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Assessment of the eel stock in Sweden, spring 2024

– Fifth post-evaluation of the Swedish eel management

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Summary

For decades, the population of the European eel has been in severe decline. In 2007, the European Union decided on a Regulation establishing measures for the recovery of the stock, which obliged Member States to implement a national Eel Management Plan by 2009. Sweden submitted its plan in 2008. According to the Regulation, Member States shall report regularly to the EU-Commission, on the implementation of their Eel Management Plans and the progress achieved in protection and restoration. The current report provides an assessment of the eel stock in Sweden as of spring 2024, intending to feed into the national reporting to the EU in August this year. This report updates and extends previous evaluation reports by Dekker (2012, 2015) and Dekker et al. (2018, 2021).

In this report, the impacts on the stock - of fishing, restocking and mortality related to hydropower generation - are assessed. Other anthropogenic impacts (climate change, pollution, increased impacts of predators, spread of parasites, disruption of migration due to disorientation after transport, and so forth) probably have an impact on the stock too, but these factors are hardly quantifiable, and no management targets have been set. For that reason, and because most factors were not included in the EU Eel Regulation, these other factors are not included in this report. Our focus is on the quantification of silver eel biomass escaping from continental waters towards the ocean (current, current potential and pristine) and mortality risks endured by those eels during their whole lifetime. The assessment is broken down on a geographical basis, with different impacts dominating in different areas (west coast, inland waters, Baltic coast).

In the last decade, a break in the downward trend in glass eel recruitment has been observed, with recruitment no longer declining consistently. Whether that relates to recent protective actions, or is due to other factors, is yet unclear. Nevertheless, recruitment levels remain at historically low levels. This report contributes to the required international assessment, but does not discuss the causing factors behind the recent recruitment trend and the overall status of the stock across Europe.

For the different assessment areas, results summarise as follows:

On the west coast, a commercial fyke net fishery on yellow eel was exploiting the stock, until this fishery was completely closed in spring 2012. A fishery-based assessment no longer being achievable, we present trends from research surveys (fyke nets). Insufficient information is currently available to assess the recovery of the stock in absolute terms. Obviously, current fishing mortality is zero (disregarding the currently unquantifiable effect of illegal fishing), but none of the other requested stock indicators (current, current potential and pristine biomass) can be presented. The formerly exploited size-classes of the stock show a recovery in abundance after the closure of the commercial fishery, and the smaller size classes show a break in their decline in line with the recent global trend of glass eel recruitment.

In order to support the recovery of the stock, or to compensate for anthropogenic mortality in inland waters, young eel has been restocked on the Swedish west coast since 2010. Noting the quantity of restocking involved, the expected effect (ca. 50 t silver eel) is relatively small, and hard to verify – in comparison to the potential natural stock on the west coast (an order of 1000 t). However, for the currently depleted stock, the contribution will likely constitute a larger share of silver eel escapement.

For inland waters, this report updates the 2021 assessment, with substantial changes in methodology being the use of a new natural recruitment model, and the full separation of Trap & Transport catches from the fisheries statistics. The assessment for the inland waters relies on a

reconstruction of the stock from information on the youngest eels in our waters (natural recruits, assisted migration, restocking). Based on 78 years of data on natural recruitment into 22 rivers, a statistical model is applied which relates the number of immigrating young eel caught in traps to the location and size of each river, the distance from the trap to the river mouth, and the year in which those eels recruited to continental waters as a glass eel (year class). The further into the Baltic, the larger and less numerous recruits generally are. Distance upstream comes with less numerous recruits.

Using the results from the above recruitment analysis, in combination with historical data on assisted migration (young eels transported upstream within a drainage area, across barriers) and restocking (young eels imported into a river system), we have a complete overview of how many young eels recruited to Swedish inland waters. From this, the production of fully grown silver eel is estimated for every lake and year separately, based on best estimates of growth and natural mortality rates. Subtracting the catch made by the fishery (as recorded) and down-sizing for the mortality incurred when passing hydropower stations (percentwise, as recorded or using a default percentage), an estimate of the biomass of silver eel escaping from each river towards the sea is derived.

Results indicate, that since 1960, the production of silver eel in inland waters has declined from over 700 to below 300 tonnes per year (t/yr). The production of naturally recruited eels is still falling; following the increase in restocking since 2010, an increase in restocking-based production is expected to be starting right around now. Gradually, restocking has replaced natural recruitment (assisted and fully natural), now making up over 90 % of the inland stock. Fisheries have taken 20-30 % of the silver eel (since the mid-1980s), while the impact of hydropower has ranged from 25 % to 60 %, depending on the year. Escapement is estimated to have varied from 72 t in the late 1990s, to 175 t in the early 2000s. The biomass of current escapement (including eels of restocked origin) is approximately 15 % of the pristine level (incl. restocked), or almost 30 % of the current potential biomass (incl. restocked). This is below the 40 % biomass limit of the Eel Regulation, and anthropogenic mortality (70 % over the entire life span in continental waters) exceeds the limit implied in the Eel Regulation (60 % mortality, the complement of 40 % survival). Mortality being that high, Swedish inland waters currently do not contribute to the recovery of the stock.

The temporal variation (in production, impacts and escapement) is partly the consequence of a differential spatial distribution of the restocking of eel over the years. The original natural (not assisted) recruits were far less impacted by hydropower, since they could not climb the hydropower dams when immigrating. Since 2010, inland restocking is increasingly concentrated to drainage areas falling to the Kattegat-Skagerrak, also including obstructed lakes (primarily Lake Vänern, and many smaller ones). Even though Trap & Transport of silver eel - from above barriers towards the sea - has contributed to reducing the hydropower impact, hydropower mortality remains the largest estimated contributor to silver eel mortality in inland waters. Without restocking, the biomass affected by fishery and/or hydropower would be only 5-10 % of the currently impacted biomass, but the stock abundance would reduce from 15 % to less than 3 % of the pristine biomass.

In summary: the inland eel stock biomass is below the minimum target, anthropogenic impacts exceed the minimum limit that would allow recovery, and those impacts have been increasing. It is therefore recommended to reconsider the current action plans on inland waters, taking into account the results of the current, comprehensive assessment.

For the Baltic coast, the 2021 assessment has been updated without major changes in methodology. Results indicate that the impact of the fishery continues to decline over the decades. The current impact of the Swedish silver eel fishery on the escapement of silver eel along the Baltic Sea coast is estimated at 0.3 %. However, this fishery is just one of the anthropogenic impacts (in

other areas/countries) affecting the eel stock in the Baltic, including all types of impacts, on all life stages and all habitats anywhere in the Baltic. Integration with the assessments in other countries has not been achieved. Current estimates of the abundance of silver eel (biomass) indicates an order of several thousand tonnes, but those estimates are extremely uncertain, due to the low impact of the fishery (near-zero statistics). Moreover, these do not take into account the origin of those silver eels, from other countries. An integrated assessment for the whole Baltic will be required to ground-truth these estimates. This would also bring the eel assessments in line with the policy to regionalise stock assessments for other (commercial) fish species (see https://ec.europa.eu/oceans-and-fisheries/fisheries/rules/multiannual-plans_en).

It is recommended to develop an integrated assessment for the entire Baltic Sea eel stock, and to coordinate protective measures with other range states.

Sammanfattning

Den europeiska ålens beståndsstorlek är starkt minskande. EU beslutade 2007 om en förordning med åtgärder för att återställa ålbeståndet i Europa. Förordningen kräver att medlemsstaterna till 2009 skulle ta fram och verkställa sina respektive nationella ålförvaltningsplaner. Sverige lämnade in sin plan hösten 2008. Enligt förordningen skall medlemsstaterna vart tredje år rapportera till Kommissionen vad som gjorts inom ramen för planen och erhållna resultat vad gäller skydd och återuppbyggnad av ålbeståndet. I föreliggande rapport presenteras en analys och uppskattning av ålbeståndet i Sverige som det såg ut våren 2024; detta med syfte att tjäna som underlag till den svenska uppföljningsrapporten till EU. Rapporten uppdaterar och utvidgar därmed tidigare års utvärderingar (Dekker 2012, 2015; Dekker et al. 2018, 2021).

Rapporten utvärderar påverkan från fiske, utsättning och kraftverksrelaterad dödlighet på ålbeståndet. Annan antropogen påverkan som klimatförändring, förorening, ökad påverkan från predatorer, parasitspridning och en eventuell störd vandring hos omflyttade ålar osv., har sannolikt också en effekt på beståndet. Sådana faktorer kan knappast kvantifieras och det finns inte heller några relaterade förvaltningsmål uppsatta. Av dessa orsaker, samt det faktum att Ålförordningen inte heller beaktar sådana faktorer, så inkluderas de inte heller i denna tekniska utvärdering. Vi fokuserar här på kvantifieringen av den, från kontinentala vatten mot havet, utvandrande blankålens biomassa (faktisk, potentiell och jungfrulig) och på den dödlighet ålarna utsätts för under sin livstid. Uppskattningen bryts ned på regional nivå, med olika typ av dominerande påverkan i olika områden (västkust, inland, ostkust).

Under de senaste åren har den sedan länge nedåtgående trenden i antalet rekryterade glasålar brutits och det över hela Europa. Om det är en effekt av de åtgärder som gjorts, eller om det finns andra bakomliggande orsaker, är fortfarande oklart. Denna rapport bidrar till den internationella bedömning som krävs, men den diskuterar inte den senaste rekryteringstrenden och ålbeståndets allmänna tillstånd i Europa.

Resultaten för de olika områdena summeras enligt följande:

Gulålen på västkusten exploaterades tidigare genom ett intensivt ryssjefiske. Det fisket är sedan våren 2012 helt stängt. Även om en viss uppföljning fortsätter genom ryssjefiske, så är den tillgängliga informationen inte tillräcklig för en beståndsuppskattning. Uppenbarligen så är fiskeridödligheten nu noll, men vi kan inte presentera några av de andra efterfrågade beståndsindikationerna (faktisk, potentiell och jungfrulig biomassa). De fiskeri-oberoende fiskeundersökningarna som görs visar emellertid att de tidigare utnyttjade storleksklasserna av

beståndet verkligen återhämtar sig, och de mindre storleksklasserna visar ett brott i sin nedgång i linje med beståndets allmänna trend över hela distributionsområdet.

Som en åtgärd för att bygga upp ålbeståndet eller för att kompensera för antropogen dödlighet på annat håll, så har unga ålar satts ut på västkusten sedan 2010. Med tanke på mängden utsatt ål, är den förväntade effekten (ca 50 ton blankål) relativt ringa och svår att verifiera – jämfört med det potentiella naturliga beståndet på västkusten efter återhämtning (i storleksordningen 1000 ton). Men för det nu utarmade beståndet kommer dock utsättningarna ha större effekt och kan bidra till utvandringen till Sargassohavet.

För inlandsvattnen så redovisar rapporten en uppdatering av 2021 års beståndsuppskattning, de största förändringarna i metodiken är användningen av en ny rekryteringsmodell, och den fullständiga separeringen av Trap & Transport från fiskeristatistiken. Beståndsuppskattningen för inlandsvattnen bygger på en rekonstruktion av beståndet utifrån information om de yngsta stadierna av rekryterande ål i våra vatten (naturliga rekryter, yngeltransport, utsättning). Baserat på 78 års data över naturlig rekrytering till 22 vattendrag, har en statistisk modell tagits fram. Den relaterar antalet uppvandrande unga ålar fångade i ålyngelsamlare till geografisk lokalisering och storlek av varje vattendrag, avstånd från mynning till ålyngelsamlare, och till vilket år dessa ålar rekryterades till kontinentala vatten som glasål, dvs. årsklass. Längre in i Östersjön är uppvandrande ålar större men färre. Längre avstånd från mynningen medför färre ålar

Genom att använda resultaten från rekryteringsanalysen ovan, i kombination med historiska data över yngeltransporter (”assisted migration”, unga ålar som med människans hjälp transporterats upp över vandringshinder) och utsatta mängder importerade ålyngel, så har vi en fullständig översikt över hur många unga ålar som rekryteras till svenska inlandsvatten. Från detta har produktionen av blankål från alla sjöar och år uppskattats. Genom att sedan dra bort mängden fångad ål (utifrån rapporterade landningar) och de som dött vid kraftverkspassager (procentuell, utifrån rapporterad andel eller standardandel), har mängden överlevande lekvandrare (lekflykt) uppskattats.

Resultaten visar att sedan 1960, har produktionen av blankål minskat från mer än 700 ton till mindre än 300 ton per år, och produktionen minskar fortfarande. Den naturliga rekryteringen av ål, uppflyttad eller fullt naturlig, har gradvis till 90 % ersatts genom utsättning av importerade ålyngel. Fisket har tagit 20-30 % av blankålen sedan 1980-talet, medan påverkan (dödlighet) från vattenkraft har varierat från 25 % till 60 %. Utvandringen av blankål till havet har varierat från 72 ton under sent 1990-tal till 175 ton under tidigt 2000-tal. Biomassan av utvandrande blankål (inklusive de av utsatt ursprung) uppskattas idag vara ungefär 15 % av den jungfruliga mängden (inkl. utsatt), eller nästan 30 % av dagens potential (inkl. utsatt). Biomassan ligger därmed under den 40 %-gräns som Ålförordningen föreskriver, och den mänskligt introducerade dödligheten (70 %) överskrider den avgörande gränsen (60 % dödlighet, motsvarande 40 % överlevnad). Med en så hög dödlighet, så bidrar svenska inlandsvatten för närvarande inte till en återhämtning av beståndet.

Variationen i produktion, påverkansfaktorer och lekflykt över tid är delvis en konsekvens av att utsättningarna av ålyngel förskjutits geografiskt över tid. De ursprungliga naturliga, dvs. inte uppflyttade, rekryterna var mycket mindre påverkade av vattenkraften, då de normalt inte kan vandra uppströms kraftverksdammar. Sedan 2010 görs utsättningarna främst i avrinningsområden som mynnar på västkusten, och därmed delvis i sjöar med hinder för nedströmsvandring (främst i Vänern, men också i flera mindre sjöar). Även om Trap & Transport av blankål – från ovan vattenkraftshinder mot havet – har bidragit till att minska vattenkraftens påverkan, är vattenkraftsdödligheten fortfarande den största beräknade bidragsgivaren till mänsklig blankålsdödlighet i inlandsvattnen. Utan ålutsättning skulle biomassan av ål påverkad av fiske och vattenkraft bara vara 5-10 % av dagens påverkade biomassa. Samtidigt skulle ålbeståndet bara vara 3 % av den jungfruliga biomassan, att jämföra med dagens 15 %.

Sammanfattningsvis: biomassan av inlandsvattnens ålbestånd uppnår inte nödvändig miniminivån, den mänskliga påverkan överskrider den lägsta gränsen för återhämtning, och de negativa effekterna har ökat. Det rekommenderas därför att nuvarande förvaltningsplan för ål i sötvatten omprövas, detta för att beakta den mer allsidiga beståndsuppskattningen i föreliggande arbete.

För ostkusten har 2021 års beståndsuppskattning uppdaterats utan förändringar i metodiken. Resultaten indikerar att fiskets inverkan snabbt minskar över tid. Dagens påverkan från det svenska blankålsfisket vid ostkusten beräknas nu till 0.3 %. Fisket är emellertid bara en av de mänskliga faktorer (i andra områden och länder) som påverkar Östersjöbeståndet av ål. Någon integrerad beståndsuppskattning i staterna runt Östersjön har inte kommit till stånd. Nuvarande uppskattning av ålbiomassan (blankål) i Östersjön är i storleksordningen några tusen ton, men denna skattning tar inte hänsyn till ursprunget av blankålar från andra länder. En integrerad, enhetlig beståndsuppskattning för hela Östersjön behövs för att verifiera denna skattning. Detta skulle ligga i linje med regionaliseringsarbetet för beståndsskattning avseende andra kommersiella målarter (de arter som fisket avser att fånga; se t.ex. https://ec.europa.eu/oceans-and-fisheries/fisheries/rules/multiannual-plans_en).

Vi rekommenderar således en integrerad beståndsuppskattning för hela Östersjöbeståndet av ål och att skyddsåtgärder samordnas mellan berörda stater.

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1. Introduction

1.1 Context

The population[1] of the European eel (*Anguilla anguilla*) has severely declined from its historic abundance levels: fishing yield has declined over the past century to below 10% of former levels, and between the 1970s and 2010s recruitment has rapidly declined to 1-10% of its former level (Dekker 2004, 2016; ICES 2023). In 2007, the European Union implemented a Regulation establishing measures for the recovery of the stock of European eel (Anonymous 2007; Council Regulation 1100/2007, hereafter referred to as the EU Eel Regulation), obliging EU Member States to develop national Eel Management Plans (EMPs) by 2009. In December 2008, Sweden submitted its EMP (Anonymous 2008), in which the entirety of Sweden is defined as a single Eel Management Unit. Subsequently, protective actions have been implemented (in Sweden and all other EU countries), and progress has been evaluated internationally in 2014 (Anonymous 2014), 2019 (Anonymous 2019), and 2022 (ICES 2022). On a national level, the state of the European eel stock in Sweden has been assessed by a series of post-evaluations, starting in spring 2012 (Dekker 2012), and every three years thereafter (Dekker 2015, Dekker et al 2018, Dekker et al 2021). This current report, in 2024, updates, extends and reviews those national reports.

1.2 Aim of this report

The EU Eel Regulation sets limits for the fishery, and for the impact of hydropower generation. Other important factors that might affect the eel stock include climate change, pollution, spread of diseases and parasites, impact of predators (anthropogenically-enhanced) and the potential disruption of migratory behaviour by transport of eels (for restocking, or by Trap & Transport). For these factors,

[1] In this report, we use the word “population” for the whole group of European eels that do or have a potential to interbreed. So far, evidence indicates that potentially all European eels across the whole distribution area of the species constitute a single population. The word “stock” is used more loosely, to indicate a group of eels in any defined area.

European policies that pre-date the EU Eel Regulation are in place, such as the Habitats Directive, the Water Framework Directive and the Common Fisheries Policy. These other policies were assumed to achieve an adequate (or the best achievable) effect for these other impacts; the EU Eel Regulation specifies no additional measures. Since this report is focused on an assessment of the eel stock in relation to the implementation of the EU Eel Regulation, the other anthropogenic impacts – listed above - will remain outside the discussion. This is in line with the approach in the Swedish Eel Management Plan, which does not plan specific actions on these factors. This should not be read as an indication that these other factors might be less relevant. However, the impacts of most of these other factors on the eel stock have hardly been quantified, and as far as they have been, they can as yet not be assessed on a regular basis. Blending in unquantified aspects into a quantitative analysis jeopardises the assessment, risking a failure to identify a possibly inadequate management of the quantified factors (fishing and hydropower mortality).

According to the EU Regulation, Member States shall report to the Commission on progress in implementing their national Eel Management Plans, and on the status of the eel stock and its protection status, every third year starting in 2012, and from 2018 onwards every sixth year. The idea behind this time schedule was, that – by 2018 – implementation would be well on track, and a lower reporting frequency would be able to document the (slow) recovery of the stock. In reality, the implementation of national Eel Management Plans does not progress that fast, and monitoring and evaluation of their effectiveness falters (Dekker 2016; Anonymous 2019). In a Joint Declaration of December 2017, the EU-Commission and Member States agreed upon the continuation of the tri-annual reporting cycle. This year (2024) is the second reporting year under this agreement, after 2021.

This report analyses the status of the Swedish eel stock and recent trends in anthropogenic impacts and their relation to the limits set in the EU Regulation and the Swedish Eel Management Plan. The intention is to assist the national reporting to the Commission. To this end, stock indicators are calculated, fitting the international reporting requirements. Prime focus will be on estimating trends in the biomass of silver eel escaping (Bcurrent, Bbest and B0) and the mortality they endured over their lifetime (ΣA); see below.

The presentation in this report will be technical in nature, and will be focused on the status and dynamics of the stock. Management measures taken, their implementation and proximate effects are not directly discussed; their net effect on the stock, however, will show up in the assessments presented in this report. Earlier, Dekker et al. (2016) analysed the effects of different management measures, in a series of scenario studies.

1.3 Structure of this report

The main body of this report is focused on the evaluation of the current stock status and protection level. To this end, assessments have been made for different areas, each of which is documented in a separate Appendix. The main report summarises the results at the national level, presents the stock indicators in the form required for international post-evaluation, and discusses general issues in the assessments. The Appendices are structed as follows:

- Appendix A presents data from the west coast.
- Appendix B presents the riverine recruitment time series and analysis of spatial and temporal trends.
- Appendix C reconstructs the inland stock from databases of historical abundance of young eels.
- Appendix D updates the Baltic coast assessment of Dekker and Sjöberg (2013), adding mark-recapture data from silver eel along the Baltic coast for the years 2012-2023.

1.4 The Swedish eel stock and fisheries

The eel stock in Sweden predominantly occurs from the Norwegian border in the Skagerrak on the west side, all along the coastline, north to about Hälsingland (61°N) in the Baltic Sea, and in most lakes and rivers draining there. Further north, the density declines to very low levels, and these northern areas are therefore excluded from most of the discussions here. In the early 20th century, there were noticeable eel fisheries also in the northernmost parts of the Baltic Sea (e.g. Olofsson 1934), but none of that remains nowadays. The current habitats and fisheries are briefly described below.

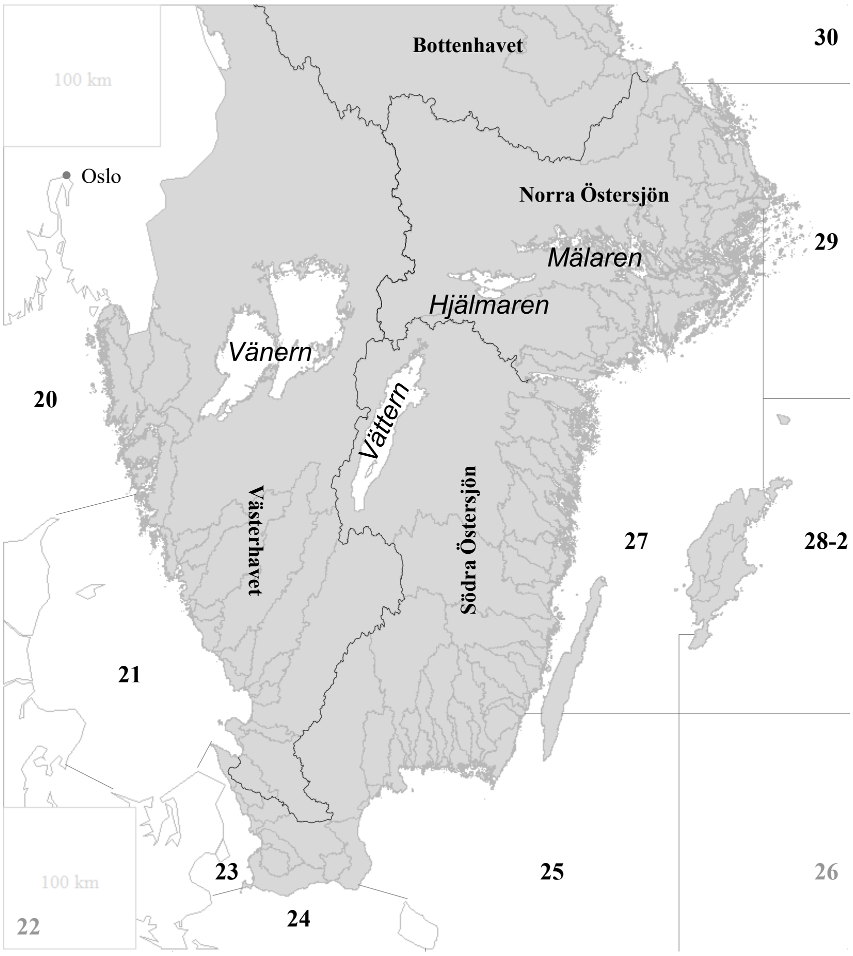


Figure 1: Map of the study area: the southern half of Sweden (north up to 61°N). The names in italics indicate the four largest lakes; the names in bold indicate the Water Basin Districts related to the Water Framework Directive (not used in this report); the numbers refer to the ICES subdivisions; the medium grey lines show the divides between the main river basins.

The West Coast from the Norwegian border to Öresund, i.e. 320 km coastline in Skagerrak and Kattegat. Along this open coast there was a fishery for yellow eels, mostly using fyke nets (single or double), but also baited pots during certain periods of the year. The west coast fishery has been closed as of spring 2012.
The coastal parts of ICES subdivisions 20 & 21 (Figure 1).

Öresund, the 110 km long Strait between Sweden and Denmark. In this open area, both yellow and silver eels are caught using fyke nets and some large pound nets. The northern part of Öresund is the last place where silver eels originating from the Baltic Sea are caught on the coast, before they disappear into the open seas.
The coastal parts of ICES subdivision 23 (Figure 1).

The South Coast from Öresund to about Karlskrona, i.e. a 315 km long coastal stretch of which more than 50 % is an open and exposed coast. Silver eels are caught in a traditional fishery using large pound nets along the beach. The coastal parts of ICES subdivision 24, and most of subdivision 25, up to Karlskrona (Figure 1).

The East Coast further north, from Karlskrona to Stockholm. Along this 450 km long coastline, silver eel (and some yellow eel) are fished using fyke nets and large pound nets. North of Stockholm, abundance and catches decline rapidly towards the north.
The coastal parts of ICES subdivisions 25 (from Karlskrona), 27, 29 and 30 (Figure 1).

Inland waters. Eels are found in most accessible waters, except in the high mountains and most of the northern parts of the country. Pound nets are used to commercially fish for eel in the biggest lakes Mälaren, Vänern and Hjälmaren, and in some smaller lakes in southern Sweden. Recreational fishing is allowed upstream of three consecutive hydropower barriers. In inland lakes, restocking of young eels has contributed to current day's production, while barriers and dams have obstructed the natural immigration of young eels. Traditional eel weirs (lanefiske) and eel traps (ålfällor) were operated at many places, and some are still being used. Hydropower generation impacts the emigrating silver eel from many lakes.

1.5 Spatial assessment units

According to the Swedish Eel Management Plan, all of the Swedish national territory constitutes a single management unit. Management actions and most of the anthropogenic impacts, however, differ between geographical areas: inland waters and coastal areas are contrasted, and so are the west coast and Baltic coast. Anthropogenic impacts include barriers for immigrating natural recruits, restocking recruits, yellow and silver eel fisheries, hydropower related mortality, Trap & Transport of young recruits and of maturing silver eels; and so forth.

The assessment in this report will be broken down along geographical lines, also taking into account the differences in impacts. This results in three blocks, with little interaction in-between. These blocks are:

1. West coast – natural recruitment and restocking, former fishery on yellow eel.
2. Inland waters – natural recruitment, assisted migration, and restocking, fishery on yellow and silver eel, impact of migration barriers (on immigrating youngsters), hydropower generation (on emigrating silver eel), and Trap & Transport. The limit between inland and coastal waters is drawn at the lowest migration barrier in each river (see further discussion in section C.1.1).
3. Baltic coast – natural recruitment and restocking, fishery on silver eel.

For each of these areas, stock indicators will be derived.

Symbols & notation used in this stock assessment

The assessments in this report derive the following stock indictors:

$B_{current}$	The biomass of silver eel escaping to the ocean to spawn, under the current anthropogenic impacts and current low recruitment.
B_{best}	The biomass of silver eel that might escape, if all anthropogenic impacts would be absent at current low recruitment.
B_0	The biomass of silver eel at natural recruitment and no anthropogenic impacts (pristine state).
A	Anthropogenic mortality (per year/age). This includes fishing mortality F, and hydropower mortality H; A=F+H.
ΣA	Total anthropogenic mortality rate, summed over the whole life span.
%SPR	Percent spawner per recruit, that is: current silver eel escapement $B_{current}$ as a percentage of current potential escapement B_{best}. %SPR can be derived either from $B_{current}$ and B_{best}, or preferably from ΣA (%SPR = $100*\exp^{-\Sigma A}$).
%SSB	Current silver eel escapement $B_{current}$ as a percentage of the pristine state B_0.

All of the above symbols may occur in three different versions. If a contribution based on restocking is explicitly included, the symbol will be expanded with a + sign ($B_{current}^+$, B_{best}^+, B_0^+, $\sum A^+$, etc.); if it is explicitly excluded, the symbol will be expanded by a – sign ($B_{current}^-$, B_{best}^-, B_0^-, $\sum A^-$, etc.); when the difference between natural and restocked immigrants is not relevant, the addition may be omitted.

1.6 Management objectives and reference points

In this section, we present a framework of quantitative reference points, to which the indicators of the current state of the eel stock can be evaluated. This will allow for the evaluation of the national (and international) Eel Management Plan(s), and inform the policy-makers of the effectiveness of their protective measures. To start with, we review the objectives and reference points applied in relevant policy documents (national and international), and in previous scientific evaluations. We then select the most informative and relevant framework, and develop that further for the current needs.

1.6.1 Management objectives

The EU Eel Regulation (Anonymous 2007) sets a long-term objective ("*the protection and sustainable use of the stock of European eel*"), delegating implementation of protective measures to its Member States (Dekker 2009, 2016). The Swedish Eel Management Plan subscribes to these objectives and emphasises stopping the decline rapidly (Anonymous 2008, section 5.1, "*we choose to dimension the measures so that they – provided similar measures are introduced over the whole area of distribution – the present recruitment decline is stopped or turned to an increase*").

1.6.2 Reference points for sustainable use and protection

To operationalise the aim to protect and recover the stock, suggested a concrete goal: rebuilding recruitment to levels "*similar to those of the 1980s* [meant is: pre-1980]." To achieve that aim, it will be essential to ensure at least a minimum spawning stock size. It is generally considered that – at low spawning stock size – the number of spawning adults can be restrictive for the production of a new year class of young fish: the stock-recruitment relationship. Although "*the ecology of the eel makes it difficult to demonstrate a stock-recruitment relationship, […] the precautionary approach requires that such a relationship should be assumed to exist for the eel until demonstrated otherwise*" (ICES 2002), and hence, a minimum level for the oceanic spawning stock must be maintained. "*In order to rebuild that oceanic spawning stock, measures should aim for increased escapement of spawners from continental waters*" (ICES 2001). Stock-wide estimates of spawning stock and recruitment for the European eel are not available and are very unlikely to be acquirable at all. Consequently, stock-wide management targets need to be translated into derived targets for local management. For this, ICES (2002) advised "*Exploitation, which provides 30% of the virgin (F=0) spawning stock biomass is generally considered to be such a reasonable provisional reference target. However, for eel a preliminary value could be 50%.*" The Eel Regulation adopted this approach, compromised between the suggested 30% and 50%, and set the objective for national Eel Management Plans as "*to reduce anthropogenic mortalities so as to permit […] the escapement […] of at least 40 % of the silver eel biomass* [relative to the notional pristine escapement]" (Art. 2.4). The long-term aim of the Eel Regulation (an escapement of 40% of the pristine escapement) will ultimately correspond to a limit lifetime anthropogenic mortality of $\Sigma A = -\ln(0.40) = 0.92$ (Dekker 2010; ICES 2010).

1.6.3 Reference points for recovery

Even though reducing anthropogenic mortalities to this minimal protection level (a maximal lifetime mortality of $\Sigma A=0.92$) may be expected to stabilise the stock, it will not be enough to recover from the current, severely depleted state. For recovery, a further reduction in mortality will be required, across the full range of the population. The further mortality is reduced, the faster the recovery can take place. Even if all anthropogenic impacts would be lowered to zero, however, full recovery is not expected within decades or centuries (Åström & Dekker 2007). In practice, some human impacts on the eel stock will be difficult to bring to zero (depending on e.g. poaching), and other impacts may be accepted because of their importance for other policies (e.g. water management systems, renewable energy production from hydropower, cultural fishing rights). Anthropogenic mortalities are therefore most unlikely to drop to zero completely – and hence, a long period of recovery is foreseen. The EU Eel Regulation (Anonymous 2007) does not specify a time frame for recovery (art. 2.4: "*the purpose [is] achieving this objective in the long term*"), and neither does the Swedish Eel Management Plan (Anonymous 2008) indicate what rate of increase is aimed for. Any mortality between $\Sigma A=0$ (maximum aspiration level, but still a slow recovery) and $\Sigma A=0.92$ (minimal aspiration level, stabilisation but no recovery) will be in line with these policies. Even though the objective clearly is to protect and recover the stock, no operational aspiration level has been specified.

1.6.4 Reference points used in the international advice by ICES

The international advice by ICES (2023) for the European eel reads "*there should be zero catches in all habitats in 2024. […] All non-fisheries related anthropogenic mortalities should be zero. The quantity and quality of eel habitats should be restored*". This advice is based on eel being listed as a Category 3 stock within the ICES framework, meaning that insufficient data is available for a full assessment and survey trends are used instead. Given that the current recruitment index is far below what would be considered the limit reference point, the automatic advice from ICES is to stop all anthropogenic impacts on the stock.

Aiming for no anthropogenic mortality and the most rapid (but still slow) recovery, this advice adopts an aspiration level above that of the Eel Regulation and the Swedish Eel Management Plan. Additionally, this advice does not facilitate the evaluation of the implementation of current protection measures. It does not allow the evaluation of the current mortalities and protection level against the objective to protect and recover. For eel, the ICES framework evaluates only the state and not the impacts.

1.6.5 Previously used reference framework

For stocks below, but still close to safe biological limits, "ICES applies a proportional reduction in mortality reference values (i.e. a linear relation between the mortality rate advised and biomass)" (FAO & ICES 2011). Though the stock is clearly far below safe biological limits, this proportional reduction in mortality reference values has been used for the evaluation of the implementation of the Eel Regulation by ICES-WGEEL (ICES 2013, 2016, 2018) and in preceding assessments of the Swedish eel stock (Dekker 2012, 2015 and Dekker et al. 2018). Even though this established a coherent reference framework for the evaluation, the proportional reduction (i.e. the proportionality of it) has been criticised for being arbitrary and leading to longer recovery times the lower the stock status is (Dekker 2019).

1.6.6 Current choice of reference framework

The Eel Regulation and the Swedish Eel Management Plan define a minimal condition for protection and express the objective to recover, but they quantify no aspiration level for setting speed to that recovery. ICES advice formulates a maximal aspiration level for protection, outside the feasible range, clearly aiming for a maximum effort – which does not allow us to evaluate the current situation against the adopted Eel Management Plan. The "proportional reduction" framework - used in several previous assessment reports - has been shown to be arbitrary and not fully consistent. Given the impossible choice between these three imperfect approaches, a different approach was taken in the previous (Dekker et al. 2021) which we continue to follow here, as follows.

