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Fourth post-evaluation of the Swedish eel management

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Executive 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 2021, intending to feed into the national reporting to the EU later this year. This report updates and extends the reports by Dekker (2012, 2015) and Dekker *et al*. (2018).

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 technical evaluation. Our focus is on the quantification of biomass of silver eel 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 2011, a break in the downward trend of the number of glass eel was observed throughout Europe, the trend since being upward, but erratic. Whether that relates to recent protective actions, or is due to other factors, is yet unclear. This report contributes to the required international assessment, but does not discuss the causing factors behind that recent trend and the overall status of the stock across Europe.

For the different assessment areas, results summarise as follows:

On the <u>west coast</u>, a 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), as in 2018. 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. After years of decline, the research surveys now indicate a break in the decline of the stock. 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 and a slight increasing trend in abundance in line with the recent 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 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 constitute a larger share, and it might contribute to the recovery of the west coast stock.

For inland waters, this report updates the 2018 assessment, not making substantial changes in methodology. 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 75 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, the mean age/size of the immigrating eel, 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 are (with the exception of Mörrumsån, 56.4°N: 100 gr, where only 30 gr would be expected). Distance upstream comes with less numerous recruits, but size is unrelated. Remarkably, the time trend differs for the various ages/sizes. The oldest recruits (age up to 7 years) declined in abundance already in the 1950s and 1960s, but remained relatively stable since. The youngest recruits (age 0) showed a steep decline in abundance in the 1980s and little decrease before and after. In-between ages show in-between trends. Though this peculiar age-related pattern has been observed elsewhere in Europe too, the cause of this is still unclear. 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 500 to below 300 tonnes per annum (t/a). 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 occur in the near future. Gradually, restocking has replaced natural recruitment (assisted and fully natural), now making 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 20 % to 60 %, depending on the year. Escapement is estimated to have varied from 25 % of the pristine level (100 t) in the late 1990s, to 50 % (200 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 (nearly 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 largely 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. Until about 2009, restocking has been practised in freely accessible lakes (primarily Lake Mälaren, 1990s), but is since 2010 concentrated to drainage areas falling to the Kattegat-Skagerrak, thus also including obstructed lakes (primarily Lake Vänern, to a lesser extent Lake Ringsjön, and many smaller ones). Trap & Transport of silver eel - from above barriers towards the sea - has added 1-5 % of silver eel to the escapement biomass. Without restocking, the biomass affected by fishery and/or hydropower would be only 10-15 % of the currently impacted biomass, but the stock abundance would reduce from 20 % to less than 5 % 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 are currently 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 2018 assessment has been updated without major changes in methodology. Minor changes include the censoring of foreign-recaptured tagged eel (treating them as though they were not captured) so as to only describe the impact of the Swedish eel fishery, and complementing the decadal estimates with triannual estimates. Results indicate that the impact of the fishery continues to decline over the decades – even declining more rapidly within the 2010s than before. The current impact of the Swedish silver eel fishery on the escapement of silver eel along the Baltic Sea coast is estimated at 1 %. 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 Baltic 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 2021; 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).

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 svårligen 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 <u>västkusten</u> 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 fiskerioberoende fiskeundersökningarna som görs visar emellertid att de tidigare utnyttjade storleksklasserna av beståndet verkligen återhämtar sig, men överlag har nedgången i beståndet fortsatt – 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 återhämtningen av beståndet på västkusten.

För <u>inlandsvattnen</u> så redovisar rapporten en uppdatering av 2018 års beståndsuppskattning, utan större förändringar i metodiken. 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 75 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, medelstorlek i ålder och storlek, och till vilket år dessa ålar rekryterades till kontinentala vatten som glasål, dvs. årsklass. Längre in i Östersjön är uppvandrande ålar större men färre. Ålarna från Mörrumsån avviker genom att de där är större än förväntat (100 g gentemot 30 g). Längre avstånd från mynningen medför färre ålar, men storleken är inte relaterad till avståndet. Anmärkningsvärt är att tidstrenderna skiljer sig åt mellan olika åldrar och storlekar. De äldsta rekryterna (ålder upp till 7 år) minskade redan under 1950- och 1960-talet, men stabiliserades sedan. De yngsta rekryterna (0+) visade en snabb minskning under 1980-talet och en mindre minskning dessförinnan och efter. Åldrarna där emellan visar på en intermediär minskningstakt. Även om en sådant anmärkningsvärt åldersrelaterat mönster har observerats också på andra håll i Europa, så är orsakerna fortfarande okända.

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 500 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 20 % till 60 %. Utvandringen av blankål till havet har varierat från 25 % (100 ton) under sent 1990-tal till 50 % (200 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 (drygt 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 i stort 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.

Fram till och med 2009 har utsättningarna främst gjorts i sjöar med fria vandringsvägar till havet (till stor del i Mälaren under 1990-talet), men görs sedan 2010 främst i avrinningsområden som mynnar på västkusten, och därmed delvis i sjöar med hinder för nedströmsvandring (främst i Vänern, men också i Ringsjön och flera mindre sjöar). Trap & Transport av blankål, från områden uppströms vattenkraftverk ner till respektive mynningsområde, har tillfört 1-5 % till lekvandringens biomassa. Utan ålutsättning skulle biomassan av ål påverkad av fiske och vattenkraft bara vara 10-15 % av dagens påverkade biomassa. Samtidigt skulle ålbeståndet bara vara 5 % av den jungfruliga biomassan, att jämföra med dagens 20 %.

Sammanfattningsvis: biomassan av inlandsvattnens ålbestånd uppnår inte nödvändig miniminivå, den mänskliga påverkan överskrider den lägsta gränsen för återhämtning, och de negativa effekterna kommer fortsätt öka. Utan ytterligare skyddsåtgärder kommer situationen att förvärras. 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 2015 års beståndsuppskattning uppdaterats utan förändringar i metodiken. Resultaten indikerar att fiskets inverkan snabbt minskar över tid, kanske snabbare mot slutet av 2010-talet än tidigare. Dagens påverkan från det svenska blankålsfisket vid ostkusten beräknas nu till 1 %. 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* (Linnaeus) is in severe decline: fishing yield has declined gradually in the past century to below 10 % of former levels, and recruitment has rapidly declined to 1-10 % over the last decades (Dekker 2004, 2016; ICES 2020a). In 2007, the European Union (Anonymous 2007) decided to implement a Regulation establishing measures for the recovery of the stock of European eel (Dekker 2008), obliging EU Member States to develop a national Eel Management Plan (EMP) by 2009. In December 2008, Sweden submitted its EMP (Anonymous 2008). Subsequently, protective actions have been implemented (in Sweden and all other EU countries), and progress has been evaluated internationally in 2012 (Anonymous 2012; Anonymous 2014) and 2020 (Anonymous 2020). In spring 2012, a first post-evaluation report was compiled, assessing the stocks in Sweden (Dekker 2012). Subsequently, in 2015 a second post-evaluation report was compiled (Dekker 2015), and in 2018, a third one (Dekker *et al*. 2018). This current report – in 2021 - updates, extends and reviews those reports.

1.2 Aim of this report

The EU 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, European

[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 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.

policies that pre-date the 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 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 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 2020). 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 (2021) is the first reporting year under this agreement.

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 ($B_{current}$, B_{best} and B_0) 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 Annex. 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.

Annex A presents data from the west coast.

Annex B presents the riverine recruitment time series and analysis of spatial and temporal trends.

Annex C reconstructs the inland stock from databases of historical abundance of young eels.

Annex D updates the assessment of Dekker and Sjöberg (2013), adding mark-recapture data from silver eel along the Baltic coast for the years 2012-2020.

1.4 The Swedish eel stock and fisheries

The eel stock in Sweden 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. On the next pages, the current habitats and fisheries are briefly described.

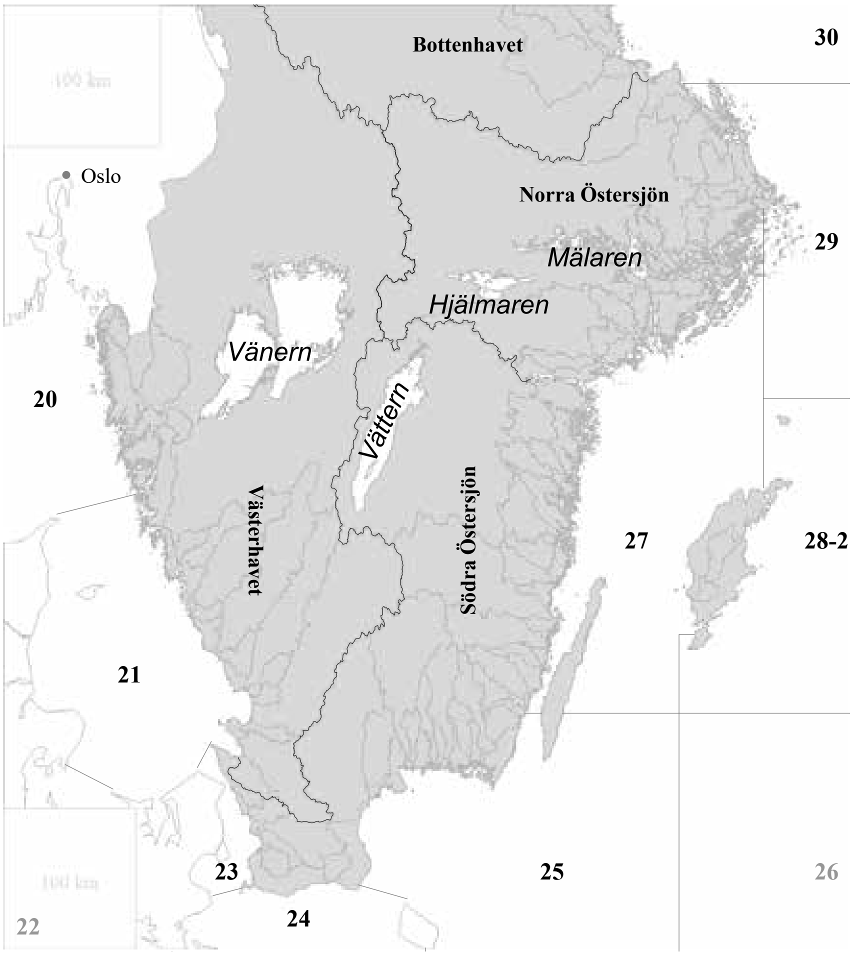

Figure 1 Map of the study area, the southern half of Sweden (north up to 61°N). The names in italics indicate the four largest lakes; the names in bold indicate the Water Basin Districts related to the Water Framework Directive (not used in this report); the numbers refer to the ICES subdivisions; the 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 lakes, except in the high mountains and the northern parts of the country. Pound nets are used to fish for eel in the biggest lakes Mälaren, Vänern and Hjälmaren, and in some smaller lakes in southern Sweden. 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 four 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 and restocking, fishery on yellow and silver eel, impact of migration barriers (on immigrating youngsters) and hydropower generation (on emigrating silver eel). 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. Trap & Transport of silver eel – only that. The presentation of Trap & Transport data has been included in Annex C, in the discussion of inland waters.
4. 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 (Anon. 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, ICES (2002) 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 $\Sigma A = -{}^{e}\log(40\%) = 0.92$ (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 ΣA=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. 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 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 ΣA=0 (maximum aspiration level, but still a slow recovery) and ΣA=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 (2020) for the European eel reads "*all anthropogenic impacts […] should be reduced to, or kept as close as possible to, zero*". SLU Aqua has recently conformed to this advice (SLU and HaV 2021). This advice is based on the consideration that there may be situations where the spawning stock is so low that reproduction is at significant risk of being impaired. In such cases, ICES may advice zero catch (i.e. zero fishing mortality) until the spawning stock biomass has clearly recovered. Aiming for minimal 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. First, 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. Secondly, it is unclear what "as close as possible to zero" exactly means in quantitative terms: is the current situation already within those limits, already "as close as possible"? Thirdly, the ICES advice framework tends to emphasise reducing fisheries, while it is the combined anthropogenic impacts affecting the stock, and it should be a policy decision which impact to address preferentially.

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 and 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 2013a, 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

What reference framework to apply in the current assessment? 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 our previous assessment reports - has now been shown to be arbitrary and not fully consistent. Given the impossible choice between these three imperfect approaches, we <u>decide</u> to take a different approach, expanding a suggestion by Dekker (2019), 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-WGEEL (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 (that is: 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, we may deduce 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). Whether that number of generations, and hence any actual level of anthropogenic mortality, is considered acceptable or not, is left open by us. 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 be preferably be replaced by one in numbers of years. Additionally, the aspiration level should not 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.

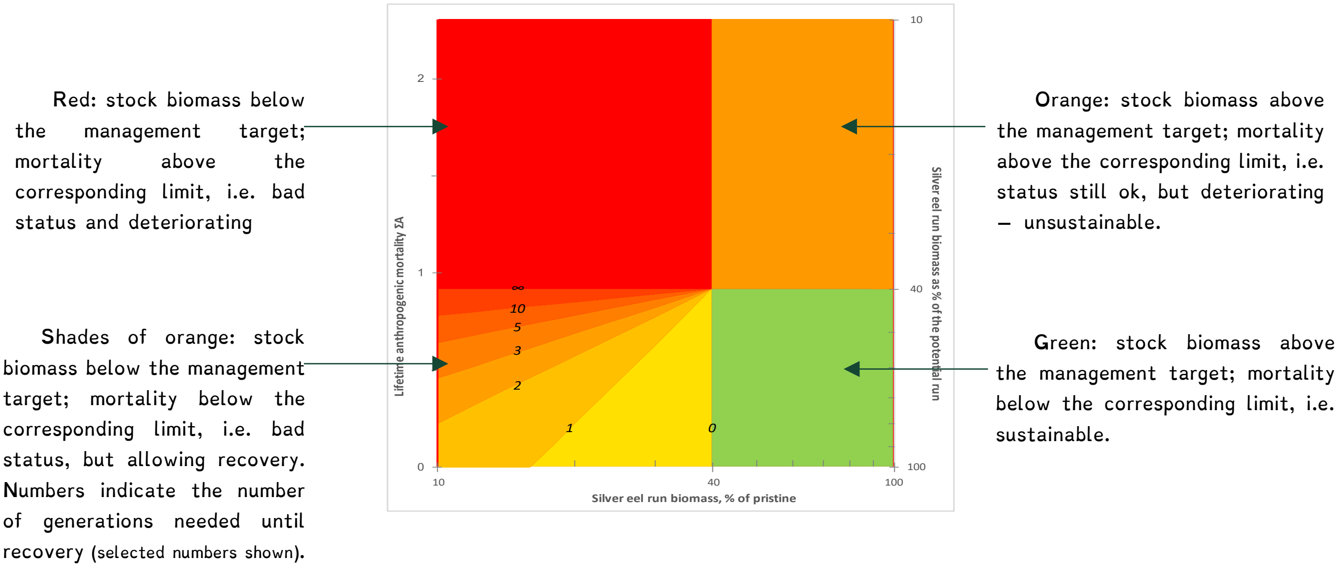

Figure 2 Precautionary Diagram, presenting the status of the stock (horizontal) and the level of anthropogenic impacts (vertical). The left axis shows the lifetime anthropogenic mortality (rate), while the right axis shows the corresponding survival rate. Note the logarithmic scale of the horizontal and right axis, corresponding to the inherently logarithmic nature of the left 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 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

There is no dedicated monitoring of natural recruitment to inland waters in Sweden, but the 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 entering the rivers where a trap is installed (Erichsen 1976; Wickström 2002). Figure 3 shows the raw observations; Annex B presents an in-depth analysis of temporal and spatial trends in these data. The results align with the international trend (ICES 2019) that - after decades of decline - the recruitment has stopped decreasing after 2011 and is now on the rise, but the trend after 2011 is rather unclear (few data points) and erratic (high variation).

Photos (from left to right): Glass eel, elver, bootlace and yellow eel. Photographers (from left to right): Jack Perks, Ad Crable, Deutsche Welle, Lauren Stoot.

[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.

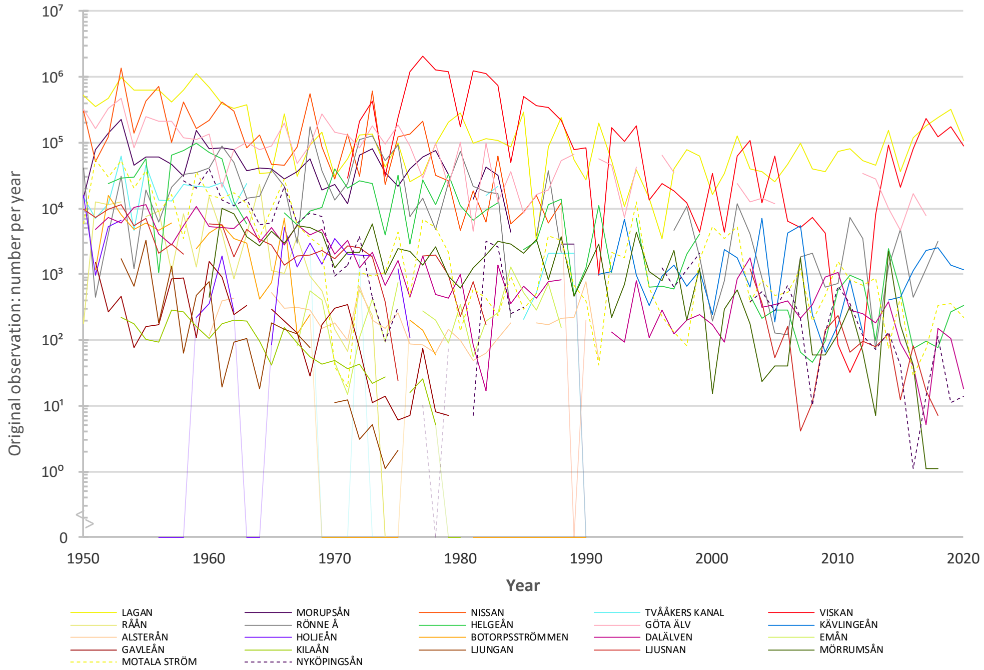

Figure 3 Trends in the number of elvers trapped at barriers, in numbers per year. Note the logarithmic character of the vertical axis. For further details, see Annex B.

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 thereafter.

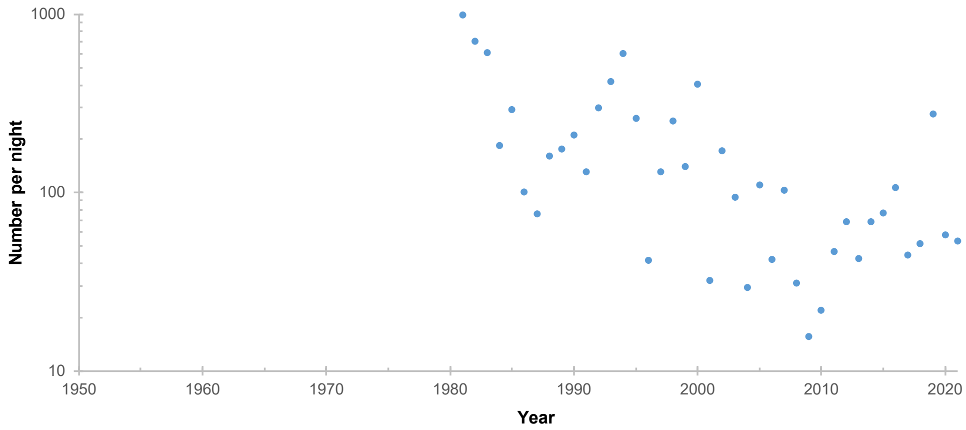

Figure 4 Time trend in glass eel recruitment at the Ringhals nuclear power plant on the Swedish Kattegat Coast. Note the logarithmic character of the vertical axis.

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

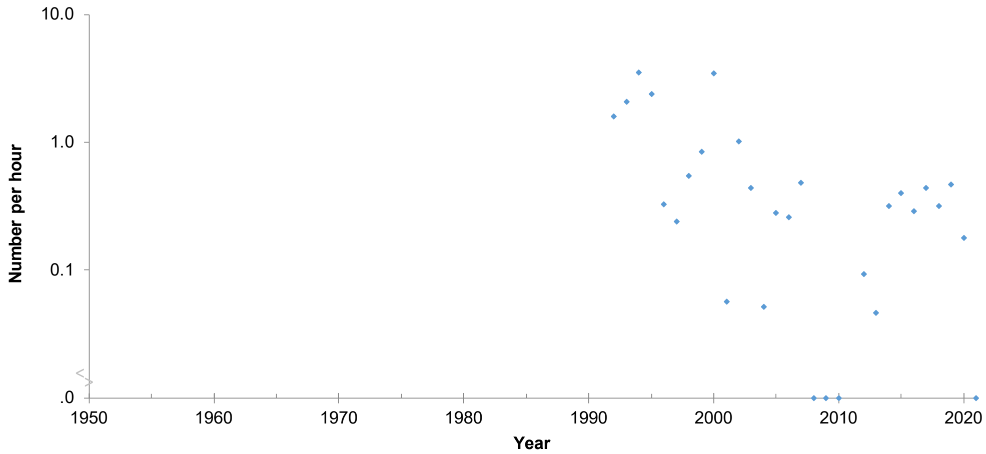

Figure 5 Catch of glass eels (number per hour trawling) by a modified Methot–Isaacs–Kidd Midwater trawl (MIKT) in the Skagerrak-Kattegat 1992–2021. In 2008-2010 and in 2021, zero glass eels were caught; in 2011, no sampling took place. Note the logarithmic character of the vertical axis.

