Aqua reports 2015:11

Assessment of the eel stock in Sweden, spring 2015

Second post-evaluation of the Swedish Eel Management Plan

Willem Dekker

Sveriges lantbruksuniversitet
Swedish University of Agricultural Sciences

Institutionen för akvatiska resurser

Assessment of the eel stock in Sweden, spring 2015
Second post-evaluation of the Swedish Eel Management Plan

Willem Dekker

Address:
Swedish University of Agricultural Sciences, Department of Aquatic Resources,
Stångholmsvägen 2, 178 93 Drottningholm, Sweden

June 2015
SLU, Department of Aquatic Resources

Aqua reports 2015:11
ISBN: 978-91-576-9331-0 (electronic version)

Publisher:
Magnus Appelberg

This report may be cited as:
Dekker, W. (2015). Assessment of the eel stock in Sweden, spring 2015. Second post-evaluation of the Swedish Eel Management Plan. Swedish University of Agricultural Sciences, Aqua reports 2015:11. Drottningholm. 93 pp.

Download the report from:
http://www.slu.se/aquareports
http://epsilon.slu.se/

E-mail
Willem.Dekker@slu.se

This report has been reviewed by:
Henrik Svedäng, Swedish University of Agricultural Sciences, Department of Aquatic Resources, Institute of Marine Research
Cedric Briand, Institution d'aménagement de la Vilaine, La Roche Bernard, France
Patrick Lambert, IRSTEA, Cestas, Bordeaux, France

This work was funded by the Swedish Agency for Marine and Water Management.

Front cover: Jacob van Maerlant (ca. 1235-1300), Der Naturen Bloeme.

Abstract

The population of the European eel *Anguilla anguilla* (L.) 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 2015, intending to feed into the national reporting to the EU; this updates and extends the report by Dekker (2012).

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, 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 (actual, potential and pristine) and mortality endured by those eels during their lifetime. The assessment is broken down on a regional basis, with different impacts dominating in different areas.

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 recruitment trend and the overall status of the stock.

On the west coast, a fykenet fishery on yellow eel was overexploiting the stock, until this fishery was completely closed in spring 2012. Though research surveys using fykenets continued, insufficient information is currently available to assess the recovery of the stock. Obviously, current fishing mortality is zero, but no other stock indicators can be presented. It is recommended to develop a comprehensive plan for monitoring the recovery of the stock.

In order to support the recovery of the stock, or to compensate for mortality elsewhere, young eel has been restocked on the west coast. No follow-up monitoring has been established. Noting the small expected effect – in comparison to the potential natural stock on the west coast, when recovered – it is recommended to reconsider this programme, or to set up adequate follow-up monitoring.

For inland waters, this report presents a major update of the 2012 assessment. In the 2012 assessment, eel production estimates were based on information from past restocking, but natural recruitment and assisted migration were ignored; these have now been included. Additionally, the impact of hydropower is now assessed in a spatially explicit reconstruction.

Based on 75 years of data on natural recruitment into 24 rivers, a statistical model is developed 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 at other places in Europe too, the cause of this is still unclear.

Using the results from the recruitment model, in combination with historical data on assisted migration (young eels transported upstream, across barriers) and restocking (imported young eels), the production of fully grown, silver eel is estimated for every lake and year separately. Subtracting the catch made by the fishery and down-sizing for the mortality incurred when passing hydropower stations, an estimate of the biomass of silver eel escaping from each river towards the sea is derived. Since 1960, the production of silver eel in inland waters has declined from 500 to 300 t/a, and natural recruitment (assisted or not) has gradually been replaced by restocking for 90%. Fisheries have taken just over 30% of the silver eel, while the impact of hydropower has ranged from 20% to 60%. Escapement is estimated to have varied from 10% (35 t) in the late 1990s, to 30% (100 t) in the 2010s. The biomass of current escapement (including eels of restocked origin) is approx. 15% of the pristine level, that is 28% of the current potential. This biomass is below the 40% limit of the Eel Regulation, and anthropogenic mortality exceeds both the short-term limit establishing recovery (15%) 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. Natural (not assisted) recruits were far less impacted by hydropower, since they could not climb the hydropower dams when immigrating. Later, restocking has been practised in unobstructed lakes (primarily Lake Mälaren, 1990s), and is now concentrated in obstructed lakes (primarily Lake Vänern, to a lesser extent Lake Ringsjön, and many smaller ones). Trap & Transport of silver eel - from above barriers towards the sea - has added 1-6% of silver eel to the escapement. Without restocking, the biomass affected by fishery and/or hydropower would be only 10% of the currently impacted biomass, but the stock abundance would reduce from 10% to only 3% of the pristine biomass.

It is recommended to reconsider the current action plans on inland waters, to take into account the results of the current, more comprehensive assessment. It is further recommended to ground-truth the current assessment on independent stock surveys.

For the Baltic coast, the 2012 assessment has been updated, using information from re-continued mark-recapture experiments. 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 is estimated at 2%. However, this fishery is just one of the anthropogenic impacts (in other areas/countries) affecting the Baltic eel stock. 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 these estimates are highly uncertain due to the low values for catch and mortality (near-zero estimation problems). 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 *Anguilla anguilla* (L.) 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 respektiva 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 2015, detta med syfte att tjäna som underlag till den svenska uppföljningsrapporten till EU. Rapporten uppdaterar och utvidgar därmed 2012-års utvärdering (Dekker 2012).

Rapporten utvärderar påverkan från fiske, utsättning och kraftverksrelaterad dödlighet på ålbeståndet. Annan antropogen påverkan som klimatförändring, förorening, 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 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.

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.

Gulålen på <u>västkusten</u> överexploaterades 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 andra beståndsindikationer. Det rekommenderas att det tas fram en allsidig plan för övervakningen/uppföljningen av ålens återhämtning.

Som en åtgärd för att bygga upp ålbeståndet eller för att kompensera för dödlighet på annat håll, så har unga ålar satts ut på västkusten. Någon riktad uppföljning av dessa utsättningar är emellertid inte etablerad. Med tanke på det förväntat lilla tillskottet från utsättningarna, jämfört med den potentiella naturliga bestånd på

västkusten efter återhämtning, så bör utsättningarna på västkusten omvärderas eller att man etablerar ett uppföljningsprogram.

För inlandsvattnen så redovisar rapporten en omfattande uppdatering av 2012-års beståndsuppskattning. 2012 var ålproduktionen enbart beräknad från tidigare utsättningar av ål, medan den naturliga rekryteringen och de ålar samlats i nedre delarna av respektive vattendrag inte beaktades. Detta är nu inkluderat. Dessutom är vattenkraftens påverkan beräknad i form av en detaljerad rekonstruktion.

Baserat på 75 års data över naturlig rekrytering till 24 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 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 rekryteringsmodellen 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 produktionen av blankål från alla sjöar och år uppskattats. Genom att sedan dra bort mängden fångad ål och de som dött vid kraftverkspassager, har mängden överlevande lekvandrare (lekflykt) uppskattats. Sedan 1960, så har produktionen av blankål minskat från 500 till 300 ton per år. Den naturliga rekryteringen av ål, uppflyttade eller ej, har gradvis ersatts till 90 % genom utsättning av importerade ålyngel. Fisket har tagit något över 30 % av blankålen, medan påverkan (dödlighet) från vattenkraft har varierat från 20 % till 60 %.Utvandringen av blankål till havet har varierat från 10 % (35 ton) under sent 1990-tal till 30 % (100 ton) under 2010-talet. Biomassan av utvandrande blankål (inklusive de av utsatt ursprung) uppskattas idag vara ungefär 15 % av den jungfruliga mängden, dvs . 28 % av dagens potential.

Biomassan av lekvandrare är därmed mindre än den 40 %-gräns som Ålförordningen föreskriver och den mänskligt introducerade dödligheten överskrider såväl den kortsiktiga gränsen för beståndets återhämtning om 15 %, som 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.

Naturliga, dvs. inte uppflyttade rekryter, var mycket mindre påverkade av vattenkraften, då de normalt inte kan vandra uppströms kraftverksdammar. På senare tid har utsättningarna gjorts i sjöar med fria vandringsvägar till havet (till stor del i Mälaren under 1990-talet), men görs sedan några år tillbaka delvis i sjöar med nedströms vandringshinder (främst i Vänern, men också i Ringsjön och flera mindre sjöar). Trap & Transport av blankål, från uppströms vattenkraftverk ner till respektive mynningsområde, har tillfört ytterligare 1-6 % till lekvandringen. Utan ålutsättning, skulle biomassan av ål påverkad av fiske och vattenkraft bara vara 10 % av vad som faktiskt påverkas nu. Samtidigt skulle ålbeståndet vara bara 3 % av den ursprungliga biomassan, att jämföra med dagens 10 %.
Det rekommenderas att nuvarande förvaltningsplan för ål i sötvatten omprövas, detta för att beakta den mer allsidiga beståndsuppskattningen i föreliggande arbete. Utöver det, rekommenderas att vår beståndsuppskattning verifieras genom oberoende beståndsstudier.

För ostkusten, så har 2012-års beståndsuppskattning uppdaterats genom att inkludera nya data från våra fortsatta fångst-återfångstexperiment. Resultaten indikerar att fiskets inverkan snabbt minskar över tid, kanske snabbare mot slutet av 2010-talet än tidigare. Dagens påverkan från fisket 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 åstadkommits. Nuvarande uppskattning av ålbiomassan (blankål) i Östersjön är i storleksordningen några tusen ton, men den skattningen är mycket osäker på grund av de låga värden på fångst och dödlighet som den grundas på (”nära noll problematik”). En integrerad, enhetlig beståndsuppskattning för hela Östersjön behövs för att verifiera våra skattningar.
Vi rekommenderar 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 beståndsuppskattnings-metoder mellan länder initieras. På så sätt kan en konsekvent och mer kostnadseffektiv beståndsuppskattning komma till stånd i hela Europa.

Table of contents

1	**Introduction**	**10**
1.1	Context	10
1.2	Aim of this report	10
1.3	Structure of this report	11
1.4	The Swedish eel stock and fisheries	11
1.5	Spatial assessment units	14
1.6	Management targets	15
2	**Recruitment indices**	**17**
3	**Restocking**	**20**
3.1	Restocked quantities	20
3.2	Restocking and stock assessments	21
3.3	Restocking and stock indicators	21
4	**Fisheries, catch and fishing mortality**	**24**
5	**Impact of hydropower on silver eel runs**	**27**
6	**Trap & Transport of silver eel**	**28**
7	**Stock indicators**	**29**
8	**Discussion**	**32**
8.1	Progress since the 2012 assessment	32
8.2	Requirements for the 2015 reporting to the EU	34
9	**Recommendations and advice**	**35**
10	**References**	**38**
Annex A	**West coast eel stock**	**41**
Annex B	**Recruitment into inland waters**	**44**
B.1	Data	44
B.2	Spatial and temporal patterns in recruitment	48
B.3	Analysis	49
B.4	Results	52
	B.4.1 Spatial trends in elver trap catches	52
	B.4.2 Temporal trends in elver trap catches	55

B.5 Predicted trends in natural recruitment into inland waters 56

Annex C Reconstruction of the inland stock 57

C.1 Data and methods 57

C.1.1 Inputs to the inland stock 58

C.1.2 Outputs from the inland stock 66

C.1.3 Conversion from recruit to silver eel 73

C.1.4 Estimation of escapement 75

C.2 Results 77

C.2.1 Silver eel production 77

C.2.2 Silver eel destination 80

C.2.3 Natural mortality M 82

C.2.4 Mortality related to hydropower H 86

Annex D Impact of the Baltic Coast fishery 89

1 Introduction

1.1 Context

The population of the European eel *Anguilla anguilla* (L.) 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; ICES 2014). 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. The common limit for all these plans is an escapement of at least 40 % of the silver eel biomass relative to the escapement if no anthropogenic influences would have impacted the stock and recruitment would not have declined. 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). This report updates, extends and reviews that report.

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, and the disruption of migratory behaviour by transport of eels. 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; 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. Blending in unquantified aspects into a quantitative analysis jeopardises the assessment, risking a failure to identify a possibly inadequate management of the quantitative factors (fisheries and hydropower mortality).

According to the EU Regulation, Member States will report to the Commission by July 2015 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 ($B_{current}$, B_{best} and B_0) and the mortality they endured over their lifetime (ΣA).

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 discussed directly; their net effect on the stock, however, will show up in the assessments.

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 2012-2014.

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 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, but none of that remains. On the next pages, the main habitats and fisheries are briefly described.

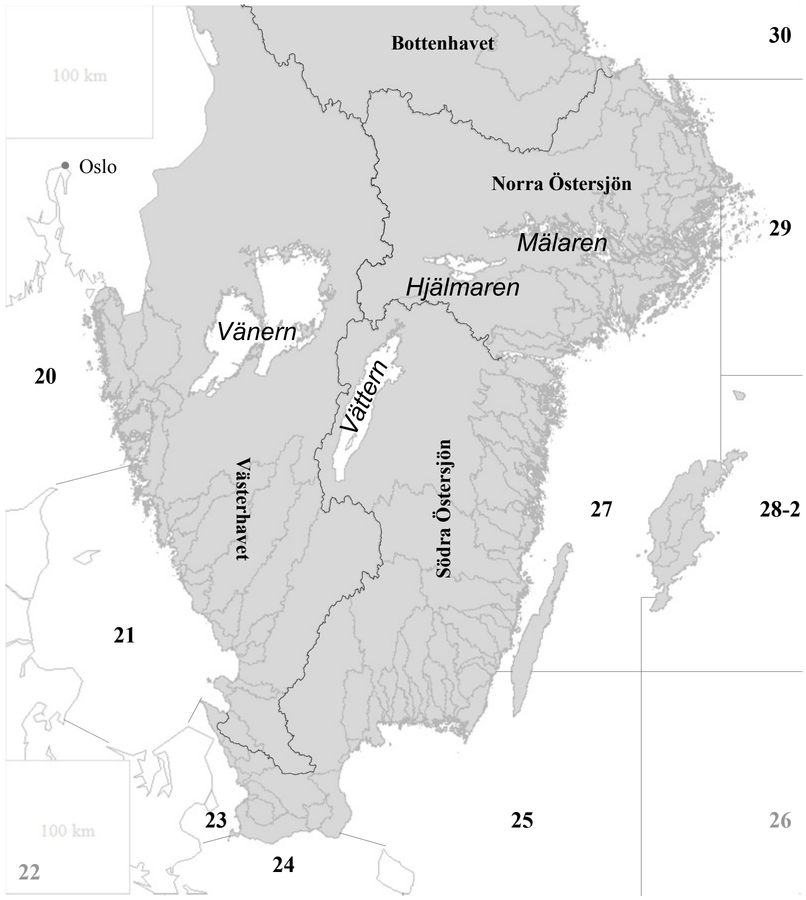
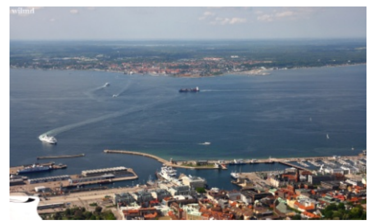


Figure 1 Map of the study area, the southern half of Sweden (north up to 61°N). The names in italics indicate the four largest lakes; the names in bold indicate the Water Basin Districts related to the Water Framework Directive (not used in this report); the numbers refer to the ICES subdivisions; the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

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

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



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

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

Inland waters. Eels are found in most lakes, except in the high mountains and the northern parts of the country. Pound nets are used to fish for eel in the biggest lakes Mälaren, Vänern and Hjälmaren, and in some smaller lakes in southern Sweden. In inland lakes, restocking of young eels has contributed to current day's yield, while barriers and dams have obstructed the natural immigration of young eels. Traditional eel weirs (lanefiske) have been operated at several places, and some are still being used. Hydropower generation impacts the emigrating silver eel.

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 hydropower generation.
3. Trap & Transport of silver eel – only that. The presentation of Trap & Transport data has been included in Annex C, in the discussion of inland waters.
4. Baltic coast – natural recruitment and restocking, fishery on silver eel.

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

Symbols & notation used in this stock assessment

The assessments in this report derive the following stock indictors:

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

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

1.6 Management targets

The EU Eel Regulation sets a long-term general objective ("the protection and sustainable use of the stock of European eel"), delegating the local management, the implementation of protective measures, the monitoring, and the local post evaluation to its Member States (Anonymous 2007; Dekker, 2009). 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 no anthropogenic influences would have impacted the stock and recruitment might 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 very informative (Dekker 2010). 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; ICES 2010, 2011, 2012, 2013a, 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. For every possible biomass limit, a corresponding long-term mortality limit can be derived. A lifetime 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 2012 post-evaluation supplied by the EU Commission includes a request to report on the quantities $B_{current}$, B_{best}, B_0 and ΣA – 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 (the linear relation is showing up as a curved line on the logarithmic scale used here).

Within ICES, there is 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 yearclasses 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 yearclasses (up to 50) spawning simultaneously, none of the risks involved in depleting short-lived species actually applies to eel.

Both the Eel Regulation and the Swedish Eel Management Plan 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).

For other anthropogenic impacts (pollution, spread of parasites, disruption of migration by transport, 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[1] 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.

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

[1] 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.

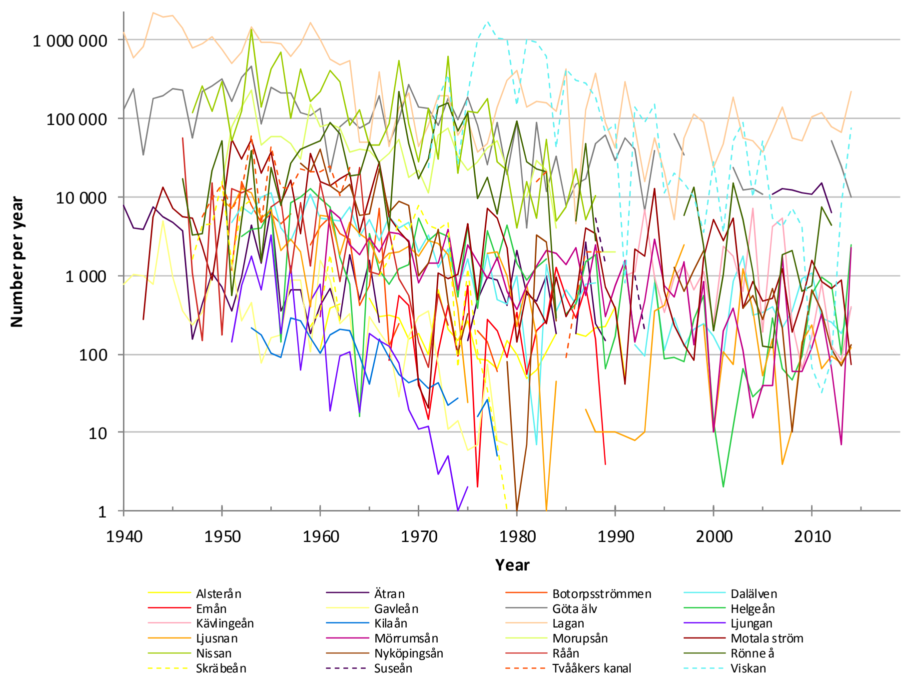


Figure 2 Trends in the number of elvers trapped at barriers, in numbers per year. Note the logarithmic character of the vertical axis. For further details, see Annex B.

The nuclear power plant at Ringhals takes in cooling water in front of the coast along the Kattegat, sucking 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*).

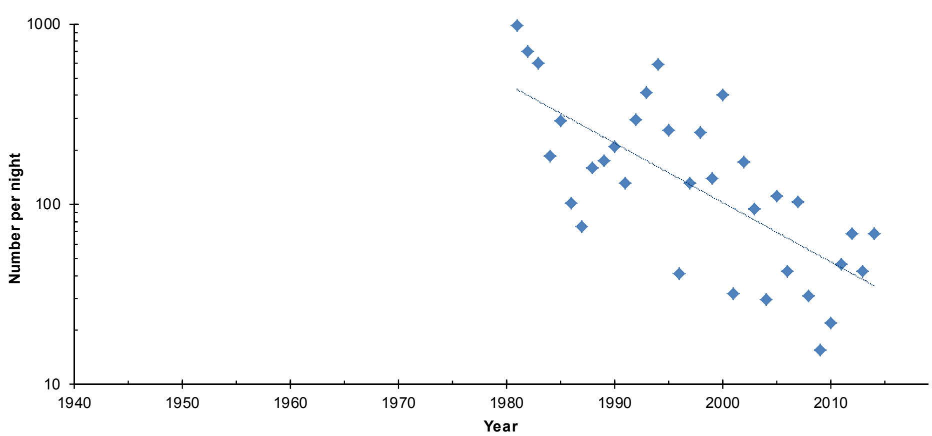


Figure 3 Time trend in glass eel recruitment at the Ringhals nuclear power plant on the Swedish Kattegat Coast. Note the logarithmic character of the vertical axis.

A modified Methot-Isaacs-Kidd Midwater trawl (MIKT) is used from R/V Argos 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*).

