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## **Assessment of the eel stock in Sweden, spring 2018**

Third post-evaluation of  
the Swedish Eel Management Plan

Willem Dekker, Andreas Bryhn, Katarina Magnusson,  
Niklas Sjöberg, Håkan Wickström



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## Executive summary

The population of the European eel is 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 will report to the Commission every third year, 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 2018, intending to feed into the national reporting to the EU; this updates and extends the report by Dekker (2012, 2015).

In this report, the impacts on the stock are assessed - of fishing, restocking and of the mortality related to hydropower generation. Other anthropogenic impacts (climate change, pollution, increased impacts of predators, spread of parasites, disruption of migration by 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 these 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 Sea coasts).

In recent years, a break in the downward trend of the number of glass eel has been observed throughout Europe. 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 that recent trend and the overall status of the stock across Europe.

For the different assessment areas, results summarise as follows:

On the west coast, a fyke net fishery on yellow eel was exploiting the stock, until this fishery was completely closed in spring 2012. Though research surveys using fyke nets continued, insufficient information is currently available to assess the recovery of the stock in absolute terms. Obviously, current fishing mortality is zero (disregarding the currently unquantifiable effect of illegal fishing), but none of the other requested stock indicators (current, current potential and pristine biomass) can be presented. The research surveys, however, indicate that the formerly exploited size-classes of the stock do recover indeed, but overall, the decline of the stock has continued – in line with the general trend of the stock across its distribution area.

In order to support the recovery of the stock, or to compensate for anthropogenic mortality elsewhere, young eel has been restocked on the west coast. 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).

For inland waters, this report updates the 2015 assessment, not making substantial changes in methodology, but improvements in some of the model parameters (notably: improved recruitment estimates and length-weight-relation) have affected all results. Though the current results thereby deviate from the 2015' results, the trends and the evaluation of the status of the stock remain the same.

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 relating 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). Further into the Baltic, recruits are larger (exception: the 100 gr recruits in Mörrumsån, 56.4°N, where only 30 gr would be expected) and less numerous; distance upstream comes with less numerous recruits, but size is not related. Remarkably, the time trend differs for the various ages/sizes. Oldest recruits (age up to 7) declined already in the 1950s and 1960s, but remained stable since; youngest recruits (age 0) showed a steep decline in the 1980s and a 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, across barriers) and restocking (imported young eels), 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. 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), and is still falling. Natural recruitment (assisted and fully natural) has gradually been replaced by restocking for 90 %. Fisheries have taken 20-30 % of the silver eel, while the impact of hydropower has ranged from 20 % to 60 %. Escapement is estimated to have varied from 25 % (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 approx. 20 % of the pristine level (incl. restocked), or almost 40 % of the current potential (incl.

restocked). This is below the 40 % limit of the Eel Regulation, and anthropogenic mortality (just over 60 %) exceeds both the short-term limit needed to establish recovery (38 %) and the ultimate limit (60 % mortality, the complement of 40 % survival). The temporal variation (in production, impacts and escapement) is largely the consequence of a differential spatial distribution of the restocked 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 unobstructed lakes (primarily Lake Mälaren, 1990s), but is since 2010 concentrated to drainage areas falling to the Kattegat-Skagerrak, thus including also obstructed lakes (primarily Lake Vänern, to a lesser extent Lake Ringsjön, and many smaller ones). Since 2010 eels are also stocked directly into the sea along the west coast. Trap & Transport of silver eel - from above barriers towards the sea - has added 1-5 % of silver eel to the escapement. 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 only 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, those impacts are currently increasing, and without further protective actions, will increase even further. It is therefore recommended to reconsider the current action plans on inland waters, and to take into account the results of the current, more comprehensive assessment.

For the Baltic coast, the 2015 assessment has been updated without changes in methodology. Results indicate that the impact of the fishery is rapidly declining over the decades – even declining more rapidly towards the 2010s than before. The current impact of the Swedish silver eel fishery on the Baltic Sea coast is estimated at 2 %. However, this fishery is just one of the anthropogenic impacts (in other areas/countries) affecting the eel stock in the Baltic. Integration with the assessments in other countries has not been achieved. Current estimates of the abundance of silver eel (biomass) are in the order of a few thousand tonnes, but those estimates 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.

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

Considering the international context, the stock indicators – in as far as they could be assessed – fit the international assessment framework, but inconsistencies and interpretation differences at the international level complicate their usage. International coordination and standardisation of the tri-annual reporting is therefore recommended. Additionally, it is recommended to initiate international

standardisation/inter-calibration of monitoring and assessment methodologies among countries, achieving a consistent and more cost-effective assessment across Europe.

## Sammanfattning

Den europeiska ålen är stadd i stark minskning. 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 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 2018, detta med syfte att tjäna som underlag till den svenska uppföljningsrapporten till EU. Rapporten uppdaterar och utvidgar därmed tidigare års utvärdering (Dekker 2012, 2015).

Rapporten utvärderar påverkan från fiske, utsättning och kraftverksrelaterad dödlighet på ålbeståndet. Annan antropogen påverkan som klimatförändring, förorening, ökad påverkan från predatorer, parasitspridning och en eventuell störd vandring hos omflyttade ålar osv., har sannolikt också en effekt på beståndet. Sådana faktorer kan knappast kvantifieras och det finns inte heller några relaterade förvaltningsmål uppsatta. Av de orsakerna 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 så har den sedan länge nedåtgående trenden i antalet rekryterande glasålar brutits och det över hela Europa. Om det är en effekt av de åtgärder som gjorts, eller om det finns andra bakomliggande orsaker, är fortfarande oklart. Denna rapport bidrar till den internationella bedömning som krävs, men den diskuterar inte den senaste rekryteringstrenden och ålbeståndets allmänna tillstånd i Europa.

Resultaten för de olika områdena summeras enligt följande:

Gulålen på västkusten exploaterades tidigare genom ett intensivt ryssjefiske. Det fisket är sedan våren 2012 helt stängt. Även om en viss uppföljning fortsätter genom ryssjefiske, så är den tillgängliga informationen inte tillräcklig för en beståndsuppskattning. Uppenbart 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. 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).

För inlandsvattnen så redovisar rapporten en uppdatering av 2015 års beståndsuppskattning, utan större förändringar i metodiken, men förbättringar av vissa modellparametrar (särskilt: förbättrade rekryteringsuppskattningar och längdvikt förhållanden) har påverkat alla resultat. Trots att nuvarande resultat avviker från resultaten från 2015, är trenderna och utvärderingen av ålbeståndets status det samma.

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 ålarna 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 märkligt åldersrelaterat mönster har observerat 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 eller standarddödlighet), har mängden överlevande lekvandrare (lekflykt) uppskattats. Resultaten visar att sedan 1960, så har produktionen av blankål minskat från mer än 500 till mindre än 300 ton per år, och produktionen minskar fortfarande. Den naturliga rekryteringen av ål, uppflyttad eller fullt naturlig, har

gradvis ersatts till 90 % genom utsättning av importerade ålyngel. Fisket har tagit 20-30 % av blankålen, 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 20 % av den jungfruliga mängden (inkl. utsatt), eller nästan 40 % av dagens potential (inkl. utsatt). Biomassan ligger därmed under den 40 %-gräns som Ålförordningen föreskriver, och den mänskligt introducerade dödligheten (drygt 60 %) överskrider såväl den kortsiktiga gränsen som krävs för beståndets återhämtning (38 %) och den avgörande slutgiltiga gränsen (60 % dödlighet, motsvarande 40 % överlevnad). 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). Numera sätts ålyngel också ut direkt i havet på västkusten. Trap & Transport av blankål, från uppströms liggande vattenkraftverk ner till respektive mynningsområde, har tillfört 1-5 % till lekvandringen. Utan ålutsättning, skulle biomassan av ål påverkad av fiske och vattenkraft bara vara 10-15 % av vad som faktiskt påverkas idag. Samtidigt skulle ålbeståndet vara bara 5 % av den ursprungliga 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 att fortsatt 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, så 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 2 %. Fisket är emellertid bara en av de mänskliga faktorer (i andra delar 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 dessa skattningen tar inte hänsyn till

ursprunget av blankålar från andra länder. En integrerad, enhetlig beståndsuppskattning för hela Östersjön behövs för att verifiera denna skattning.

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

Från ett internationellt perspektiv passar beståndsindikatorerna, så långt de nu kan uppskattas, väl in i ramen för arbetet med den internationella beståndsuppskattningen. Skillnader i tolkning och bristande överensstämmelse mellan länder komplicerar dock användningen av indikatorerna. Vi rekommenderar därför en internationell koordinering och standardisering av den rapportering till EU som återkommer vart tredje år. Dessutom rekommenderas att en internationell standardisering och interkalibrering av övervaknings- och beståndsuppskattningsmetoder mellan länder initieras. På så sätt kan en konsekvent och mer kostnadseffektiv beståndsuppskattning komma till stånd i hela Europa.

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# 1 Introduction

## 1.1 Context

The population<sup>1</sup> 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 2004a, 2016; ICES 2017a). 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 by 2009. In December 2008, Sweden submitted its Eel Management Plan (Anonymous 2008). Subsequently, protective actions have been implemented (in Sweden and all other EU countries), and progress has been reported in 2012 (Anonymous 2012; Anonymous 2014). 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). This report 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 parasites, impact of predators (possibly anthropogenically-enhanced) and the potential disruption of migratory behaviour by transport of eels (for restocking, or by Trap & Transport). For these factors, European policies that pre-date the Eel Regulation are in place, such as the Fauna and Flora 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;

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<sup>1</sup> 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.

the Eel Regulation has no additional measures. Since this report is focused on an assessment of the eel stock in relation to the implementation of the Eel Regulation, these other factors 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 impact of most of these other factors on the eel stock has 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 (fisheries and hydropower mortality).

According to the EU Regulation, Member States shall report to the Commission no later than the 30 of June 2018 on the implementation of their Eel Management Plans and the effect it has had on stock and fisheries. This report analyses the status of the 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 facilitate 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 ( $B_{\text{current}}$ ,  $B_{\text{best}}$  and  $B_0$ ) and the mortality they endured over their lifetime ( $\Sigma A$ ); 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.

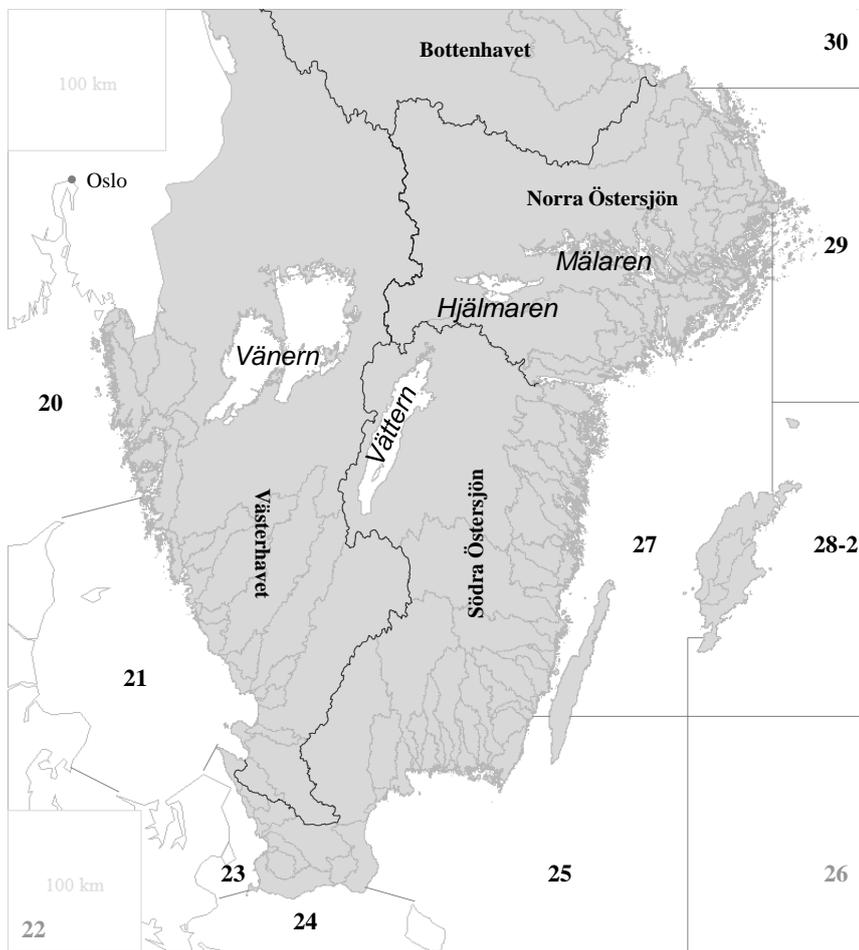
Annex B presents the riverine recruitment time series and analysis spatial and temporal trends.

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 2012-2017.

#### 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 coast, 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 substantial 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.

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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).



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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).



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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).



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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).



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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) have been operated at many places, and some are still being used.

Hydropower generation impacts the emigrating silver eel.



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## 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 west coast versus 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).
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 indicators:

$B_{\text{current}}$	The biomass of silver eel escaping to the ocean to spawn, under the current anthropogenic impacts and current low recruitment.
$B_{\text{best}}$	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. This includes fishing mortality $F$ , and hydropower mortality $H$ ; $A=F+H$ .
$\Sigma A$	Total anthropogenic mortality rate, summed over the whole life span.
%SPR	Percent spawner per recruit, that is: current silver eel escapement $B_{\text{current}}$ as a percentage of current potential escapement $B_{\text{best}}$ . %SPR can be derived either from $B_{\text{current}}$ and $B_{\text{best}}$ , or preferably from $\Sigma A$ (%SPR = $100 \cdot \exp(-\Sigma A)$ ).
%SSB	Current silver eel escapement $B_{\text{current}}$ 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 ( $B_{\text{current}}^+$ ,  $B_{\text{best}}^+$ ,  $B_0^+$ ,  $\Sigma A^+$ , etc.); if it is explicitly excluded, the symbol will be expanded by a – sign ( $B_{\text{current}}^-$ ,  $B_{\text{best}}^-$ ,  $B_0^-$ ,  $\Sigma A^-$ , etc.); when the difference between natural and restocked immigrants is not relevant, the addition may be omitted.

## 1.6 Management targets

The EU Eel Regulation sets a long-term general objective (“the protection and sustainable use of the stock of European eel”), delegating local management, the implementation of protective measures, monitoring, and local post evaluation to its Member States (Anonymous 2007; Dekker, 2009, 2016). A limit is set for the biomass of silver eel escaping from each management area: at least 40 % of the silver eel biomass relative to the escapement if 1. no anthropogenic influences would have impacted the stock and 2. recruitment would not have declined. Since current recruitment is far below pre-1980 levels and is assumed to be so due to anthropogenic impacts, return to this level is not expected before decades or centuries, even if all anthropogenic impacts are removed (Åström & Dekker 2007). In the current situation of low stock abundance and declining recruitment, the stock is below the biomass level aimed for, and – despite management actions taken – may only just have started to recover. In this situation, biomass limits and biomass assessments are not informative (Dekker 2016). They only indicate that the stock is in bad condition, not whether protective actions can be expected to achieve recovery.

In addition to the biomass limits of the Eel Regulation, a parallel system focused on mortality limits has been developed (Dekker 2010, 2016; ICES 2010, 2014). The rationale for this parallel system is that protective actions primarily affect the stock through their effect on mortality rates, that biomass only increases as a consequence of reduced anthropogenic mortality, and above all: that mortality rates reflect the effect of protective actions immediately, while biomass levels in most cases will only increase gradually over a number of years (Dekker 2016). For every possible biomass limit, a corresponding long-term mortality limit can be derived. A lifetime anthropogenic mortality of  $\Sigma A=0.92$  corresponds to a lifetime survival from anthropogenic mortalities of 40 %, which will – if and when recruitment restores to historical values – result in a biomass of escaping silver eels of 40 % of the pristine level. The template for the 2018 post-evaluation supplied by the EU Commission includes a request to report on the quantities  $B_{\text{current}}$ ,  $B_{\text{best}}$ ,  $B_0$  and  $\Sigma A$  – enabling the application of this framework.

A lifetime mortality of  $\Sigma A=0.92$  can be shown to match the 40 % biomass limit in the long run. At very low biomass, however, ICES (2009) reduces the anthropogenic mortality advised, to reinforce the tendency for stocks to rebuild. In general, ICES applies a reduction in mortality reference values that is proportional to the biomass (i.e. a linear relation between the mortality rate advised and biomass). This results in a Precautionary Diagram, as modified by ICES (2012). This diagram is applied below (Figure 7); the linear relation is showing up as a curved line on the logarithmic scale used here).

Within ICES, there has been discussion whether this reference framework is applicable to eel, or a stricter protection must be advised (ICES 2013a, Technical

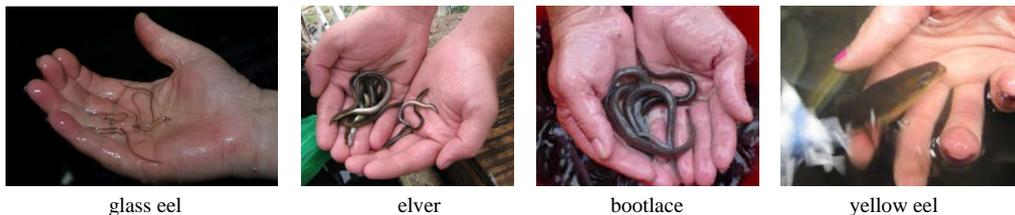
Minutes from the Review Group on Eels). The argument for that is that eel is semelparous (each eel reproduces only once in its lifetime), which makes the stock vulnerable to short-term fluctuations. Therefore, it is argued, a framework for short-lived species should be applied, in which anthropogenic mortality is reduced to zero immediately whenever spawning stock biomass is below the threshold – not gradually reduced in proportion to the spawning stock biomass. ICES (2014), however, argued that it is the number of year classes that contribute to the spawning in any particular year - rather than the number of years an individual eel spawns - that determines the vulnerability to short-term fluctuations. The eel being an extremely long-lived species with many year classes (up to 50) spawning simultaneously (ICES 2014), none of the risks involved in depleting short-lived species actually applies to eel.

Both the Eel Regulation (Anon. 2007) and the Swedish Eel Management Plan (Anon. 2008) have set a long-term goal. The Eel Regulation aims to reduce anthropogenic impacts to achieve a recovery “in the long term” (Art. 2.4). The Swedish Eel Management Plan subscribes to the objectives of the Eel Regulation and emphasises a rapid increase of silver eel escapement, to a level at which the stock decline is expected to stop or turned into an increase (section 5.1) – but the Swedish EMP does not aim at full recovery in the shortest possible time, does not aim at recovery at maximum speed. In accordance with these, the ‘long-lived’ reference framework is applied here, as before (Dekker 2012, 2015).

For other anthropogenic impacts (predation, pollution, spread of parasites, disruption of migration by transport, possibly 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.

## 2 Recruitment indices

There is no dedicated monitoring of natural recruitment to inland waters in Sweden, but the trapping of elvers<sup>2</sup> 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 2 shows the raw observations; Annex B presents an in-depth analysis of temporal and spatial trends in these data.



glass eel

elver

bootlace

yellow eel

(Photos: Jack Perks, Ad Crable, Deutsche Welle, Lauren Stoot)

<sup>2</sup> 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.

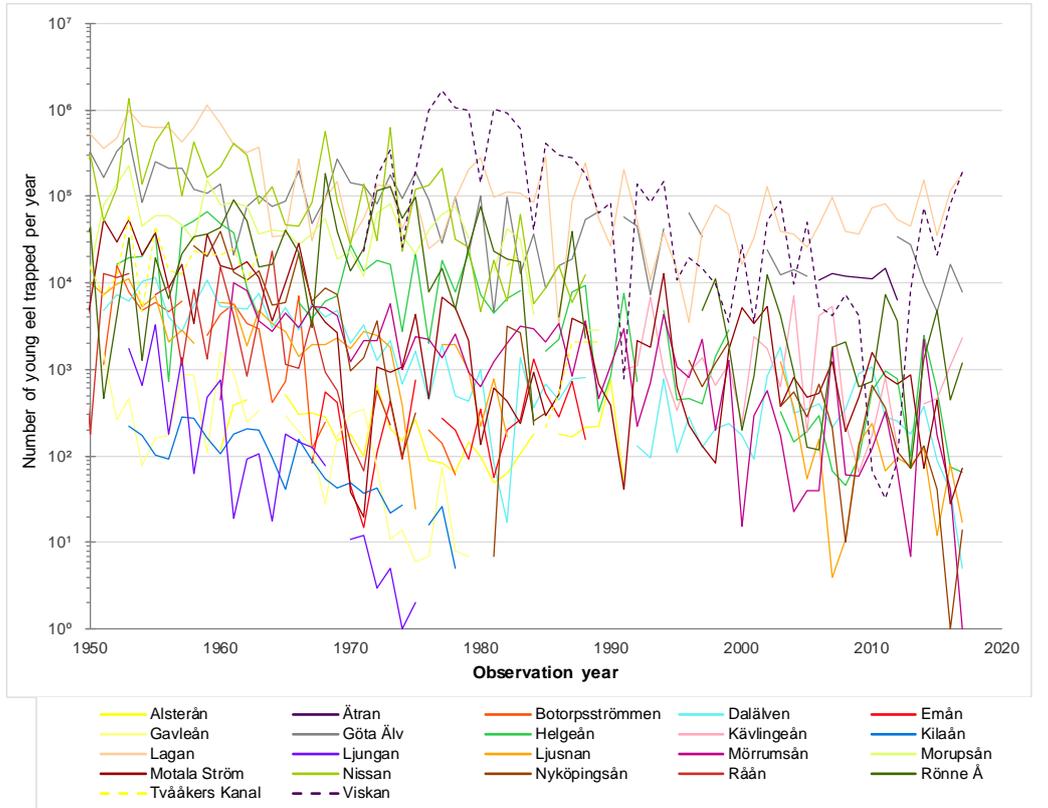


Figure 2 Trends in the number of elvers trapped at barriers, in numbers per year. Note the logarithmic character of the vertical axis. For further details, see Annex B.