According to the FAO Technical Guidelines for Responsible Fisheries, policy makers are expected to "*Establish a recovery plan that will* ***rebuild the stock over a specific time period*** *with reasonable certainty*" (FAO 1996, point 48.b, formatting added). When a rebuilding target has been specified, and an appropriate time period has been selected, a corresponding level of anthropogenic mortality can be deduced (using a scientific model of stock dynamics and anthropogenic impacts). While the ultimate rebuilding target gives no guidance for taking momentaneous actions (it describes an ultimate goal, far into the future; Dekker

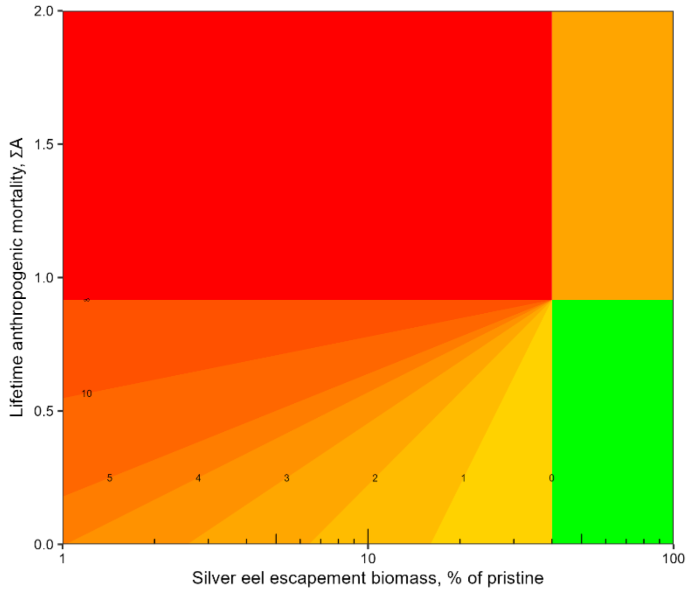
2016), the corresponding anthropogenic mortality level directly translates into contemporary protective actions (which can be implemented and evaluated immediately). Hence, stock management is generally evaluated in two dimensions: the stock status itself in relation to the ultimate target (in biomass, horizontal), and the momentaneous impacts (as mortality rate, vertical) – as in the Precautionary Diagram (Figure 2). This then allows evaluating current management, by comparing the actual mortality level to the mortality level needed for recovery within the specified time period. For the eel, Dekker (2019) noted that a time period specified in 'number of years' hardly allows the deduction of an acceptable mortality level (because of lack of full insight in eel stock dynamics across the whole population). A time period expressed as 'number of generations until recovery', however, translates logically and straightforwardly into an acceptable mortality level. In summary: given an ultimate rebuilding target and a specified aspiration level (formulated as a specific time period or number of generations until recovery), a corresponding mortality level can be calculated. Current management is then evaluated, depending on whether the actual mortality is above or below that reference mortality level. Based on this line of reasoning, ICES (2019) pleaded for the adoption of a time-period (as number of generations) by the relevant policy makers. However, no such time-period has been adopted yet.

Here, we reverse the above line of reasoning: in the absence of a specified period until full recovery (i.e. fully achieving the EU aim to restore 40% of the pristine spawning stock biomass), we cannot derive a corresponding limit mortality – but given any actual mortality, we can calculate the corresponding expected period until full recovery. For every possible combination of stock status and mortality, it is possible to derive the number of generations needed for full recovery (Figure 2, the shades of orange in the lower-left quadrant reflect the number of generations needed until full recovery). No information is needed on actual generation time for this, given that anthropogenic mortality is reported as lifetime mortality ΣA (ICES 2021), but for reference the average generation time in Swedish waters is around 17 years. Whether that number of generations until recovery, and hence any actual level of anthropogenic mortality, is considered acceptable or not, is left open by us. Instead, in line with the principle of role-separation between science and policy-making, that decision on acceptability and aspiration level is left to the policy-makers. This approach has the additional advantage, that we do not suggest there is a sharp boundary between acceptable (recovery within the specified time period) and unacceptable – which there is not. The shades of orange represent a continuous range of feasible aspiration levels.

We note that this reference framework might be perceived as a bit theoretical. The quantification of the aspiration level in terms of numbers of generations would preferably be replaced by one in numbers of years. Additionally, the aspiration level should not be misunderstood to mean that the stock will truly recover within the specified time – other factors (other impacts, climatic factors), as well as the (lack of) protective measures in other areas/countries, might intervene. Rather than an accurate prediction, this framework should be seen as a uniform way to quantify an otherwise intangible issue such as aspiration, enabling the comparison between regions/countries, which potentially even can lead to effective post-evaluation of the chosen aspiration.

Red: stock biomass below the management target; mortality above the corresponding limit, i.e. bad status and deteriorating

Shades of orange: stock biomass below the management target; mortality below the corresponding limit, i.e. bad status, but allowing recovery. Numbers indicate the number of generations needed until recovery (selected numbers shown).



Orange: stock biomass above the management target; mortality above the corresponding limit, i.e. status still ok, but deteriorating – unsustainable.

Green: stock biomass above the management target; mortality below the corresponding limit, i.e. sustainable.

Figure 2: Precautionary Diagram, presenting the status of the stock (horizontal) and the level of anthropogenic impacts (vertical). Note the logarithmic scale of the horizontal axis, corresponding to the inherently logarithmic nature of the vertical axis. Background colours explained in call-outs. The numbers on the borders between the shades of orange, in the lower-left quadrant, indicate the number of generations needed until full recovery to the management target (40%). (After Dekker 2019, strongly modified).

1.7 Spatial coverage, whole stock versus management units

The discussion of a reference framework, given above, predominantly focused on the whole stock of the European eel, distributed all over Europe and the Mediterranean. While the actual recovery of the stock likely depends on the protection across the whole distribution area, an effective evaluation of the stock abundance and its protection status is currently only achievable at the level of individual Eel Management Units (Dekker 2016). The discussion above, including the evaluation of mortality levels and recovery times, and the Precautionary Diagram (Figure 2), however, can equally well be applied to the whole population, as to any (collection of) sub-stocks or spatial management units.

The actual recovery (and the number of generations until full recovery) crucially depends on the overall status of the whole population. The aim of the current evaluation is focused on the Swedish part of the stock only, and our results apply to the Swedish assessment only. Consequently, any indication of an expected or predicted number of generations (and consequently, any evaluation of the protection status) will only be valid, if the anthropogenic mortality and the protection status evaluated here for Sweden, would apply equally across the whole stock – which they do not. Because of that, the "number of generations until recovery" should not be seen as a realistic prediction of the time needed for the recovery of the stock, but as a coherent way to quantify the shared aspiration to recover the stock within reasonable time, and an individual country's contribution to that. The Swedish Eel Management Plan (Anonymous 2008, section 5.1) is aware of the contrasts of scales, formulating "*we choose to dimension the measures so that they – provided similar measures are introduced over the whole area of distribution – the present recruitment decline is stopped or turned to an increase*". The condition "*provided similar measures are introduced over the whole area of distribution*" thus applies to the evaluations in this current report as well.

1.8 Fisheries and non-fishing anthropogenic impacts

For anthropogenic impacts other than fisheries and hydropower-related impacts (i.e. for pollution, spread of parasites, potential disruption of migration by transport, increased predation pressure, and so forth), no targets have been set in the national Eel Management Plan or the European Regulation, and no quantitative assessment is currently achievable. Hence, the current report discusses these impacts only marginally. This should not be misread as an indication that we consider them of less importance.

2. Recruitment indices

Trapping of elvers[2] below barriers in rivers (for transport and release above the barriers, a process known as 'assisted migration') provides information on the quantities of young eels entering the rivers where a trap is installed (Erichsen 1976; Wickström 2002). Figure 3 shows the raw observations; Appendix B presents an in-depth analysis of temporal and spatial trends in these data. The results align with the international trend (ICES 2023) that - after decades of decline - the recruitment has stopped decreasing after 2011, but the trend after 2011 is rather unclear (few data points) and erratic (high variation), with recruitment remains around historically low levels.

The nuclear power plant at Ringhals takes in cooling water from the coast of the Kattegat, drawing in glass eel too. This is one of the rare cases where true, unpigmented glass eel is observed in Sweden. An Isaacs-Kidd Midwater trawl (IKMWT) is fixed in the current of incoming cooling water, fishing passively during entire nights (Figure 4). Results indicate a steady decline in glass eel numbers per night from 1980 (beginning of the series) to 2010, and a stabilisation around a historically low level thereafter.

A modified Methot-Isaacs-Kidd Midwater trawl (MIKT) is used during the International Bottom Trawl Survey (IBTS Quarter 1, called the ICES-International Young Fish Survey before 1993; Hagstrom & Wickström 1990). No glass eels were caught in 2008, 2009, 2010 and 2021. In 2011, there was no sampling due to technical problems. Similar to the Ringhals index, the IBTS results indicate a steady decline from 1990 (beginning of the series) to 2010, and a stabilisation thereafter (Figure 5).

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[2] Terminology: In this report, the words glass eel, elver and bootlace eel are used to indicate the young eel immigrating from the sea to our waters. Glass eel is the youngest, unpigmented eel, that immigrates from the sea; true glass eel is very rare in Sweden. At the international level, the term 'elver' usually indicates the youngest pigmented eels; whether it also includes the unpigmented glass eel depends on the speaker (a.o. English versus American). Bootlace eel is a few years older, the size of a bootlace. The Swedish word 'yngel' includes both the elver and the bootlace, by times even the glass eel. In some Swedish rivers, the immigrating eel can be as large as 40 cm.

In this report, we make a distinction between truly unpigmented glass eel (by definition: at age zero) and any other immigrating eel (continental age from just over zero to approx. seven years). The latter category comprises the pigmented elver, the bootlace, but also the larger immigrating eel having a length of 40 cm or more. To avoid unnecessarily long wording, all pigmented recruits will collectively be indicated as "elvers", or the size/age of the eel will be clearly specified.

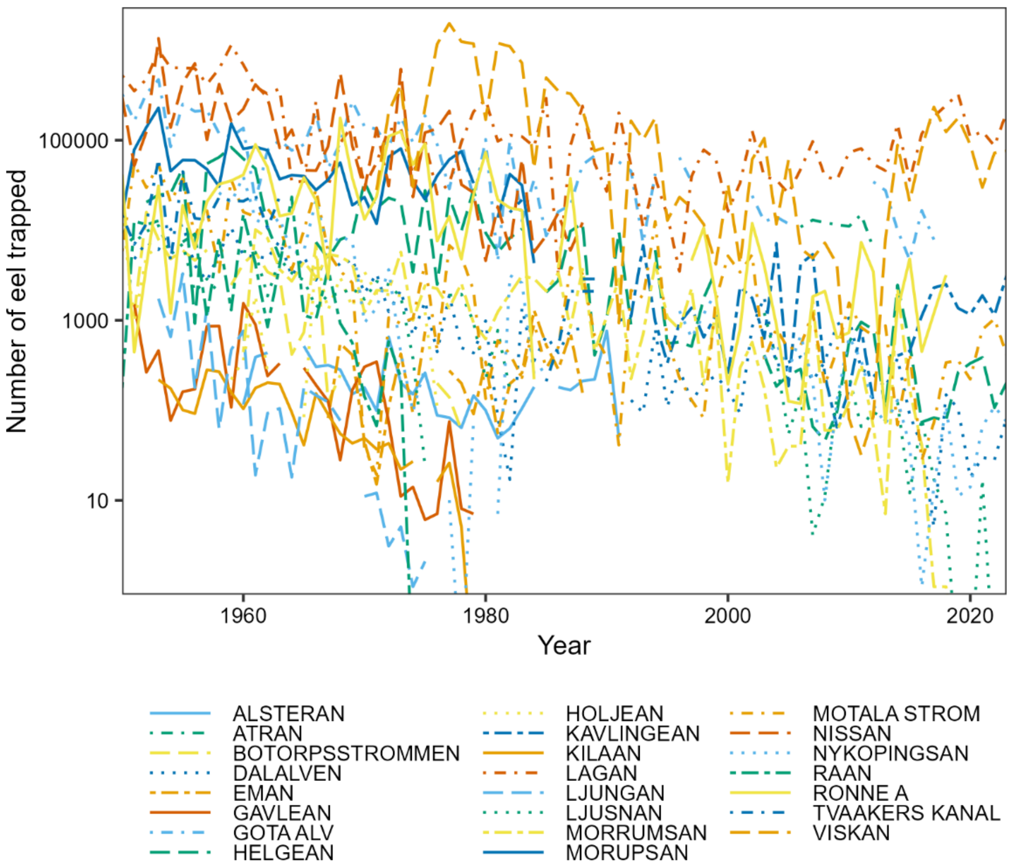

Figure 3: Trends in the number of elvers trapped at barriers, in numbers per year. Note the logarithmic character of the vertical axis. For further details, see Appendix B.

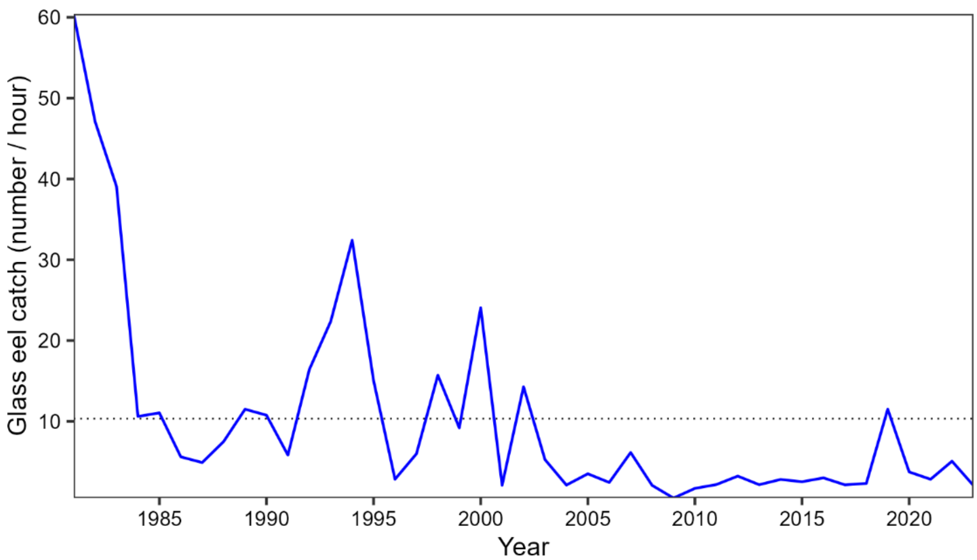

Figure 4: Time trend of glass eel catches per trawling hour in a survey at the Ringhals nuclear power plant on the Swedish Kattegat Coast. The horizontal dotted line represents the average for the displayed years.

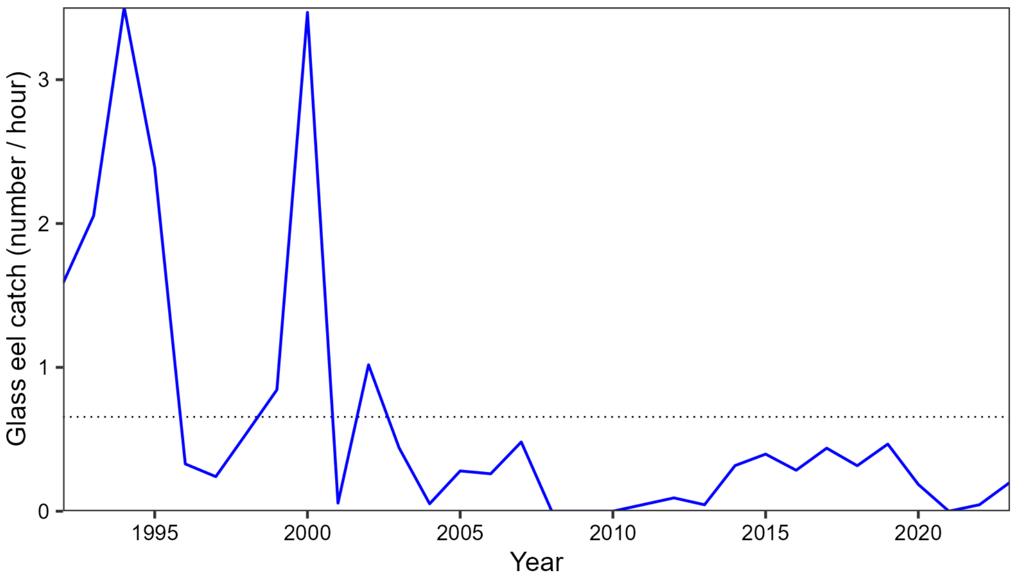

Figure 5: Time trend of glass eel catches per trawling hour by the IBTS survey in the Skagerrak-Kattegat 1992–2023. Catches are made by a modified Methot–Isaacs–Kidd Midwater trawl (MIKT). In 2008-2010 and in 2021, zero glass eels were caught; in 2011, no sampling took place. The horizontal dotted line represents the average for the displayed years.

3. Restocking

Restocking (stocking) is the practice of importing young eel from abroad (France, England before Brexit, in historical times also Denmark) or transferring them from the west coast to inland waters or Baltic coast and releasing them into natural waters. The size of the young eels varies from glass eel, to on average five-to-seven year old bootlace eels (ca. 40 cm length, 100 gr individual weight, so-called ‘sättål’). In order to facilitate temporal and spatial comparisons, all quantities of young eels have been converted to glass eel equivalents (see Appendix C for details). Restocking of young eel started in Sweden in the early (Trybom & Schneider 1908), and has been applied in inland waters as well as on the coast.

3.1 Restocked quantities

Table 1 provides an overview of the number of eel used for restocking in recent years (2010-2023). Appendix C gives full detail (spatial and temporal) for the inland waters; Appendix A for the coastal waters.

Table 1: Number of eels restocked, by area.

Year	West coast	Inland waters	Baltic coast
2010	180 000	1 694 510	62 000
2011	543 000	1 977 984	103 000
2012	553 000	1 924 022	89 000
2013	581 600	1 953 984	122 000
2014	738 611	2 022 432	192 000
2015	785 250	944 144	137 000
2016	1 383 035	1 334 362	153 800
2017	459 574	394 074	93 707
2018	1 250 408	1 584 371	273 860
2019	1 217 307	1 419 808	235 000
2020	916 725	1 964 979	225 000
2021	0	387 982	55 000
2022	184 483	526 149	85 000
2023	169 840	538 834	70 000

3.2 Restocking and stock assessments

Where eels have been restocked, the yellow eel stock consists of a mix of naturally recruited and restocked individuals. This may or may not complicate the assessment of the size of the stock and of anthropogenic mortalities.

For the coastal fisheries (both west coast and Baltic coast), the assessment is based on fisheries related data (landings, size composition of the catch, tag recaptures). The fisheries exploit the mix of natural and restocked individuals, and therefore, the estimates of stock size and mortalities relate to the mixed stock. Trends in restocking and natural recruitment are shown as relative indices, not in absolute numbers in the stock. Since the absolute number of natural recruits is generally unknown, the sum of natural and restocked recruits is unknown. Hence, the recruitment data have not been used in the coastal assessments.

The contribution from restocking to the coastal stocks is thought to be relatively small in comparison to the natural stock. For the west coast current restocking (170 000 in 2023) will potentially produce less than 50 t, which is a negligible quantity in comparison to the production potential from natural recruitment (pristine production estimated at 1 154 t or more; Dekker 2012). However, for the current depleted state of the stock, restocking could make a larger contribution to production. For the years 2013-2021, Myrenås (2024) found that restocked yellow eel made up on average 26% of survey catches at a restocking release site at the west coast. For the Baltic coast, the potential production of silver eel B_{best} was estimated at 3 770 t (Dekker 2012), and current restocking (order of 70 000 per year recently) will potentially produce less than 20 t. Thus, it is doubtful whether these small additions made by coastal restocking to the natural stock will be noticeable in the long run.

For the inland waters, the reconstruction of the silver eel production identifies explicitly which eels were derived from restocking, and which ones from other sources. The restocking-based production is in an order of 200-300 t, while the natural silver eel production in 2023 is estimated at 24 t.

All in all, none of the assessments are biased by quantities of eel being restocked, and all assessments relate to the stock comprising both natural and restocked individuals.

3.3 Restocking and stock indicators

Over the decades, restocking has been practised with various objectives in mind (Dekker & Beaulaton 2016): to support/extend a fishery, to compensate for other anthropogenic impacts, or to support the recovery of the stock. The classical objective for restocking in Sweden has been to support the fishery, where releasing young eel upstream of hydropower barriers mitigates the negative effect of these barriers on upstream eel recruitment. More recently, restocking has also been intended to support recovery of the stock through governmental restocking in unobstructed unexploited waters (Anonymous 2008), a program that has been paused since 2020. Furthermore, restocking is also done to compensate for other anthropogenic mortalities, e.g. restocking on the coast in the programme 'Krafttag Ål', meant to compensate for the impact of hydropower generation on eel (Dekker & Wickström 2015). In short, multiple objectives of restocking (i.e. supporting the fishery, stock recovery, and compensating for other anthropogenic impacts) have been and still are in use.

Though the framework of stock indicators (see section 1.6, above) allows for the inclusion of restocking (ICES 2010), different indicators can be calculated depending on the setting and objectives. In particular, the indicator of anthropogenic mortality ΣA, expressing the relation of the actual silver eel escapement $B_{current}$ to the current potential escapement if no anthropogenic actions had influenced the stock B_{best}, can be interpreted in two different ways. If the silver eel produced from restocking is included in the estimate of B_{best} (say B_{best}^{+}), that is $\Sigma A^{+} = -\ln(B_{current}^{+}/B_{best}^{+})$, the resulting mortality indicator expresses the mortality exerted on any part of the stock, both natural and restocked. If, however, the restocking is *not* included in the calculation of B_{best} (say B_{best}^{-}), the resulting indicator $\Sigma A^{-} = -\ln(B_{current}^{+}/B_{best}^{-})$ reflects the effect of management actions (comparing the actual escapement to one without any anthropogenic impact), but does *not* express the mortality actually experienced by any eel in the stock. Instead, ΣA^{-} expresses the net effect of all anthropogenic impacts, including detrimental impacts and the compensatory effect of restocking. In this situation (ΣA^{-}), restocking could be used as a substitute for the required reductions in anthropogenic impacts (e.g. compensating for anthropogenic mortality in one area, by restocking in another area). Noting that this would oppose the conditions specified for the Precautionary Approach (Dekker 2019; FAO 1996, point 48.g), we will not follow this approach, and provide estimates of ΣA^{+} only. For the status of the stock relative to pristine conditions ($\%SSB = 100*B_{current}/B_0$), this report provides estimates *with* and *without* including restocking into the estimate of B_0 in parallel (Figure 8).

4. Fisheries, catch, and fishing mortality

Statistics of catch and landings of commercial fisheries have been kept since 1914, but the time series are far from complete, and the reporting system has changed several times. Until the 1980s, statistics were based on detailed reports collected by fishery officers (fiskerikonsulenter); since that time, sales slips from traders have been collected by the Swedish Statistics Sweden (SCB). For the sales slips, the reported county refers to the home address of the trader, not to the location of fishing. In recent years (since 1999), fishers have reported their landings directly to the responsible national agencies. Where data series overlapped, precedence has been given to the more detailed individual reports. For the analysis of the impact of the silver eel fishery along the Baltic coast, however, a breakdown of landings by county is required for all years (Figure 6). Due to a lack of county-specific reporting on eel landings in the years 1978-1998, county-specific eel landings for these years were reconstructed based on the assumption that each county's relative share of landings remained constant (Dekker and Sjöberg, 2013). For the reconstruction of the inland stock, more detailed data (catch by lake) are required; see Appendix C section C.1.2 for further detail.

For the fishery on the west coast, estimates of fishing mortality were derived by Dekker (2012), based on the estimate in the Swedish Eel Management Plan ($\Sigma F=2.33$, averaged over the years 2000-2006) and the assumption that the stock had not changed considerably in recent years. In spring 2012 however, the fishery was closed completely, i.e. $\Sigma F=0$. Thus, in this report, no new assessment is made; the old estimates have been copied without change. In addition, Appendix A presents trends in stock abundance estimates, based on fishery-independent surveys.

For the fishery in inland waters, Appendix C presents a full update of data for the assessment of the inland stock. The initial assessment in the EMP was based on the assumption that lake productivity can be estimated from habitat characteristics. Over the decades, restocking lakes has resulted in substantially increased catches, contradicting this assumption. Dekker (2012) took the restocking data as the starting point for a reconstruction of lake productivity, but did not include natural and assisted immigration. Dekker (2015) extended that analysis, adding estimates of natural, assisted and restocked recruits, as well as the impact from the fishery and hydropower, in a spatially and temporally explicit reconstruction. That analysis was repeated in 2018 (Dekker 2018), with some minor modifications, and again in 2021 (Dekker et al., 2021). The current assessment largely copies this methodology, while applying a new natural recruitment model (Appendix B), and introducing some small updates in the handling of Trap & Transport data. Trends in catch and fishing impact are presented in Table 2; the trend in the catch is depicted in Figure 7.

For the fishery on the Baltic coast, Dekker and Sjöberg (2013) provided an assessment based on historical mark-recapture data and landings statistics. That analysis has been updated, adding recent mark-recapture data; see Appendix D for details. Since this assessment covers only the silver eel stage within the coastal Swedish Baltic waters, and the actual origin of those silver eel is unknown, the reported fishing mortality does not represent a lifetime mortality, but a partial mortality (F in Swedish waters, say: F_{SE} - not ΣF). Trends in landings and fishing impact are presented in Table 2; the trend in the landings is depicted in Figure 6.

For the fisheries in inland waters and along the Baltic coast, the percentage of yellow eel in the catch is small, and those yellow eels are generally close to the silver eel stage. Hence, the catch in silver eel equivalents is almost identical to the reported total catch.

In recent years, silver eel from lakes situated above hydropower generation plants have been trapped and transported downstream by lorry, bypassing the hydropower-related mortality (see Chapter 6, below). Statistics on these quantities sometimes were, sometimes were not included in the official statistics. In the 2021 assessment (Dekker et al., 2021), these catches were included in the overview of total inland landings. In contrast, here we have now removed all Trap & Transport catches from the landings data shown in Table 2. See Chapter 6 on Trap & Transport for an overview of these catches.

For the recreational fishery, only fragmentary information is available (Anonymous 2008); since 2007, recreational fishery on eel is no longer allowed (except in some designated waters, generally above three hydropower generation plants. See FIFS 2004:37, Annex 6 for details).

Table 2: Fisheries statistics, by year and area. For the west coast and the inland waters, the lifetime fishing mortality ΣF is reported; for the Baltic coast, only the impact of the Swedish fishery F_{SE} can be assessed.

	Landings (tonnes)			Fishing mortality (rate)		
Year	West coast	Inland waters	Baltic coast	West coast ΣF	Inland waters ΣF	Baltic coast F_{SE}
2006	239	128	366	2.66	0.40	0.055 (2006–2009)
2007	170	114	418	1.91	0.33	
2008	164	118	389	1.86	0.33	
2009	107	97	310	1.19	0.25	
2010	108	110	307	1.20	0.28	0.0091 (2010–2019)
2011	83	96	271	0.93	0.24	
2012	0	101	239	0	0.26	
2013	0	89	271	0	0.23	
2014	0	92	213	0	0.26	
2015	0	71	158	0	0.21	
2016	0	80	181	0	0.28	
2017	0	85	143	0	0.34	
2018	0	89	143	0	0.41	
2019	0	72	94	0	0.35	
2020	0	91	95	0	0.47	0.0034 (2020–2023)
2021	0	90	77	0	0.45	
2022	0	57	60	0	0.24	
2023	0	82	93	0	0.34	

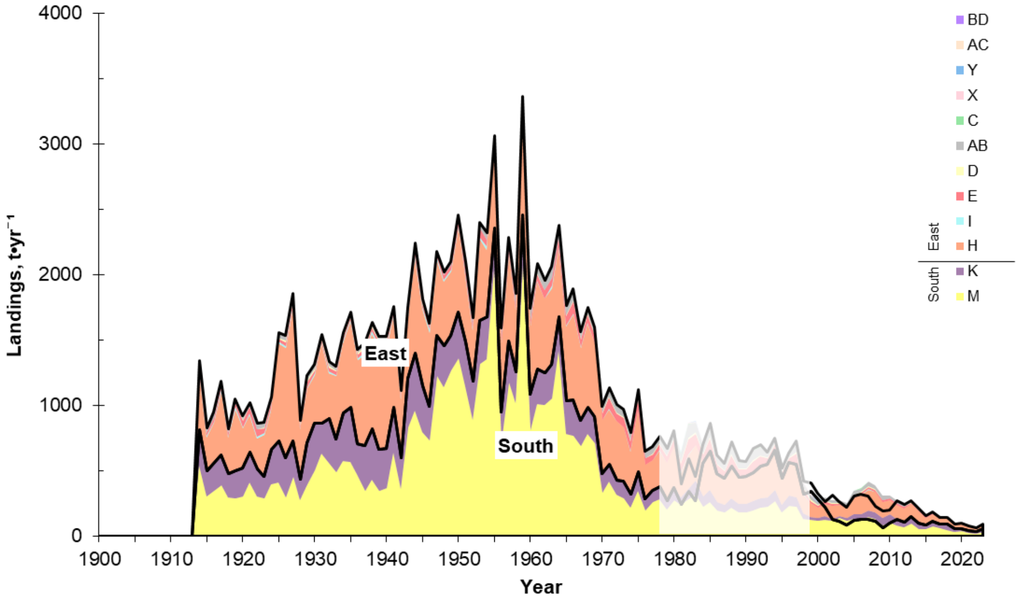

Figure 6: Trend in landings from the Baltic coast fisheries, by county (colours) and area (black lines). In the years 1978-1998 (faded), due to lack of detailed records, it has been assumed that the percent-wise contribution of each county had remained constant. Note that the total landings on the Baltic coast come predominantly from five counties (AB Stockholm, E Östergötland, H Kalmar, K Blekinge and M Skåne) and that the contribution from other areas is barely visible in this graph.

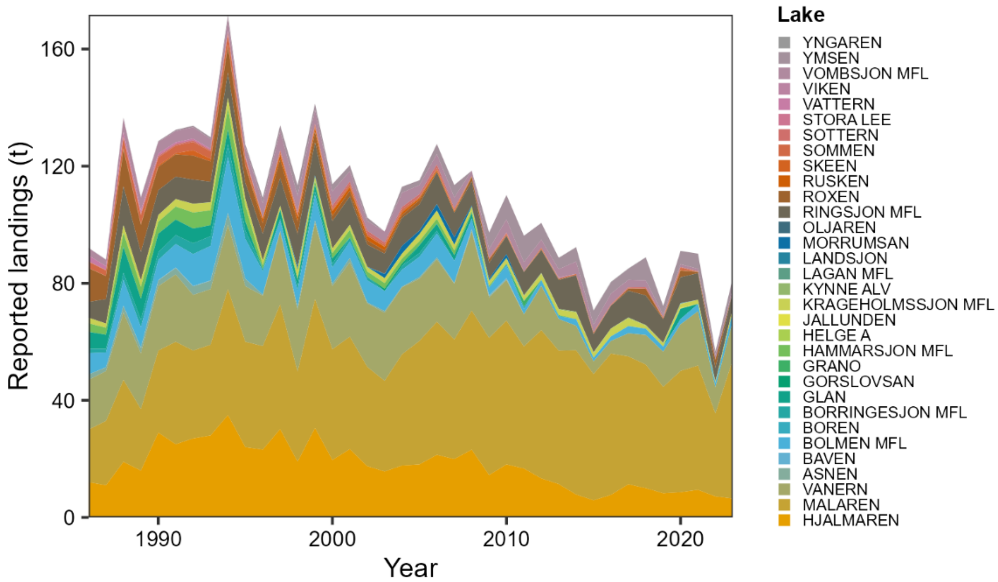

Figure 7: Trends in landings of eel from inland waters. Before 1996, only the combined total for all smaller lakes (all except Vänern, Mälaren, and Hjälmaren) are known. Therefore, for the years 1986-1995, the landings of the smaller lakes have been reconstructed based on that year's share of each lake's estimated silver eel production vs total silver eel production. Statistics before 1986 are only available for Vänern, Mälaren, and Hjälmaren.

5. Impact of hydropower on silver eel runs

A reconstruction of the inland stock is presented in Appendix C. That includes a spatially and temporally explicit reconstruction of the impact of individual hydropower stations. The data in Table 3 are estimates taken from this reconstruction. The estimates refer to the actual situation, i.e. taking into account the removal of eels for the Trap & Transport programme, reducing the hydropower mortality. For the release of the Trap & Transport eels, see Chapter 6.

From the detailed reconstruction in Appendix C, it becomes clear that the temporal variation shown in Table 3 is effectively the consequence of a temporal change in the spatial distribution of the stock, caused by reduced natural recruitment and altering restocking practices. A reduction in natural recruitment means that a smaller fraction of silver eel production originates from waters that are not impacted by hydropower mortality, increasing the relative impact of hydropower on silver eel escapement. Furthermore, since 2010, restocking first shifted relatively more towards lakes with hydropower stations downstream (e.g. Lake Vänern), which results in a further rising estimate of the overall impact from hydropower on the inland eel stock.

Table 3: Estimates of the impact of hydropower generation plants on the silver eel run (2006-2023).

	Biomass of silver eel killed (tonnes)			Hydropower mortality ΣH (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2006		98			0.46	
2007		119			0.52	
2008		141			0.62	
2009		183			0.76	
2010		196			0.84	
2011		207			0.88	
2012		213			0.98	
2013		208			0.97	
2014		188			0.94	
2015		183			0.94	
2016		154			0.97	
2017		126			0.95	
2018		107			0.97	
2019		110			1.00	
2020		97			0.99	
2021		97			0.94	
2022		109			0.77	
2023		115			0.86	

6. Trap & Transport of silver eel

In recent years, silver eel from lakes situated above hydropower generation plants have been trapped and transported downstream by lorry, bypassing the hydropower-related mortality (Table 4). The initial catch of silver eel for this programme conforms to a normal fishery, but seeing as how the eel are released again downstream, free to continue their migration to their spawning area, we have opted not to include the catches of Trap & Transport in the fishery statistics (Chapter 4). Instead, we here report on the Trap & Transport data separately.

The effect of the Trap & Transport programme has an indirect effect on the magnitude of the estimated hydropower mortality, by reducing the number of eel that migrate downstream past hydropower turbines, thereby leading to a reduction in hydropower mortality. Thus, the hydropower mortality as shown in Chapter 5 also includes the impact of Trap & Transport.

Table 4: Quantities of silver eel released on the coast (or below the lowest barrier in rivers), in the context of the Trap & Transport programme (2010-2023).