3 Restocking

Restocking (stocking) is the practice of importing young eel from abroad (England, France, 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 Annex C for details). Restocking of young eel started in Sweden in the early 1900s (Trybom and 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 numbers used for restocking in most recent years. Annex C gives full detail (spatial and temporal) for the inland waters; Annex A for the coastal waters.

Table 1 *Number of eels restocked, by area. To the left, the actual numbers released, by the year in which they were released. To the right, the same but expressed in glass eel equivalents, by their year class, i.e. the hypothetical number and year that they would have been a glass eel.*

	Actual numbers				Glass eel equivalents		
Year	West coast	Inland waters	Baltic coast	year class	West coast	Inland waters	Baltic coast
2000		1 437 378	566 722	2000	9 771	838 688	178 647
2001		969 108	376 597	2001	8 974	1 264 889	442 664
2002	24 255	1 117 322	486 184	2002		333 019	444 244
2003	12 502	463 751	516 713	2003	15 879	882 555	284 893
2004	21 625	939 356	368 156	2004		899 453	198 664
2005	6 195	915 822	187 667	2005		992 907	397 871
2006		940 781	375 847	2006	7 939	796 358	210 943
2007	7 500	777 033	201 576	2007		1 035 343	422 304
2008		1 121 863	398 927	2008		583 360	220 932
2009		564 254	212 002	2009	190 548	1 791 196	65 633
2010	180 000	1 694 510	62 000	2010	574 819	2 096 453	109 036
2011	543 000	1 977 984	103 000	2011	585 405	2 034 154	94 215
2012	553 000	1 924 022	89 000	2012	615 681	2 067 906	129 149
2013	581 600	1 953 984	122 000	2013	824 237	2 140 580	160 907
2014	778 611	2 017 432	152 000	2014	899 015	1 002 795	77 278
2015	849 250	944 144	73 000	2015	1 569 939	1 409 346	56 953
2016	1 483 035	1 334 362	53 800	2016	528 848	416 814	56 854
2017	499 574	394 074	53 707	2017	1 550 220	1 680 590	63 368
2018	1 464 408	1 584 371	59 860	2018	1 489 773	1 503 074	47 637
2019	1 407 307	1 419 808	45 000	2019	1 139 820	2 063 187	68 809
2020	1 076 725	1 948 979	65 000	2020			

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 assessments.

The contribution from restocking to the coastal stocks is small in comparison to the natural stock. For the west coast, the potential production of silver eel $B_{pristine}$ was estimated at 1 154 t or more (Dekker 2012), but current silver eel production is certainly much lower (though not exactly known). Current restocking (1 million in 2020) will potentially produce considerably less than 100 t, which is a negligible quantity in comparison to the ultimate potential, even though it might be more in comparison to the current low silver eel production. 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 0.1 million per year recently) will potentially produce considerably less than 10 t. It is doubtful whether these small additions made by 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, which ones from other sources. The restocking-based production is in an order of 200-300 t, while the natural silver eel production in 2020 is estimated at 25 t.

All in all, none of the assessments is 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 mitigate the effect of migration barriers, to compensate for other anthropogenic mortalities, or to support the recovery of the stock. The classical objective for restocking in Sweden has been to support the fishery; assisting migration of natural recruits intended to mitigate the effect of migration barriers on stock and fisheries. Current restocking is intended to support recovery of the stock (governmental restocking in unobstructed, unexploited waters; Anon 2008), or to compensate for other anthropogenic mortalities (restocking on the coast, compensating for the impact of hydropower generation, in the programme

‘Krafttag Ål’ (KTÅ) on hydropower and eel; Dekker & Wickström 2015). That is: both objectives of restocking (increasing the stock, resp. compensating for other anthropogenic mortality) have been and still are in use. Whatever way we define our indicators in this report (taking restocking into account or not), there will be areas where they do and do not apply, leading to confusing results.

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 1979-1999, 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 Annex 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, Annex A presents trends in stock abundance estimates, based on fishery-independent surveys.

For the fishery in inland waters, Annex 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. The current assessment copies the methodology of Dekker (2015, 2018). 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 Annex D for details. Since this assessment covers the silver eel stage only, 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 has 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. The data in Table 2 have been corrected, and now represent the total catch, whatever the destination. See also Chapter 6 on Trap & Transport.

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}
2000	154	114	263	1.79	0.29	
2001	226	120	297	2.53	0.30	
2002	216	102	273	2.41	0.26	
2003	192	98	275	2.15	0.25	
2004	216	113	254	2.43	0.30	0.054
2005	214	115	346	2.39	0.32	
2006	239	128	366	2.66	0.36	
2007	170	114	418	1.91	0.31	
2008	164	118	389	1.86	0.31	
2009	107	97	310	1.19	0.24	
2010	108	110	307	1.20	0.26	
2011	83	96	271	0.93	0.22	
2012	0	101	239	0	0.23	
2013	0	103	271	0	0.25	0.020
2014	0	111	213	0	0.28	
2015	0	88	158	0	0.24	
2016	0	97	181	0	0.30	0.016
2017	0	102	143	0	0.36	
2018	0	105	146	0	0.43	
2019	0	89	99	0	0.40	0.003
2020	0	94	101	0	0.45	

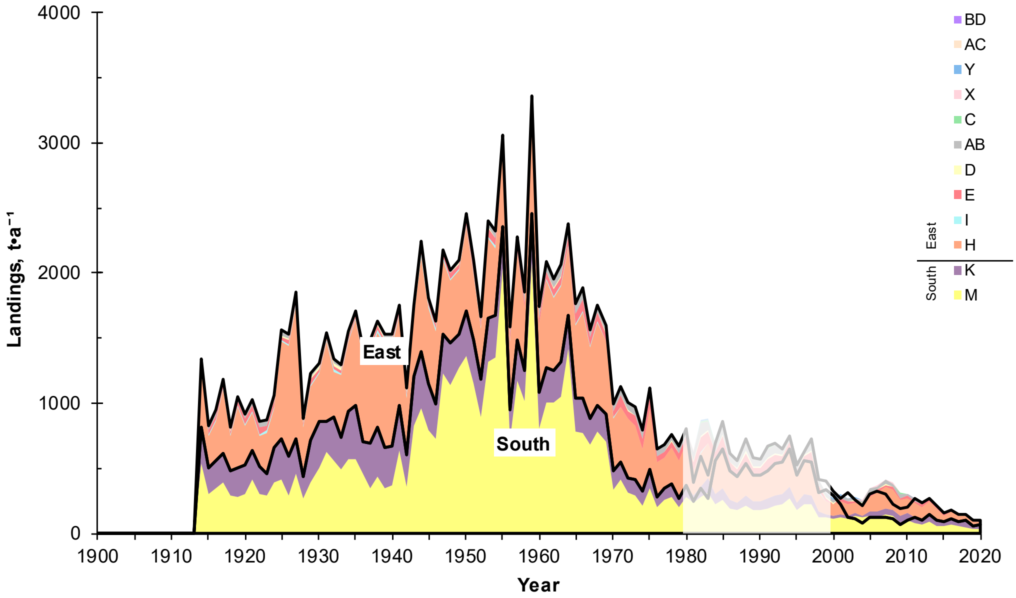

Figure 6 Trend in landings from the Baltic coast fisheries, by county (colours) and area (black lines). In the years 1978-1998 (faded), due to lack of detailed records, it has been assumed that the percent-wise contribution of each county had remained constant. Note that the total landings on the Baltic 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.

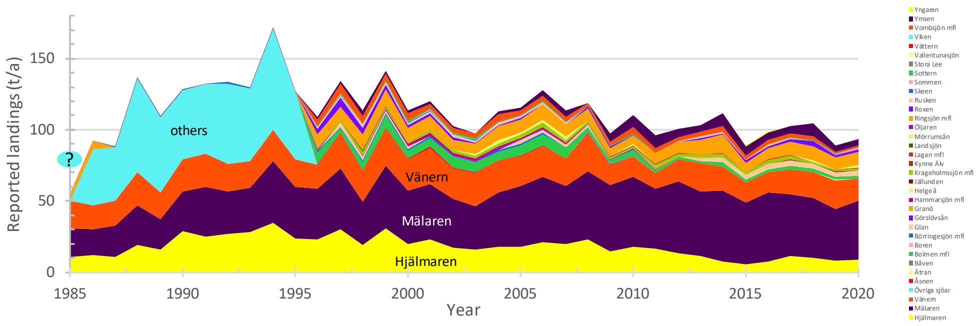

Figure 7 Trends in landings of eel from inland waters. Before 1996, only the totals for all lakes (except the three largest ones) are known; statistics before 1986 are only available for the three largest ones.

5 Impact of hydropower on silver eel runs

A reconstruction of the inland stock is presented in Annex C. That includes a spatially and temporally explicit reconstruction of the impact of individual hydropower stations. The data in Table 3 are 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. However, the release of those eels is not considered here, i.e. the estimates in Table 3 represent the true mortality exerted on migrating silver eel. For the release of the Trap & Transport eels, see Chapter 6.

From the detailed reconstruction in Annex 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 altering restocking practices. Since 2010, restocking first shifted relatively more towards lakes with hydropower stations downstream (i.c. Lake Vänern), which results in a 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.*

	Biomass of silver eel killed (tonnes)			Hydropower mortality ΣH (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000		213			0.84	
2001		181			0.69	
2002		163			0.57	
2003		138			0.47	
2004		107			0.37	
2005		88			0.31	
2006		74			0.27	
2007		90			0.32	
2008		115			0.41	
2009		155			0.53	
2010		169			0.59	
2011		193			0.65	
2012		205			0.73	
2013		194			0.73	
2014		178			0.72	
2015		175			0.75	
2016		149			0.76	
2017		119			0.72	
2018		97			0.69	
2019		95			0.75	
2020		93			0.83	

6 Trap & Transport of silver eel

In recent years, silver eel from lakes situated above hydropower generation plants has been trapped and transported downstream by lorry, bypassing the hydropower-related mortality. The initial catch of silver eel for this programme conforms to a normal fishery; this impact has been included in the fishery statistics (Chapter 4). The release of these silver eels, however, contributes to the overall escapement. Therefore, those data are reported here separately (see Table 8 on page 79 for further details).

The effect of the Trap & Transport programme cannot be expressed as a (negative) mortality rate. The silver eel released is neither strictly related to the stock in inland waters (where they come from), nor to the stock in coastal waters (where they are released into). To express the Trap & Transport programme as a mortality rate, one would have to compare the biomass affected to the biomass in the stock. Since the relevant stock cannot be identified uniquely, there is no unique way to express the Trap & Transport as a (negative) mortality rate.

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.*

	Biomass of silver eel (tonnes)			As mortality (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000						
2001						
2002						
2003						
2004						
2005						
2006						
2007						
2008						
2009						
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			

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*. (2017) 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.

In this report, the impact of the Swedish fisheries on the run of silver eels along the Baltic coast is assessed (Annex 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 (on page 99, below) 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 six 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. Annex 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, and a slight recovery in abundance can be observed for the size classes below the (former) minimum legal size (37 cm until 2007, then 40 cm until the closure in 2012). 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, Annex 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 (actually, 2020 is the peek year). However, this will not bring the anthropogenic impacts below the minimum limit that will allow the stock to recover.

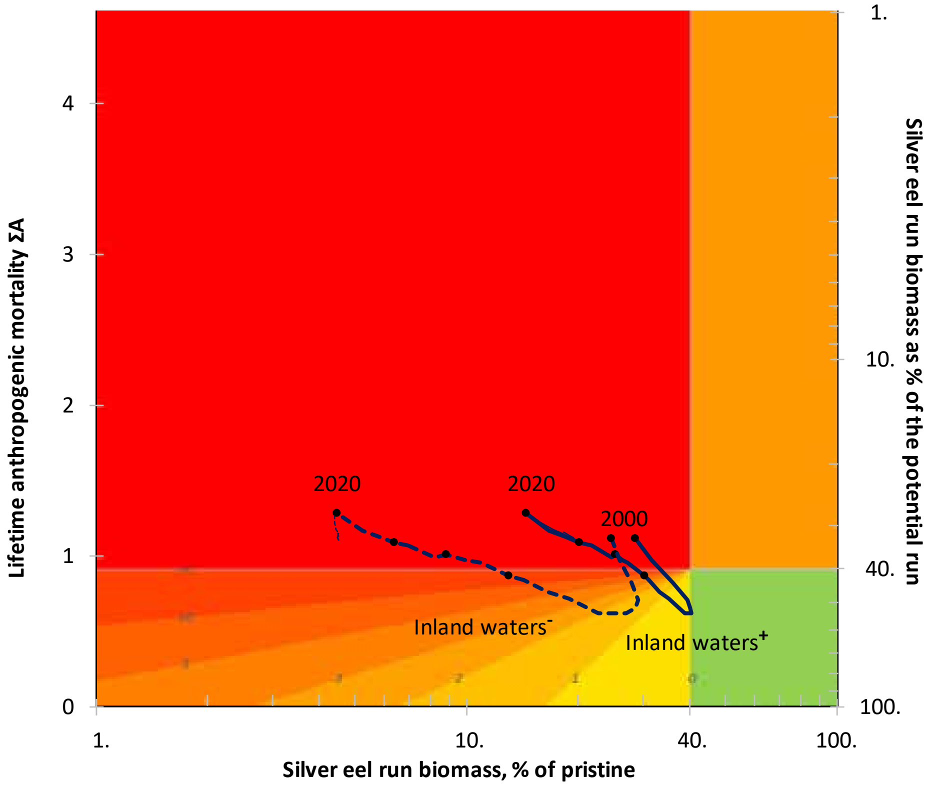

Figure 8 Precautionary Diagram for the Swedish eel stock in inland waters. For the west coast and the fisheries along the Baltic coast, no stock indicators are currently available. For inland waters, the true mortality is shown (that is: not interpreting restocking as compensation for other mortalities), 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 bullets mark the indicators in the previous assessment years 2011, 2014 and 2017. (For the details of the diagram, see Section 1.6.6 and Figure 2).

For the Baltic coast, the assessment in Annex 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 fishery only, disregarding other anthropogenic impacts in earlier life stages, 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 2017b), 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 2018-2020, the fishing mortality F_{SE} is estimated at approx. 0.003 a^{-1} (0.3%); the average landings were 115 t/a. Estimates of the silver eel run along the Swedish coast range from 180 t/a (Stockholm) to 23438 t/a (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 and 2018 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 Annex 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. For Trap & Transport, the biomass released is specified, for the West coast and the Baltic separately. 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 p. 18, above)*

	West coast						Inland waters												Baltic coast						T&T		
							with restocking +				without restocking -				Mortality rates										W	B	
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	$B_{current}$		year
2000					1.79		162	489	567	28.5	73	222	300	24.5	0.27	0.84	1.11	33.1	3507								2000
2001					2.53		184	485	581	31.6	77	204	300	25.7	0.28	0.69	0.97	37.9	3473								2001
2002					2.41		211	477	589	35.9	83	188	300	27.7	0.24	0.57	0.81	44.3	3497								2002
2003					2.15		233	468	594	39.2	87	175	300	29.0	0.23	0.47	0.70	49.7	3495								2003
2004					2.43		240	460	596	40.2	85	163	300	28.4	0.28	0.37	0.65	52.1	3516								2004
2005					2.39		240	443	594	40.4	81	149	300	26.9	0.30	0.31	0.61	54.1	3424								2005
2006					2.66		236	437	600	39.3	74	138	300	24.7	0.34	0.27	0.62	53.8	3404								2006
2007					1.91		239	443	617	38.7	68	126	300	22.6	0.30	0.32	0.62	54.0	3352								2007
2008					1.86		226	459	644	35.1	57	115	300	18.9	0.30	0.41	0.71	49.3	3381								2008
2009					1.19		222	475	669	33.2	49	106	300	16.5	0.23	0.53	0.76	46.8	3460								2009
2010					1.20		213	487	689	30.9	43	98	300	14.2	0.25	0.59	0.84	43.2	3463						5		2010
2011	12	1154	1154	1	0.93	39	212	493	702	30.1	39	91	300	13.0	0.21	0.65	0.86	42.3	3499						5	3	2011
2012					0		192	487	704	27.2	33	83	300	10.9	0.23	0.73	0.95	38.6	3531						9	2	2012
2013					0		181	464	689	26.3	30	76	300	9.9	0.24	0.73	0.97	37.9	3500						10	4	2013
2014					0		168	435	666	25.2	26	69	300	8.8	0.28	0.72	1.00	36.7	3558						15	7	2014
2015					0		158	402	639	24.6	25	63	300	8.2	0.24	0.75	0.98	37.4	3613						13	6	2015
2016					0		130	358	601	21.6	21	57	300	6.9	0.30	0.76	1.07	34.4	3590						13	6	2016
2017					0		114	317	564	20.2	19	53	300	6.4	0.36	0.72	1.08	33.9	3628						13	6	2017
2018					0		97	282	532	18.2	17	50	300	5.7	0.43	0.69	1.12	32.5	3624						11	6	2018
2019					0		84	252	505	16.7	16	47	300	5.2	0.40	0.75	1.16	31.4	3671						11	5	2019
2020					0		72	243	498	14.5	13	45	300	4.5	0.45	0.83	1.28	27.9	3670						11	8	2020

9 Discussion

9.1 Comparison to the 2018 assessment

For the west coast stock, Dekker *et al*. (2018) did not present an assessment, advocating that a comprehensive monitoring plan should be developed. Andersson *et al*. (2019) effectively did so, concluding 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 (Annex 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 Annex A updates the time series, the current assessment is essentially a repeat of the 2018 results.

For the inland stock, the current assessment updates and improves the assessment of Dekker (2015) and Dekker *et al*. (2018). This year's assessment replicates the previous methodology, and updates the data (with some additional minor corrections). As such, the current assessment is essentially a repeat of the 2018 results. The outcomes confirm the 2018 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, and rising).

For the silver eel fisheries on the Baltic coast, the current assessment methodology is identical to the 2015 and 2018 assessments; the database has just been extended. As before, estimates of fishing impact are derived, pooled by decade. However, noting that data shortage does not play such an important role as before (i.e. the re-continuation of the mark-recapture experiments in 2012 has now led to a small dataset), this report has now added more details, presenting tri-annually pooled results. 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 56) 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. Further shortening of the season – if and when that occurs – might challenge the value of future tagging experiments.

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 2021 reporting to the EU

To our knowledge, no template for reporting stock indicators has been circulated by the EU Commission, other than a letter calling upon Member States to improve the reporting on the implementation of the Eel Regulation. 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 2015 and 2018 requirements. 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). Results may inform a future national evaluation whether there is a need to revise and improve the Swedish Eel Management Plan. The presented national stock indicators were and will further be used for international evaluations (ICES 2013a, 2015, 2018 and coming), informing the international discussions. While ICES (2020) 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 west coast: 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 now show an upward trend in abundance. To achieve the management targets of the 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).

Restocking on the west coast adds upon the natural silver eel production. Though this contribution is only a small quantity in comparison to the natural pristine stock, it is probably a considerable addition to the current depleted state.

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 are increasing. 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, 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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Annex A West coast eel stock

Until quite recently, the west coast eel stock has been exploited by an extensive fyke net fishery; in spring 2012, this fishery north of 56°25’N (near Torekov, Skåne region) has been closed completely. 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 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 (Svensk Fiskeri tidskrift 1891), 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 9). 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 and 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/a in 1900-1920, to 200 t/a in early 1930 (Figure 9). Technical development of fyke nets and boats allowed catches to remain around 250 t/a, although the number of coastal fishers decreased (Figure 10). 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 and Dekker 2020). 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 9).

Most of the eel was exported (Figure 11); 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/a), but decreased again when glass eel replaced the yellow eel in the restocking programme.

Relatively low investment costs, high eel price, and good opportunities for trade generated an increased interest in eel fishery (Magnusson and Dekker 2020). In early 1900, the eel fishery was usually combined with a fishery for other coastal species, and agriculture. Catch was maintained despite decreasing number of fishers from the early to mid-1900s due to a more intensified fishery. 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.

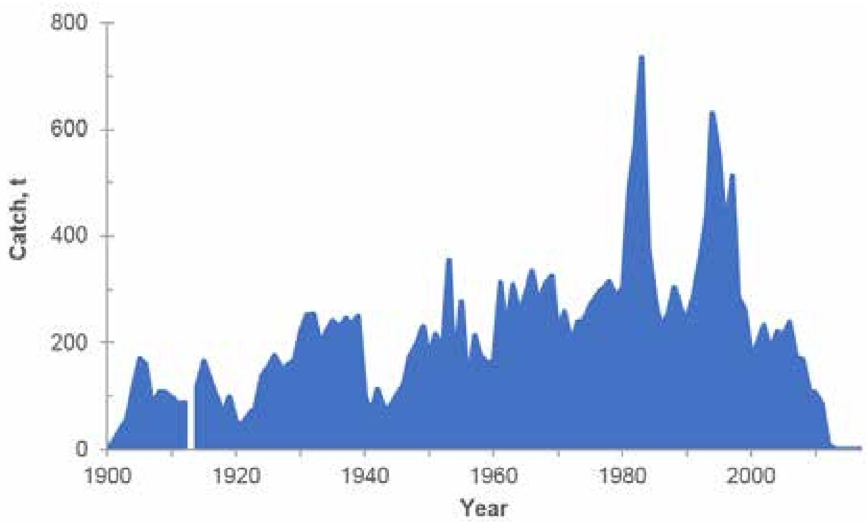

Figure 9 Time trend in eel catch in the Kattegat and Skagerrak from 1900 to 2017 (catch in the period 1970-1984 is solely based on landings data).