The nuclear power plant at Ringhals takes in cooling water in front of the coast along 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 3).

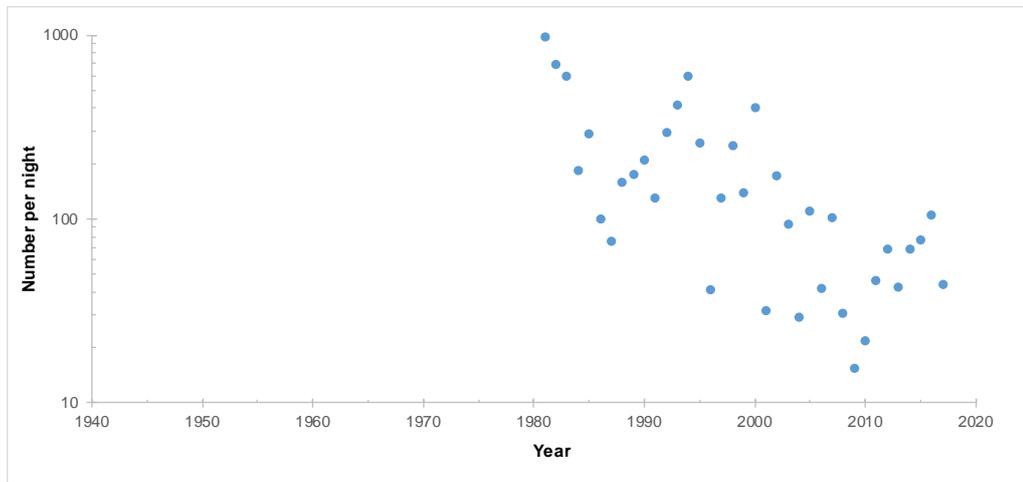


Figure 3 Time trend in glass eel recruitment at the Ringhals nuclear power plant on the Swedish Kattegat Coast. Note the logarithmic character of the vertical axis.

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 and 2010. In 2011, there was no sampling due to technical problems (Figure 4).

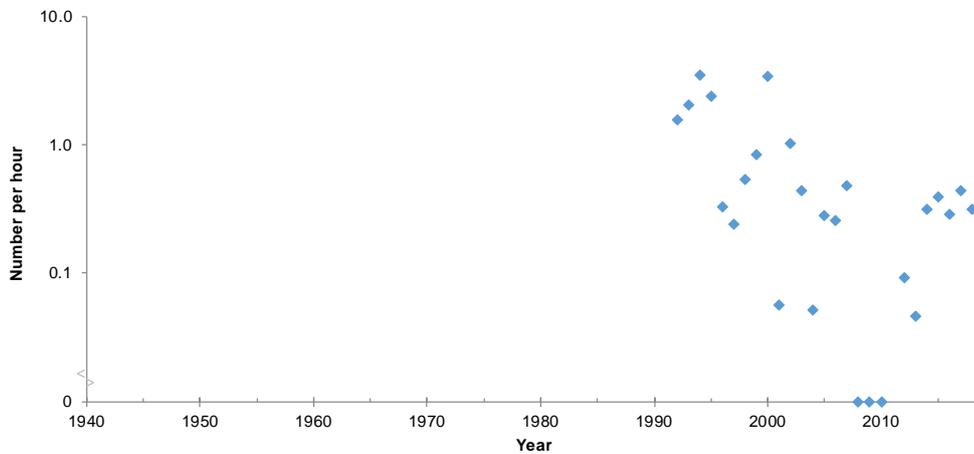


Figure 4 Catch of glass eels (number per hour trawling) by a modified Methot-Isaacs-Kidd Midwater trawl (MIKT) in the Skagerrak-Kattegat 1992–2011. In 2008–2010, zero glass eels were caught; in 2011, no sampling took place. Note the logarithmic character of the vertical axis.

## 3 Restocking

Restocking (stocking) is the practice of importing young eel from abroad (England, France, in historical times also Denmark) and releasing them into outdoor 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). 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 (next page) 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.

Year	Actual numbers			year class	Glass eel equivalents		
	West coast	Inland waters	Baltic coast		West coast	Inland waters	Baltic coast
2000		1 437 378	566 722	2000	9 600	834 967	178 040
2001		969 108	376 597	2001	8 824	1 254 604	441 519
2002	24 255	1 117 322	486 184	2002		331 332	442 889
2003	12 502	463 751	516 713	2003	15 838	880 273	284 157
2004	21 625	939 356	368 156	2004		897 128	198 150
2005	6 195	915 822	187 667	2005		990 340	396 843
2006		940 781	375 847	2006	7 919	794 300	210 397
2007	7 500	777 033	201 576	2007		1 066 454	421 212
2008		1 121 863	398 927	2008		581 853	220 361
2009		564 254	212 002	2009	190 055	1 786 565	65 463
2010	180 000	1 694 510	62 000	2010	573 333	2 089 301	108 754
2011	543 000	1 977 984	103 000	2011	583 892	2 030 630	93 972
2012	553 000	1 924 022	89 000	2012	614 089	2 062 562	128 815
2013	581 600	1 953 984	122 000	2013	822 106	2 129 771	160 491
2014	778 611	2 017 432	152 000	2014	896 691	1 000 207	77 078
2015	849 250	944 144	73 000	2015	1 565 881	1 405 703	56 805
2016	1 483 035	1 334 362	53 800	2016	527 481	415 741	56 707
2017	499 574	394 074	53 707	2017		3 372	

### 3.2 Restocking and stock assessments

Where eels have been restocked, the yellow eel stock consists of a mix of natural and restocked individuals. This may or 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  $B_{\text{best}}$  was estimated at 1 154 t (Dekker 2012), and current restocking (0.5 million in 2017) will potentially produce considerably less than 100 t. For the Baltic coast, the potential

production of silver eel  $B_{\text{best}}$  was estimated at 3 770 t (Dekker 2012), and current restocking (0.05 million in 2017) 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.

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 300 t, while the natural silver eel production in 2017 is estimated at 27 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. Though the framework of stock indicators 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  $\Sigma A$ , expressing the relation of the actual silver eel escapement  $B_{\text{current}}$  to the current potential escapement if no anthropogenic actions had influenced the stock  $B_{\text{best}}$ , can be interpreted in two different ways. If the silver eel produced from restocking is included in the estimate of  $B_{\text{best}}$  (say  $B_{\text{best}}^+$ ), that is  $\Sigma A^+ = -\ln(B_{\text{current}}^+/B_{\text{best}}^+)$ , 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  $B_{\text{best}}$  (say  $B_{\text{best}}^-$ ), the resulting indicator  $\Sigma A^- = -\ln(B_{\text{current}}^+/B_{\text{best}}^-)$  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,  $\Sigma A^-$  expresses the net effect of all anthropogenic impacts, including detrimental impacts and the compensatory effect of restocking.

Within the ICES framework for advice, the limit mortality level is related to the spawning stock biomass: below a certain threshold biomass level, lower mortality limits are advised (the upward curve between the orange and the red area in Figure 7). When restocking is applied to augment the natural stock, the silver eel production will increase – consequently, a higher mortality limit will apply. At the same time, the interpretation of restocking as a compensatory measure for other anthropogenic mortalities results in an estimate of  $\Sigma A$  that does *not* represent the actual mortality experienced by any eel in the stock, but represents the combined effect of true mortalities and the beneficial effect of restocking. Due to the higher mortality limit, the true anthropogenic mortality on the natural recruits can even be allowed to be *higher* than without restocking. Applying both

a relaxed mortality limit, as well as interpreting restocking as a compensation for other anthropogenic mortalities appears to be a case of double banking.

ICES (2012) used stock indicators reported by individual countries, to derive a population-wide assessment of the status of the European eel stock. Because different countries used different calculation procedures, the resulting international indicators were based on a mix of approaches. For instance, Germany (Oeberst and Fladung 2012) included restocking in its estimates of  $B_{\text{current}}$ , but not in  $B_{\text{best}}$ ; hence, the estimate of  $\Sigma A$  reflected the combined effect of detrimental impacts and beneficial restocking, but not a true mortality rate. Sweden (Dekker 2012) included restocking in the estimates of both  $B_{\text{current}}$  and  $B_{\text{best}}$ ; hence, the estimate of  $\Sigma A$  constituted a true mortality rate, but did not reflect the effect of restocking.

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. 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, there will be areas where they do and do not apply, leading to confusing results.

The Eel Regulation considers both restocking and reducing anthropogenic mortalities as contributions to the protection of the stock. Interpreting restocking as a compensatory measure and discounting the estimate of  $\Sigma A$  for it, however, might lead to situations where large quantities of eel are restocked into areas of high mortality. This would result in a net increase of the biomass of silver eel escaping (compared to the situation without restocking), but a high number of restocking would be required to cope with the high mortality. Using a mortality indicator that interprets mortality as a compensation for other mortalities, i.e.  $\Sigma A^- = -\ln(B_{\text{current}}^+/B_{\text{best}}^-)$ , the indicator would not flag this situation. To avoid this, the positive effect of restocking will not be included in our estimates of mortality  $\Sigma A$ , and – where possible - biomasses of silver eel are expressed separately for eels of natural and of restocked origin. That is: we use  $\Sigma A^+ = -\ln(B_{\text{current}}^+/B_{\text{best}}^+)$ . For the status of the stock relative to pristine conditions ( $\%SSB = 100 \cdot B_{\text{current}}/B_0$ ), this report provides estimates *with* and *without* including restocking into the estimate of  $B_0$  (Figure 7).

## 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 Statistical Bureau 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, fishers have reported their landings directly to the responsible national agencies. Where data series overlapped, precedence has been given here 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. Dekker and Sjöberg (2013) present the assessment of the impact of the fishery, including a reconstruction of the breakdown by county for the years 1979-1999. Figure 5 shows this reconstruction (shaded). 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 EMP ( $\Sigma F=2.33$ , averaged over the years 2000-2006) and the assumption that the stock had not changed considerably in recent years. In spring 2012, the fishery has been closed completely, i.e.  $\Sigma F=0$ . 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 and methods 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 is repeated in this report,

with some modifications (see Annex B and Annex C for details). Trends in catch and fishing impact are detailed in Table 2; the trend in the catch is depicted in Figure 6.

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  $\Sigma F$ ). Trends in catch and fishing impact are detailed in Table 2; the trend in the catch is depicted in Figure 5.

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 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. 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, the recreational fishery is no longer allowed (except in some designated waters, generally above three hydropower generation plants)).

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

Year	Landings (tonnes)			Fishing mortality (rate)		
	West coast	Inland waters	Baltic coast	West coast $\Sigma F$	Inland waters $\Sigma F$	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	0.1
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	0.02
2014	0	111	213	0	0.29	
2015	0	88	158	0	0.17	
2016	0	97	181	0	0.21	
2017	0	102	143	0	0.25	

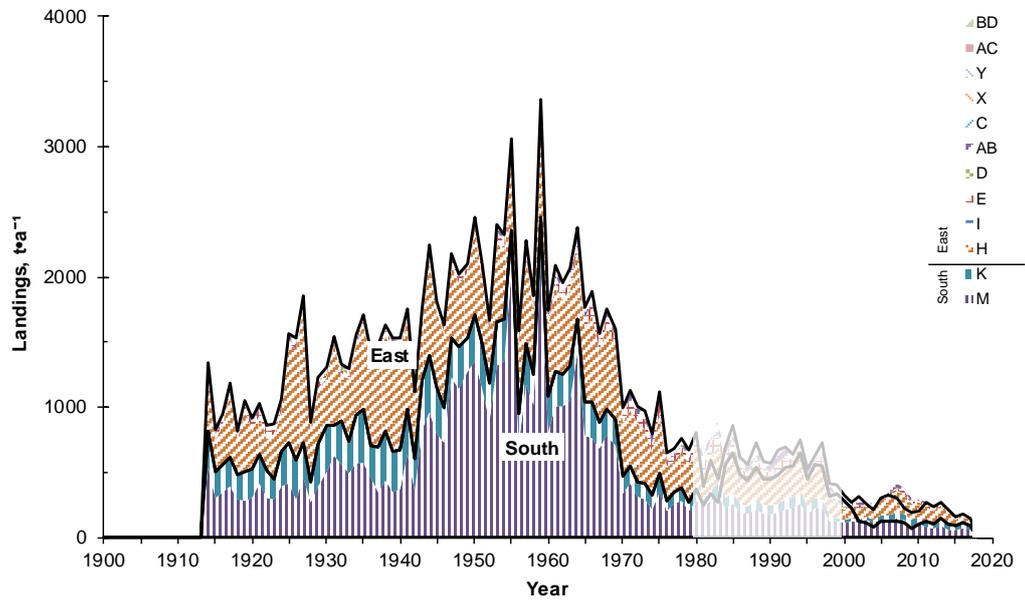


Figure 5 Trend in landings from the coastal fisheries, by county (colours) and area (black lines). In the years 1978-1998 (faded), due to lack of detailed records, it has been assumed that the percent-wise contribution of each county had remained constant. Note that the total landings on the Baltic coast come predominantly from six counties (AB, E, H, K, M, O) and that the contribution from other areas is barely visible in this graph.

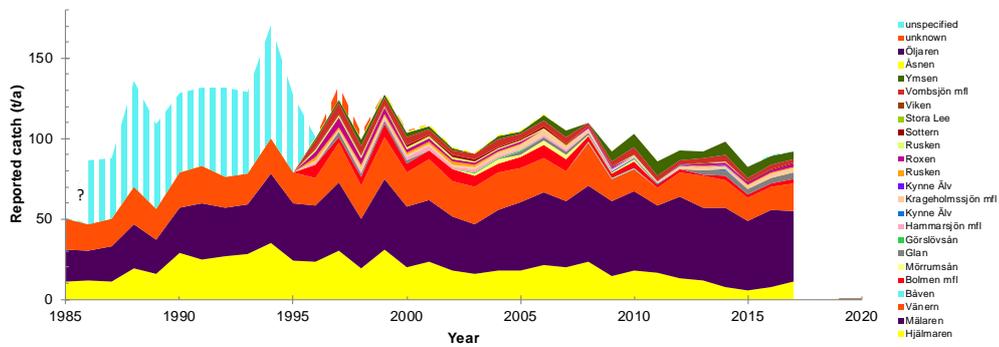


Figure 6 Trends in landings from inland waters. Before 1996, only the totals for all lakes (except the three largest ones) are known; statistics before 1986 are not available.

## 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. In recent years, restocking has shifted relatively more towards lakes with hydropower stations downstream, 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.*

Year	Biomass of silver eel (tonnes)			Hydropower mortality $\Sigma H$ (rate)		
	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000		195			0.83	
2001		166			0.68	
2002		151			0.57	
2003		128			0.46	
2004		99			0.36	
2005		82			0.31	
2006		69			0.27	
2007		86			0.32	
2008		111			0.42	
2009		152			0.54	
2010		166			0.60	
2011		191			0.67	
2012		203			0.75	
2013		193			0.75	
2014		177			0.75	
2015		193			0.81	
2016		168			0.83	
2017		141			0.80	

## 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 7 on page 83 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.*

Year	Biomass of silver eel (tonnes)			As mortality (rate)		
	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			

## 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.

As there are legal fisheries in inland lakes as well as along the Baltic coast, there are probably also trading channels used by law-breaking agents. Along the Swedish west coast, eel fishing is prohibited since 2012, and trade in illegally fished eel is less likely. Recreational eel fishing is only allowed in some designated waters, generally above three hydropower generation plants. However, selling the catch is not allowed. Whether and how unauthorized trade occurs is not known by us or any other agency.

There has been a long tradition of fixed eel fisheries in streams, but the extent of this legal and/or illegal fishery is not known.

As eel fisheries along the west coast have been closed and eel fishers elsewhere give in, there are probably high numbers of fyke nets available on the market, easily acquired for illegal fishing activities. When fishing illegally, such gears are not marked, i.e. there are no floats on the surface to observe, and specialised skills are required to find them under water.

The number of eels in seized fishing gears were not always counted and, as it is impossible to know for how long an illegal gear has been in use before being disclosed, all estimates are very unreliable. However, based on information on the number of

fykenets seized, the catch per day/night as observed in SLU's test-fishing, and a speculative assumption (range presented) on the number of fyke nets used, we come to the following range of estimates. A conservative estimate of 500 fyke nets fishing along the east coast for four months yields about 5.5 t/a, while a high estimate of 5000 fyke nets fishing for six months results in about 83 tonnes. The corresponding estimates for the west coast, using three times higher CPUE than on the east coast, results in 18 and 270 tonnes respectively.

As close to 350 illegal fyke nets were disclosed in 2017 only, from a very restricted part of the Baltic coast, the estimate of 500 fyke nets in total is probably far below reality.

In comparison, the Swedish commercial catch in the Baltic Sea amounted to 184 tonnes in 2016; thus a realistic estimate of the total IUU in Sweden could be of the same magnitude as the reported commercial landings.

To improve the estimates of IUU landings of eel, we recommend the following: the number of eels in seized fyke nets should be counted and preferably weighed, at least in total per site. Similar controls should be done also in freshwater. The occurrence and use of fixed eel fisheries based on historical permits should be investigated as well as the extent of the legal recreational fishery for eel.

Having only an order-order-of-magnitude estimate for a recent years – 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, eco-system effects), which might have contributed to the decline of the eel stock. Limiting or reducing the predator abundance might enhance the status of the eel stock. In this context, cormorants (*Phalacrocorax carbo carbo*, *P. carbo sinensis*, and *P. aristotelis*) as well as seals (*Phoca vitulina*, *Pusa hispida*, and *Halichoerus grypus*) have been discussed.

In a recent literature review, Hansson et al. (2017) showed that, in the southern Baltic Sea, the eel consumption by cormorants was in the same order of magnitude as the fishing impact (and no impact from seals). For inland waters, the cormorant impact has been studied in several lakes, but 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 104, 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. 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 Anon (2008). In spring 2012, the fishery has been closed, and since then, 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, indicating that the decreasing recruitment in past years leads to a further decline in stock abundance, especially of the size classes below the (former) minimum legal size. The closure of the fishery in 2012 has led to a better survival into larger size classes, and a relative recovery of their abundance, but that abundance cannot be quantified in absolute terms.

For inland waters, Annex C presents a comprehensive and fully updated assessment, from which most stock indicators were derived. For the pristine biomass (the biomass of silver eel in the absence of any anthropogenic mortality, at historical recruitment), the previous estimate (300 t plus the contribution from restocking) is copied from Dekker (2012) - now using the updated estimates of the contribution from restocking. Mid-term extrapolations 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.

For the Baltic coasts, the assessment in Annex D covers the impact of the Swedish silver eel fishery. Other impacts on the same eels (in earlier life stages, often 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, Dekker (2012, 2015) derived estimates of lifetime anthropogenic mortality  $\Sigma A$  from the analysis in Dekker & Sjöberg (2013); estimated  $B_{best}$  from the ratio of landings to  $\Sigma A$ ; 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  $\Sigma A$  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), we report them as partial indicators here, and leave the estimates of  $\Sigma A$ ,  $B_{best}$  and  $B_0$  missing. Over the years 2010-2017, the fishing mortality  $F_{SE}$  is estimated at approx. 2 %; the average catch was 223 t/a, resulting in an estimate of the silver eel run along the Swedish coast ranging from 973 t/a (Södermanland) to 4108 t/a (Blekinge). 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.

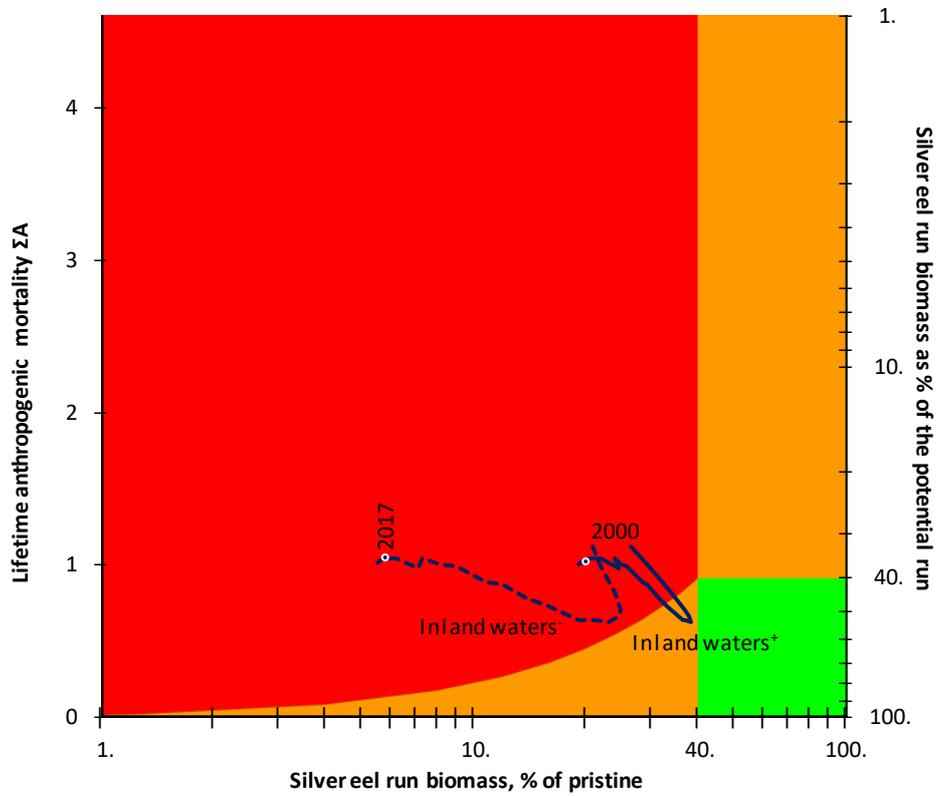


Figure 7 Precautionary Diagram for the Swedish eel stock in inland waters. For the west coast and the fisheries along the Baltic coast, no stock indicators are currently available. For inland waters, the true mortality is shown (that is: not interpreting restocking as compensation for other mortalities), giving separate curves for the current biomass with (+, solid) or without (-, dashed) the contribution from restocking. (For the details of the diagram, see Dekker 2010, 2016).