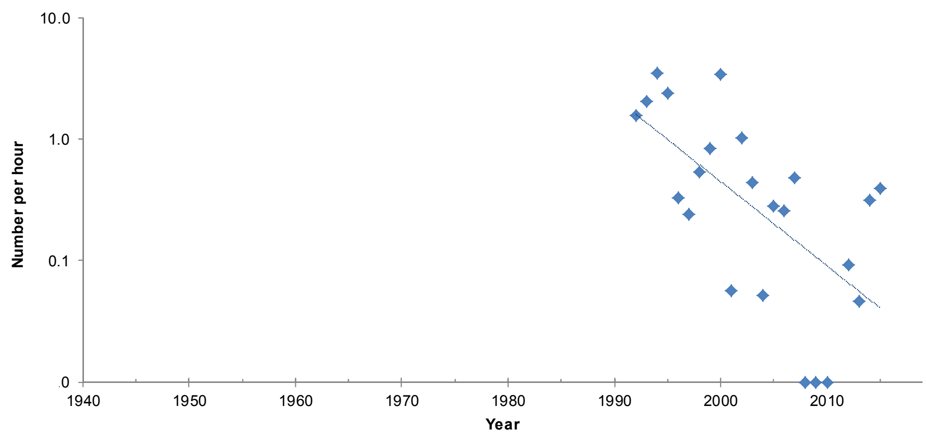


Figure 4 Catch of glass eels (number per hour trawling) by a modified Methot–Isaacs–Kidd Midwater trawl (MIKT) in the Skagerrak-Kattegat 1992–2011. In 2008-2010, zero glass eels were caught; in 2011, no sampling took place. Note the logarithmic character of the vertical axis.

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. Restocking of young eel started in Sweden in the early 1900s, and has been applied in inland waters as well as on the coast.

3.1 Restocked quantities

Table 1 provides an overview of the numbers applied for restocking in most recent years. Annex C gives full detail (spatial and temporal) for the inland waters; Annex A for the coastal waters.

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

	Actual numbers				Glass eel equivalents		
Year	West coast	Inland waters	Baltic coast	Year class	West coast	Inland waters	Baltic coast
2000		1 477 542	566 722	2000	9 590	1 221 368	265 388
2001		1 033 108	312 597	2001	8 806	1 086 338	104 498
2002	24 255	1 272 182	454 184	2002		1 202 817	409 449
2003	12 502	495 751	484 713	2003		334 385	397 440
2004	21 625	1 165 971	336 156	2004	15 640	1 105 576	247 245
2005	6 195	947 822	155 667	2005		919 298	162 312
2006		972 781	343 847	2006		1 011 346	358 524
2007	7 500	821 498	169 576	2007	7 820	830 750	174 406
2008		1 130 187	366 927	2008		1 056 273	382 589
2009		599 690	180 002	2009		611 540	184 245
2010	180 000	1 726 510	30 000	2010	187 683	1 800 172	31 281
2011	543 000	2 011 984	71 000	2011	566 178	2 097 855	74 031
2012	553 000	1 956 022	57 000	2012	576 605	2 039 480	59 433
2013	581 600	1 985 984	90 000	2013	606 426	2 070 679	93 842
2014	778 611	2 049 432	120 000	2014	811 846	2 136 812	125 122

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. Since the absolute number of natural recruits is generally unknown, the sum of natural and restocked recruits is unknown. Hence, these 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 Bbest was estimated at 1154 t (Dekker 2012), and current restocking (0.8 million in 2014) will potentially produce less than 100 t. For the Baltic coast, the potential production of silver eel Bbest was estimated at 3770 t (Dekker 2012), and current restocking (0.1 million in 2014) will potentially produce 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 2014 is estimated at 35 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 ΣA, expressing the relation of the actual silver eel escapement $B_{current}$ to the potential escapement if no anthropogenic actions had influenced the stock B_{best}, can be interpreted in two different ways. If the silver eel produced from restocking is included in the estimate of B_{best} (say B_{best}^{+}), that is $\Sigma A = -\ln(B_{current}^{+}/B_{best}^{+})$, the resulting mortality indicator expresses the mortality exerted on any part of the stock, both natural and restocked. If, however, the restocking is *not* included in the calculation of B_{best} (say B_{best}^{-}), the resulting indicator $\Sigma A = -\ln(B_{current}^{+}/B_{best}^{-})$ reflects the effect of management actions (comparing the actual escapement to one without any anthropogenic impact), but does *not* express the mortality actually experienced by any eel in the stock.

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 ΣA 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. Different countries using 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 $B_{current}$, but not in B_{best}; hence, the estimate of ΣA 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 $B_{current}$ and B_{best}; hence, the estimate of ΣA constituted a true mortality rate, but did not reflect the effect of restocking.

The classical objectives for restocking in Sweden has been to support the fishery, but 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 on hydropower and eel KTÅ; 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 ΣA 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 restockings would be required to cope with the high mortality. Using $\Sigma A = -\ln(B_{current}^{+}/B_{best}^{-})$, the indicator would not flag this situation. To avoid this, the positive effect of restocking will not be included in our estimates of mortality ΣA, and – where possible - biomasses of silver eel are expressed separately for eels of natural and of restocked origin. That is: we use $\Sigma A = -\ln(B_{current}^{+}/B_{best}^{+})$. For the status of the stock relative to pristine conditions ($\%SSB = 100*B_{current}/B_0$), this report provides estimates *with* and *without* including restocking into the estimate of B_0 (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 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 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 has been made; the old estimates have been copied without change.

For the fishery in inland waters, Annex C presents a full update of data and methods for the assessment of the inland stock. The 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. In this report, Annex B estimates the number of natural recruits, while Annex C reconstructs the inland stock, taking into account the contributions from natural, assisted and restocked recruits, as well as the impact from the fishery and hydropower, in a spatially and temporally explicit reconstruction.

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 now been updated, adding recent mark-recapture data; see Annex D for details.

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.

Table 2 *Fisheries statistics, by year and area.*

	Landings (tonnes)			Fishing mortality ΣF (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000	154	114	263	1.79	0.44	
2001	226	120	297	2.53	0.47	
2002	216	102	273	2.41	0.40	
2003	192	98	275	2.15	0.38	
2004	216	113	254	2.43	0.47	
2005	214	115	346	2.39	0.50	0.1
2006	239	128	366	2.66	0.59	
2007	170	114	418	1.91	0.49	
2008	164	118	389	1.86	0.50	
2009	107	97	310	1.19	0.36	
2010	108	110	307	1.20	0.39	
2011	83	96	271	0.93	0.32	
2012	0	101	239	0	0.33	0.02
2013	0	103	271	0	0.34	
2014	0	111	213	0	0.38	

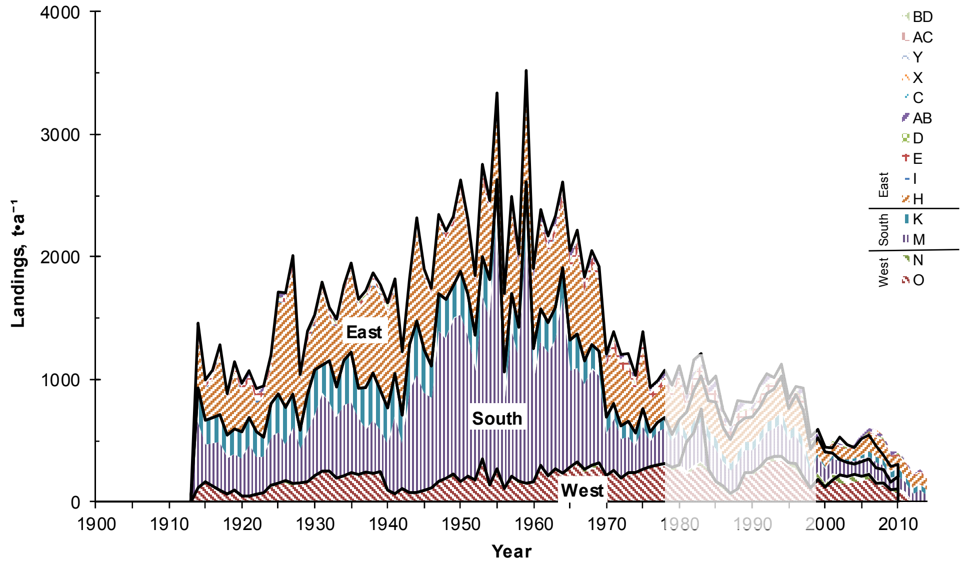


Figure 5 Trend in landings from the coastal fisheries, by county (colours) and area (black lines). In the years 1978-1998 (faded), due to lack of detailed records, it has been assumed that the percent-wise contribution of each county had remained constant. Note that the total landings on the Baltic coast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

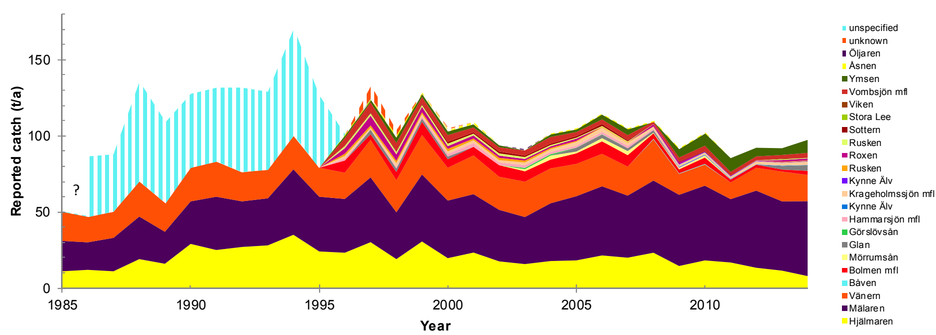


Figure 6 Trends in landings from inland waters. Before 1996, only the totals for all lakes (except the three largest ones) are known; statistics before 1986 are not available (yet).

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 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.*

	Biomass of silver eel (tonnes)			Hydropower mortality (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000		156			1.41	
2001		131			1.09	
2002		121			0.85	
2003		103			0.67	
2004		78			0.53	
2005		64			0.45	
2006		50			0.38	
2007		65			0.45	
2008		80			0.57	
2009		115			0.73	
2010		126			0.81	
2011		146			0.87	
2012		160			0.99	
2013		156			0.97	
2014		147			0.96	

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 6 on page 65 for further details).

The silver eel in the Trap & Transport programme 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). Hence, no unique comparison can be made between the quantity released and the stock they relate to – the Trap & Transport cannot be expressed as a (negative) mortality rate.

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

	Biomass of silver eel (tonnes)			As mortality (rate)		
Year	West coast	Inland waters	Baltic coast	West coast	Inland waters	Baltic coast
2000						
2001						
2002						
2003						
2004						
2005						
2006						
2007						
2008						
2009						
2010	5.2					
2011	4.9		2.6			
2012	8.6		1.4			
2013	10.4		3.4			
2014	13.9		5.1			

7 Stock indicators

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. Though monitoring has continued, no assessment has been made. Obviously, fishing mortality is zero, but the recent biomass indicators are unknown.

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 only the impact of the Swedish silver eel fishery. Other impacts on the same eels, in earlier life stages and other countries, have not been included – no integrated assessment for the whole Baltic stock has been established. For the Swedish eel fishery, Dekker (2012) derived estimates of ΣA from the analysis in Dekker & Sjöberg (2013); estimated B_{best} from the ratio of landings (mean 377 t/a over the years 2006-2008) to ΣA; and calculated $B_{current}$ as what is left after the catch has been taken from B_{best}. Over the years 2010-2014, the hazard ΣA is estimated at approx. 2%; the average catch was 260 t/a, resulting in an estimate of B_{best} of over 10 000 t. This appears not to be a realistic estimate. See Annex D for further details. For the time being, the 2012-estimate B_{best}=3770 t is maintained.

For the Trap & Transport programme, only the biomass of silver eel affected is reported.

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

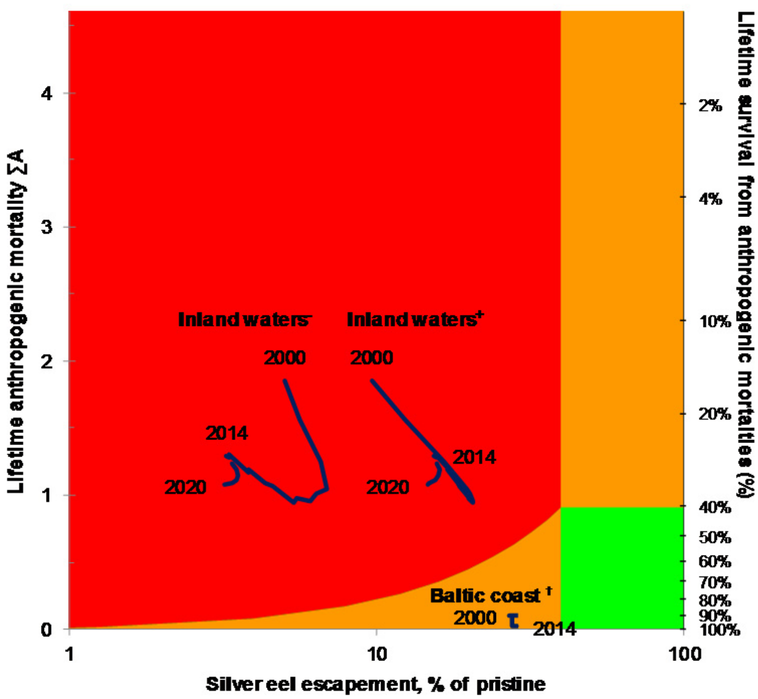


Figure 7 Precautionary Diagram for the Swedish eel stock in inland waters and along the Baltic coast. For the west coast, no stock indicators are currently available. For inland waters, the true mortality is shown (not interpreting restocking as compensating for other mortalities), giving separate curves for the current biomass with or without the contribution from restocking, $\%SSB^{+}$ resp. $\%SSB^{-}$.

† For the Baltic coast, only the impact of the Swedish silver eel fishery is included; impacts in other life stages, in other areas/countries, are not.

Table 5 *Stock indicators by area and year. For inland waters, biomass indicators are given with (+) and without (-) the contribution from restocked eels. All mortality estimates refer to true mortality (both on natural and restocked eels), not interpreting restocking as a compensation for other mortalities. For all coastal waters, ΣH=0, hence ΣF=ΣA. For Trap & Transport, the biomass released is specified, for the West coast and the Baltic separately. All biomass indicators expressed in tonnes, mortality indicators as rate per lifetime, %SPR (relative survival) and %SSB (relative state of the stock) in percent.*

	West coast						Inland waters												Baltic coast						T&T		
							with restocking +				without restocking -				Mortality rates										W	B	
year	$B_{current}$	B_{best}	B_0	%SSB	∑A	%SPR	$B_{current}^+$	B_{best}^+	B_0^+	$\%SSB^+$	$B_{current}^-$	B_{best}^-	B_0^-	$\%SSB^-$	∑F	∑H	∑A	%SPR	$B_{current}$	B_{best}	B_0	%SSB	∑A	%SPR	$B_{current}$		year
2000					1.79		50	320	524	9.6	15	96	300	5.0	0.44	1.41	1.85	15.8	3507	3770	12500	28.1					2000
2001					2.53		67	319	538	12.5	17	80	300	5.6	0.47	1.09	1.56	21.0	3473	3770	12500	27.8					2001
2002					2.41		91	314	545	16.6	20	68	300	6.6	0.40	0.85	1.24	28.9	3497	3770	12500	28.0					2002
2003					2.15		109	309	550	19.7	21	59	300	6.9	0.38	0.67	1.05	35.1	3495	3770	12500	28.0					2003
2004					2.43		110	300	548	20.0	19	52	300	6.3	0.47	0.53	1.01	36.5	3516	3770	12500	28.1	0.10	90.5			2004
2005					2.39		112	291	543	20.6	18	47	300	6.1	0.50	0.45	0.96	38.5	3424	3770	12500	27.4					2005
2006					2.66		107	285	541	19.8	16	44	300	5.5	0.59	0.38	0.98	37.6	3404	3770	12500	27.2					2006
2007					1.91		113	291	549	20.5	16	41	300	5.4	0.49	0.45	0.95	38.8	3352	3770	12500	26.8					2007
2008					1.86		104	303	563	18.5	14	40	300	4.6	0.50	0.57	1.07	34.4	3381	3770	12500	27.0					2008
2009					1.19		107	319	580	18.5	13	39	300	4.4	0.36	0.73	1.09	33.6	3460	3770	12500	27.7					2009
2010					1.20		101	333	595	17.1	12	38	300	3.9	0.39	0.81	1.19	30.5	3463	3770	12500	27.7			5		2010
2011	12	1154	1154	1.0	0.93	39	105	340	603	17.5	12	37	300	3.8	0.32	0.87	1.17	31.0	3499	3770	12500	28.0	0.02	98.0	5	3	2011
2012					0.00		94	345	608	15.5	10	37	300	3.3	0.33	0.99	1.30	27.3	3531	3770	12500	28.2			9	1	2012
2013					0.00		94	339	604	15.6	10	36	300	3.3	0.34	0.97	1.28	27.8	3499	3770	12500	28.0			10	3	2013
2014					0.00		91	330	595	15.3	10	35	300	3.3	0.38	0.96	1.29	27.5	3557	3770	12500	28.5			14	5	2014
2015					0.00		91	313	578	15.7	10	35	300	3.4	0.36	0.94	1.24	28.9									2015
2016					0.00		89	289	555	16.0	10	34	300	3.5	0.33	0.92	1.19	30.5									2016
2017					0.00		84	261	528	15.9	11	33	300	3.5	0.31	0.89	1.14	31.9									2017
2018					0.00		78	235	504	15.4	11	32	300	3.5	0.30	0.88	1.11	32.9									2018
2019					0.00		72	213	483	14.9	10	30	300	3.4	0.29	0.88	1.09	33.6									2019
2020					0.00		69	201	473	14.6	10	28	300	3.2	0.30	0.86	1.08	34.1									2020

8 Discussion

8.1 Progress since the 2012 assessment

Following the reporting on the status of the stock and the implementation of protective actions in 2012, ICES (2013b) conducted a technical evaluation of the progress reports submitted by the EU Member States to the European Commission.

For the west coast stock, ICES (2013b) wrote: "The assessment is a mortality-based assessment. Only fishery mortality has been considered. Since 2012 the fishery has been closed and other mortalities should now be evaluated. […] The major source of mortality has been reduced to zero in 2012. As it was a yellow eel commercial fishery, it will not change silver eel biomass before those saved yellow eel mature. It is expected to have a huge impact to future silver eel escapement."

The current report does not present any progress on the west coast assessment. Since the closure of the fishery, restricted monitoring using fykenets has been continued. However, past dynamics of the stock were poorly understood (Dekker 2012); trends in recruitment (diminishing), catch per unit effort (increasing) and landings (stable) contradicted each other. An in-depth analysis of old and new data will be required to clarify the dynamics of this stock.

Because of the many years since the last data in the assessment (2006), the current report does not extrapolate that old assessment and does not present new stock indicators for the west coast anymore. For the management of the west coast stock itself, the absence of stock indicators will not make a big difference, since the maximum protection level (closing the fishery) is already achieved. However, without follow-up monitoring, the effect of this closure remains putative. For the country as a whole, and the more so for the international assessment, the absence of stock indicators from the west coast constitutes a serious loss of information, impairing the derivation of international stock indicators and the international post-evaluation.

For the inland stock, ICES (2013b) wrote: "The Swedish Inland assessment is based on numbers of female silver eels. B_0 is calculated using historic catch data and assuming comparable fisheries mortality as current. Natural mortality is assessed to be low. $B_{current}$ is calculated based on a model. No field data [for groundtruthing] are available in the reports. …. Little action has been taken to reduce mortality at Hydropower stations. Reduced F and only limited trap and transfer, in combination with continued high H will actually increase H."

In the current report, an update and completion of the 2012 assessment is presented, now including the contributions from natural recruitment and assisted migration too, and covering all hydropower stations at full spatial detail. The ground-truthing information (notably the presence of an extensive database of electrofishing data in rivers and streams) has not been included in the assessment. The assessment presented here is focused on production data (inputs to and outputs from the stock) – not on the resulting stock abundance (standing stock density). Actually, the French 2012 assessments (Jouanin et al. 2012) took the reverse approach: based on standing stock estimates (mostly: electrofishing), an estimate of the quantities of silver eel produced is derived, without ground-truthing those production data. The conversion between standing stock data and production estimates (to and fro) requires information on growth, natural mortality and silvering. Results shown here indicate a considerable spatial variation in the parameters of those processes, as well as a huge uncertainty in the magnitude of natural mortality (Figure 45). Further progress will require field sampling, dedicated analysis, and the development - in international cooperation - of comprehensive assessment tools that can integrate information on production (rate) and abundance (state).

For the silver eel fisheries on the Baltic coasts, ICES (2013b) wrote: "The assessment relies on mark-recapture of silver eels within the EMU, but silver eel migrating within the EMU can come from the whole Baltic. The biomass found is thus part of the whole Baltic stock. The mortality indicator only takes into account commercial fishery occurring in the EMU. No other impacts are considered. It is inconsistent to consider only EMU impact while assessing part of the whole stock. Moreover given the mortality is based on past mark-recapture data and that B_{best} is assumed to be constant, the declining trend in anthropogenic mortality may be in fact be due to a declining B_{best} while having a constant (or increasing) fishery mortality. An update on mark-recapture data should give the data to evaluate the current fishing mortality. […] the way the trend of ΣF and thus the trend $B_{current}$ are calculated seems without groundtruthing."