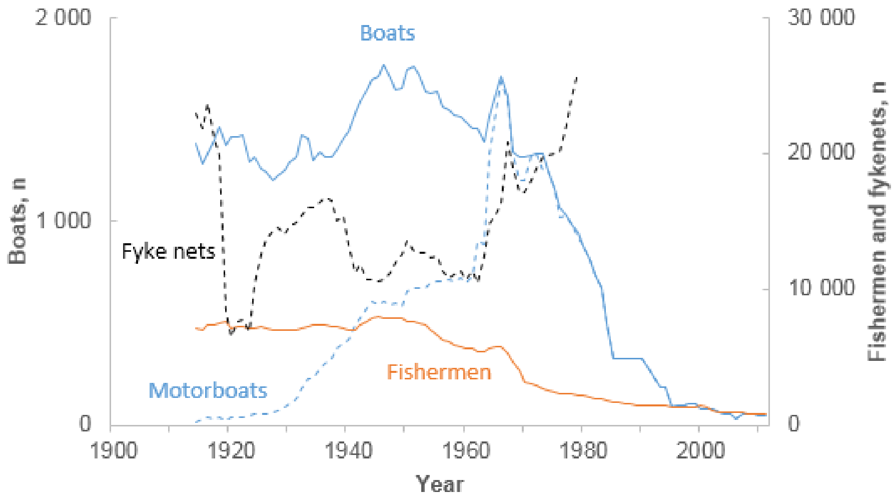

Figure 10 Time trend in number of small boats, fyke nets and fishers on the Swedish west coast.

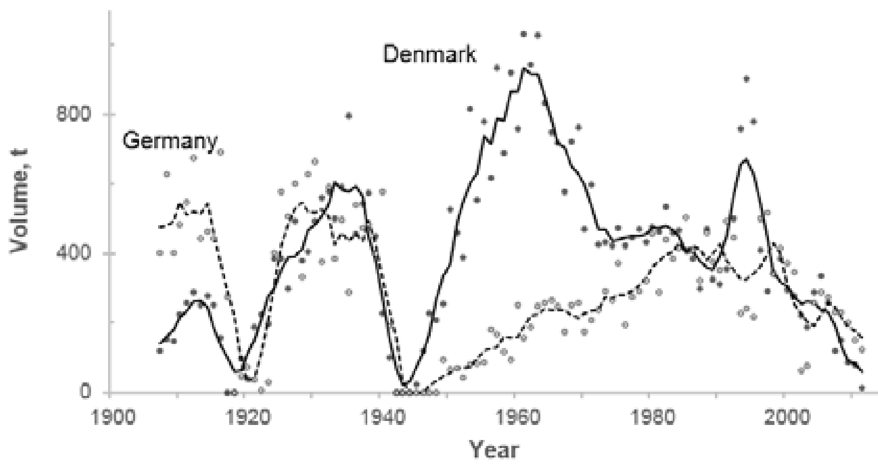

Figure 11 Time trend in total Swedish eel export to the two major receiving countries Denmark and Germany.

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 emigration of (young) eel from the west coast towards the Baltic has not been considered in past assessments; most likely, the fishery-dependent assessment has misclassified the effect of emigration 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 the EU-wide monitoring network DCF/EU-Map, in the month of August, at several sites along the Swedish west coast (Table 6). During the course of these surveys, the fyke nets are checked once every 1-4 days, and the catch is recorded. Records include the number of fish caught, as well their species 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.

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. Then, for the Barsebäck area, CPUE of each size class was calculated as the average CPUE of the five different sample site locations in the area.

The resulting CPUE time series (Figure 12) 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 show a slightly increasing trend again. 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 2020), which had been declining since the 1980s until reaching a breakpoint in 2011, after which recruitment increased again. 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, while the increasing recruitment after 2011 across Europe (ICES 2020) will also (begin to) contribute. Thus, although no definitive abundance trend can be derived here, the CPUE trends appear to show that increased recruitment and decreased fishing mortality has resulted in a current increasing trend in eel abundance along the Swedish west coast.

Table 6 Overview of the sample site locations of the fisheries-independent surveys along the Swedish west coast. The catch-per-unit-effort data of the five sample sites at Barsebäck have been averaged in Figure 12.

Area	Sample site location	First sampling
Barsebäck	Barsebäckshamn	1988
	Barsebäcksverket	1988
	Lundåkrabukten	1988
	Sjöbobadet/Golfbanan	1988
	Tjuvakroken	1988
Fjällbacka	Musön	1988
Kullen	S Skälderviken	2002
Vendelsö	Vendelsö	1988
Hakefjorden	Älgöfjorden	2002

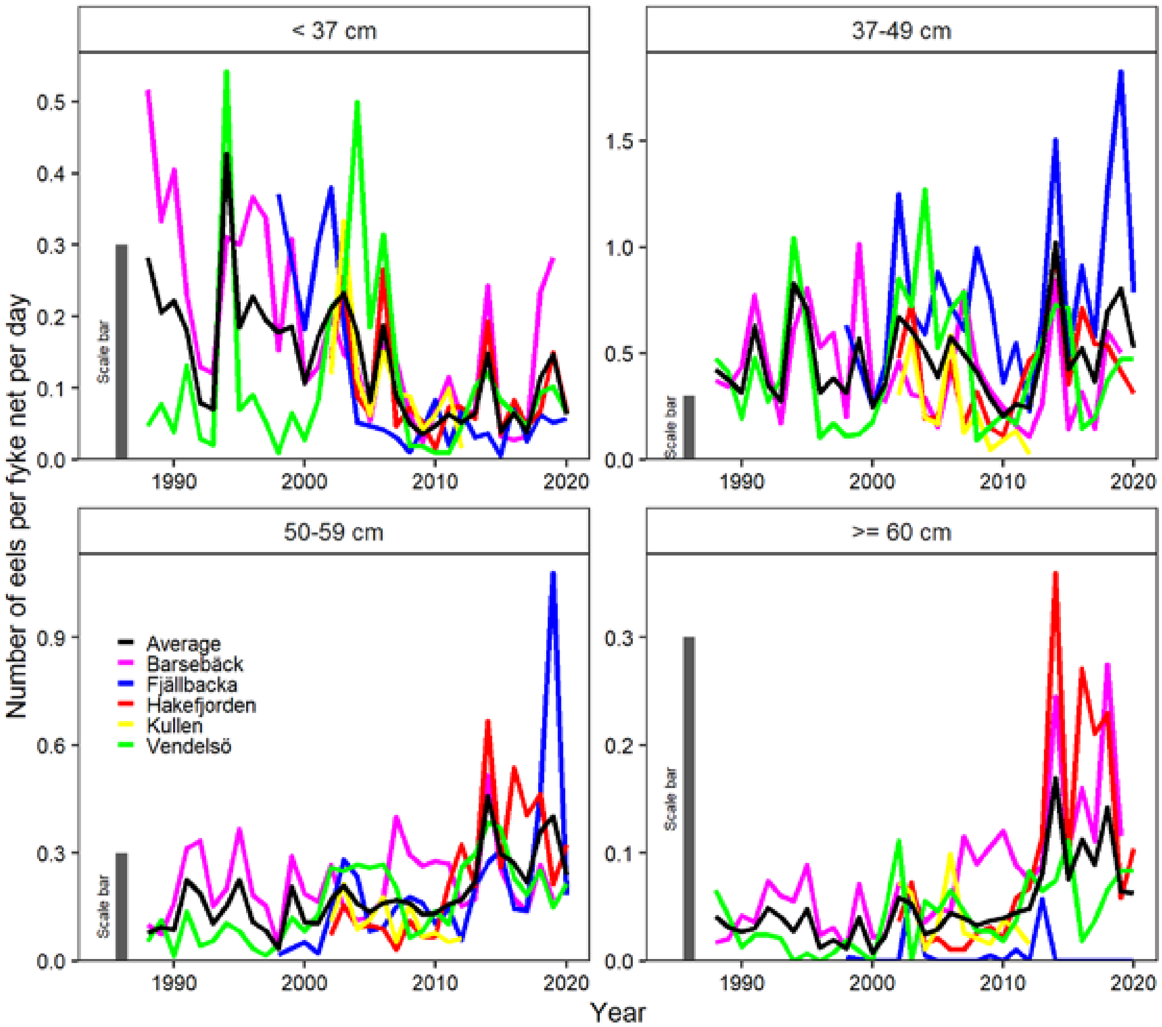

Figure 12 Catch-per-unit-effort of eel by size-class (total length) from fisheries-independent fyke net surveys at various areas along the Swedish west coast, including their average. Note that the range of the y-axis differs between subfigures, as is also indicated by the scale bar.

A.3 Restocking in coastal waters

Restocking practices have influenced the stock on the west coast in two ways. Firstly, 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. Secondly, since the mid-1970s, glass eel has been imported and released (predominantly) on the west coast (and inland waters). Until the year 2000, the amount of young eel extracted effectively exceeded the amount of glass eel released (Figure 13, extrapolate year class to year of release), but since then, the extraction has come to an end. In the 2010s, on average 1.0 million glass eels have been restocked per year. This quantity is expected to produce an amount of silver eel of ca. 60 t/a, some 15 years later. Noting that the fishing yield on the west coast was in the order of 200 t/a, and that the potential (natural) production is estimated in the order of 1000 t/a (Dekker 2012), the addition based on the restocking will be relatively small, and therefore difficult to detect.

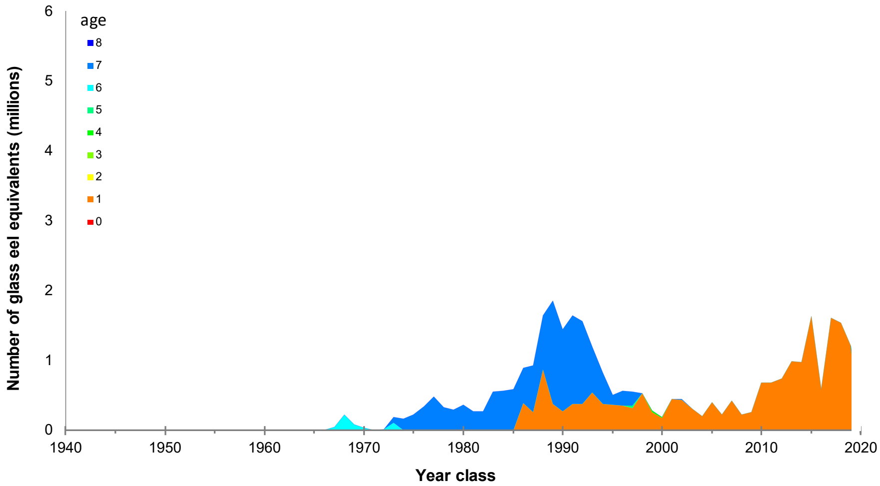

Figure 13 Time trend in the number of eel restocked in coastal waters, shown as glass eel equivalents per year class (not year of restocking). The colour indicates at what age the eels were restocked, with all numbers converted to glass eel equivalents. Before 1970, almost no eel had been restocked on the coast.

The references for this Annex are included in the reference list of the main report, on page 46.

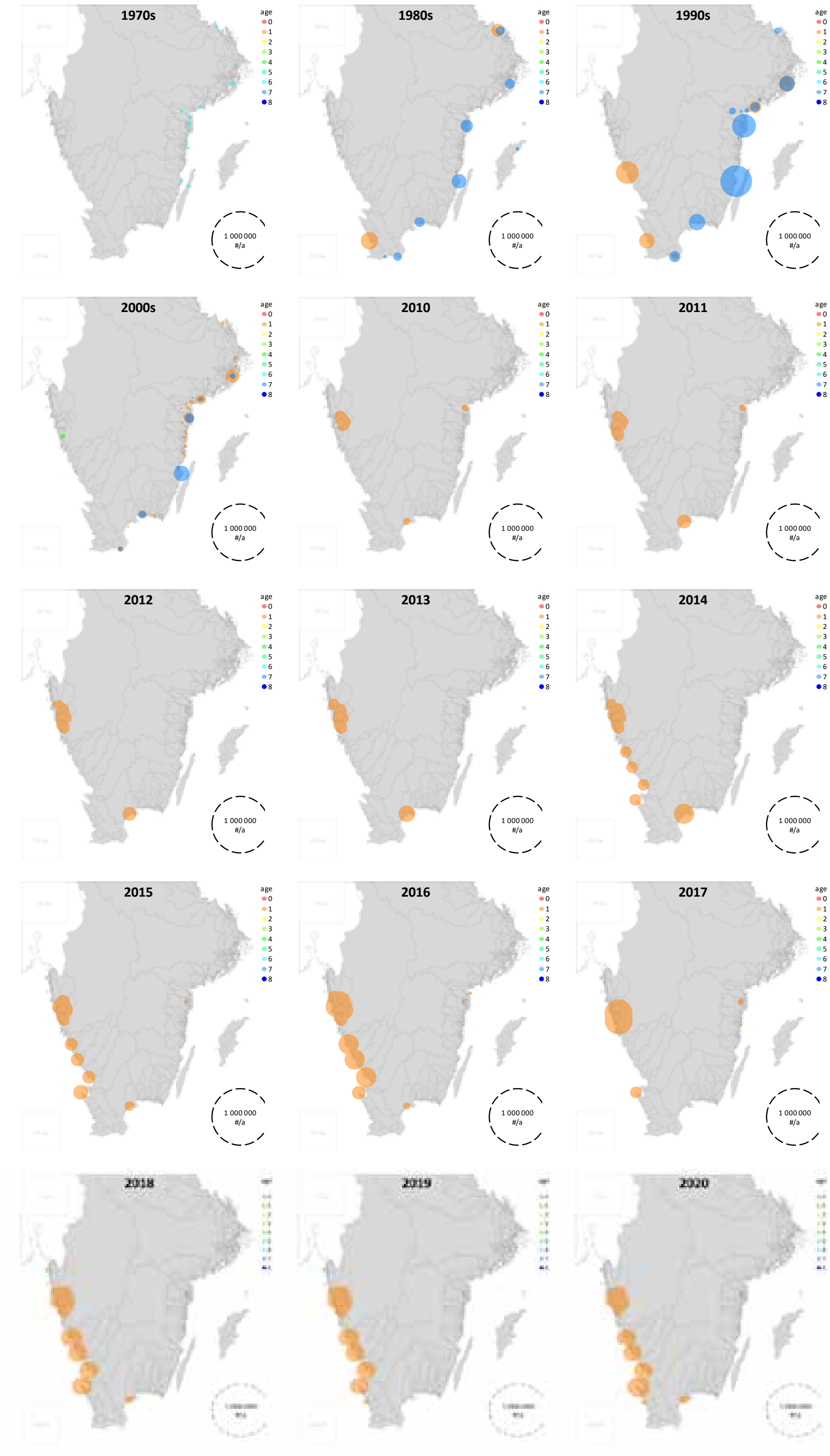

Figure 14 Spatial distribution of the restocking in coastal waters, expressed in glass eel equivalents per year. Restocking actions are shown for decades (1970s – 2010s) or individual years (2014 – 2020). The colour of the symbols indicates at what age the eels were restocked, and their size indicates the number of eel restocked in terms of 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.

Annex B Recruitment into inland waters

The reconstruction of the silver eel production in inland waters (Annex C) requires information on the natural immigration of glass eels, elvers and bootlace eels into inland waters. There is no dedicated monitoring of natural recruitment to inland waters in Sweden, but 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 Annex 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, data

In historical times, eel fisheries occurred in most inland waters in Sweden (e.g. Nordberg (1977) describes the fisheries in the river Ljungan since late-medieval times), up to the far north (Olofsson 1934 describes an eel fishery at Vändträsket in Alån, , north-west of Luleå), exploiting young eel recruiting naturally from the Baltic 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 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 15, Table 7), multi-decadal data series are available. The starting years of these series vary from before 1900 to 1991; some series were discontinued earlier (from 1973 to 1991); and seven series

[3] The estimated number of natural recruits is thus based on data related to rivers with an obstructing weir. 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.

are continued as of today. The number of concurrently operated sites rose from four in 1950 to ten in 1955, to twenty-one in the early 1970s, and then declined to around ten in the years since 1990. In the years since the previous assessment (Dekker *et al*. 2018), the number of reporting stations has declined from eleven to seven, but the stop at some of these stations might be temporary. Recorded data consist of annual catch per station, 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 Råån), converting weight to numbers using number-per-kilogram from nearby rivers (Motala ström and Rönneå, respectively).

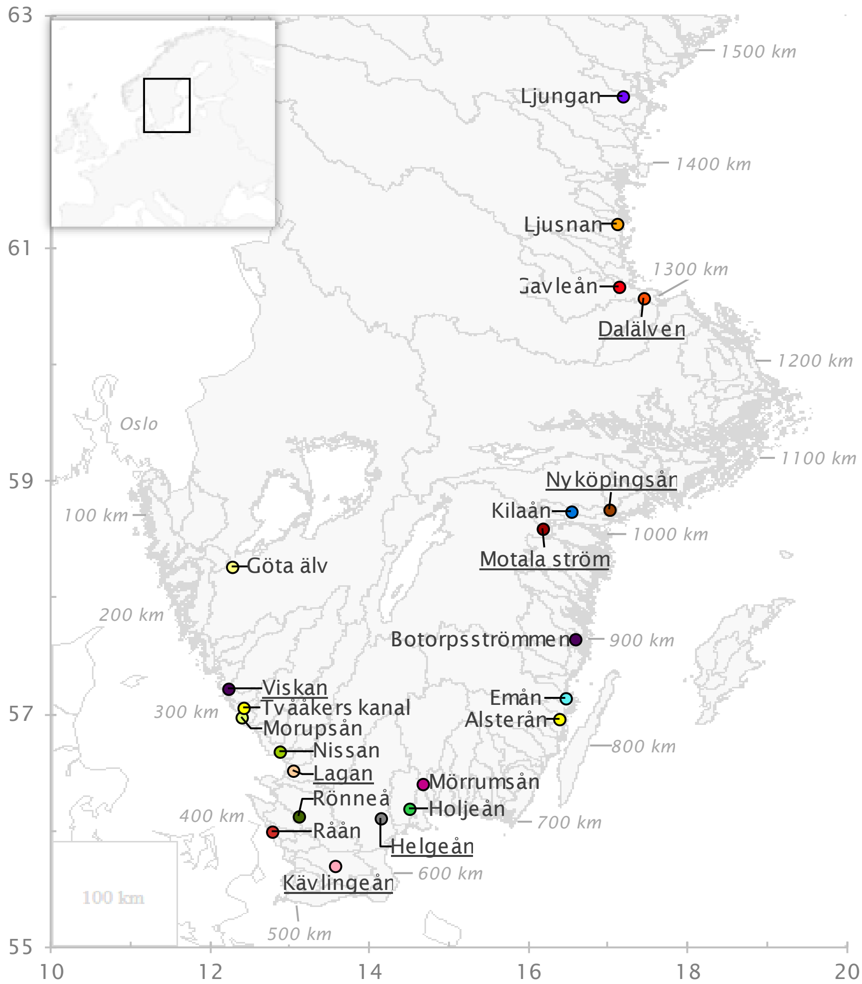

Figure 15 Map of the study area, showing sampling sites, drainage areas and distances along the coast from Oslo. Underlined sites are continuing their sampling up to today.

Table 7 Characteristics of the sites, the observation series, and the eels. The column 'Valid obs.' gives the number of 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	41.8	4.0
Ätran	1932	2012	7	317	51	6	10	0.5	1.3
Botorpsströmmen	1951	1990	37	897	6	0	6	40.5	5.1
Dalälven	1951	ctd	67	1312	348	11	14	59.3	6.0
Emån	1967	1988	21	842	30	4	13	43.8	5.4
Gavleån	1920	1979	28	1327	21	4	7	50.0	5.6
Göta älv	1900	**2017**	56	221	518	77	23	9.7	2.6
Helgeån	1952	ctd	62	623	46	35	12	31.2	2.2
Holjeån	1947	1976	24	645	8	26	20	20.9	3.9
Kävlingeån	1991	ctd	29	449	4	49	20	17.2	2.9
Kilaån	1948	1980	27	1023	1	31	19	50.0	5.6
Lagan	1925	ctd	71	363	77	4	37	0.5	0.4
Ljungan	1953	1979	23	1464	138	20	9	69.1	5.9
Ljusnan	1950	**2018**	47	1362	230	1	18	43.8	5.3
Mörrumsån†	1960	**2018**	59	663	27	32	119	98.3	6.2
Morupsån	1950	1990	38	303	1	11	11	0.4	0.0
Motala ström	1942	ctd	71	1008	93	5	11	49.8	5.6
Nissan	1947	1990	41	350	41	4	13	0.4	0.1
Nyköpingsån	1958	ctd	48	1024	22	4	11	49.8	5.6
Råån	1946	1973	23	416	2	4	13	1.8	1.1
Rönne å	1946	**2018**	60	389	24	37	31	1.8	1.1
Tvååkers kanal	1948	1989	30	303	1	7	26	0.5	0.1
Viskan	1971	ctd	50	276	35	5	1	0.5	0.1

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

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=334) 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 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 records before 1950 (n=133), the total number of valid observations, including the 35 zero observations, comes at n=942.