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,  $\Sigma H=0$ , hence  $\Sigma F=\Sigma A$ . 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.

year	West coast						Inland waters								Baltic coast						T&T		year					
	B <sub>current</sub>	B <sub>best</sub>	B <sub>0</sub>	%SSB	$\Sigma A$	%SPR	with restocking +				without restocking -				Mortality rates				B <sub>current</sub>	B <sub>best</sub>	B <sub>0</sub>	%SSB		$\Sigma A$	%SPR	B <sub>current</sub>	B	
							B <sub>current</sub> <sup>+</sup>	B <sub>best</sub> <sup>+</sup>	B <sub>0</sub> <sup>+</sup>	%SSB <sup>+</sup>	B <sub>current</sub> <sup>-</sup>	B <sub>best</sub> <sup>-</sup>	B <sub>0</sub> <sup>-</sup>	%SSB <sup>-</sup>	$\Sigma F$	$\Sigma H$	$\Sigma A$	%SPR										
2000					1.79		162	471	567	28.6	70	204	300	23.4	0.28	0.79	1.07	34.4	3507									2000
2001					2.53		183	469	581	31.4	73	188	300	24.4	0.30	0.65	0.94	38.9	3473									2001
2002					2.41		209	462	589	35.5	79	174	300	26.2	0.25	0.54	0.79	45.2	3497									2002
2003					2.15		230	455	594	38.7	82	162	300	27.2	0.24	0.44	0.68	50.5	3495									2003
2004					2.43		236	448	596	39.5	80	151	300	26.6	0.29	0.35	0.64	52.7	3516									2004
2005					2.39		235	433	594	39.6	75	138	300	25.1	0.31	0.30	0.61	54.4	3424									2005
2006					2.66		231	428	600	38.5	69	128	300	23.0	0.35	0.26	0.62	54.0	3404									2006
2007					1.91		234	434	617	38.0	63	117	300	21.0	0.30	0.31	0.62	54.0	3352									2007
2008					1.86		221	451	644	34.4	52	107	300	17.5	0.30	0.41	0.71	49.1	3381									2008
2009					1.19		218	467	669	32.5	46	98	300	15.2	0.23	0.53	0.76	46.6	3460									2009
2010					1.20		208	480	689	30.2	39	91	300	13.1	0.26	0.59	0.85	42.9	3463						5			2010
2011	12	1154	1154	1	0.93	39	207	486	702	29.5	36	84	300	11.9	0.22	0.65	0.87	41.9	3499					5	3			2011
2012					0		187	481	704	26.6	30	77	300	10.0	0.23	0.73	0.96	38.1	3531					9	2			2012
2013					0		177	459	689	25.7	27	70	300	9.0	0.25	0.74	0.98	37.4	3499					10	4			2013
2014					0		164	430	666	24.6	24	64	300	8.1	0.28	0.73	1.01	36.3	3557					15	7			2014
2015					0		161	398	639	25.1	24	59	300	7.9	0.17	0.79	0.96	38.5	3612					13	6			2015
2016					0		135	355	601	22.5	21	54	300	6.9	0.21	0.81	1.02	36.2	3589					13	6			2016
2017					0		120	314	564	21.2	19	51	300	6.4	0.24	0.78	1.02	36.0	3627					13	6			2017

## 9 Discussion

### 9.1 Comparison to the 2015 assessment

For the west coast stock, Dekker (2015) did not present an assessment, advocating that a comprehensive monitoring plan should be developed. Andersson et al. (2018) effectively did so, concluding that no realistic option 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 to monitor 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.

For the inland stock, the current assessment updates and improves the assessment of Dekker (2015). Comparing to that previous assessment, current results were mostly affected by two changes. First, the analysis of recruitment trends has been revised, which now leads to slightly higher (and consistently non-negative) estimates of the natural recruitment. Secondly, the Length-Weight-relation has been updated, now better reflecting the observations. Both of these changes result in a somewhat higher estimate of the stock biomass, but do not affect the yield (biomass) in the fisheries (observed), or the impact (mortality) of hydropower (observed and/or assumed). As a consequence, estimated fishing mortality is somewhat lower, and the biomass of the stock (in percentage of the pristine status) slightly higher. This does not alter the evaluation of the status of the stock (biomass below the minimum target, anthropogenic impacts exceeding a sustainable level and rising).

The assessment of the inland stock in this report is based on a detailed reconstruction, taking the young eel (natural recruits, assisted migration and restocking) as a starting point for the reconstruction. Dekker (2015)

recommended to ground-truth results on independent stock surveys of yellow eel (electro-fishing in streams, fyke-netting in lakes). In the years since, a start of that ground-truthing analysis has been made. Mixing well-surveyed but mostly un-reconstructed habitats (rivers), with hardly-surveyed but here reconstructed habitats (lakes), however, requires an extremely complex analysis. Though progress has been made, no results can be shown yet.

For the silver eel fisheries on the Baltic coasts, the current assessment methodology is identical to the 2015 one; the database has been extended. As before, estimates of fishing impact are derived, pooled by decade. In 2015, however, only 10 tagging experiments (94 recaptures out of 1353 releases) were available for the 2010s, and population estimates were highly influenced by uncertainty. In 2018, 6 more experiments (65 recaptures out of 989 releases) now contribute to the 2010s results, and data uncertainty is less of an issue: no problematic divisions of near-zero by near-zero occurred. Noting that the main result (estimated fishing impact) in the current assessment is almost identical to the previous estimate, it appears that data shortage is no major issue, when pooling a whole decade. This implies, however, that a rapid evaluation of management measures (if and when needed) will require a more intense mark-recapture programme.

Recent tagging experiments (Figure 52) were more evenly spread along the coast than the historical experiments (Dekker & Sjöberg 2013; their Figure 4), and the distance from release to recapture showed a meaningful trend. The number of days between tagging and recapture, however, recently declined – most likely as a consequence of restrictions on the length of the fishing season. First tags, applied at the start of 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 2015, 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. In contrast to the 2015-assessment, no partial indicators (covering the Swedish fishery only) have been reported in place of the requested full-lifetime indicators.

## 9.2 Requirements for the 2018 reporting to the EU

A template for reporting stock indicators has been circulated by the EU-Commission. Additionally, the 2015 reporting and subsequent international evaluation indicate what information is required. 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 the previous section. Only the current assessment of the inland stock does produce all requested indicators.

The templates ask for quantities of silver eel (or “silver eel equivalents”), split over the different mortality factors. Table 2 and Table 3 present that information for the fishery resp. the impact from hydropower. However, it should be noted that these quantities do not constitute independent impacts. An individual eel can be derived from restocking, later on be fished, and finally released near the sea to prevent hydropower-related mortality. For example, changing the quantities restocked will affect the fishery, the Trap & Transport-programme, the hydropower mortality and the escapement; reductions in the fishery will for the major part be annihilated by the subsequent mortality in the hydropower; and so forth. Hence, care should be taken in the interpretation of those Tables, and double counting be avoided.

## 10 Recommendations and advice

In this report, an assessment of the Swedish part of the European eel stock is presented, extending and updating the results of the previous assessments (Dekker 2012, 2015). The national stock indicators were and will be used for the international assessment (ICES 2013a, 2015), on which the international advice is based. In compiling the international assessment, national stock indicators were taken at face value, and conclusions and advice focused on the status of the international stock. In 2018, an evaluation and review of national assessments is scheduled, focused on the quality of the assessments. This chapter fills the gap between national assessment and international advice, 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 (and hence  $\Sigma A$ ) is zero (disregarding illegal catches), but current, current potential and pristine biomasses ( $B_{\text{current}}$ ,  $B_{\text{best}}$  and  $B_0$ ) could not be determined. However, current stock biomass is undoubtedly far below the recovery target, and stock surveys indicate that the stock in general is still in decline. 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 (anthropogenic mortality is zero).

Restocking on the west coast (to support recovery and/or to compensate for mortality in inland waters) is expected to contribute to the stock, but - given the small quantities applied and the small expected effect in comparison to natural recruits - that effect will be too small to quantify.

For the inland stock: status indicators point out that the stock biomass is below the limit level, anthropogenic impacts (fishery and hydropower, together) exceed the minimum limit that would allow recovery, and those

anthropogenic impacts are increasing. Management actions include assisting migration, restocking, fishing restrictions and Trap & Transport. These measures have strong interactions: adjusting one measure, 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 limits, and/or effectively losing natural habitats. Most current management actions are based on the 2008 assessments (included in the national Eel Management Plan; Anonymous 2008), which is fully outdated. It is recommended

- to urgently reduce anthropogenic impacts on the inland stock, and/or
- to develop an updated, comprehensive 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, but this fishery is just one of the anthropogenic impacts affecting the Baltic eel stock. No comprehensive assessment 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. Fishing restrictions have reduced the fishing impact, but 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 mark-recapture experiments, and to embed this in a pan-Baltic, comprehensive assessment.
- to coordinate national protective measures with other range states, i.e. integrated management in the Baltic.

Considering the international context, assessments and indicators for the Swedish part of the European eel stock are produced in this report, fitting the international assessment framework. For the west coast, however, no assessment could be made; for inland waters and the Baltic coast fishery, results could not be verified on independent ground-truth. Assessments and assessment methodologies were largely determined by the availability of data and budget. Though a consistent set of stock indicators is achieved within Sweden, inconsistencies and interpretation differences at the international level complicate their usage – in particular: un-standardised assessment

methodologies and conflicting ways of calculating and interpreting stock indicators are noted. To address this situation, it is recommended

- to coordinate and standardise the coming tri-annual reportings internationally more thoroughly,
- to initiate international standardisation/inter-calibration of monitoring and assessment methodologies among countries, achieving a consistent and more cost-effective assessment across Europe.

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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 län) has been closed completely. We discuss the historical development of that fishery, and present recent information on the west coast eel stock, including recent restocking.

### A.1 Development of the west coast eel fishery

There are two different time-series reported 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 fishermen (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 late 1800 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 8). Around this time, fyke net fishery 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, fishermen 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 8). Technical development of fyke nets and boats allowed catches to remain around 250 t/a although the number of coastal fishermen decreased (Figure 9). The first fyke nets were hand-made, heavy and large, and required high maintenance (frequent cleaning, tarring, and drying). Some fishermen had two sets of fyke nets and replaced the used ones with newly cleaned nets, while others switched to fishing for other species. 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.

The growth of the west coast eel fishery opened the opportunity for Danish traders to develop a trading route from the Swedish west coast to Denmark and Germany. In early 1900, Danish traders visited fishermen along the Swedish west coast to buy live eel for export to Germany. 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 corves 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 the 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 8).

Most of the eel was exported (Figure 10); 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. 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 fishermen from early to mid-1900 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.

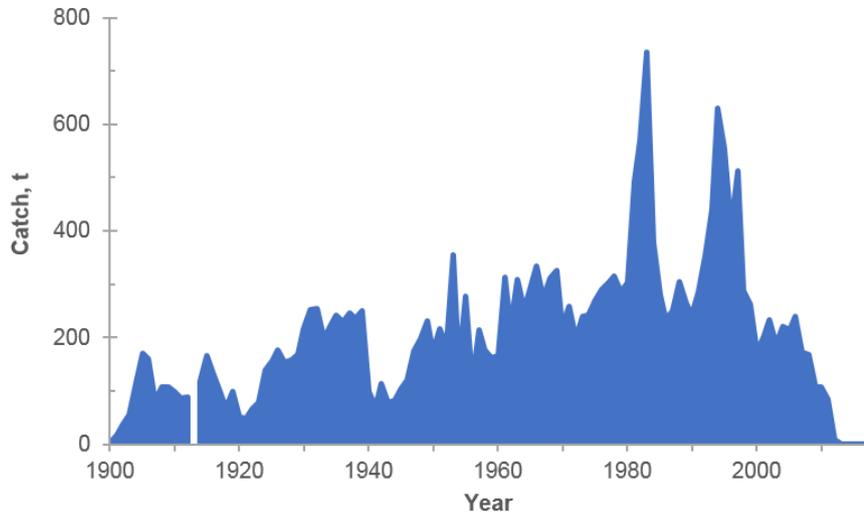


Figure 8 Time trend in eel catch in Kattegatt and Skagerrak from 1900 to 2017 (catch in the period 1970-1984 is solely based on landings data).

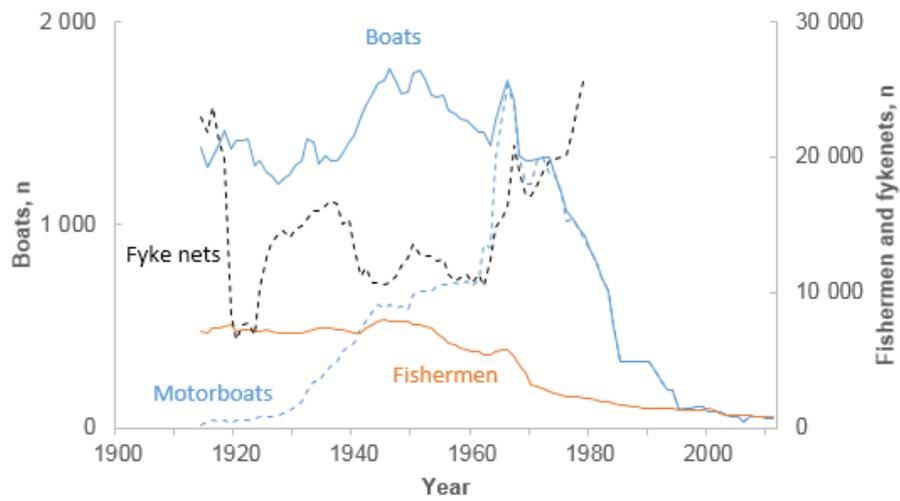


Figure 9 Time trend in number of small boats, fyke nets and fishermen on the Swedish west coast.

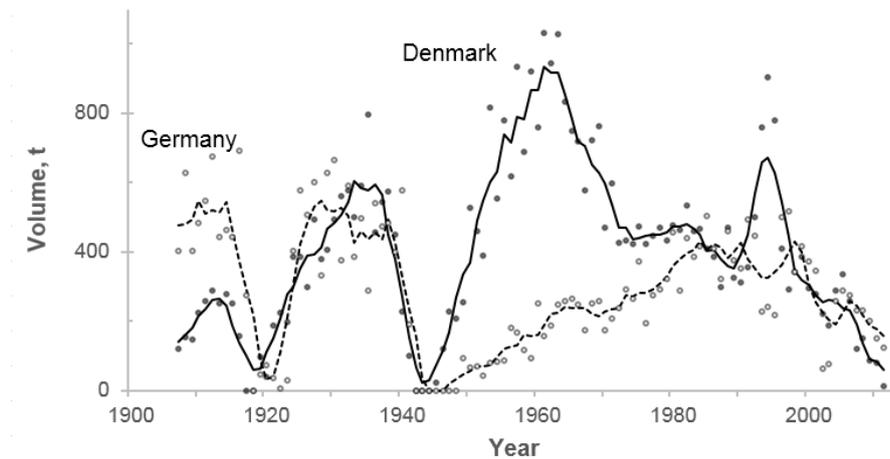


Figure 10 Time trend in total Swedish eel export to the two major receiving countries Denmark and Germany.

## A.2 Trends in the west coast eel stock

In the Swedish Eel Management Plan (Anonymous 2008), a fishery-dependent assessment was presented, analysing length-frequency data and catch statistics from that fishery. When the 2012 post-evaluation report was compiled (Dekker 2012), it was already known that the fishery would be closed, i.e. that the fishery-based assessment could not be continued.

Since the closure of the fishery in spring 2012, the stock is recovering. The current status of the stock most likely reflects: the past trend in recruitment, the overexploitation in the past, and the recovery since 2012. Unravelling these processes from 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.

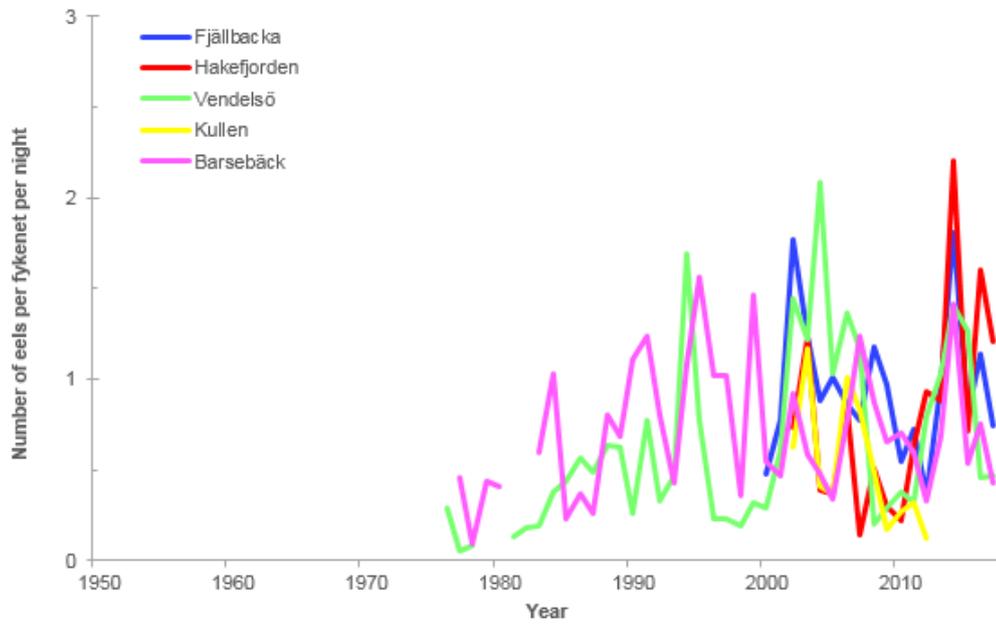


Figure 11 Time trend in the catches of the fishery-independent fyke net survey at various places along the west coast.

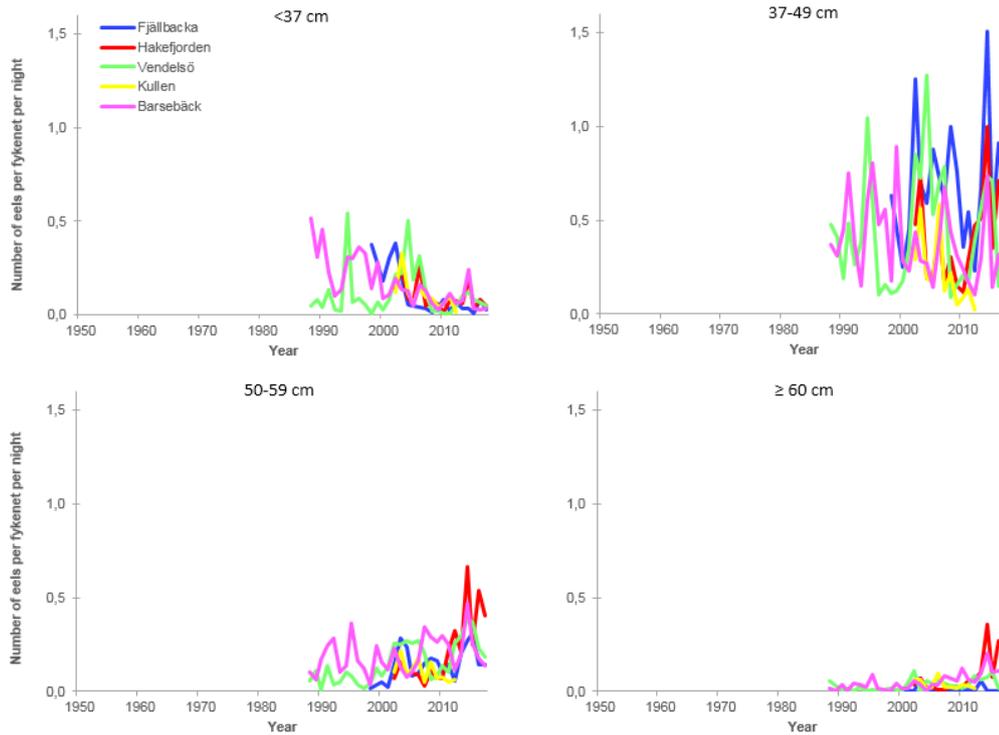


Figure 12 Time trend by size-class (total length) in the fishery-independent fyke net survey at various places along the west coast.

### A.3 Restocking in coastal waters

Restocking has interacted with the stock on the west coast in two ways. Since the early 1950s, medium sized eel has been harvested on the west coast, and transported to the east coast. And since the mid-1970s, glass eel has been imported and released on the west coast. Until the year 2000, the amount of young eel extracted effectively exceeded the amount of glass eel released (Dekker 2012, Figure 13), but since then, the extraction has come to an end. In the 2010s, on average 0.8 million glass eels have been restocked per year. This quantity is expected to produce an amount of silver eel of ca. 50 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 production is estimated in the order of 1000 t/a (Dekker 2012), the addition based on the restocking will be relatively small, and therefore hard to detect.