In the current report, an update based on new mark-recapture experiments is presented, no longer extrapolating from past experiments. The results confirm the declining trend in fishing mortality. The very low estimates of fishing mortality, however, now complicate the estimation of the biomass of silver eel migrating along the coast considerably (near-zero estimation problems). 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.

8.2 Requirements for the 2015 reporting to the EU

A template for reporting stock indicators to the EU has been circulated informally. Additionally, the 2012 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 to avoid double counting.

9 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 2012 assessment (Dekker 2012). The national stock indicators were and will be used for the international assessment (ICES 2013a), on which the international advice is based. However, in compiling the international assessment, national stock indicators are taken at face value, used in good faith. No review of the data quality, methods and national achievements was given. 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 ΣA) is zero, but current, potential and pristine biomasses ($B_{current}$, B_{best} and B_0) could not be estimated from the currently available data. After the fishing ban, routine fykenet surveys have been continued, but the recovery of the stock is not adequately quantifiable. The effect of restocking (to support recovery and/or to compensate for mortality in inland waters) is not monitored, and - given the small expected effect in comparison to natural recruits - that effect will be hard to quantify. To achieve the management targets of the Eel Regulation and the national Eel Management Plan, no further action can be taken. It is recommended

- to develop a comprehensive plan for monitoring the expected recovery after the fishing ban.
- to reconsider the effect of restocking on the coast, or to develop a follow-up monitoring.

For the inland stock: status indicators point out that the stock biomass is below the limit level, and anthropogenic impacts (fishery and hydropower) exceed the current limit, even exceed the limit that would apply if the stock biomass had been at a sustainable level. These indicators are derived from a detailed reconstruction of the silver eel production over the past decades, but ground-truthing the results has not been achieved and the quality of the landings data is doubtful. Management actions include assisting migration, restocking, fishing restrictions and Trap & Transport. These measures have strong interactions: adjusting one measure, any positive effect is likely to be largely annihilated by the 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. The current management limits are based on outdated assessments. It is recommended

- to develop an updated, comprehensive management plan for the inland stock.
- to improve the quality of the landings data, possibly reconsidering the registration system.
- to improve the quality of the assessment, by ground-truthing the results on independent stock surveys (electro-fishing in streams, fyke-netting in lakes).

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. Stock biomass is likely below the threshold. Fishing restrictions have reduced the fishing impact even further, but that affects the escapement biomass only marginally. The assessment of the fishing impact is based on re-continued mark-recapture experiments. Due to the low (and decreased) impact of the fishery, the number of recaptures is very low, making the estimates of biomass indicators highly uncertain (in contrast to the more accurate estimates of fishing mortality). To improve the biomass estimates, a comprehensive assessment of the targeted stock will be required, i.e.: an assessment of the production of silver eel in the whole Baltic area. It is recommended

- to continue the mark-recapture experiments, and to embed this assessment 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 of ICES-WGEEL. 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. Further inconsistencies are likely to emerge, due to the absence of an official template for the 2015 reporting. To address this situation, it is recommended

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

10 References

Anonymous 2007 Council Regulation (EC) No 1100/2007 of 18 September 2007 establishing measures for the recovery of the stock of European eel. Official Journal of the European Union L 248/17.
http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2007:248:0017:0023:EN:PDF (English)
http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2007:248:0017:0023:SV:PDF (Swedish)

Anonymous 2008 Förvaltningsplan för ål. Bilaga till regeringsbeslut 2008-12-11 Nr 21 2008-12-09 Jo2008/3901 Jordbruksdepartementet. 62 pp. [Swedish eel management plan.]

Anonymous 2012 Report on the implementation of the Swedish Eel Management Plan. Letter from the Ministry of Rural Affairs in Sweden to the Director General for Maritime Affairs and Fisheries, d.d. 26 June 2012. Ref L2012/1703, 16 pp.

Anonymous 2014 Report from the Commission to the Council and the European Parliament on the outcome of the implementation of the Eel Management Plans. COM(2014) 640 final, Council document 14619/14. 15 pp.

Åström M. and Dekker W. 2007 When will the eel recover? A full life-cycle model. ICES Journal of Marine Science, 64: 1–8.

Bevacqua D., Melià P., De Leo G.A. and Gatto M. 2011 Intra-specific scaling of natural mortality in fish: the paradigmatic case of the European eel. Oecologia **165**: 333–339.

Boström, M. K., Öhman, K. (2014). Mellanskarvens i Roxen. Förändringar i fisksamhället och mellanskarvens (*Phalacrocorax carbo sinensis*) föda [Cormorants in Lake Roxen. Changes in the fish community and the cormorant's diet]. Aqua reports 2014:10. Sveriges lantbruksuniversitet, Öregrund. 44 pp. (in Swedish)

Calles O. and Christianson J. 2012 Ålens möjlighet till passage av kraftverk - En kunskapssammanställning för vattendrag prioriterade i den svenska ålförvaltningsplanen samt exempel från litteraturen [The eels opportunity to pass hydropower stations]. Elforsk rapport 12:37, 77 pp. (in Swedish) Report published by Elforsk AB, at http://www.elforsk.se/Rapporter/?rid=12_37_

Clevestam P. D. & Wickström H. 2008 Rädda ålen och ålfisket! – Ett nationellt bidrag till en europeisk förvaltningsplan. Vetenskaplig slutrapport från pilotprojekt till Fonden för fiskets utveckling [Safe the eel and the eel fishery – a national contribution to a European management plan. Scientific end-report for the pilot project under the Fund for fisheries' development]. Swedish Board of Fisheries. Dnr: 231-1156-1104 (in Swedish).

Dekker W. 2000 A Procrustean assessment of the European eel stock. ICES Journal of Marine Science 57: 938-947.

Dekker W. 2004a Slipping through our hands - Population dynamics of the European eel. PhD thesis, 11 October 2004, University of Amsterdam, 186 pp. http://www.diadfish.org/doc/these_2004/dekker_thesis_eel.pdf

Dekker W. 2004b What caused the decline of Lake IJsselmeer eel stock since 1960? ICES Journal of Marine Science 61: 394-404.

Dekker W. 2008 Coming to Grips with the Eel Stock Slip-Sliding Away. *In* International Governance of Fisheries Eco-systems: Learning from the Past, Finding Solutions for the Future. *Edited by* M.G. Schlechter, N.J. Leonard and W.W. Taylor. American Fisheries Society, Symposium **58**, Bethesda, Maryland. pp 335-355.

Dekker W. 2009 A conceptual management framework for the restoration of the declining European eel stock. Pages 3-19 in J.M. Casselman & D.K. Cairns, editors. Eels at the Edge: science, status, and conservation concerns. American Fisheries Society, Symposium 58, Bethesda, Maryland.

Dekker W. 2010 Post evaluation of eel stock management: a methodology under construction. IMARES report C056/10, 67 pp.

Dekker W. 2012 Assessment of the eel stock in Sweden, spring 2012; first post-evaluation of the Swedish Eel Management Plan. Aqua reports 2012:9. Swedish University of Agricultural Sciences, Drottningholm. 77 pp.

Dekker W. & Beaulaton L. 2016 Faire mieux que la nature - the history of eel restocking in Europe. Environment and History, accepted 19-Feb-2015. http://www.ingentaconnect.com/content/whp/eh.

Dekker W., Deerenberg C. & Jansen H. 2008 Duurzaam beheer van de aal in Nederland: Onderbouwing van een beheersplan. IMARES rapport C041/08, 99 pp.

Dekker W. & Sjöberg N. B. 2013 Assessment of the fishing impact on the silver eel stock in the Baltic using survival analysis. Canadian Journal of Fisheries and Aquatic Sciences 70 (12), 1673-1684.

Dekker W. and Wickstrom H. 2015 Utvärdering av målen för programmet krafttag ål [Evaluation of the objectives of the programme ’ krafttag ål’, in Swedish]. Energiforsk rapport 2015:103, 49 pp. (in Swedish). Report published by Energiforsk AB, at http://www.elforsk.se/Programomraden/Vattenkraft/Rapporter/?download=report&rid=2015_103_.

Erichsen, L., 1976. Statistik över ålyngeluppsamling i svenska vattendrag [Statistics on the elver catches in Swedish rivers]. Information från Sötvattenslaboratoriet, Drottningholm 8. 36 pp. (in Swedish)

Hagström O. & Wickström H. 1990 Immigration of Young Eels to the Skagerrak-Kattegat Area 1900 to 1989. Internationale Revue der gesamten Hydrobiologie und Hydrographie, 75(6), 707-716.

ICES 2009 Report of the ICES Advisory Committee, 2009. ICES Advice, 2009. Books 1 - 11. 1,420 pp.

ICES 2010 Report of the Study Group on International Post-Evaluation on Eels (SGIPEE), 10–12 May 2010, Vincennes, France. ICES CM 2010/SSGEF:20. 42 pp.

ICES 2011 Report of the Study Group on International Post-Evaluation on Eels. (SGIPEE), 24–27 May 2011, London, UK. ICES CM 2011/SSGEF:13. 39 pp.

ICES 2012 Report of the 2011 session of the Joint EIFAC/ICES Working Group on Eels. Lisbon, Portugal, from 5 to 9 September 2011. ICES CM 2011/ACOM:18. Rome, FAO/Copenhagen, ICES. 2012. 841p.

ICES 2013a Report of the Joint EIFAAC/ICES Working Group on Eels (WGEEL), 18–22 March 2013 in Sukarietta, Spain, 4–10 September 2013 in Copenhagen, Denmark. International Council for the Exploration of the Sea, ICES CM 2013/ACOM:18. 851 pp.

ICES 2013b Report of the Workshop on Evaluation Progress Eel Management Plans (WKEPEMP), 13–15 May 2013, Copenhagen, Denmark. ICES CM 2013/ACOM:32. 757 pp.

ICES 2014 Report of the Joint EIFAAC/ICES/GFCM Working Group on Eels (WGEEL), 3-7 November in Rome, Italy. International Council for the Exploration of the Sea, ICES CM 2014/ACOM:18. 203+704 pp.

Jouanin C., Briand C., Beaulaton L., Lambert L. 2012. Eel DensityAnalysis (EDA 2.x). Un modèle statistique pour estimer l'échappement des anguilles argentées (*Anguilla anguilla*) dans un réseau hydrographique. Convention ONEMA-Cemagref. Partenariat 2011. Domaine : Espèces aquatiques continentales, Action 11.1. Rapport d'étape.

Kuhlin L. 2014 Info om Svensk vattenkraft [Information on Swedish hydropower]. Database of hydropower stations in Sweden, excel file "vattenkraft-2014-02-13.xlsx" made available by its author in personal communication, Feb 2014. See also http://vattenkraft.info/.

Lagenfelt I. 2012 Blankålsvandring i Göta älv, Telemetristudie 2010-2011 [Silver eel migration in river Göta Älv, telemetry study in 2010-2011]. Länsstyrelsen Västra Götalands län, report 2012-95, 26 pp. (in Swedish) Available at http://www.lansstyrelsen.se/vastragotaland/SiteCollectionDocuments/Sv/publikationer/2012/2012-95.pdf

Leonardsson K. 2012 Modellverktyg för beräkning av ålförluster vid vattenkraftverk [Model for the calculation of eel losses at hydropower stations]. Elforsk rapport 12:36, 83 pp. (in Swedish) Report published by Elforsk AB, at http://www.elforsk.se/Rapporter/?rid=12_36_

Mandel J. 1959 The analysis of Latin squares with a certain type of row-column interaction. Technometrics 1, 379-387.

Milliken, G.A. & Johnson, D.E., 1989, Analysis of messy data, Vol. 2: nonreplicated experiments. Van Nostrand Reinhold, New York, 199 pp.

Oeberst R. & Fladung E. 2012 German Eel Model (GEM II) for describing eel, *Anguilla anguilla* (L.), stock dynamics in the river Elbe system. Informationen aus der Fischereiforschung. 59: 9–17.

SAS 2014 SAS/STAT software, Version 9.4 of the SAS System for Windows, SAS Institute Inc., Cary, NC, USA.

Sjöberg N.B. 2015 Eel migration – results from tagging studies with relevance to management. Doctoral thesis, Stockholm University, 140 pp. [not yet available on line]

SMHI 2014 Flödesstatistik för Sveriges vattendrag [river statistics for Swedish drainage systems] http://www.smhi.se/klimatdata/hydrologi/vattenforing/om-flodesstatistik-for-sveriges-vattendrag-1.8369, accessed May 1st 2014.

Strömberg A., Lunneryd S.-G. & Fjälling A. 2012 Mellanskarv – ett problem för svenskt fiske och fiskodling? [Cormorants – a problem for the Swedish fisheries and aquaculture?] Aqua reports 2012:1. Sveriges lantbruksuniversitet, Öregrund. 31 pp. (in Swedish)

Svärdson G. 1976. The decline of the Baltic eel population. Reports of the Institute Freshwater Research Drottningholm 143: 136-143.

Trybom F. 1881 Om s.k. ålvinner vid Elfkarleby och plantering af ålyngel. Kongliga Landtbruks-akademiens Handlingar och Tidskrift 3: 189-192.

Wickström H. 2002. Monitoring of eel recruitment in Sweden. In: Dekker W. (ed) Monitoring of glass eel recruitment. Netherlands Institute of Fisheries Research, IJmuiden, the Netherlands, report C007/02-WD, Volume 2A, pp. 69-86.

Ăström M. and Dekker W. 2007 When will the eel recover? A full life-cycle model. ICES Journal of Marine Science, 64: 1–8.

Annex A West coast eel stock

Until recently, the west coast eel stock has been exploited by an extensive fykenet fishery; in spring 2012, this fishery has been closed completely. 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. Since 2012, however, no budget has been made available for this. Therefore, this Annex presents the primary monitoring data only.

The references for this Annex are included in the reference list of the main report, on page 38.

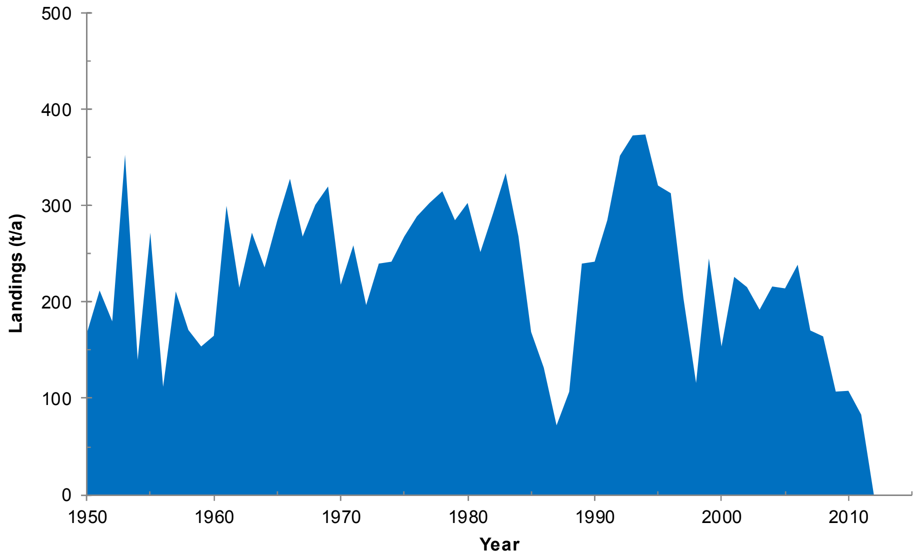


Figure 8 Landings from the west coast, by year. In spring 2012, the fishery was closed.

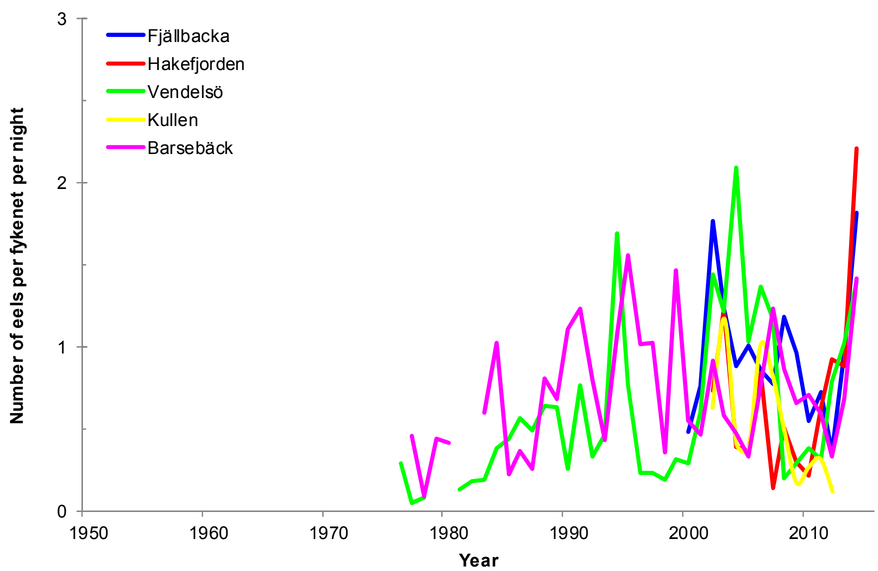


Figure 9 Time trend in the catches of the fishery-independent fykenet survey at various places along the west coast.

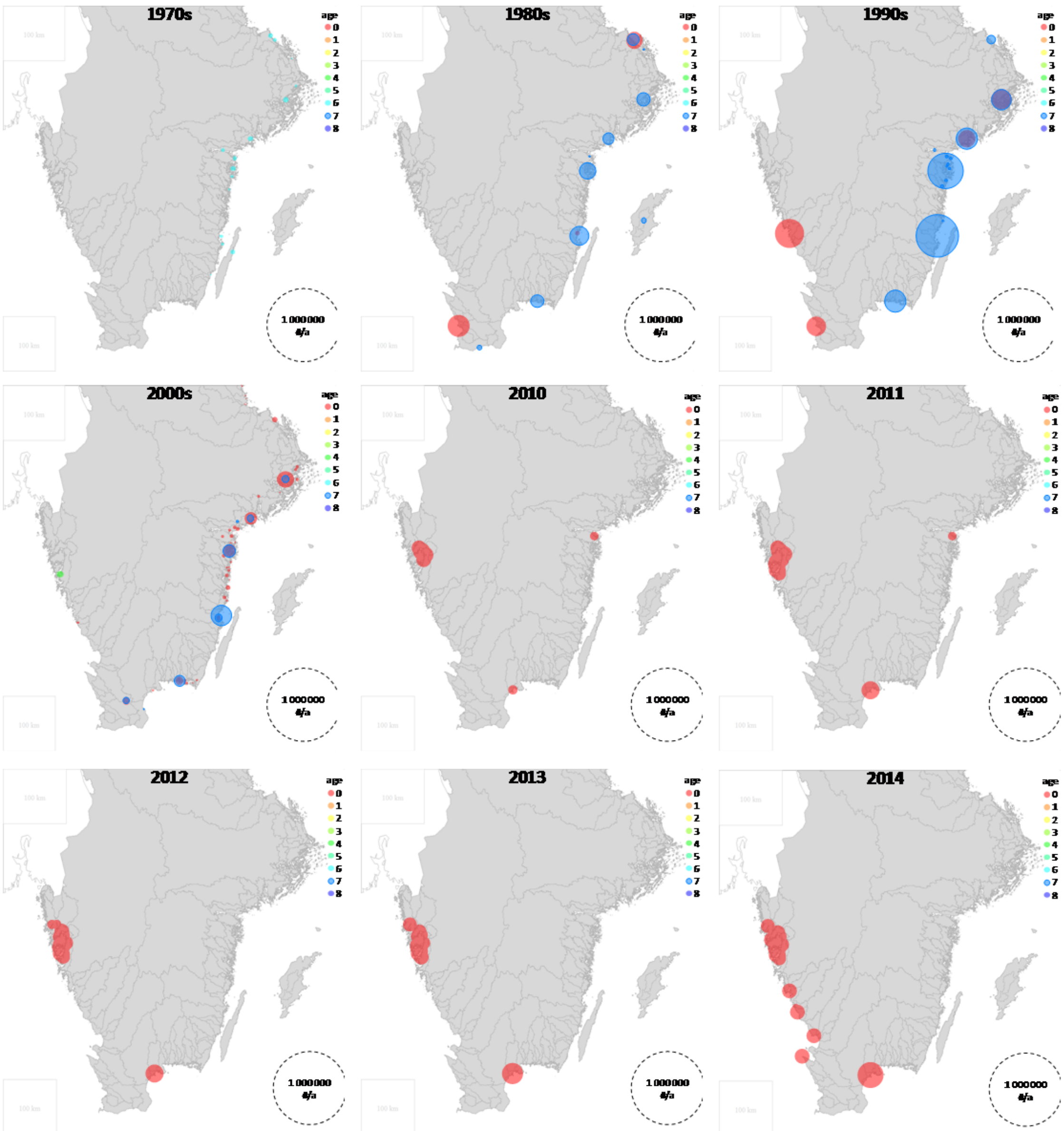


Figure 10 Spatial distribution of the restocking applied on in coastal waters, expressed in glass eel equivalents per year, for decades (1970s – 2000s) or individual years (2010 - 2014). Before 1970, no eel has been restocked on the coast. The colour of the symbols indicates at what age the eels were restocked, though all age groups 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 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).