Characteristics of the 22 sampling sites are given in Table 7, 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 Primary results and common trend

Throughout the twentieth century, young eels have been collected and transported upstream in many rivers in Sweden. Summed over the years since 1950 and over all 22 sites, a total of more than 50 million eels have been trapped and transported - the largest shares coming from the rivers Lagan (22 million eels) and Göta älv (59 t). Catches peaked in 1953, the sum of all sites reaching 3.3 million and 10.2 t, respectively. 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. Several sites have reported to have caught not a single eel in specific years, especially in the 1960s and 1970s (Figure 16).

The absolute number of eels being caught per site varies greatly, and the time trends in these results appears to be erratic; many series have stopped reporting. The common trend in these results is hard to detect from the raw observations (Figure 16).

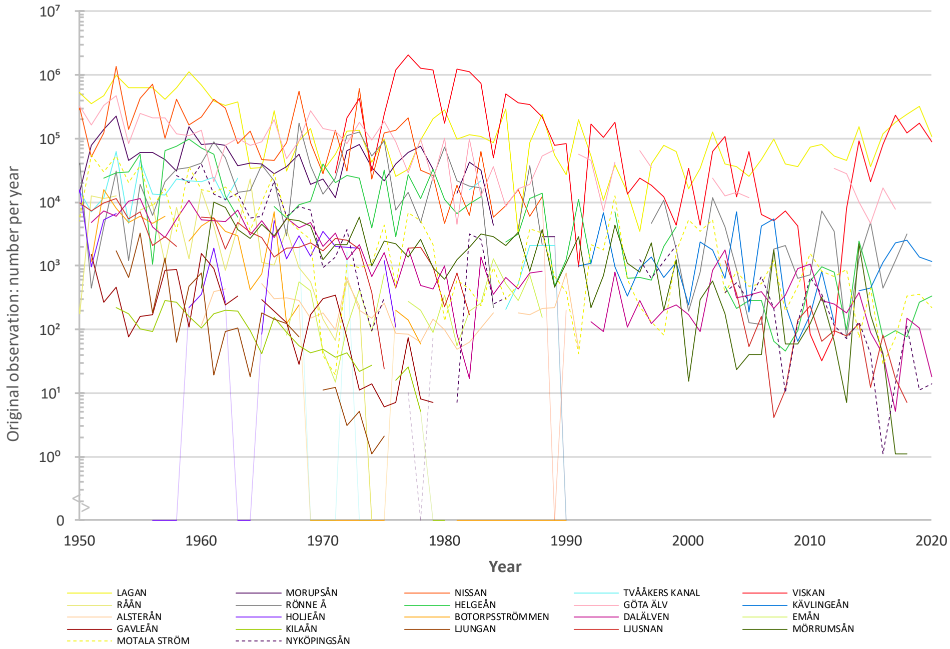

Figure 16 Primary observations of the number of young eel caught in the traps, by year and site. Where only the weight of the annual catch was reported, these have been converted to the corresponding numbers (see text). Note the logarithmic scale of the vertical axis.

Standardising across the sampling sites to a common average per year (Figure 17), however, it appears that the average number of recruiting young eels has been in decline from 1945 until 2010 (on average 7% down per year), albeit with a considerable year-to-year variation. Since 2011, an upward trend has been observed. The most-recent year classes have only just (or not yet) arrived at the more northern locations (where older eels predominate). At the more southern/western locations, the younger ages have been around for a number of years, and the trend is now clearly upwards. It is too early to say, whether the older-aged eel at the more northern locations will follow the same trend, or might continue to decline.

Inspection of the standardised series (Figure 17) indicates that zero observations (the reporting of not even a single elver, in a trap in a certain year) has occurred over all decades. Additionally, there is some indication that the older elvers (blue, in Figure 17) declined earlier than the younger ones, especially in the 1960s and 1970s. These issues will be taken up, in the development of a predictive statistical model, below.

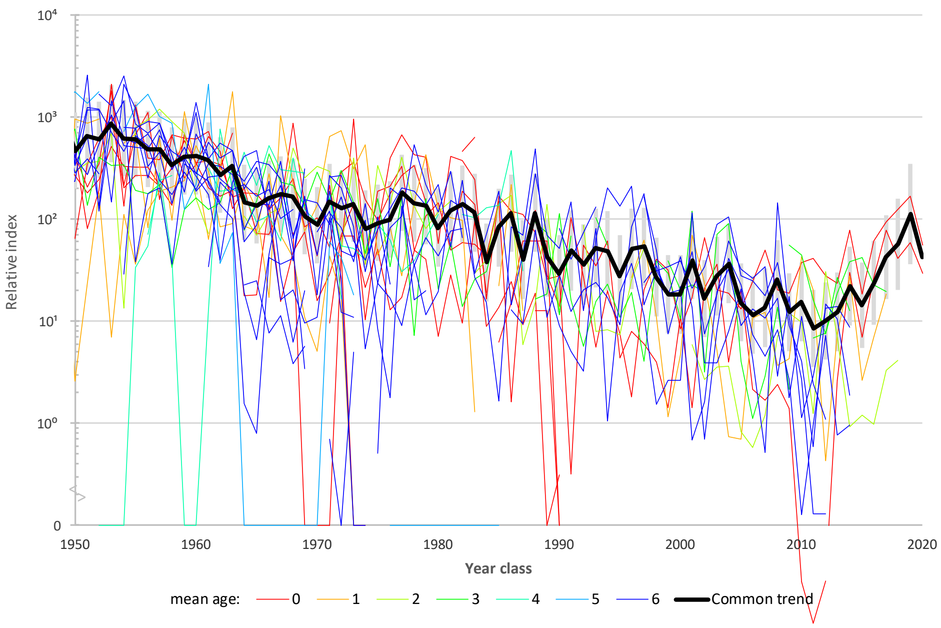

Figure 17 Standardised observations: each data series has been scaled to a mean of 100 over the whole time period. In addition, the common trend has been added (in grey: confidence interval of the mean, 1 std.). The colour of the lines indicates the mean age of the young eel being caught at each location. Note the logarithmic scale of the vertical axis.

B.1.3 Statistical analysis

The aim of the statistical analysis of the recruit time series is to describe (and test) the trends in recruitment over the years, in relation to the location along the coast (outside, or (far) inside the Baltic), to the distance upstream from the river mouth, to the (average) age of the eel, and to known site characteristics; and each of these, possibly in interaction with the time trend. By including only known site characteristics (that is: not treating '*site'* as such as an indicator of unidentified characteristics, as was done in the standardisation of the time series, shown above), we enable the use of our results for the prediction of eel abundance and trends in any other river (with or without elver trap), for the purpose of assessing the stock in all inland waters in Sweden (c.f. Dekker 2015).

We analyse the time series by a generalised linear model with a log-link and Gamma error distribution, using 'proc glimmix' of SAS (2014); this 'proc' allows fitting splines in interaction with other variables. To handle zero observations, one eel is added to all observations. Main effects in the model are:

1. 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 $W=a \cdot L^b$, where a=0.000559 and b= 3.297428, and a linear growth rate of 4.2 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). For both the main effect, and for the interactions with other explanatory variables (see below), a smooth spline over the year classes is fitted, using the default settings

of SAS: a cubic B-spline basis with three equally spaced knots positioned between the first and last year class.

2. The size of the river, coded by the mean annual discharge; in m^3/s. 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. Expecting a proportional relation between the discharge and the amount of eel caught, we include the logarithm of the discharge in our log-linear models. Unlike for the other explanatory variables, there is no hypothesis on the interaction between discharge and year class, and hence, this interaction is not included.
3. The location of the river, (far) outside or (far) inside the Baltic, coded as the shortest distance from Oslo to the river mouth; in km. For each location, the length of the convex hull around the coastline of southern Sweden was calculated on the map supplied by SAS (2014). We include the distance-from-Oslo in the log-linear predictor, that is: an exponential decline in eel numbers with increasing distance-from-Oslo.
4. The location of the trap within the river, coded by the distance upstream, from the river mouth towards the trap, derived from the GIS databases of SMHI (2014); in km. We include the distance-upstream in the log-linear predictor, that is: an exponential decline in eel numbers with increasing distance-upstream.
5. The average Age of the eel, derived from the observed mean weight per site, as specified above (age itself was not measured directly); in years. We include the age as a continuous covariate in the log-linear predictor, that is: an exponential decline in eel numbers with increasing age.

For each of the variables above, except for year class, the number of independent observations is very restricted: only 22 different values occur: one for each site, repeated exactly in all observation years. Because of that limited number of distinct values, we fit simple linear relations for these variables; preliminary model runs fitting even slightly more flexible relations (a spline, as specified for year class) resulted in unrealistic predictions at intermediate values, in-between the 22 observations, up to several orders of magnitude above or below the observations. For year class, a total of 71 equally-spaced observations occurs, repeated over (max) 22 sites. Since the catch in any year at any site contains several age groups and year classes, a smooth trend over the years is expected, disrupted by unpredictable local effects. Hence we fit a spline over the year classes (six degrees of freedom), both for the main effect, and for the interactions with other variables. Preliminary model runs fitting a class variable for the main effect (71 degrees of freedom, allowing irregular variation from year to year) added less than 1 % to the explained deviance, and did not lead to contradictory conclusions. Preliminary model runs treating year class as a class variable in the interactions too, exhausted the available information considerably, and hence, did not result in any statistically significant outcome.

The immigrating eels observed at sites further into the Baltic tend to be older and larger than the ones near the outlet. Hence, age is well correlated with distance-to-Oslo (R^2=0.733; $p<0.0001$). Distance-upstream shows no such relation to age (R^2=0.001; p=0.86); all other correlations between explanatory variables are small and insignificant.

The trap in the River Mörrumsån, however, is exceptional: it is located at 663 km from Oslo, where – by comparison to other sites - an average individual size of approx. 30 gram would be expected, but a size of around 100 grams is observed. Most likely, the altitude of 78-119 m at the Mörrumsån traps (in contrast to an altitude of 1 – 37 m for all other sites) is slowing down the upriver migration by some years, giving the eel time to grow.

For each of the main effects, a partial residual plot is shown (Figure 18 and Figure 19), giving partial predictions (for the first year class in each decade) and partial residuals (for each observation, whatever the year class); although year class 2020 belongs to a new decade, this single year class is not shown separately as a decade of its own, but included in the preceding decade (it would show erratic results, due to the extremely low number of observations involved). For these plots, all main effects, except the explanatory variable under consideration, were set at a rounded value close to their average (discharge=100 m^3/s; distance-to-Oslo=700 km; distance upstream=20 km; age=3) and (partial) predicted values calculated for each of the so standardised observations. Partial residuals were then calculated as the partial prediction multiplied by the antilog of the observed deviance residual.

B.2 Results

The year-to-year variation has been considerable at all sites (Figure 16, Figure 17), with an inter-quartile range for individual observations of 46 % - 260 % relative to the previous year's observation at the same site. Fitting a main-effects model (spline(year class) + log(discharge) + distance-from-Oslo + distance-upstream + age) explains 7 % of the total deviance; adding interactions between spline(year class) and respectively distance-from-Oslo, distance-upstream and age, taken together, explains less than 1 % extra. The interaction between distance-from-Oslo and spline(year class) is not statistically significant; the other interactions are. Results and model diagnostics are shown below, with all interactions in the model, even the insignificant interaction with distance-from-Oslo.

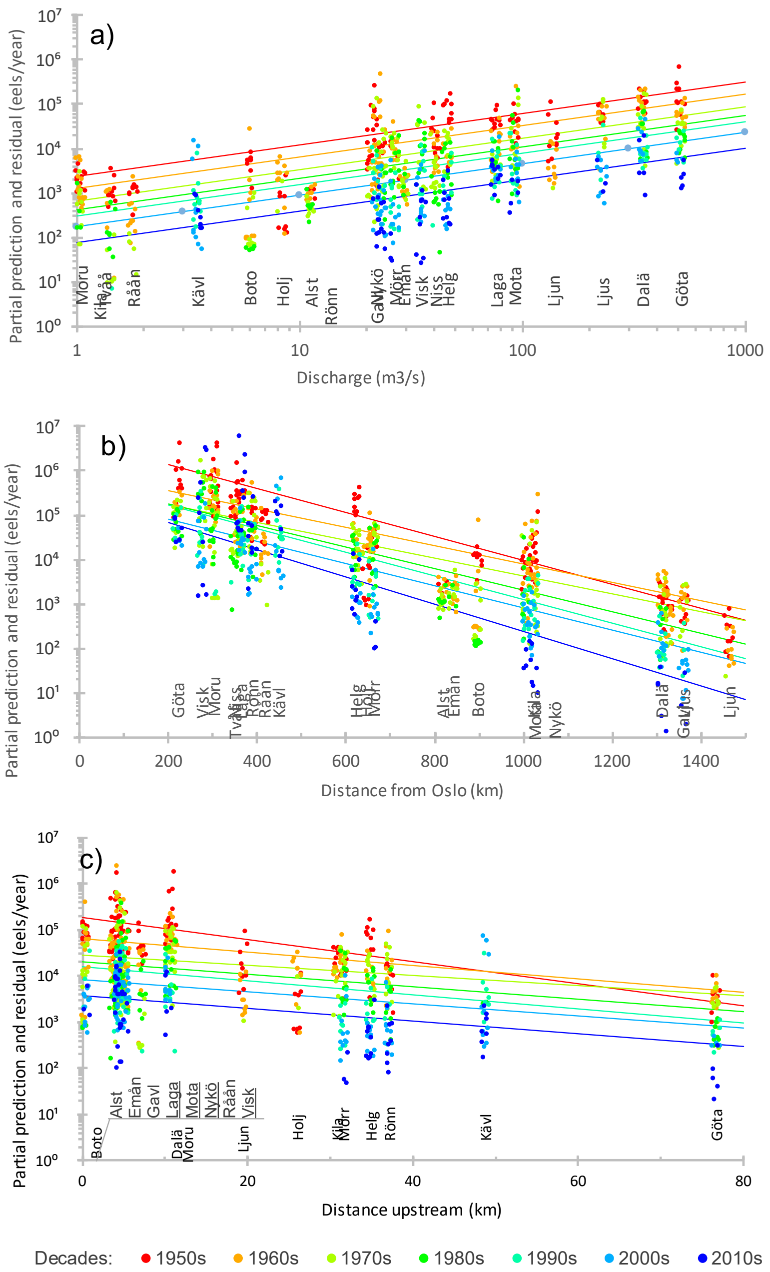

Figure 18 Partial predictions and partial residuals, by year class; for a) Discharge, b) Distance-from-Oslo, and c) Distance-upstream. Though partial residuals have been calculated for each individual year class, the colours in this plot apply to whole decades. Partial predictions (regression lines) are given for the first year of each decade only (1950, 1960…). For clarity, all dots have been displaced horizontally by a horizontal random jitter of max ±5 % of the discharge, resp. ±10 km from Oslo and ±0.5 km upstream. The position of each sampling site has been indicated along the bottom; site names have been shortened to four characters (see Table 7).

The number of eels trapped per year is positively related to the discharge at the site of capture (Figure 18.a), but the relation is less than proportional; rather, the quantity is related to discharge$^{0.688}$. Our analysis did not test whether the relation to discharge changed over the decades. Inspection of the partial residuals (Figure 18.a) indicates that the smallest streams Morupsån and Kilaån - both heavily modified little streams in an agricultural landscape - reported catches considerably above the statistical expectation. For discharges up to 10 m^3/s, the partial residuals show hardly any relation between river discharge and the number of eels, while for discharges above 10 m^3/s, the relation is more close to proportionality. It seems quite likely that the Morupsån and Kilaån sites have been selected for their higher catches, despite their small river size. If so, Morupsån and Kilaån are not representative for other small rivers, with low discharge. If we would allow for a non-linear relation between discharge and eel catch (as the partial residuals suggest), we would predict a considerable recruitment of young eel into many small rivers all over the country – which we do not believe to be real. Instead, we fit the linear relation, as shown.

For the site position in the Baltic, a steep reduction in eel abundance is observed with increasing distance-from-Oslo (Figure 18.b) - declining 152- to 4348-fold over 1300 km, depending on the decade. Expecting a decline first and foremost at the sites furthest into the Baltic, the decrease appears to have started at the other end, at the sites more close to Oslo, and only recently at the sites further into the Baltic. The trend with increasing distance into the Baltic is statistically significant, but the change in this trend over the decades is not.

The number of eel caught decreases with the Distance-upstream of the trapping site (Figure 18.c), numbers decreasing 2- to 35-fold over 80 km distance upstream, depending on the decade. Expecting a decline first and foremost at the sites furthest into the river, the upriver trend appears to change over the decades in a rather erratic way, going up and down without a clear trend.

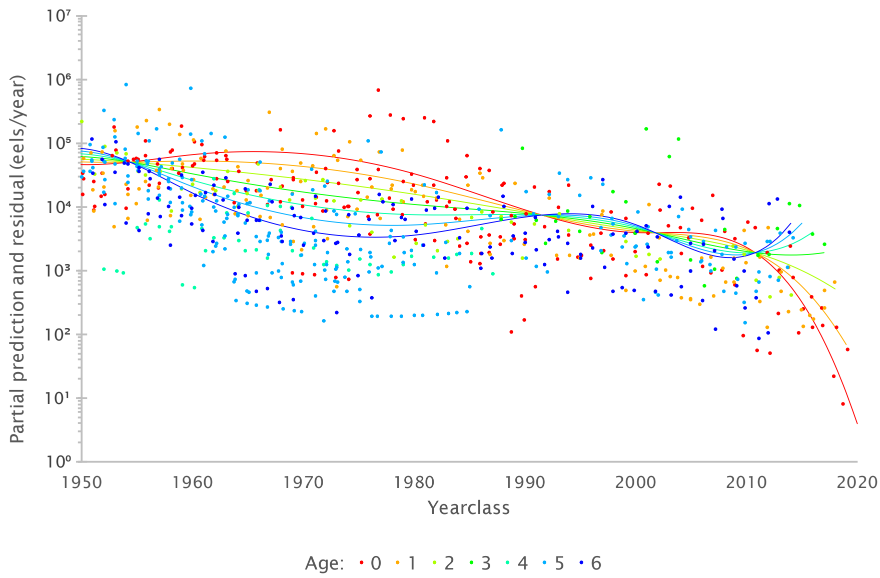

Figure 19 Partial predictions and partial residuals per year class, by mean Age (in interaction). Unlike the other plots, the colour in this plot codes for the (rounded) mean Age at each site - not for decades. For clarity, all dots have been displaced horizontally by a horizontal random jitter of ±0.25 years max.

The relation between eel abundance, mean Age in the catch and the year class is shown in Figure 19. In the 1950s and 1960s, the number of older eels caught in the traps declined

40- to 60-fold, while the number of youngest eels remained at a high level. In later decades, younger and younger ages followed, with the youngest ages declining foremost in the late-1970s through to the 1990s, decreasing about 15-fold from 1970 to 2000. In the years after 2010, the youngest age groups have shown an increase in abundance, but that upturn has not had time to progress fully into the older ages yet. The regression model, fitting smooth functions, does not pick up that signal yet (see discussion below).

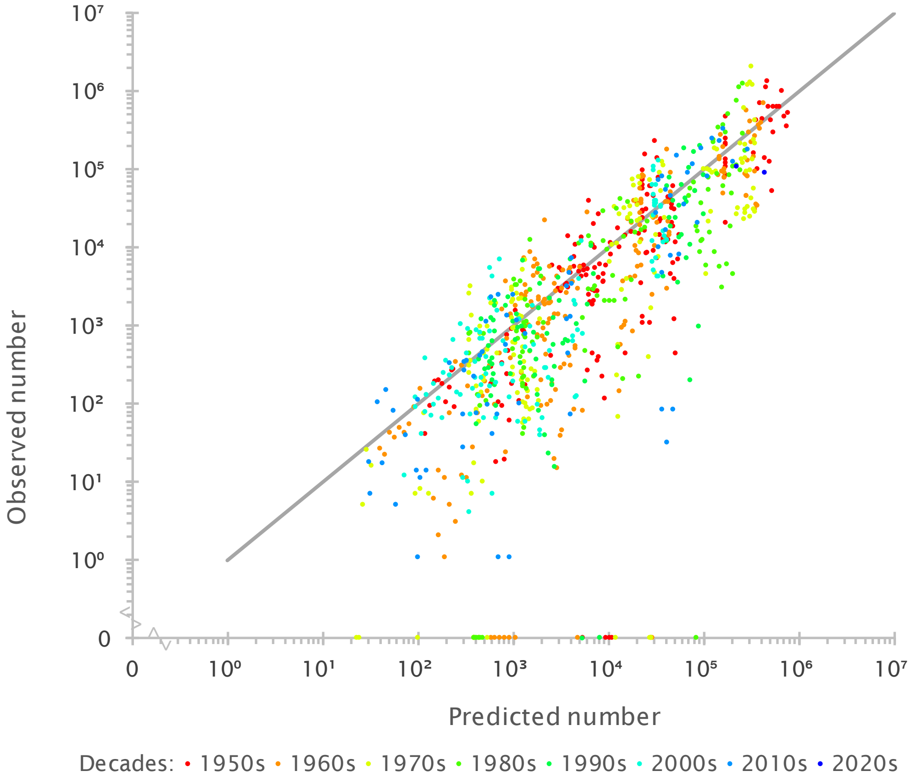

Figure 20 Relation between observed values and values predicted by the statistical model, coloured by decade. The solid line represents the main diagonal, where observed and predicted values are equal.