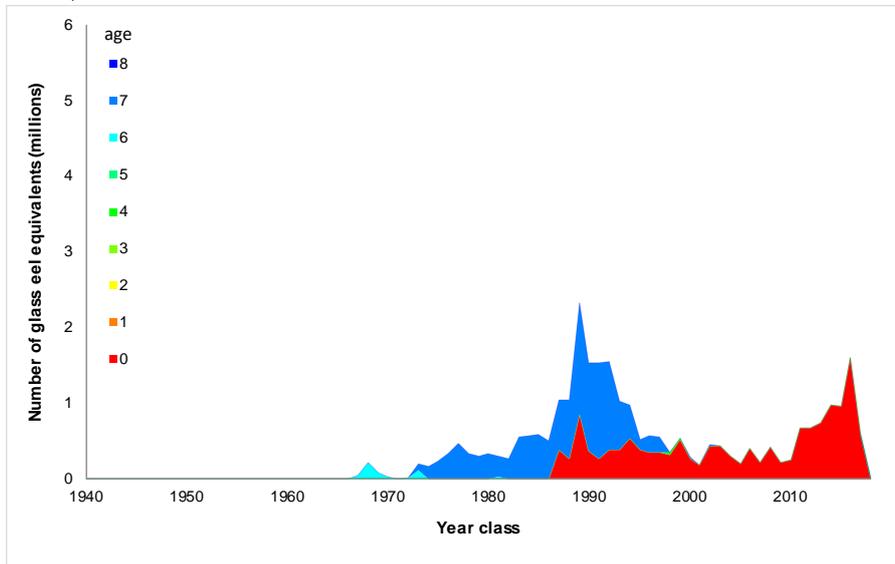


Figure 13 Time trend in the numbers of eel used for restocking in coastal waters, expressed in glass eel equivalents per year. Before 1970, almost no eel had been restocked on the coast. The colour of the symbols indicates at what age the eels were restocked, though all numbers have been converted to glass eel equivalents.

The references for this Annex are included in the reference list of the main report, on page 47.

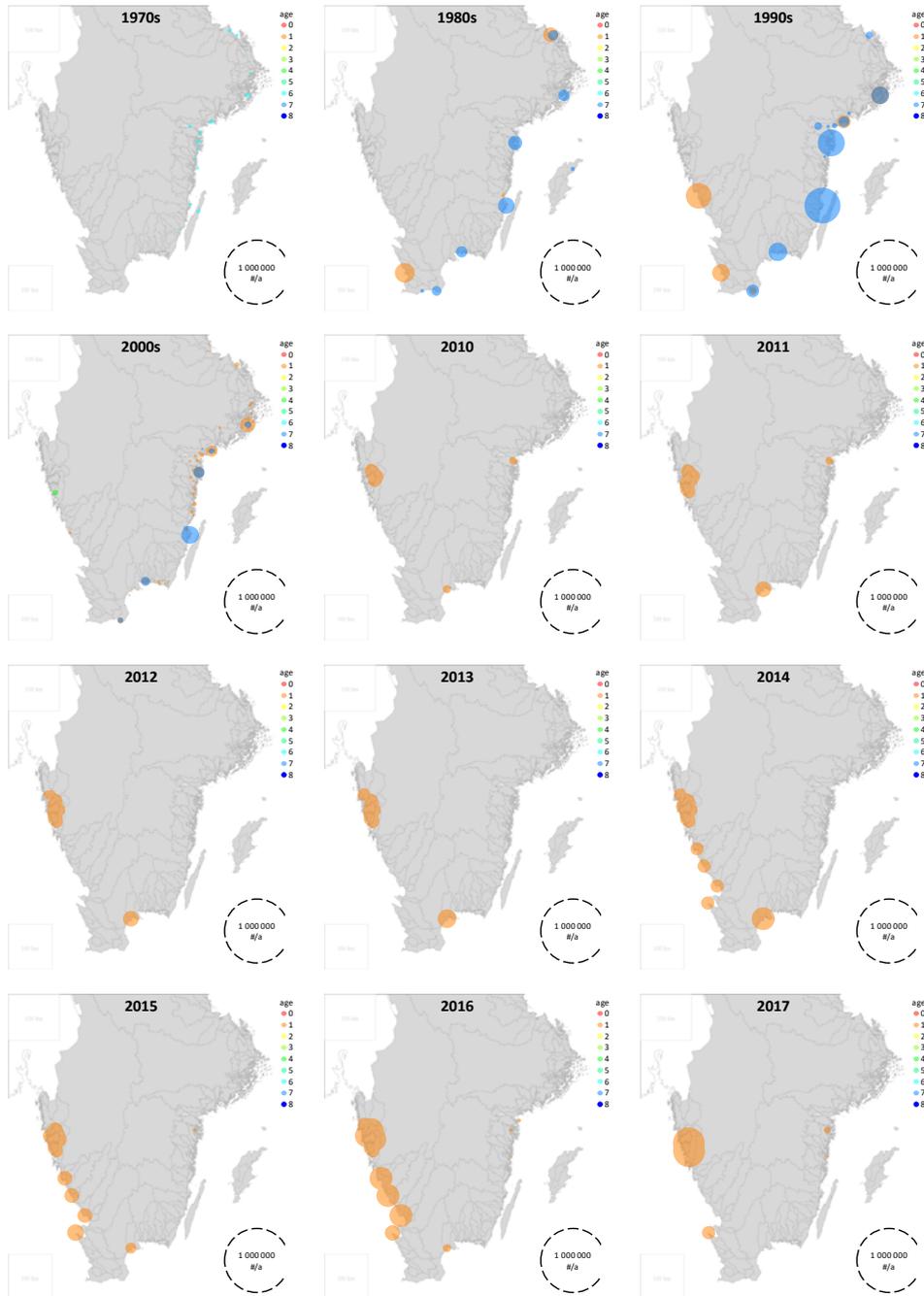


Figure 14 Spatial distribution of the restocking in coastal waters, expressed in glass eel equivalents per year, for decades (1970s – 2000s) or individual years (2010 - 2017). Before 1970, no eel has been restocked on the coast. The colour of the symbols indicates at what age the eels were restocked, though all numbers have been converted to glass eel equivalents.

## Annex B Recruitment into inland waters

The reconstruction of the inland silver eel production (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, which block the whole river, there will be only few eels passing upstream. Hence, considering the set of elver traps as an unbiased and efficient sampling of the natural immigration, this Annex analyses the spatial pattern and temporal trend in these data. This will enable interpolation (for years with missing observations in rivers *with* a trap) and extrapolation (to all rivers *without* a trap).

For the preceding assessment, Dekker (2015) analysed the same data (up to 2014), applying a statistical model that was comparable, but not identical to the one presented below. Following the publication of Dekker (2015), it was realised that the model was inconsistent in the way statistical interaction terms were shaped (Mandel-interactions for *upstream* and *Oslo* in interaction with year class, resp. a cubic spline for the interaction between *age* and year class). Additionally, that analysis frequently gave rise to estimates of natural recruitment somewhat below the actual number of elvers in the traps, which could yield negative estimates of stock biomass in the assessment. Subsequent analysis of a range of alternative models indicated that the main conclusion (on time trends by age, and on potential density dependent effects) were the same, almost regardless of the way the model was exactly chosen. Furthermore, in-depth analysis of the data identified a plausible cause for the negative biomass estimates. A manuscript presenting a consistent statistical analysis, and an in-depth analysis of the lowest observations is in preparation (Dekker and Wickström, in prep.) – here, we copy and update parts of that manuscript. In particular, the data up through 2017 are included, and special

attention is given to the predictions up to 2017, extrapolations even up to 2020. At the end of this Annex, the shortcomings of those extrapolations are discussed.

## **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 in the river Ljungan), up to the far north (Olofsson 1934), 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 12, Table 6), multi-decadal data series are now available. The starting year of these series varies from before 1900 to 1991; some series were discontinued (from 1973 to 1991); and eleven series 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. 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 resp. Rönneå).

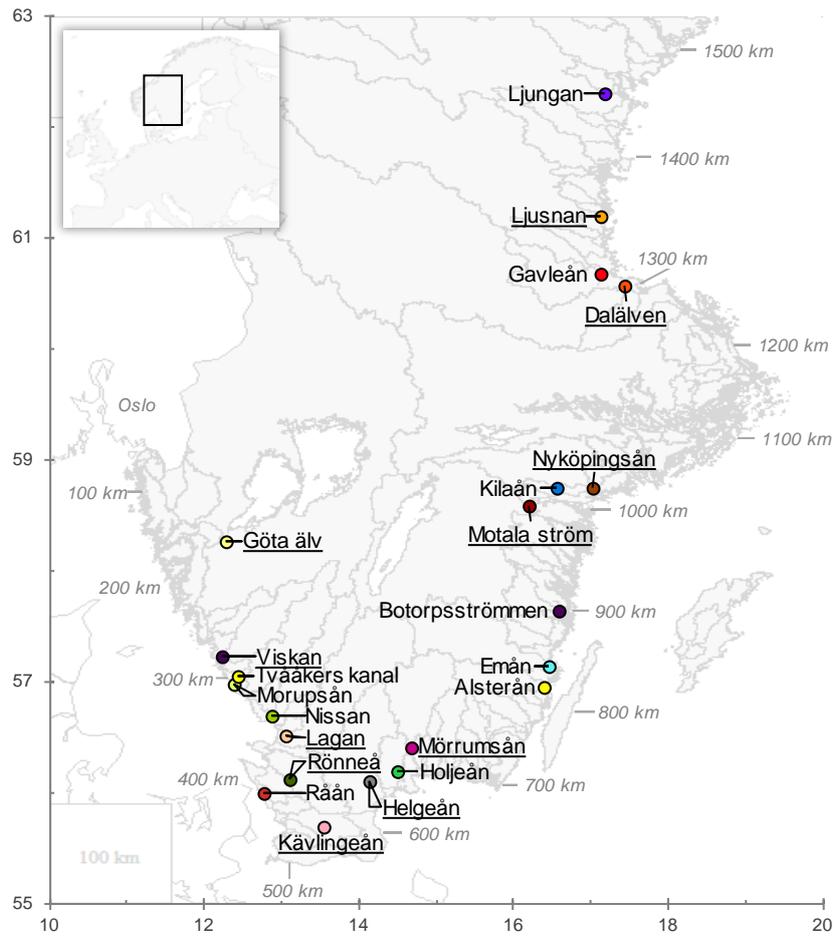


Figure 13 Map of the study area, showing sampling sites, drainage areas and distances along the coast from Oslo. Underlined sites are continuing their sampling up to today.

Table 6 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.

Site	First year	Last year	Valid obs.	Distance Oslo, km	Discharge m <sup>3</sup> /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	1978	33	897	6	0	6	40.5	5.1
Dalälven	1951	ctd	58	1312	348	11	14	59.3	6.0
Emån	1967	1989	21	842	30	4	13	43.8	5.4
Gavleån	1920	1979	23	1327	21	4	7	50.0	5.6
Göta älv	1900	ctd	52	221	518	77	23	9.7	2.6
Helgeån	1952	ctd	58	623	46	35	12	31.2	2.2
Holjeån	1956	1976	20	645	8	26	20	20.9	3.9
Kävlingeån	1991	ctd	25	449	4	49	20	17.2	2.9
Kilaån	1948	1978	24	1023	1	31	19	50.0	5.6
Lagan	1925	ctd	67	363	77	4	37	0.5	0.4
Ljungan	1951	1975	20	1464	138	20	9	69.1	5.9
Ljusnan	1950	ctd	40	1362	230	1	18	43.8	5.3
Mörrumsån†	1960	ctd	57	663	27	32	119	98.3	6.2
Morupsån	1950	1990	38	303	1	11	11	0.4	0.0
Motala ström	1942	ctd	61	1008	93	5	11	49.8	5.6
Nissan	1947	1990	41	350	41	4	13	0.4	0.1
Nyköpingsån	1958	ctd	44	1024	22	4	11	49.8	5.6
Råån	1946	1973	23	416	2	4	13	1.8	1.1
Rönne å	1946	ctd	57	389	24	37	31	1.8	1.1
Tvååkers kanal	1948	1989	30	303	1	7	26	0.5	0.1
Viskan	1971	ctd	46	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 not been continued 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=874.

Characteristics of the 22 sampling sites are given in 6, 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 was 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 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 Statistical analysis

The aim of the statistical analysis 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, which would have explained an additional 1.6 % of the deviance), we keep the option to use our results for the prediction of eel abundance and trends in any other river, for the purpose of assessing the stock in all inland waters in Sweden (c.f. Dekker 2015).

We analyse our data 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 applied, using the default settings of SAS: a cubic B-spline basis with three equally spaced knots positioned between the minimum and maximum year class.
2. The size of the river, coded by the 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 sother 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 67 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 (67 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, contrasting 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 14 and Figure 15), giving partial predictions (for the first year class in each decade) and partial residuals (for each observation, whatever the year class). 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<sup>3</sup>/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**

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 almost 53 million eels, 156 tonnes of young eel, have been 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 respectively 10.2 t. 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.

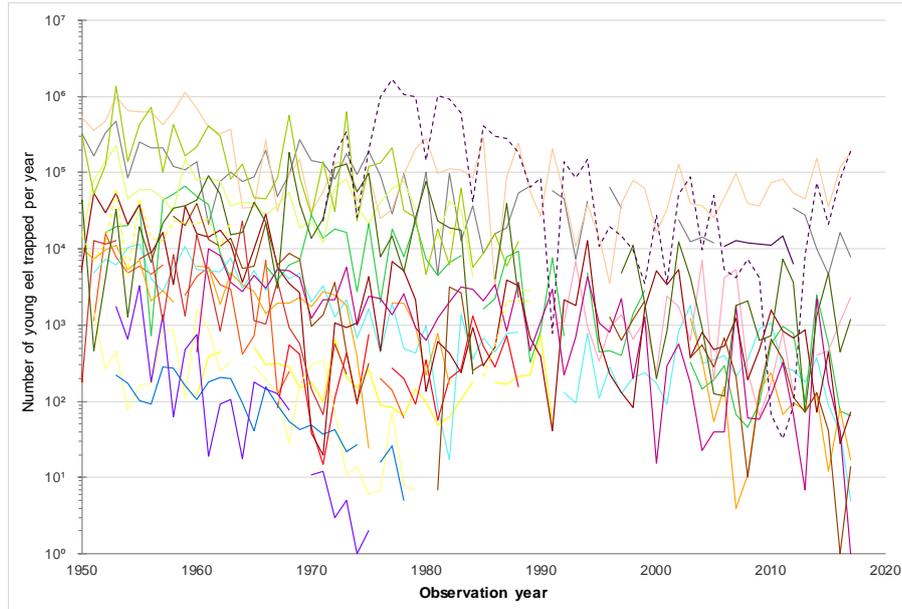


Figure 14 Observed trends in the catch of immigrating young eel, per site and year; when only catch weights were recorded, these have been converted to numbers. Incompletely covered seasons or otherwise invalid observations have been excluded. Sites continuing their sampling up to today are underlined; site names have been shortened to four characters. This figure presents the raw data ordered by the year the observations were made - not by the year class of the eel, as most other figures do.

The year-to-year variation has been considerable at all sites (Figure 13), 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.

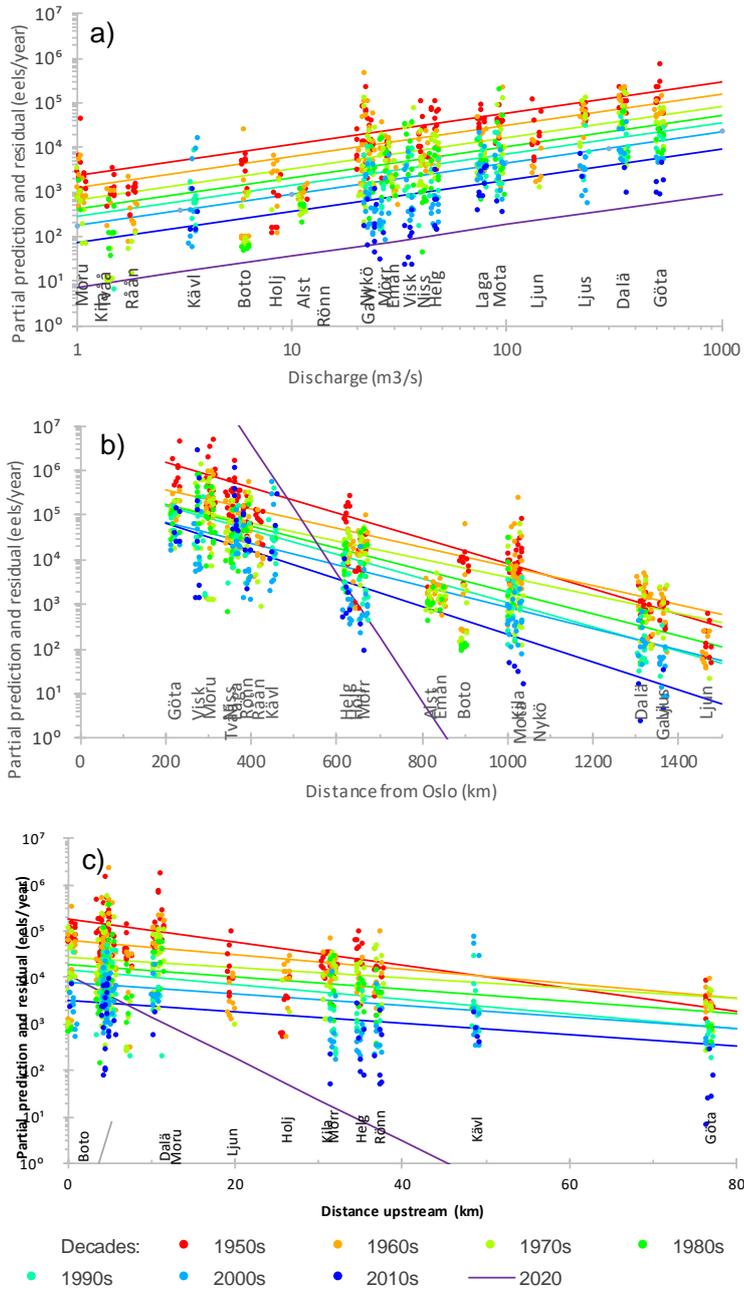


Figure 15 Partial predictions and partial residuals, by year class; for a) Discharge, b) Distance-from-Oslo, and c) Distance-upstream. Though partial residuals have been calculated for each individual year class, the colours in this plot apply to whole decades. Partial predictions (regression lines) 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  $\pm 5\%$  of the discharge, resp.  $\pm 10$  km from Oslo and  $\pm 0.5$  km upstream. The position of each sampling site has been indicated along the bottom; sites continuing their sampling up to today are underlined; site names have been shortened to four characters (see6).

The number of eels trapped per year is positively related with the discharge at the site of capture (Figure 14.a), but the relation is less than proportional; rather, the quantity is related to discharge<sup>0.688</sup>. Inspection of the partial residuals (Figure 15.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<sup>3</sup>/s, there is hardly any relation between river discharge and the number of eels, while for discharges above 10 m<sup>3</sup>/s, the relation is more close to proportionality. Our analysis did not test whether the relation to discharge changed over the decades.

For the site position in the Baltic, a steep reduction in eel abundance is observed with increasing distance-from-Oslo (Figure 14.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 14.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.

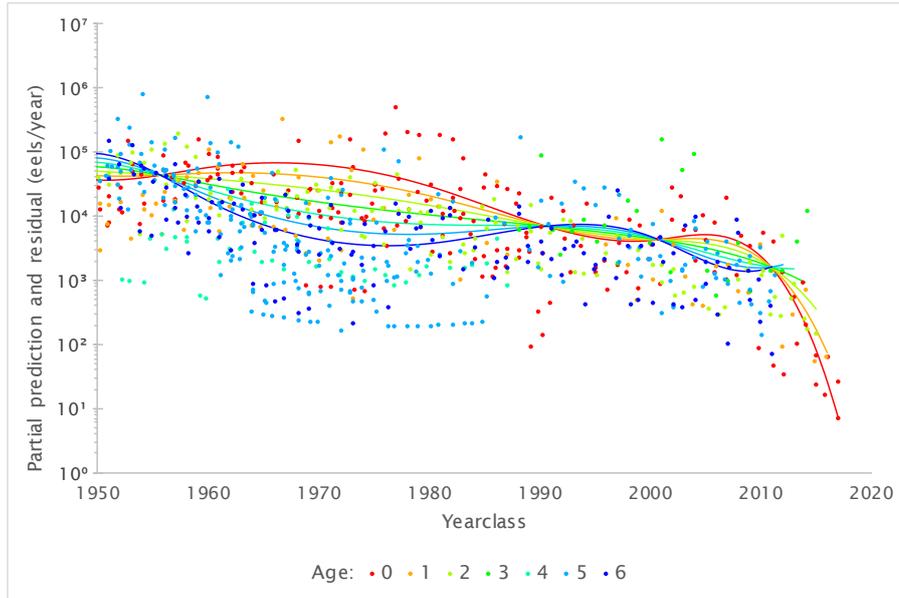


Figure 16 Partial predictions and partial residuals for year class, mean Age and their interaction. Unlike the other plots, the colour in this plot codes for the (rounded) mean Age at each site - not for decades. For clarity, all dots have been displaced horizontally by a horizontal random jitter of  $\pm 0.25$  years max.

The relation between eel abundance, mean Age in the catch and the year class is shown in Figure 15. 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 into the older ages yet. The regression model, fitting smooth functions, does not pick up that signal (see discussion below).