B.1 Data

A database of historical trapping, transporting and releasing of elvers across barriers in rivers is held at SLU-Aqua, specifying site, year, quantity caught per year (number and/or biomass per year). For years when only the biomass of the elvers 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 (Figure 11). Additionally, an estimate of the mean age of the elvers was derived from the observed mean weight; the length-weight relation; and the average growth rate (see Annex C).

Data series from 24 different trap locations are available (Figure 13), and releases from these traps have been made at more than 160 locations. Individual data series <u>start</u> in-between 1900 (river Göta Älv, though the elver trapping started earlier) and 1991 (river Kävlingeån) and <u>stop</u> in-between 1975 (river Ljungan) and today (12 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. In this Annex, the trapping data are analysed.

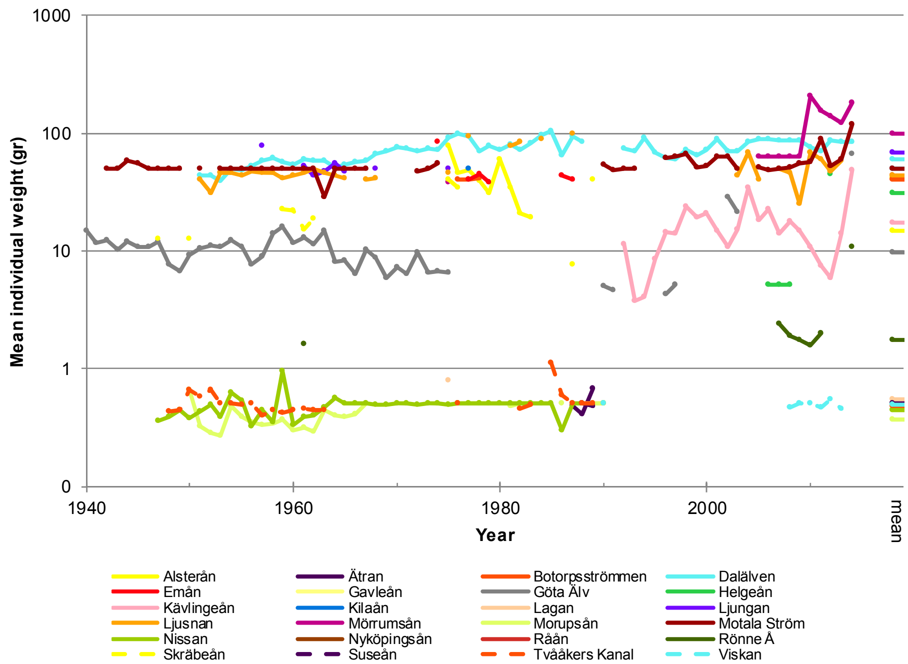


Figure 11 Mean individual weight of eels trapped for assisted migration, per year and river. To the right of the plot, the average per location over all observed years is given. In cases when the total number trapped was not recorded, but the total biomass was, numbers were reconstructed using these means.

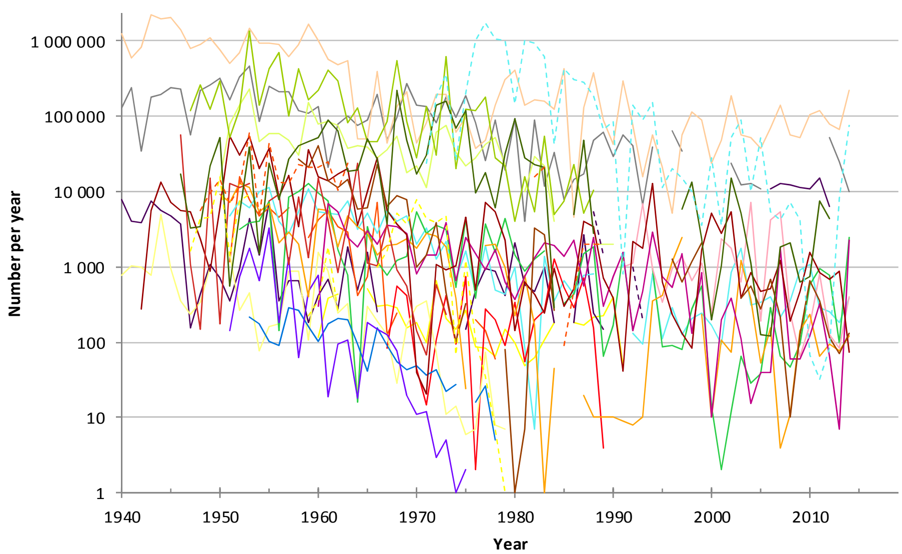


Figure 12 Trends in the number of elvers trapped at barriers, in numbers per year. The location of the traps is identified in Figure 13; the colours in this graph match those in the other, location-specific figures. Note the logarithmic character of the vertical axis. Legend as in Figure 11.

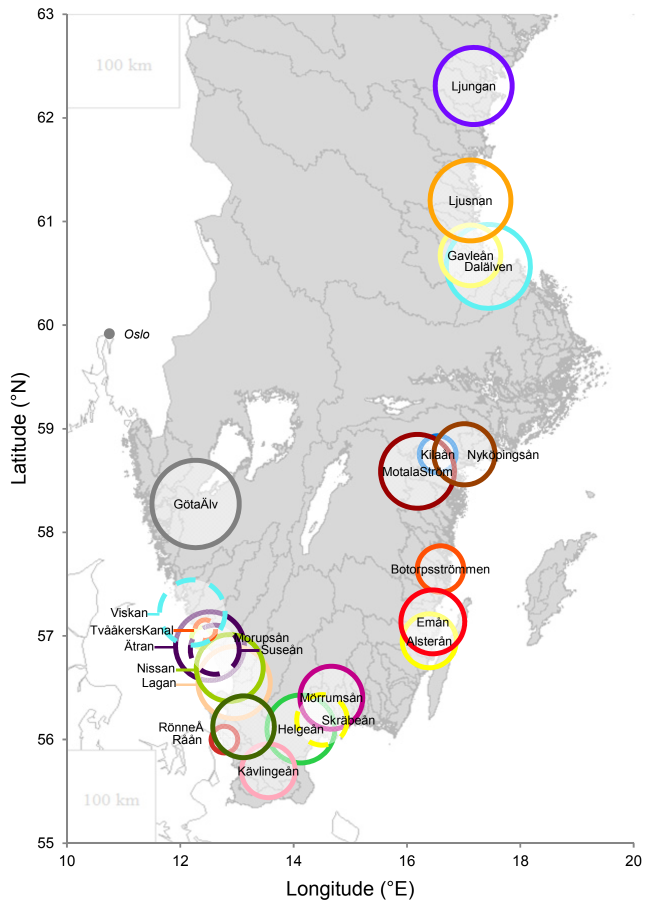


Figure 13 Locations of the elver traps. The size of the symbols is proportional to the logarithm of the river discharge at each location; the colours match those in the other, location-specific figures.

Characteristics of the trapping locations include: latitude, longitude, the distance into the Baltic Sea (calculated as the shortest route around the coast from the river mouth to the city of Oslo, in km), and finally the distance upstream where the trap is placed (km). Mean annual discharge data (m^3/s) for each river were derived from the Swedish meteo office (SMHI 2014).

The different sites capture different sizes of eel: from young-of-the-year on the west coast, to on average five-to-seven year old elvers (ca. 40 cm length, 100 gr individual weight) in the Baltic (Figure 16). Though sampling series started in very different years, sites catching small (<10 gr), medium and large (>30 gr) elvers have been operated throughout all decades.

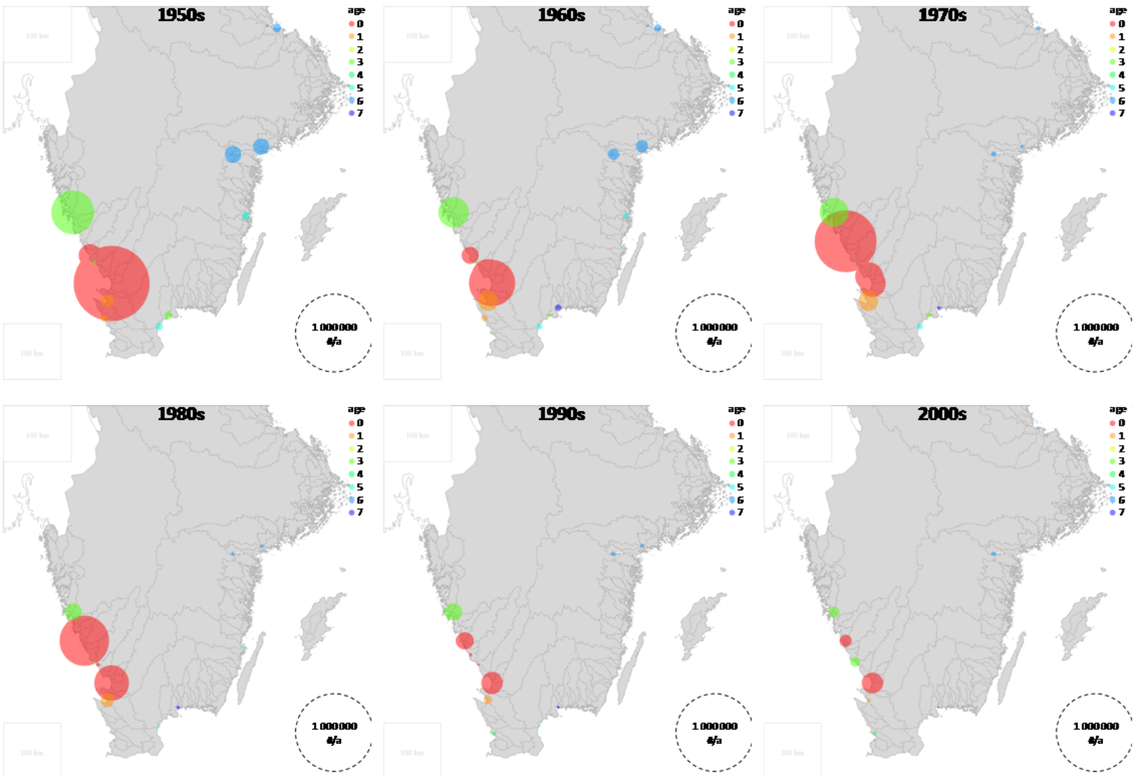


Figure 14 Spatial distribution of the observed numbers of elvers caught in the traps, averaged per decade, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class. In later decades, the numbers at many locations are that low, that the symbols are invisible in these maps.

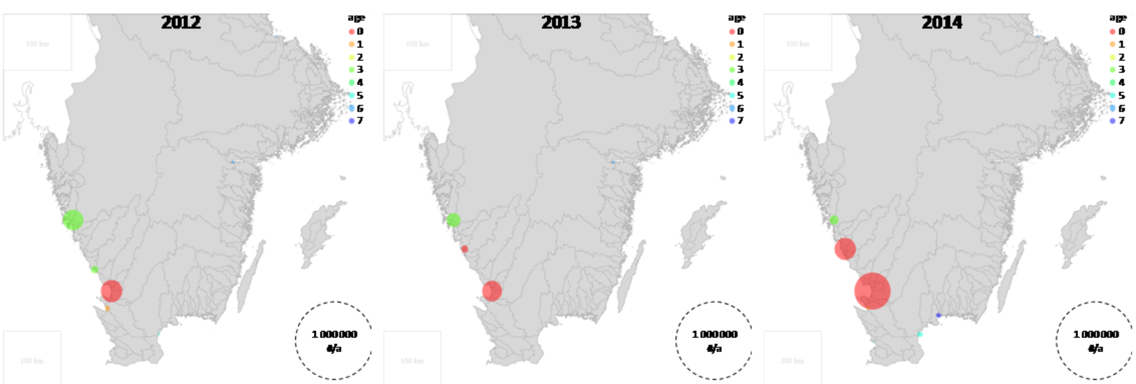
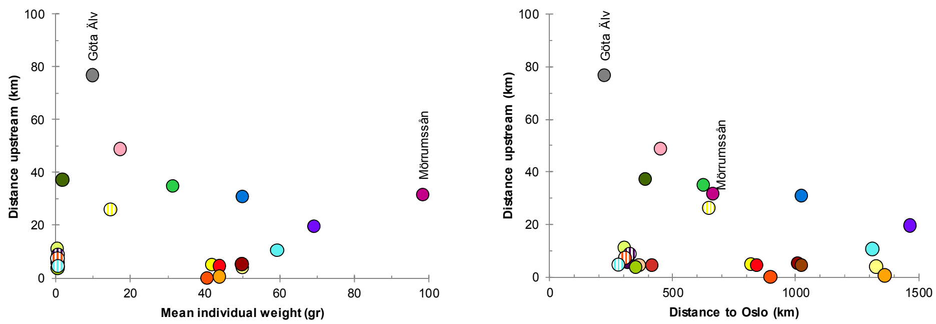


Figure 15 Spatial distribution of the observed numbers of elvers caught in the traps, in the years 2012-2014, expressed in glass eel equivalents per year. These figures are sorted by the year in which the immigration took place, not by year class. The numbers at many locations are that low, that the symbols are invisible in these maps.



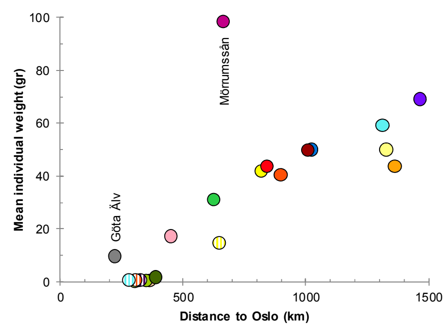


Figure 16 Observed relations between the mean size of the elvers (averaged over all observed years) and the location of the trap, both within the river (distance upstream) and along the coast (distance from Oslo). The colours match those in the other, location-specific figures. Two relative outliers, Göta Älv and Mörrumsån, have been labelled explicitly.

B.2 Spatial and temporal patterns in recruitment

Most time series of glass eel recruitment in Europe are closely correlated in time (Dekker 2000), though the decline since 1980 was a bit steeper in the North Sea area than along the Atlantic coasts (ICES 2014). The trends for bootlace eel and elvers, however, were quite different from those for the glass eel: the downward trend started much earlier (in 1960 or before) and the decline occurred more gradually (Svärdson, 1976; Dekker 2004b; ICES 2014). A number of hypotheses explaining the difference in trends between glass eel and bootlace have been raised:

a. Svärdson (1976) suggested that glass eel immigration into the Baltic might have declined earlier than elsewhere. Because most bootlace monitoring in Europe takes place in the Baltic area, the spatial pattern shows up as a size-related pattern in the international data. If so, all time-series in the Baltic will show an earlier decline, irrespective of the size of the eel and the location of the trap.
b. Dekker (2004b) discussed what processes could explain the observed decline in medium-sized eel in Lake IJsselmeer at a time that glass eel immigration into Lake IJsselmeer was as abundant as before, and suggested a gradually increasing natural mortality in the young stage. The older the eel in the surveys in Lake IJsselmeer, the

earlier the decline started, and the further the decline had progressed. The observed size-related pattern could be related to an increasing mortality in the yellow eel stage, caused by an unidentified process. If such a process operates in the Baltic too, the recruit series of older/larger eels in the Baltic will have declined earlier than the younger/smaller ones, irrespective of the location being monitored.

c. Sjöberg (2015) hypothesised that migration of young eels into the Baltic might be a density-dependent process, in the sense that the West coast is populated first, and only excess recruitment moves on into the Baltic. If so, the recruit series further into the Baltic will have declined earlier/more, irrespective of the age/size of the elvers.
d. Sjöberg (2015) further hypothesised that the decline might affect the upriver migration, in the sense that coastal habitats might be preferred, and only the remaining recruits migrate into the rivers. If so, the elver traps further upstream will have shown an earlier/stronger decline, irrespective of the distance into the Baltic and/or the size of the elvers concerned.

In analysing the available information from the elver traps, a model is applied that accommodates for each of the above hypotheses. In particular, we will fit a flexible time trend (a), differentiated by age/size (b), which allows for an earlier decline further into the Baltic (c) or further upstream (d). To this end, the data are analysed by a Generalised Additive Model GAM, in which time-trends are represented by a smooth function over the yearclasses, differentiated or not by age; density-dependent effects are covered by an additive model with multiplicative interactions. These models are detailed below.

B.3 Analysis

For each observation (one site in one year), the number of elvers was converted to the equivalent number of glass eels of the corresponding yearclass:

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

where year = the year the observation was made, age = the mean age at each site, and M = natural mortality between the glass eel and the elver stage. For M, an average value of 0.10 per year was assumed (see also the discussion on M in section C.2.3). Age was estimated from the average observed mean weight (see Figure 16, and the discussion of growth and weight in Annex C). The conversion to glass eel equivalents enables the comparison between differently aged elvers coming from the same yearclass (e.g. 6-years-old elvers in 2006 will be compared to 2-years-old elvers from 2002, instead of to 2-years-old elvers from 2006). The correction for natural mortality in the elver stage standardises the observations on a common unit (numbers of glass eels), but it will not affect the results any further, since *age* is included as an explanatory variable in all analyses.

These data were analysed using 'proc GAM' of SAS/STAT software Version 9.4 of the SAS System for Windows (SAS 2014).

The number of glass eel equivalents was log-transformed, enabling analysis by an additive model, and normalising the error-distribution. Proc GAM can handle non-normal data and non-linear relations without transformation of the observations, but the combination of a Gamma distribution (fitting our observations best) and a multiplicative model (in line with the hypotheses) is not enabled (a gamma error goes with a negative reciprocal link). Therefore, a transformation of the dependent variable was preferred. No true zero-observations occur in the database; apparently, sampling is stopped before catch numbers actually decline to zero.

The general form of the model reads:

$$\log\left(glass\ eel\ equivalents_{yearclass,site,i}\right) \\ = \alpha + \beta\ \times covariates_{site} +\ spline(age_{site}, yearclass_i) + \varepsilon_i$$

where

yearclass	the year in which the elver recruited as a glass eel
site	the site at which the observation was made
i	the observation serial number
α and β	model parameters, estimated
spline()	a smoothing function, estimated
age	the age, estimated from the average weight of the elvers at each site
ε_i	the error term of observation *i*, from a normal distribution
covariates	explanatory variables, including any or all of *logDischarge*, *upstream*, and *Oslo*.
logDischarge	the mean annual discharge for the river (m^3/s), derived from SMHI (2014); log-transformed
upstream	the distance from the river mouth to the elver trap, in km.
Oslo	distance from Oslo to river mouth, shortest route (convex hull) around the coastline, in km.

The smoothing function is estimated either by a Cubic Smoothing Spline (univariate) or by a Thin-Plate Smoothing Spline (bivariate), determining the degrees of freedom (the degree of smoothing) on Generalised Cross Validation GCV. The GCV-method will automatically select the smoothest function (lowest number of degrees of freedom) adequately fitting the data. These are the default options.

For each site and year, only a single observation is available (the annual total number). Testing interaction terms of *yearclass* (75 levels) with each of the variables *age*, *upstream* and *Oslo* is no option, since that would drain the information available in the data rapidly (it would exhaust the degrees of freedom). For the interaction of *yearclass* with *upstream* resp. *Oslo*, the basic model was extended by "Mandel's bundle of straight lines" (Mandel 1959; Milliken and Johnson 1989), testing the interaction between *upstream* resp. *Oslo* and the strength of the yearclass (continuous), rather than an indicator of the yearclass itself (class variable). This adequately represents the hypotheses c. and d., as specified in the section above, which relate to the strength of any yearclass, rather than to the temporal order of the yearclasses themselves.

For "Mandel's bundle of straight lines", estimates of the yearclass strengths were derived from an initial model run, regressing the observations on *logDischarge*, *upstream*, *Oslo*, and the sum of two univariate splines, for *age* and *yearclass* (but no interaction *age*yearclass*, no bivariate spline). This two-step procedure (fitting a preliminary model, followed by a final model using estimates from the first) is in line with the procedure described in Mandel (1959) resp. Milliken and Johnson (1989).

For the presentation of results in graphs, the estimates of the time trend (Figure 17) were subdivided into ten yearclass-strength-classes of equal size (scaled between their minimum in 2011 and maximum in 1945). In Figure 17, each of these classes is indicated by a different colour. Below (in section B.4), partial predictions and partial residuals will be presented (Figure 18, Figure 19 and Figure 20). Those plots will use the same colour coding as Figure 17. That is: the colours of the points and lines in those figures represent the strength of the yearclass for each data point.