Model diagnostics (not shown) did not reveal statistical problems, except for the relation between observed and predicted numbers, specifically at low abundance. While a strict proportionality is expected, Figure 20 indicates that - below a predicted number of approximately a hundred to a thousand eels - observations are increasingly below the expectation; these low observations stem predominantly from the 1970s, a few from the 2010s. Zero observations occur below an *expected* number of 10^5 eels, especially below 10^3. Detailed inspection of these zero- and unexpectedly-low observations indicates, that most of these occur in years shortly before observation series were stopped (Figure 21.bottom). In the last five years before data series stopped, no single observation reached the statistically expected number (except Morupsån 1986, at four years before the end of this series, following a year of non-operation of the trap). Otherwise, results did not show any relation to either the seniority of the observation series (Figure 21.top), or their further longevity (Figure 21.bottom). In our interpretation, this probably indicates that – when catches were somewhat lower than what was hoped for - the operation of the traps might have slackened (emptying the traps less often during the season, or cleaning the traps less thoroughly), which in itself led to a further reduction in the catch, which then in turn led to the decision to stop the trapping altogether – a self-fulfilling circular process. A sufficiently high catch might be required to motivate the continued successful operation of the trap, and vice versa: a disappointingly low catch could easily initiate a self-destructive negligence of the trap operation.

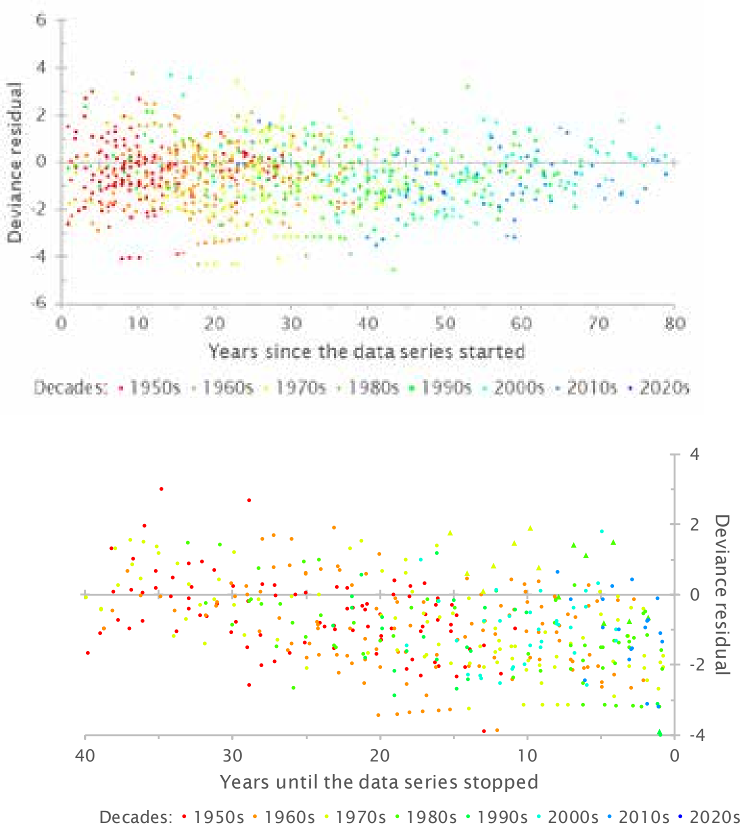

Figure 21 Partial residuals plotted as a function of (top) the number of years since the data series was started, resp. (bottom) the number of years until the data series was stopped; note that neither of these numbers of years is included in the analysis model. The bottom panel includes only the data series that stopped before the final year 2020. For clarity, all dots have been displaced by a horizontal random jitter of ±0.25 years max.

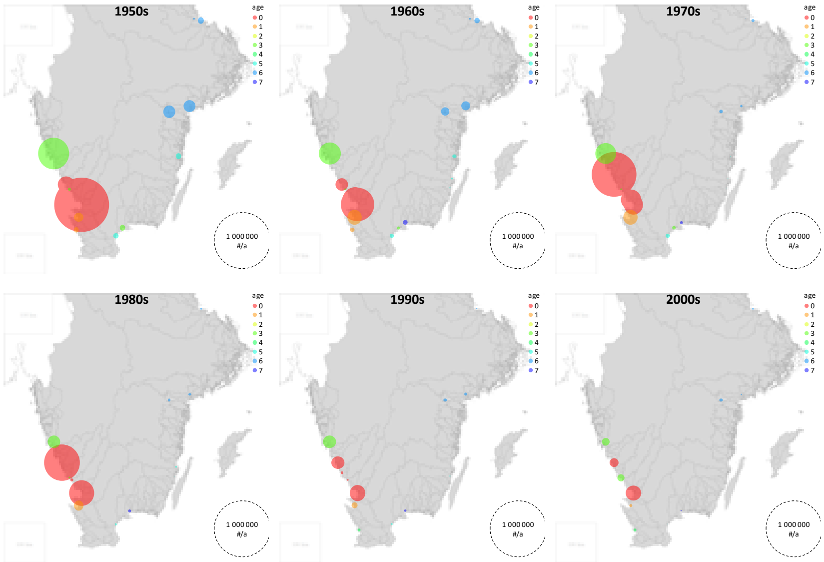

Figure 22 Spatial distribution of the <u>observed numbers of elvers</u> caught in the traps, averaged per decade, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class.

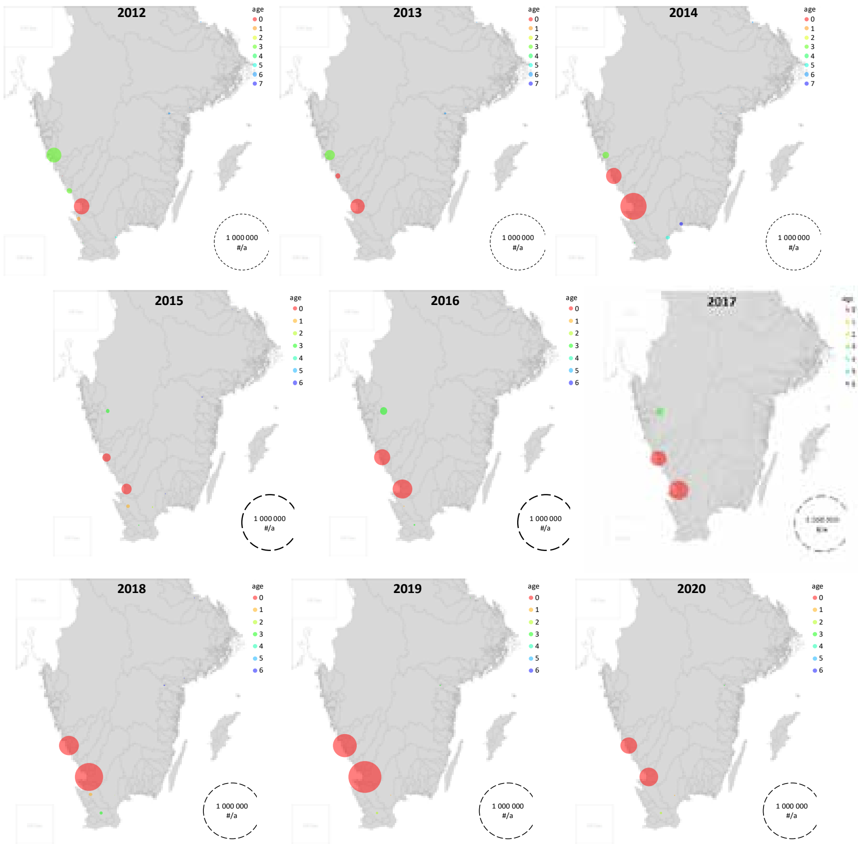

Figure 23 Spatial distribution of the <u>observed numbers of elvers</u> caught in the traps, in the years 2012-2020, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class. The numbers at many locations are that low, that the symbols become invisible in these maps.

B.3 Extrapolating trends in natural recruitment

The reconstruction of the inland silver eel production (Annex 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 in all years passed, and for a range of coming years (during which current management actions gain their full effects). For the recruitment in rivers without a trap in earlier years, the current analysis results in quite plausible predictions. For the predictions of the most recent years and forward in time, however, aberrant predictions are obtained, due to the limited amount of information available for the very last year classes. For the very last year class, only very few observations are available (Viskan and Lagan); 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 (2015-2020) only in the time still coming. For those most recent year classes, the model is relatively over-specified. Extrapolating those most-recent year classes thus results in implausible estimates (Dekker *et al*. 2018 worked these out in detail; see also the implausible trend in the youngest ages in most recent years, in Figure 19). Because of this, the model is lending itself badly for extrapolation forward in time (based on the very last year class), and produces implausible results for the more northern rivers in most recent years.

Year class 2012 is the last year class already recruited at all trapping sites, and therefore, this year class is selected as the basis for extrapolation forward in time. Selecting year class 2012 as the basis for this extrapolation, however, implies that the extrapolation largely misses the breakpoint in 2011, and the upward trend since is not taken into account. On the one hand, the extrapolation will be based on a weaker year class than the most recent ones (year class 2019 was approximately ten times stronger than year class 2010), and consequently, the extrapolation will be conservative. On the other hand, the future trend in natural recruitment is uncertain anyhow (we assume no further change will occur, while – on a very subjective basis - a continued trend towards higher numbers seems more plausible), and the extrapolation serves no other point than showing the full (life-time) effect of current management measures. Whatever reference year we would have chosen, the extrapolation results would have had limited value.

The references for this Annex are included in the reference list of the main report, on page 46.

Annex 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 six 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 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 glass eel equivalents:

$$glass\ eel\ equivalents_{year-age} = number_{year,age} \times exp^{+M \times age}$$

where year = the year the observation was made, age = the mean age of the eels, number is the number of recruiting eels, and M = natural mortality between the glass eel and the immigrating stage. For M, an average value of 0.10 per year 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, the Trap & Transport programme is a source of eel for the inland stock, albeit consisting of fully-grown silver eel released at the outer margin of the inland waters rather than youngsters released within.

Natural recruitment

The statistical analysis of Annex B estimates 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 annex. 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, natural recruitment is estimated by the statistical prediction, not by the actual observation in the elver trap – a consistent approach across all rivers, yielding an estimate even in the years that a trap was not operated (e.g.: during hydropower repair works). In many cases, the actual catch exceeded the statistical prediction (i.e. a positive residual, on theoretical grounds expected in half the number of cases). The removal of trapped eels for assisted migration then leads to a negative estimate of the remaining local stock size at the trapping location. For the whole drainage area, however, the sum of the negative stock abundance estimate at the trap and the increased abundance at the point of release leads to a non-negative estimate for the area as a whole.

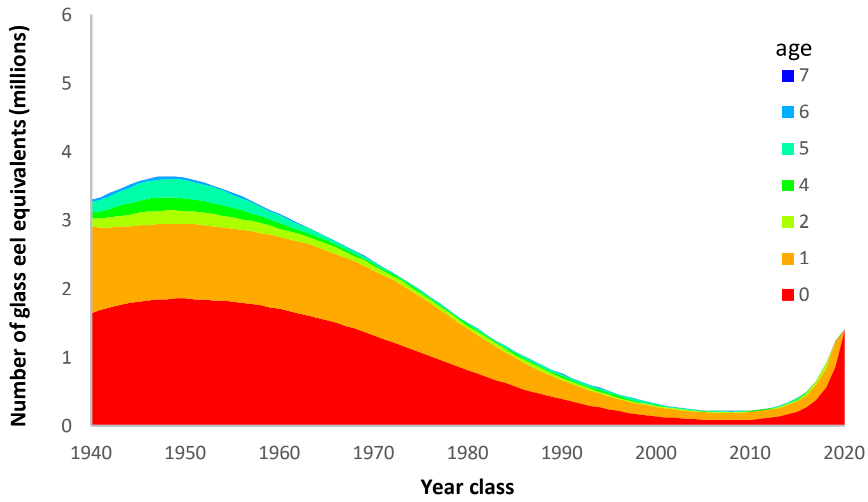

Figure 24 Time trend in the estimated number of naturally recruiting eels, expressed as glass eel equivalents per year class.

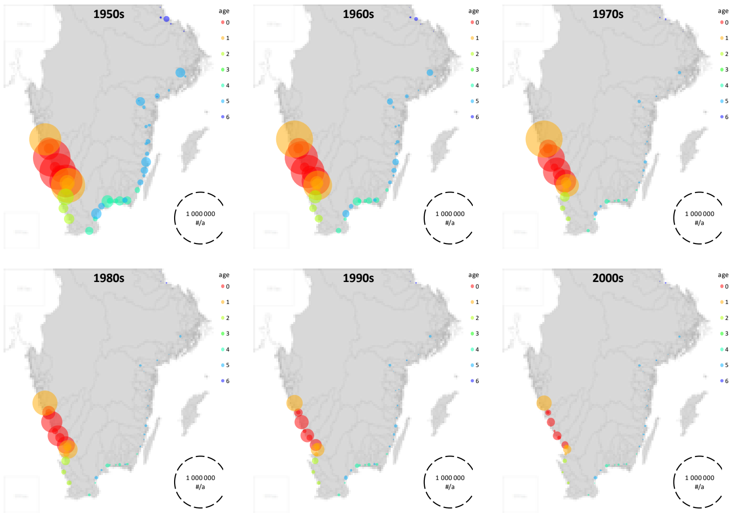

Figure 25 Spatial distribution of the estimates of natural recruitment, per decade, expressed in glass eel equivalents. These plots show the total number per decade (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the immigration took place, not by year class.

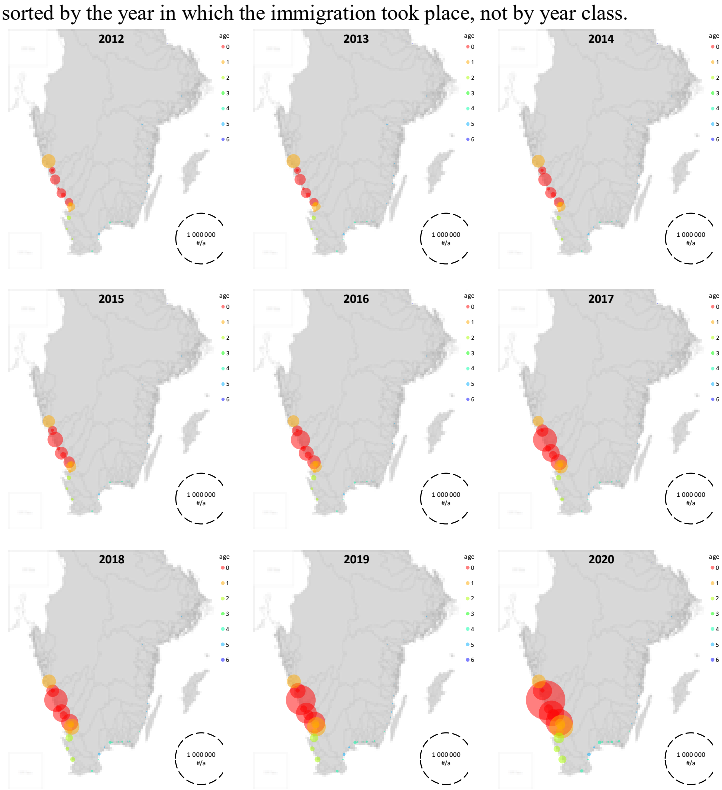

Figure 26 Spatial distribution of the estimates of natural recruitment, in the years 2012-2020, expressed in glass eel equivalents. These plots show the total number per year (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the immigration took place, not by year class.

Assisted migration

A database of historical transports of young eels across barriers in rivers is held at SLU-Aqua, specifying site, year, 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 (p. 87).

Trapping of young eels was (and is) often 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 (11 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.

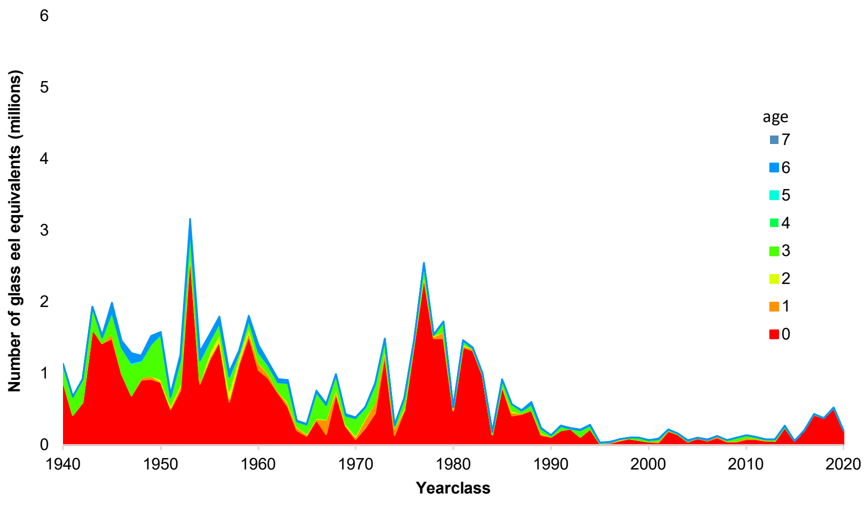

Figure 27 Time trend in the number of eels released from assisted migration. Though this plot is subdivided by age of the eel, all quantities are expressed in glass eel equivalents per year class.

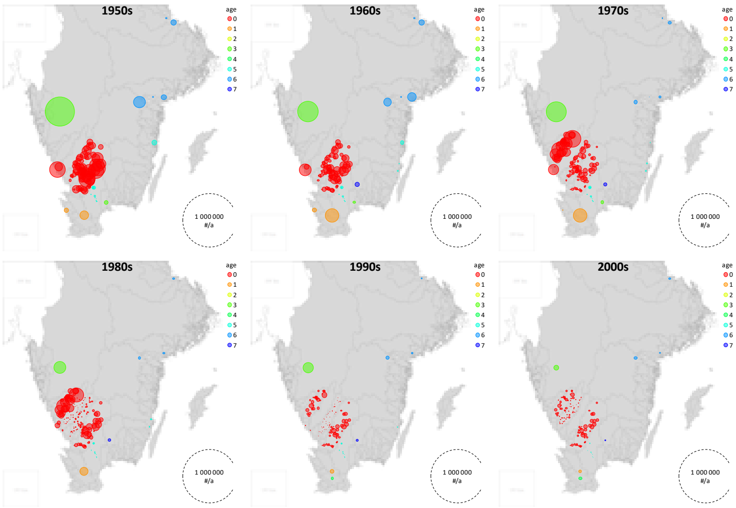

Figure 28 Spatial distribution of the release from assisted migration, per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that the figures are sorted by the year in which the release took place, not by year class.

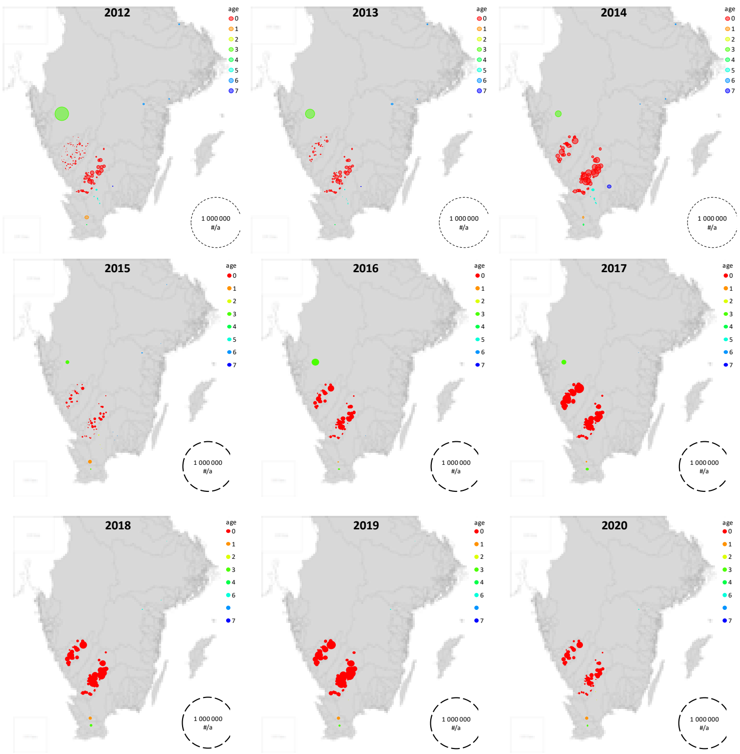

Figure 29 Spatial distribution of the release from assisted migration, in the years 2012-2020, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the release took place, not by year class.

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 rivers have been restocked.

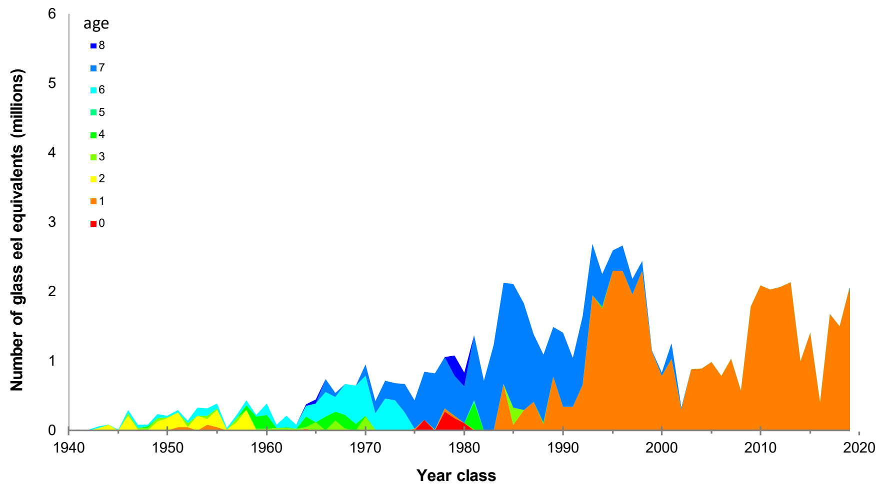

Figure 30 Time trend in the numbers of eel used for restocking. Though this plot is subdivided by age of the restocking material, all quantities are expressed in glass eel equivalents per year class.