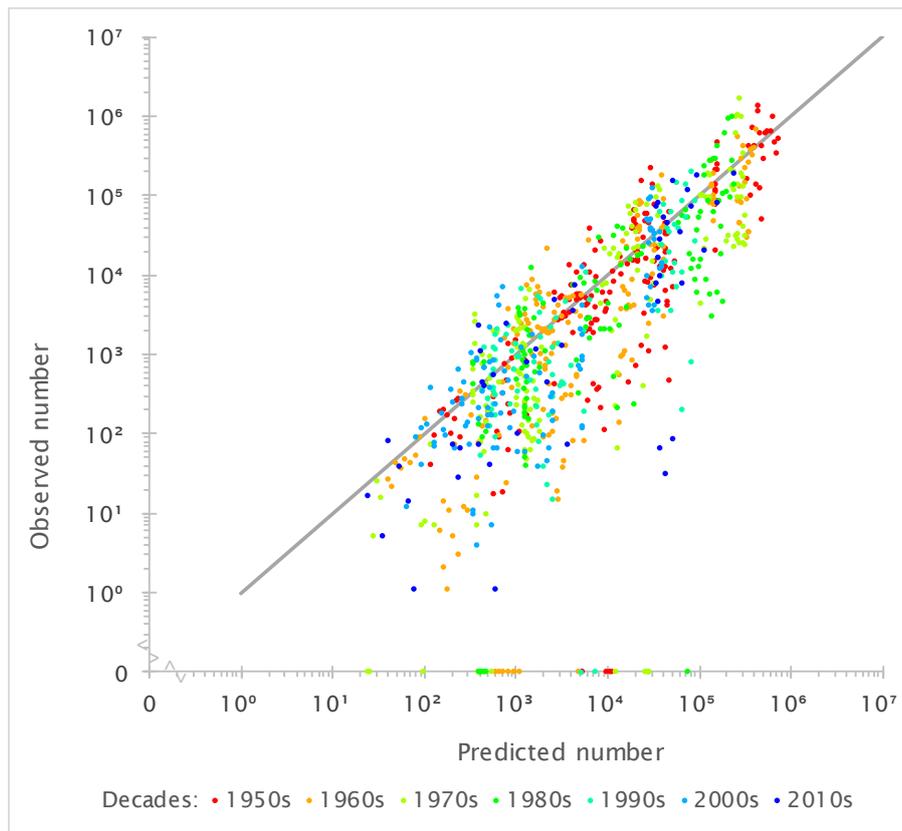
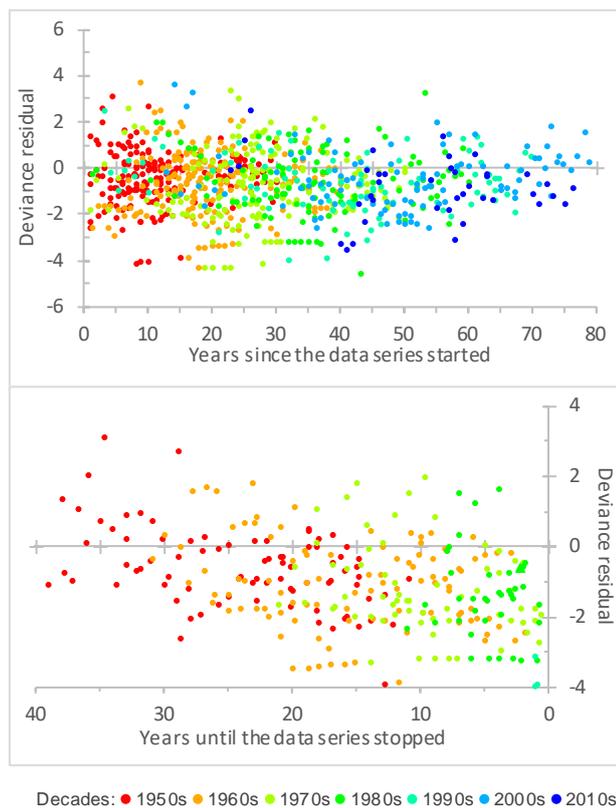


Figure 17 Relation between observed values and values predicted by the statistical model, coloured by decade. The solid line represents the main diagonal, where observed and predicted values are equal.

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 16 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 17.b). 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 17.a), or their further longevity (Figure 17.b).



*Figure 18* Partial residuals plotted as a function of a) the number of years since the data series was started, resp. b) 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 2015. For clarity, all dots have been displaced by a horizontal random jitter of  $\pm 0.25$  years max.

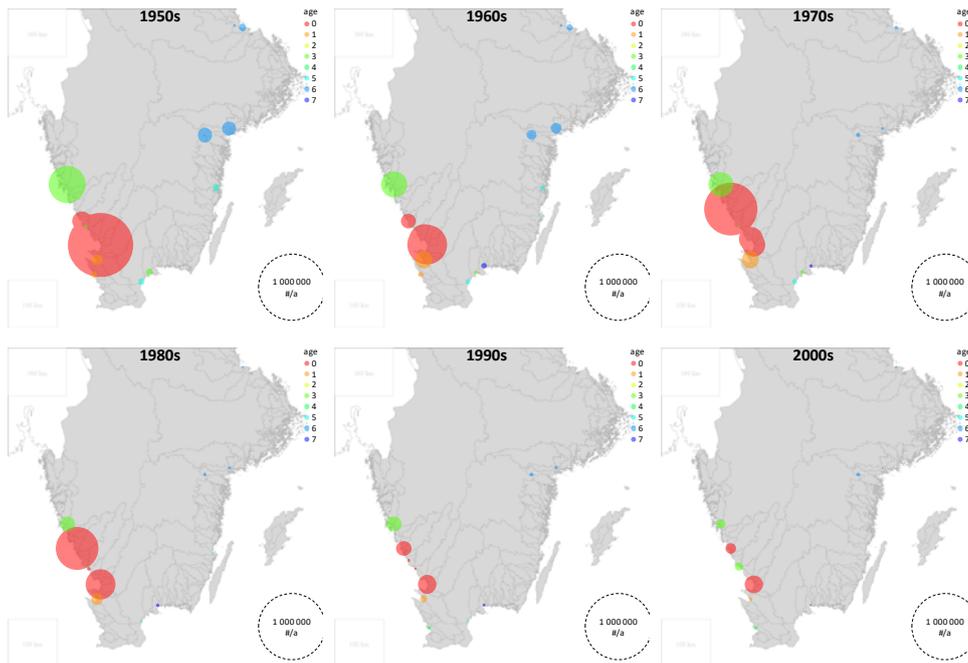


Figure 19 Spatial distribution of the observed numbers of elvers caught in the traps, averaged per decade, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class.

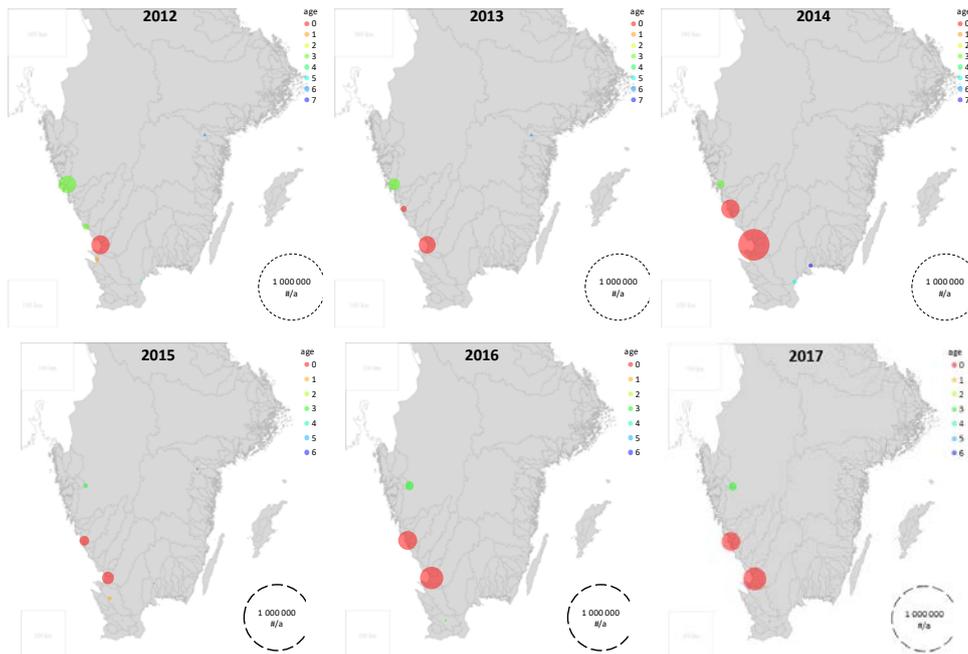


Figure 20 Spatial distribution of the observed numbers of elvers caught in the traps, in the years 2012-2017, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class. The numbers at many locations are that low, that the symbols become invisible in these maps.

### **B.3 Predicted trends in natural recruitment into inland waters**

The reconstruction of the inland silver eel production (Annex C) is based on estimates of the natural immigration of glass eels, elvers and bootlace eels into all rivers. To this end, the model of the spatial and temporal patterns in the elver trap catches, presented above, was used to generate statistical predictions for all rivers in all years. For the recruitment in rivers without a trap, in earlier years, plausible predictions were generated. For the predictions of the most recent years, however, aberrant predictions were obtained. For the very last year class, only very few observations are available (Viskan and Lagan); other elver trapping sites tend to catch incoming recruits of an older age, and these sites are therefore expected to catch the 2017 year class only in the time still coming. Because of the extremely low number of observations for the most recent year class 2017 (and some before), the model is relatively over-specified. Figure 14 and Figure 15 show the predicted regression line even for year class 2020, three years forward in time, beyond even the last observation. Clearly, those regression lines deviate considerably from all others; the predictions for Viskan and Lagan, however, add up 1. an extremely high abundance closer to Oslo (Figure 14.b), 2. an extremely high abundance close to the river mouth (Figure 14.c), and 3. an extremely weak year class in 2017 (Figure 15) – adding up to a realistic prediction of Viskan and Lagan in 2017. Obviously, the extreme relations for 2020 do not generate plausible predictions for sites further from Oslo, deeper into a river. Because of this, the model is lending itself badly for extrapolation to other rivers and years based on the very last year class. Since year class 2010 was the last one already recruited at all trapping sites, this year class was selected as the basis for extrapolation.

The references for this Annex are included in the reference list of the main report, on page 47.

## **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:

$$\text{glass eel equivalents}_{\text{year-age}} = \text{number}_{\text{year,age}} \times \exp^{+M \times \text{age}}$$

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 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), 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 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. 60 t. (see below), ignoring these smaller rivers introduces a bias of approximately 3 % of 60 t.  $\approx$  2 t. only.

For the rivers *with* an elver trap, natural recruitment is estimated by the statistical prediction, not by the actual observation – 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.

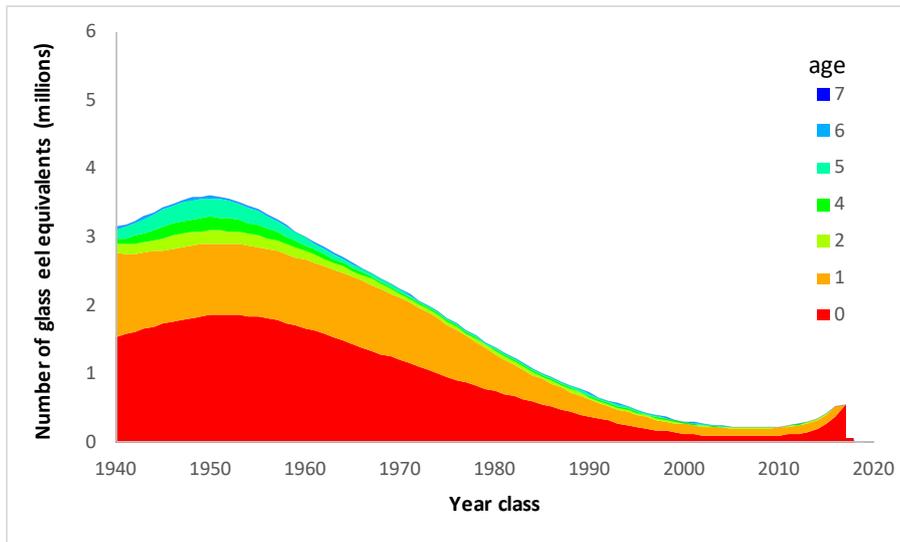


Figure 21 Time trend in the estimated number of naturally recruiting eels, expressed as glass eel equivalents per year class.

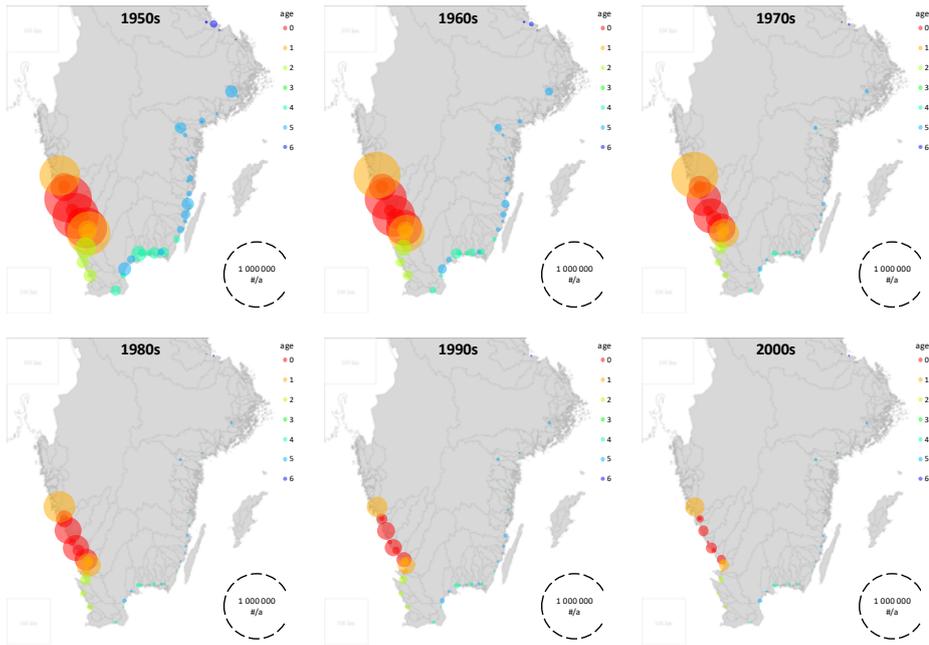


Figure 22 Spatial distribution of the estimates of natural recruitment, per decade, expressed in glass eel equivalents. These plots show the total number per decade (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the immigration took place, not by year class.

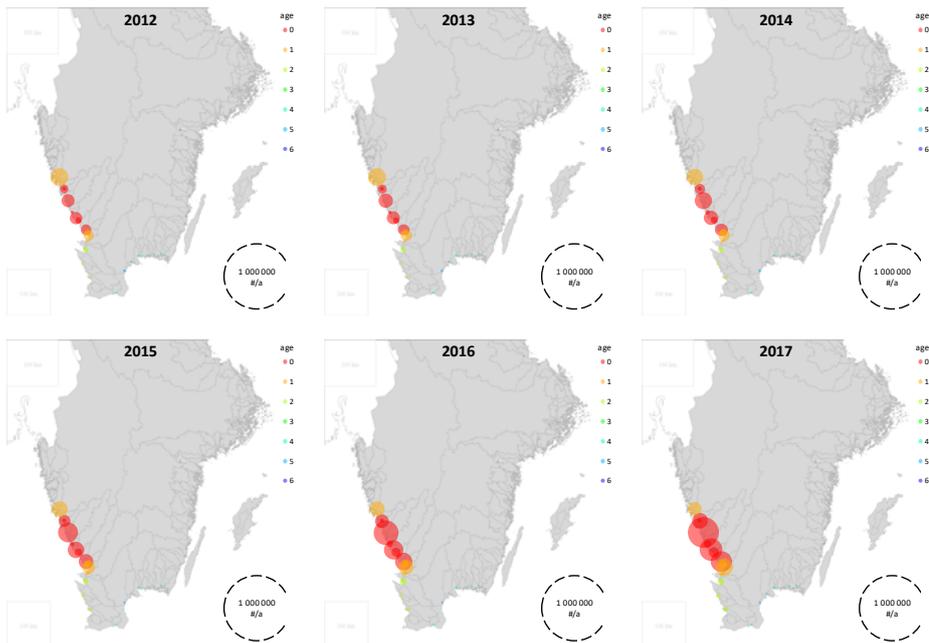


Figure 23 Spatial distribution of the estimates of natural recruitment, in the years 2012-2017, expressed in glass eel equivalents. These plots show the total number per year (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are 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. 92).

Trapping of young eels was 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.

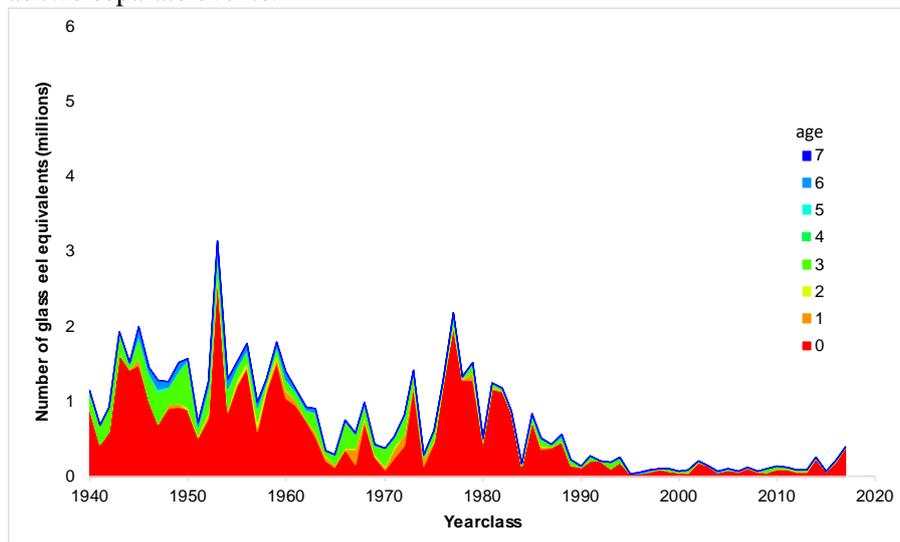


Figure 24 Time trend in the number of eels released from assisted migration. Though this plot is subdivided by age of the eel, all quantities are expressed in glass eel equivalents per year class.

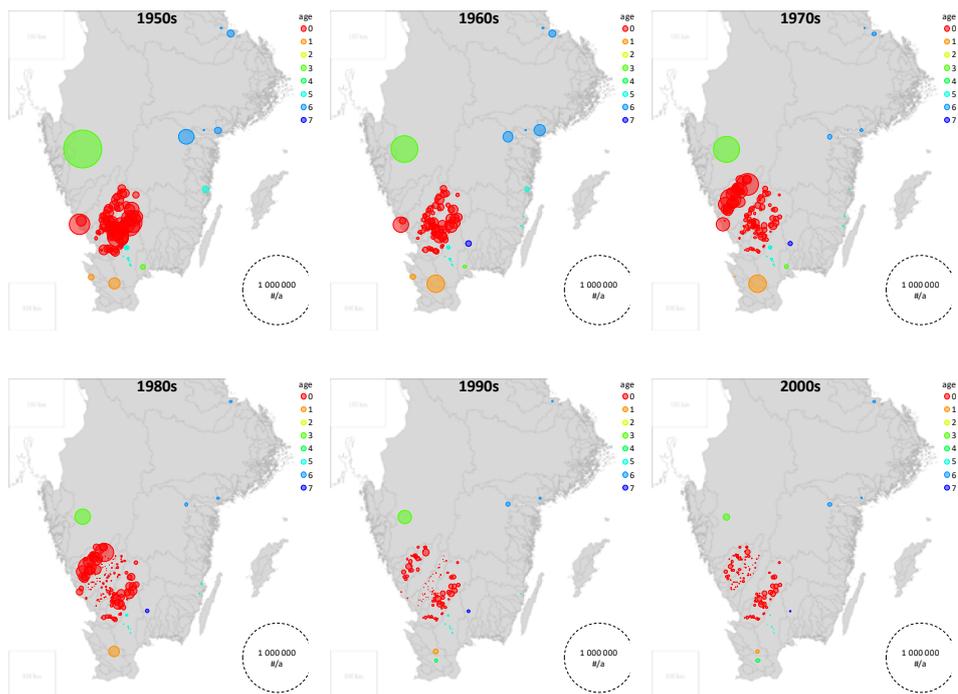


Figure 25 Spatial distribution of the release from assisted migration, per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that the figures are sorted by the year in which the release took place, not by year class.

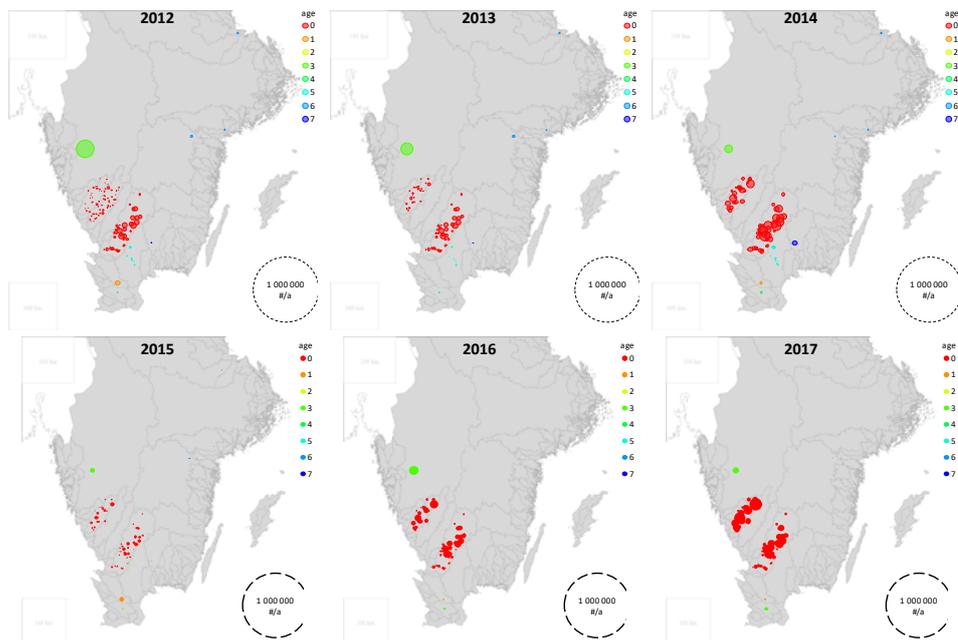


Figure 26 Spatial distribution of the release from assisted migration, in the years 2012-2017, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the release took place, not by year class.