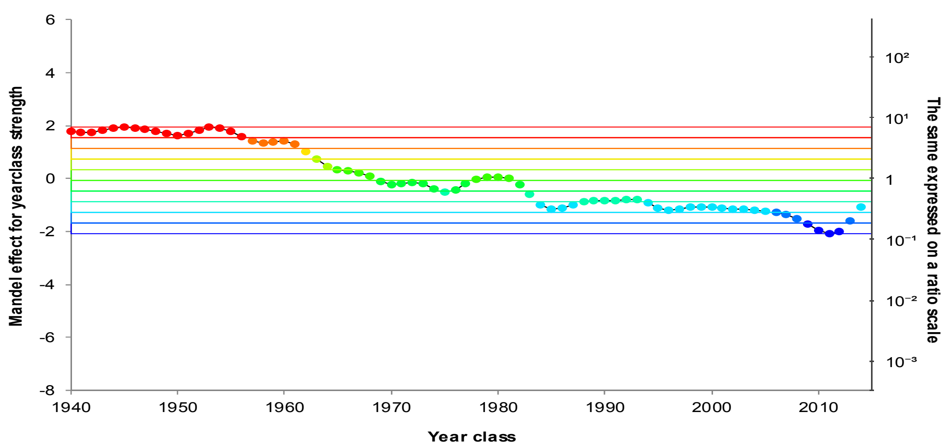


Figure 17 Initial estimates of year class strength, derived from a main-effects-only model.

For the interaction between *yearclass* and *age*, a bivariate spline was applied. This adequately represents the hypotheses a. and b., as specified in the section above, which consider time-trends (in yearclass strength, resp. in elver mortality), but do not necessarily relate to the strength of the yearclasses themselves.

For 60 main rivers south of 62.5°N (Indalsälven) and all years since 1940, a statistical prediction was estimated for the number of elvers that could be caught in a trap – for rivers *with* or *without* an actual trap. Since the statistical model includes the distance from the dam where the trap is located to the river mouth, the lowest dam in each river was identified; the statistical prediction of natural recruitment thus relates to the stock arriving at the first dam. For an additional 35 smaller rivers where no dam was 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).

For each of the explanatory variables *discharge*, *Oslo*, *upstream* in interaction with yearclass strength *Mandel*(*yearclass*), and for the interaction *yearclass*×*age*, plots of partial predictions and partial residuals are shown below, where

$$partial\ prediction = \beta \times covariate_i$$

$$partial\ residual \quad = \beta \times covariate_i + \varepsilon_i$$

and *covariate* indicates <u>one single</u> explanatory term , as plotted on the horizontal axis.

The model described here deviates slightly from the one used in Dekker & Wickström (2015), in which the trend in elver mortality was represented by a parametric model rather than a GAM model. Fitting *yearclass* as a class variable (df=75) in interaction with *age* (continuous) almost identical predictions. Results were not or just marginally statistically significant, due the excessive number of degrees of freedom involved in this parametric model. To overcome that, the current non-parametric analysis was developed.

B.4 Results

B.4.1 Spatial trends in elver trap catches

The number of elvers trapped at barriers in the rivers varies widely, from one elver to over two million elvers per trap per year (*Figure 12*). The major part of that variation is related to the size of the river where the trap is located, i.e. *discharge* (Figure 18). From Morupsån and Tvååkers-Kanal (both ± 1 m^3/s) to Göta Älv (565 m^3/s), the number of elvers goes up by a factor of 200 on average. The regression results indicate that the number of elvers is proportional to $discharge^{0.885 \pm 0.036}$, that is: the elver catch is not exactly proportional to the size of the river, but almost so.

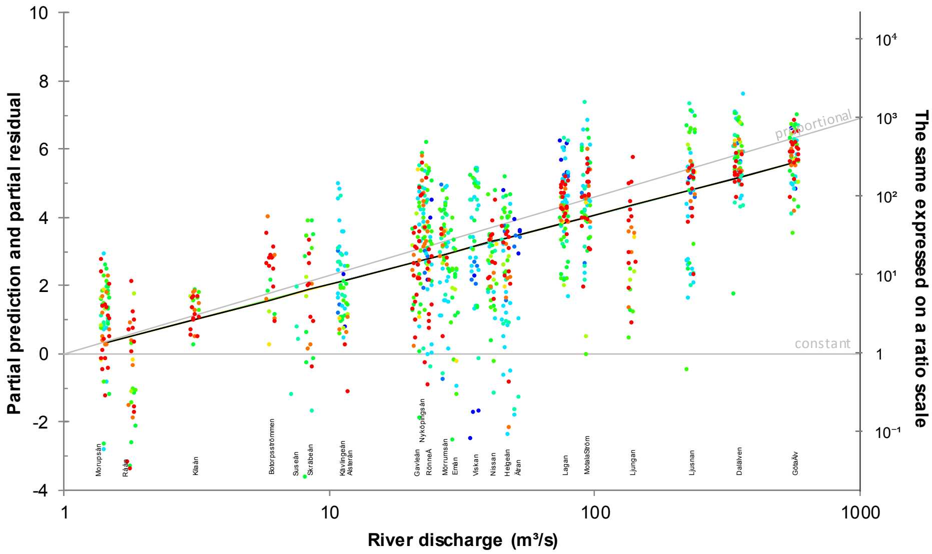


Figure 18 Partial predictions (the regression line) and partial residuals (dots, representing the observations after correction for the effect of other explanatory variables), regressed against the discharge of the river where the traps are located. To make individual dots distinguishable, a small jitter has been added to all data points in horizontal direction. The colours classify the strength of the year class concerned (see Figure 17), ranging from the strongest (red, in 1945) to the weakest year class (blue, in 2011). Two reference lines (in light grey) identify a situation in which there would be no relation between the elver catch and the river discharge, resp. a strictly proportional relation.

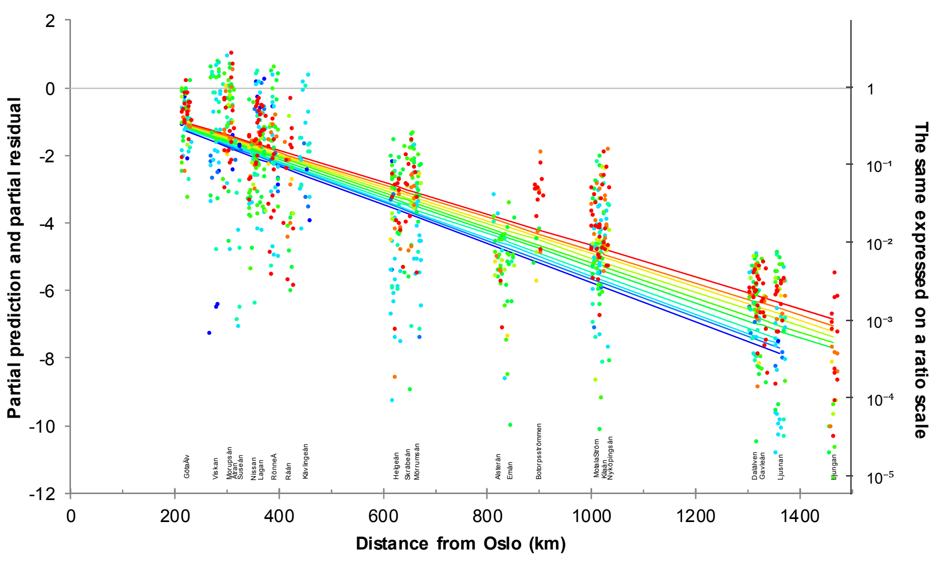


Figure 19 Partial predictions and partial residuals regressed against the distance along the coast, from Oslo to the river where the traps are located. For further details, see the caption of Figure 18.

For the distance-from-Oslo (Figure 19), observations range from Göta Älv (221 km from Oslo) to Ljungan (1464 km) – predictions differing by a factor of 330. Stronger year classes (red line and dots in Figure 19) are relatively more abundant far into the Baltic than weak year classes (blue line) - nearly 5 times more abundant. This is in line with the hypothesis of potentially density dependent migration into the Baltic.

For four smaller rivers close to the Norwegian border (Strömsån 73 km; Enningsdalalven 88 km; Örekilsälven 140 km; and Bäveån 176 km – in combination 0.7% of the total drainage area, 1.1% of the discharge), an extrapolation is required beyond the observed data range. Because of their proximity to Oslo, the predictions for these four rivers would sum to an equal amount of elvers as that for all other 57 rivers together. Because of the uncertainty in extrapolations, these four rivers have been excluded from the assessment completely. The reconstruction in this Annex thus applies to the area above the first dam in each river in inland Sweden, except for these four rivers.

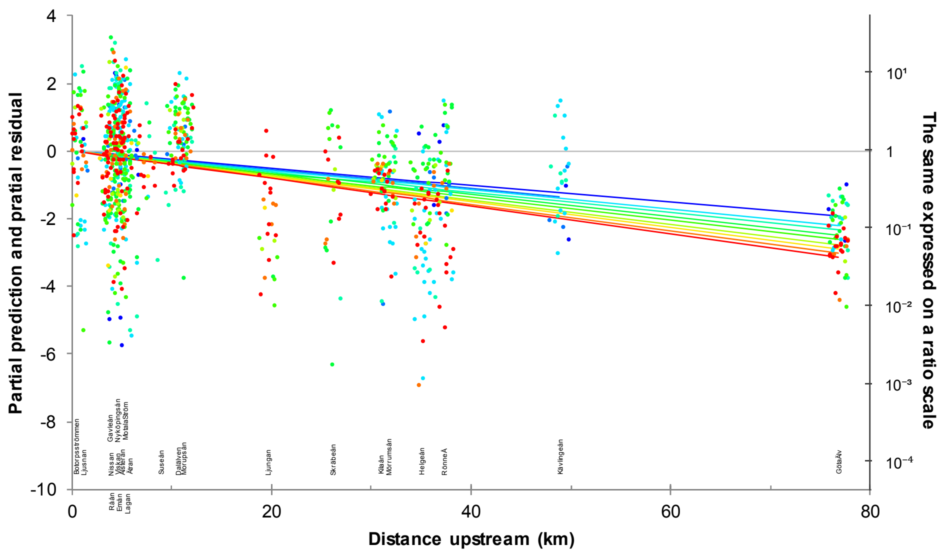


Figure 20 Partial predictions (the regression lines) and partial residuals (dots, representing the observations after correction for the effect of other explanatory variables), regressed against the distance from the mouth of the river to the traps. To make individual dots distinguishable, a small jitter has been added to all data points in horizontal direction. The colours classify the strength of the year class concerned (see Figure 17), ranging from the strongest (red, in 1945) to the weakest year class (blue, in 2011).

For the distance between the trap and the river mouth, results are dominated by Göta Älv (Figure 20), the only trap at more than 50 km upstream. The regression lines indicate that trap catches are ± 75% lower there, than at the river mouth. Comparing the effect of the distance upstream between strong (red) and weak (blue) year classes indicates that the weaker year classes were relatively more abundant far upstream – contradicting the expectations of potentially density-dependent up-river migration.

B.4.2 Temporal trends in elver trap catches

The estimated time trends differ considerably, depending on the mean age of the elvers being trapped (Figure 21). For the oldest recruits (age 7, blue line), a declining trend was estimated from 1950 until 1970 (declining by a factor of 8), followed by a somewhat varying low level, moderately declining since. For age 0 (red line), however, estimates declined from 1950 to 1970 by only a factor of 2, showed a steep decline from 1980 to 1990 by a factor 6, and a slow decline by a factor 2 since. In 1977, the 7-year old elvers had declined a factor 12 *more* than the 0-year old ones. Over the whole time interval, the range of yearclass strengths for the 7-year old elvers is spanned by a factor 11, while for the 0-year old elvers, the ranged is spanned by a factor 56 – that is: the youngest ones declined more. Intermediate age groups show intermediate results, both in the level of decline, as in the period when that occurred: the older the elver, the earlier the decline. This pattern does not fit Svärdson’s hypothesis that all glass eel recruitment into the Baltic had started to decline long before that happened to the rest of Europe; the major decline in the youngest elvers occurred in the 1980s, as was the case in the rest of Europe. The earlier decline of the older elvers does fit Dekker’s (2004b) hypothesis of increased mortality in the yellow eel stage, but the levelling-off for the oldest ages after 1980 then indicates that mortality is no longer raised.

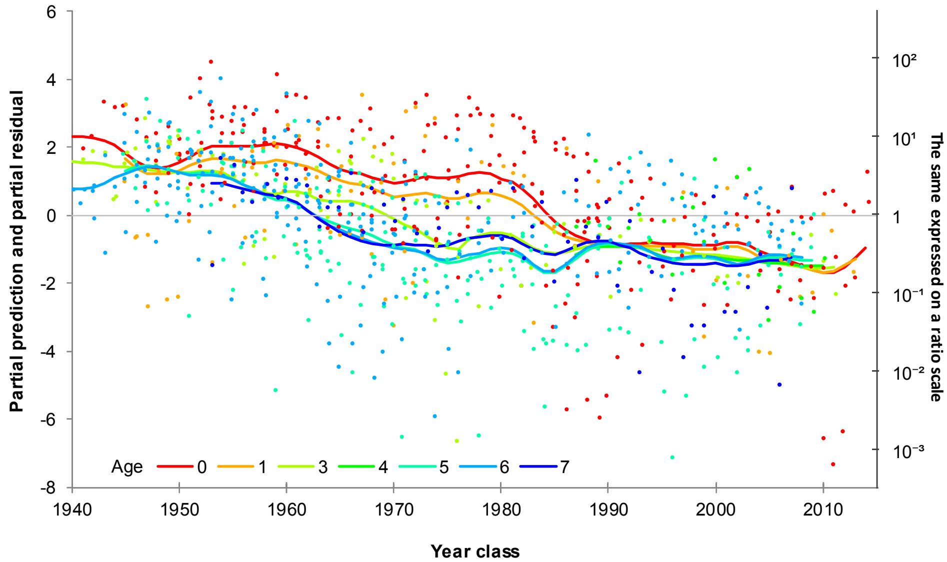


Figure 21 Partial predictions (smooth regression lines) and partial residuals (dots, representing the observations after correction for the effect of all other explanatory variables), regressed against the interaction of *year class* (horizontal axis) and *age* (colour), by means of a bivariate spline. To make individual dots distinguishable, a small jitter has been added to all data points in horizontal direction. *Note*: the colours in this plot do *not* represent the strength of the year classes (as in Figure 18 to Figure 20), but the mean age of the elvers being trapped.

B.5 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 was used to generate statistical predictions for all rivers in all years. Although the results of the analysis in this Annex favour some hypotheses more than others, the predictions used in the reconstruction of silver eel production (Annex C) are those based on the full model. Details will be presented in Annex C.

The references for this Annex are included in the reference list of the main report, on page 38.

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 reliable in all detail. Production estimates for individual lakes in specific years will certainly be much less reliable than nation-wide estimates, or decadal averages, and so forth. Hence, the presentation of results will be restricted to nation-wide averages and/or decadal means.

C.1 Data and methods

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

C.1.1 Inputs to the inland stock

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

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

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

where year = the year the observation was made, age = the mean age of the eels, number is the number of recruiting eels, and M = natural mortality between the glass eel and the immigrating stage. For M, an average value of 0.10 per year was assumed (the same value as used in the remainder of the analysis; when testing different values of M, the conversion to glass eel equivalents was adapted accordingly). This standardises all data sources of young eel 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 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

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. 35 t. (see below), ignoring these smaller rivers introduces a bias of approximately 3% of 35 t. ≈ 1 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, 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.

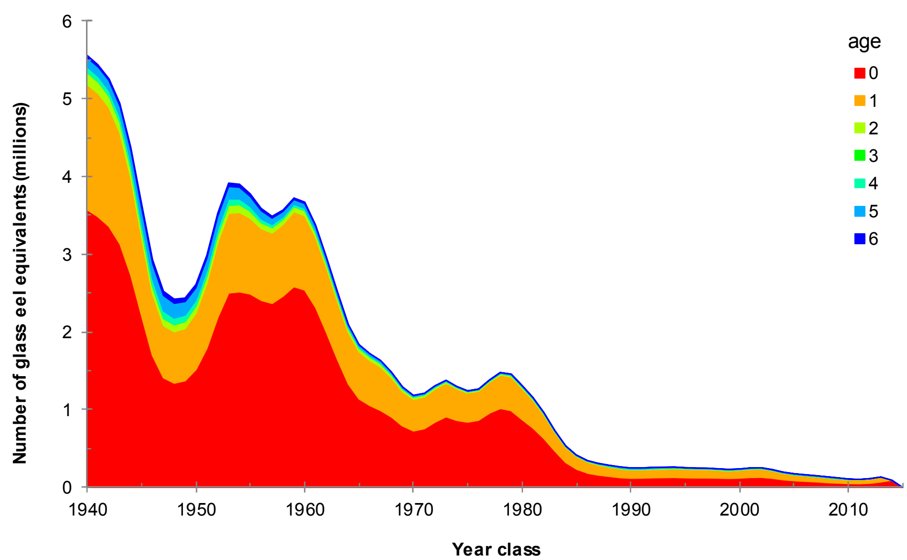


Figure 22 Time trend in the estimated number of <u>naturally recruiting eels</u>, expressed as glass eel equivalents per year class.

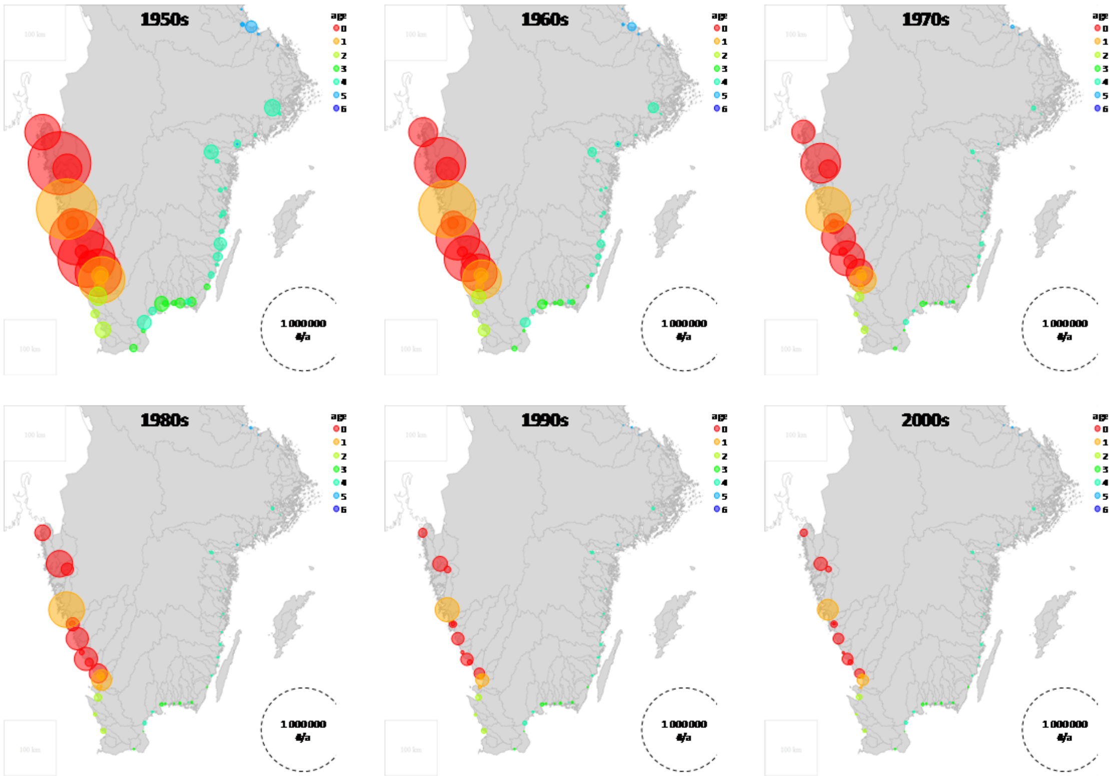


Figure 23 Spatial distribution of the estimates of natural recruitment, per decade, expressed in glass eel equivalents. These plots show the total number per decade (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the immigration took place, not by year class.

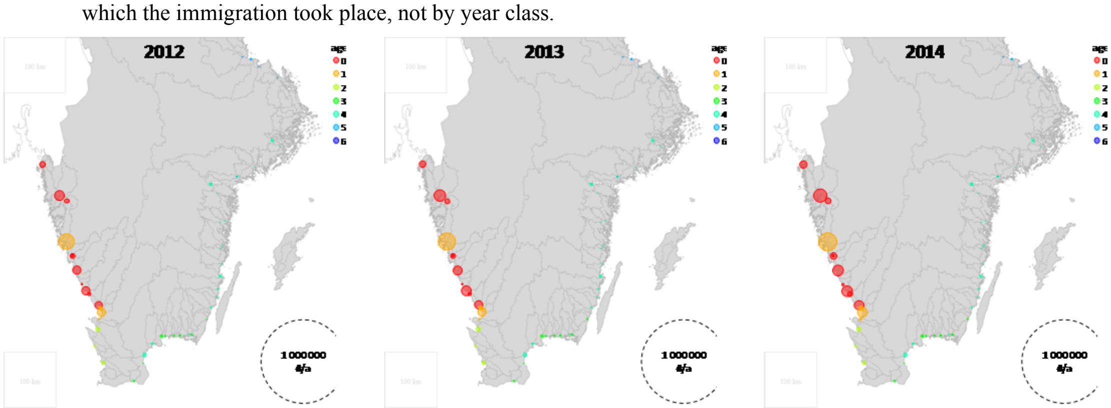


Figure 24 Spatial distribution of the estimates of natural recruitment, in the years 2012-2014, expressed in glass eel equivalents. These plots show the total number per year (as predicted by the model of Annex B), plotted at the location of the lowest barrier in each river. Note that these figures are sorted by the year in which the release 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 per year). For years 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 (Figure 11). Additionally, an estimate of the mean age of the immigrating eel was derived from the observed mean weight, the length-weight relation and the average growth rate (see p. 73).