	Biomass of silver eel (tonnes)		
Year	West coast	Inland waters	Baltic coast
2010	5.2		
2011	4.9		3.1
2012	8.6		1.6
2013	10.4		3.8
2014	14.6		7.2
2015	13.0		6.0
2016	13.0		6.0
2017	12.7		5.7
2018	10.9		6.2
2019	10.8		4.8
2020	10.6		7.9
2021	12.5		12.0
2022	13.5		8.2
2023	13.6		5.5

7. Other anthropogenic impacts

In addition to what has been described in the previous sections, several other anthropogenic actions do have an impact on the stock. This chapter discusses those.

7.1 Illegal, unreported and unregulated fisheries

During the last few years, media have repeatedly reported on an extensive Illegal, Unreported or Unregulated catch of eels (IUU). This information has mainly been based on reports from the responsible agencies, such as the Swedish Agency for Marine and Water Management, the Swedish Coast Guard and the different County Boards. These agencies have reported on an increasing number of confiscated fyke nets, sometimes with notes of how many eels were caught. No full data compilation has been made, but most seizures appear to have been made in the County of Blekinge, followed by Östergötland, Västra Götaland and Kalmar counties. However, the distribution of this illegal fishery is probably biased, as most controls were made in Blekinge County.

Dekker *et al*. (2018) compiled a first, preliminary estimate of the order of magnitude of IUU fisheries, in Swedish inland waters and along Swedish coasts. This indicated that the total IUU in Sweden may be of the same magnitude as the reported commercial landings.

Having only an order-of-magnitude estimate for a recent year – not well quantified, and not for the range of years covered by our assessments – there is no option to include this information in our quantitative analyses.

7.2 Cormorants and other predators

In the EU Eel Regulation (Anonymous 2007), "combating predators" is listed as one option (amongst many others) to protect and enhance the eel stock. In recent years, there has been societal discussion whether and to what extent natural predators have increased in numbers due to anthropogenic actions (protected status and/or indirect, ecosystem effects), which might have contributed to the decline of the eel stock. Limiting or reducing the predator abundance will enhance the status of the eel stock. In this context, cormorants (*Phalacrocorax carbo carbo* and *P.*

carbo sinensis) as well as seals (*Phoca vitulina*, *Pusa hispida*, and *Halichoerus grypus*) have been discussed.

In a literature review, Hansson et al. (2018) showed that, in the southern Baltic Sea, the eel consumption by cormorants in 2010 was in the same order of magnitude as the fishing impact. Additionally, they calculated that the impact of seals is negligible, but that is obviously contradicted by frequent observations of direct predation. For inland waters, the cormorant impact has been studied in several lakes with incongruent results (sometimes showing big impacts, sometimes small), and no country-wide overview has been compiled. Dekker (2015) summarised that information, and developed a tentative assessment ("a few percent of the approx. 3000 t of fish biomass consumed"), coming to the conclusion that this did not discredit his assessment for the inland water. However, the temporal increase in cormorant abundance was not addressed, and cormorant abundance has continued to increase since (Lundström 2024).

In this report, the impact of the Swedish fisheries on the run of silver eels along the Baltic coast is assessed (Appendix D), but no assessment is made of the yellow eel stock (in Sweden and other areas/countries) from which this silver eel run is derived. Though an integrated assessment for both yellow and silver eel - for the whole Baltic, and covering all impacts, including increased predation pressures - is urgently required, there is no option to achieve that in the current report.

For the assessment of the inland stock, section C.2.3 updates the tentative analysis of Dekker (2015) concerning the effect of cormorant predation on the inland stock assessment.

8. Stock indicators

In this section, stock indicators, as requested by the EU, are presented for the different parts of the stock in Swedish waters. Table 5, below, provides the indicators in full detail.

For the west coast, no estimates of stock size are available. The 2012-indicators were based on the 2000-2006 assessment made in Anonymous (2008). Since spring 2012 (fishing closure), fishing mortality has been zero (disregarding the potential effect of illegal fishing). The intensity of the fishery-independent monitoring programme (sampling four sites each year) is insufficient to allow a direct estimation of the stock abundance, or an assessment of the relation between stock abundance and habitat characteristics. Hence, the size of the west coast stock remains unquantified. Appendix A provides a basic trend-analysis of west-coast monitoring data, indicating that the decreasing recruitment observed over the past years appears to have now come to a halt. The closure of the fishery in 2012 has also led to a better survival into larger size classes, and a relative recovery of their abundance. However, for the reasons mentioned above, abundance cannot be quantified in absolute terms.

For inland waters, Appendix C presents a comprehensive and fully updated assessment, from which most stock indicators in Table 5 (below) were derived. For the pristine biomass (the biomass of silver eel in the absence of any anthropogenic mortality, at historically high recruitment as before 1980), the previous estimate (300 t plus the contribution from restocking) is copied from Dekker (2012). Mid-term extrapolations (one lifetime ahead in time) assume that the status quo is continued (unchanged recruitment and restocking numbers, unchanged fishing and hydropower mortality). These mid-term extrapolations show the expected effect of the trends in recruitment and restocking in most recent years.

The indicators for the inland stock apply to all inland waters, with the exception of a number of smaller rivers (4 % of the total drainage area), in which no barrier, no fishery and no hydropower generation occurs. Additionally, four smaller drainage areas close to the Norwegian border (0.7 % of the total drainage area) have been excluded. For these north-western rivers, an extremely high natural recruitment is predicted, based on extrapolation from other rivers, but no independent evidence exists. No assisting of migration, restocking or fishery occurs in these four rivers.

The indicators for the inland stock (Figure 8) show that the stock biomass is below the long-term goal, anthropogenic impacts (fishery and hydropower, together) exceed the minimum limit that would allow recovery, and those anthropogenic impacts are increasing. The spatial shift in the restocking in around 2010 (major restocking in Mälaren and other eastward flowing waters – major restocking in Vänern and other westward flowing waters) is the main driver behind this rise in mortality. Extrapolations for the coming years indicate that mortality is expected to diminish slightly in the years coming (2021 is the peak in recent years). However, this will not bring the anthropogenic impacts below the minimum limit that will allow the stock to recover.

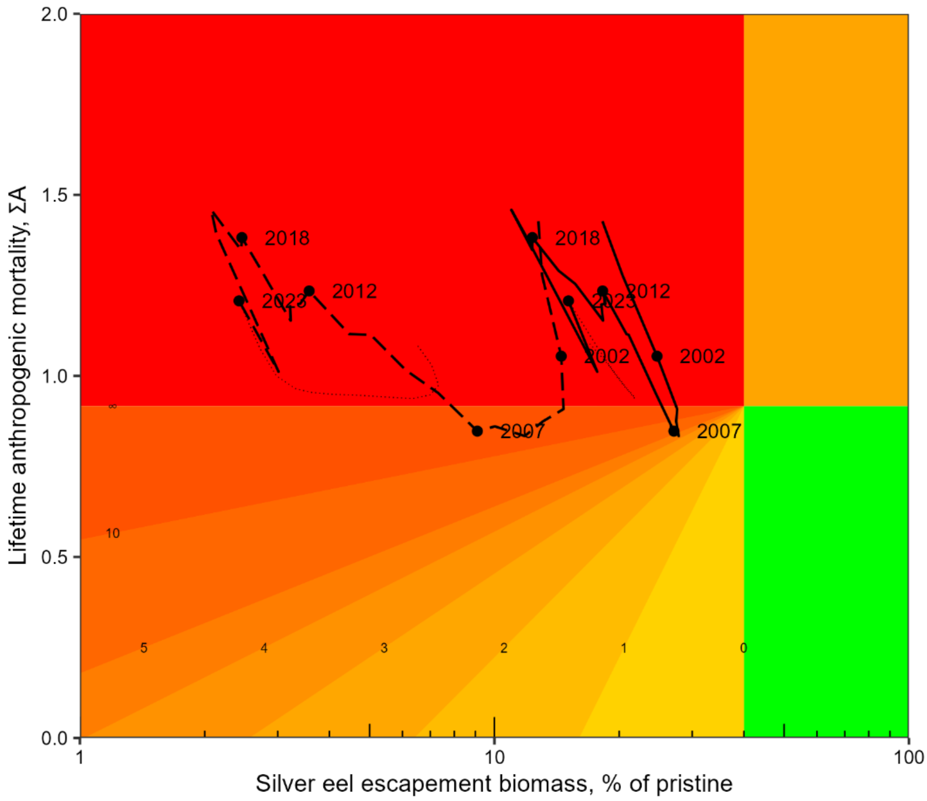

Figure 8: Precautionary Diagram for the Swedish eel stock in inland waters. For the west coast and the fisheries along the Baltic coast, no stock indicators are currently available. For inland waters, the true mortality is shown (that is: not interpreting restocking as compensation for other mortalities), giving separate curves for the current biomass with (solid) or without (dashed) the contribution from restocking to the biomass of the silver eel run. The thin dotted lines show the estimated future trends. For the details of the diagram, see Section 1.6.6 and Figure 2.

For the Baltic coast, the assessment in Appendix D covers the impact of the Swedish silver eel fishery. Other impacts on the same eels (in earlier life stages or further along their migration route, e.g. residing in other countries) have not been included

– no integrated assessment for the whole Baltic stock has been established yet. For the Swedish eel fishery on the Baltic coast, previous assessments derived estimates of lifetime anthropogenic mortality ΣA from the analysis in Dekker & Sjöberg (2013); estimated B_{best} from the ratio of landings to ΣA; and calculated $B_{current}$ as what is left after the catch had been taken from B_{best}. However, those estimates covered the Swedish coastal fishery only, disregarding other anthropogenic impacts in earlier life stages and earlier habitats, and therefore, the results represented a partial assessment – neither the estimate of ΣA nor the estimates of B_{best} and B_0 reported in 2012 and 2015 truly represented the requested indicators for the silver eel run along the Baltic coast of Sweden. Noting that the presentation of partial indicators (in place of lifetime indicators) gave rise to confusion (ICES 2017), Dekker *et al*. (2018) reported them as partial indicators, and left the estimates of ΣA, B_{best} and B_0 missing. Here, we repeat this approach following the same argumentation. Over the years 2020-2023, the fishing mortality rate F_{SE} is estimated at approx. 0.003 yr^{-1} (≈0.3%); the average landings were 81 t/yr. Estimates of the silver eel run along the Swedish coast range from 556 t/yr (Stockholm) to 10555 t/yr (Blekinge). However, the combination of low landings numbers and extremely low recapture rates increasingly questions the reliability of these estimates for the silver eel run. For the time being, the 2012-estimate of the silver eel run (before fishing) of 3770 t is maintained (as was done in 2015, 2018, and 2021 too). It is evident that this constant value is in the right order of magnitude, but does not track the recent trends in the stock. Given the uncertainties in the data, which result in highly unlikely outcomes, we cannot provide any better. See Appendix D for further details.

For the Trap & Transport programme, only the biomass of silver eel affected is reported, but no corresponding mortality rates.

In the absence of stock indicators for the west coast and incompleteness of those for the Baltic coast, no indicators for the whole country can be derived.

Table 5: Stock indicators by area and year. For inland waters, biomass indicators are given with (+) and without (-) the contribution from restocked eels. All mortality estimates refer to true mortality (both on natural and restocked eels), not interpreting restocking as a compensation for other mortalities. For all coastal waters, ΣH=0, hence ΣF=ΣA. All biomass indicators expressed in tonnes, mortality indicators as rate per lifetime, %SPR (relative survival) and %SSB (relative state of the stock) in percent. (All symbols are explained in the text box at Section 1.5)

	West coast						Inland waters												Baltic coast						
							with restocking +				without restocking -				Mortality rates										
year	$B_{current}$	B_{best}	B_0	%SSB	∑A	%SPR	$B_{current}^{+}$	B_{best}^{+}	B_0^{+}	%SSB^{+}	$B_{current}^{-}$	B_{best}^{-}	B_0^{-}	%SSB^{-}	∑F	∑H	∑A	%SPR	$B_{current}$	B_{best}	B_0	%SSB	∑A	%SPR	year
2006					2.66		165	390	619	26.7	30	71	300	10.0	0.40	0.46	0.86	42.29	3404						2006
2007					1.91		174	406	643	27.1	27	64	300	9.1	0.33	0.52	0.85	42.85	3352						2007
2008					1.86		163	422	665	24.5	22	57	300	7.3	0.33	0.62	0.95	38.57	3381						2008
2009					1.19		159	439	688	23.1	18	51	300	6.1	0.25	0.76	1.02	36.22	3460						2009
2010					1.20		150	455	709	21.1	15	46	300	5.0	0.28	0.84	1.11	32.85	3463						2010
2011	12	1154	1154	1	0.93	39	148	452	711	20.8	13	41	300	4.5	0.24	0.88	1.12	32.77	3499						2011
2012					0		129	442	705	18.2	11	37	300	3.6	0.26	0.98	1.23	29.09	3531						2012
2013					0		128	425	692	18.5	10	34	300	3.4	0.23	0.97	1.20	30.14	3499						2013
2014					0		121	402	670	18.1	10	32	300	3.2	0.26	0.94	1.20	30.20	3557						2014
2015					0		117	370	640	18.3	10	31	300	3.2	0.21	0.94	1.15	31.58	3612						2015
2016					0		94	329	599	15.7	9	30	300	2.9	0.28	0.97	1.25	28.54	3589						2016
2017					0		80	292	562	14.3	8	30	300	2.7	0.34	0.95	1.29	27.51	3627						2017
2018					0		66	262	532	12.3	7	29	300	2.5	0.41	0.97	1.38	25.11	3627						2018
2019					0		64	246	517	12.3	7	28	300	2.4	0.35	1.00	1.35	25.98	3676						2019
2020					0		57	245	518	11.0	6	27	300	2.1	0.47	0.99	1.46	23.26	3675						2020
2021					0		62	249	524	11.8	6	26	300	2.1	0.45	0.94	1.39	24.83	3693						2021
2022					0		95	261	536	17.7	9	25	300	3.0	0.24	0.77	1.01	36.37	3710						2022
2023					0		84	281	556	15.1	7	24	300	2.4	0.34	0.86	1.21	29.90	3677						2023

9. Discussion

9.1 Comparison to the 2021 assessment

For the west coast stock, Dekker *et al*. (2021) did not present an assessment, following the conclusion of Andersson et al. (2019) that no realistic option currently exists to assess the stock in full detail (absolute stock size, past and present anthropogenic mortality). However, analysis of trends in fishery-independent surveys (Appendix A) does allow monitoring the local stock after the closure of the fishery in 2012, and results confirm the relative recovery of the previously exploited part of the stock. This relative recovery, however, is superimposed on the long-term decline of the whole stock. Though Appendix A updates the time series, the current assessment is essentially a repeat of the 2021 results.

For the inland stock, the current assessment updates and improves the assessment of Dekker (2015), Dekker *et al*. (2018), and Dekker et al. (2021). This year's assessment updates the data, adds a new natural recruitment model, and no longer includes Trap & Transport catches in the fisheries statistics, while largely replicating the rest of the previous methodology. The outcomes confirm the 2021 evaluation of the status of the stock: the biomass is below the long-term target, and anthropogenic impacts exceed the minimum level that will allow the stock to recover.

For the silver eel fisheries on the Baltic coast, the current assessment methodology is identical to the 2021 assessment; the database has just been extended. As before, estimates of fishing impact are derived, pooled by decade. If and when a more rapid evaluation will be required in future (evaluating annual changes), a more intense mark-recapture programme will be needed.

Recent tagging experiments (Figure D2) were more evenly spread along the coast than the historical experiments (Dekker & Sjöberg 2013; their Figure 4), and the trend in the distance from release to recapture showed a meaningful relation to the trend in fishing impact. The number of days between tagging and recapture, however, appears to have declined when compared to before the 1990s – possibly due to restrictions on the length of the fishing season. First tags applied during the fishing season can be recaptured until the end of the season, but not thereafter. A further shortening of the season in the second half of 2023 might further decrease the number of days between tagging and recapture.

As in previous years, the current assessment covers the impact of the Swedish coastal fishery only. Other anthropogenic impacts (on earlier life stages, and possibly in other countries) have not been considered. Ground-truthed information on the production of silver eel across the Baltic has not been collated and cross-Baltic cooperation in management and assessment has yet not been achieved. Development of the cross-Baltic cooperation is urgently needed, but cannot be achieved within the context of this national assessment.

9.2 Requirements for the 2023 reporting to the EU

Preparations have been made for a data call (jointly by ICES and the EU Commission) later this year, in which the stock indicators will be included; the indicators in this call are essentially in-line with the earlier data calls. We therefore worked towards the requirements specified for the 2021 Data Call. Comparing those requirements to the results in this report, it shows that all requested indicators have been considered, but not all have been produced – see the discussion in Chapter 8, above. Only the current assessment of the inland stock does produce all requested indicators.

10. Recommendations and advice

In this report, an assessment of the Swedish part of the European eel stock is presented, and evaluated against the objectives of the Swedish Eel Management Plan (Anonymous 2008) and the EU Eel Regulation (Anonymous 2007). This report extends and updates the results of the previous assessments (Dekker 2012; 2015; Dekker *et al*. 2018; Dekker et al., 2021). Results can inform a future revision and improvement of the Swedish Eel Management Plan. The presented national stock indicators were and will further be used for international evaluations (ICES 2013, 2016, 2018, 2021 and coming), informing the international discussions. While ICES (2023) provides advice on the status of the stock across its entire distribution area, this chapter evaluates the status in Sweden against the objectives of the Swedish Eel Management Plan, filling the gap between national assessment and international evaluation, providing advice on national assessment and management.

For the <u>west coast</u>: the status of the stock is not well known. Following the closure of the fishery in 2012, fishing mortality is zero (disregarding any illegal catches), but no exact quantification of current ($B_{current}$), current potential (B_{best}) or pristine biomasses (B_0) could be made. However, the current stock biomass is undoubtedly far below the long-term recovery target, and stock surveys indicate that the stock in general is only just recovering after the commercial fishing closure in 2012; the breakpoint in glass eel recruitment in 2011 (observed across the continent) is reflected on the west coast, too: the smallest size classes no longer show a decreasing trend in abundance. To achieve the management targets of the EU Eel Regulation and the national Eel Management Plan, no further action can be taken on the west coast regarding fishing (fishing mortality is virtually zero, and there is no impact of hydropower on the coastal stock). The impacts of other pressures on the stock, such as natural predation, could be better studied. If the biomass trends of the west coast are to be better quantified in the future, it is recommended to intensify the data collection along the west coast.

Restocking on the west coast adds upon the natural silver eel production. Though this contribution is only a relatively small quantity in comparison to the natural pristine stock, a recent evaluation of the Swedish restocking programme (Myrenås, 2024) has found that restocking can represent a more valuable addition to local silver eel production, given the current depleted state of the west coast stock. However, the majority of west coast eel likely continue to originate from natural recruitment.

For the inland stock: status indicators point out that the stock biomass is below the long-term goal, anthropogenic impacts (fishery and hydropower, together) exceed the minimum limit that would allow recovery, and those anthropogenic impacts have been increasing in recent years. Implemented management actions include Assisted Migration, restocking, fishing restrictions, and Trap & Transport. These measures have strong interactions: if one measure is adjusted, any positive effects are likely to be largely annihilated by other impacts. Management actions resulting in a reduction of the inland stock (e.g.: diminished restocking) will decrease the amount of eel that is impacted, but at the cost of increasing the distance to the biomass goal, and/or effectively losing natural habitats thereby reducing biodiversity. Most current management actions are based on the 2008 assessment (included in the national Eel Management Plan; Anonymous 2008), which is fully outdated. It is therefore recommended:

- To urgently reduce anthropogenic impacts on the inland stock to a level that will allow recovery, and
- To revise and improve the management plan for the inland stock, in line with the objectives of the Eel Regulation and the national Eel Management Plan (sustainable management and recovery of the stock).

For the Baltic coast: the impact of the silver eel fishery is far below the mortality limit implied by the national Eel Management Plan (recovery), but this fishery is just one of the anthropogenic impacts (earlier in the eel's life, at other places in the Baltic) affecting the stock. No comprehensive assessment across the Baltic has been achieved, and management across the Baltic area has not been integrated. Hence, the reported indicators relate to the Swedish fishery only. Stock biomass is likely below the threshold of the Eel Regulation (40% of pristine). Fishing restrictions implemented since the adoption of the national Eel Management Plan have reduced the fishing impact, but – noting the limited impact of the Swedish fishery in comparison to other impacts earlier in life (such as fisheries and hydropower impacts, all over Baltic inland waters) - that affects the escapement biomass only marginally. To improve the assessment and management of the stock targeted by the Swedish fishery, more knowledge is needed about the habitat use and impacts experienced before the silver eel become susceptible to the Baltic coast fishery. For this, a comprehensive assessment of the eel stock in the whole Baltic area will be required. It is recommended:

- To continue the assessment of the stock and impacts in Swedish waters, and to embed this in a pan-Baltic, comprehensive assessment.
- To coordinate the national assessments and national protective measures with other range states, i.e. integrated management in the Baltic.

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Appendix A: West coast eel stock

Until recently, the west coast eel stock has been exploited by an extensive fyke net fishery, mainly targeting yellow eel. However, in spring 2012, this fishery north of 56°25'N (near Torekov, Skåne region) has been closed completely. Here we discuss the historical development of that fishery, and present recent information on the development of the west coast eel stock since the closure, including recent restocking.

A.1 Development of the west coast yellow eel fishery

There are two different time-series compiled by SCB: one that is solely based on sales statistics and the location of the receiver of the catch (1970-1999) or landing harbour (from 2000), and another where these data are combined with catch information from fishers (1985-2012). In this section we use the latter because it better reflects the actual eel catch in the area (except for the years 1970-1984).

Increasing foreign demand for eel in the late 1800s resulted in an increased interest for eel fishing in Sweden, and opened the opportunity to develop a commercial eel fishery on the Swedish west coast. The catch data suggest that the eel stock on the Swedish west coast was underexploited in early 1900 (Figure A1). Around this time, fyke net fisheries for eel had limited geographic coverage and eel was captured using baited pots and bucks or longlines in summer or using spears in winter (Haneson & Rencke 1923). These methods did not provide sufficient volumes for trade, so a fyke net fishery was introduced through an exchange of equipment and knowledge of fishing methods from the coasts of Sweden, Denmark and Germany. For example, fishers could get free fishing gears in exchange for selling their catch to German traders in early 1900 (Göteborgs och Bohus läns hushållningssällskap 1866-1961).

As the fyke nets increased in popularity, the fishing area expanded, and reported catches increased from 100 t/yr in 1900-1920, to 200 t/yr in early 1930 (Figure A1). Technical development of fyke nets and boats allowed catches to remain around 250 t/yr, although the number of coastal fishers decreased (Figure A2). The first fyke nets were hand-made, heavy and large, and required high maintenance (frequent cleaning, tarring, and drying). Some fishers had two sets of fyke nets and replaced the used ones with newly cleaned nets, while others switched to fishing for other species during the cleaning. The cotton-nets were gradually replaced by fyke nets made of nylon requiring less maintenance, which could be kept in the water for a longer period, thereby extending the fishing season. In addition, rowing boats were gradually replaced by motorboats, which allowed quick transportation to fishing grounds and extension of the fishing area. The increase in cheap fyke nets and plastic boats may also have increased catch in the recreational fishery.

In early 1900, German and Danish traders visited fishers along the Swedish west coast to buy live eel for export to Germany. This increased the demand and thereby the price on the west coast, which allowed the west coast fishery to expand (Magnusson & Dekker 2021). With time, the transport by boat was replaced by tanker trucks on land. The trade was relatively easy as the eel could be kept alive in fish-bins for long periods of time until being picked up by the tradesmen, and therefore eel fishing made a good complementary income to other small scaled fisheries or agricultural activities.

Reported eel catch dropped temporarily during World War I and II, when export was prohibited, and peaked in 1980-2000 (the peak in early 1980 may be inflated due to changes in the reporting system, Figure A1).

Most of the eel was exported (Figure A3); local demand for yellow eel on the Swedish west coast was low. There was no sale over the counter in the shops, though yellow eel could be specifically ordered. The local demand for small eel increased in 1970-2000 for restocking purposes (<105 t/yr), but decreased again when glass eel replaced the yellow eel in the restocking programme.

From the middle of the 20th century onwards, catch was maintained despite decreasing number of fishers from the early to mid-1900s due to a more intensified fishery (Figure A1, Figure A2). The increasing eel catch on the Swedish west coast from early to late 1900, and the lack of a clear break-point with declining catch, suggest that the west coast eel stock was not overexploited, but may have reached an exploitation level close to its limit in 1980-1995.

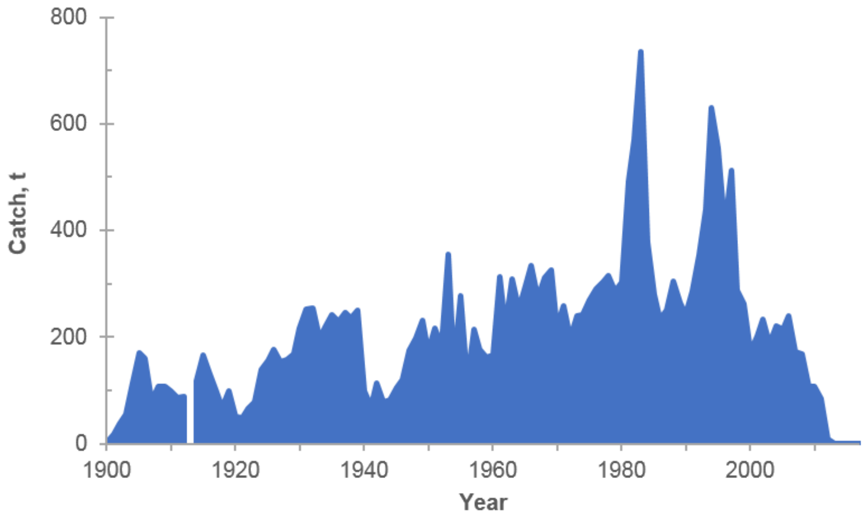

Figure A1: Time trend in eel catch in the Kattegat and Skagerrak from 1900 to 2012 (catch in the period 1970-1984 is solely based on landings data).

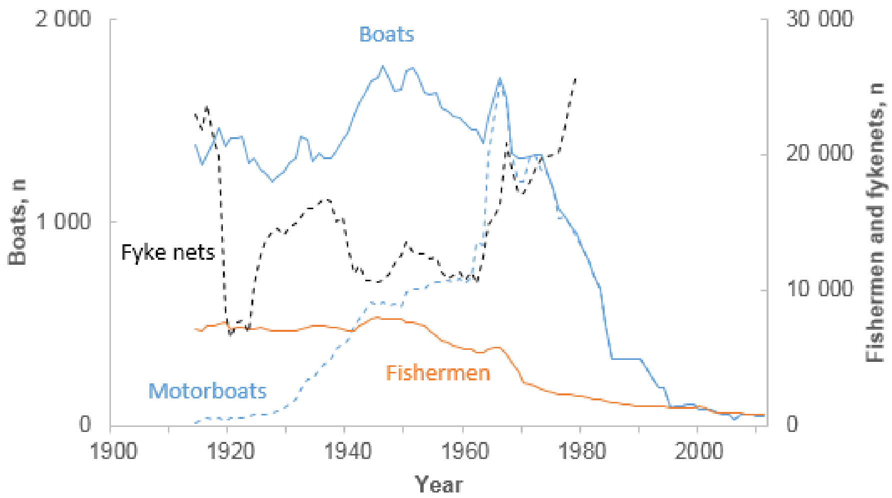

Figure A2: Time trend in number of small boats, fyke nets and fishers on the Swedish west coast.

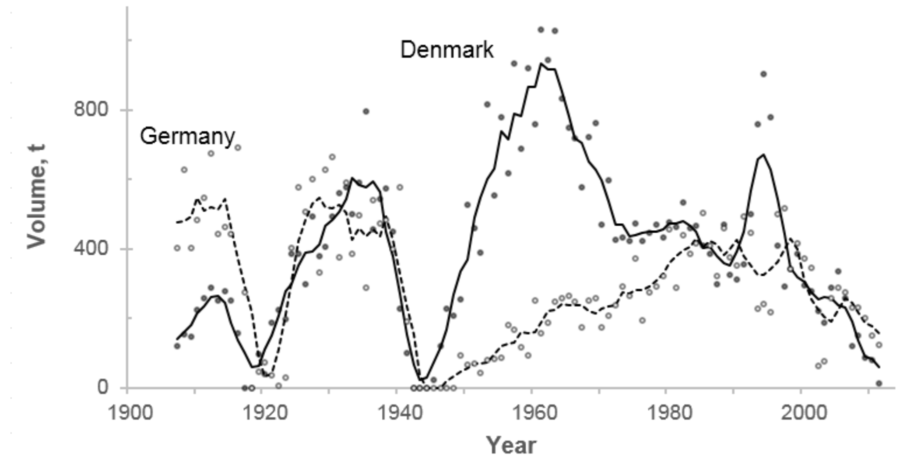

Figure A3: Time trend in total Swedish eel export to the two major receiving countries Denmark and Germany.

A.2 Trends in the west coast eel stock

In the Swedish EMP (Anonymous 2008), a fishery-dependent assessment was presented, analysing length-frequency data and catch statistics from that fishery. Since spring 2012, this fishery has been closed, and the fishery-based assessment could not be continued. Instead, the development of the west coast eel stock has been deduced from fisheries-independent fyke net surveys. Unravelling the trends observed in these fishery-independent data will require a complex analysis. Additionally, the potential emigration of (young) eel from the west coast into inland waters and towards the Baltic Sea has not been considered in past assessments. The previous fisheries-dependent assessments will instead have misclassified such emigration losses from the system as fishing mortality. Hence, a comprehensive analysis of the available fishery-independent data is required, which has not been achieved yet. Therefore, this section presents the primary monitoring data.

Standardised fisheries-independent fyke net surveys are conducted annually by SLU Aqua as part of different monitoring programs, at several sites along the Swedish west coast (Barsebäck, Fjällbacka, Vendelsö, and Hakefjorden). The surveys are conducted in August and the fyke nets, placed in shallow areas at 0-6 m depth, are checked once every 1-4 days. Records include the number of fish caught as well as species identity and individual length, among other things (Andersson et al., 2019). Furthermore, each sample site also reports their total fishing effort, in number of days (24 hours) per fyke net. This allows for the construction of catch-per-unit-effort time-series, which can be used as proxies for eel abundance time-series.

The sampling program has changed somewhat during the years, and from 2021 onwards sampling is done following a depth stratified sampling program for coastal fish (Havs- och vattenmyndigheten 2024). Three of four sites (Barsebäck, Hakefjorden and Fjällbacka) are affected by this change. As a consequence total effort has decreased (from approximately 210, 192 and 192 to 40, 140 and 40 fyke net days per year and site in Barsebäck, Fjällbacka and Hakefjorden, respectively), something which may affect the precision in catch-per-unit-effort estimates.

Data on eel catches were extracted from the fyke net survey database, and used to construct size-specific CPUE time-series. For each sample site, catches of eel were subdivided into four different size classes: under 37 cm, 37 up to 50 cm, 50 up to 60 cm, and 60 cm and larger. Next, for each sample site and each size class, CPUE time-series (eels caught per fyke net per day) were constructed by dividing catch with effort.

The resulting CPUE time series (Figure A4) shows different patterns in eel abundance over time, depending on size class. For eels smaller than 37 cm, the CPUE time-series shows an initial decreasing trend to an overall minimum around the year 2009, after which CPUE appears to stop declining. For eels sized 37-49 cm, the CPUE trend over time is less clear. For eels sized 50-59 cm and over 60 cm, CPUE was stable until 2010, after which a clear increasing trend in CPUE can be observed.

The results of the CPUE analysis match up with trends in eel recruitment and fishing mortality. The CPUE trend of eel smaller than 37 cm largely appears to follow the trend in glass eel recruitment across Europe (ICES 2023), which had been declining since the 1980s until reaching a breakpoint in 2011, after which recruitment stopped declining. The increase in CPUE of larger eel after 2010 appears to match up with the closure of the eel fishery along the Swedish west coast in 2011. Thus, although no definitive abundance trend can be derived here, the CPUE trends appear to show that decreased fishing mortality has resulted in a current increasing trend in eel abundance along the Swedish west coast.

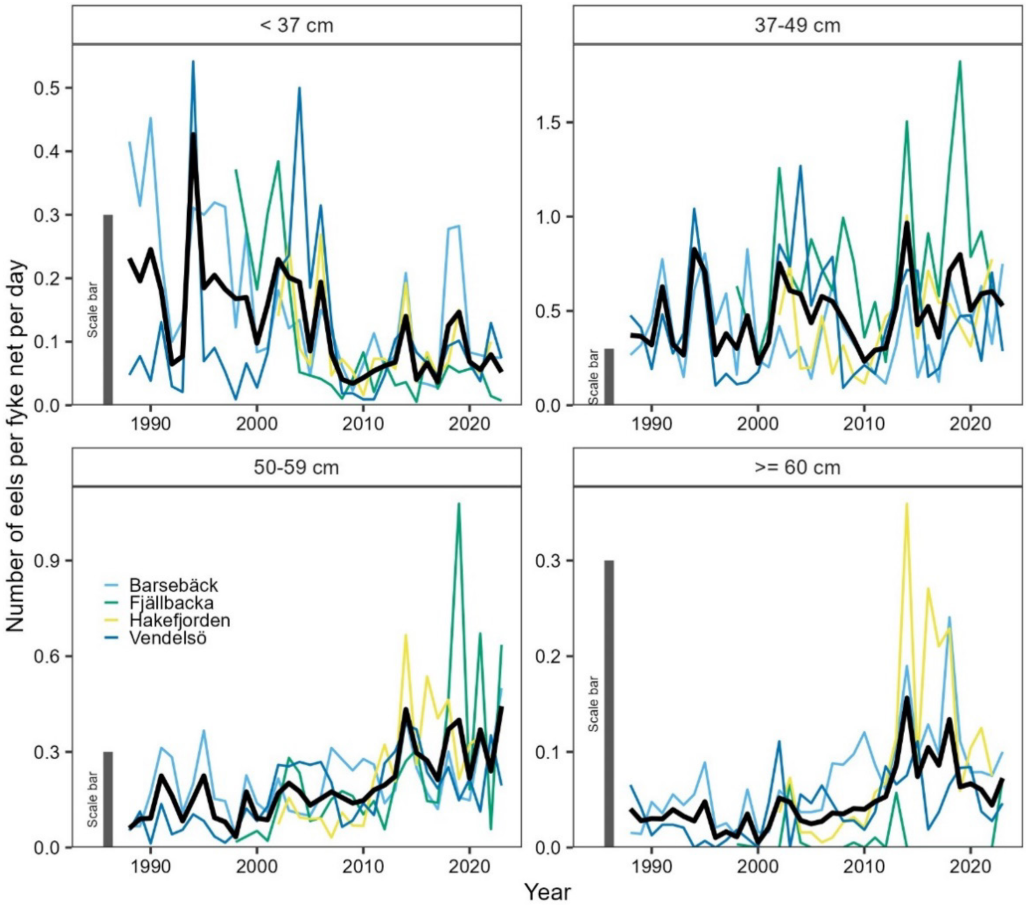

Figure A4: Catch-per-unit-effort of eel by size-class (total length) from fisheries-independent fyke net surveys at various areas along the Swedish west coast, including their average over time (black bold line). Note that the range of the y-axis differs between subfigures, as is also indicated by the scale bar.