### 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), 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 have been restocked.

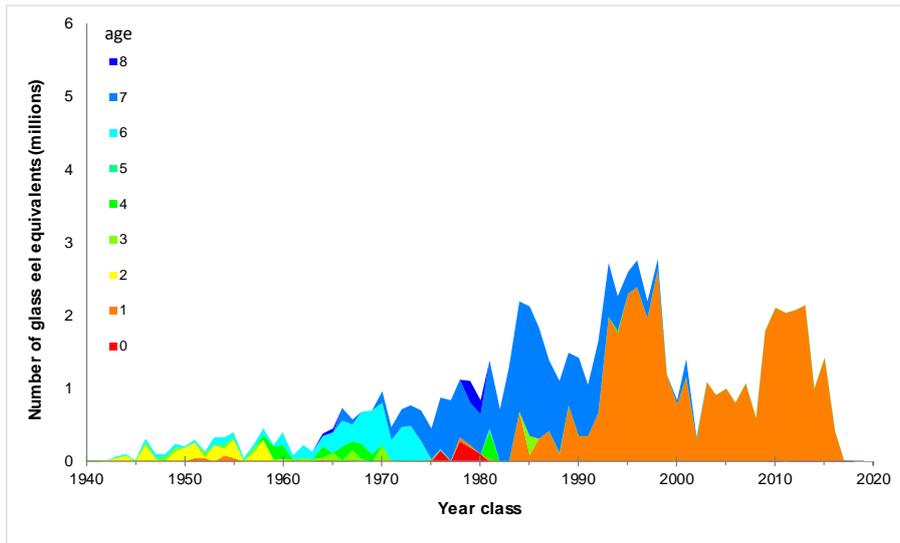


Figure 27 Time trend in the numbers of eel used for restocking. Though this plot is subdivided by age of the restocking material, all quantities are expressed in glass eel equivalents per year class.

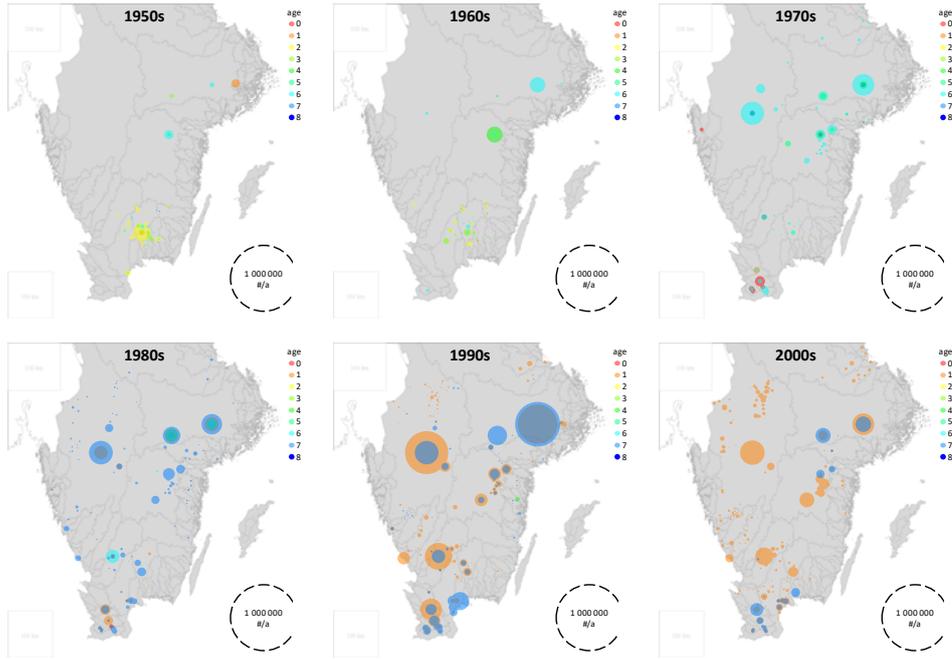


Figure 28 Spatial distribution of the restocking per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that these figures are sorted by the year in which the restocking actually took place, not by year class.

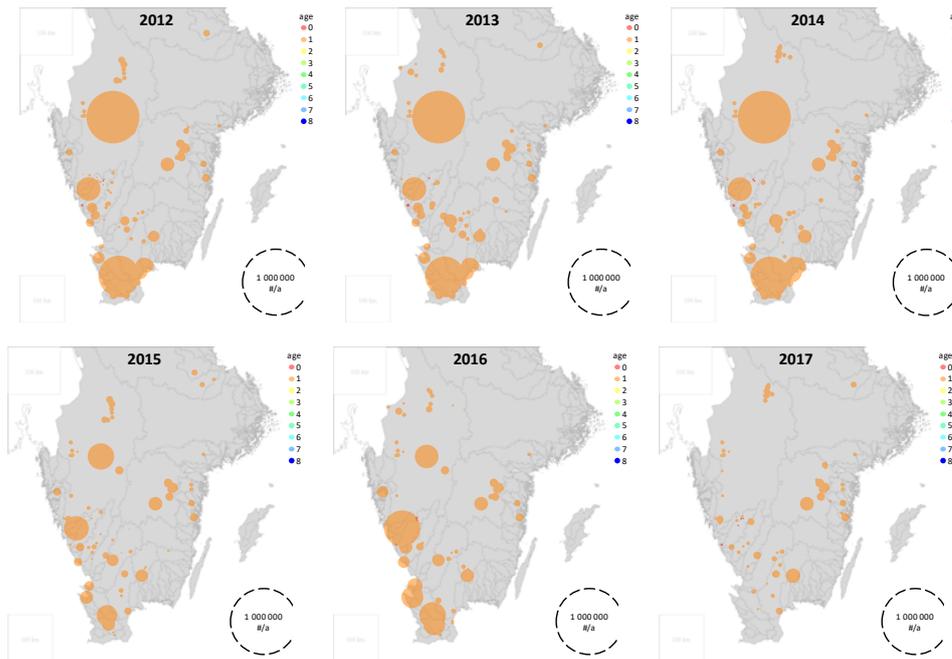


Figure 29 Spatial distribution of the restocking in the years 2012-2017, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the restocking took place, not by year class.

*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 7 Quantities of silver eel in the *Trap & Transport* programmes, in numbers (N) and biomass (kg)

River	Year		2010		2011		2012		2013		2014		2015		2016		2017	
	N	kg	N	kg	N	kg	N	kg	N	kg	N	kg	N	kg	N	kg	N	kg
Motala Ström			546	676	930	1283	2531	3167	4746	5931	3534	4821	3749	5141	3630	4894		
Mörrumsån			1616	1883	135	154	212	269	286	329	816	938	284	327	298	343		
Kävlingeån			685	548	214	167	439	325	1057	909	301	241	604	544	449	445		
Rönne Å										415		250		316		541		
Lagan	423	365	653	367	72	110	932	921	1447	1484	705	681	885	866	1128	1111		
Nissan											95	83	109	96	326	334		
Ätran					369	295	120	96	365	292	163	130	17	14	321	257		
Göta Älv	4590	4841	4250	4499	7803	8237	9039	9393	12355	12417	11669	11890	11277	11743	10508	10448		

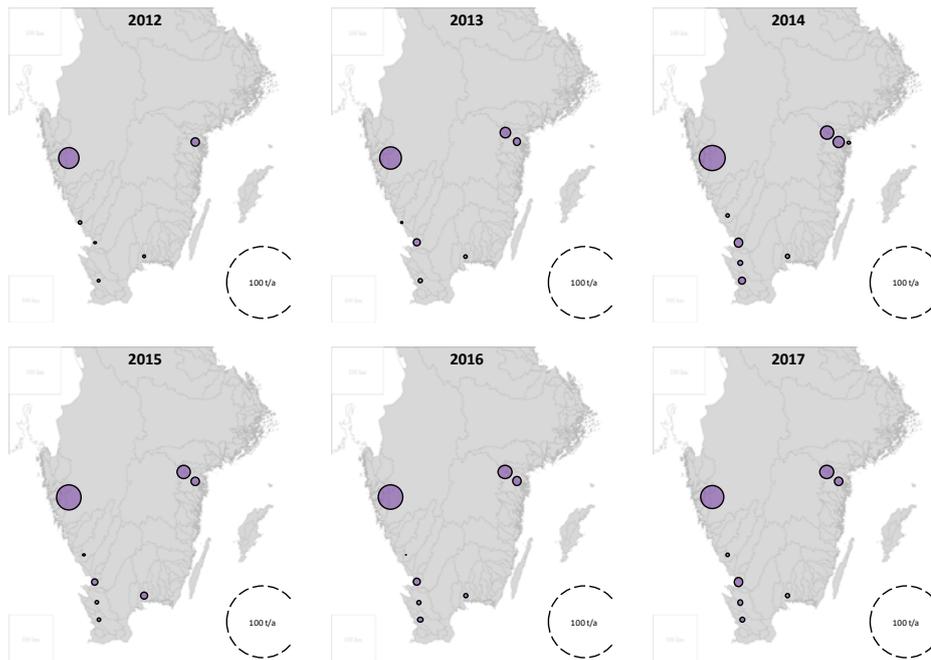


Figure 30 Spatial distribution of the releases from the Trap & Transport programmes, in the years 2012-2017.

### 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 several times. The Swedish Fishery Board (Fiskeriverket, now Havs- och Vattenmyndighet) and the Swedish Statistics Bureau 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 30). Elsewhere, data are available per lake and/or for varying groups of lakes (Figure 31). 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).

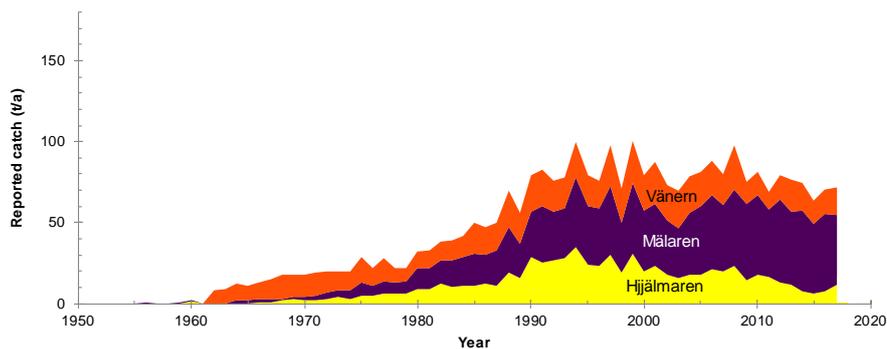


Figure 31 Time trend in the reported landings from the fishery, for the larger lakes, and years since 1950. For smaller lakes, data are only available since 1986.

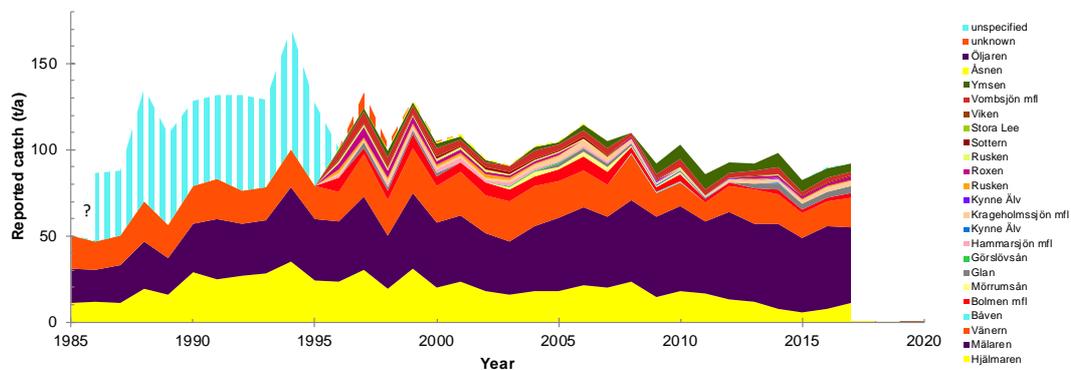


Figure 32 Time trend in the reported landings from the fishery, for all lakes, and years since 1985. Note the time interval on the horizontal axis, deviating from most other figures.

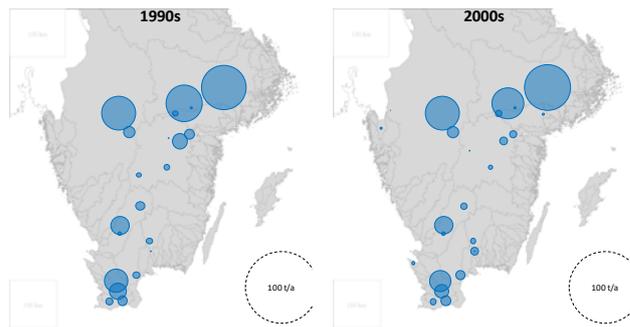


Figure 33 Spatial distribution of the reported landings from fisheries, in the 1990s and 2000s. For earlier decades, insufficient information is available.

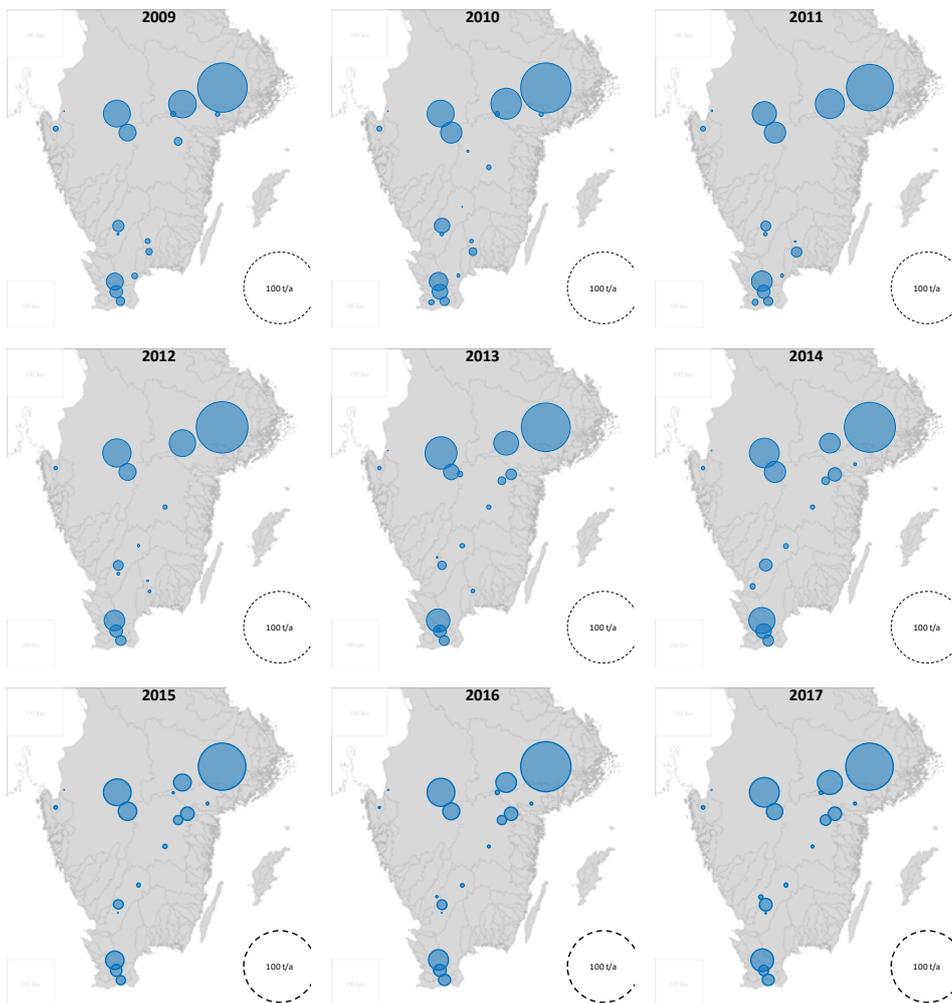


Figure 34 Spatial distribution of the reported landings from the fisheries, for the years since 2009.

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 33), 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.

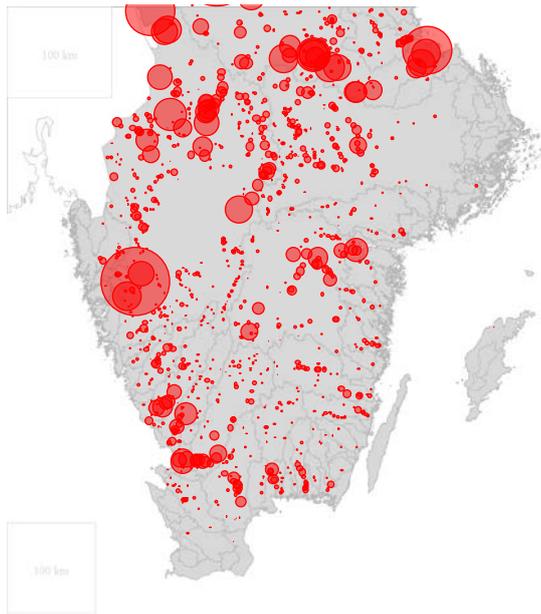
Until 1998, information was collected by regional fisheries officers (fiskerikonsulenter, länsstyrelsen) in direct contact to individual fishers, most often on an annual basis. Since 1999, this was replaced by a system of obligatory reporting by individual fishers directly to the Swedish Board of Fisheries, 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.

Recently, an effort has been made to disclose information on landings in historical archives, with a focus on the years 1960-1995. Since that information has not been fully processed yet, the current assessment is still based on the official, less-detailed statistics for that period.

### *Impact of hydropower generation*

#### Location of hydropower stations

A database of hydropower generation plants was made available by Kuhlin (2018), documenting location and year of construction (Figure 34). 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 1505 hydropower stations listed by Kuhlin (2018), 519 stations are relevant for the current reconstruction (eel occurring upstream).



*Figure 35* 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 35, 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 is: 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  $H$  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.

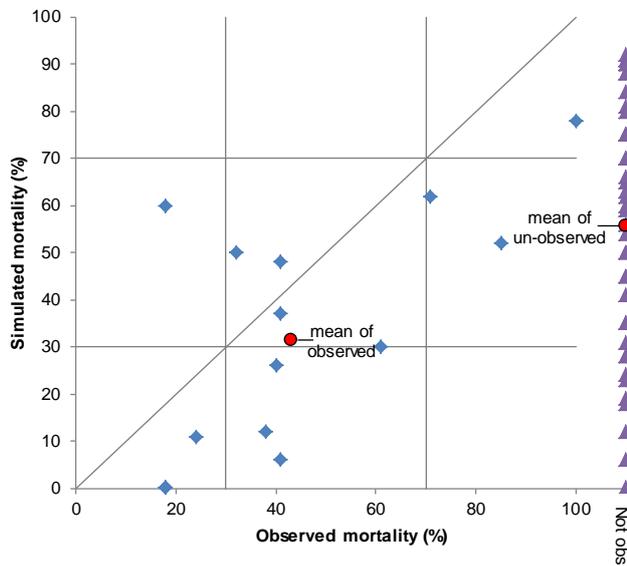


Figure 36 Relation between the observed (horizontal) and simulated (vertical) mortality, for eel passing a hydropower turbine. Data from Calles and Christianson (2012), applying the simulation model of Leonardsson (2012); original plot of data tabulated by the source.

### 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

From 2010 to 2017, 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. In total, a number of 2 850 eels have been analysed. 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  $b=3.297428$ . This differs from the parameter values used in Dekker (2015), which overestimated the weight of the very youngest eel (glass eel), and under-

estimated the weight of the larger eels. As a consequence, the 2015-assessment incorrectly assigned the quarantined eel used for restocking to age group 0; the current parameter settings assign them more correctly to age group 1. We note that this shifts all restocked eels one year class back in time, which affects all tables and graphs referring to the year classes. For eel of about the size of a silver eel, this brings the individual weight up by about 20 %, which affects all biomass estimates based on numbers of youngsters, but not the recorded fishing yield; correspondingly, the fishing mortality is estimated to be somewhat lower.

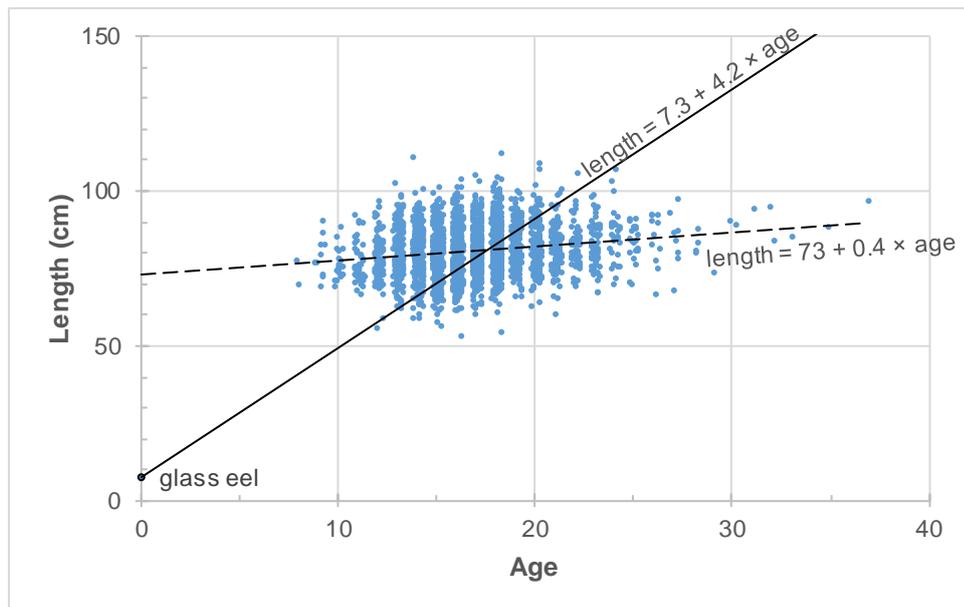


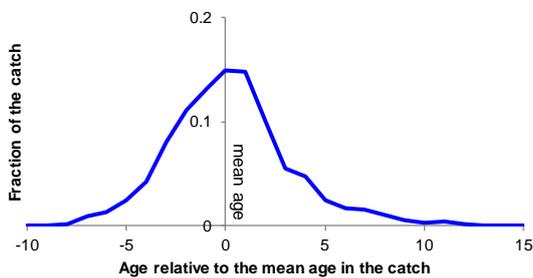
Figure 37 Length and age for 2 850 silver eels, sampled between 2010 and 2017 in 6 lakes. To show so many data points, a small jitter has been added to all data points in horizontal direction. Two regression lines are given: a growth-line (drawn) forced through the length/age of glass eel (7.3 cm cm 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 36, dotted line) is much less than the mean growth rate during the yellow eel stage of 4.2 cm/year (Figure 36, 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 38* 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, 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,  $M=0.10$ .