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) – most 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 also in other river systems.

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 (12 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.

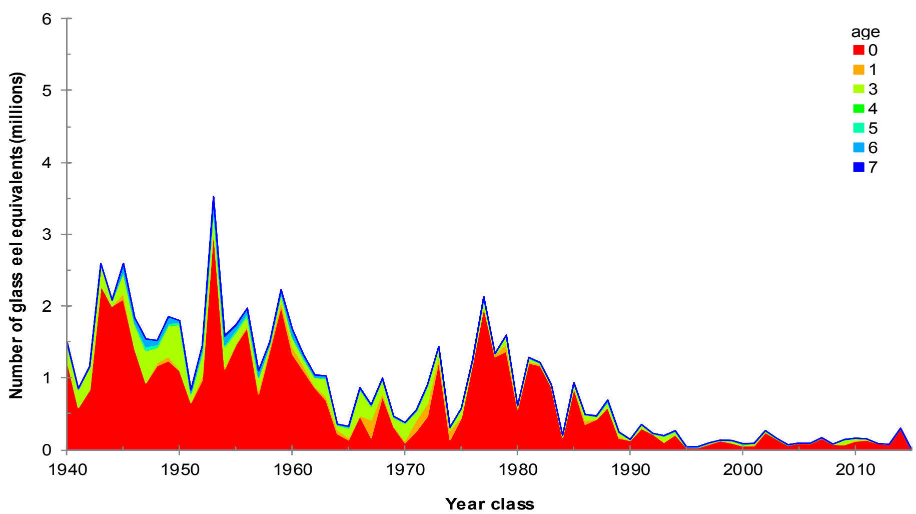


Figure 25 Time trend in the number of eels released from assisted migration. Though this plot is subdivided by age of the eel, all quantities are expressed in glass eel equivalents per year class.

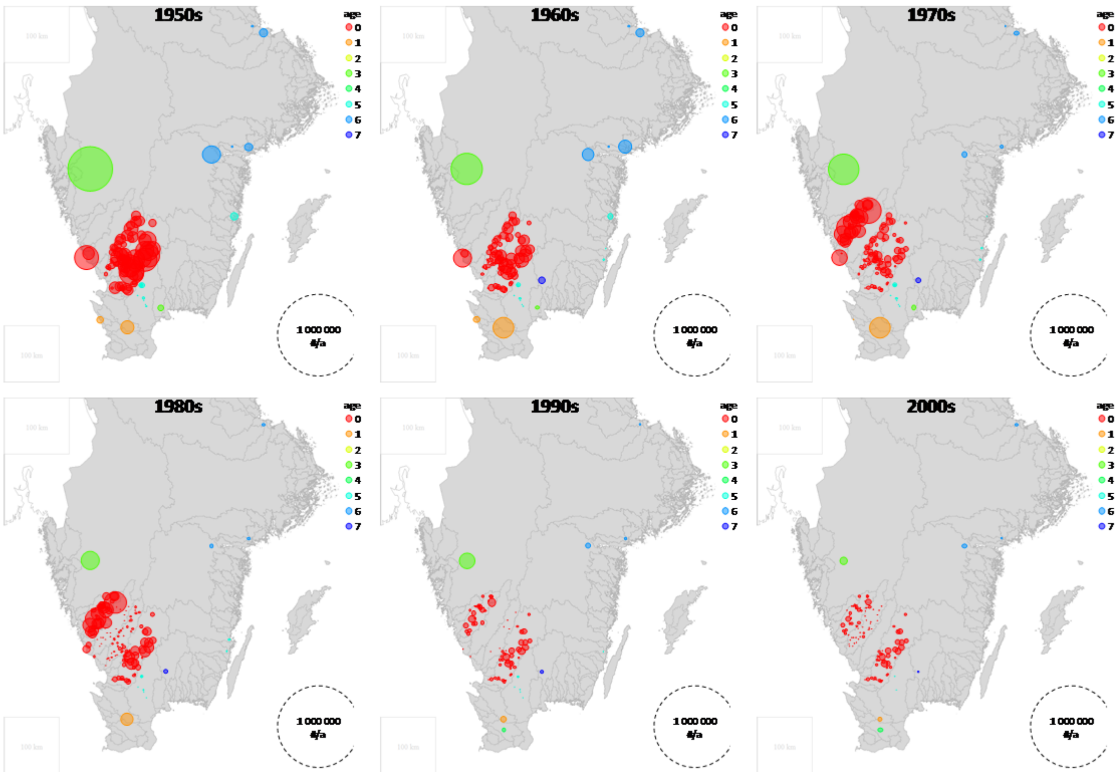


Figure 26 Spatial distribution of the release from assisted migration, per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that the figures are sorted by the year in which the release took place, not by year class.

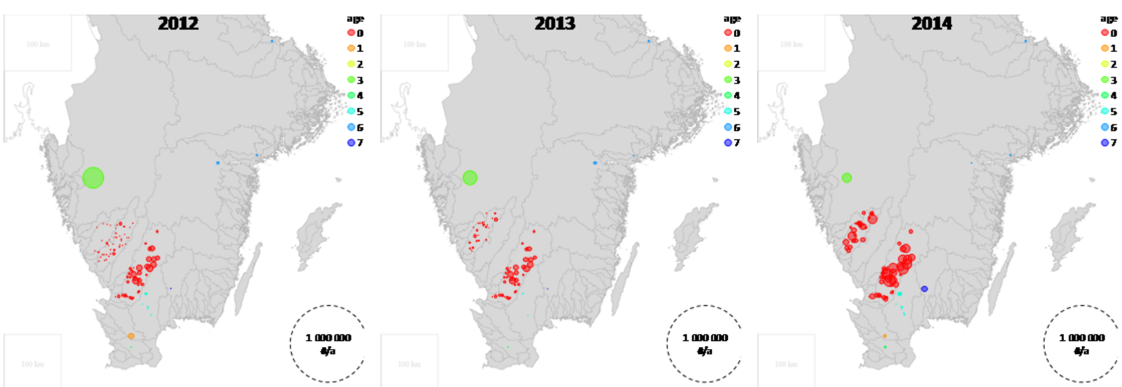


Figure 27 Spatial distribution of the release from assisted migration, in the years 2012-2014, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the release took place, not by year class.

Restocking

A data base of eel restocking 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.

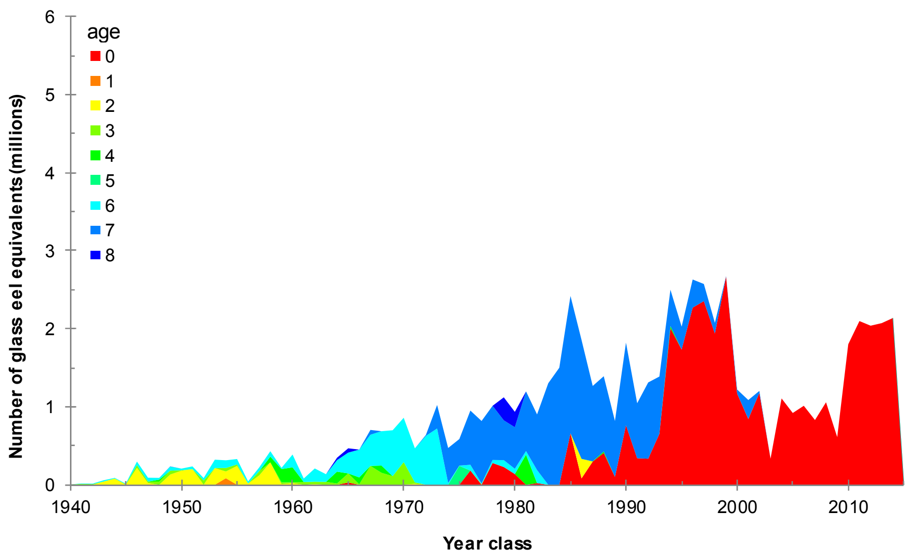


Figure 28 Time trend in the numbers of eel used for restocking. Though this plot is subdivided by age of the restocking material, all quantities are expressed in glass eel equivalents per year class.

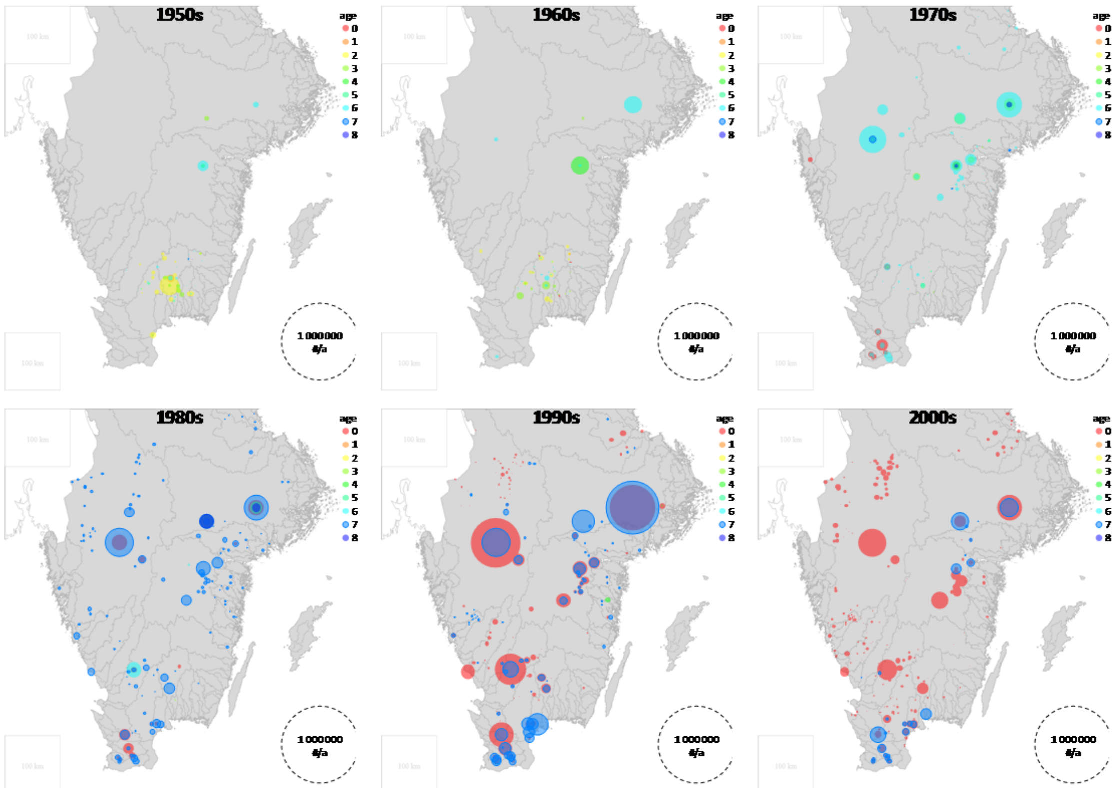


Figure 29 Spatial distribution of the restocking per decade, expressed in glass eel equivalents. These plots show the total number per decade. Note that these figures are sorted by the year in which the restocking actually took place, not by year class.

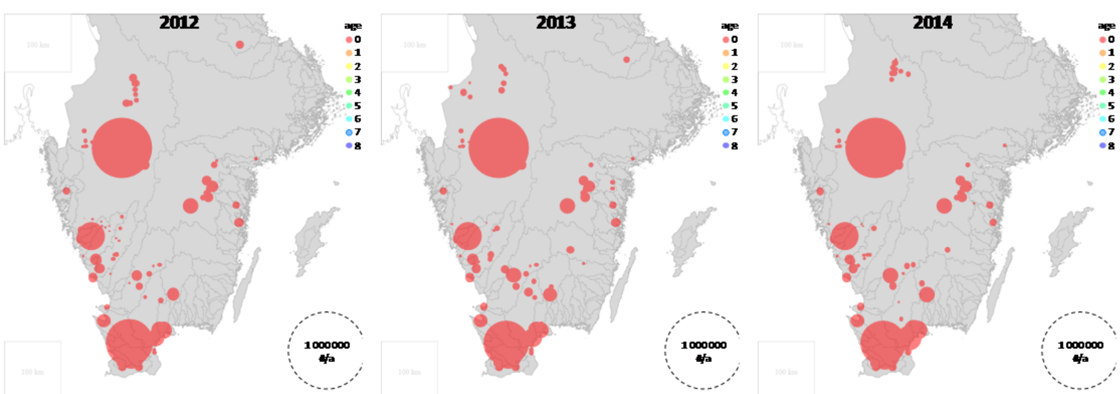


Figure 30 Spatial distribution of the restocking in the years 2012-2014, expressed in glass eel equivalents. These plots show the total number per year. Note that these figures are sorted by the year in which the restocking took place, not by year class.

Trap and 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 6 *Quantities of silver eel applied in the Trap & Transport programmes, in numbers and biomass (kg)*

Year	2010		2011		2012		2013		2014	
River	Number	Biomass	Number	Biomass	Number	Biomass	Number	Biomass	Number	Biomass
Motala Ström			545	676	928	1 283	2 526	3 167	3 788	4 708
Mörrumsån			1 613	1 883	135	154	212	269		
Rönne Å									733	415
Lagan	422	365	652	367	72	110	931	921	1 445	1 484
Ätran					369	253	120	82		
Göta Älv	4 582	4 841	4 243	4 499	7 790	8 237	9 024	9 393	12 335	12 417
Total	5 005	5 206	7 053	7 425	9 293	10 037	12 812	13 832	18 300	19 024

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 31). Elsewhere, data are available per lake and/or for varying groups of lakes (Figure 32). In summing across lakes, one has grouped many different sets, sometimes even spanning different drainage areas. 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).

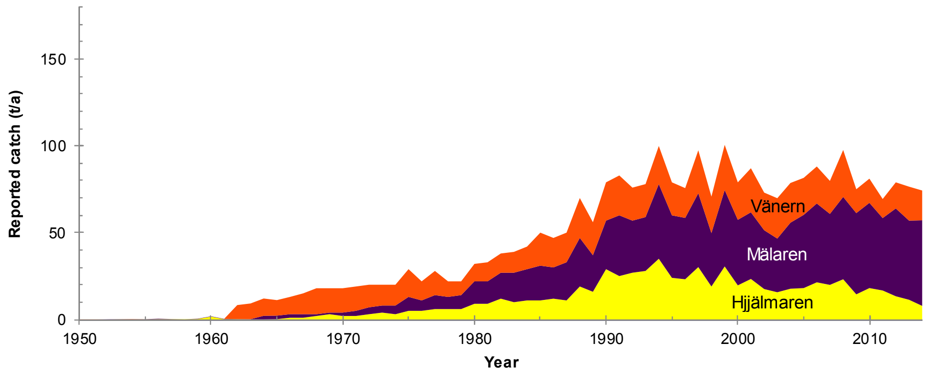


Figure 31 Time trend in the reported landings from the fishery, for the larger lakes, and years since 1950. For smaller lakes, data are only available since 1986.

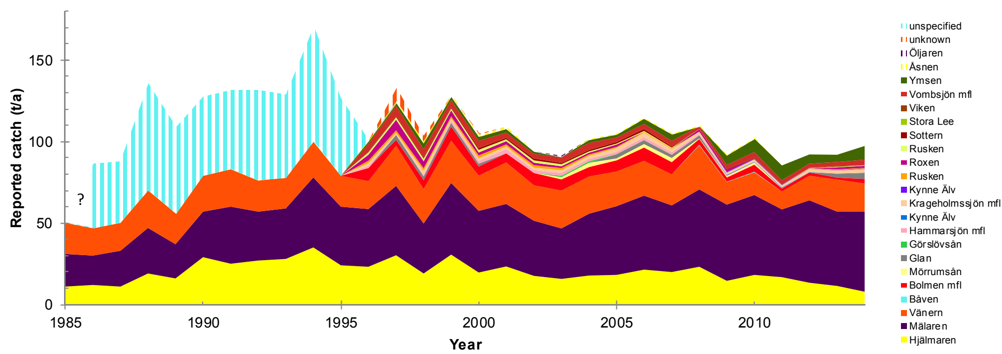


Figure 32 Time trend in the reported landings from the fishery, for all lakes, and years since 1985. Note the time interval on the horizontal axis, deviating from most other figures.

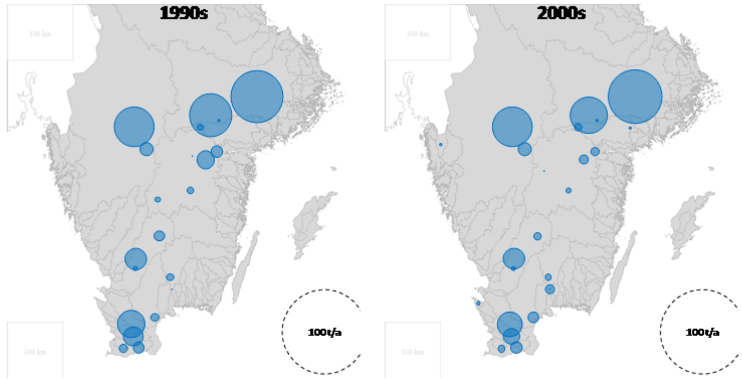


Figure 33 Spatial distribution of the reported landings from fisheries, in the 1990s and 2000s. For earlier decades, insufficient information is currently available.

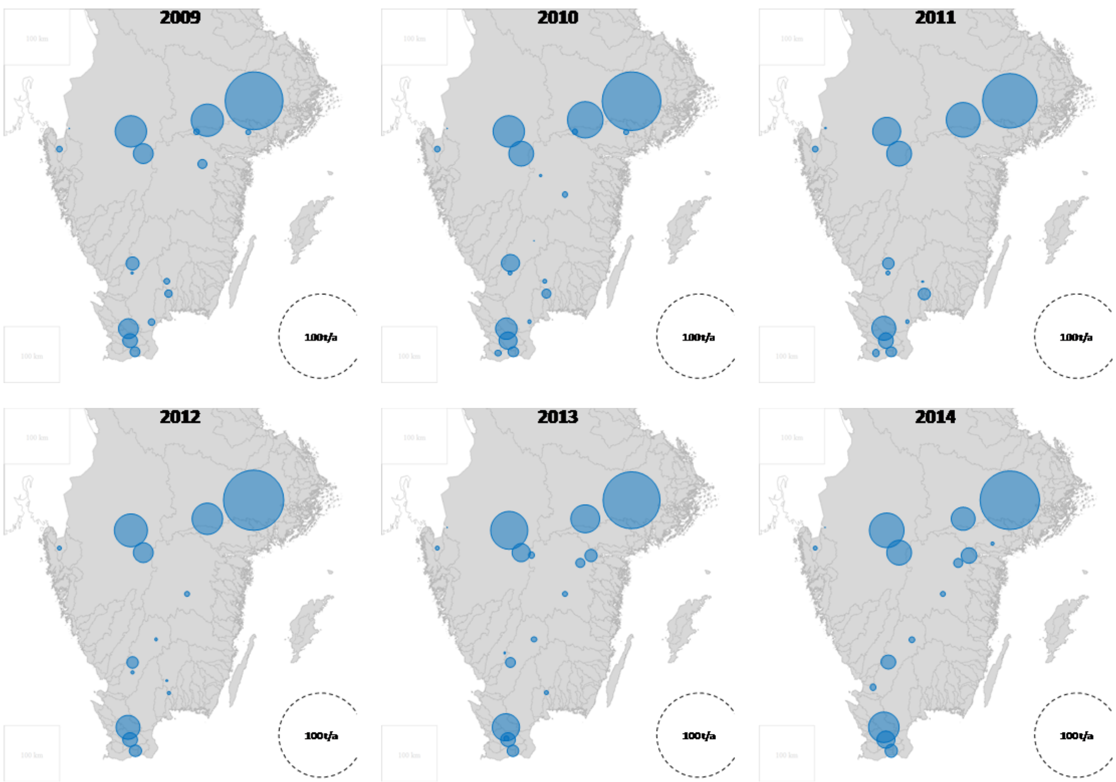


Figure 34 Spatial distribution of the reported landings from the fisheries, for the years since 2009.

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 (regleringsdam). 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, especially in recent years. For several lakes (e.g.: Båven, Glan, Roxen, Rusken, Sommen, Sottern; Figure 34), years with and without reported landings alternate. For other lakes, years with and without reported landings for individual fishers alternate (not shown), while the licensing system requires a continuous operation. Personal communication to individual fishers almost invariably yields more consistent information, higher landings figures.

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.

For the coastal fisheries, information on landings is reported in EU logbooks, in individual reports on a daily basis *(is that correct, daily?)*. Inspection of those data does not reveal quality issues of the kind observed in the data on inland fisheries, while the data are much more detailed than the inland data, and require much more effort from the fisher to complete the forms.

Noting the doubtful quality of the information on landings from inland waters, while a well-tried reporting system is available in the coastal fisheries, it is recommended to reconsider the system of reporting used for the inland fisheries, to consider the potential inclusion of the inland fisheries in the EU logbook system.