A.3 Restocking in coastal waters

Restocking practices have influenced the stock on the west coast in two ways. First, since the early 1950s, medium sized eel have been harvested on the west coast, and transported to the east coast (and inland waters) and released there. Second, since the mid-1970s, glass eels have been imported and released (predominantly) on the west coast (and inland waters) (Figure A5 & Figure A6, age 1 eel represent imported glass eel, age 4 and up mostly represent transported young eel).

Until the year 2000, the amount of young eel transported from west to east coast effectively exceeded the amount of imported glass eel released (Figure A5, extrapolate year class to year of release), but since then, the extraction has come to an end. In the 2010s, after passing of the Swedish EMP, government-funded releases of imported glass eel were initiated. Releases of these eel in coastal waters were mostly performed along the west coast, where commercial fishing for eel had been stopped, and anthropogenic mortality was therefore expected to be minimal. However, this government-funded release of imported glass eel has been paused since 2020, until the effect of past restockings have been better studied.

In the 2010s, on average 1.0 million glass eel equivalents have been restocked per year in coastal waters. In the 2020s, this was reduced to an average of 450,000 per year. 1.0 million glass eel equivalents are expected to produce an amount of silver eel of ca. 60 t/yr, some 15 years later. Noting that the fishing yield on the west coast was in the order of 200 t/yr, and that the potential (natural) production is estimated in the order of 1000 t/yr (Dekker 2012), the addition based on the restocking could be expected to be relatively small, and therefore difficult to detect. Myrenås (2024) found that in survey catches in a west coast area that has seen frequent restockings (the coastal waters around Stenungsund), for the years 2013-2021, an average of 26% of caught yellow eel were of restocked origin, suggesting that restocking under the current depleted stock state can make a more valuable contribution to eel numbers, but that the majority of eel continue to have their origin from natural recruitment.

The references for this Appendix are included in the reference list of the main report, on page 48.

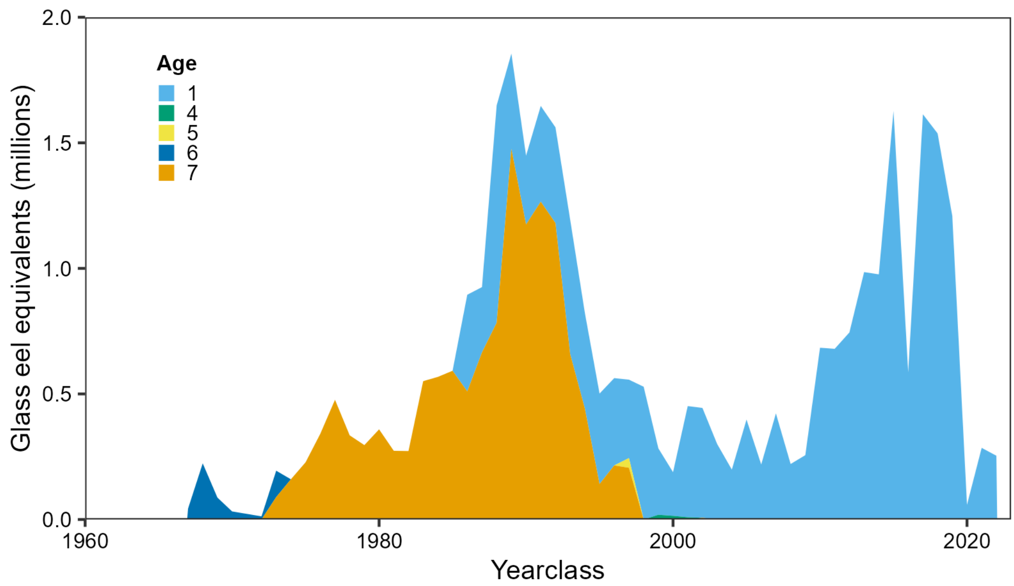

Figure A5: Time trend in the number of eel restocked in coastal waters, shown as glass eel equivalents per year class (not year of restocking), for the year class range 1967-2022. The colour indicates at what age the eels were restocked, with all numbers converted to glass eel equivalents. Before 1970, almost no eel had been restocked on the coast.

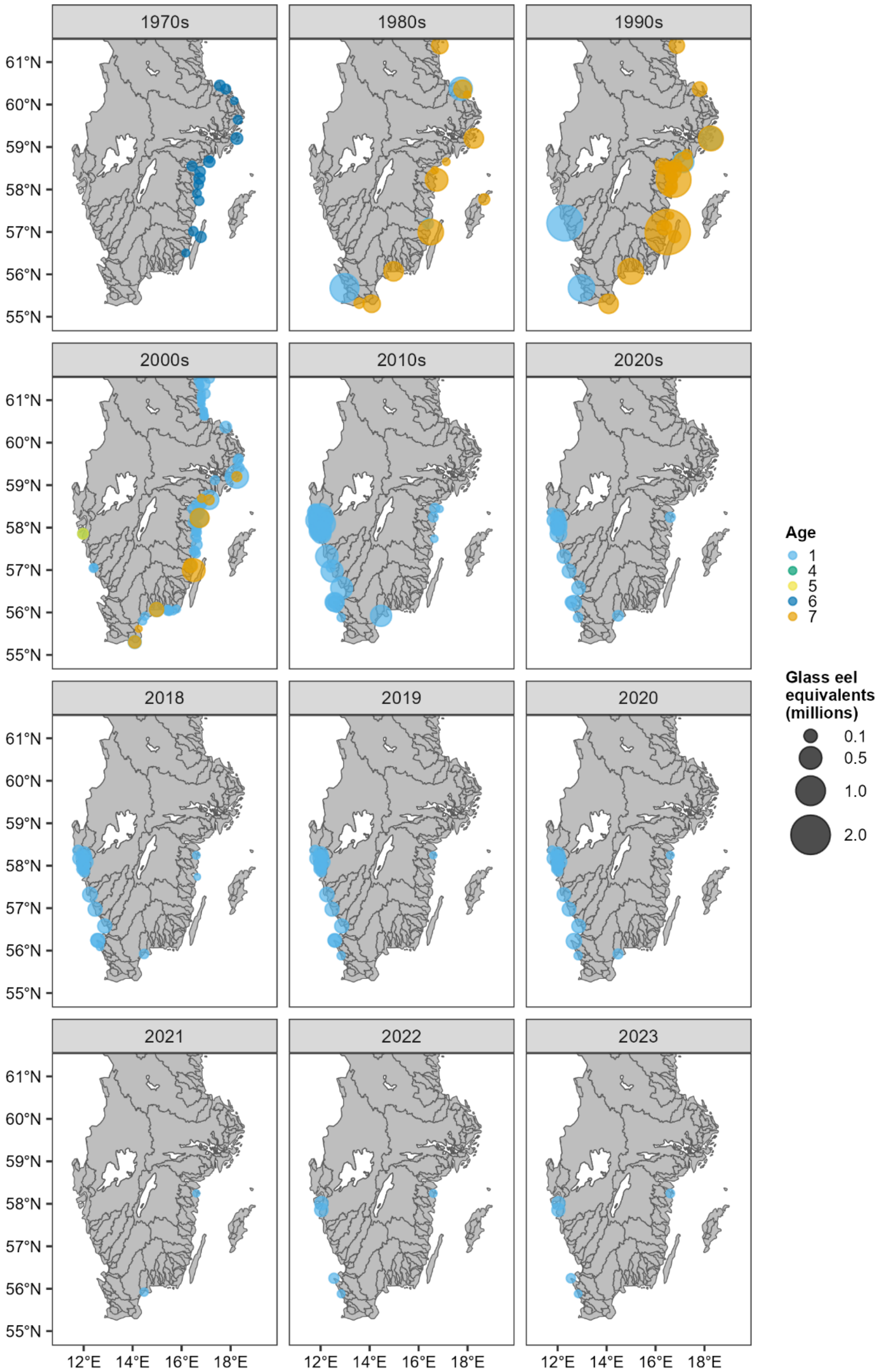

Figure A6: Spatial distribution of the <u>restocking in coastal waters</u>, expressed in glass eel equivalents per year. Restocking actions are shown for decades (1970s – 2020s) or individual years (2018 – 2023). The colour of the symbols indicates at what age the eels were restocked, and their size indicates the number of eel restocked in terms of million glass eel equivalents. Before 1970, almost no eel had been restocked on the coast. Note that these figures are sorted by the year in which the restocking took place, not by year class.

Appendix B: Recruitment into inland waters

The reconstruction of the silver eel production in inland waters (Appendix C) requires information on the natural immigration of glass eels, elvers and bootlace eels into inland waters. Elver trapping for transporting across barriers (assisted migration) provides information on the quantities entering the rivers where a trap is placed (Erichsen 1976; Wickström 2002). Since most traps are located at barriers that block the whole river, there will be very few eels passing upstream. Eels that fail to enter the trap, will remain in the area below the dam, and probably try again soon after (or become part of the coastal stock, to be included in the assessment of that). Hence, considering the set of elver traps as an unbiased and efficient sampling of the natural immigration into rivers, this Appendix analyses the spatial pattern and temporal trend in these data[3]. This will enable interpolation (for years with missing observations in rivers with a trap) and extrapolation (to all rivers without a trap). To begin with, we present the elver traps and locations, the primary results, and a simple trend analysis. Subsequently, we develop the more complex statistical model, that will enable the required statistical extrapolation to rivers without traps, and we discuss the best choice for that extrapolation.

B.1 Material and methods

B.1.1 Study sites

In historical times, eel fisheries occurred in most inland waters in Sweden, up to the far north. For instance, Nordberg (1977) describes the fisheries in the river Ljungan since late-medieval times, and Olofsson (1934) describes an eel fishery at Vändträsket in Alån, north-west of Luleå. These inland eel fisheries relied on young eel recruiting naturally from the Kattegat and Baltic Sea into the rivers. When rivers became progressively blocked for water management or hydropower generation, the damage done to these fisheries was mitigated either by catching and transporting

[3] The estimated number of natural recruits is thus based on data related to rivers with an obstructing barrier. Extrapolation to unobstructed rivers will be cumbersome or impossible. However, the number of completely free-flowing rivers is extremely limited, comprising only 4% of the total surface area in Sweden. Eels that have entered a free-flowing river, will be able to migrate back to the coast at will, and will thus effectively been included in our assessment of the coastal stock. Hence, these unobstructed rivers are ignored here.

immigrating eel from below the barrier where they were blocked to areas upstream (so-called: assisted migration), or by importing young eel from abroad (restocking).

Local water court decisions to mitigate the damage to the eel stock often included an obligation to report on the numbers (or weight) of eel caught, transported and released upstream. The capture of young eel below the barrier was achieved by means of a fixed trap (c.f. Wickström 2002). Noting that the traps were operated consistently for many years (and if changes were made, these were reported), the catches are considered indicative for the abundance of the eel immigrating at the sites concerned. For 22 sites (Figure B1, Table B1), multi-decadal data series are available. The starting years of these series vary from before 1900 to 1991, and some series were discontinued quite long ago (from 1973 to 1991. The number of concurrently operated sites rose from four in 1950 to ten in 1955, to twenty-one in the early 1970s, then declined to around ten in the years since 1990. Currently, eight sites remain active (Figure B1).

Recorded data consist of annual catch per site, in number and/or in weight. Subsamples were taken (though not in all years and not at all sites) to derive an estimate of the number of eels per kilogram. For each site and year, we derived, in order of priority:

1. Catch numbers as actually counted.
2. Catch weight as recorded, converted to numbers on the basis of number-per-kilogram, recorded for that year and that site.
3. Catch weight as recorded, converted to numbers on the basis of number-per-kilogram, as recorded in other years at the same site
4. For two rivers where subsampling has never taken place (Nyköpingsån and Rååån), converting weight to numbers using number-per-kilogram from nearby rivers (Motala ström and Rönneå, respectively).

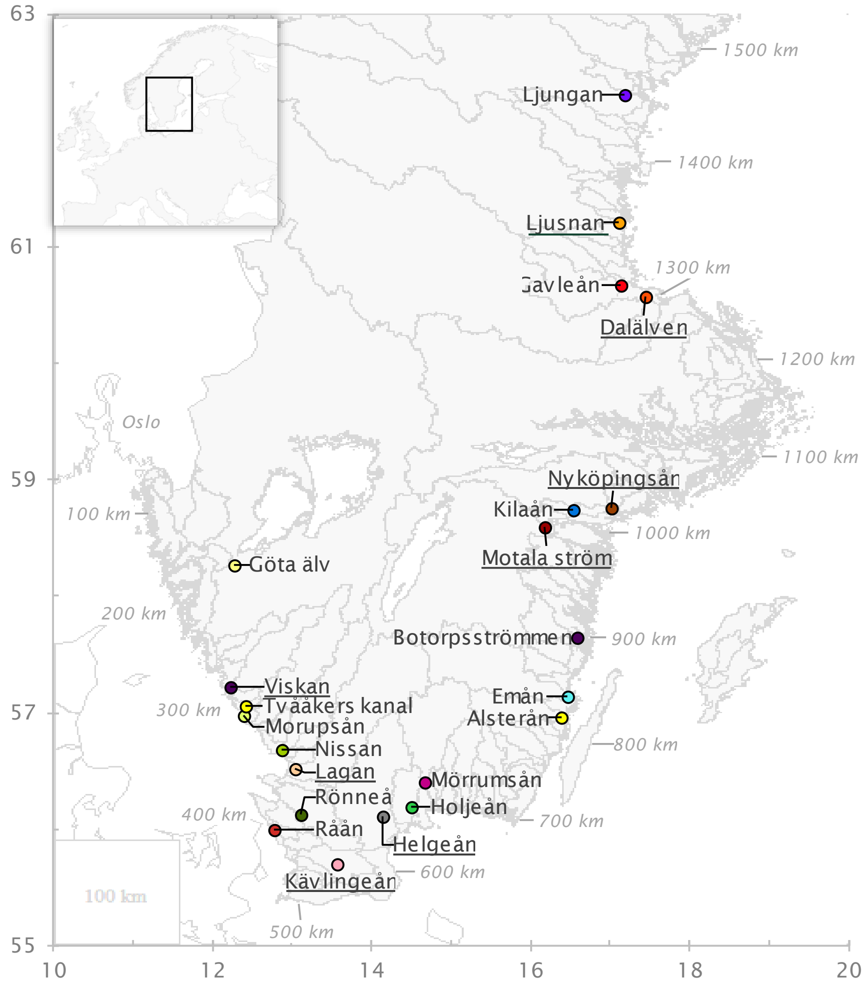

Figure B1: Map of the study area, showing sampling sites, drainage areas and distances along the coast from Oslo. Underlined sites are continuing their sampling up to today.

In some years, reports indicated that the trap had not worked properly; that the hydropower station had been kept on hold for repair; that the trapping had been continued but not for the whole season; or any other reason raising doubt on the validity of the observation. All of these records (n=331) have been flagged as invalid, and excluded from further analysis. In a few cases, an exact zero catch was reported, either in number (n=15) or in weight (n=20), without any indication of invalidness (sometimes, comments even said it was truly zero). This occurred seventeen times for Botorpsströmmen, six times for Tvååkers kanal, five times for Holjeån, two times for Kilaån, and one time each for Ljungan, Morupsån, Nissan, Nyköpingsån and Råån. All of these zeroes occurred before 1990, and all of these series have been stopped (except Nyköpingsån) in the 1970s (but Nissan in 1990 and Tvååkers kanal in 1989). We double-checked these zero records; though we doubt the correctness of the observation as such (see Results, below), the original

Table B1: Characteristics of the sites, the observation series, and the eels. The column 'Valid obs.' gives the number of valid observations since 1950, excluding the years of incomplete or otherwise invalid observations; ctd=continued.

Site	First year	Last year	Valid obs.	Distance Oslo, km	Discharge m^3/s	Distance upstream, km	Altitude m	Weight gr	Age years
Alsterån	1960	1991	29	819	11	5	5	22.2	4.2
Ätran	1932	2012	7	317	51	6	10	2.8	1.4
Botorpsströmmen	1922	1990	30	897	6	0	6	40.6	5.4
Dalälven	1951	ctd	70	1312	348	11	14	62.2	6.3
Emån	1967	1988	21	842	30	4	13	46.4	5.6
Gavleån	1920	1979	28	1327	21	4	7	50.0	5.8
Göta älv	1900	2017	56	221	518	77	23	8.9	2.7
Helgeån	1952	ctd	65	623	46	35	12	4.9	2.0
Holjeån	1947	1976	19	645	8	26	20	20.9	4.1
Kävlingeån	1991	ctd	32	449	4	49	20	12.9	3.3
Kilaån	1948	1979	26	1023	1	31	19	50.0	5.8
Lagan	1925	ctd	74	363	77	4	37	0.8	0.4
Ljungan	1951	1979	23	1464	138	20	9	60.5	6.2
Ljusnan	1950	ctd	51	1362	230	1	18	44.6	5.6
Mörrumsån†	1960	2018	60	663	27	32	119	65.4	6.5
Morupsån	1950	1989	37	303	1	11	11	0.3	0.0
Motala ström	1942	ctd	74	1008	93	5	11	51.9	5.9
Nissan	1947	1990	40	350	41	4	13	0.4	0.1
Nyköpingsån	1922	ctd	51	1024	22	4	11	51.9	5.9
Rååån	1946	1973	24	416	2	4	13	2.3	1.2
Rönne å	1917	2018	60	389	24	37	31	2.3	1.2
Tvååkers kanal	1948	1989	24	303	1	7	26	0.5	0.1
Viskan	1971	ctd	53	276	35	5	1	0.4	0.0

† For Mörrumsån, data from four traps have been combined; see text for details.

data sources did truly report a zero, and hence, we kept the observation as a valid record. Excluding the relatively scarce and less well documented valid records before 1950 (n=133), the total number of valid observations, including the 35 zero observations, comes at n=954.

Characteristics of the 22 sampling sites are given in Table B1, and described in detail in Wickström (2002). Most sites are located just below the most downstream barrier in each river. In Göta älv, however, there is one hydropower station (Lilla Edet, built in 1918) in-between the trap and the sea; in Kävlingeån, there are two (Lilla Harrie 1509 and Bösmöllan 1896). In Mörrumsån, there are five dams (in upstream order: Marieberg 1918, Hemsjö nedre 1917, Hemsjö övre 1906, Ebbemåla 1907, Fridafors Nedre 1893), one above the other, and eels have been collected at all these dams. Since none of these barriers in any of these rivers were erected in our study period, it is unlikely that they have affected the observed time trends. Moreover, noting that huge quantities of eel have been caught in the traps above these barriers, and that the size of those eels did not deviate from expectations, it is rather unlikely that those lower barriers affected the absolute number of eels either. Most likely, the trap location was chosen exactly because of the local abundance of eels, that is: because the lower barriers did not affect the migration further upstream very much.

For Mörrumsån, records do not always indicate at which of the four dams in this river the eel was collected, or records indicate that catches from different traps were merged. The different traps in the river Mörrumsån vary in distance upstream 21 - 32 km; in altitude 78 - 119 m; eel weight varied 63 - 180 gr; corresponding ages are 6 - 9 years. We treated all Mörrumsån data as a single, valid data series, using the characteristics of the trap producing the major share of the catch (Hemsjö övre, 69% of the catch).

For one site, in the River Ätran in-between the rivers Morupsån and Nissan, a long data series is available (since 1932), but reported catches were consistently considered disappointingly low. In 2006, the trap was renewed, and moved to another location in the same river; subsequently, catches increased almost a thousand-fold. Most likely, the earlier trap was not properly placed; because of that, all data until 2006 were flagged as invalid. The new trap was operated from 2006 until 2012, after which the whole dam was removed. The low number of valid observations for this site did not make it worthwhile to include this series in our analysis.

B.1.2 Data and common trend

Summed over the years since 1950 and over all 22 sites, a total of more than 45 million eels have been recorded in the eel traps as valid observations - the largest shares coming from the rivers Lagan (14 million eels, equalling around 14.5 million glass eel equivalents) and Göta älv (50 t, equalling around 7 million glass eel equivalents). Catches peaked in 1953, the sum of all sites reaching 3.3 million and 10.7 t, respectively (around 3.5 million glass eel equivalents). Individual non-zero observations (one site, one year) varied from just one single eel per season (Ljungan 1974, and Nyköpingsån 2016) to almost 1.7 million eels (Viskan 1977) respectively 0.5 t (Göta älv 1953). That is: our data span more than six orders of magnitude.

The absolute number of eels being caught per site varies greatly, and the time trends in these results appears to be erratic. Several sites have reported to have caught not a single eel in specific years, especially in the 1960s and 1970s. As a result, the common trend in these catches is hard to detect from the raw observations (Figure B2).

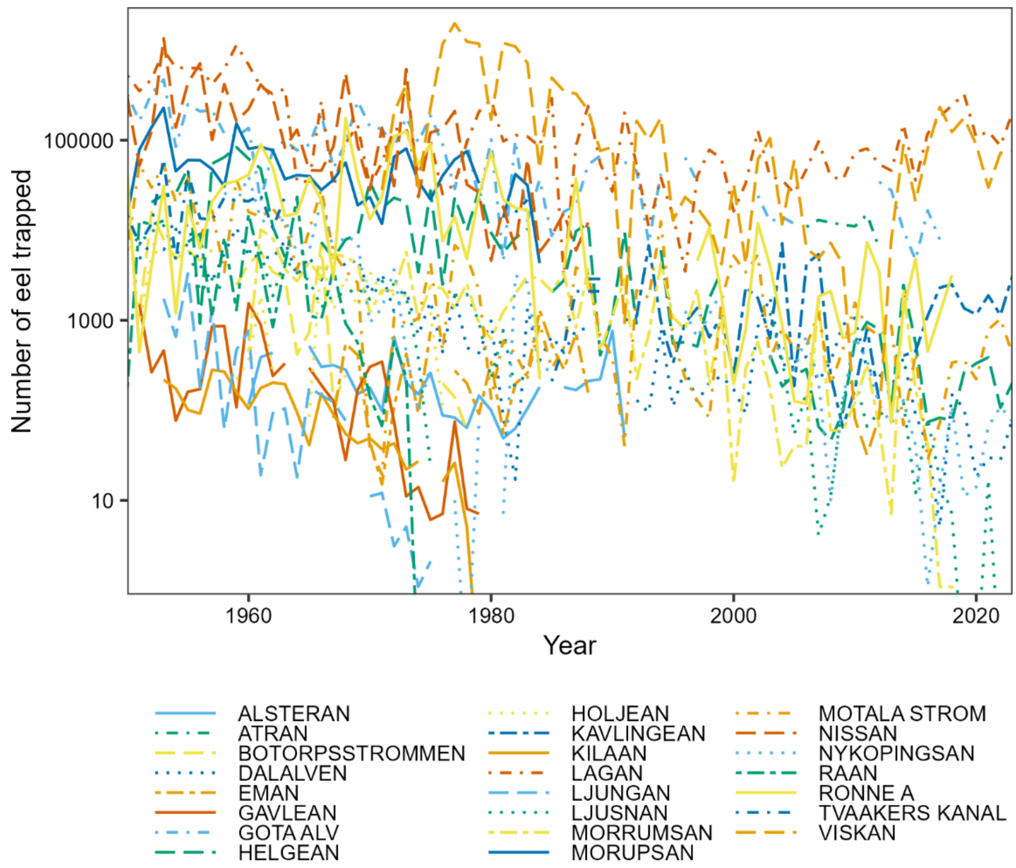

Figure B2: Primary observations of the number of young eel caught in the eel traps, by year and site, during 1950-2023. Observations considered invalid due to non-standard circumstances have been discarded. Where only the weight of the annual catch was reported, these have been converted to the corresponding numbers (see text). Note the logarithmic scale of the vertical axis.

In order to try and visualise a common trend, the time series from each trap site has been standardised to a standard score (observation minus mean, divided by standard deviation), after which the mean of all standard scores was considered to be the common trend (Figure B3). This common trend suggests that the average number of recruiting young eels has been in decline from 1950 until 2010, albeit with a considerable year-to-year variation. Since 2011, the common trend appears to show an increase in recruitment. However, the youngest year classes have only just (or not yet) arrived at the more northern locations (where older eels predominate), so the trend for the youngest year classes remains highly uncertain. Additionally, the decreasing number of available time series could further bias the recent trend.

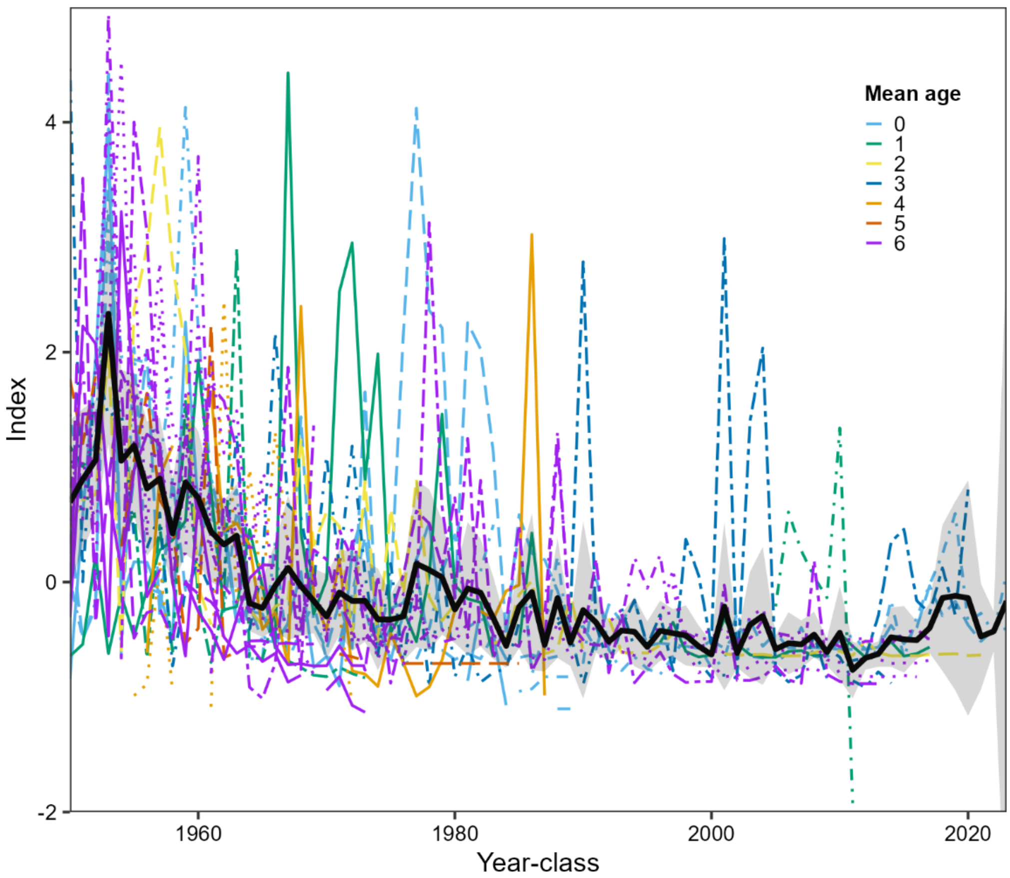

Figure B3: Standardised observations of the number of young eel caught in the eel traps (1950-2023): each data series has been normalised as a standard score (observation minus mean, divided by standard deviation). In addition, the common mean trend has been added (black bold line), including the confidence interval of the mean (grey area). The colour of the lines indicates the mean age of the young eel being caught at each location.

Visualising the common trend for the eel trap catches could give an indication of the general trend of eel recruitment to Swedish inland waters, but it comes with several shortcomings. First of all, the standardisation and subsequent common trend is based on the mean of each trap's time series and the deviation from that mean. Sites that have been active for different time periods over the course of the eel's recruitment decline will have the value of their means, and deviations therefrom, affected by the level of recruitment during the time period of the trap's activity. For instance, a trap active since 1900 will have a high mean value of catches and current catches will be far below this mean, whereas a trap started in 1970 will have fewer years of high recruitment and subsequently catches that will be closer to the mean. Furthermore, recruitment up to the locations of the eel traps will likely be affected by the river discharge, the distance of the trap to the river mouth, as well as the distance of the river mouth to the Skagerrak (the pathway by which the glass eel reach the Swedish coast). Therefore, in order to obtain a more accurate recruitment estimate, eel recruitment to Swedish inland waters will be estimated by a mixed linear model.

B.1.3 Statistical analysis

In the previous assessment (Dekker et al., 2021), recruitment of eel to inland waters was estimated with a generalised linear model. Here, we now opt for a mixed linear model instead, to try and better account for inherent differences between the different sites. The response variable is the number of captured eels at a given eel trap in a given year. To handle zero observation, one eel was added to each observation. Potential explanatory variables to include into the model are river and eel-specific variables:

5. The year class, to which the catch belongs, i.e. the year the observation was made, minus the mean age, rounded to the nearest integer. Observed mean weight (g) in the catch is converted to the corresponding age (years), assuming a length-weight-relation W=a·Lb, where a=0.000559 and b= 3.297428, and a linear growth rate of 4.25 cm per year from the glass eel length of 7.3 cm onwards (parameters matching the means of all our data on inland eel sampling, see also Section C.1.3).
6. The mean age of the eel (in years), derived from the observed mean weight per site, as specified above (age itself was not measured directly).
7. The river's mean annual discharge (in m^3/s), representing the size of the river. Multi-annual average discharge values per river (measured or modelled) were taken from SMHI (2014). We selected the nearest (or otherwise most representative) stream gauges for each trapping site.
8. The shortest distance from Oslo to the river mouth (measured in km), representing the location of the river, (far) outside or (far) inside the Baltic. For each river mouth, distance to Oslo was calculated using the length of the convex hull around the coastline of southern Sweden.
9. The distance upstream between the river mouth and the eel trap (in km), representing the location of the trap within the river, derived from the GIS databases of SMHI (2014).

We examined combinations of these five covariates using model selection techniques and tests for multi-collinearity. To meet the assumption of homoscedasticity of residuals, we transformed the response variable to the log scale. Expecting a proportional relation between the discharge and the amount of eel caught, we subsequently also log-transformed the river discharge covariate. Due to significant nonlinear yearly variation in the number of eels reaching Swedish rivers, the covariate year class was best represented using three polynomials, compared to two and one polynomials, based on the Akaike Information Criterion (AIC). Including interactions with year-class and other variables led to multi collinearity, as did including mean age and distance to Oslo in the same model. Consequently, we fitted the final model without interactions. Lastly, we excluded mean age as an explanatory variable from the model, given that mean age was highly correlated to distance to Oslo, and distance to Oslo was a better predictor of eel numbers. Furthermore, to address non-independence in the data, we included the trap site as a random intercept. The final model estimating number of eel caught for observation i and site j is:

$$\log(\text{number}+1)_{ij} = \beta_0 + \beta_1 \log(\text{discharge})_j + \beta_2 \text{d2mouth}_j + \beta_3 \text{d2Oslo}_j + \beta_4 \text{poly}(\text{yearclass}_{ij}, 3) + u_j + \epsilon_{ij}$$

where β_0 is the fixed intercept, $\text{poly}(\text{yearclass}_{ij}, 3)$ is the third-degree polynomial of year class, u_j is the random intercept for site j, and ϵ_{ij} is the residual error. D2mouth refers to distance of the trap to the river mouth, and d2Oslo refers to the distance from the river mouth to Oslo according to a convex hull around the coastline of Sweden. The model was fitted using the 'lme4' package and the lmer function in R (Bates et al. 2015). Fixed effects were assessed using ANOVA (type II) with Wald F-tests and Kenward-Roger degrees of freedom from the 'car' package (Fox et al. 2012), while random effects were assessed using likelihood ratio tests using the 'lmerTest' package (Kuznetsova et al. 2017). These tests are recommended for mixed effect models (Bolker et al. 2009).

Finally, for each year class, we used the fitted model to generate predictions of eel recruitment numbers to the rivers of Sweden's major catchment areas (Swedish: huvudavrinningsområden). For the rivers included in the original model fit, these predictions were made with the covariate distance-to-mouth set equal to the distance between the given eel trap site and the river mouth. Rivers not included in the original model fit do not have an eel trap, so for these rivers predictions were made with the covariate distance-to-mouth set equal to the distance between the first downstream barrier and the river mouth. Furthermore, for rivers not included in the original model fit, predictions were made by setting the random intercept u_j to zero.

B.2 Results

The type two ANOVA revealed that the fixed effects: log discharge, distance from Oslo, and year-class, were significant at an alpha level of 0.05. Even though distance to the river mouth did not show significance, we deemed this variable necessary to be able to estimate recruitment to a specific location in a given river, and therefore retained it in the model (Table B2). The random intercept for river proved significant as well (Table B3), and was retained in the model.

Looking at the predicted versus observed values of the number of trapped eel (Figure B4), it appears that the model performs reasonably well in predicting number of eel trapped, as no clear biases are visible. This is further reinforced when looking at the residual values versus the fitted values (Figure B5). There is no sign of heteroscedasticity as the residuals appear randomly spread around zero without clear systematic patterns or structures.

Table B2: Results of the type II analysis of variance.

Variable	F	df	Residual df	Sig.
log(Discharge)	28.1	1	19.1	<0.001
Distance to river mouth	3.58	1	18.8	0.074
Distance to Oslo	96.0	1	19.3	<0.001
poly(Year class, 3)	223	3	997	<0.001

Table B3: Output of the backward elimination of random effects from the step function in the 'lmerTest' package, where each row shows model fit statistics after elimination of a given variable from the model. The "Variable" column shows the full model ("none") and the variables tested for elimination. "n par" represents the number of parameters in the model. The "logLik" column presents the model's log-likelihood, while "AIC" refers to the Akaike Information Criterion. "LRT" stands for the likelihood ratio test, "df" denotes the degrees of freedom, and Sig." represents the p-value associated with the likelihood ratio test.

