Swedish west coast is now, under the current record low recruitment levels (ICES, 2024), in a fully recovered state. However, the whole European eel population is threatened also by multiple other factors such as the direct mortality caused by hydropower turbines and pumping stations, loss of inland habitats due to intensive damming of water course, water pollution and climate change (Cutts et al., 2024; Dekker, 2004; Drouineau et al., 2018; Righton et al., 2025). Hence, fishing closures are just one of the conservation measures that could be implemented to restore the current depleted European eel population (Cutts et al., 2024). Still, our study clearly shows that fishing closure is a local conservation measure that can help the recovery of the wide-ranging, panmictic European eel population.

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CRediT authorship contribution statement

Rob van Gemert: Writing – review & editing, Writing – original draft, Project administration, Methodology. **Philip Jacobson:** Writing – review & editing, Writing – original draft, Visualization, Methodology. **David Gilljam:** Writing – review & editing, Methodology, Conceptualization. **Torbjörn Säterberg:** Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Per B. Holliland:** Writing – review & editing, Writing – original draft.

Declaration of Competing Interest

The authors have no conflicts of interest to declare.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.fishres.2025.107564](https://doi.org/10.1016/j.fishres.2025.107564).

Data availability

<https://zenodo.org/records/17084120>

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