#### 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 27 on restocking; Figure 31 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 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 recruited yet, 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 actions. Not knowing those future trends and actions, 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, 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).

For two factors, however, the extrapolation deviated from this general principle of a status quo extension from the last observation year onward. Neither of these two deviating factors affects the estimates of biomasses and mortalities until 2017; only the predictions into the future are affected. For the restocking, 2017 was an exceptional year: in spring 2017, an outbreak of the virus EVEX occurred in a major quarantine facility, and all infected glass eel had to be destroyed. As a consequence, the restocking programme

2017 was much lower (Figure 20), and deviated in spatial distribution (Figure 28), from the years before. Since it is unlikely that further virus outbreaks will occur in the near future, we based our extrapolation on the preceding year 2016 (i.e. restocking of year class 2015), which was a moderate year, in line with the years shortly before. For the natural recruitment, the last year class is 2017, but that year class has been observed at very few places yet (Viskan and Lagan); other elver trapping sites tend to catch incoming recruits at an older age, and these sites are therefore expected to catch the 2017 year class only in the coming years. Because of the extremely low number of observations for the most recent year class 2017 (and some before), the model is relatively over-specified, lending itself badly for extrapolation to other rivers and years based on the very last year class. year class 2010 was the last one already recruited at all trapping sites, and therefore, this year class was selected as the basis for extrapolation.

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 2017, 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 2017, natural recruitment – including the amount assisted in their migration upstream - is estimated at a total number of 62 million glass eel/glass eel equivalents, with a minimum of 0.2 million eels in 2007 and a maximum of 3.3 million in 1950. The corresponding silver eel production is estimated at 18 497 t, minimum 46 t/a, maximum 556 t/a. In 2010, 0.2 million glass eel equivalents were natural recruits. Total silver eel production from natural recruits (assisted or not) in 2017 is estimated at 46 t.

From 1960 until 2017, a total of 29 million eels have been caught for assisted migration upstream, with a minimum of 0.035 million of year class 1995 and a maximum of 2.2 million of year class 1977. The corresponding silver eel production is estimated at 8 888 t, minimum 19 t/a in 2017, maximum 295 t/a in 1969. In 2017, 0.4 million glass eel equivalents were

assisted upstream. Total silver eel production from the 2017 assisted migration is estimated below 20 t.

From 1960 until 2017, a total number of 67 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 9 538 t, minimum 15 t/a in 1960, maximum 404 t/a in 2012. Of year class 2017, 0.4 million glass eel equivalents have been restocked (mean since 2010: 1.6 million). The corresponding silver eel production (before fishery and hydropower impacts) is estimated at approximately 260 t.

Overall silver eel production declined from 500-600 t in the 1960s and 1970s, to about 400 t/a since 2010. Natural recruits, freely immigrating or assisted upstream, have been gradually replaced by (imported) restocking and the natural recruits now make up only 5-10 % of the total production in inland waters. Peak restocking in the 1990s brought recent production to a temporary maximum of 480 t/a in 2010; lower restocking in the early 2000s will reduce production to 240 t/a by 2020, and thereafter production will return to about 300 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.

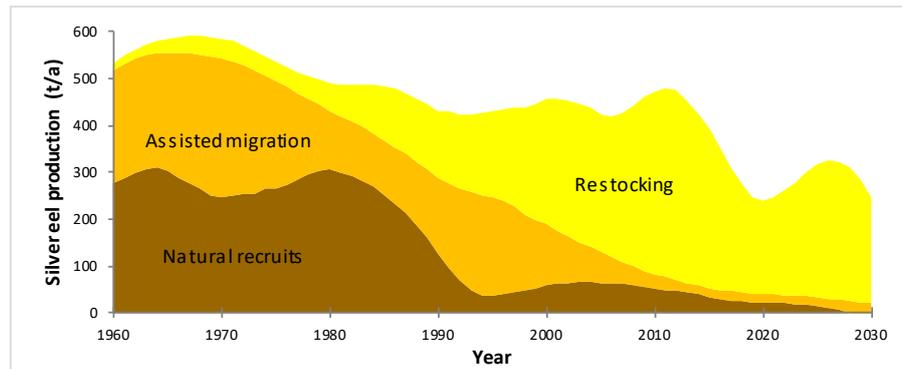


Figure 39 Production of silver eel by year and by origin of the eel, that is: the estimated total production before the impact of fishery and hydropower. For these results, a natural mortality rate of  $M=0.10$  was assumed.

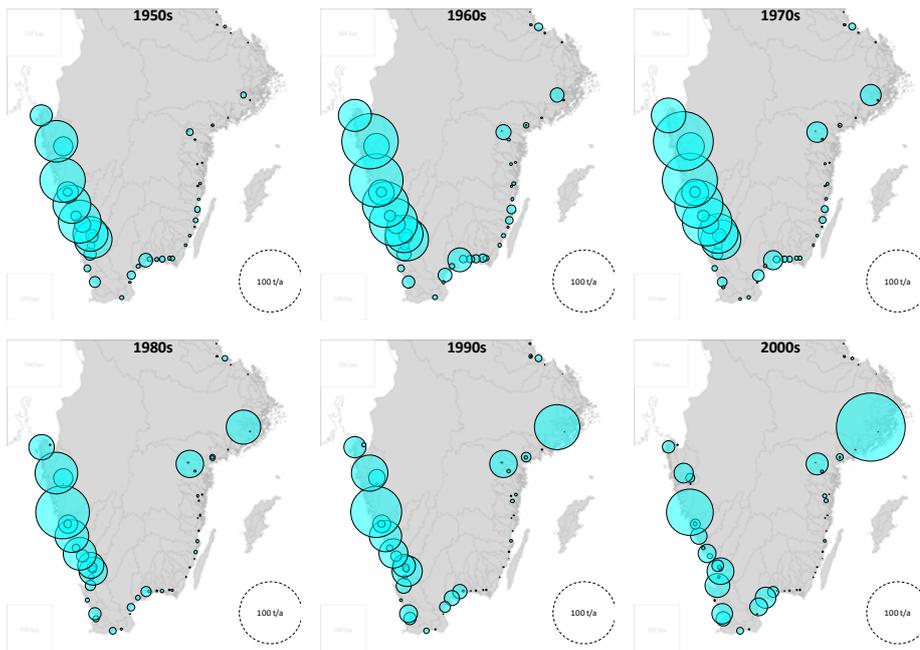


Figure 40 Spatial distribution of the predicted production of silver eel (before fishery and hydropower impacts), per decade and per river drainage system. The production for each river drainage area is plotted at the place of the river mouth, while in reality, the production will have taken place all over the drainage area.

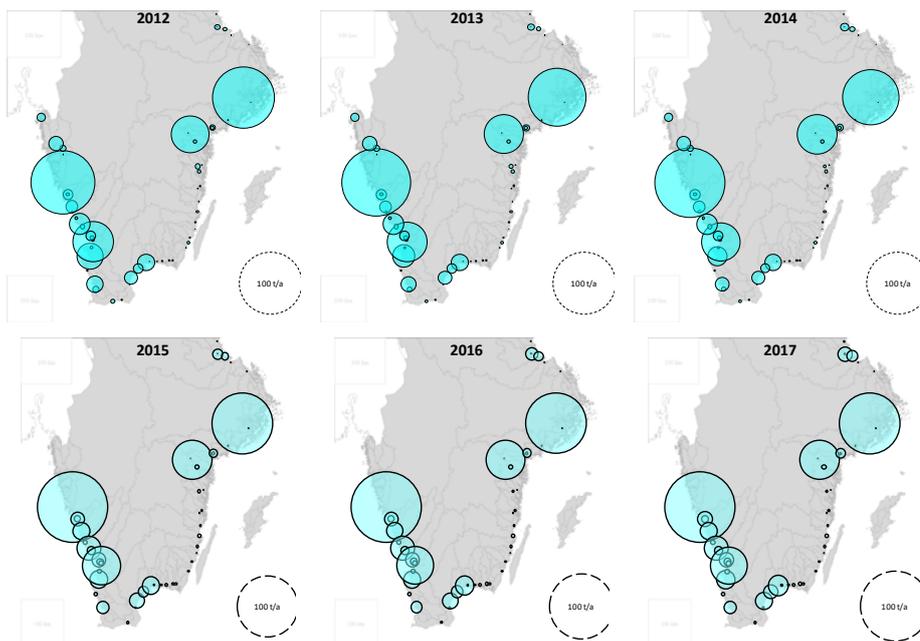


Figure 41 Spatial distribution of the estimated production of silver eel (before fishery and hydropower impacts), per year since 2012 and per river drainage system. The whole production estimated for each river drainage area is plotted at the place of the river mouth.

### C.2.2 Silver eel destination

Figure 41 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 41 matches the predicted total production, presented in Figure 38.

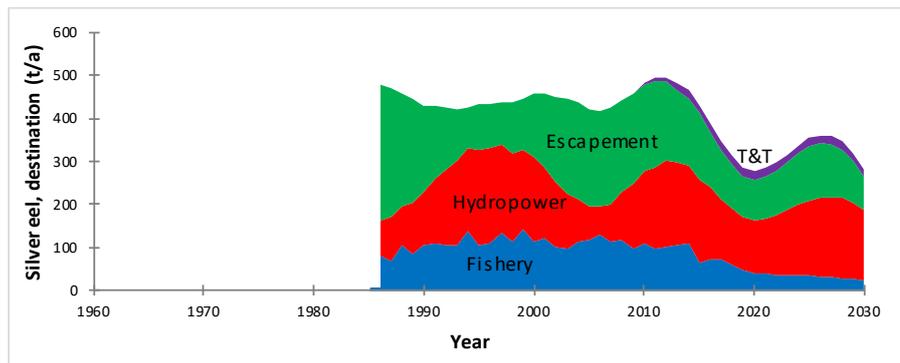


Figure 42 Time trends in the destination of the silver eel produced in inland waters. Data before 1986 are incomplete.

For the fishery, the landings have varied between 64 t (in 2015) and 133 t (in 1997). This is on average 25 % of the production; with unaltered management, the impact is expected to decline to ca. 10 % (Figure 44). The catch in 2017 was 72 t.

For the hydropower, the estimated impact varied between 70 t (in 2006) and 223 t (in 1995), that is approximately 35 % of the total production (range 20 % - 50 %). The estimated impact in 2017 was 141 t. 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 rise to 60 %.

Reconstructed escapement of silver eel ranged from 94 t (in 1994) to 316 t (in 1986), on average 40 % of the total production (range 22 % - 66 %). The 2017 escapement is estimated at 115 t.

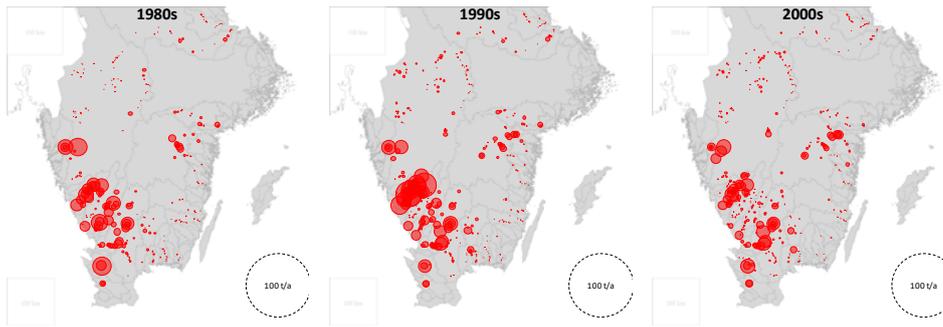


Figure 43 Spatial distribution of the estimated impact of hydropower, per hydropower station per decade. For the 1980s, estimates are based on the years from 1986 onwards; for the earlier years, no estimates could be derived because of the absence of information on the landings from fisheries.

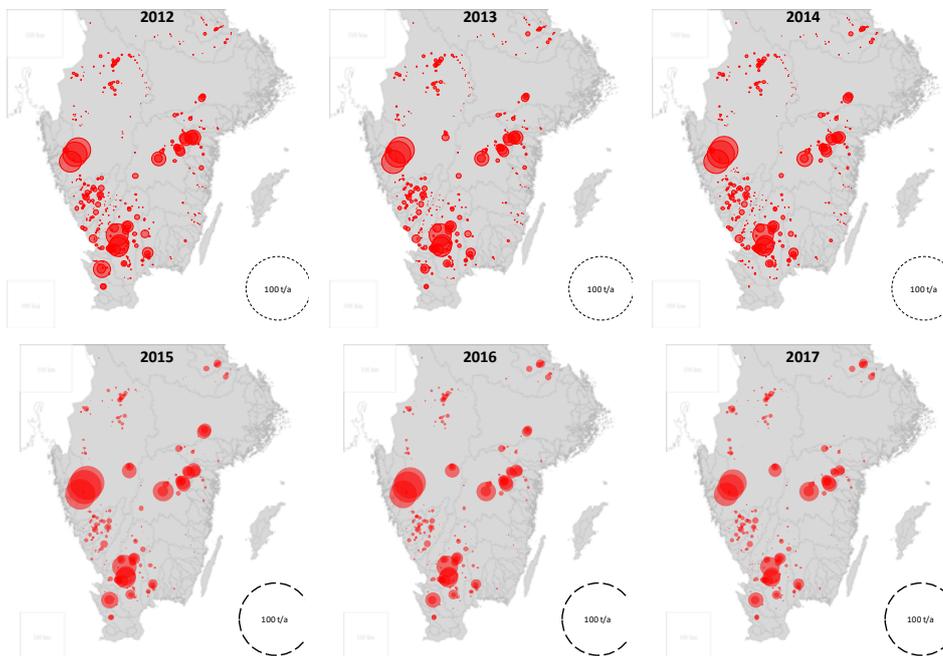
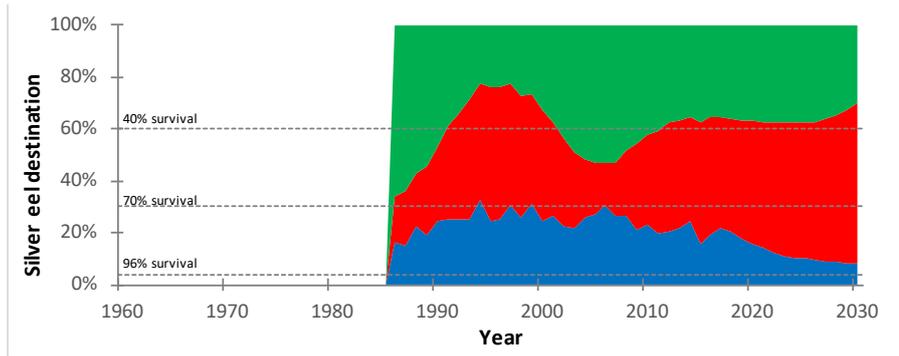


Figure 44 Spatial distribution of the estimated impact of hydropower, per hydropower station per year, since 2012.



*Figure 45* 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 limit mortality for a healthy stock ( $B_{\text{current}} > 40\% * B_0$ ). The reference line “70 % survival” applies in the current, depleted state, accounting for restocking. The reference line “96 % survival” applies in the current, depleted state, *not* accounting for restocking.

Expressing anthropogenic impacts in terms of mortality rates (Figure 45), 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 45 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  $\Sigma A$  of fishing mortality and hydropower related mortality represents the actual mortality exerted on any part of the stock, whether natural or restocked.

Taking the effects of restocking into account, the total estimate has ranged from +0.98 (in 1994) to -1.05 (in 2015); the 2017 value is estimated at -0.91. 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 0.41 (in 1986) to 1.50 (in 1994); the 2017 mortality is estimated at 1.05. These estimates express the mortality exerted on the natural recruits, as well as on the restocked eels.

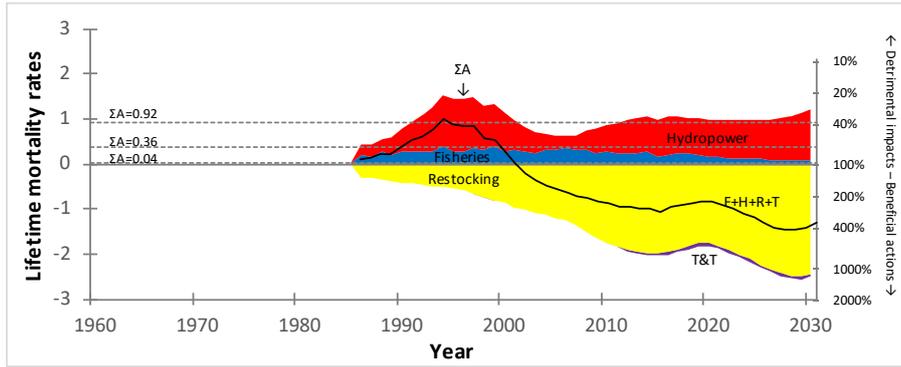
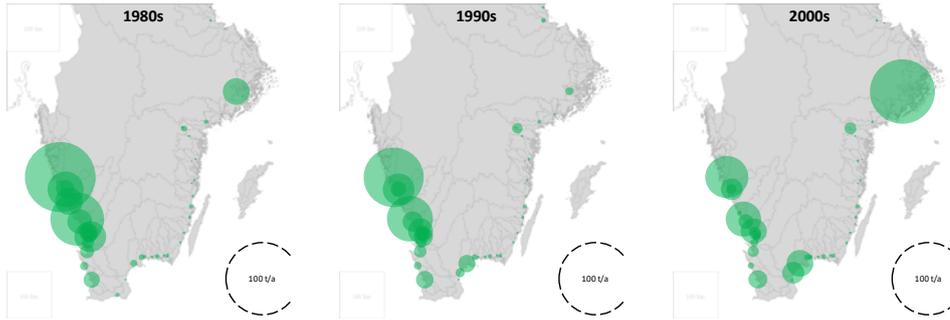


Figure 46 Time trend in the estimated anthropogenic mortalities: fisheries, hydropower, restocking and Trap & Transport (T&T). The mortality exerted by Restocking and Trap & Transport are negative; that is: these actions increase the amount of silver eel escaping. The line marked "F+H+R+T" represents the sum of all anthropogenic actions, including Restocking and Trap & Transport;  $\Sigma A$  represents the mortality exerted on the stock, whether natural or restocked.

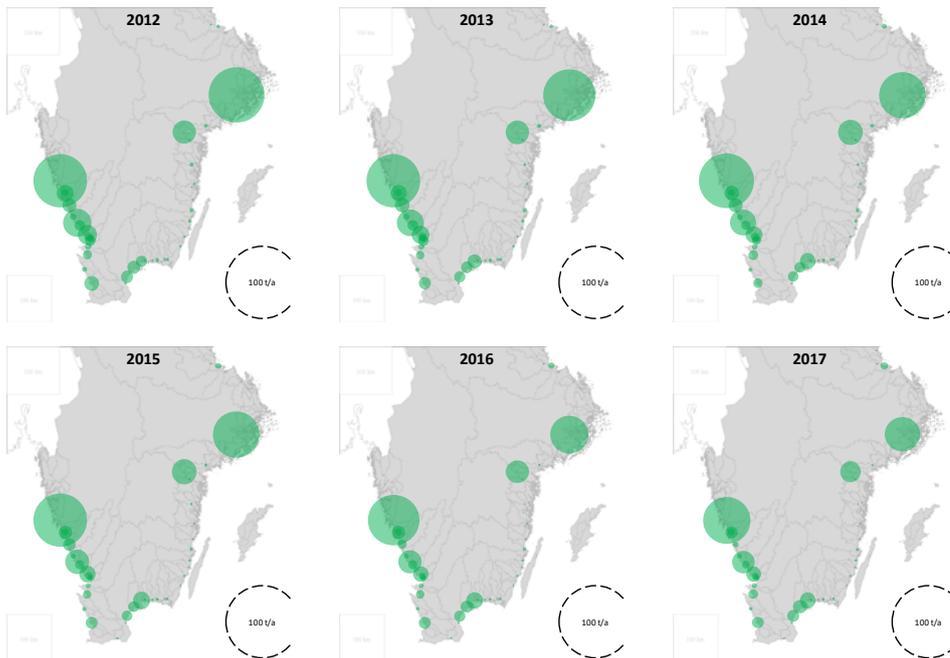
Fishing and hydropower-related mortality have their impact on the silver eel stage; hence, the horizontal axis represents the year the mortality occurred, i.e. the silvering year. For the interpretation of restocking as a negative mortality, however, the year the restocking was done precedes the silvering year by a lifetime; for these too, the results refer to the silvering year.

The reference line  $\Sigma A=0.92$  represents the limit mortality for a healthy stock ( $B_{\text{current}} > 40\% * B_0$ ). The reference level for mortality is related to the actual status of the stock. Hence, different levels apply, whether one takes into account or not the presence of restocked eels; that choice affects the view on the current status.