Variable	n par	logLik	AIC	LRT	df	Sig.
none	9	-2057.2	4132.5	-	-	-
(1 | river)	8	-2173.2	4362.5	232	1	<0.001

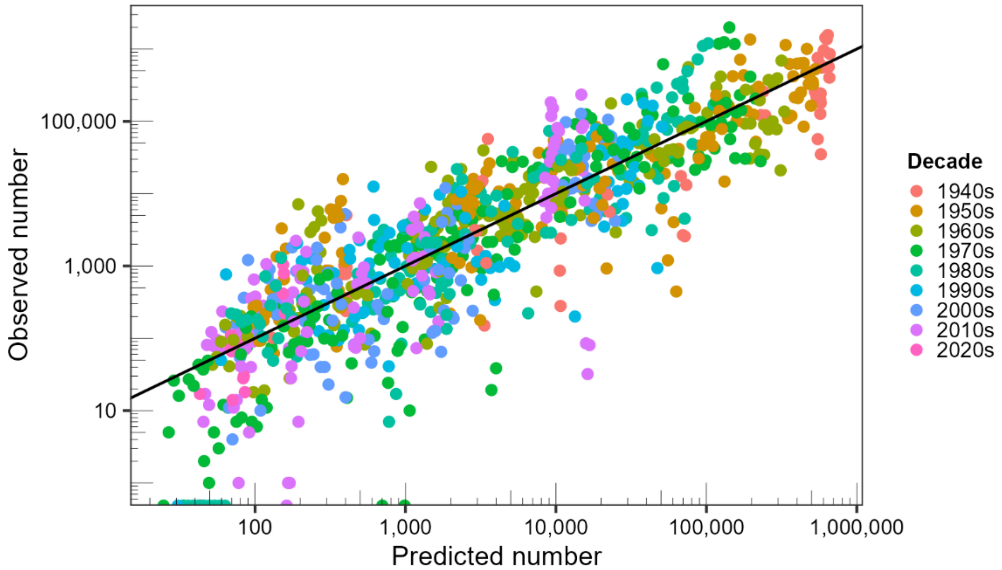

Figure B4: Relation between observed values and values predicted by the statistical model, coloured by decade. The solid line represents the main diagonal, where observed and predicted values are equal.

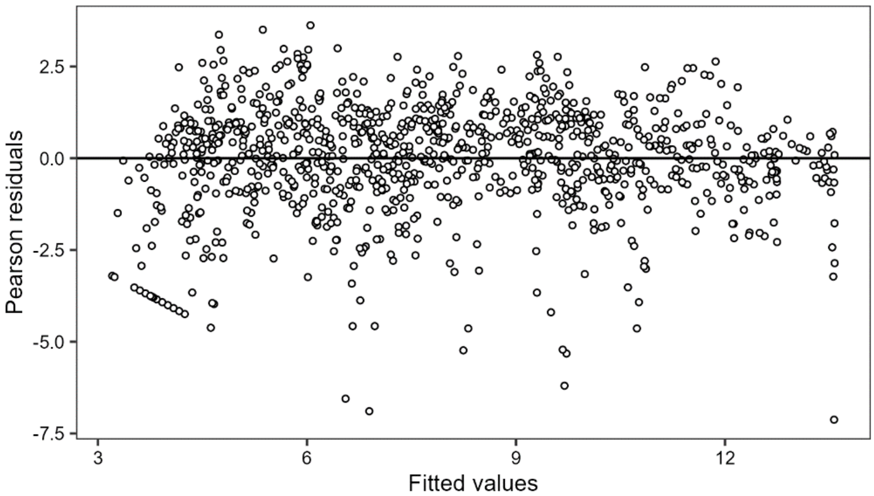

Figure B5: The model's fitted values versus residual values.

To gain insight into each predictor's effect, as well as the model's fit, the working residuals were plotted together with the partial regression against the x-levels of the focal predictor (Figure B6), following the description in Fox & Weisberg (2018). For each predictor, the partial regression was obtained as the fitted model values across the range of the focal predictor while setting the remaining predictors at their mean. Working residuals were calculated by subtracting the fitted model values from the observed values, and added to the partial regression. A well specified model should have the working residuals randomly scattered around the partial regression, which appears to be the case for this model (Figure B6). Additionally, observing the isolated effect of each predictor, the model predicts a decrease in eel catches over time, though the decrease levels off in recent year classes (Figure B6a), a decrease in eel catches with an increasing distance to Oslo (Figure B6b), an increase in eel catches with an increase in river discharge (Figure B6c), and a decrease in eel catches with an increasing distance of the trap to the river mouth (Figure B6d).

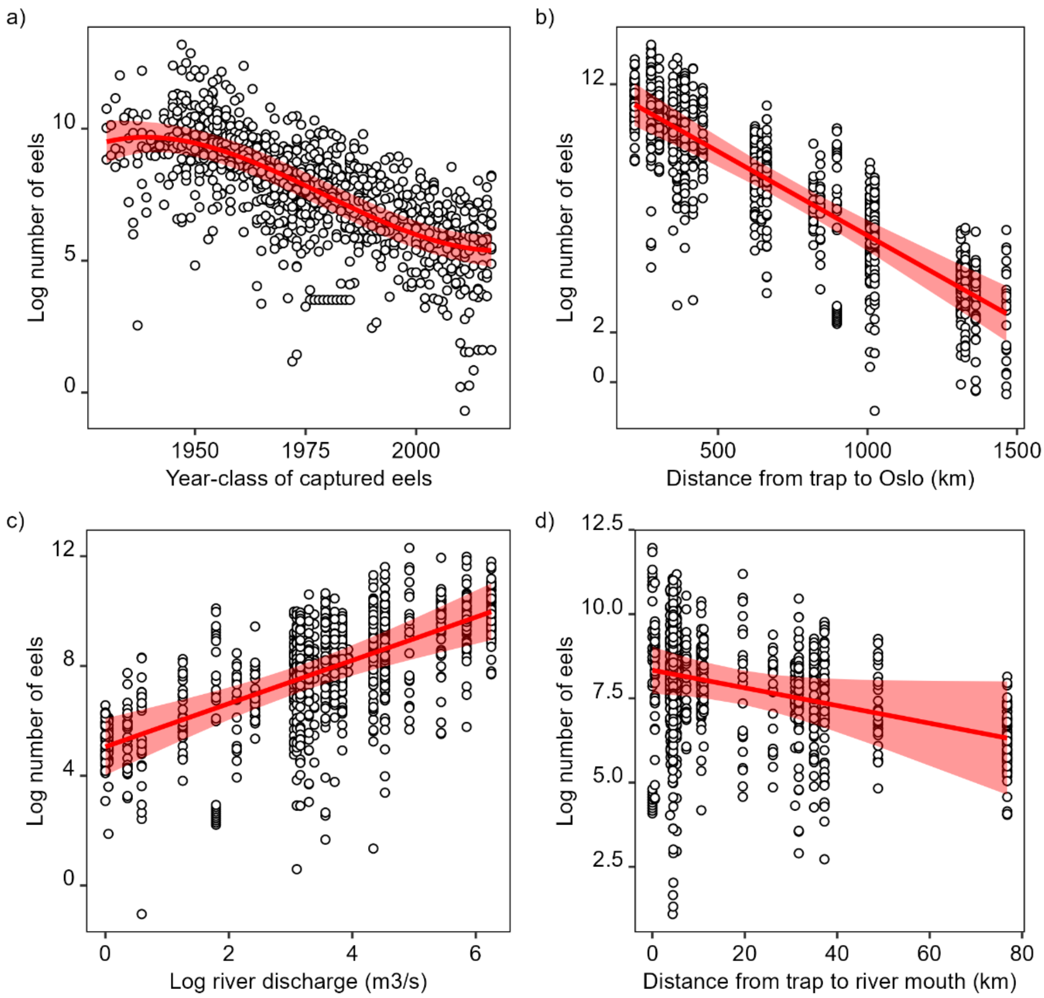

Figure B6: Covariate predictions (red line) and working residuals (points), showing the predicted isolated effect of each predictor, and the deviation of the observed values from this effect. The lines are the fitted values across the range of the focal predictor while setting the remaining predictors at their mean, i.e. partial regressions. The shaded areas are the 95% confidence intervals. Working residuals were calculated by subtracting the fitted values from the observed values, and added to the partial regression

B.3 Extrapolating trends in natural recruitment

The reconstruction of the inland silver eel production (Appendix C) requires (amongst others) estimates of the natural immigration of glass eels, elvers and bootlace eels into all rivers, while monitoring data are available for a very limited number of rivers only, and not for all years. To this end, the model of the spatial and temporal patterns in the elver trap catches, presented above, is used to generate statistical predictions, for all rivers and all past year classes.

In previous assessments of the Swedish eel stock (most recently Dekker et al., 2021), the natural recruitment of eel to Swedish inland waters was determined solely by the statistical model estimates of recruitment, even for year classes where trap data was available. However, for rivers with eel traps, the catches of those traps that were transported upstream as assisted migration would still be subtracted from the natural recruitment figure, resulting in a negative recruitment if the elver trap catches in a given river and year were higher than the statistical estimate of natural recruitment there. Here, we instead opt to use the trap catches as the assumed natural recruitment for those rivers and year classes where trap data is available. We then use the statistical model predictions of recruitment for those rivers and year classes where no trap data is available.

For the model predictions of the most recent years, however, the model is relatively over-specified, as not all recent year classes have been observed yet in all remaining eel traps. The year class of 2017 is the most recent year class that has been observed in all active eel traps, while the most recent year class (2023) has only been observed in Viskan and Lagan so far. Other (more northern) sites tend to catch incoming recruits of an older age, and these sites are therefore expected to catch the most recent year classes (2018-2023) only in the time still coming. To avoid implausible model estimates of recent year classes due to model overspecification, the recruitment estimate of the 2017 year class was used to extrapolate recruitment to more recent and future year classes.

B.4 The potential for using electrofishing data to estimate natural recruitment

As described above, natural recruitment to the inland stock is currently largely based on a statistical model fitted to data from elver traps located at dams in a number of rivers. However, many of the elver traps have stopped operating and it is uncertain to what extent the data collection from elver traps will continue in the future (Appendix B.1.1). Hence, alternative methods of estimating recruitment may be needed in the future.

As part of the European Data Collection Framework (DCF) program, SLU Aqua collects eel recruitment data using standardized electrofishing. 15 sites are electrofished annually and trends in recruitment can thus be followed. This electrofishing data shows a similar pattern as that of the recruitment model: mean eel density per site tends to decrease along a north-south gradient on the Swedish west coast, i.e. density decreases with the distance to Oslo (Figure B7). However, the electrofishing time series can only be used as an index of recruitment as they do not represent absolute numbers of recruiting individuals and are merely standardized indices of the density of recruiting eels in a number of water courses.

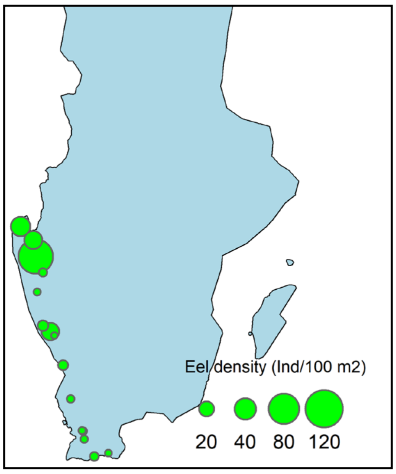

Figure B7: Eel density estimates for 15 sites sampled within the DCF program along the Swedish west coast. Density estimates refer to average eel density for the years being sampled (Arödsån (n=13), Dybäcksån (n=7), Höje å (n=12), Knapabäcken (n=8), Kollerödsbäcken (n=6), Kvarnabäcken (n=11), Kynne älv (n=12), Kävlingeån site 1 (n=11), Kävlingeån site 2 (n=10), Löftaån (n=11), Nybroån (n=7), Rössjöholmsån (n=13), Strömsån (n=13), Viskan (n=2), Örekilsälven (n=12)).

Currently an attempt at estimating natural recruitment using the Swedish national registry of electrofishing (SERS 2024), of which the DCF time series described above is a part, is in progress. To this end, catches from the electrofishing data base (Figure B8), data from the hydrographic network of Sweden (Lantmäteriet 2018), and other data sources related to potential eel habitats (SMHI 2024) are integrated in an analysis referred to as Eel Density Analysis (Jouanin et al. 2012; de Eyto et al. 2016; Briand et al. 2022). The overarching aim with this analysis is to relate response variables such as eel occurrence, eel density and potentially also eel size classes to various predictors such as the distance to Oslo along the Swedish coast, the distance to the sea from the electrofishing site, and year of survey. Estimates from the model are thereafter extrapolated to areas with no data and by multiplying density estimates with areas of wetted surfaces it is possible to derive estimates of eel abundances for a specific watercourse and hence to derive estimates of total recruitment.

As a first step, an eel density model is under development for the wetted areas below the first hydropower dam in each river reach (Figure B9). The reason for fitting a model to data for these areas only is that hydropower dams most likely constitute, if not impassable, at least very difficult to pass migration barriers for eels. Hence, most of the recruitment to Swedish inland waters is likely constrained to wetted areas below the first migration barrier in each water course (Table B4; Figure B9). Moreover, as many glass eels have historically been restocked above migration barriers, and earlier assessments indicate that most of the silver eel production stem from restocking (Dekker et al. 2021), electrofishing results for wetted areas above the first hydropower dam are likely mainly representing local densities of restocked eel and are hence unrelated to recruitment. Preliminary figures also indicate that it is more likely to detect an eel below as compared to above the first hydropower dam (Table B4) despite that there is much more habitat available above as compared to below these hydropower dams (Figure B9).

Table B4: Summary of eel occurrence and eel density estimates in the SERS data base (SERS 2024). Eel detected refers to whether an eel was detected or not during an electro fishing operation, upstream first barrier refers to whether an electrofishing operation was conducted upstream (TRUE) or downstream (FALSE) the most downstream hydropower station in a watercourse, mean density refers to the mean of all density estimates where eels where detected.

Eel detected	Upstream first barrier	Number of electrofishing operations	Mean density (ind/100m^2)	Sd (Density)
FALSE	FALSE	24264		
FALSE	TRUE	51905		
TRUE	FALSE	3938	9.61	25.20
TRUE	TRUE	3329	4.42	11.22

Work will continue toward further developing an eel density model for Swedish waters, and we have good hope that its results will be available to include in the next assessment of 2027.

The references for this Appendix are included in the reference list of the main report, on page 48.

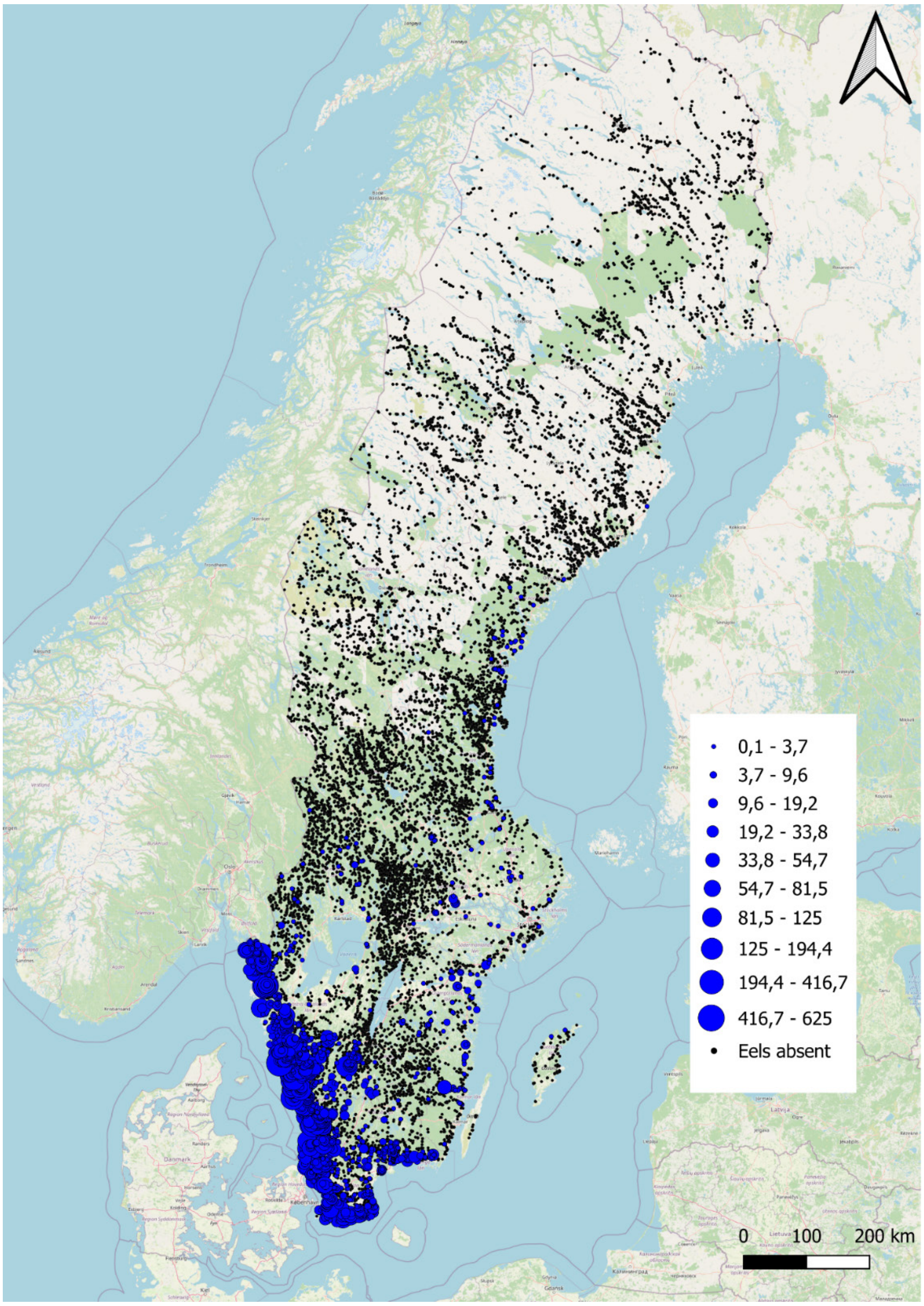

Figure B8: Eel density estimates (ind/100m^2) from the Swedish national registry of electrofishing (SERS 2024). Black circles refer to electrofishing operations where no eels were detected. The size of blue circles is shows the density of eels for all electrofishing operations where eels were detected.

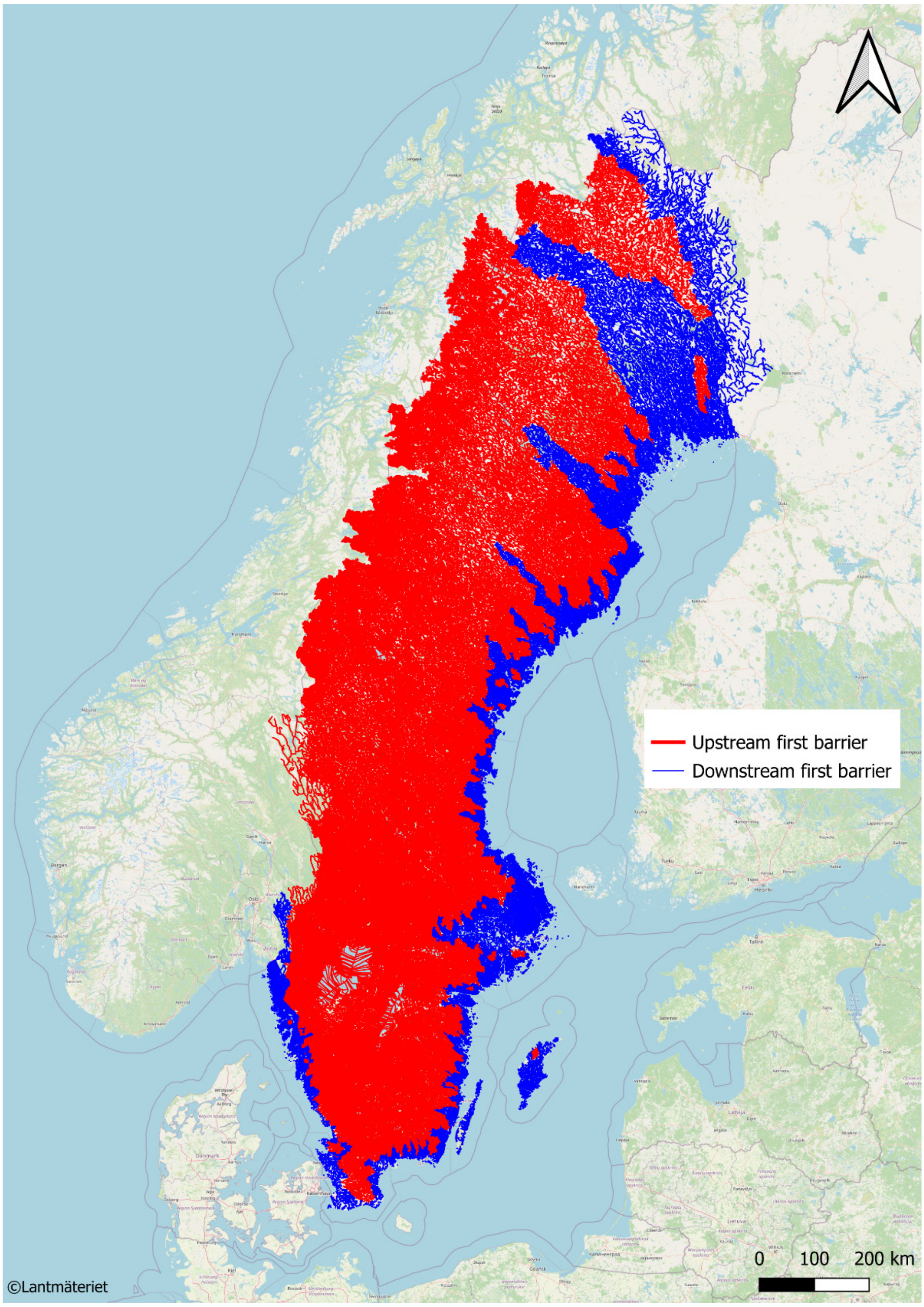

Figure B9: The Swedish river network (Lantmäteriet 2018). Red colored river segments are located above the first hydropower station in each river reach. River segments in blue are located below the first hydropower station in each river reach and are hence potential habitats available to naturally recruiting eels.

Appendix C: Reconstruction of the inland stock

In Swedish inland waters, most anthropogenic interactions with the eel stock happen to relate to either the youngest (glass eel, elvers and bootlace eel) or the oldest stages (silver eel, or yellow eel close to the silver eel stage) – impacts during the long growing stage are much more infrequent. Developing a simple conversion between the youngest and the oldest stages, the silver eel production over the past seven decades is reconstructed, taking into account natural recruitment, assisted migration (within-river transport), and restocking (import from abroad), in a spatially explicit reconstruction. Subtracting the fishing harvest and down-sizing for the mortality incurred when passing hydropower stations, an estimate of the biomass of silver eel escaping to the sea is derived.

A reconstruction of the silver eel production from historical data on their youngest ages, requires an extrapolation over many years, assumptions on growth and mortality, and a comparison between reconstructed (production) and actually observed (landings) variables. Though this makes the best use of the available information, we cannot pretend that the results will be fully accurate in all detail. Production estimates for individual lakes in specific years will certainly be much less reliable than nation-wide estimates, or decadal averages, and so forth. Hence, the presentation of results will be largely restricted to nation-wide averages and/or decadal means.

C.1 Data and methods

The reconstruction is based on a) historical time series on natural immigration of young eel, assisted migration and restocking (‘inputs’ to the inland stock), b) historical time series on fishing yield and hydropower plant construction (‘outputs’ from the inland stock) and c) the conversion from young eel to silver eel (from input to output).

C.1.1 Inputs to the inland stock

There are three sources of young eels in Sweden: natural immigration, assisted migration (man-made transport within river systems), and restocking (imports from abroad, or from the coast). In this section, these data will be presented with regard to their spatial and temporal patterns.

The size of the young eels in the assisted migration and restocking varies from young-of-the-year (glass eel and newly pigmented elver), to on average five-to-seven year old bootlace eels (ca. 40 cm length, 100 gr individual weight). In order to facilitate temporal and spatial comparisons, all quantities of young eels have been converted to number of glass eel equivalents:

$$GEE_{y-a} = N_{y,a} \exp^{Ma}$$

where GEE is the number of glass eel equivalents of year class $y - a$, y is the year the observation was made, a is the mean age of the eels, N is the number of eels of age a and observed in year y that is to be converted to glass eel equivalents, and M is the assumed natural mortality rate between the glass eel and the immigrating stage. For M, an average value of 0.10 yr^{-1} was assumed (the same value as used in the remainder of the analysis; when testing different values of M, the conversion to glass eel equivalents was adapted accordingly). This standardises all data sources of young eel of different sizes/ages on the same units of numbers of glass eel equivalents.

In addition to the three sources of young eel, fully grown silver eels are released into outdoor waters within the framework of a Trap & Transport programme, in which silver eels are caught above a migration obstacle (hydropower generation plant), transported downstream (sometimes directly to the sea, sometimes below the lowest hydropower station in the river), and released. The Trap & Transport programme is considered here as two separate events: the initial catch (interpreted as a normal fishery, a withdrawal from the stock) and the final release (an addition of silver eel to the stock). The release most often takes place in the lower river stretch, or on the coast nearby. Because of the strong link of the Trap & Transport programme to the management of the inland stock, the coastal releases are included here in the inland assessment. Hence, here the Trap & Transport catches are considered a sink of eel for the inland stock, and the Trap & Transport releases are considered a source of eel for the inland stock.

Natural recruitment

The output of Appendix B is an estimate of the number of natural recruits arriving at the first dam in each river each year, for 60 main rivers south of 62.5°N (Indalsälven) and all years since 1940. For an additional 35 (smaller) rivers where no dam is found (4 % of total drainage area, 3 % of total discharge), no prediction could be made (that would have required a consistent extrapolation beyond the range of observations, towards the river mouth). None of these smaller rivers has been restocked, or has a fishery or hydropower stations. Thus, these smaller rivers hardly interfere with the reconstruction in this Appendix. Noting that total production of silver eels derived from natural recruits and assisted migration for most recent years is estimated at approx. 30 t. (see below), ignoring these smaller rivers introduces a bias of approximately 3 % of 30 t. ≈ 1 t. only.

For the rivers *with* an elver trap, in previous assessments (most recently Dekker et al., 2021), natural recruitment was estimated solely by the statistical prediction, not by the actual observation in the elver trap. Here, we instead opt to use the trap catches as the assumed natural recruitment for those rivers and year classes where trap data is available. We then use the statistical model predictions of recruitment for those rivers and year classes where no trap data is available, to arrive at a time trend of estimated natural recruitment to Swedish inland waters (Figure C1, Figure C2).

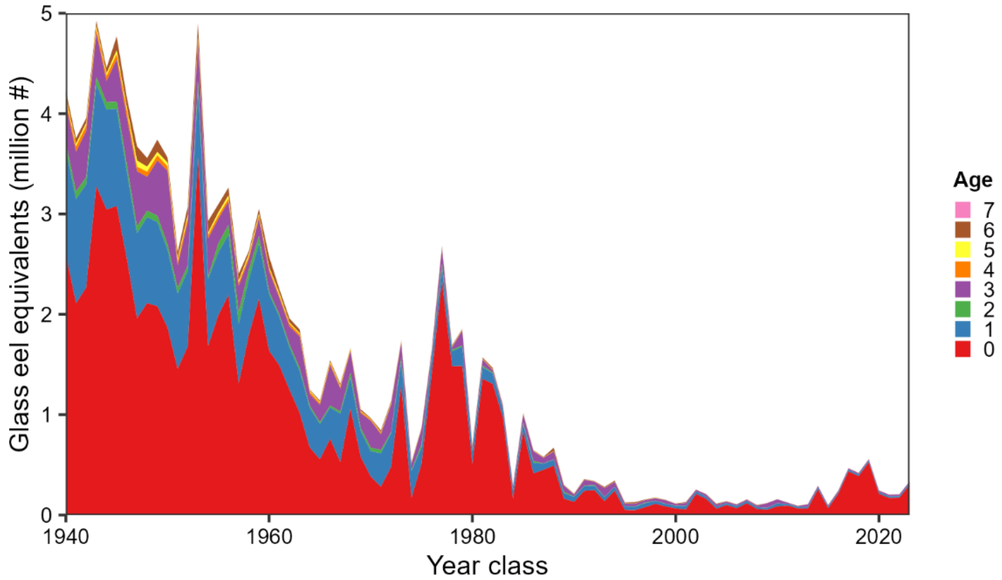

Figure C1: Time trend in the estimated number of <u>*naturally recruiting*</u> *eels. Though this plot is subdivided by age of the eel, all quantities are expressed in glass eel equivalents per year class.*

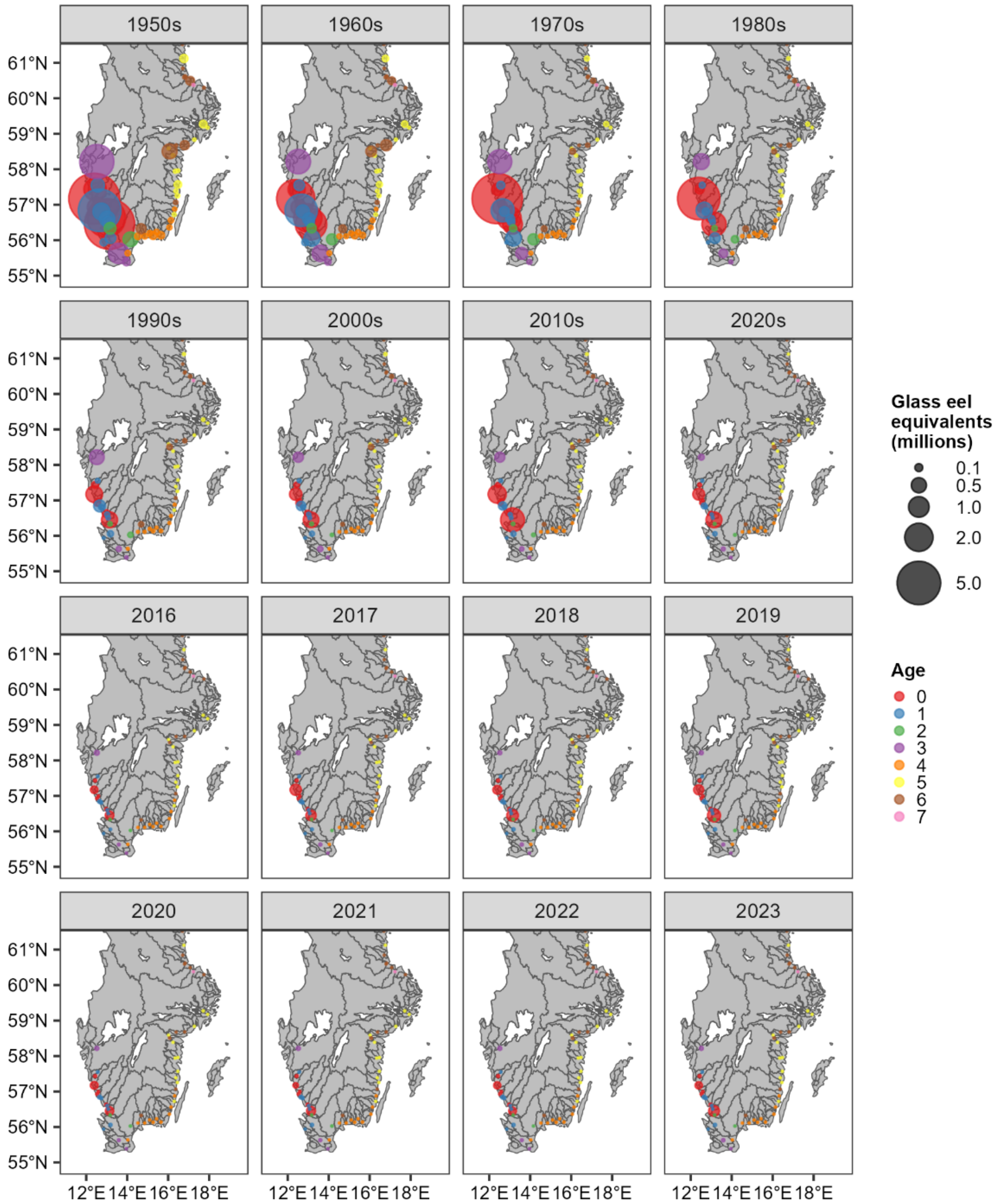

Figure C2: Spatial distribution of the estimates of *<u>natural recruitment</u>*, *per decade (top two rows) and per recent year (bottom two rows, 2016-2023), expressed in glass eel equivalents. These plots show the total number per decade and per recent year (as predicted in Appendix B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the immigration took place, not by year class.*

Assisted migration

A database of historical transports of young eels upstream barriers in rivers is held at SLU-Aqua, specifying site, year, and quantity caught per year (number and/or biomass). When only the biomass of the eel was recorded but not the number, the biomass was converted into numbers using the mean individual weight as observed in other years at the same location. Additionally, an estimate of the mean age of the immigrating eel was derived from the observed mean weight, the length-weight relation and the growth rate (Section C.1.3).

Trapping of young eels was (and is) related to Water Court decisions, obliging anyone obstructing the free migration route to trap and release the eel upstream. For most sites, an explicit redistribution plan is available (though often partly or completely out of practice now), specifying what percentage is released at which location (latitude/longitude and name of lake/river) – often, releases were proportional to the upstream habitat area in each tributary. For Trollhättan, in the river Göta Älv, the releases were also included in the database on restocking, because these eels were not only released within the Göta Älv drainage, but in other river systems too.

Data series from 24 different trap locations are available, and releases from these traps have been made at more than 160 locations. Individual data series start in-between 1900 (river Göta Älv, though the operation of the trap started earlier) and 1991 (River Kävlingeån) and stop in-between 1975 (River Ljungan) and today (8 series continue). Both the trapping (removal from the stock) and the release (addition to the stock) were included in the assessment, as two separate events (Figure C3, Figure C4).