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 (2014), documenting location and year of construction (Figure 35). 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 (2014), 519 stations are relevant for the current reconstruction (eel occurring upstream).

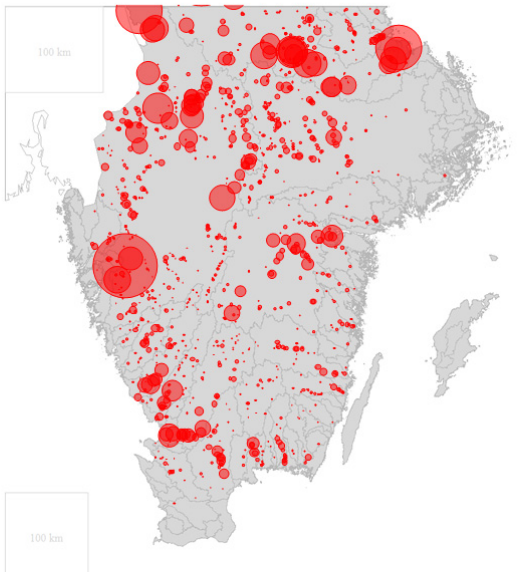


Figure 35 Spatial distribution of the 519 hydropower generation plants having an eel stock upstream. The size of the symbols in this figure is proportional to the capacity of each station.

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 36, 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, a range of options for the hydropower-related mortality is explored.

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%. In the current assessment, three options will be explored: 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).

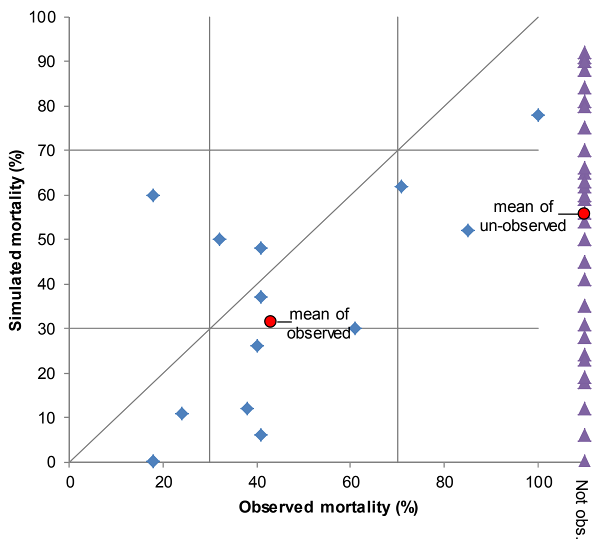


Figure 36 Relation between the observed (horizontal) and simulated (vertical) mortality, for eel passing a hydropower turbine. Data from Calles and Christianson (2012), applying the simulation model of Leonardsson (2012); original plot of data tabulated by the source.

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 2003 to 2013, samples have been collected from the commercial catch, predominantly from the larger lakes. These eels have been analysed for length, weight, maturity and age. In total, a number of 2 122 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 73 mm) divided by the number of years in-between (the age read). The data (Figure 37, drawn lines) indicate a large variation in growth rate between lakes, with the two most southern lakes (Bolmen and Ringsjön) showing the lowest growth rate. The variation in growth among the northern lakes, however, is of the same order as that between southern and northern lakes, while the variation in latitude for those northern lakes is small. We make the conservative assumption that growth is not related to latitude.

Comparing between sampling years (not shown) indicates very little temporal variation. Since our samples cover only ten years, while a period of 75 years is reconstructed, it is assumed that growth has been constant over the whole period.

In conclusion, we apply a constant growth of 44 mm/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*=0.000000444 and *b*=3.23.

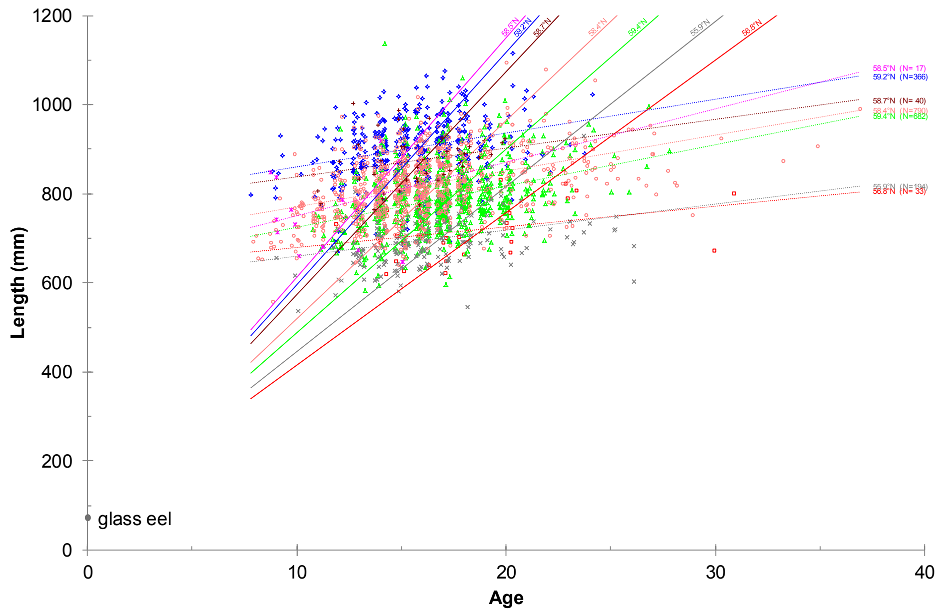


Figure 37 Length and age for 2 122 silver eels, sampled between 2003 and 2013 in 7 lakes. To show so many data points, a small jitter has been added to all data points in horizontal direction. For each lake, two regression lines are given: a growth-line (drawn) forced through the length/age of glass eel (73 mm at age=0), and an unforced silver-eel-size-line (dotted). For each lake, the latitude and the total number of observations is given.

Silvering

Sampling data indicate a latitudinal trend in mean size at silvering (Figure 37, dotted lines), from approximately 700 mm in the south (56°N) to 900 mm in the north (60°N). 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 men length per year of increment in age (on average 7.5 mm/year; Figure 37, dotted lines) is much less than the mean growth rate during the yellow eel stage of 44 mm/year (Figure 37, drawn); 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.

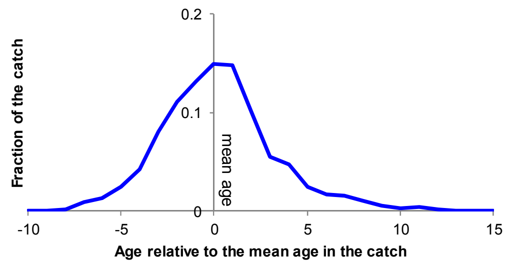


Figure 38 Relative age composition of the catches in inland waters, where age is expressed relative to the observed mean age.

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) are explored and the outcome discussed. 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 28 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. 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).

The analysis of recruitment trends (Annex B) took 1940 as its starting point. Most young eels 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 2014, with an extrapolation to 2030 to show the fate of the 2014 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 1940 until 2013, natural recruitment – including the amount assisted in their migration upstream - is estimated at a total number of 187 million glass eel equivalents, with a minimum of 198 thousand eels in 2011 and a maximum of 7.5 million in 1940. The corresponding silver eel production is estimated at 14 356 t, minimum 35 t/a, maximum 527 t/a. In 2014, 0.4 million glass eel equivalents were natural recruits. Total silver eel production from natural recruits (assisted or not) in 2014 is estimated at 35 t.

From 1940 until 2013, a total of 65 million eels have been caught for assisted migration upstream, with a minimum of 40 thousand of yearclass 1995 and a maximum of 3.5 million of yearclass 1953. The corresponding silver eel production is estimated at 8 339 t, minimum 20 t/a, maximum 290 t/a. In 2014, 0.3 million glass eel equivalents were assisted upstream. Total silver eel production from the 2014 assisted migration is estimated at 20 t.

From 1940 until 2014, a total number of 70 million glass eel equivalents has been restocked, with a minimum of 5000 glass eel equivalents for yearclass 1940 and a maximum of 2.7 million for yearclass 1999. The corresponding silver eel production is estimated at 7 091 t, minimum 14 t/a, maximum 308 t/a. Of yearclass 2014, 2.1 million glass eel equivalents have been restocked. The corresponding silver eel production (before fishery and hydropower impacts) is estimated at 295 t.

Overall silver eel production declined from approximately 500 t in the 1960s and 1970s, to 300 t/a at the end of the 1980s, and varied around 300 t/a since. 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 current production at a temporary maximum of 330 t/a; lower restocking in the early 2000s will reduce production to 200 t/a by 2020, but the 2014 level of restocking can be expected to restore total production to 320 t by 2030.

From 2010 until 2014, a total number of 52.5 thousand silver eels have been trapped and transported downstream, with a minimum of 4.8 thousand (5 t) in 2010 and a maximum of 18.3 thousand (19 t) in 2014.

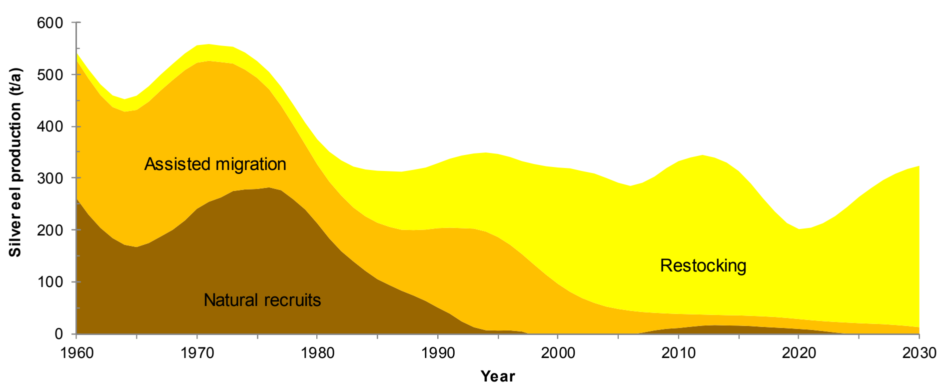


Figure 39 Production of silver eel by year and by origin of the eel, that is: the estimated total production before the impact of fishery and hydropower. For these results, a natural mortality rate of M=0.10 was assumed.

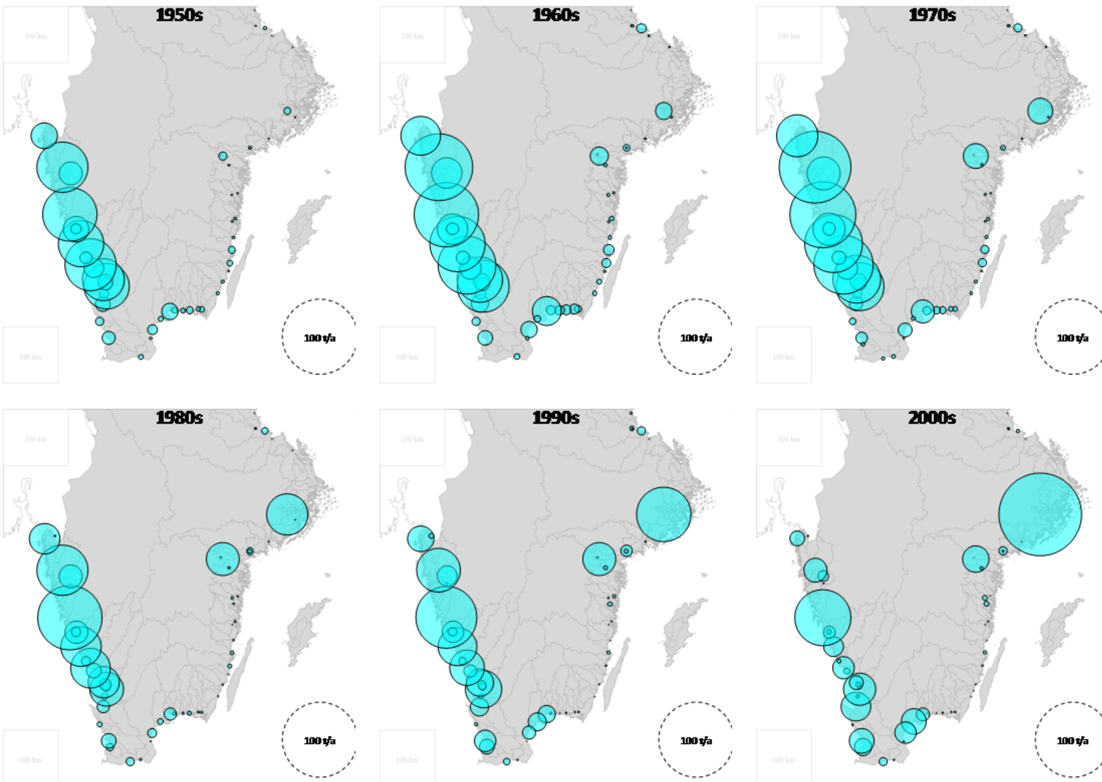


Figure 40 Spatial distribution of the predicted production of silver eel (before fishery and hydropower impacts), per decade and per river drainage system. The production for each river drainage area is plotted at the place of the river mouth, while in reality, the production will have taken place all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

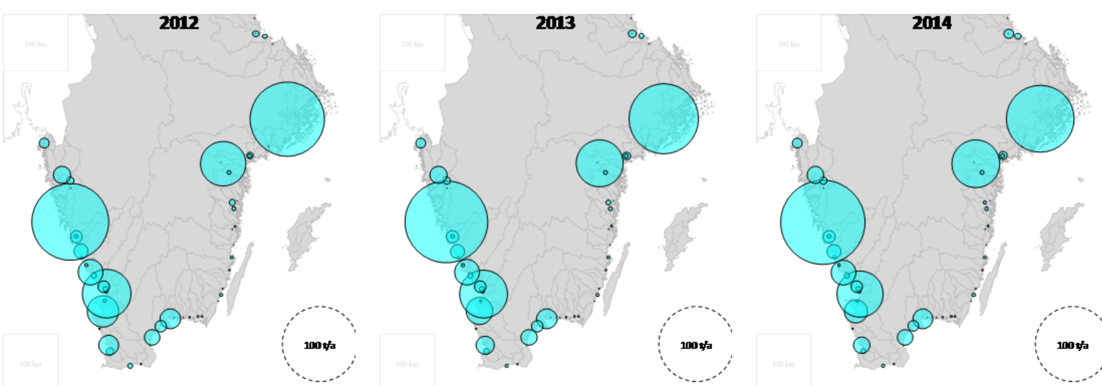


Figure 41 Spatial distribution of the estimated production of silver eel (before fishery and hydropower impacts), per year since 2012 and per river drainage system. The whole production estimated for each river drainage area is plotted at the place of the river mouth.

C.2.2 Silver eel destination

Figure 42 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.2.4, below.

Fishing data being incomplete up to 1986, results are only available for the period after. The total biomass of silver eel in Figure 42 matches the predicted total production, presented in Figure 39.

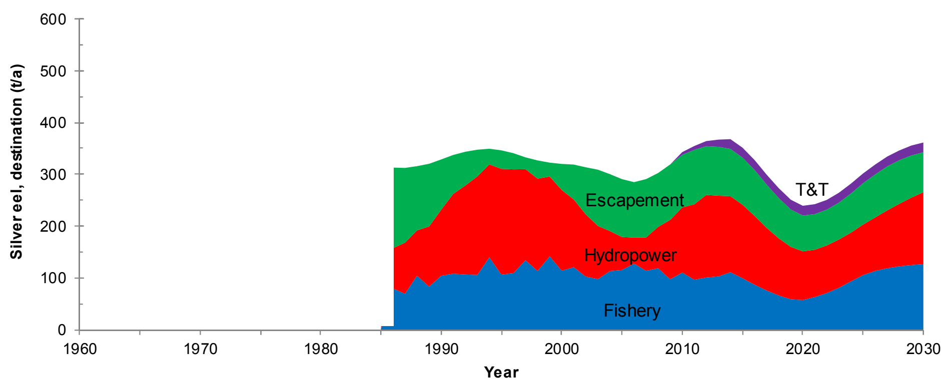


Figure 42 Time trends in the destination of the silver eel produced in inland waters. Data before 1986 are incomplete.

For the <u>fishery</u>, the landings have varied between 85 t (in 2011) and 133 t (in 1997). This is on average 33% of the production, with rather little variation over the years (Figure 43). The catch in 2014 was 111 t.

For the <u>hydropower</u>, the estimated impact varied between 50 t (in 2006) and 204 t (in 1995), that is approximately 40% of the total production (range 18% - 59%). The estimated impact in 2014 was 147 t.

Predicted <u>escapement</u> of silver eel ranged from 23 t (in 1997) to 156 t (in 1986), on average 27% of the total production (range 7% - 50%). The 2014 escapement is estimated at 91 t.

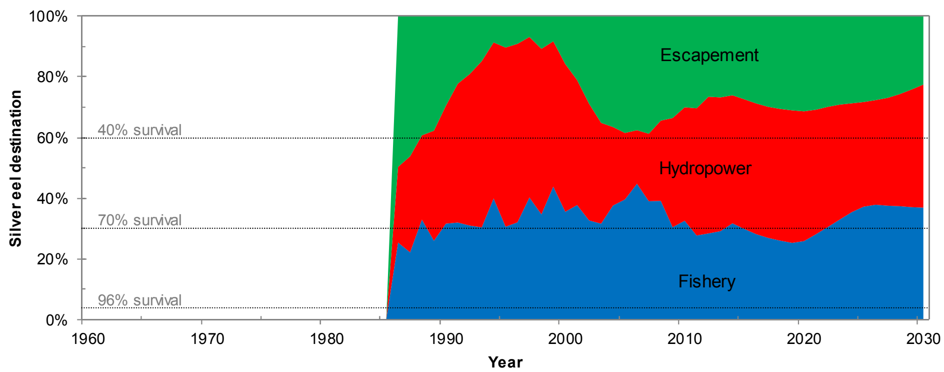


Figure 43 Time trend in the estimated anthropogenic mortality (and escapement), expressed in percentage impacts on the silver eel production.
The reference line "40% survival" represents the limit mortality for a healthy stock ($B_{current}$ > 40% * B_0). The reference line "70% survival" applies in 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 44), 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 also the discussion in section 3.3 above). The presentation in Figure 44 allows for both interpretations. Including the effect of restocking (yellow), the sum of fishing mortality, hydropower related mortality, restocking and T&T is represented by a drawn line (F+H+R+T); *without* restocking, the sum ΣA of fishing mortality and hydropower related mortality represents the actual mortality exerted on any part of the stock, whether natural or restocked.

Taking the effects of restocking into account, the total estimate has ranged from +1.90 (in 1997) to -1.04 (in 2011); the 2014 value is estimated at -0.94. 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 0.70 (in 1986) to 2.68 (in 1997); the 2014 mortality is estimated at 1.29. These estimates express the mortality exerted on the natural recruits, as well as on the restocked eels.

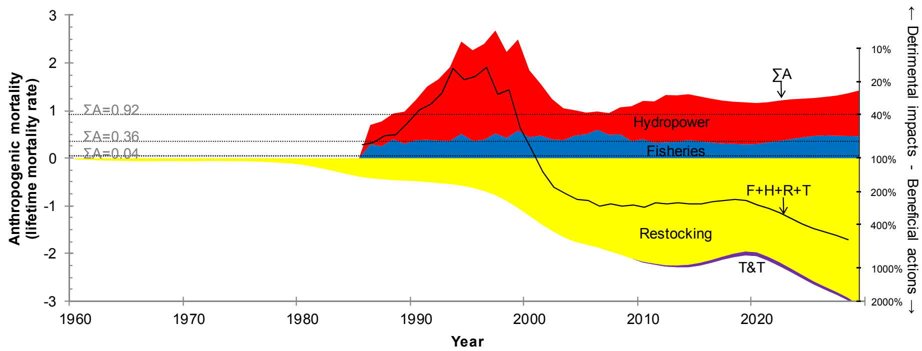


Figure 44 Time trend in the estimated anthropogenic mortalities: fisheries, hydropower, restocking and Trap & Transport (T&T). The mortality exerted by Restocking and Trap & Transport are negative; that is: these actions increase the amount of silver eel escaping. The line marked "F+H+R+T" represents the sum of all anthropogenic actions, including Restocking and Trap & Transport; ΣA represents the mortality exerted on the stock, whether natural or restocked.
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 ΣA=0.92 represents the limit mortality for a healthy stock ($B_{current} > 40\% * B_0$). 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 ΣA=0.36 applies in the current, depleted state, taking into account restocking. The reference line ΣA=0.04 applies in the current, depleted state, *not* taking into account restocking. A mortality of ΣA=0.11 conforms to the 90% survival, the management limit of the Swedish Eel Management Plan.

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 45 compares results, for two selected years: 1995 and 2014, that is: the year with maximum impact from hydropower, and the most recent year.