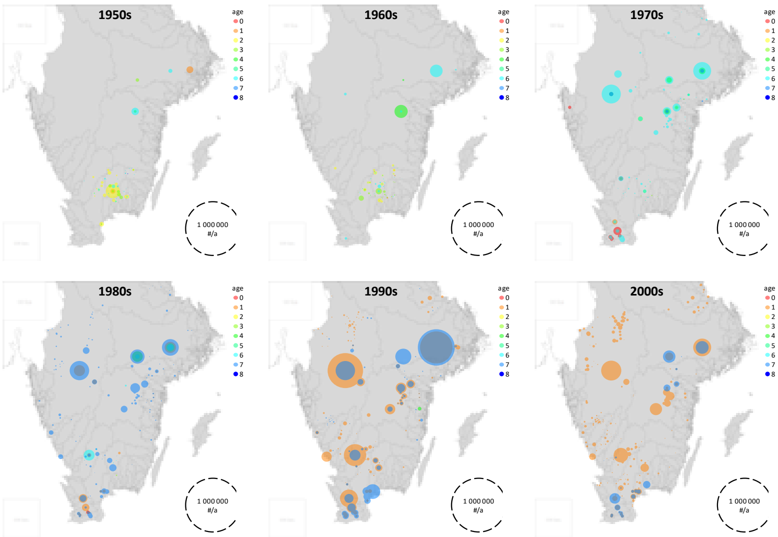

Figure 31 Spatial distribution of the restocking per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that these figures are sorted by the year in which the restocking actually took place, not by year class.

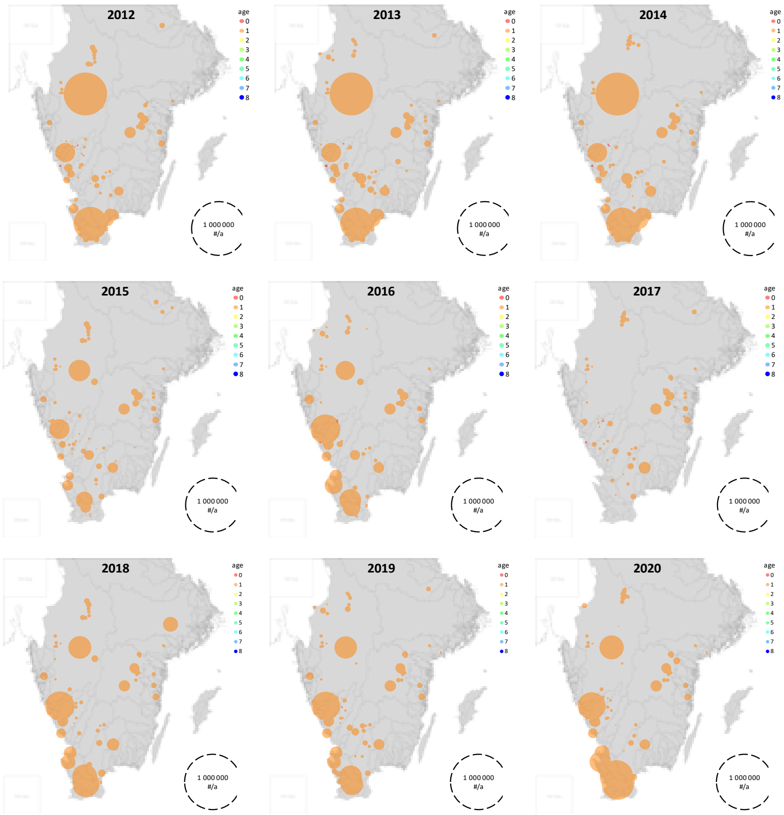

Figure 32 Spatial distribution of the restocking in the years 2012-2020, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the restocking took place, not by year class.

Trap & Transport of silver eel

In recent years, silver eel from lakes situated above hydropower generation plants has been trapped and transported downstream by lorry, bypassing the hydropower-related mortality. 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 (see below), and data have been collected and processed accordingly. The release of silver eel downstream, however, often occurs just outside the area considered in this reconstruction. 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 assessment, though results are reported separately from the silver eel escaping directly from the inland waters to the sea.

Table 8 Quantities of silver eel in the Trap & Transport programmes, in biomass (kg)

River year	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Motala ström		676	1 283	3 167	5 931	4 821	5 141	4 894	4 629	4 573	5 171
Mörrumsån		1 883	154	269	329	938	327	343	943		2 144
Helgeån											153
Kävlingeån		548	167	325	909	241	544	445	669	248	384
Rönne Å					415	250	316	541			
Lagan	365	367	110	921	1 484	681	866	1 111	586	923	769
Nissan						83	96	334	154		187
Ätran			295	96	292	130	14	257	24		45
Göta älv	4 841	4 499	8 237	9 393	12 417	11 890	11 743	10 448	10 173	9 870	9 606
Total	5 206	7 973	10 246	14 171	21 777	19 035	19 046	18 373	17 179	15 614	18 459

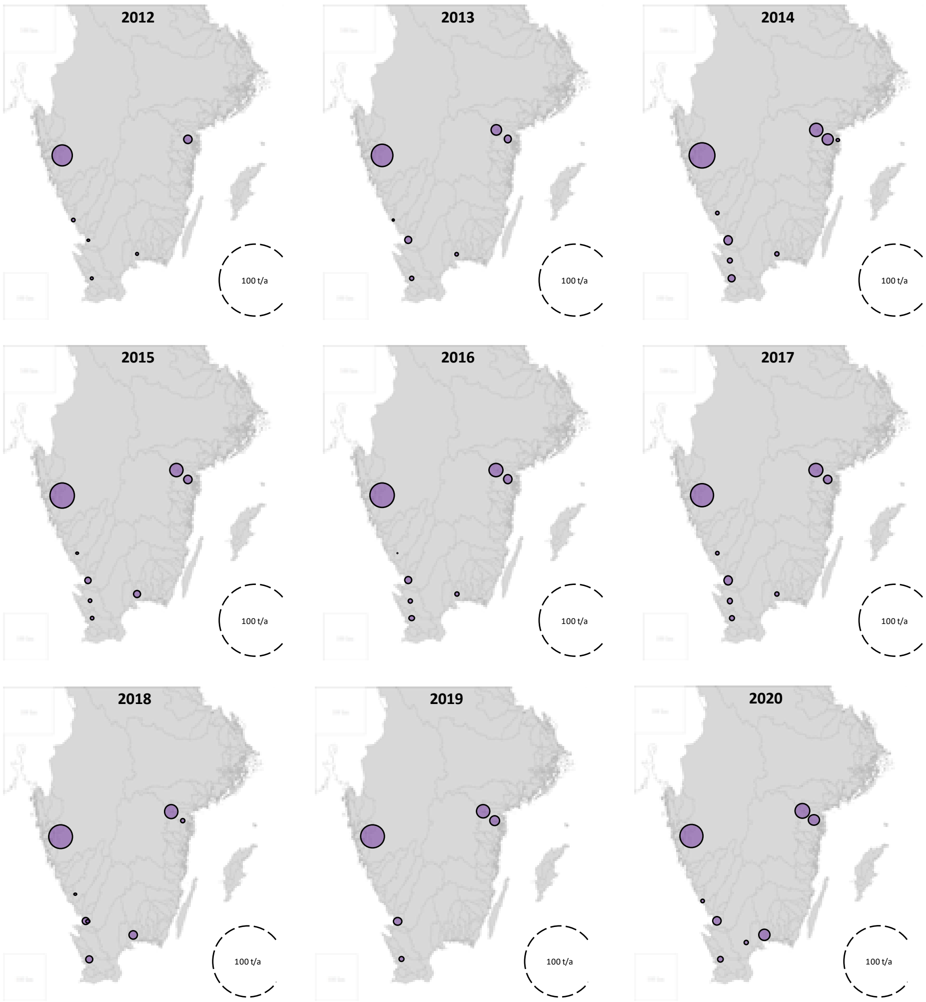

Figure 33 Spatial distribution of the releases from the Trap & Transport programmes, in the years 2012-2020.

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; before 1960, landings were extremely low, probably negligible in comparison to the rest of the inland fisheries (Figure 34). Elsewhere, data are available per lake and/or for varying groups of lakes (Figure 35). In summing across lakes, one has grouped many different sets, sometimes even spanning different drainage areas. 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).

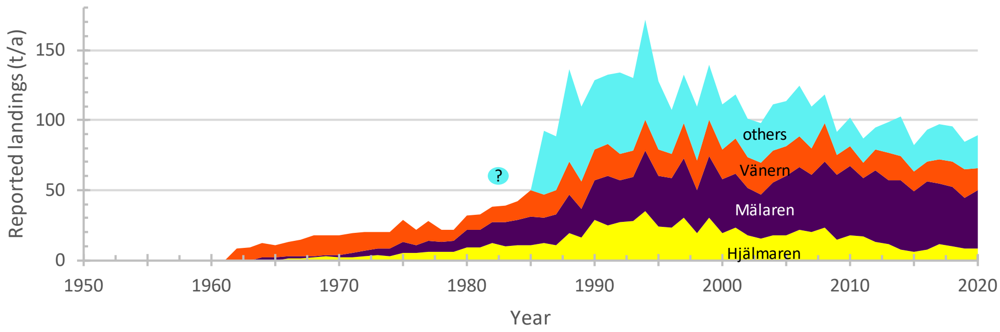

Figure 34 Time trend in the reported landings from the fishery, for the larger lakes, and years since 1950. For smaller lakes, no data are available before 1986.

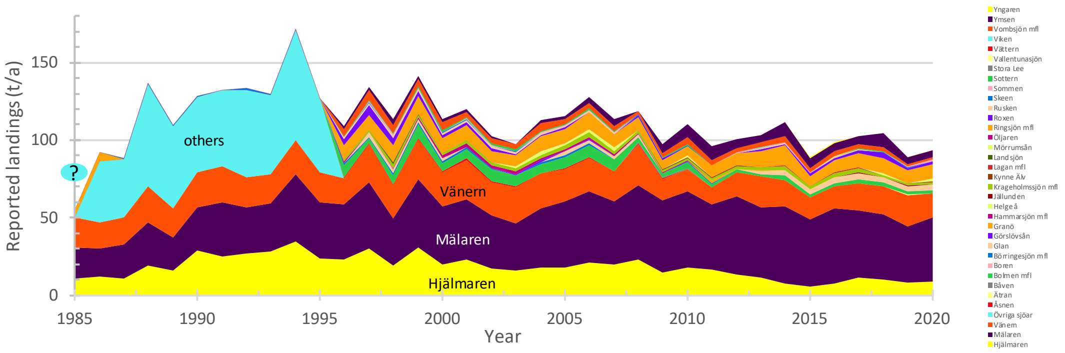

Figure 35 Time trend in the reported landings from the fishery, for all lakes, and years since 1985. Note the time interval on the horizontal axis, deviating from most other figures.

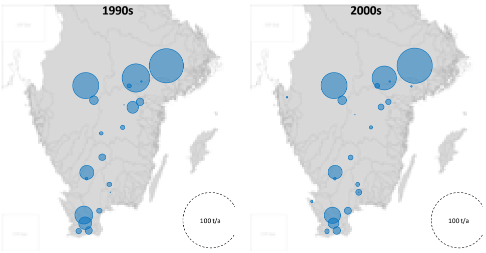

Figure 36 Spatial distribution of the reported landings from fisheries, in the 1990s and 2000s. For earlier decades, insufficient information is available.

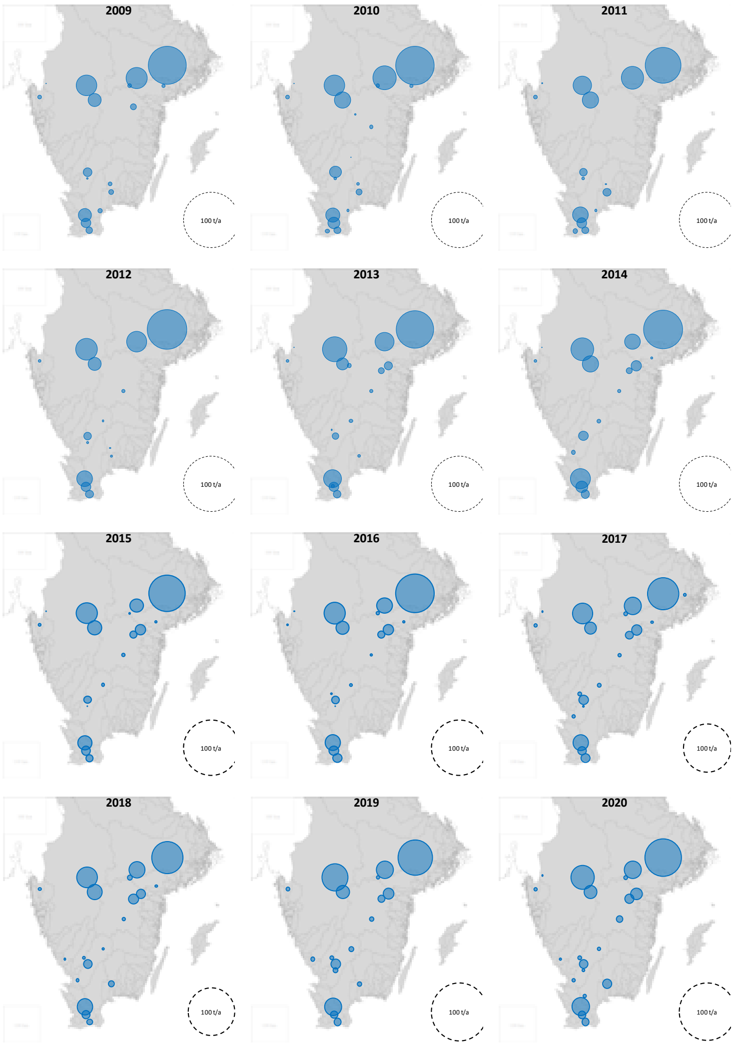

Figure 37 Spatial distribution of the reported landings from the fisheries, for the years since 2009.

For the years 1986 to 1995, the available data relate to the total landings for all smaller lakes combined, and to 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; Figure 37), 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 requires disproportional much effort. It is therefore recommended to include all of the catches in the regular fisheries statistics, and to keep special registration for the releases only.

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

Location of hydropower stations

A database of hydropower generation plants was made available by Kuhlin (2021), documenting location and year of construction (Figure 38). 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 1454 hydropower stations listed by Kuhlin (2021), 503 stations are relevant for the current reconstruction (eel occurring upstream).

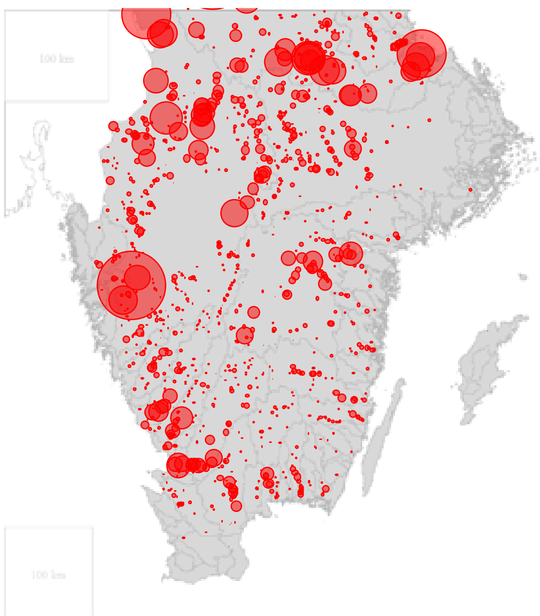

Figure 38 Spatial distribution of the 519 hydropower generation plants having an eel stock upstream. The size of the symbols in this figure is proportional to the capacity of each station.

Mortality per hydropower station

The mortality of eel passing a hydropower station in Sweden is not well known. Calles and Christianson (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 39, our plotting of their data). While the simulation almost 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); and 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.

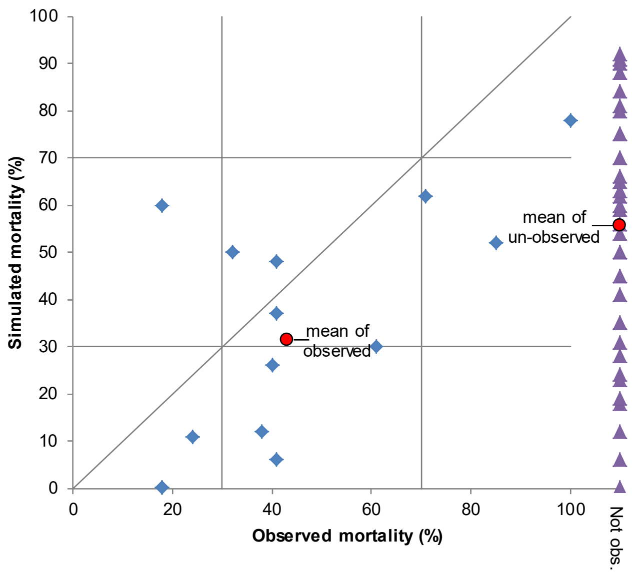

Figure 39 Relation between the observed (horizontal) and simulated (vertical) mortality, for eel passing a hydropower turbine. Data from Calles and Christianson (2012), applying the simulation model of Leonardsson (2012); original plot of data tabulated by the source.

Mortality on the route towards the sea

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 - 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 2017, a number of 2 850 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. 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 4.2 cm/year (the mean of all observations) for all years and sites.

Individual weights were calculated as

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

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

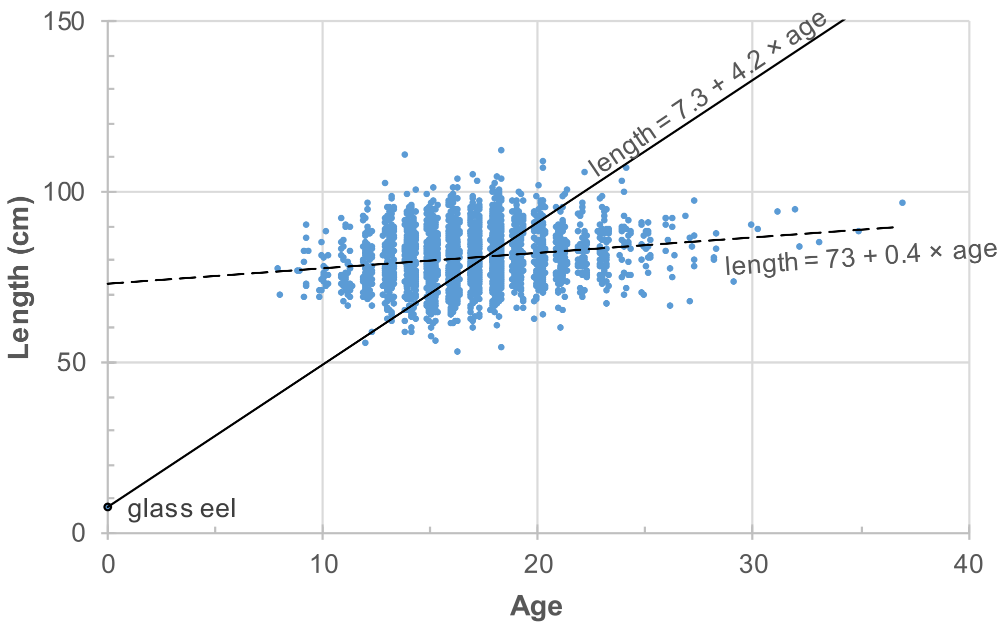

Figure 40 Length and age for 2 850 silver eels, sampled between 2010 and 2017 in 6 lakes. To show so many data points, a small jitter has been added to all data points in horizontal direction. Two regression lines are given: a growth-line (drawn) forced through the length/age of glass eel (7.3 cm cm at age=0), and an unforced silver-eel-size-line (dashed). Note that the intercepts and slopes of the two regression lines appear to differ by a factor of exactly ten, but that is not exactly so – it is a coincidence.

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 size, even where sampling had actually taken place.

At each sampling site, the age of the individual eels ranges from almost ten years below, to fifteen years above the mean age. 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 0.4 cm/year; Figure 40, dotted line) is much less than the mean growth rate during the yellow eel stage of 4.2 cm/year (Figure 40, solid 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.

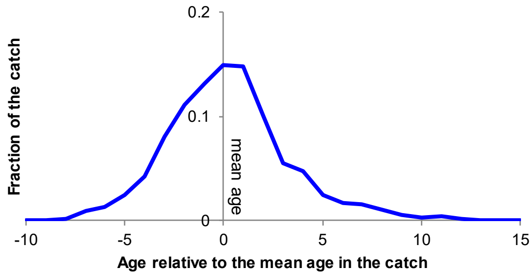

Figure 41 Relative age composition of the catches in inland waters, where age is expressed relative to the observed mean age.