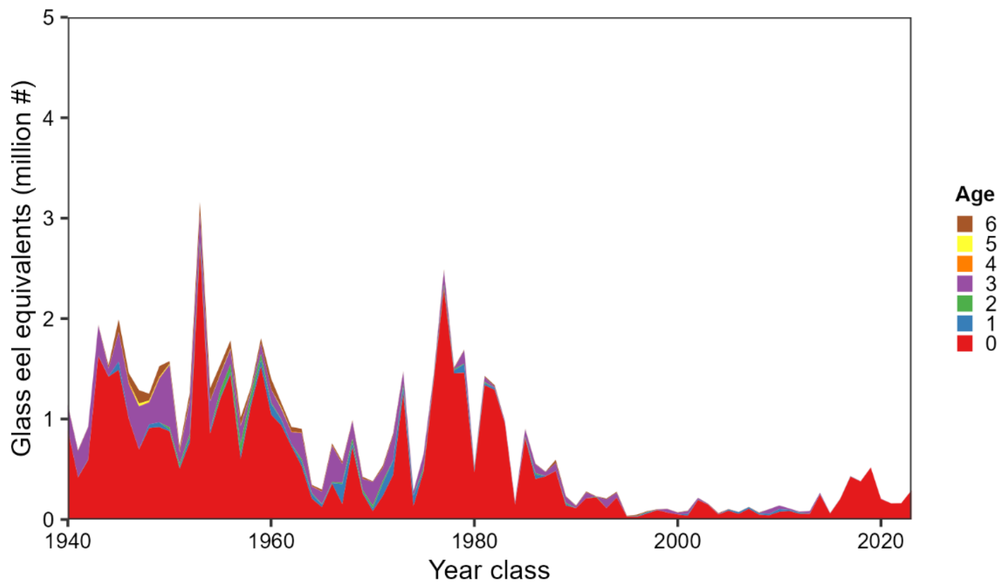

Figure C3: Time trend in the number of eels released from assisted migration. Though this plot is subdivided by age of the eel, all quantities are expressed in glass eel equivalents per year class.

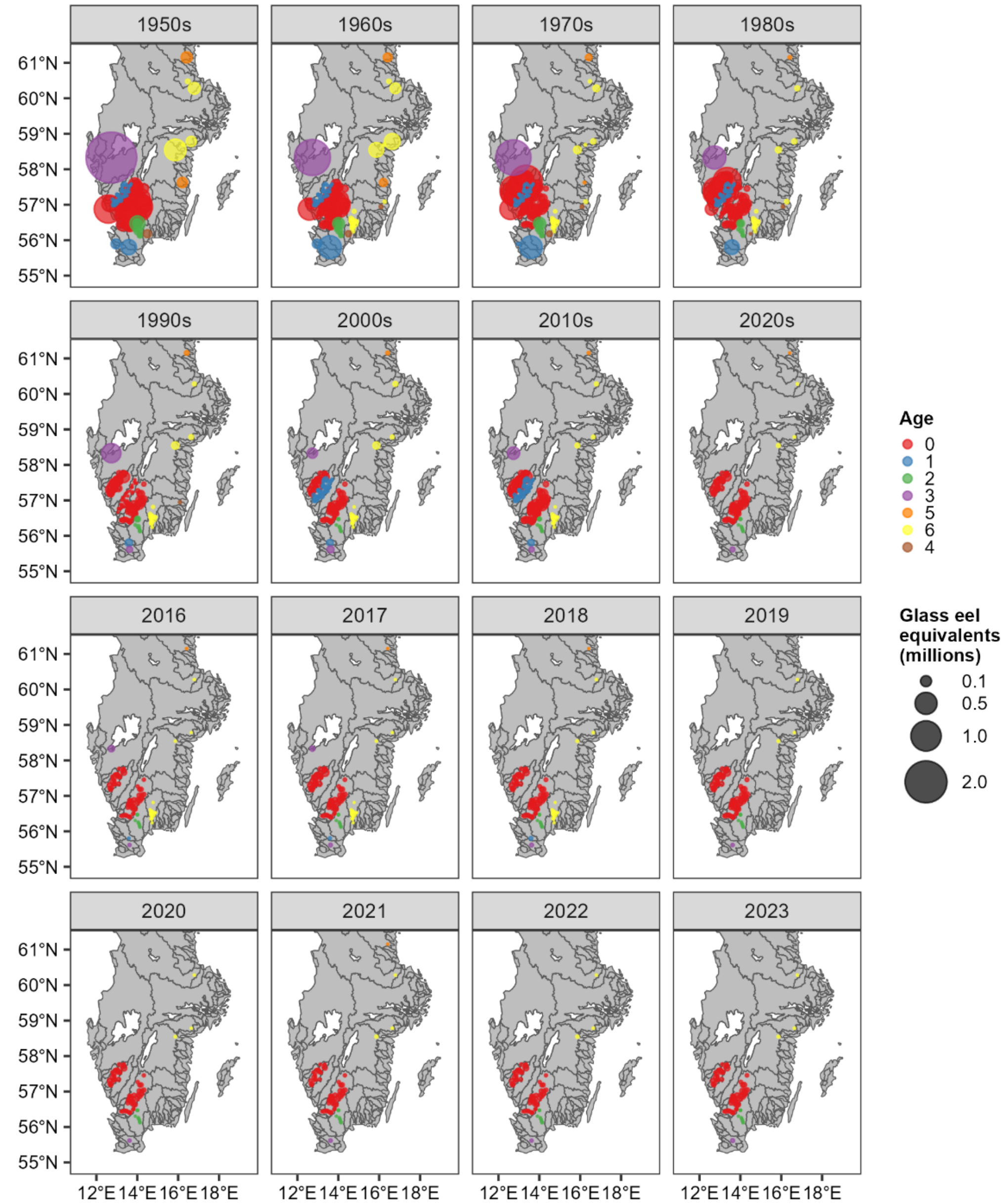

Figure C4: Spatial distribution of the release from *<u>assisted migration</u>**, per decade (top two rows) and per recent year (bottom two rows, 2016-2023), expressed in glass eel equivalents. These plots show the total number per decade and per recent year. Note that the figures are sorted by the year in which the release took place, not by year class.*

Restocking

A data base of eel restocking data is held at SLU Aqua, specifying year, quantity (number), life stage (glass eel, elvers, bootlace), origin (national sources in detail, or international source country), and destination location (latitude/longitude as well as name of the lake/river). The data series start in the early 1900s - that is the start of the restocking in Sweden - and run continuously until present. In total, over 500 different locations in 70 different catchment areas have been restocked (Figure C5, Figure C6).

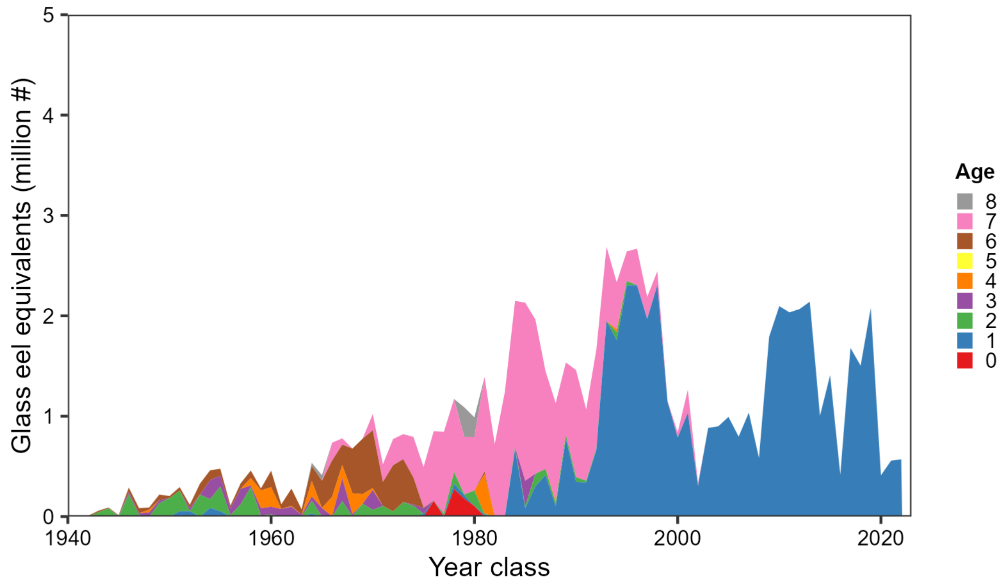

Figure C5: Time trend in the numbers of eel used for <u>*restocking*</u> *in inland waters. Though this plot is subdivided by age of the restocking material, all quantities are expressed in glass eel equivalents per year class.*

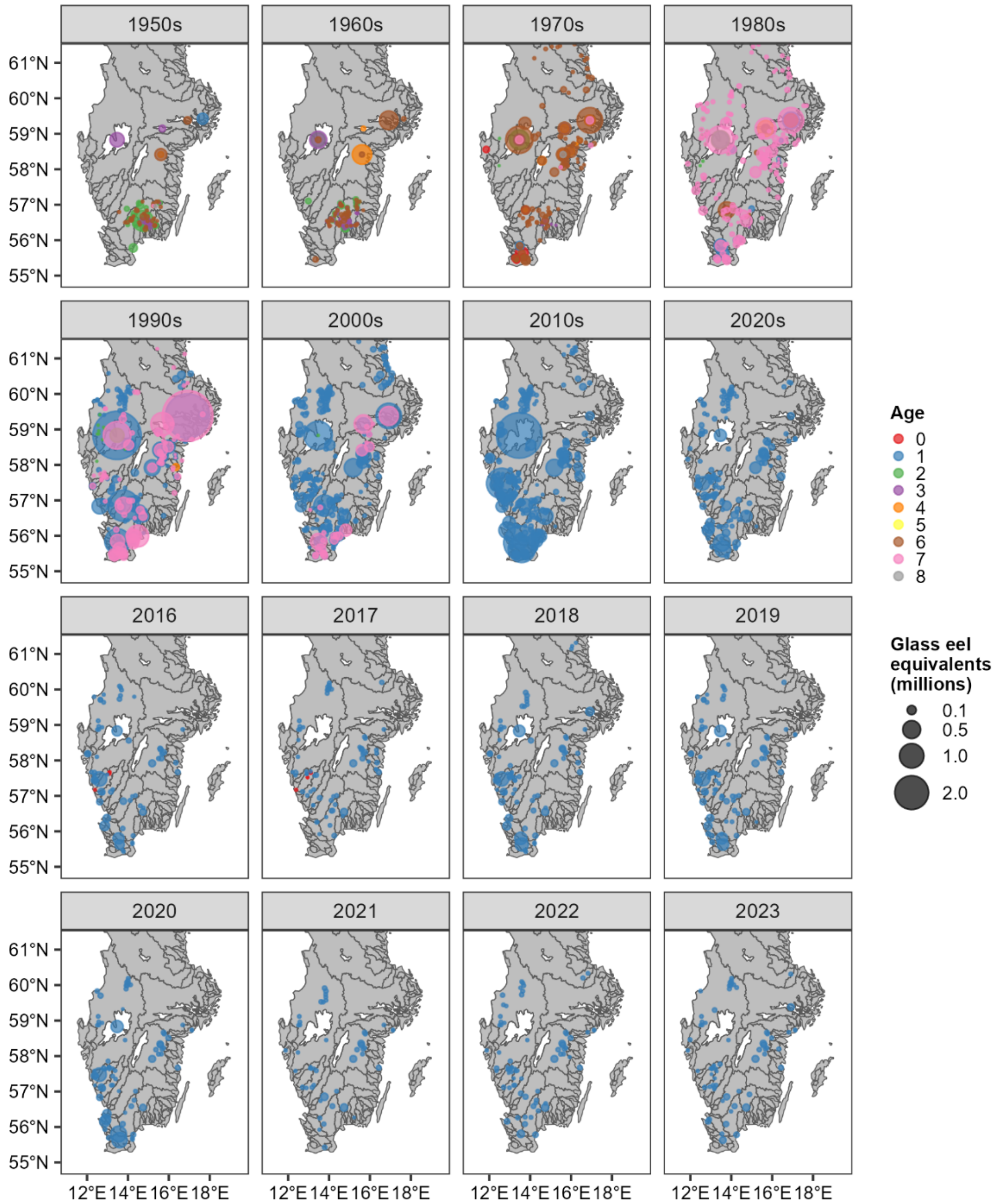

Figure C6: Spatial distribution of the restocking per decade (top two rows) and per recent year (bottom two rows, 2016-2023), expressed in glass eel equivalents. These plots show the total number per decade and per recent year. Note that these figures are sorted by the year in which the restocking actually took place, not by year class.

Trap & Transport of silver eel

In recent years, silver eel from lakes situated above hydropower generation plants have been trapped and transported downstream by lorry, bypassing the hydropower-related mortality (Figure C7, Figure C8). These transports have been organized cooperatively by the government, the energy companies, and the fishers involved. Data on quantity of silver eel, trapping location and release location, date, and details on samples from the catch were available.

The initial catch of silver eel for this programme conforms to a normal fishery, but since the eel are released again later, this fishery is not included further below in the presented commercial landings and estimated fishing mortality. The downstream release of silver eel often occurs just outside the inland area considered in this reconstruction. However, noting the inland origin of these eels, and the involvement of inland fishers and inland operating energy companies, the Trap & Transport programme is included in the current inland assessment, though results are reported separately from the silver eel escaping directly from the inland waters to the sea.

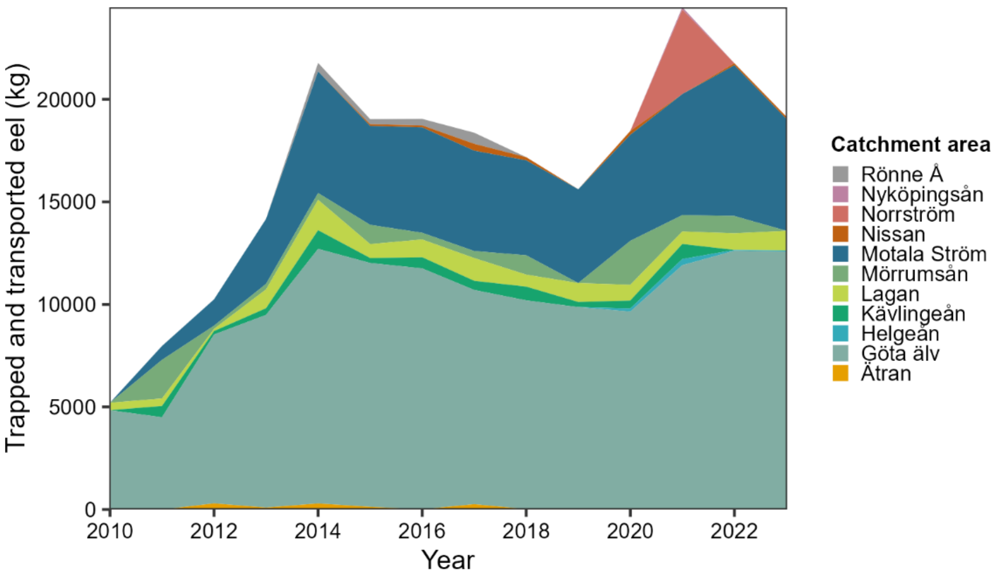

Figure C7: Quantities of silver eel in the Trap & Transport programmes, in biomass (kg), shown per catchment area.

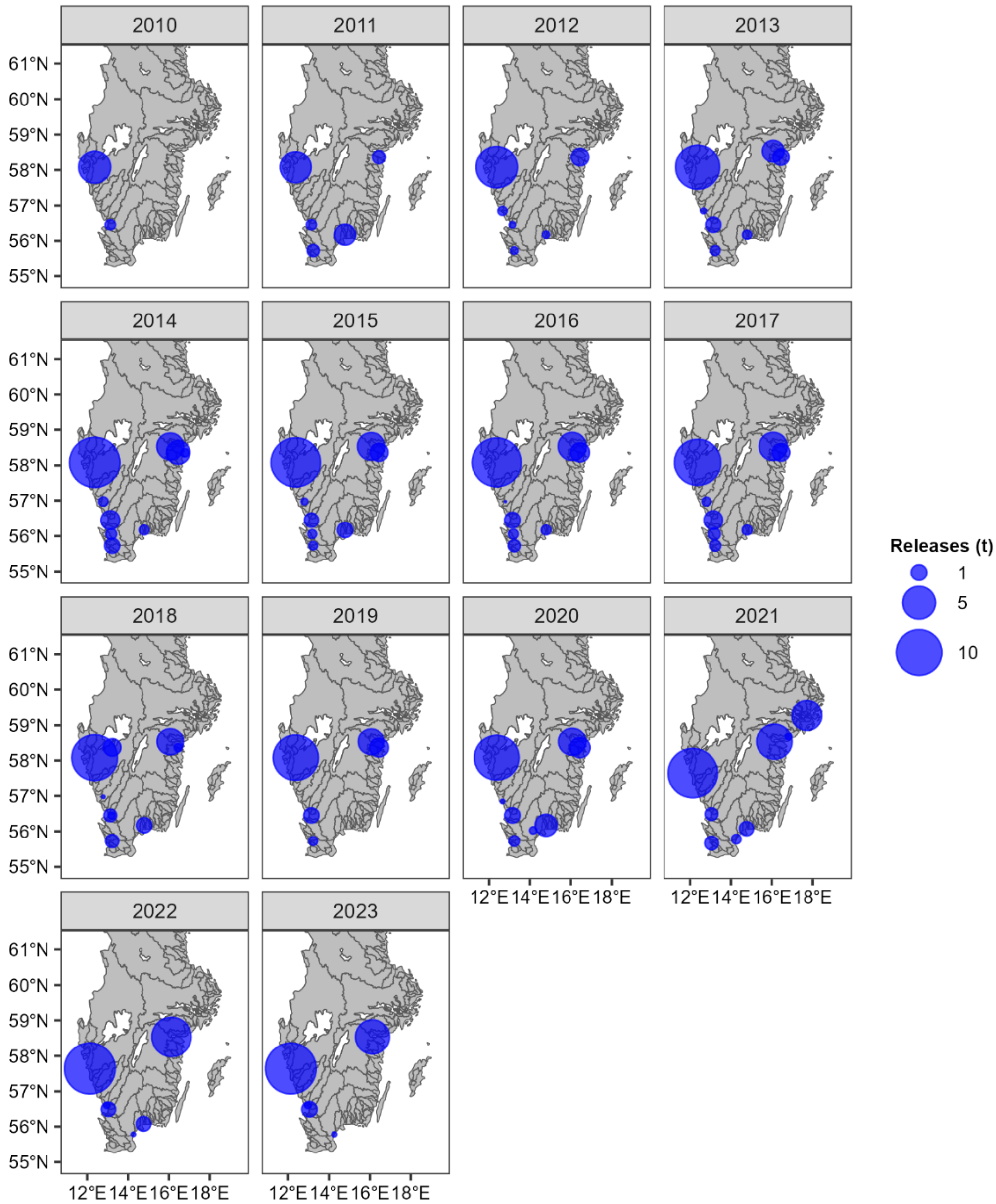

Figure C8: Spatial distribution of the releases from the Trap & Transport programmes, in the years 2010-2023.

C.1.2 Outputs from the inland stock

Fisheries

Statistics of catch and landings have been kept since the late 1800s, but the time series are far from complete, and the reporting system has changed many times. The Swedish Board of Fisheries (Fiskeriverket, now Havs- och Vattenmyndigheten) and SCB have kept databases of annual landings, sometimes based on (daily) logbook registrations, but more often on monthly or annual reporting by individual fishers.

For the larger lakes (Mälaren, Hjälmaren and Vänern), continuous data series exist since the early 1960s, and these series are considered to be complete and reliable. Up to the 1980s, reported landings from the larger lakes were relatively low, after which reported landings increased considerably (Figure C9). Elsewhere, data are available per lake and/or for varying groups of lakes (Figure C10, Figure C11). For the current assessment, historical records were merged into the smallest sets of lakes that allowed unique assignment of all data (e.g., if, in some years, landings were recorded for lake A and lake B separately, but in other years they were merged, we merged the data for those lakes in all years). Only two sets of lakes could not be assigned to a unique drainage area; these have been arbitrarily assigned to the biggest lakes within each set. This concerns: the grouping of Hammarsjön (biggest), Råbelovssjön (both Helgeån drainage), Ivosjön, Levrasjön and Oppmannasjön (all three Skräbeån drainage), respectively Krageholmssjön (biggest), Skönadalssjön (both draining into Svartån, in-between Nybroån and Segeån), Ellestadssjön, Hackebergasjön, Snogeholmssjön and Sövdesjön (all four Kävlingeån drainage).

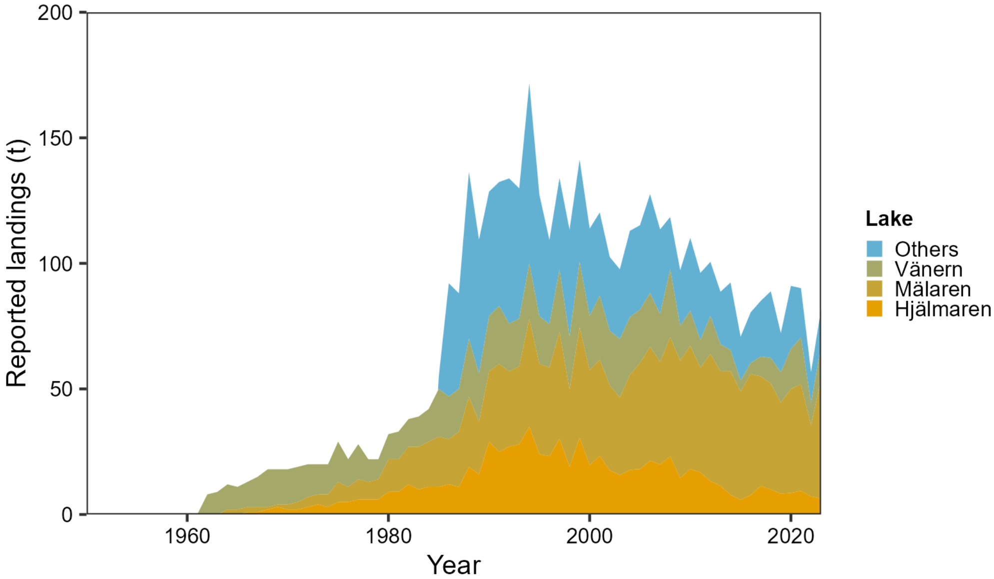

Figure C9: Time trend in the reported landings from the inland fishery, for the larger lakes, and years since 1962. For smaller lakes, no data are available before 1986.

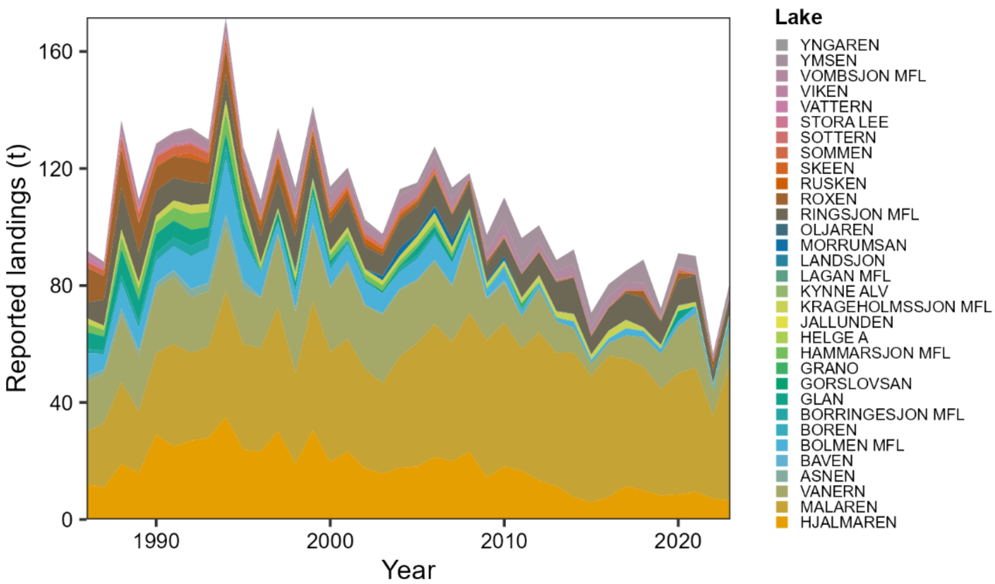

Figure C10: Time trend in the reported landings from the fishery, for all lakes and years since 1986. For the years 1986-1995, landings of the smaller lakes (all lakes except Vänern, Mälaren, and Hjälmaren) are not available per lake, but aggregated as the total landings of all smaller lakes. Therefore, for the years 1986-1995, the landings of the smaller lakes have been reconstructed based on that year's share of each lake's estimated silver eel production vs total silver eel production.

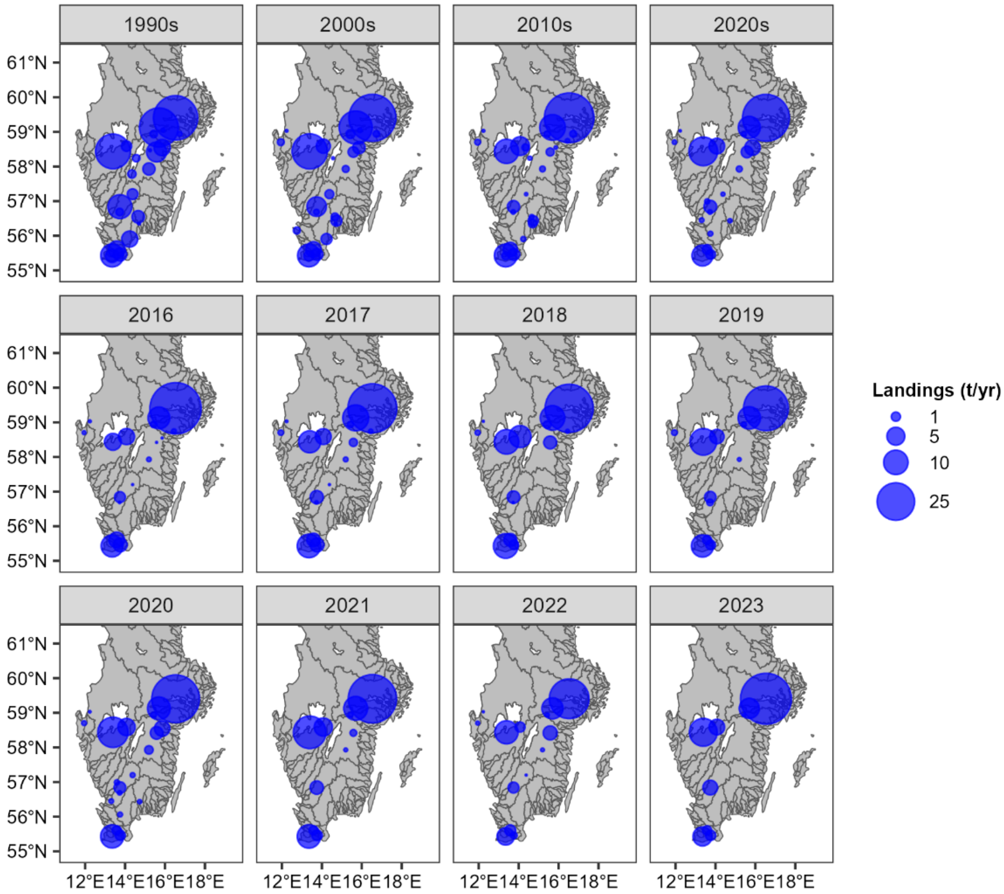

Figure C11: Spatial distribution of the reported <u>landings from fisheries</u>, shown as the annual average per decade (top row), the annual total for recent years (bottom two rows, 2016-2023). For the decades preceding the 1990s, insufficient information is available.

For the years 1986 to 1995, the available commercial fisheries landings data consist of the total landings for all smaller lakes combined, and includes the three largest lakes separately (Mälaren, Hjälmaren and Vänern). For all smaller lakes in this range of years, the landings per individual lake have been reconstructed from the annual totals, on the assumption that fishing impact has been constant across the lakes (though it could vary from year to year). If fishing impact is constant across lakes, the catch will be proportional to the production of silver eel, as in:

$$Catch_{lake,year} = Catch_{total,year} \times \frac{Production_{lake,year}}{Production_{total,year}}$$

for each lake and year. The current assessment reconstructs the production of silver eel available to the fishery by lake and year, from information on natural recruitment, restocking and assisted migration. For the eel derived from restocking or assisted migration, the release location is known (latitude/longitude as well as lake name); it is assumed that within-river migration has not notably altered the spatial distribution – or more often, that downstream migration in the silver eel stage brought the eel back to the lake from which it had migrated upstream after release so many years ago. Downstream migration in the yellow eel stage is unlikely, noting that most lakes have a barrier directly downstream (regleringsdamm). Release (restocked eel or assisted migration) directly into a river occurred less frequently, and those eels have been assumed to have remained in the river, outside reach of the lake fisheries. River fisheries have been abundant in old times, especially using weirs ("lanefiske") across rivers to catch the emigrating silver eel; the only remaining one (at Havbältan in Mörrumsån) is included in our data as a special fishery of minor magnitude.

Catch reporting

Inspection of the landings data raises doubts on the quality of the available information. For several lakes (e.g., Båven, Glan, Roxen, Rusken, Sommen, Sottern) years with and without reported landings alternated (in the 1990s and 2000s). For other lakes, years with and without reported landings for individual fishers alternated (not shown), while the licensing system required continuous operation. Personal communication to individual fishers almost invariably yielded more consistent information, higher landings figures. The reliability of the historical data series is therefore not beyond doubt.

Additionally, the Trap & Transport programme for silver eel has complicated the statistics considerably. Essentially, the Trap & Transport consists of a fishery, a transport and a release. The initial fishery removes silver eels from the local stock, as all fisheries do. The licensing of and the statistics on this fishery are sometimes covered by the conventional fishery system, sometimes registered separately. Completing and correcting the fishery data for this programme has been done to the best of our ability, but requires a disproportional amount of effort, with the possibility that errors still remain. It is therefore recommended that fishers should be able to report the catch of eel destined for Trap & Transport separately.

Until 1998, information was collected by regional fisheries officers (fiskerikonsulenter, länsstyrelsen) in direct contact to individual fishers, most often on an annual basis. Since 1999, this was replaced by a system of obligatory reporting by individual fishers directly to the Swedish Board of Fisheries, now to the Swedish Agency for Marine and Water Management, mostly on a monthly basis. The switch in 1999 from annual reports by region, to monthly reports to a national agency, appears to have come with a loss of quality, i.e. the geographical scale, rather than the frequency of reporting introduced the quality problems.

Impact of hydropower generation

A database of hydropower generation plants was made available by Kuhlin (2024), documenting location and year of construction (Figure C12). Detailed information on ownership, turbine types and capacity were available but not used. Details on local river characteristics (channel size, discharge) were not available. Of the 1501 hydropower stations listed by Kuhlin (2024), 539 stations are relevant for the current reconstruction, as the assisted migration and restocking release data indicates that these stations have or have had eel occurring upstream.

The mortality of eel passing a hydropower station in Sweden is not well known. Calles & Christiansson (2012) list an evidence-based estimate of mortality for 15 stations. Leonardsson (2012) developed a simulation model for the passage of turbines, relating the mortality to the turbine type and local river characteristics. Calles and Christianson (2012) applied this simulation model to a total of 56 stations (see Figure C13, our plotting of their data). While the simulation almost

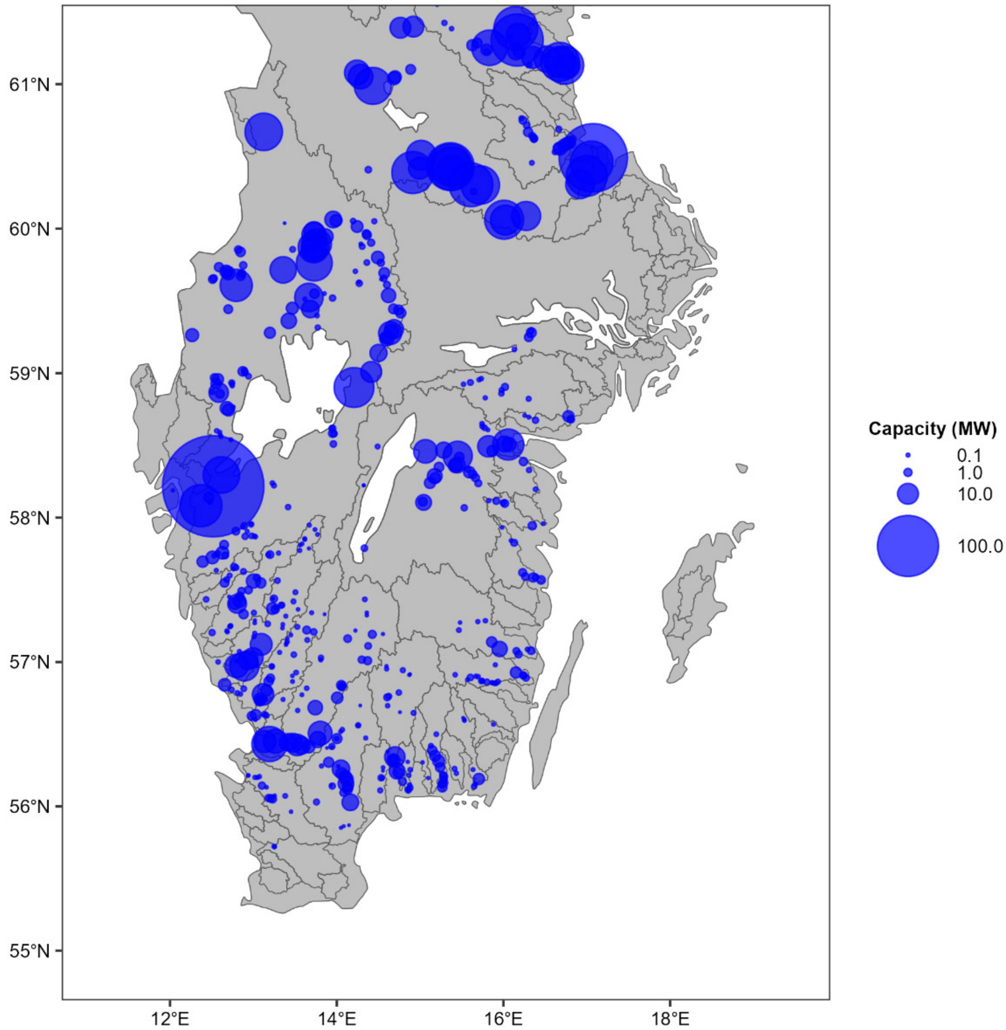

Figure C12: Spatial distribution of the 539 hydropower generation plants having an eel stock upstream, as indicated by assisted migration and restocking release data. The size of the symbols in this figure is proportional to the capacity of each station.

systematically underestimates the mortality in the observed cases (mean mortality: observed=43%, simulated=31%, R^2=0.46, 12 out of 15 cases have observed>simulated), the simulated mortality for the unobserved stations was substantially higher than for the observed stations (mean of simulated mortality: unobserved stations = 56%, observed stations = 31%). That indicates that observations have been made preferably at locations where the simulation happens to predict a low mortality - most likely: observations have been made at locations where the actual mortality is indeed below average. Rather than valuing and correcting for this bias, Dekker (2015) explored a range of options for the hydropower-related mortality. The Swedish Eel Management Plan (Anonymous 2008) assumed a standard mortality of 70% for all hydropower stations, irrespective of turbine type or river characteristics, which is higher than the mean observed and simulated. The observations and simulations discussed above suggest a much lower value, as low as 31%. Dekker (2015) explored three options:

a) Constant mortality of 70% (equivalent to an instantaneous mortality rate of H=1.2 per station).

b) Constant mortality of 30% (H=0.35 per station).

c) Best estimates, using either the observed mortality, or the simulated mortality, or a default value of 70% (whichever is available, in order of precedence).