The reference line  $\Sigma A=0.36$  applies in the current, depleted state, taking into account restocking. The reference line  $\Sigma A=0.04$  applies in the current, depleted state, *not* taking into account restocking. A mortality of  $\Sigma A=0.11$  conforms to the 90% survival, the management limit of the Swedish Eel Management Plan.



*Figure 47* 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 years, no estimates could be derived because of the absence of information on the landings from fisheries.



*Figure 48* Spatial distribution of the estimated escapement of silver eel per year, since 2012.

### 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 48 compares results, for two selected years: 1995 and 2017, 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 48.a&b) range from just over 200 t/a to around 900 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 48.c&d), results are quite different. For  $M=0.05$ , production is estimated at c. 900 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 almost 800 t (for  $M=0.05$ ) to far less than 100 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 the estimates of anthropogenic mortality (Figure 48.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  $\Sigma A$  reflects the sensitivity of  $F$  to  $M$ . The cumulative effect of fisheries and hydropower (1.16 – 1.17 in 1995; 0.84 – 1.51 in 2017) exceeds the minimal mortality limit ( $\Sigma A=0.92$  for a healthy stock,  $\Sigma A=0.36$  for the currently depleted stock with, and  $\Sigma A=0.04$  without restocking). The restocking did not compensate for these mortalities in 1995, but does more than so in 2017, for all values of  $M$  tested. Though the estimate of  $\Sigma A$  is sensitive to the assumption on  $M$ , the evaluation remains that anthropogenic mortality exceeds the limit for the current, depleted stock.

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.

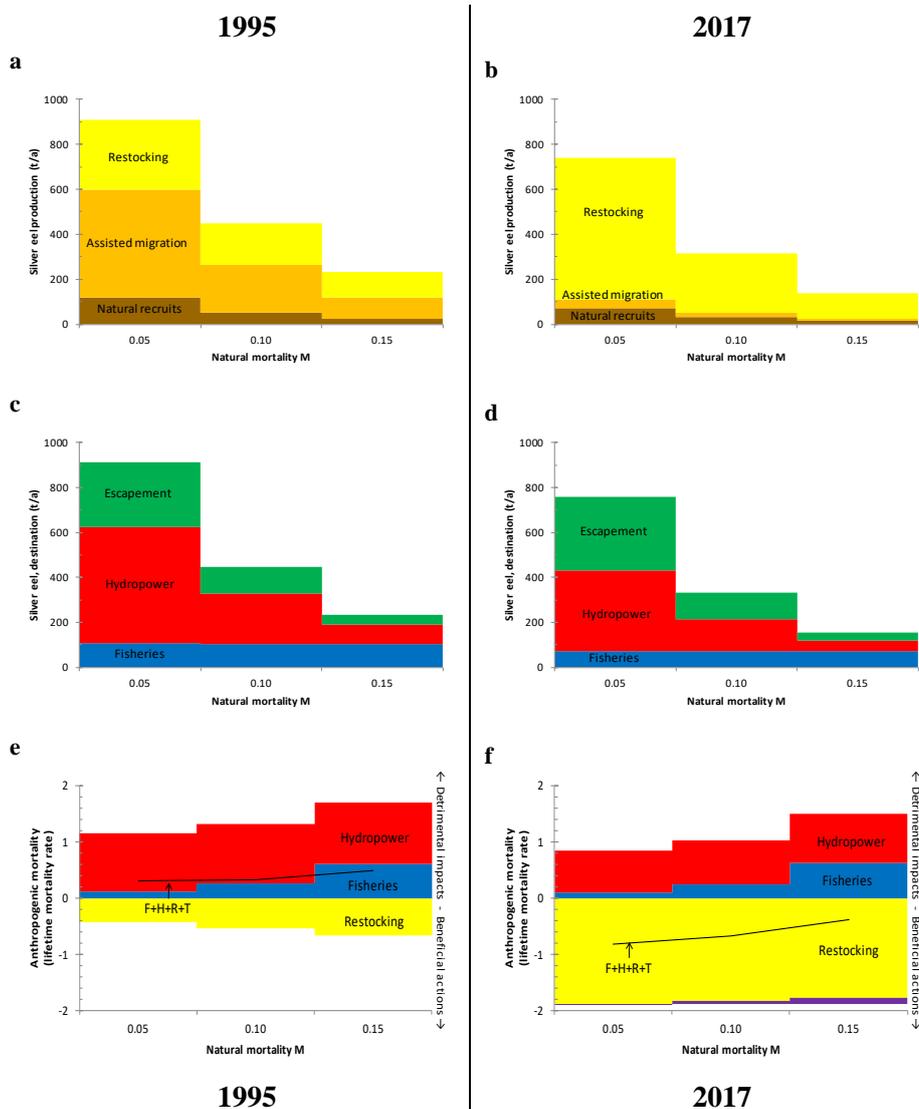


Figure 49 Comparison of results for 3 different values of natural mortality, showing results for 1995 (left) and 2017 (right). Within each sub-plot, the columns show results for the three options  $M=0.05$ ,  $M=0.10$  and  $M=0.15$ , respectively; comparisons are to be made within each subplot, between the columns.

Top row: predicted silver eel production (compare Figure 38);

Middle row: predicted silver eel destination (compare Figure 41);

Bottom row: anthropogenic mortality rates (compare Figure 45).

### *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 38 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 47.

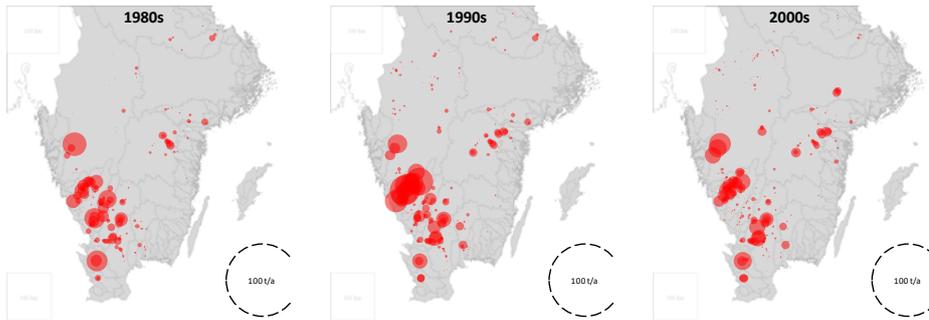


Figure 50 Spatial distribution of the estimated impact of hydropower, per hydropower station per decade. For the 1980s, estimates are based on the years from 1986 onwards; for the earlier years, no estimates could be derived because of the absence of information on the landings from fisheries.

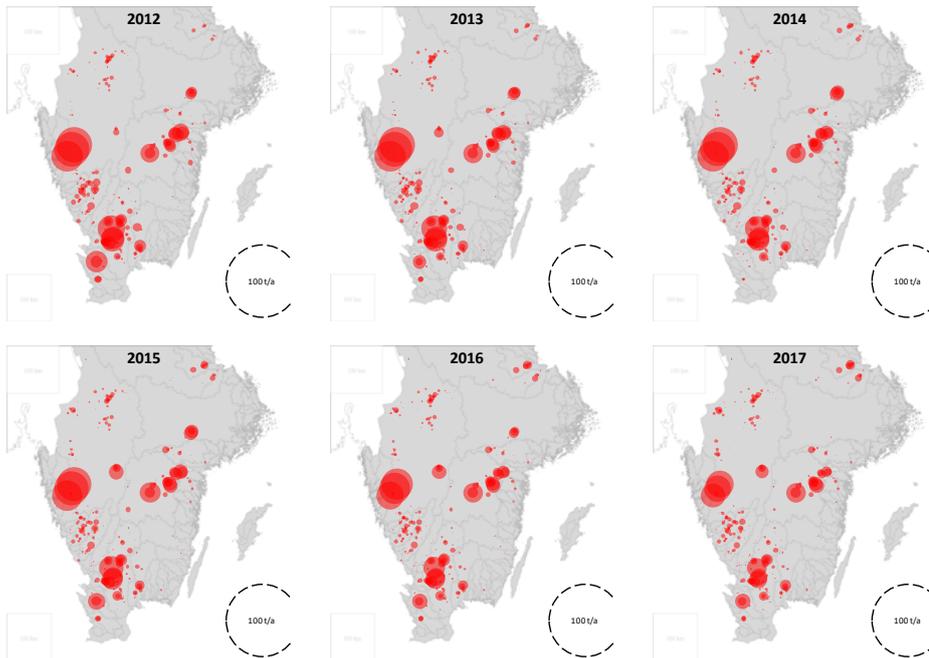


Figure 51 Spatial distribution of the estimated impact of hydropower, per hydropower station per year, since 2012.

## Annex D Impact of the Baltic Coast fishery

Dekker and Sjöberg (2013) analysed the impact of the silver eel fisheries on the Baltic Coast over the past 60 years, 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 assessment (Dekker 2015) updated that analysis, adding the data from the re-continued tagging programme. This Annex now presents a second update, including data up to and including 2017 (Figure 51). No changes in the methodology of Dekker & Sjöberg (2013) have been made.

From 2015 through to 2017, six additional experiments have been conducted (Figure 52), tagging 989 silver eels in total, of which 65 have been recaptured today.

Estimates of the hazard and survival curves are given in Figure 53 and Figure 54. Compared to previous decades, the hazard of being recaptured in the fishery has declined considerably. This is in line with the trend in landings data (Figure 5), declining from 354 t in 2011 to 143 t in 2017.

Figure 57 presents the results of the population estimate by county (län), for the 2010s in particular. This reconstruction uses the estimate of the fishing mortality, that is the hazard (Figure 55) from Survival Analysis (Figure 54), and combines that with the landings (Figure 5) split by county (Figure 56), to derive an estimate of the population size (Figure 57). For most counties, population estimates are in the order of 1500-2000 t, with the exception of Blekinge (4108 t). For Södermanland, a catch of only 1446 kg was recorded; the population is estimated at less than 1000 t only.

Over all counties with a catch > 10 t, the average hazard has declined from over 50 % in the 1950s, to  $\pm 10$  % in the 2000s, and 2.0 % in the 2010s. Over all counties with a catch > 100 t, the 2010s estimate comes at 2.5 %. The decline in hazard from the 2000s to the 2010s is somewhat larger than in previous decades, possibly reflecting the effect of fishing restrictions

implemented in recent years. The ratio of catches to the estimate of fishing mortality (a proxy for the catch per unit of effort), however, has changed dramatically – varying between 2000 t and 4000 t per unit of mortality over the 1950s to 2000s, it jumped to nearly 10 000 t in the 2010s. This might indicate that the recapture of tags and/or the tag return rate (the percentage of recaptured tags that is actually reported) is much lower than before, for whatever reason. Inspection of the spatial distribution of the returned tags (Figure 52), and the mean distances between release and recapture (Figure 58) hints at a lower recapture rate, rather than a lower tag return rate.

The number of days at large for tagged eel has been fairly similar over the decades (Figure 58). It should be noted that – in recent years - the maximum number of days at large is related to the length of the fishing season allowed: the maximum days runs from the start of the season to the end of the season, and restrictions in the season length necessarily lead to a lower maximum period at large. Should the season be shortened even further, then this would decrease the maximum number of days at large accordingly.

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. Restocking in coastal waters, on p. 57, above.

The references for this Annex are included in the reference list of the main report, on page 47.

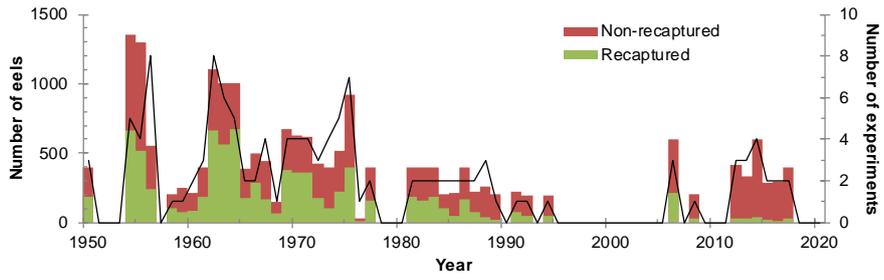
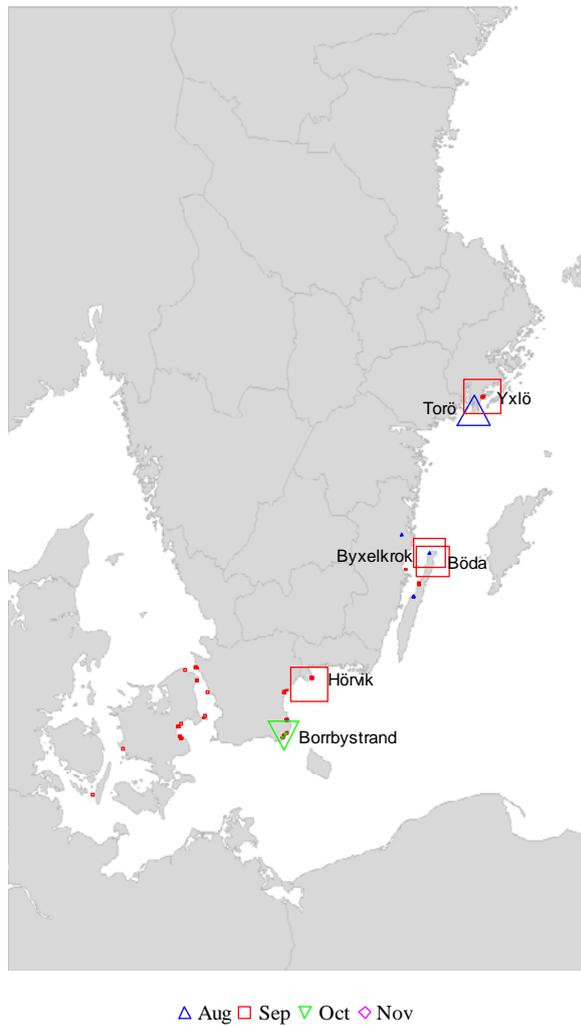


Figure 52 Time trend in the number of tagging experiments and the number of eels being tagged.



△ Aug □ Sep ▽ Oct ◇ Nov

Figure 53 Location of the tagging experiments in the years 2015-2017. The size of the larger symbols is proportional to the number of eels released. The small dots represent recaptures of single eels.

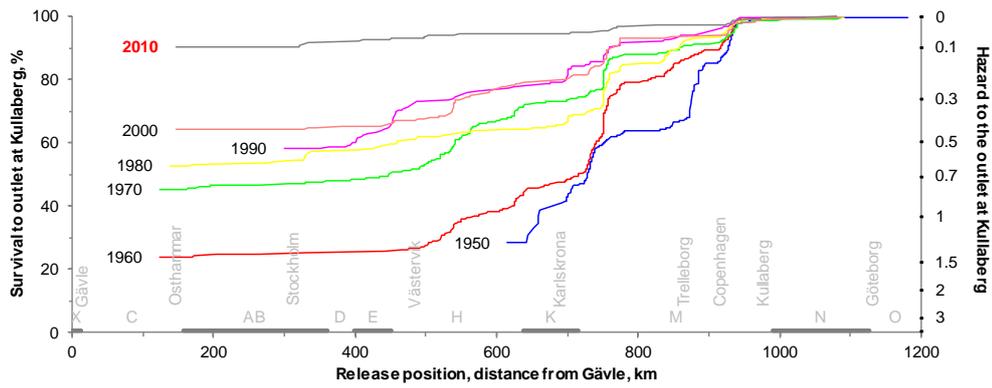


Figure 54 Hazard and survival by decade, estimated by the Kaplan-Meier method. The horizontal axis gives the distance from Gävle, just north of the northernmost release. The left vertical axis expresses the net survival observed in the recapture data; the right vertical axis expresses the same in terms of the accumulated hazard over the remaining interval.

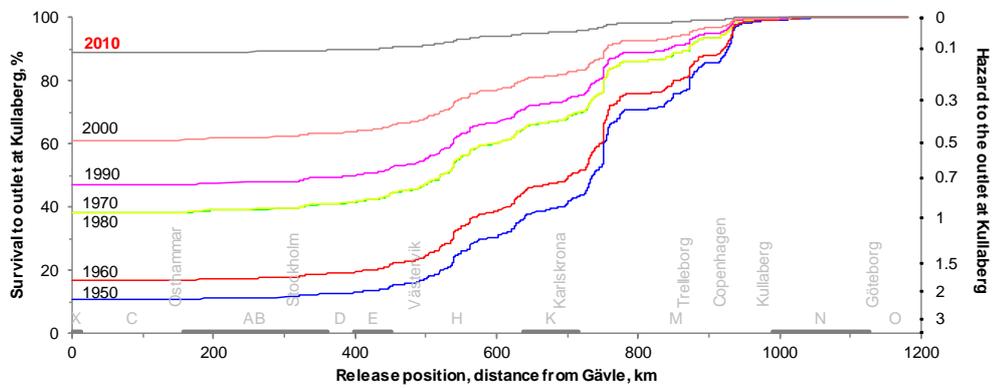


Figure 55 Hazard and survival, estimated by Cox proportional hazards model, by decade, without time-dependent covariates. The left vertical axis expresses the net survival from the release position  $t_0$  to the outlet of the Baltic at Kullaberg; the right vertical axis expresses the same in terms of accumulated hazard over that interval.

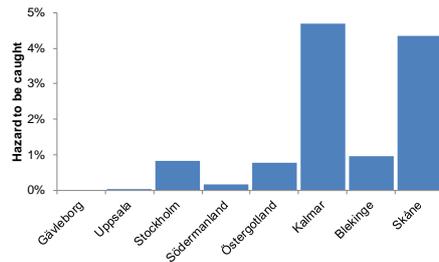


Figure 56 Hazard by county (län), in the 2010s.

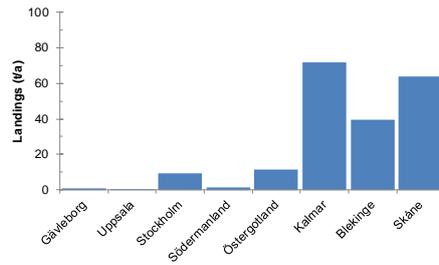


Figure 57 Landings by county (län), in the 2010s.

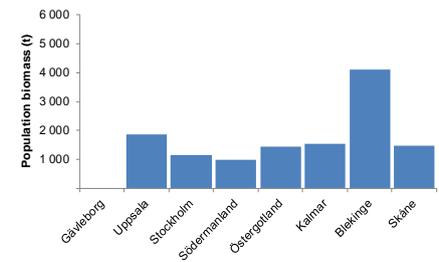


Figure 58 Estimated population size by county (län), in the 2010s. Since catches and hazards in Gävleborg were effectively zero, no estimate is derived there.

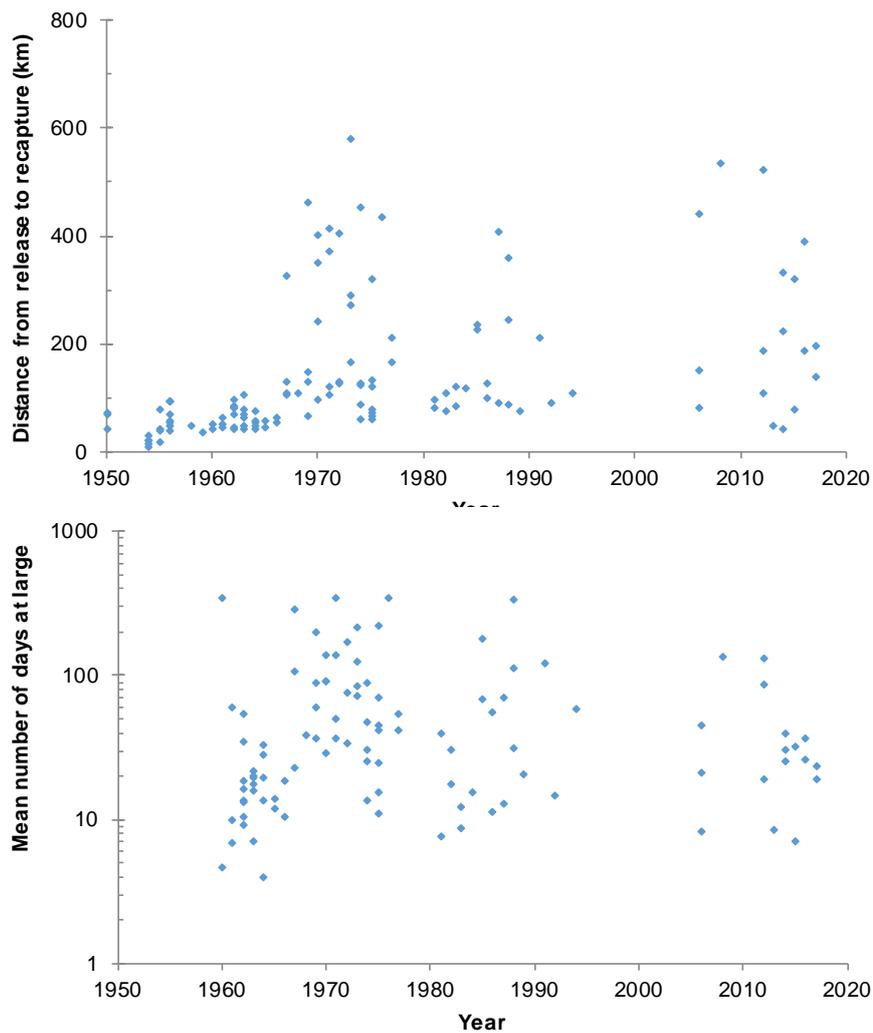


Figure 59 Mean distance and mean number of days at large between tag release and tag recapture, by year. Each dot represents a tagging experiment. Note the logarithmic vertical axis in the second graph.