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 per annum at the glass eel stage, decreasing to 0.015 per annum at the silver eel size, with a lifetime average of about 0.2 per annum. 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. 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 30 on restocking; Figure 34 on fishing yield from the larger lakes) that the more eel has been restocked, the higher the production has been. Therefore, it is very unlikely that density dependent growth and/or mortality have been limiting the production to any degree. 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– even, individual batches released at any place can remain separate in the assessment.

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 match spatially, resulting in local over- or underestimates.

Summing results by river drainage area, however, is smoothing out any spurious spatial patterns.

At the bottom line, this reconstruction yields an estimate of the quantity of silver eel starting 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 the spawning places. 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).

The calculation is additive in character (additive sources of youngsters, additive contributions from different rivers/lakes, additive contributions from various age-classes, and so forth; except for the hydropower impacts), but the natural recruitment is estimated by a multiplicative model (i.e. by a linear model of log-transformed data). In cases where the multiplicative statistical model yields an overestimate or an upward extrapolation is made above the normal range of observations, the mix of additive and multiplicative components leads to unrealistically high estimates. For that reason, extrapolations were avoided as much as possible. In particular, the assessment area was restricted to inland waters above the first migration barrier, and four smaller rivers near the Norwegian border (beyond the most north-western observation) were excluded.

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 (Annex 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 2030 to show the fate of the most recent recruits (natural or restocked).

C.2 Results

C.2.1 Silver eel production

This section presents results for the assumption on natural mortality that M=0.10 – other options for M will be discussed in section C.2.3 below.

From 1960 until 2020, natural recruitment – including the amount assisted in their migration upstream - is estimated at a total number of 77 million glass eel equivalents, with a minimum of 0.2 million eels in 2009 and a maximum of 3.6 million in 1951. The corresponding silver eel production is estimated at 11 619 t, minimum 25 t/a, maximum 375 t/a. In 2012, 0.26 million glass eel equivalents were natural recruits. Total silver eel production from natural recruits (assisted or not) in 2020 is estimated at 25 t.

From 1960 until 2020, a total of 32 million eels have been caught for assisted migration upstream, with a minimum of 0.037 million of year class 1995 and a maximum of 2.6 million of year class 1977. The corresponding silver eel production is estimated at 8 888 t, minimum 20 t/a in 2017, maximum 2958 t/a in 1969. In 2020, 0.2 million glass eel equivalents were assisted upstream. Total silver eel production from assisted migration in 2020 is estimated at nearly 21 t.

From 1960 until 2020, a total number of 72 million glass eel equivalents has been restocked, with a minimum of 0.08 million glass eel equivalents for year class 1961 and a maximum of 2.6 million for year class 1996. The corresponding silver eel production is estimated at 10 173 t, minimum 15 t/a in 1960, maximum 404 t/a in 2012. Of year class 2019, 2.1 million glass eel equivalents have been restocked (mean since 2010: 1.6 million).

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[4] For natural recruitment, the very last observation year 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 2012 (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. This is further discussed in Section B.3 on page 83.

Total silver eel production (before fishery and hydropower impacts) in 2020 is estimated at approximately 198 t.

Overall silver eel production declined from 500-650 t in the 1960s and 1970s, to less than 500 t/a since 2010, and an estimated 243 t/a in 2020. Natural recruits, freely immigrating or assisted upstream, have been gradually replaced by (imported) restocking and the natural recruits now make up only 10 % of the total production in inland waters. Peak restocking in the 1990s brought recent total production to a temporary maximum of 475 t/a in 2010; lower restocking in the early 2000s reduced production to 243 t/a in 2020, but increased restocking thereafter will return production to about 340 t/a.

From 2010 until 2017, a total number of 0.1 million silver eels have been trapped and transported downstream, with a minimum of 0.005 million (5 t) in 2010 and a maximum of 0.02 million (22 t) in 2014.

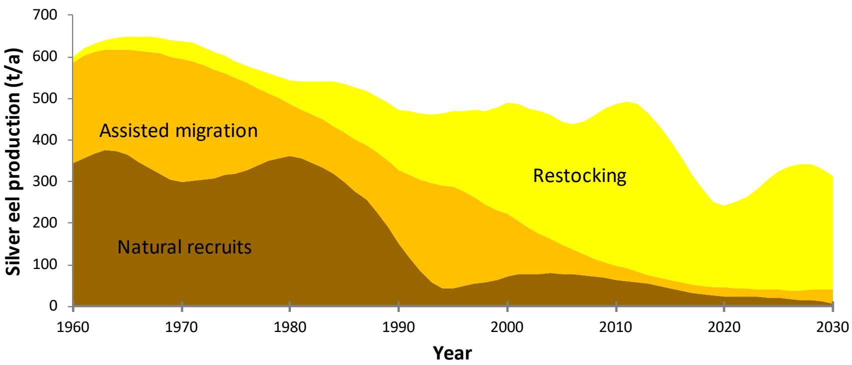

Figure 42 Production of silver eel by year and by origin of the eel, that is: the estimated total production before the impact of fishery and hydropower. For these results, a natural mortality rate of M=0.10 was assumed.

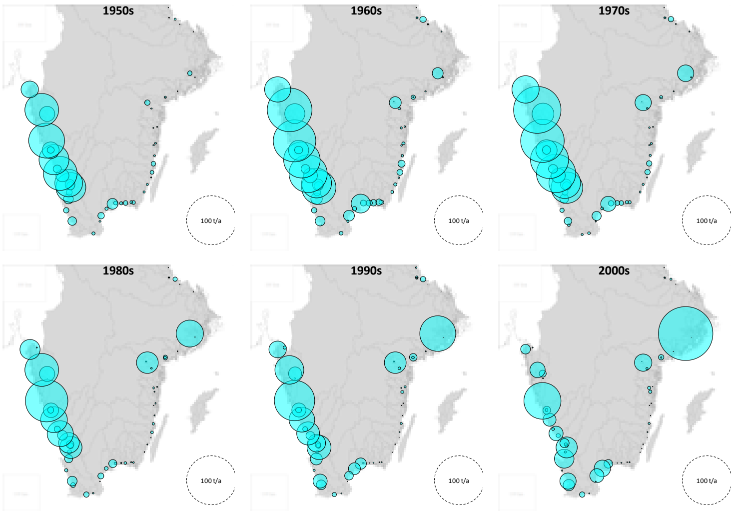

Figure 43 Spatial distribution of the predicted production of silver eel (before fishery and hydropower impacts), per decade and per river drainage system. The production for each river drainage area is plotted at the place of the river mouth, while in reality, the production will have taken place all over the drainage area.

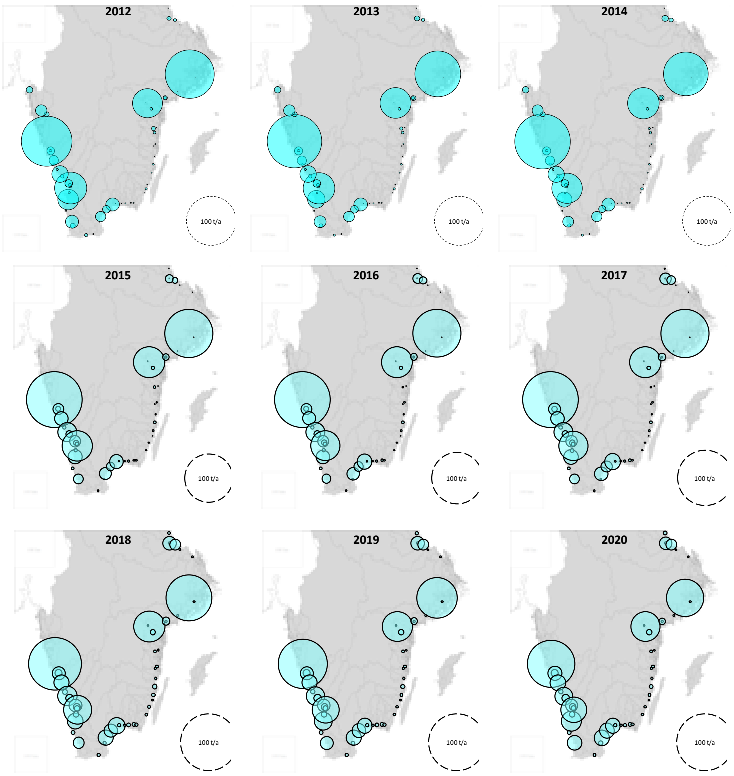

Figure 44 Spatial distribution of the estimated production of silver eel (before fishery and hydropower impacts), per year since 2012 and per river drainage system. The whole production estimated for each river drainage area is plotted at the place of the river mouth.

C.2.2 Silver eel destination

Figure 45 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, below.

Fishing data being incomplete up to 1986, results are only available for the period after. The total biomass of silver eel in Figure 45 matches the predicted total production, presented in Figure 42.

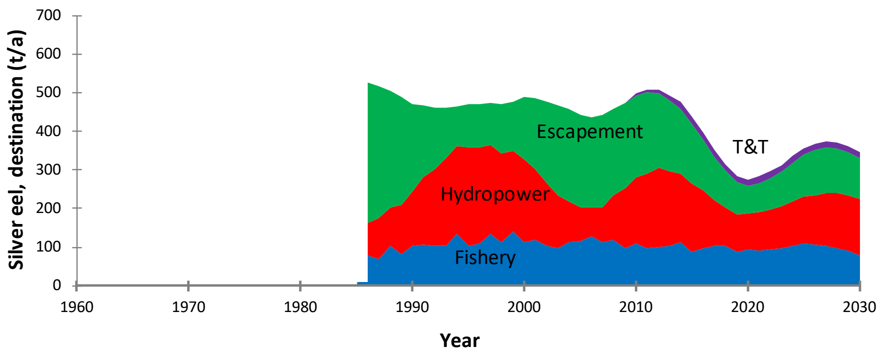

Figure 45 Time trends in the destination of the silver eel produced in inland waters. Data before 1986 are incomplete.

For the <u>fishery</u>, the landings have varied between 89 t (in 2015) and 134 t (in 1997). This is on average 25 % of the production, with rather little variation over the years (Figure 48). The catch in 2020 was 94 t.

For the <u>hydropower</u>, the estimated impact varied between 74 t (in 2006) and 255 t (in 1995), that is approximately 45 % of the total production (range 20 % - 50 %). The estimated impact in 2020 was 93 t, 56%. 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 <u>escapement</u> of silver eel ranged from 72 t (in 2020) to 364 t (in 1986), on average 40 % of the total production (range 22 % - 66 %). The 2020 escapement is the lowest estimate on record. The increase in restocking since 2010 is expected to contribute to a net rise in escapement from 2021 onwards, to a peak of 120 t in 2027. *Without* the contribution from restocking, estimated escapement ranged from 278 t (in 1986), to only 13 t (in 2020). The recent rise in natural recruitment (since year class 2011) will contribute to the escapement only after they have grown to silver eel size, at the very end of the prediction interval shown.

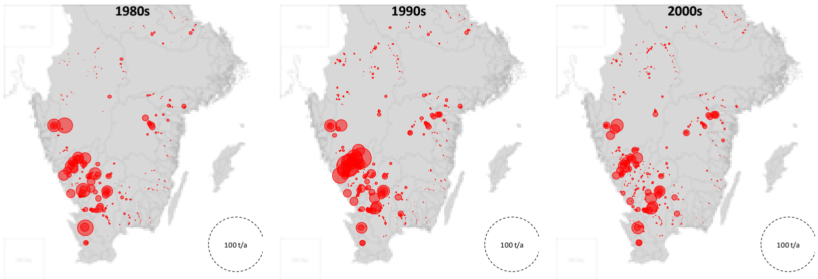

Figure 46 Spatial distribution of the estimated impact of hydropower, per hydropower station per decade. For the 1980s, estimates are based on the years from 1986 onwards; for the earlier years, no estimates could be derived because of the absence of information on the landings from fisheries.

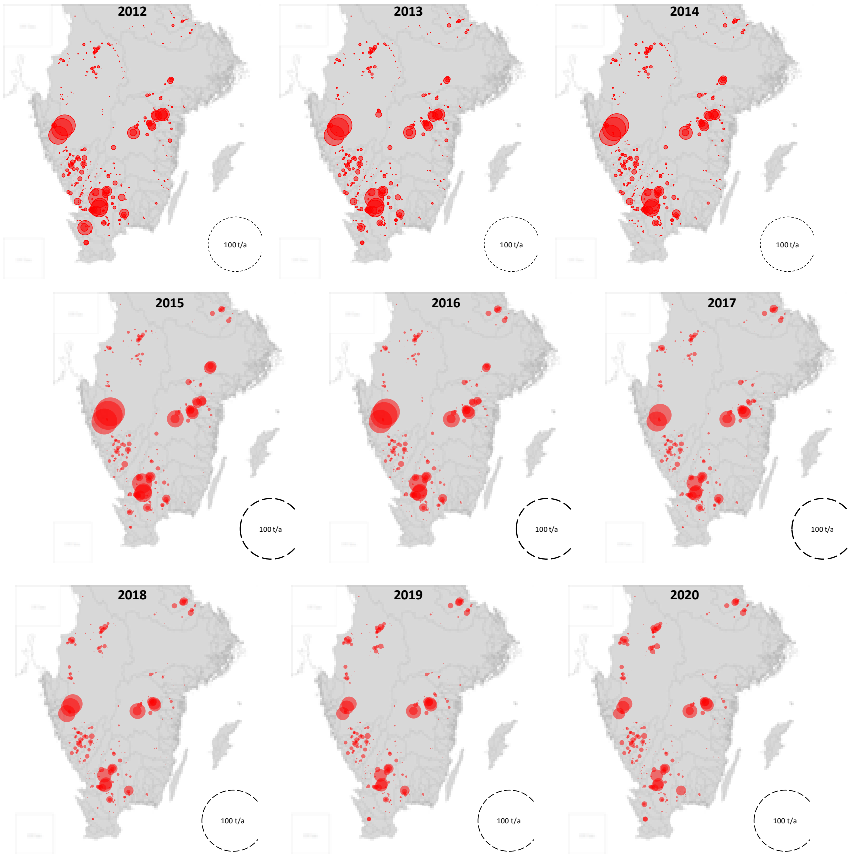

Figure 47 Spatial distribution of the estimated impact of hydropower, per hydropower station per year, since 2012.

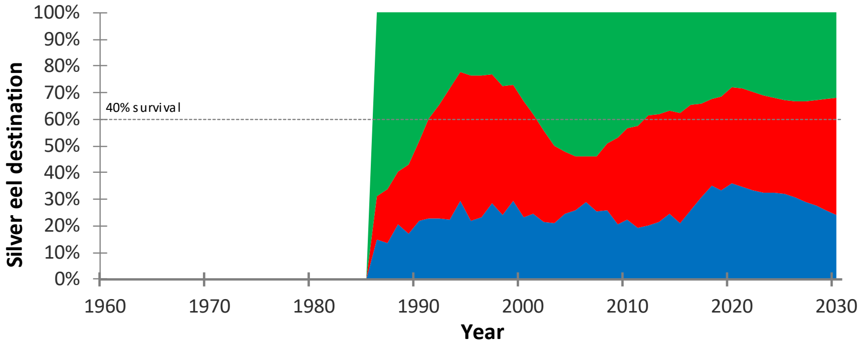

Figure 48 Time trend in the estimated anthropogenic mortality (and escapement), expressed in percentage impacts on the silver eel production.
The reference line “40 % survival” represents the ultimate limit mortality, for a healthy stock ($B_{current}$ > 40 % * B_0).

Expressing anthropogenic impacts in terms of mortality rates (Figure 49), one can either consider the mortality on the available stock (whatever their origin, natural or restocked), or one can consider restocking as a compensatory action (see the discussion in section 3.3 above). The presentation in Figure 49 allows for both interpretations. Including the effect of restocking (yellow), the sum of fishing mortality, hydropower related mortality, restocking and T&T is represented by a drawn line (F+H+R+T); *without* restocking, the sum ΣA of fishing mortality and hydropower related mortality represents the actual 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 +1.04 (in 1994) to -0.92 (in 2015); the 2020 value is estimated at -0.470. 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. As a consequence, the ratio of the restocking to the natural recruits is increasing.

Considering the anthropogenic mortality *without* restocking, total anthropogenic mortality has ranged from 0.37 (in 1986) to 1.51 (in 1994); the 2020 mortality is estimated at 1.28. These estimates express the mortality exerted on the natural recruits, as well as on the restocked eels.

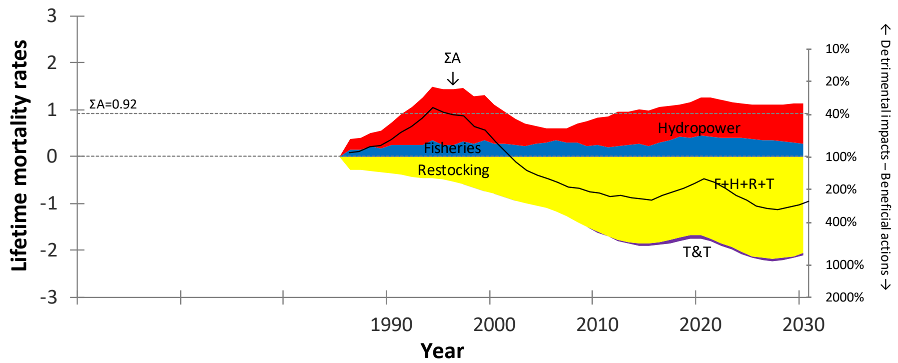

Figure 49 Time trend in the estimated anthropogenic mortalities: fisheries, hydropower, restocking and Trap & Transport (T&T). The mortality exerted by Restocking and Trap & Transport are negative; that is: these actions increase the amount of silver eel escaping. The line marked "F+H+R+T" represents the sum of all anthropogenic actions, including Restocking and Trap & Transport; ΣA represents the mortality exerted on the stock, whether natural or restocked. The reference line ΣA=0.92 represents the ultimate limit mortality, for a healthy stock (Bcurrent > 40 % * B0). A mortality below that level is expected to allow recovery.

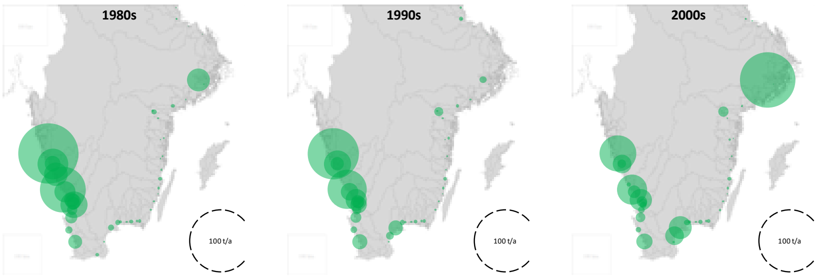

Figure 50 Spatial distribution of the estimated escapement of silver eel per decade. For the 1980s, estimates are based on the years from 1986 onwards; for the earlier decades, no estimates could be derived because of the absence of information on the landings from fisheries.

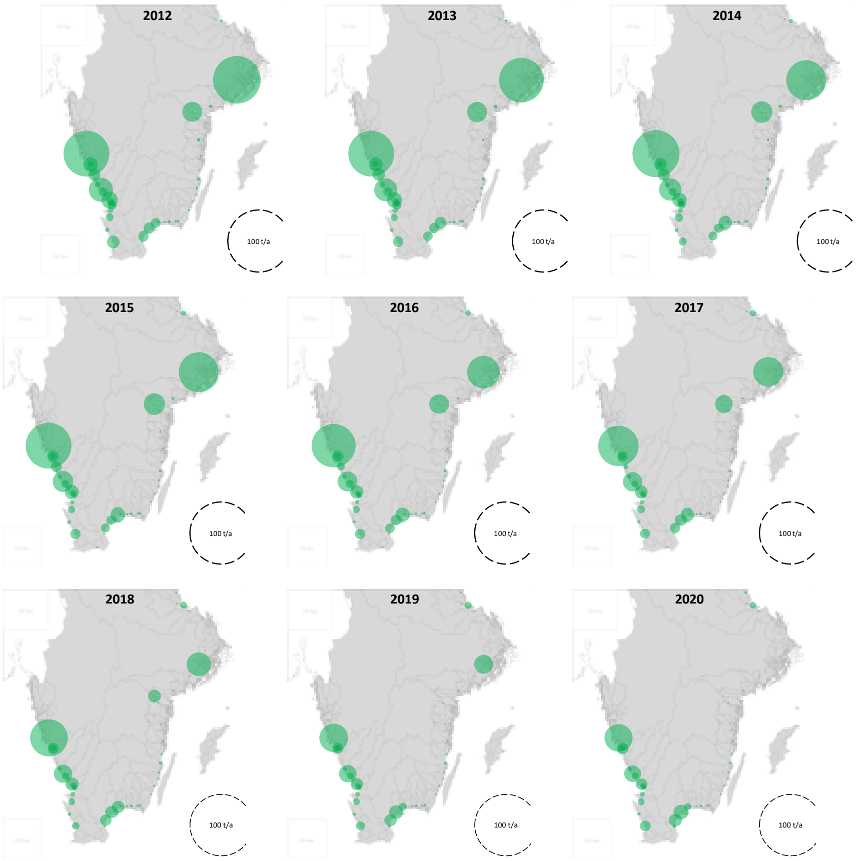

Figure 51 Spatial distribution of the estimated escapement of silver eel per year, since 2012.