Comparison of the outcome of these three options indicated, that the net results were very close to each other. A major part of the silver eel production (ca. one-third) is derived from areas where no hydropower generation takes place (primarily Mälaren). Another one-third is from areas with four or more hydropower stations, where the number of hydropower stations, more than the mortality per individual station, determines the net impact (i.e. even at a low impact per hydropower station, the accumulated impact of four or more stations is considerable). Of the remaining one-third, a major share is produced in the river Göta älv, where actual mortality estimates have been obtained for all three power stations downstream of lake Vänern. As a consequence, Dekker (2015) concluded that the uncertainty in the value of the hydropower impact per station has very little relevance for the reconstruction of the status of the stock and the assessment of anthropogenic impacts. In the current assessment, only option c (best available information) will be used, that is: the base option of the 2015 assessment.

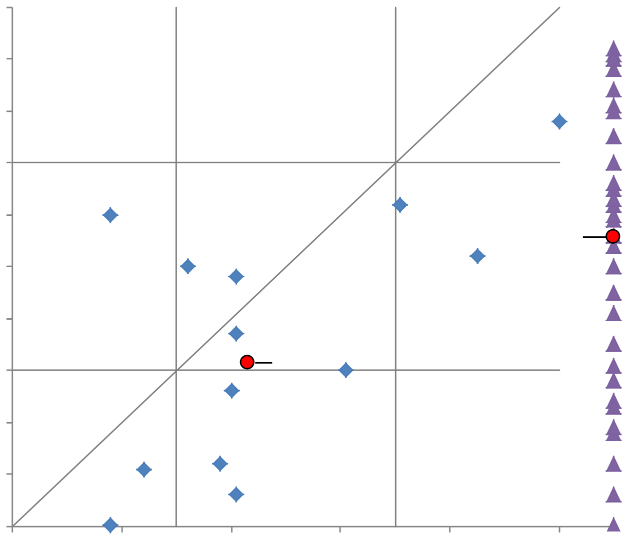

Figure C13: Relation between the observed (horizontal) and simulated (vertical) mortality, for eel passing a hydropower turbine. Data from Calles and Christianson (2012), applying the simulation model of Leonardsson (2012); original plot of data tabulated by the source.

The river network in Sweden is described in detail by the GIS datasets made available by SMHI (2014). For all locations where young eel had recruited or had been released, the route towards the sea was traced and the list of hydropower stations on that route derived. Individual routes pass up to 24 hydropower stations. For each hydropower station, the biomass of the escaping silver eel was reduced by a certain percentage – according to the assumed turbine mortality as specified in the paragraph above – and the biomass reduction was flagged as mortality due to hydropower generation. Summing the biomasses over all hydropower station gives an estimate of the total hydropower related mortality, while the remaining biomass gives an estimate of the escapement towards the sea.

C.1.3 Conversion from recruit to silver eel

Since 2010, samples have been collected from the commercial catch, predominantly from the larger lakes, in the context of the DCF-sampling. These eels have been analysed for length, weight, maturity and age. From 2010 to 2023, a number of 3 460 eels have been analysed in total. Because samples have been taken only in the most recent decade and by far do not cover all river systems, simple relations between variables were assumed; obviously, this is a simplification of reality.

However, noting the high uncertainty in other model parameters (foremost: natural mortality), simple and traceable relations are preferred here.

Growth and length-weight relation

Annual growth in length in the yellow eel stage was calculated as the difference between final length (measured in the silver eel stage) and the glass eel length (fixed at 7.3 cm) divided by the number of years in-between (the age read). The data indicate a large variation in growth rate between lakes, but no systematic relation to latitude or local lake conditions (see also e.g. Myrenås & Jacobson 2024). Noting that we apply growth estimates to all natural recruits, all restocking and all assisted migration, wherever it may have occurred in the past 7 decades, we make the conservative assumption that growth is constant. In conclusion, we apply a constant growth of 4.25 cm/year (the mean of all observations) for all years and sites (Figure C14, red line).

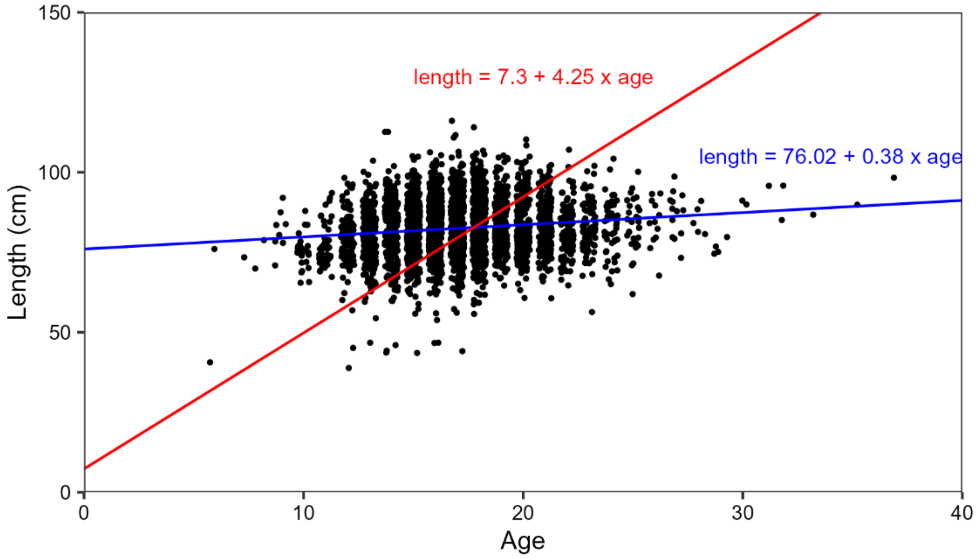

Figure C14: Length and age for 3 460 silver eels, sampled between 2010 and 2023 in eight lakes. To show so many data points, a small jitter has been added to all data points in horizontal direction. Two regression lines are given: a growth-line (red) forced through the length/age of glass eel (7.3 cm at age=0), and an unforced silver-eel-size-line (blue).

Individual weights were calculated as

$$W = a \times L^b$$

where W=weight (g), L=length(cm), *a*=0.000559 and *b*=3.297428.

Silvering

Sampling data indicate a latitudinal trend in mean size at silvering, from approximately 700 mm in the south (56°N) to 900 mm in the north (60°N), but the short-range variation is huge (Dekker et al. 2011, Figure 14). A linear latitudinal trend was consistently applied to all years and locations in the reconstruction to predict mean silvering length, even where sampling had actually taken place:

$$\text{mean silvering length (cm)} = 70 + 5 \times (latitude - 56)$$

At each sampling site, the age of the individual eels ranges from over ten years below, to twenty years above the mean age (Figure C15). In converting recruits into silver eels, the average age-distribution was applied at all sites, taking into account the mean age at each site (which is related to length and - in turn - to latitude).

For the silver eel, the increase in mean length per year of increment in age, on average 0.38 cm/year (Figure C14, blue line), is much less than the mean growth rate during the yellow eel stage of 4.25 cm/year (Figure C14, red line); the silvering process itself appears to be length-selective. The mean observed increment in length with age was applied to calculate length at silvering, taking age relative to the mean age at any site.

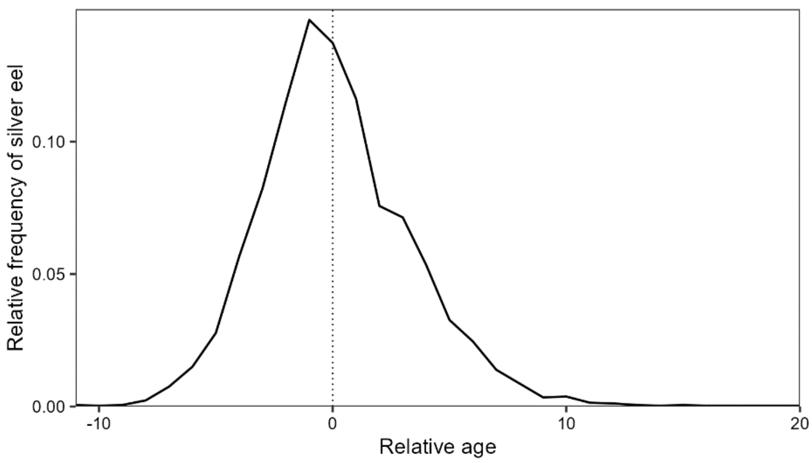

Figure C15: Relative age composition of the commercial landings of silver eel in inland waters, where age is expressed relative to the observed mean age of silver eel in the catch.

Natural mortality

Natural mortality for the inland stock is unknown. A value of M=0.1385 is frequently applied in many studies around Europe, giving Dekker (2000) as a reference – but Dekker (2000) just assumed that value. Bevacqua et al. (2011) performed a meta-analysis, relating reported natural mortality to local stock density, annual average water temperature and individual's body mass. Applied to average conditions in Sweden, their results indicate a mortality of approximately 0.3 yr^{-1} at the glass eel stage, decreasing to 0.015 yr^{-1} at the silver eel size, with a lifetime average of about 0.2 yr^{-1}. Preliminary assessment runs, using a natural mortality rate between 0.1385 and 0.2, however, indicated that the reconstructed eel production would be far less than the actually observed catch, resulting in negative estimates of the size of the silver eel run. Hence, results for a range of plausible values (M=0.05, M=0.10 and M=0.15) were explored by Dekker (2015). Unless otherwise stated, presented results refer to the middle option, M=0.10 yr^{-1}. In addition, section C.2.3 (below) will explore the sensitivity of results towards the assumption on the value of natural mortality.

C.1.4 Estimation of escapement

Given the time series of restocking and assisted migration, and the analysis of the spatial and temporal pattern in natural recruitment, silver eel production is derived from the growth, silvering pattern and natural mortality:

$$Production = f(recruits, growth, mortality, maturation\ )$$

Inspection of the data indicates (Figure C5 on restocking; Figure C9 on fishing yield from the larger lakes) that the more eel has been restocked, the higher the production has been. Therefore, it is unlikely that density dependent growth and/or mortality have been limiting the production. As a consequence, the production from natural recruitment, assisted migration and restocking can be assessed independent of each other and resulting figures be summed afterwards. In fact, individual batches released at any place can remain separate in the assessment in their estimate of silver eel production.

The data sources use different geographical positioning systems (exact latitude/longitude, lake or river name, the sum of smaller lakes) and eels might have moved around during their yellow eel phase. Consequently, the assessment of inputs to and outputs from the stock might not always perfectly match spatially, possibly resulting in local over- or underestimates. Summing results by river drainage area, however, should smooth out any spurious spatial patterns.

At the bottom line, this reconstruction yields an estimate of the quantity of silver eel starting their downstream migration by river and year. The fisheries are targeting this stock of silver eel (or the yellow eel, shortly before they silver), resulting in an effective silver eel run of:

$$Silver\_eel\_run = Production - Catch$$

Passing hydropower generation stations reduces the silver eel run to:

$$Escapement = Silver\_eel\_run \times \exp^{-\sum H}$$

where the hydropower-related mortality $\sum H$ is summed over all hydropower stations on the route towards the sea - which is a different sum for each location (and year) - and *Escapement* is the silver eel biomass escaping towards the sea, on their route towards their spawning area. It is assumed that – other than fisheries and hydropower – no other mortality during the migration towards the sea occurs.

Rearranging the above yields

$$\begin{aligned} Escapement &= (Production - Catch) \times \exp^{-\sum H} \\ &= Production \times \exp^{-\sum H} - Catch \times \exp^{-\sum H} \end{aligned}$$

The latter splits the production data (first term) from the fishery data (latter term) and *post-hoc* sums them up; this allows processing different spatial entities for different data sets (e.g. point-locations for release of recruits versus lake-totals for fisheries).

Recent recruitment/restocking will contribute to the escapement of silver eels for about fifteen years from now, but some slow-growers or late-maturing eels may be found for up to twenty-five years or more. By that time, the stock will be dominated by year-classes that have not yet recruited now, and will be under the influence of management measures taken in coming years. That is: the effect of today's actions can only be assessed by analysing their effect in the future, but future trends are also influenced by yet unknown developments. Not knowing those future trends and developments, the result of today's actions are assessed by extrapolating the status quo indefinitely into the future. It is assumed that coming recruitment is equal to the last observed value (constant numbers; applies to natural recruitment[4], assisted migration and restocking, as well as Trap & Transport of silver eel) and that future fisheries and hydropower generation have an impact equal to the most recent estimate (constant mortality rate). Keeping the status quo unchanged, results for future years will express the expected effect of today's actions, but will not provide an accurate prediction of the real developments (continued upward or downward trends, extra actions, and autonomous developments).

The analysis of recruitment trends (Appendix B) took 1940 as its starting point. Most young eels, which recruited in 1940, will have grown to the silver eel stage before 1960. Hence, results on silver eel (production and destination, mortality) will be presented from 1960 through 2020, with an extrapolation to 2035 to show the fate of the most recent recruits (natural or restocked).

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[4] For natural recruitment, the very last observation year of the recruitment model gives no plausible and reliable estimate, which can be used for extrapolation, because it would relate to the very last year class, which has been observed at very few stations yet. For the natural recruitment, year class 2017 (the last year class observed at all stations, even the more northerly ones, where the eel recruits at an older age) is used as the basis for extrapolation from the recruitment model. This is further discussed in Appendix B.

C.2 Results

C.2.1 Silver eel production

Figure C16 presents an overview of the production of silver eels over time from inland waters, including their source, while Figure C17 presents a spatial overview. This section presents results for the assumption on natural mortality that M=0.10 yr^{-1}. Other options for M will be discussed in section C.2.3 below.

From 1960 until 2023, natural recruitment – including the amount assisted in their migration upstream - is estimated at a total number of 81 million glass eel equivalents, with a minimum of 0.15 million eels in 2009 and a maximum of 5.0 million in 1977. The corresponding total silver eel production from these glass eel equivalents is estimated at 8 674 t, minimum 17 t/yr, maximum 486 t/yr. In 2023, an estimated 0.63 million glass eel equivalents were natural recruits. Total silver eel production from natural recruits (assisted or not) in 2023 is estimated at 24 t.

From 1960 until 2023, a total of 31 million eels have been caught for assisted migration upstream, with a minimum of 0.030 million in 1995 and a maximum of 2.4 million in 1977. The corresponding silver eel production is estimated at 6 030 t, minimum 60 t/yr in 1995, maximum 448 t/yr in 1977. In 2023, 0.28 million eel were assisted upstream. Total silver eel production from assisted migration in 2023 is estimated at nearly 19 t.

From 1960 until 2020, a total number of 76 million glass eel equivalents have been restocked in inland waters, with a minimum of 0.090 million glass eel equivalents in 1967 and a maximum of 3.4 million in 1997. The corresponding silver eel production is estimated at 11 621 t, minimum 17 t/yr in 1960, maximum 411 t/yr in 2011. In 2023, 0.57 million glass eel equivalents have been restocked in inland waters. Total silver eel production in 2023 originating from restocking is estimated at approximately 256 t.

Overall silver eel production declined from 500-700 t/yr in the 1960s and 1970s, to less than 500 t/yr since 2010, and an estimated 281 t in 2023. Natural recruits, freely immigrating or assisted upstream, have been gradually replaced by (imported) restocking and the natural recruits now make up less than 10 % of the total production in inland waters. Peak restocking in the 1990s brought recent total production to a temporary maximum of 455 t in 2010. Lower restocking in the early 2000s reduced total production to 245 t in 2020, but increased restocking thereafter is expected return production to about 336 t in 2027. Thereafter, the contribution of restocking to total production is expected to decrease again due to lower restocking in the 2020s, while the contribution of natural recruitment and assisted migration to total production is expected to increase again due to increased natural recruitment.

From 2010 until 2023, a total number of 462 t of silver eels have been trapped and transported downstream, with a minimum of 5 t in 2010 and a maximum of 25 t in 2021.

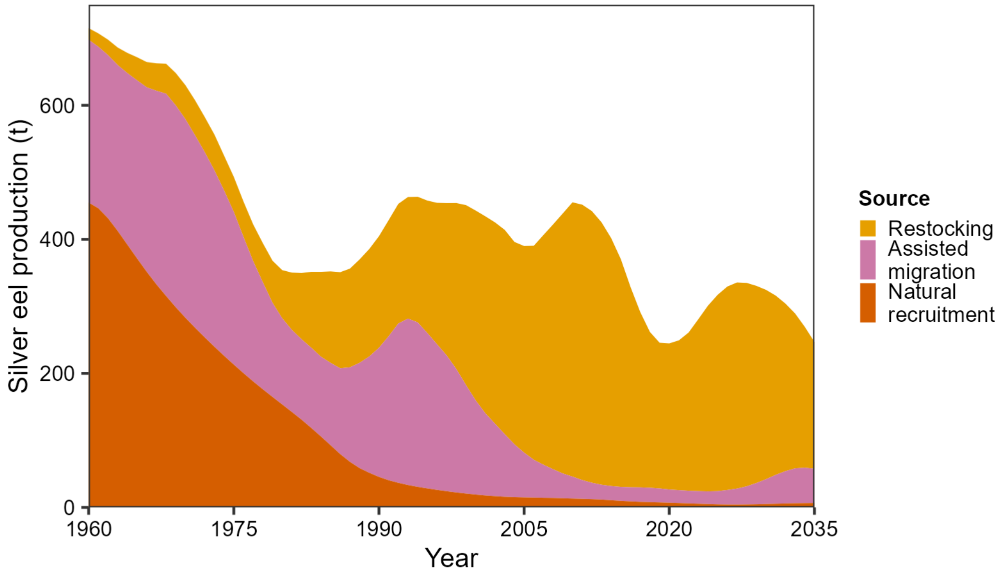

Figure C16: Production of silver eel by year and by origin of the eel, i.e.: the estimated total production before the impact of fishery and hydropower. For these results, a natural mortality rate of M=0.10 yr^{-1} was assumed. Estimates are extrapolated into the future, up to the year 2035.

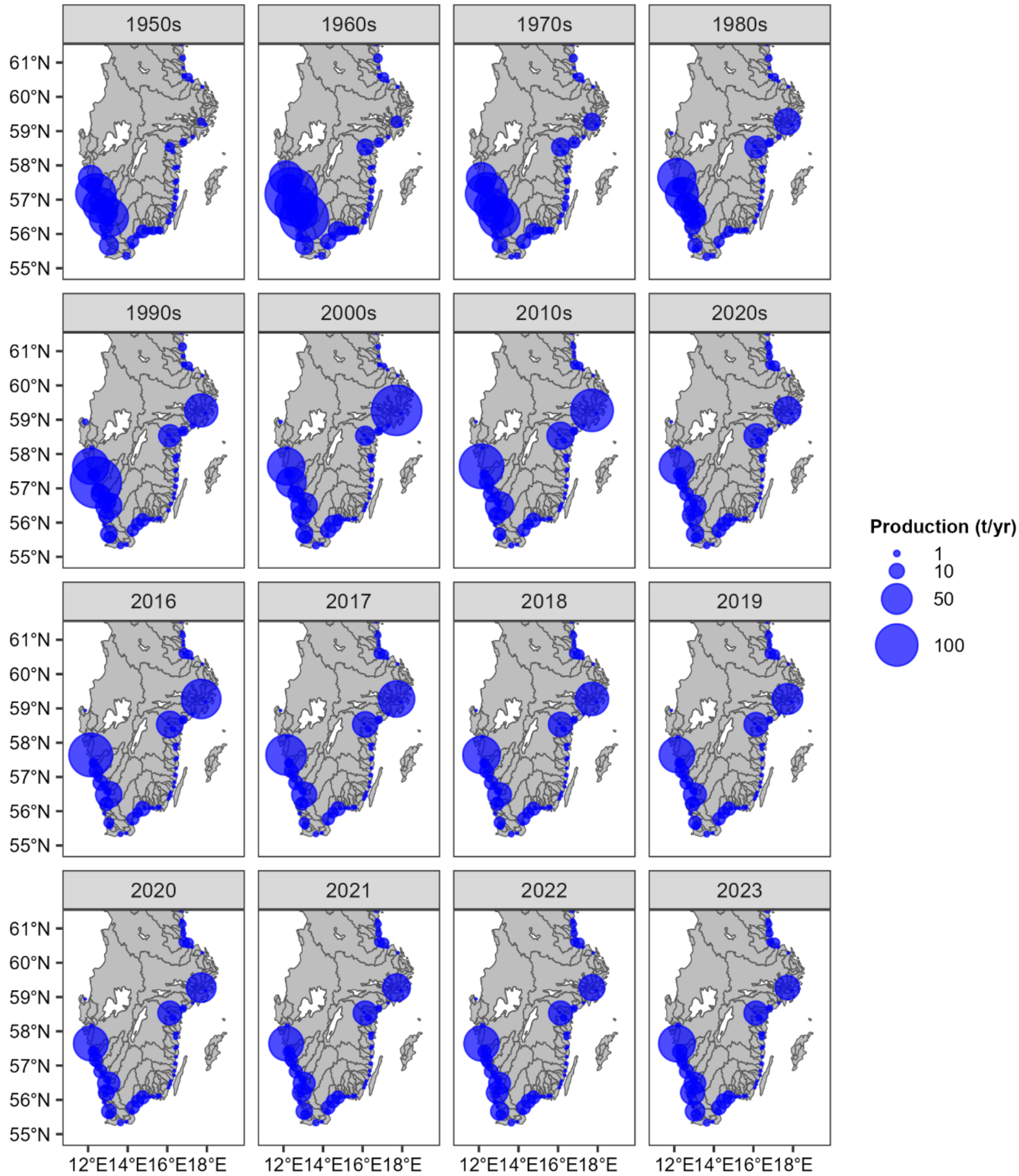

Figure C17: Spatial distribution of the predicted <u>*production*</u> *of silver eel (before fishery and hydropower impacts), shown per river drainage system, as the annual average per decade (top two rows) and as the annual total per recent year (bottom two rows, 2016-2023). The production for each river drainage area is plotted at the place of the river mouth, while in reality, the production will have taken place all over the drainage area.*

C2.2 Silver eel destination

Figure C18 presents the results concerning the destination of the silver eels produced in inland waters, in which the impact of hydropower is estimated from (in order of priority) local experiments, a simulated value reported in Calles and Christianson (2012), or a default impact of 70 % per station; – other options for M will be discussed in section C.2.3.

Fishing data being incomplete up to 1986, results are only available for the period after. The total biomass of silver eel in Figure C18 matches the predicted total production, presented in Figure C16.

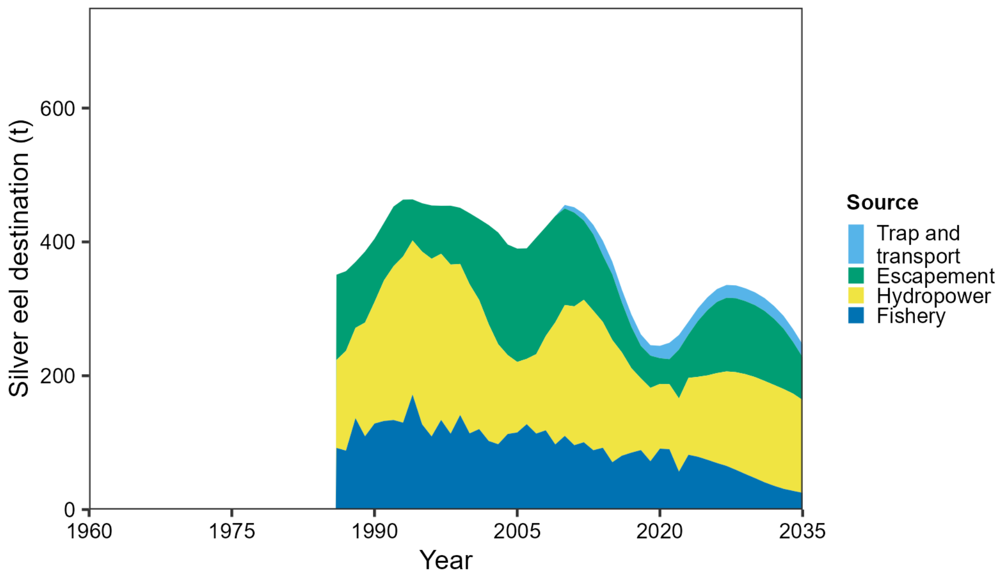

Figure C18: Time trends in the destination of the silver eel produced in inland waters. Here, escapement from Trap & Transport releases is shown separately from 'natural' escapement. Data before 1986 are incomplete. Estimates are extrapolated into the future, up to the year 2035.

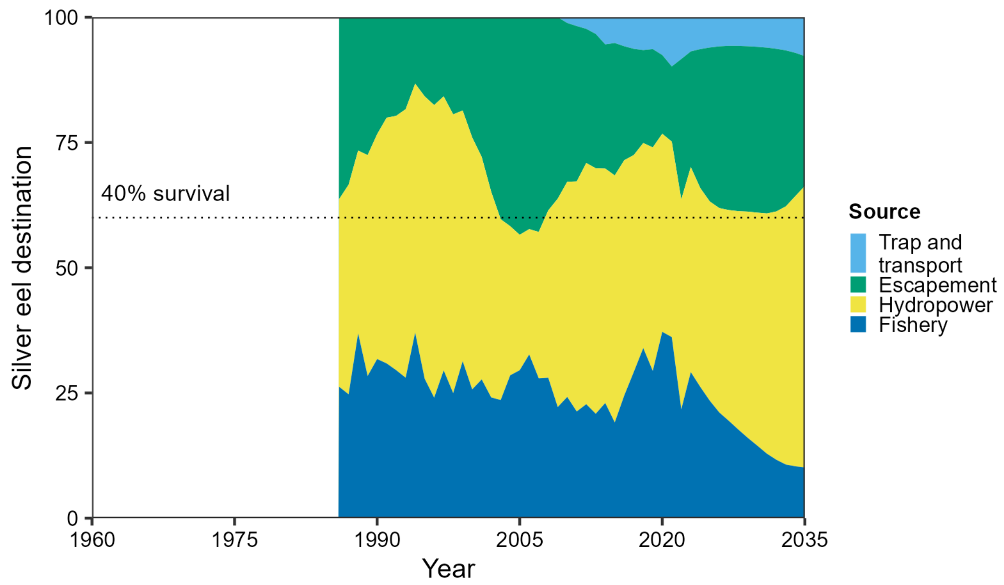

*Figure C19: Time trend in the destination of silver eel produced in inland waters, expressed in percentage of total production. Here, escapement from Trap & Transport releases is shown separately from 'natural' escapement. The reference line "40 % survival" represents the minimum survival considered necessary for stock recovery ($B_{current}$ > 40 % * B_{best}).*

For the fishery, the landings have varied between 57 t (in 2022) and 172 t (in 1994). This is on average 25 % of the production, with rather little variation over the years (Figure C19). The catch in 2023 was 82 t. In the future, assuming a fishing pressure consistent with the 2023 level, it is expected that the landings of the fishery will decrease, primarily due to an expected decrease in silver eel production in lake Mälaren, where restocking has strongly decreased in recent years (Figure C6).

For the hydropower, the estimated impact varied between 97 t (in 2021) and 265 t (in 1996), that is approximately 45 % of the total production (range 25 % - 60 %). The estimated impact in 2023 was 115 t, 41%. Due to the change in restocking locations since 2009 (from major focus on Mälaren, to major focus on Vänern), the impact of hydropower is expected to remain high in the years coming.

Reconstructed escapement of silver eel ranged from 57 t (in 2021) to 174 t (in 2007), on average 25 % of the total production (range 10 % - 40 %). The increase in restocking since 2010 and the increase in natural recruitment since 2015 is expected to contribute to a net rise in escapement from 2024 onwards, to a peak of 130 t in 2028. Without the contribution from restocking, estimated escapement ranged from 75 t (in 1986), to only 6 t (in 2021).

Expressing anthropogenic impacts in terms of mortality rates (Figure C20), one can either consider the mortality on the available stock (whatever their origin,

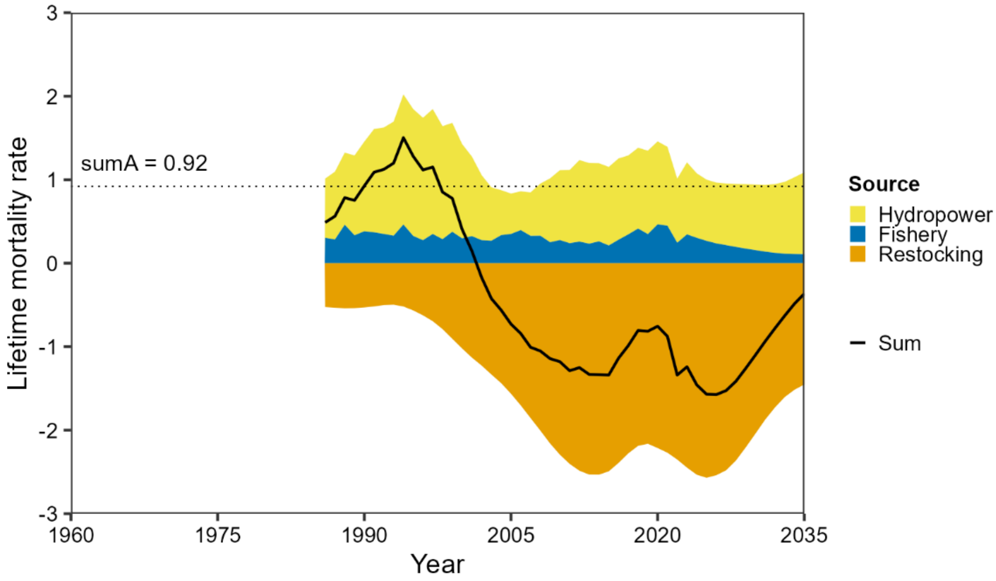

Figure C20: Time trend in the estimated anthropogenic mortalities: fisheries, hydropower, and restocking. The mortality exerted by restocking is negative; that is: restocking increases the amount of silver eel escaping. The solid line represents the sum of all anthropogenic actions, including restocking. The reference line sumA=0.92 represents the limit lifetime mortality considered necessary for stock recovery. Here, the positive mortality effect of the Trap & Transport programme is not directly shown, but is instead indirectly included through a reduction in hydropower mortality.

natural or restocked), but one can also consider restocking as a compensatory action that has an associated negative mortality (so a production of biomass). The presentation in Figure C20 allows for both interpretations. Including the effect of restocking, the sum of fishing mortality, hydropower related mortality, and restocking is represented by a drawn line. Without restocking, the sum of fishing mortality and hydropower related mortality (ΣA) represents the estimated true anthropogenic mortality exerted on any part of the stock, whether natural or restocked. Although we do present both estimates (with and without the effect of restocking interpreted as a negative mortality), we acknowledge that restocking should not be considered as a substitute for precautionary measures (Dekker 2019).

Taking the effects of restocking into account, the total estimate has ranged from +1.50 (in 1994) to -1.34 (in 2015); the 2023 value is estimated at -1.24. Note that negative mortality rates indicate a situation where the effect of compensatory actions surpasses the effects of detrimental impacts. The high and rising estimate for the compensatory effect from restocking is for the major part the consequence of the very low magnitude of natural recruitment (assisted or not), which has led to a low biomass of naturally recruited eels impacted by fishery and/or hydropower.

Considering the anthropogenic mortality without restocking, total anthropogenic mortality has ranged from 0.85 (in 2007) to 2.02 (in 1994); the 2023 mortality is estimated at 1.21. These estimates express the mortality exerted on the natural recruits, as well as on the restocked eels.

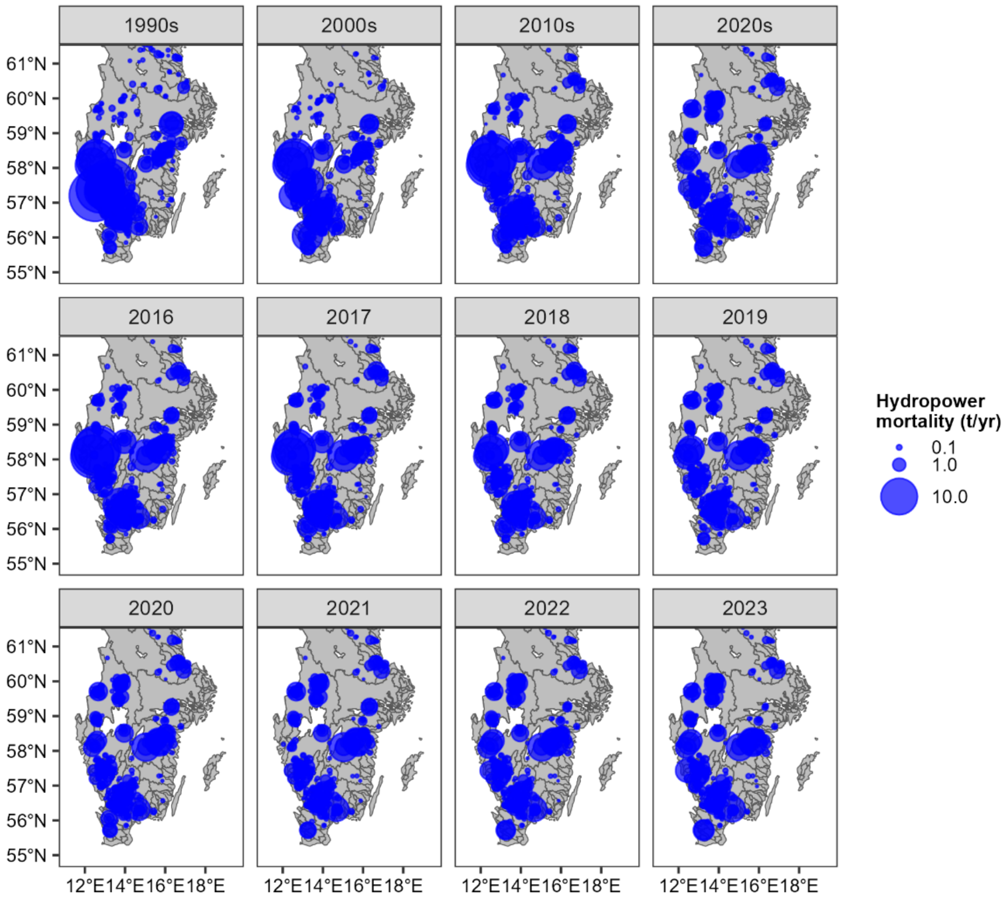

Figure C21: Spatial distribution of the estimated impact of hydropower per hydropower station, shown as the annual average per decade (top row) and the annual total for recent years (bottom two rows, 2016-2023). For the decades preceding the 1990s, no estimates could be derived because of the absence of information on the landings from fisheries before 1986.