Depending on the value of M, production estimates (Figure 45.a&b) range from approximately 150 t/a to around 700 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 45.c&d), results are quite different. For M=0.05, production is estimated at circa 700 t; for M=0.15 at only 150 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 600 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 45.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), a smaller effect on the estimate of hydropower mortality H (a factor up to 2), and a very small effect on the estimate of restocking (expressed as a negative mortality). The estimate of total anthropogenic mortality ΣA reflects the sensitivity of F to M. The cumulative effect of fisheries and hydropower exceeds the minimal threshold of ΣA=0.92, for all values of M tested. The restocking did not compensate for these mortalities in 1995, but does more than so in 2014, for all values of M tested. Though the estimate of ΣA is sensitive to the assumption on M, the judgement remains that anthropogenic mortality exceeds the level for the current, depleted stock - even exceeds the limits that would apply for a healthy 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.

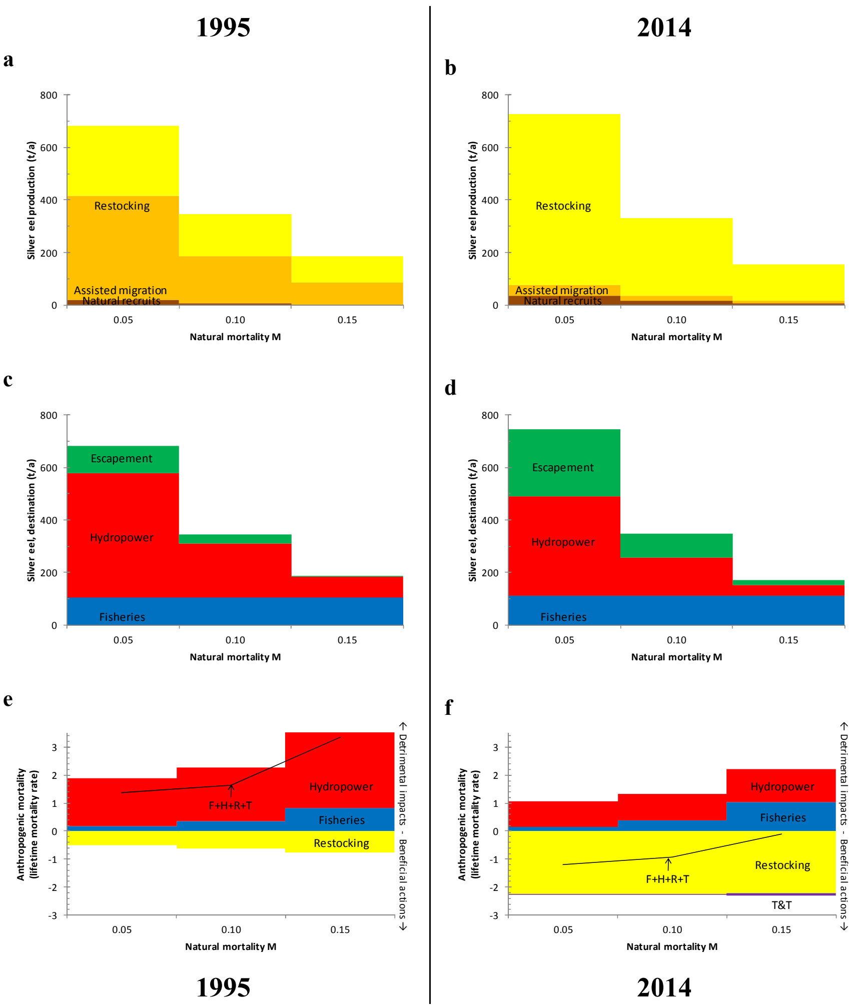


Figure 45 Comparison of results for 3 different values of natural mortality, showing results for 1995 (left) and 2014 (right). Within each sub-plot, the columns show results for the three options M=0.05, M=0.10 and M=0.15, respectively; comparisons are to be made within each subplot, between the columns.
Top row: predicted silver eel production (compare Figure 39);
Middle row: predicted silver eel destination (compare Figure 42);
Bottom row: anthropogenic mortality rates (compare Figure 44).

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. 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 39 indicates that silver eel production has varied between 300 and 500 t per year; 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 per year) to the amount of eel considered to have died of natural causes in the current reconstruction (order of magnitude of 400-1000 t per year) indicates that the available information on cormorant predation does not contradict the current results.

C.2.4 Mortality related to hydropower H

The results presented in this Annex so far are based on the 'best information' available for the impact of hydropower, i.e. (in order of priority) estimates from local experiments if available (obs), a simulated value (sim) reported in Calles and Christianson (2012), or a default impact of 70% per station. In this section, the sensitivity of results to this assumption is explored. To this end, the whole analysis was rerun, using either this 'best information' option, a uniform default value of 30%, or a uniform default value of 70%. Clearly, the higher the assumed value for the mortality H per hydropower station, the more biomass is estimated to be killed, and the less biomass is estimated to escape. These continuous relations between H and biomasses enable a presentation of all three options for H in a single graph (Figure 46). This graph shows the results for the year 2014, split by the number of hydropower stations along the route from the place where the eel was released as young recruit, towards the sea.

Of the 350 t of silver eel predicted to have been produced in the year 2014, 102 t is from areas with no hydropower station downstream (primarily Mälaren), only 21 t from areas with one or two hydropower stations downstream, 125 t is from areas having three hydropower stations downstream (primarily Vänern), and 100 t is from areas having four or more hydropower stations downstream.

Obviously, for the areas with no barriers downstream, it makes no difference which value for H is chosen. For the areas with four or more barriers, the choice of H appears to make little difference too (5 t); even for a low assumption on H, the number of stations to pass is such that very few eels survive all of them. For areas with three barriers (i.e. Vänern, and some less important areas), the most optimistic and the most pessimistic estimate differ by 27 t. That is: the discussion on an appropriate assumption for H narrows down to a correct estimate for the three stations between Vänern and the sea: Lilla Edet (obs=38%, sim=12%), Trollhättan (obs=32%, sim=50%) and Vargön (obs=24%, sim=11%). All these three observed mortalities being close to 30%, the difference between using the observed H or using a default of 30% is clearly marginal (4 t).

The discussion on H narrows down to a discussion on three hydropower stations only, and the mortality of all these three has been shown in field experiments to be close to 30%. Assuming that H=70% (as was done in the Swedish Eel Management Plan, ÅFP, Anon 2008) would contradict the available information, while the other two assumptions give almost identical results. It is therefore concluded, that the uncertainty in the value of H has very little relevance for the current reconstruction and impact assessment.

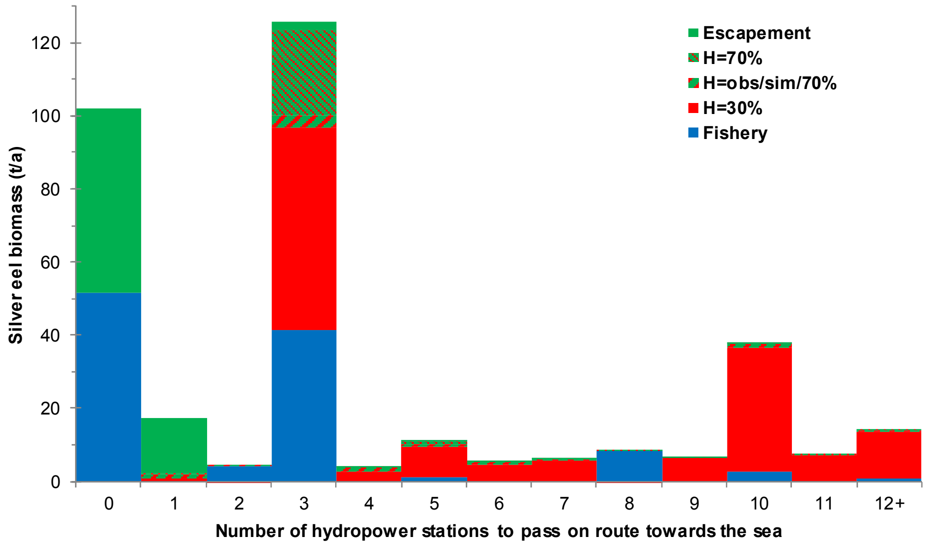


Figure 46 Comparison of the effect of <u>three different assumptions</u> on the impact H of individual hydropower stations on the silver eel run. This plot shows the predicted destination of the <u>silver eel running in 2014</u>. Hydropower mortality is plotted in red, escapement in green. The parts that either will have been killed by hydropower, or have escaped freely (depending on the assumptions) are shown hatched green/red. Under the most mild assumption (H=30%), only the red bars are killed; both hatched bars contribute to the escapement. Under the middle assumption (H=obs/sim/70%), the heavily hatched part is killed too, but the finely hatched parts escape. Under the most severe assumption (H=70%), even the finely hatched part is killed, and only the solid green escapes.

The references for this Annex are included in the reference list of the main report, on page 38.

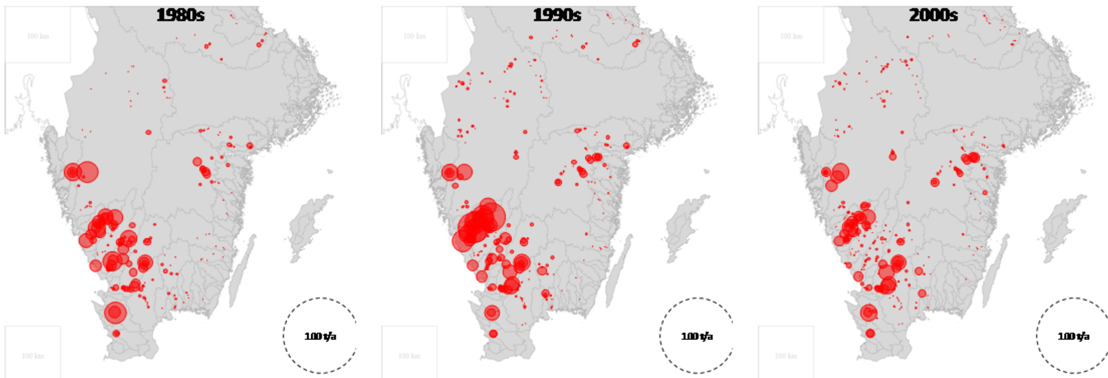


Figure 47 Spatial distribution of the estimated impact of hydropower, per hydropower station per decade. For the 1980s, estimates are based on the years from 1986 onwards; for the earlier years, no estimates could be derived because of the absence of information on the landings from fisheries.

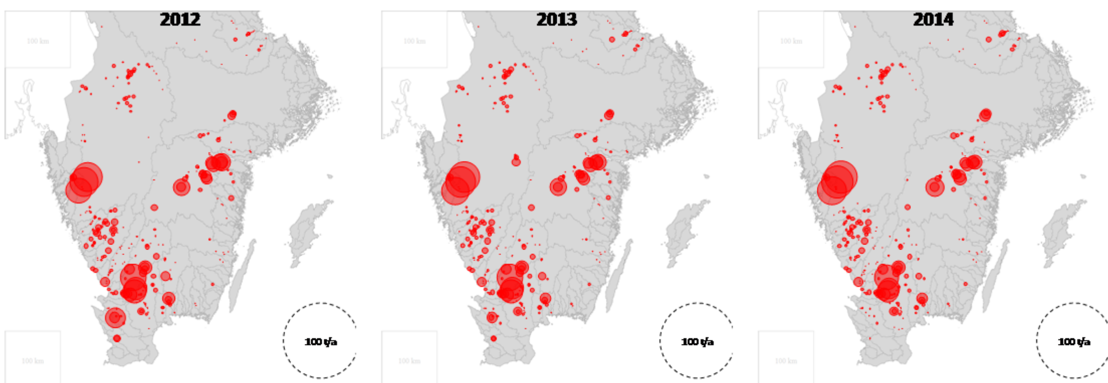


Figure 48 Spatial distribution of the estimated impact of hydropower, per hydropower station per year, since 2012.

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. Since 2012, the silver eel tagging programme has been re-continued (Figure 49), and this Annex now presents an updated analysis. No changes in the methodology of Dekker & Sjöberg (2013) have been made.

From 2012 to 2014, ten experiments have been conducted (Figure 50), tagging 1 353 silver eels in total, of which 94 have been recaptured.

Estimates of the hazard and survival curves are given in Figure 51 and Figure 52. 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 213 t in 2014.

Figure 54 to Figure 56 present the results of the population reconstruction by county (län), for the 2010s in particular. This reconstruction uses the estimate of the fishing mortality, that is the hazard (Figure 54) from Survival Analysis (Figure 52), and combines that with the landings (Figure 5) split by county (Figure 55), to derive an estimate of the population size (Figure 56). As in Dekker and Sjöberg (2013), problems arise due extremely low estimates. For Gävleborg, for instance, the hazard of being recaptured is estimated at 0.001%, approximately one out of 100 thousand tagged eels being recaptured. The landings in Gävleborg recorded in 2014 amounted to 0.89 t only. Assuming that the fishery has taken only one out of 100 thousand eels, those 0.89 t of catch apparently have come from a total population in Gävleborg of 89 000 t. Clearly, this is not a realistic estimate, caused by the extreme uncertainties of dividing a near-zero catch by a near-zero hazard. For the counties further south, most population estimates are in the order of 2000 t, with the exception of Blekinge

(5400 t). For Södermanland, a catch of only 1645 kg was recorded; the population is estimated at 1055 t only.

Over all counties with a catch > 10 t, the average hazard has declined from over 50% in the 1950s, to ±10% in the 2000s, and 1.75% in the 2010s. Over all counties with a catch > 100 t, the 2010s estimate comes at 2.19%. 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 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 50), and the mean distances between release and recapture (Figure 53) hints at a lower recapture rate, rather than a lower tag return rate. Alternatively, the 2014 experiments might be too fresh, too recent, to allow for any recaptures yet. However, re-running the model without the 2014-releases (not shown) does estimate a somewhat higher survival, but only marginally so. That is: the evidence points towards a lower recapture rate, rather than a lower tag return rate.

The 2012 assessment (Dekker 2012) estimated fishing mortality by year based on mark-recapture estimates from the years 2006-2008, extrapolating towards 2011 on the assumption that fishing mortality and catch would be proportional. Noting that the proportionality between mortality and catch obviously does not apply in most recent years, no extrapolation is made in this report. Instead, we report on decadal means, matching the survival analysis.

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 references for this Annex are included in the reference list of the main report, on page 38.

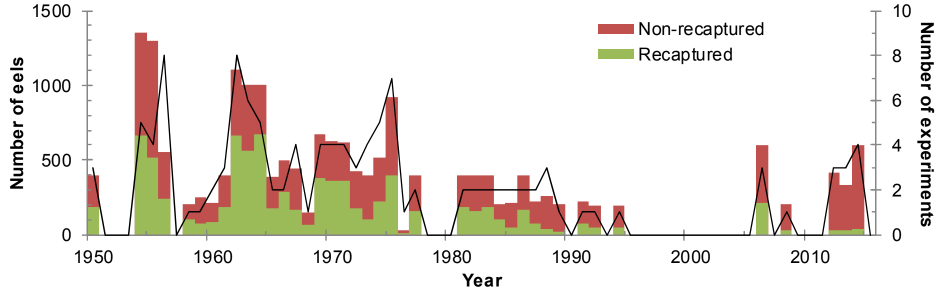


Figure 49 Time trend in the number of tagging experiments and the number of eels being tagged.

Figure 50 Location of the tagging experiments in the years 2012-2014. The size of the larger symbols is proportional to the number of eels released. The small dots represent recaptures of single eels.

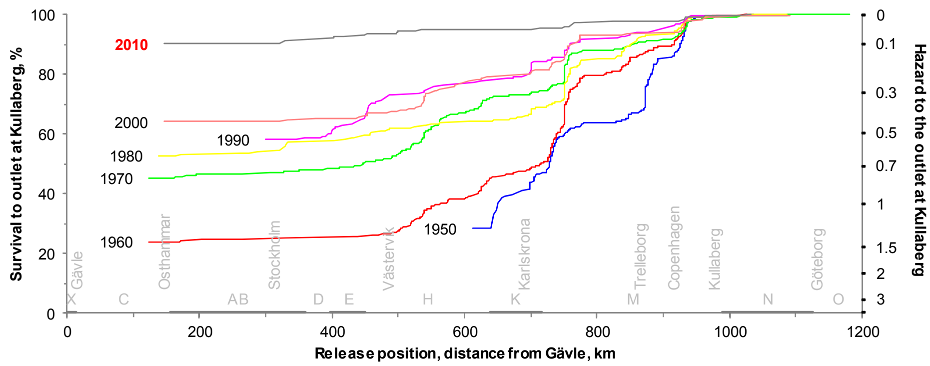


Figure 51 Hazard and survival by decade, estimated by the Kaplan-Meier method. The horizontal axis gives the distance from Gävle, just north of the northernmost release. The left vertical axis expresses the net survival observed in the recapture data; the right vertical axis expresses the same in terms of the accumulated hazard over the remaining interval.

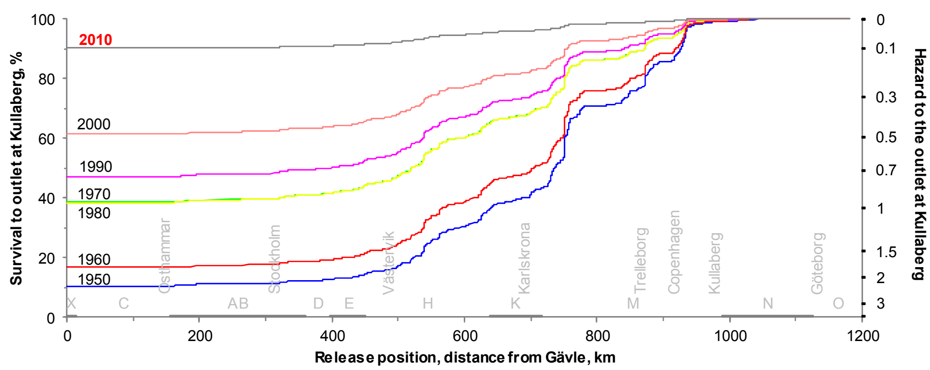


Figure 52 Hazard and survival, estimated by Cox proportional hazards model, by decade, without time-dependent covariates. The left vertical axis expresses the net survival from the release position t_0 to the outlet of the Baltic at Kullaberg; the right vertical axis expresses the same in terms of accumulated hazard over that interval.

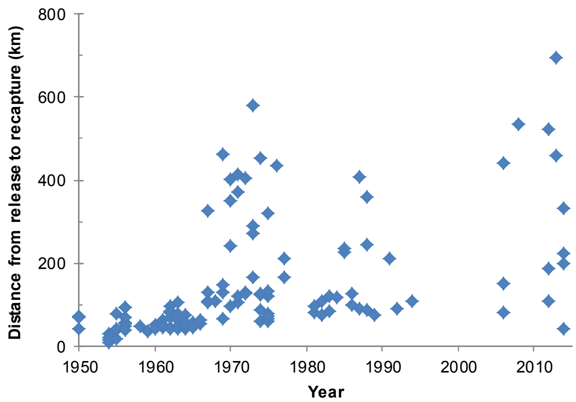


Figure 53 Mean distance between tag release and tag recapture, by year. Each dot represents a single tagging experiment.

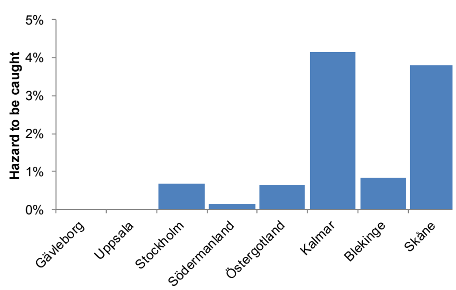


Figure 54 Hazard by county, in the 2010s.

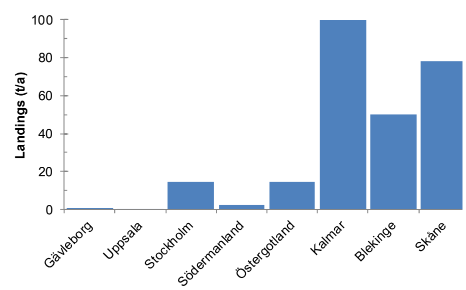


Figure 55 Landings by county, in the 2010s.

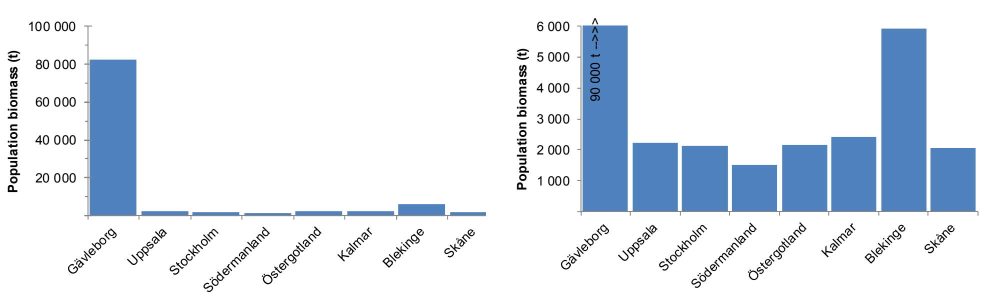


Figure 56 Estimated population size by county, in the 2010s. The vertical axis in the left graph allows plotting all counties, while the right graph is rescaled, excluding the unrealistic estimate for Gävleborg.