Considering individual rivers/lakes (*Figure 51*), 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 completely after 2009 (except for a single restocking in 2018), 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.

C.2.3 Natural mortality M

Parameter value

The results presented in this Annex so far are based on an assumption on the level of natural mortality, M=0.10. 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 52 compares results, for two selected years: 1995 and 2020, 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 52.a&b) range from just over 240 t/a to around 960 t/a. 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 52.c&d), results are quite different. For M=0.05, production is estimated at c. 960 t; for M=0.15 at slightly more than 200 t. The fishery taking just over 100 t – irrespective of the assumption on M - the estimates of the silver eel run migrating downstream ranges from 860 t (for M=0.05) to not more than 140 t (for M=0.15). For M=0.10, the estimated production for a few lakes and years ends up below the recorded catch, resulting in a negative estimate for the silver eel run, the hydropower mortality and the escapement to the sea. For M=0.15, negative estimates occur in many cases (including Mälaren and Vänern, for many years).

For the estimates of anthropogenic mortality (Figure 52.e&f), the assumption on M has a large effect on the estimate of fishing mortality F (variation by a factor of 5 or more), little effect on the estimate of hydropower mortality H (a factor up to 1.1), 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 (1.28 – 1.82 in 1995; 0.98 – 2.28 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 our inland waters appears to be extremely good. An alternative explanation could be that natural recruitment is much higher than estimated in Annex B, but micro-chemical analysis of otoliths has corroborated that natural recruits (including assisted migration) constitute not more than 10 % of the catch (Clevestam and Wickström 2008).

In the absence of conclusive evidence on the true value of M, the main results in this Annex are based on the assumption M=0.10, 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) do not critically depend on this assumption.

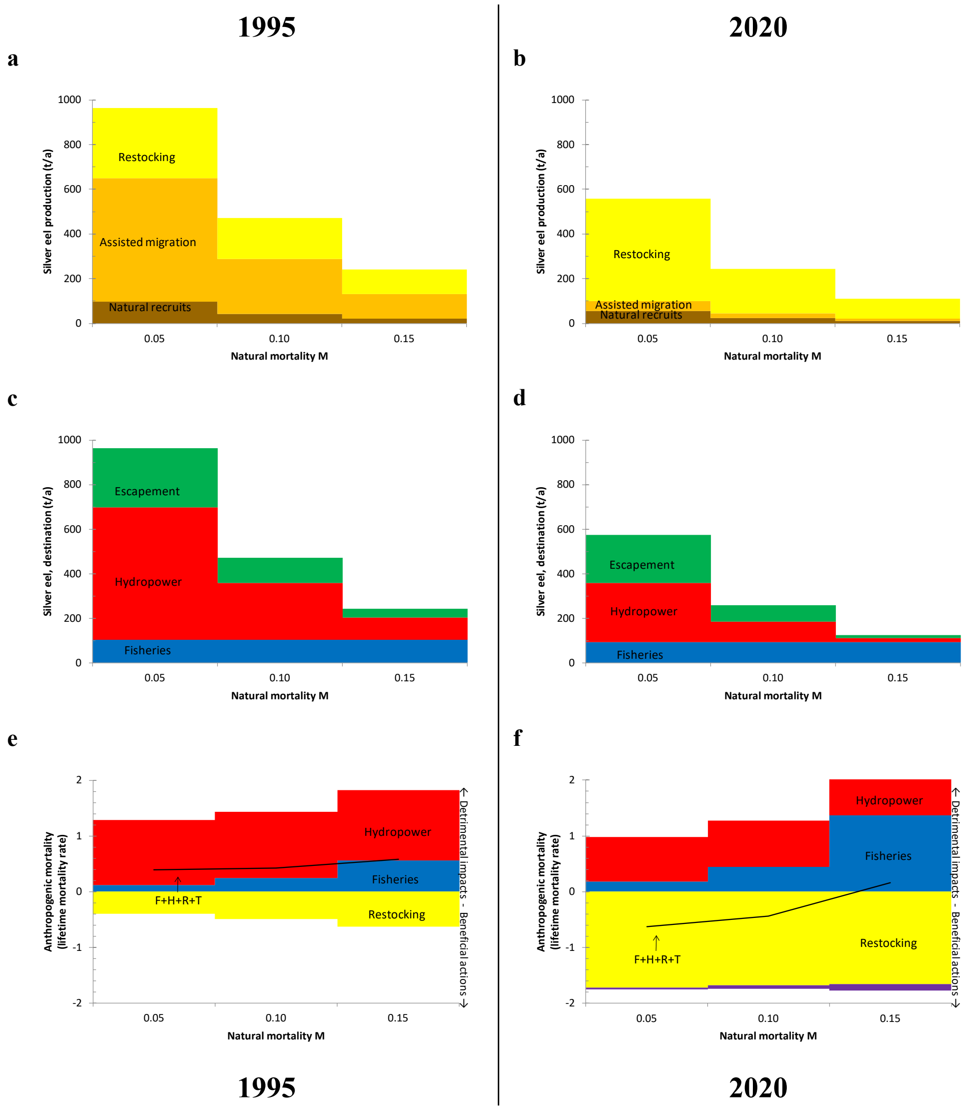

Figure 52 Comparison of results for three different values of natural mortality, showing results for 1995 (left) and 2020 (right). Within each sub-plot, the columns show results for the three options M=0.05, M=0.10 and M=0.15, respectively; comparisons are to be made within each subplot, between the columns.
Top row: predicted silver eel production (compare Figure 42);
Middle row: predicted silver eel destination (compare Figure 45);
Bottom row: anthropogenic mortality rates (compare Figure 49).

Cormorant predation

Over the years, the numbers of cormorants feeding in inland waters has risen considerably, 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 42 indicates that silver eel production has varied between 300 and almost 600 t/a; hence, it is estimated that 400 to 1000 t of yellow eel has died of natural causes.

According to Strömberg *et al*. (2012), the number of breeding cormorants is in the order of 40-45 thousand pairs, of which approximately 20 % 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 can be estimated at some 3000 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 and Öhman 2014), but of 293 tags in eels released in Lake Roxen, 7.5 % 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. 3000 t of fish biomass consumed.

The contrast between the estimate of the biomass consumed by cormorants (order of magnitude of a few percent of 3000 t/a) to the amount of eel considered to have died of natural causes in the current reconstruction (order of magnitude of 400-1000 t/a) indicates that the available information on cormorant predation does not contradict the current results.

The references for this Annex are included in the reference list of the main report, on page 46.

Annex 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 (Dekker 2012) used those estimates, extrapolating the 2006-2008 results to 2011 on the assumption that landings and fishing mortality were proportional. The 2015 and 2018 assessments (Dekker 2015, Dekker *et al*. 2018) updated the analysis, adding the data from the re-continued tagging programme. This Annex now presents a third update, including data up to and including 2020. No major changes in the methodology of Dekker & Sjöberg (2013) have been made, except that non-Swedish captures of tagged eel have now been considered as uncaptured, for the sake of only assessing the impact of the Swedish component of the Baltic Sea eel fishery.

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 55). 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 56 gives a spatial overview of the eel tagging experiments that have been performed since the latest triannual eel assessment.

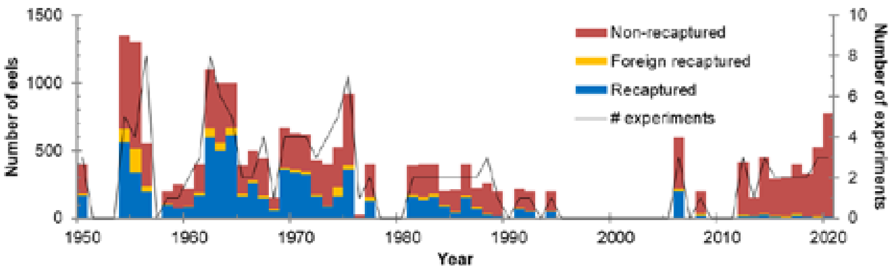

Figure 53: Number of eel tagging experiments performed over time (line) and their annual number of tagged eel (bars). The annual number of tagged eel is subdivided into eel recaptured by the Swedish eel fishery (blue), eel recaptured by a foreign fishery (orange), and non-recaptured eel (red).

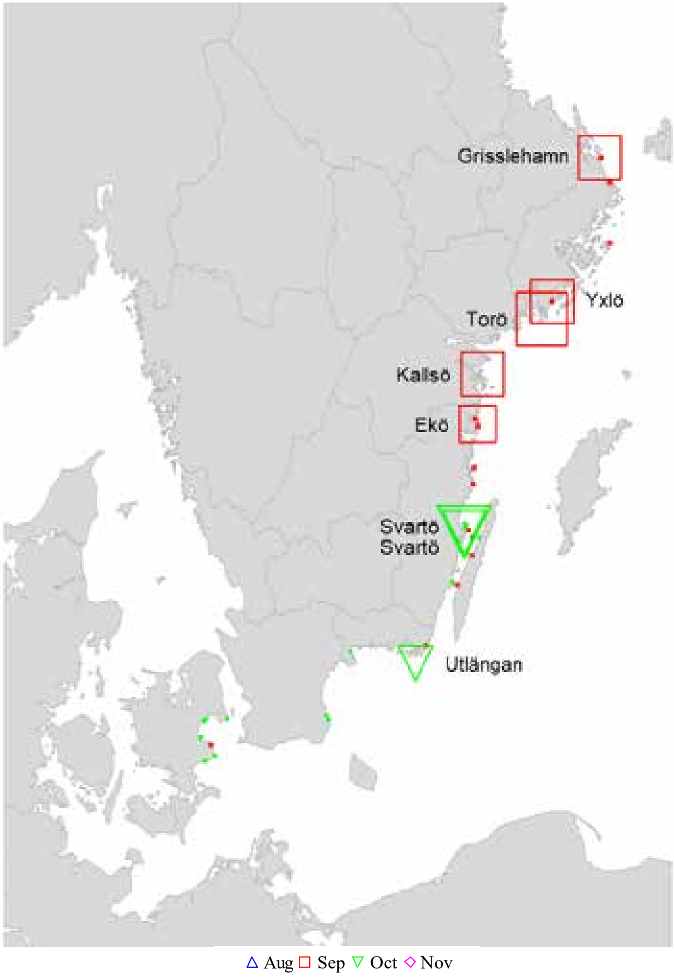

Figure 54: Location of eel tagging experiments in the years 2018-2020. The size of the larger symbols is proportional to the number of eels released. The small dots represent recaptures of single eels. Colour indicates month of release of the tagged eel.

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. Dekker & Sjöberg (2013) listed four different models of increasing complexity for estimating survival and hazard functions. 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.

D.2 Results

Figure 55 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 2010 decade, 18 tagging experiments were performed with an average recapture of 4.2% by the Swedish eel fishery, and an additional recapture of 2.0% by the Danish eel fishery. In comparison, in the 2000 decade 4 tagging experiments were performed with an average recapture of 27.4% 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 a slight increasing trend from the 2000s again (Figure 57). Mean number of days at large appears to roughly follow the same trend, except that no increasing trend after 2000 can be observed (Figure 57).

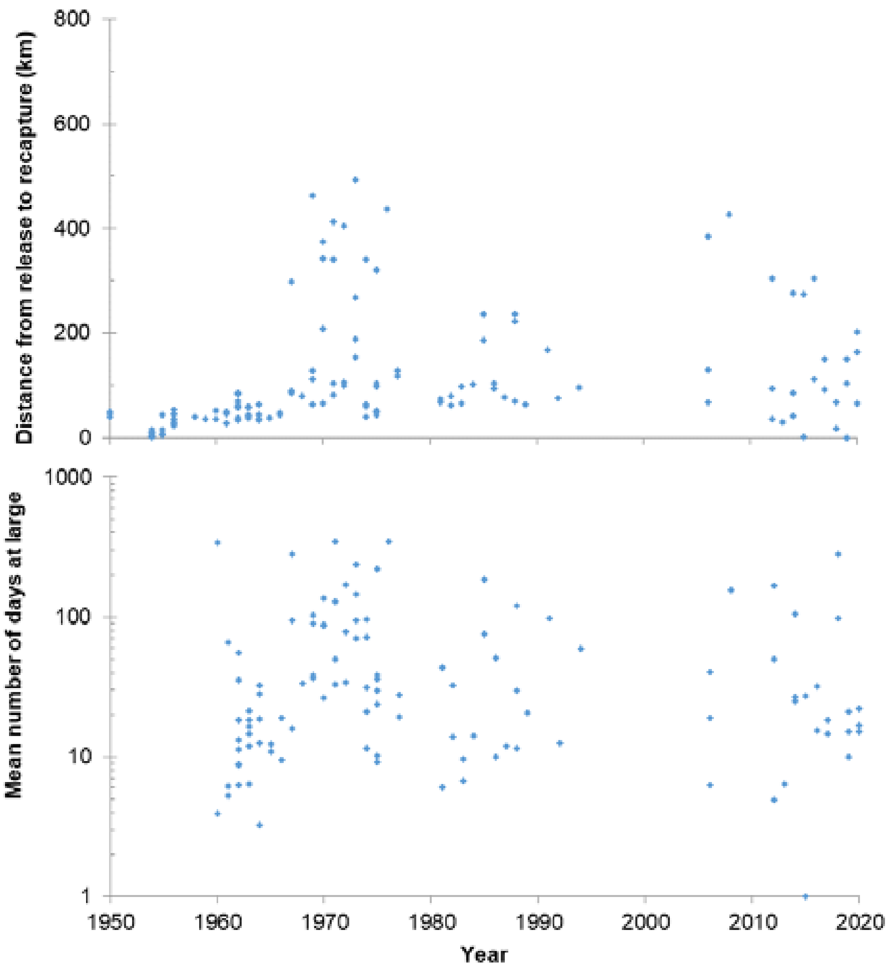

Figure 55: Mean distance covered (top) and mean number of days at large (bottom) between release and recapture, shown for each tagging experiment (year on the x-axis refers to year of experiment). Recaptures outside of Sweden's EEZ have been omitted. Note the logarithmic y-axis on the bottom graph.

Figures 58-61 show the results of the survival analysis. Each figure shows the results in two forms: firstly on a per-decade level, and secondly for the three most recent sets of three consecutive years. The decadal results show the long-term trend developments, whereas the triannual results show the recent short-term trend developments. Because the triannual results are based on less data than the decadal results, these come with a greater uncertainty than the decadal results.

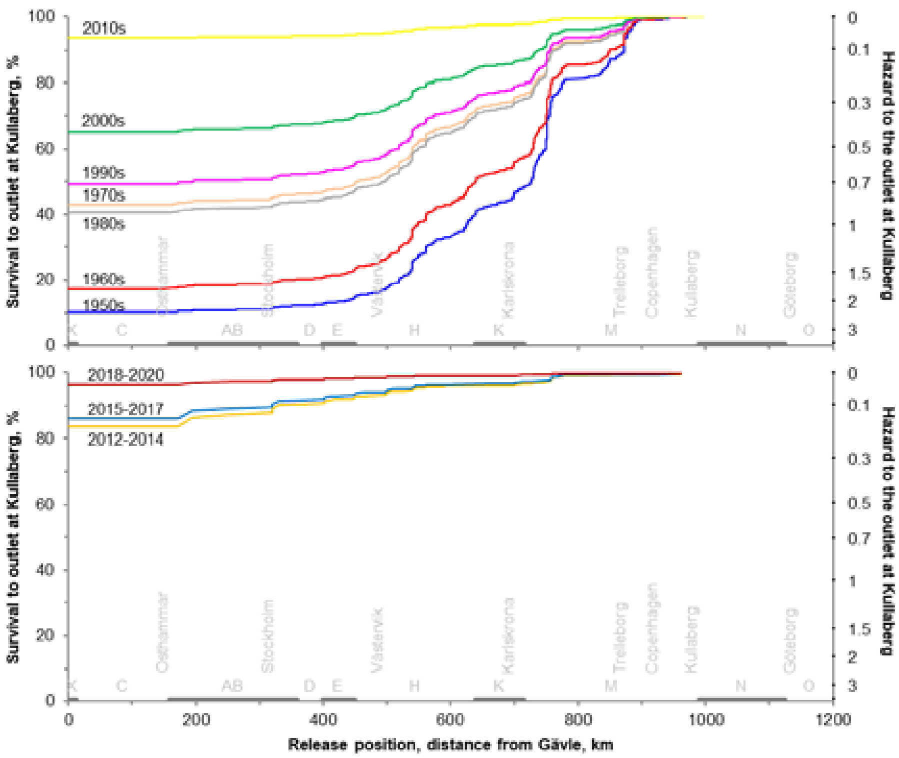

Figure 56: Survival and hazard over distance along the Swedish coast, as estimated by the Cox proportional hazards model. Top: survival and hazard estimated per decade. Bottom: survival and hazard estimated for the three most recent sets of three consecutive years. The left y-axis shows the estimated 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 58. 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 three-year periods between 2012 and 2020. Average hazard of capture (that is: fishing mortality) in the 2010 decade was estimated at only 0.009, and in the 2018-2020 period decreased even further to an estimated 0.003. To compare, in the 2000 decade average hazard of capture was still 0.05.

County or region (Swedish: län or region) specific estimates of capture hazard for tagged eel are given in Figure 59. These show a declining trend in hazard over time for every county, both on a decadal scale and on a triannual scale.

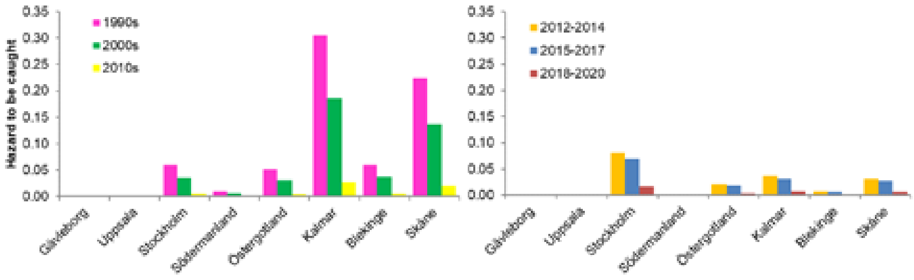

Figure 57: Hazard by county/region (län/region), shown for the three most recent decades (left), and the three most recent sets of three consecutive years (right).

Similarly, county-specific landings of eel also show a declining trend over time (Figure 60) on both a decadal and triannual scales, with Blekinge county as the only exception.

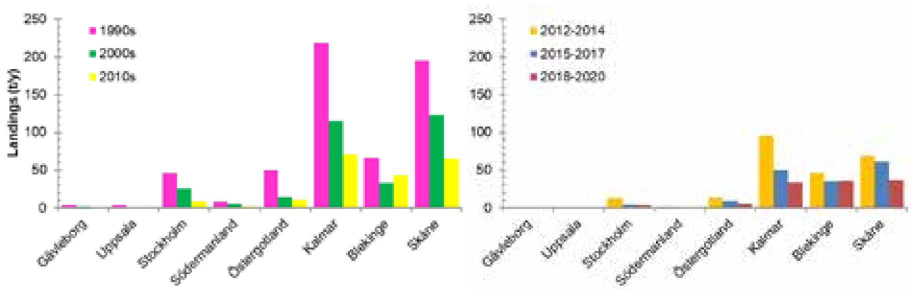

Figure 58: Landings of eel along the Baltic Sea coast by county (län), shown for the three most recent decades (left), and the three most recent sets of three consecutive years (right).

County-specific estimated stock biomass of silver eel, estimated by dividing county-specific landings (Figure 60) with county-specific hazard (Figure 59), shows a general increasing trend over time (Figure 61). Blekinge county shows an especially-large estimated silver eel biomass for the years 2018-2020, due to its very low estimate of hazard (Figure 59) during this time, which in turn is the result of the very low recapture rate in Blekinge in these years. 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.

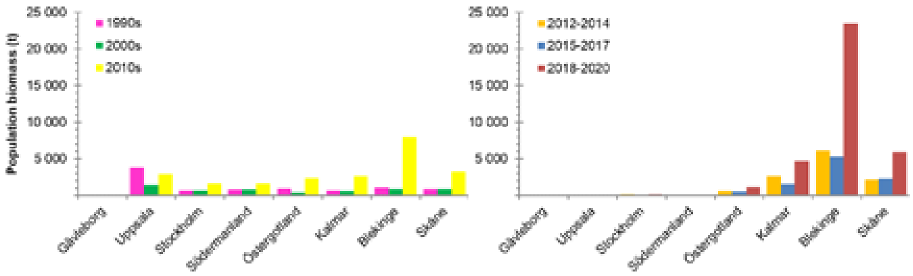

Figure 59: Estimated stock size by county (län), shown for the three most recent decades (left), and the three most recent sets of three consecutive years (right). Since catches and hazards in Gävleborg were effectively zero, no estimate is derived there.

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 58), and silver eel stock biomass shows an increasing trend over time. This positive trend is reflected on both the long term (decadal results) and the recent short term (triannual results).

The number of days at large for tagged eel has been similar over the decades (Figure 57). 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. 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 (4.2% in the 2010 decade) means 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 can for instance be observed by comparing the decadal estimates of stock biomass with the triannual estimates (Figure 61. For the 2010s, stock biomass estimates in the Uppsala, Stockholm, and Södermanland counties are much higher than their estimates in any of the three triannual periods. Here, the decadal estimates are based on a greater underlying amount of data (recaptures), and are thus more reliable. 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.

The restocking of eel on the Baltic coast has been described in section A.3, on p. 55, above.

The references for this Annex are included in the reference list of the main report, on page 46.