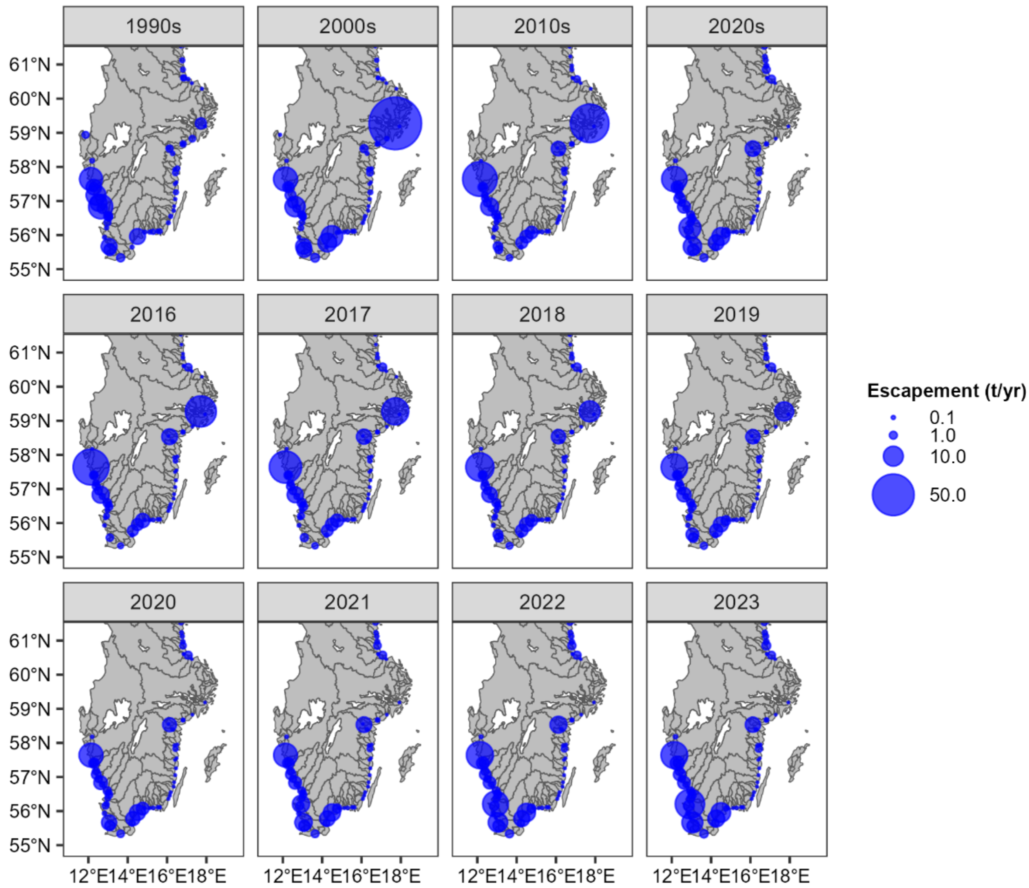

Figure C22: Spatial distribution of the estimated <u>escapement of silver eel</u>, shown as the annual average per decade (top row), and as the annual total per recent year (bottom two rows, 2016-2023). For the decades preceding the 1990s, no estimates could be derived because of the absence of information on the landings from fisheries before 1986.

Considering individual rivers/lakes (Figure C22), it is noted that the most recent estimate of net escapement vanishes in some areas – in particular, the net escapement from Lake Mälaren is now estimated at/below zero. While the restocking into Lake Mälaren ceased almost completely after 2009 (except for a single restocking in 2018 and in 2023), it is expected that the production of silver eel will reduce substantially one life time later, in the 2020s. The actual landings from the commercial fishery, however, show no such drop yet – and hence, our estimate of net escapement comes at a zero/negative value. Though this clearly illustrates the limits of our reconstruction model for individual lakes/rivers, it is also evident that the end of the restocking after 2009 will affect the net escapement negatively, sooner or later.

C2.3 Natural mortality M

Parameter value

The results presented in this Appendix so far are based on an assumption on the level of natural mortality, M=0.10 yr^{-1}. In this section, the sensitivity of results to this assumption is explored. To this end, the whole analysis was rerun, using either a value of M=0.05 or M=0.15. Obviously, all results will change, depending on the value of M. Figure C23 compares results, for two selected years: 1995 and 2023,

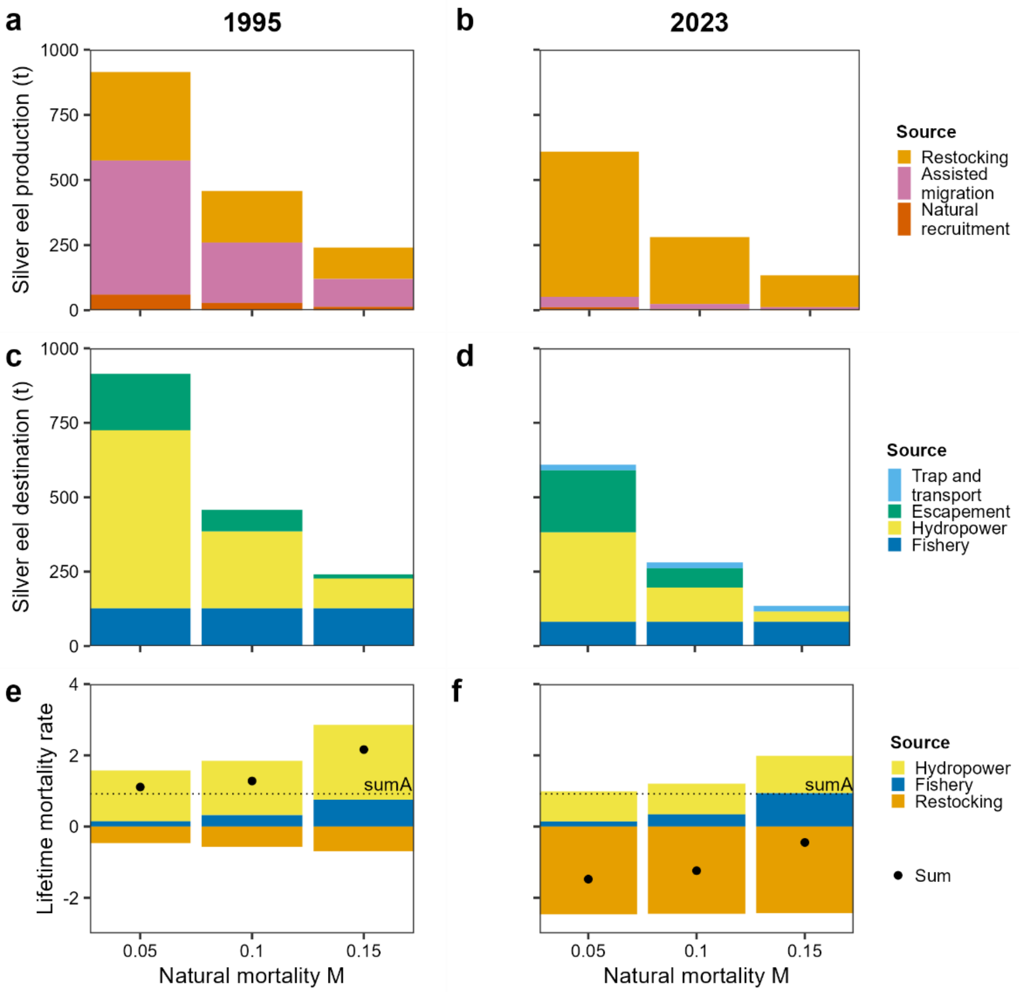

Figure C23: Comparison of results for three different values of natural mortality, showing results for 1995 (left) and 2023 (right). Within each sub-plot, the columns show results for the three options M=0.05, M=0.10 and M=0.15, respectively; comparisons are to be made within each subplot, between the columns.
Top panels (a & b): predicted silver eel production (compare Figure C16);
Middle panels (c & d): predicted silver eel destination (compare Figure C18);
Bottom panels (e & f): anthropogenic mortality rates (compare Figure C20).

that is: a year in the mid-1990s, when both fishing mortality and the impact of the hydropower were at their maximum, and the most recent year. Depending on the value of M, production estimates (Figure C23a&b) can differ by around a factor of three. The relative contributions from natural immigration, assisted migration and restocking, however, are hardly affected. That is: for the production estimates, M operates as a scaling factor, but otherwise does not influence the results considerably. Neither the spatial (not shown) nor the temporal patterns (not shown) are affected considerably by the assumption on M.

For the destination of the silver eel (Figure C23c&d), results are quite different. Fishery landings simply reflect the reported landings, and are thus unaffected by the assumption on M. As a result, as silver eel production decreases with an increasing value of M, silver eel escapement decreases greatly. For M=0.10 yr^{-1}, the estimated production for a few lakes and years ends up below the recorded catch, resulting in a local negative estimate for the silver eel run, the hydropower mortality and the escapement to the sea. For M=0.15 yr^{-1}, negative local estimates occur in many cases (including Mälaren and Vänern, for many years), and on the whole-stock level almost no silver eel is estimated to escape out to sea, both for 1995 (13.8 t) and 2023 (18.4 t).

For the estimates of anthropogenic mortality (Figure C23e&f), the assumption on M has a large effect on the estimate of fishing mortality F (variation by a factor of 5 or more), a moderate effect on the estimate of hydropower mortality H (a factor up to 3), and a very small effect on the estimate of restocking (expressed as a negative mortality). The estimate of total anthropogenic mortality ΣA reflects the sensitivity of F to M. The cumulative effect of fisheries and hydropower (1.57 – 2.86 in 1995; 0.99 – 1.99 in 2020) exceeds the minimal mortality limit (ΣA=0.92 for a healthy stock) in all cases. Though the estimate of ΣA is sensitive to the assumption on M, the evaluation remains that anthropogenic mortality exceeds the limit that will allow any recovery.

At the bottom line, the recorded landings do set an upper limit to the assumptions on M, at a level that is surprisingly low in comparison to conventional estimates/assumptions. Survival from young recruit to silver eel in Swedish inland waters appears to be relatively good. An alternative explanation could be that natural recruitment is much higher than estimated in Appendix B, but micro-chemical analysis of otoliths has corroborated that natural recruits (including assisted migration) constitute not more than 10 % of the catch (Clevestam & Wickström 2008).

In the absence of conclusive evidence on the true value of M, the main results in this Appendix are based on the assumption M=0.10 yr^{-1}, i.e. a rounded value that does not contradict the landings statistics, closest to the more conventional, much higher assumptions. The main conclusion (current levels of anthropogenic impacts, by fisheries and hydropower, do not allow recovery) does not critically depend on this assumption.

Cormorant predation
Over the years, the numbers of cormorants feeding in inland waters has risen considerably (Lundström, 2024), and cormorants are known to feed on eel too (Strömberg et al. 2012). Concerns have been expressed on their predation impact on eel, which might counteract protective actions and reduce fishing yield. The available information on the abundance of cormorants is by far not enough to allow inclusion of cormorant predation in the current reconstruction, which covers more than 65 years and all inland waters in detail. In the current reconstruction, all predation mortality (and other natural causes) is included in a single, constant parameter M for natural mortality. The question arises whether that adequately covers the (increasing) mortality by cormorants.

The assessment of the eel stock given here is based on detailed data concerning the youngest life stages (natural recruits, assisted migration and restocking), and a conversion from youngster to fully-grown silver eel. The conversion to silver eel is based on a simple growth model, and an assumed, constant rate of natural mortality M=0.10, affecting the stock throughout its yellow eel phase. For those eels that are predicted to have died of natural causes at some time during their yellow eel phase, the total biomass comes at 125 % - 200 % (depending on the mean size of the silver eel, 70-90 cm) of the biomass of silver eel produced; only 10 % - 15 % of the initial numbers of youngsters are predicted to survive to the silver eel stage. Figure C16 indicates that silver eel production has varied between 245 and 715 t/yr; hence, it is estimated that the biomass of yellow eel that dies of natural causes has varied between 300 and 1430 t/yr.

According to Lundström (2024), the number of breeding cormorants is in the order of 75 thousand pairs, of which approximately 14 % is found in inland waters. Daily food consumption is estimated at approx. 0.5 kg per individual per day, the year round. Hence, the total fish biomass (of whatever species) eaten by cormorants in inland waters can be estimated at some 3800 t. It is not well known what fraction of the diet consists of eel, especially since the number of eels found in diet samples is almost zero (Boström & Öhman 2014), but of 293 tags in eels released in Lake Roxen, 3.8 % was later recovered in the cormorant colony. Most likely, eel otoliths have been missed, or had fallen apart in the diet analysis (Maria Boström, pers. comm.). No quantitative estimate of the eel consumption by cormorants can be given, but it seems unlikely to be more than a few percent of the approx. 3800 t of fish biomass consumed.

The contrast between the estimate of the biomass consumed by cormorants (order of magnitude of a few percent of 3800 t/yr) to the amount of eel considered to have died of natural causes in the current reconstruction (order of magnitude of

300-1430 t/yr) indicates that the available information on cormorant predation does not contradict the current results.

The references for this Appendix are included in the reference list of the main report, on page 48.

Appendix D: Impact of the Baltic Coast fishery

Dekker and Sjöberg (2013) analysed the impact of the silver eel fisheries on the Baltic Coast, using Survival Analysis for analysing half a century of mark-recapture data, up to 2008. The 2012 assessment used those estimates, extrapolating the 2006-2008 results to 2011 on the assumption that landings and fishing mortality were proportional (Dekker 2012). The 2015 assessment updated the analysis, adding the data from the then re-continued tagging programme (Dekker 2015). No major changes in the methodology of Dekker & Sjöberg (2013) have been made in the 2018 and 2021 assessments (Dekker et al., 2018; Dekker et al., 2021) except that since the 2021 assessment, non-Swedish captures of tagged eel are considered as uncaptured, for the sake of only assessing the impact of the Swedish component of the Baltic Sea eel fishery. This Appendix now presents a new update, including data up to and including 2023.

D.1 Data and methods

The impact of the Baltic Coast eel fishery is assessed using data from Swedish eel tagging experiments. The frequency of these tagging experiments has varied over the years, but in recent years has numbered around 2 to 3 per year (Figure D1). Each experiment tags a number of eel at or close to silvering (typically ranging between 150 to 300 individuals per experiment) with an external tag (silver plates until 1968, Carlin tags since 1967, see Dekker & Sjöberg, 2013), and releases the eel back into the Baltic Sea. Fishers capturing tagged eel report these back, including the location of capture, incentivized by a financial reward. Figure D2 gives a spatial overview of the eel tagging experiments that have been performed since the latest triannual eel assessment.

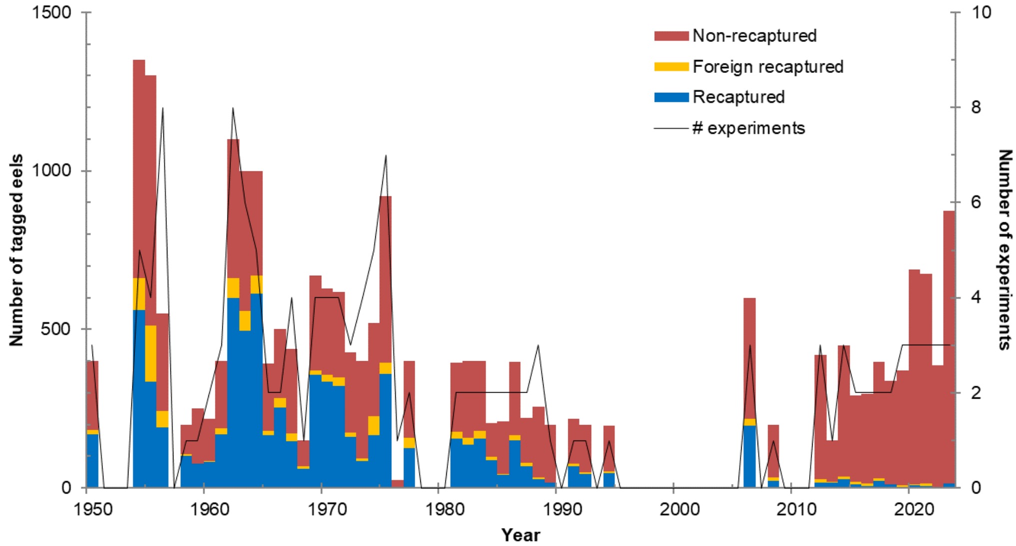

Figure D1: Number of eel tagging experiments performed over time (line) and their annual number of tagged eel (bars). The annual number of tagged eel is subdivided into eel recaptured by the Swedish eel fishery (blue), eel recaptured by a foreign fishery (orange), and non-recaptured eel (red).

Survival analysis is applied to assess the impact of the Swedish Baltic Sea fishery on the escapement of silver eel. Dekker & Sjöberg (2013) first used survival analysis to study the impact of silver eel fisheries along the Baltic coast, and it has been used in each triannual assessment since. They listed four different models of increasing complexity for estimating survival and hazard functions (Dekker & Sjöberg, 2013). Here, we have used their Cox proportional hazards model without time-dependent covariates.

Previously, foreign recaptures of tagged eel had been included in the survival analysis as well. However, the aim of this component of the assessment is to assess the impact of the Swedish Baltic Coast eel fishery. Therefore, tagged eel that have been recaptured outside of Swedish waters (and were thus not captured by the Swedish eel fishery) have now been censored in the survival analysis. This means that they were treated as though they had not been captured.

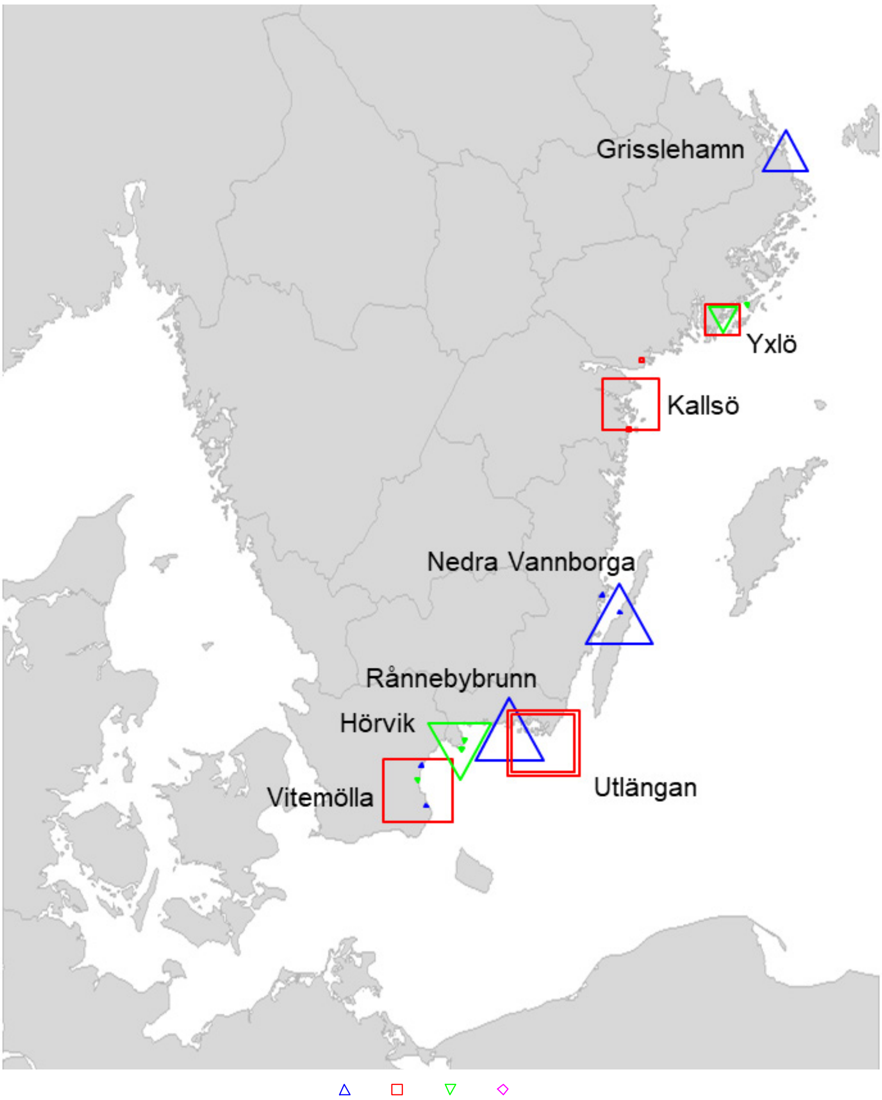

Figure D2: Location of eel tagging experiments in the years 2021-2023. The size of the larger symbols is proportional to the number of eels released. The small dots represent recaptures of single eels. Colour indicates month of release of the tagged eel.

D.2 Results

Figure D1 gives an overview of the number of recaptured and non-recaptured tagged eel over the years. Eel that have been recaptured in foreign (non-Swedish) waters, and have thus not been captured by the Swedish eel fishery, are listed separately from those recaptured in Swedish waters. In recent years the percentage of tagged eel that has been recaptured has been much lower than in the years before 2008. In the 2020 decade, 12 tagging experiments have so far been performed with an average recapture of 1.1% by the Swedish eel fishery, and an additional recapture of 0.5% by the Danish eel fishery. In comparison, in the 2000 decade 4 tagging experiments were performed with an average recapture of 27.5% by the Swedish eel fishery, and an additional 3.9% by the Danish eel fishery. Mean distance covered until recapture shows an increasing trend up to the 1970s, then a declining trend toward the 1990s, and again a declining trend after the 2000s (Figure D3). Mean number of days at large appears to roughly follow the same trend, except that no increasing trend after 2000 can be observed (Figure D3).

Figure D4 & Figure D5 show the results of the survival analysis. Each figure shows the results on a per-decade level, with the most recent decade (2020s) consisting of only three years of data.

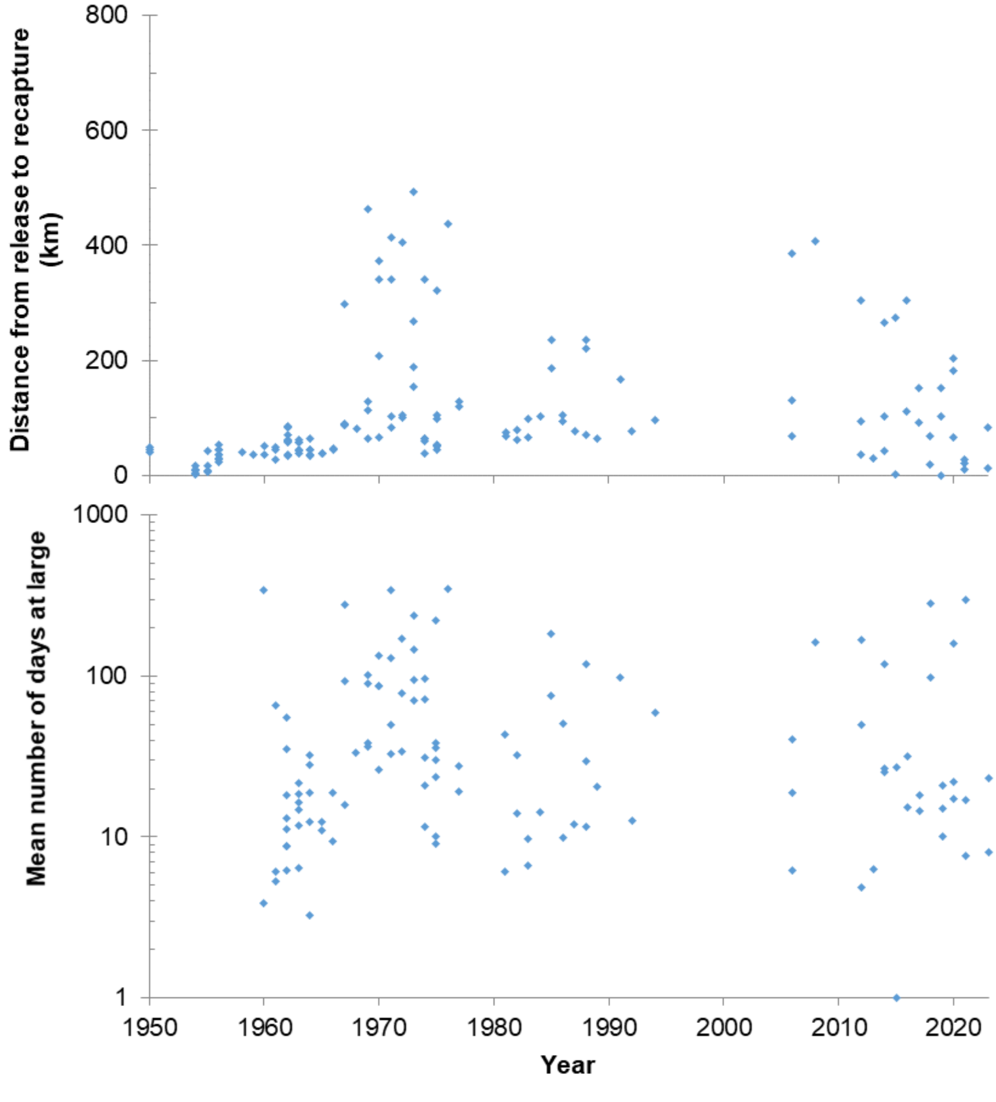

Figure D3: Mean distance covered (top) and mean number of days at large (bottom) between release and recapture, shown for each tagging experiment (year on the x-axis refers to year of experiment). Recaptures outside of Sweden's EEZ have been omitted. Note the logarithmic y-axis on the bottom graph.

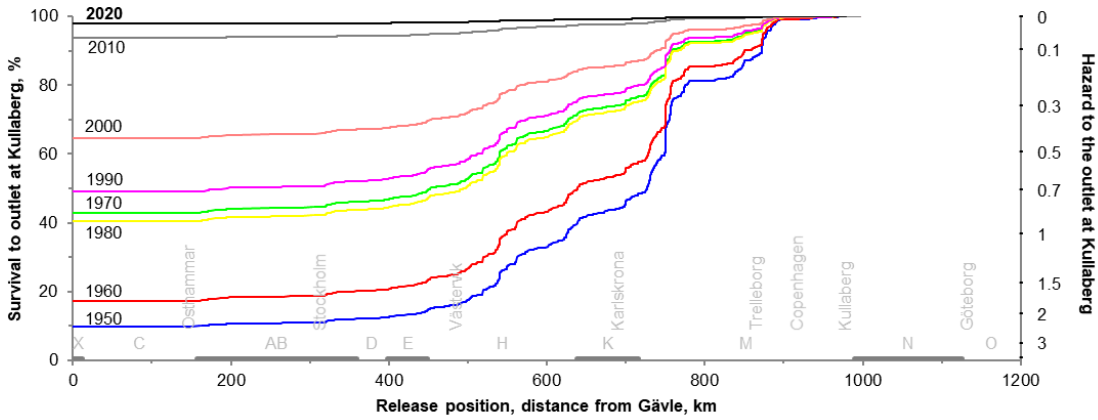

Figure D4: Survival and hazard over distance along the Swedish coast, per decade, as estimated by the Cox proportional hazards model. The left y-axis shows the estimated net survival from a given position along the Swedish coast up to the outlet of the Baltic Sea at Kullaberg, the right y-axis shows the associated accumulated hazard over that interval.

Estimates of survival and hazard curves are given in Figure D4. Over the decades, the hazard of a tagged eel to be recaptured in the Baltic coast eel fishery has decreased considerably, and this decrease in hazard has continued in the 2020s. Average hazard of capture (i.e. fishing mortality) in the 2010 decade was estimated at only 0.009, and in the 2020s decreased even further to an estimated 0.003. To compare, in the 2000 decade average hazard of capture was still 0.055.

County or region (Swedish: län or region) specific estimates of capture hazard for tagged eel are given in Figure D5a. These show a declining trend in hazard over time for every county. Similarly, county-specific landings of eel also show a declining trend over time (Figure D5b).

County-specific estimated stock biomass of silver eel, estimated by dividing county-specific landings (Figure D5b) with county-specific hazard (Figure D5a), shows a general increasing trend over time (Figure D5c). Blekinge county shows an especially-large estimated silver eel biomass, due to its very low estimate of hazard (Figure D5a), which in turn is the result of the very low recapture rate in Blekinge. Whether this low recapture rate shows a true underlying trend, or is due the random absence of tagged eel in the catch, the temporal mismatch of releases and commercial fishing, or the non-reporting of recaptured eel, is unclear.

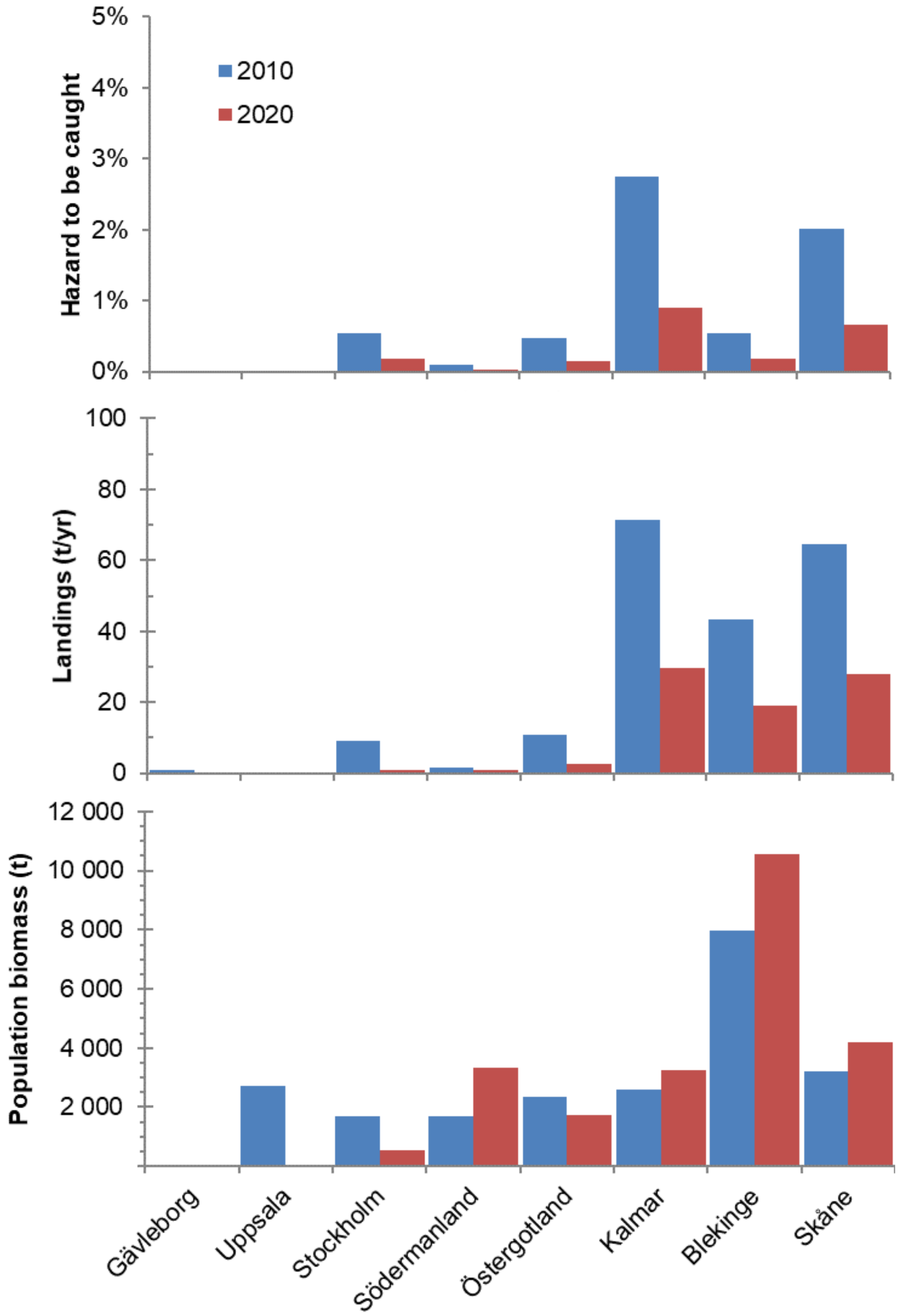

Figure D5: Annual hazard (a), landings (b), and estimated biomass (c) of silver eel along the Swedish Baltic coast, by county/region (län/region), shown for the two most recent decades (2010s, blue; 2020s, red). Since catches and hazards in Gävleborg were effectively zero, no estimate is derived there.

D.3 Discussion

The results of the survival analysis estimate a positive trend in the status of silver eel along the Swedish Baltic coast: commercial fishing pressure on silver eel has been greatly reduced over time as reflected by the survival and hazard curves (Figure D4), and silver eel stock biomass shows an increasing trend over time.

The number of days at large for tagged eel has been similar over the decades (Figure D3). It should be noted that – in recent years - the number of days at large is also related to the length of the fishing season allowed: recaptures can only be made from the start of the season to the end of the season, and restrictions in the season length will thus likely lead to a lower average period at large, with eel that would otherwise have been captured during the now-closed season successfully migrating out of the Baltic Sea. In other words, with a closed season, a decrease in number of days at large is not indicative of a greater fishing pressure. Should the season be shortened even further, then this would likely decrease the average number of days at large accordingly.

The reason that hazard of capture is estimated to have decreased is because the recapture rate of tagged silver eel has decreased considerably over the years. This raises the question: has recapture rate truly decreased, or are recaptures less likely to be reported? The concurrent decline in silver eel landings, along with no strange patterns in time at large and distance at large, favours the explanation that recapture rate has indeed declined.

The current low recapture percentage of tagged silver eel (4.4% in the 2010 decade, 1.1% so far in the 2020s) indicates that the Swedish commercial fishing impact on silver eel in the Baltic Sea is comparatively low, aiding in silver eel escapement and stock recovery. However, it also means that the estimates of the survival analysis are becoming increasingly uncertain. This increasing uncertainty of the survival analysis due to reductions in commercial fishing pressure on silver eel means that, to be able to continue to reliably monitor the trend in silver eel stock status along the Baltic coast, the potential value of a fisheries-independent monitoring programme of silver eel along the Baltic coast should be considered urgently.

This estimate of the anthropogenic mortality on the Baltic coast in Sweden applies to the silver eel in front of our coast, not to the preceding lifetime in other Baltic countries where they grew up as yellow eel. As such, this assessment is not able to estimate the lifetime anthropogenic mortality on eel in this area, only the mortality component of the Swedish Baltic coast fishery.

The restocking of eel on the Baltic coast has been described in Appendix A, above.

The references for this Appendix are included in the reference list of the main report, on page 48.