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European Eel (*Anguilla anguilla*) stock size, anthropogenic mortality and silver eel escapement in the Netherlands 2006-2020.

T. van der Hammen, F. Soudijn, J. Volwater, J.C. van Rijssel, A.B. Griffioen, C. Chen and H.V. Winter

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Postbus 20401
2500 EK, Den Haag
dhr. L. Gorissen

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Stichting Wageningen Research
Centre for Fisheries Research (CVO)
P.O. Box 68
1970 AB IJmuiden
Phone. +31 (0)317-487418

Visitor address:
Haringkade 1
1976 CP IJmuiden

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Samenvatting

Sinds de jaren 1980 zijn de glasaalintrek en de aalpopulatie zeer sterk teruggelopen. ICES (the International Council for Exploration of the Sea, www.ices.dk), die op verzoek van de Europese Commissie (EC) advies uitbrengt over de status en het beheer van visbestanden, heeft daarom aan het eind van de jaren 1990 advies gegeven voor het opstellen van een internationaal herstelplan. Dit heeft ertoe geleid dat de Europese Unie in 2007 de "verordening van de Raad tot vaststelling van maatregelen voor het herstel van het bestand van Europese aal (EC 1100/2007)" heeft ingevoerd. Deze verordening (de 'Aalverordening') verplicht de lidstaten om een nationaal aalbeheerplan op te stellen en te implementeren. Het doel van deze aalbeheerplannen is daarbij als volgt omschreven:

> *"Doel van de beheerplannen voor aal is het verminderen van de antropogene sterfte, zodat er een grote kans bestaat dat ten minste 40% van de biomassa van schieraal kan ontsnappen naar zee, gerelateerd aan de beste raming betreffende de ontsnapping die plaats zou hebben gevonden indien de mens geen invloed had uitgeoefend op het bestand. De beheerplannen voor aal worden opgesteld met het oog op het bereiken van die doelstelling op lange termijn (Artikel 2.4 van de Aalverordening)."*

De maatregelen in het Nederlandse aalbeheerplan zijn vanaf juli 2009 geïmplementeerd (*Tabel 1*).

Tabel 1 Overzicht van de maatregelen in het Nederlandse aalbeheerplan.

Maatregel aalbeheerplan
• Terugzetten van aal op zee en op binnenwater door sportvissers
• Verbod op recreatieve visserij, gebruikmakend van professionele vistuigen.
• Gesloten aal visseizoen 1 september tot 1 december
• Decentraal aalbeheer in de provincie Friesland (op basis van quotum).
• Stoppen met uitgave van peurvergunningen op Staatswateren.
• Onderzoek naar het kweken van aal in gevangenschap.
• Oplossen van migratieknelpunten bij sluizen, gemalen en andere kunstwerken.
• Aangepast turbinebeheer bij de drie grote waterkrachtcentrales, verminderen sterfte met minstens 35%
• Visserijvrije zones in gebieden die belangrijk zijn voor aal migratie.
• Sluiten van de visserij in de belangrijkste grote rivieren, met als aanleiding dioxineverontreiniging (april 2011).
• Uitzet van glas- en pootaal.

De Aalverordening verplicht lidstaten ook om over de effectiviteit van de aalbeheerplannen te rapporteren aan de EC. Deze verplichting gold voor de eerste drie rapportages elke drie jaar (tot en met 2018), en daarna elke zes jaar. Echter, aangezien de huidige stand van de aalpopulatie nog steeds zorgwekkend is, hebben de lidstaten afgesproken om voorlopig drie jaarlijks te blijven rapporteren aan de EC.

De onderhavige rapportage heeft een aantal updates ondergaan in vergelijking met de vorige rapportage (van de Wolfshaar et al., 2018). Ten eerste is de driejaarlijkse periode verschoven met één jaar. De meest recente periode van rapporteren is daardoor 2018-2020 (en niet 2017-2019), zodat de meest recente data is meegenomen in deze rapportage. Daarnaast zijn de verschillende onderliggende modellen verbeterd. Als gevolg van de verschuiving zijn de schattingen uit voor de eerdere driejaarlijkse periodes opnieuw berekend waarbij gebruik is gemaakt van de verbeterde modellen.

In deze rapportage wordt het aalbeheerplan geëvalueerd in het licht van het bovenstaande beheerdoel uit de Aalverordening (*Artikel 2.4)*. De methodiek die bij deze evaluatie is gehanteerd, komt voort uit ICES SGIPEE (Study Group on International Post-Evaluation on Eels, 2010a, 2011). De evaluatie is uitgevoerd door middel van modellen, vangstgegevens, veldwaarnemingen en statistische analyses, welke worden beschreven in de rapportage. Het geheel van deze inspanning resulteert in schattingen van een aantal, door de EC gevraagde, bestandsindicatoren voor vijf driejaarlijkse periodes (2006-2008, 2009-2011, 2012-2014, 2015-2017 en 2018-2020). De belangrijkste bestandsindicatoren zijn B_0, $B_{current}$ en LAM. B_0 is de biomassaschatting van uittrekkende schieraal in een pristine situatie. Voor Nederland is

deze vastgesteld op 10.400 ton op basis van wetenschappelijke onderzoeken, de beoordeling van de resultaten van die twee onderzoeken door een onafhankelijke commissie van deskundigen en de beoordeling van dit geheel door ICES (ICES 2010b).

De 10.400 ton hebben betrekking op de binnenwateren. Voor alle wateren (dus ook de kustwateren en de visserijzone) gaat het om 13.000 ton (B_0); 40% hiervan is 5200 ton. Voor de kustwateren en de visserijzone is er geen monitoringsprogramma, daarom worden alleen de "binnenwateren" meegenomen. $B_{current}$ is de schatting van de daadwerkelijke schieraalbiomassa die uittrekt naar zee. De doelstelling voor de lange termijn (*artikel 2.4)* is dat de verhouding tussen $B_{current}$ en B_0 hoger is dan 0.40 (40%). *LAM* geeft de hoogte van de totale antropogene sterfte aan. Deze bestaat voornamelijk uit barrièresterfte en visserijsterfte.

Effecten van het Nederlandse aalbeheerplan op de Nederlandse aalpopulatie

De schattingen van de bestandsindicatoren laten zien dat de maatregelen uit het Nederlandse aalbeheerplan hebben geleid tot een toename van de uittrekkende schieraalbiomassa ($B_{current}$) en een teruggang in antropogene sterfte (*LAM*) tussen 2006-2008 en 2015-2017 (*Tabel 2*). In het bijzonder de eerste periode na de implementatie van de maatregelen uit het Nederlandse aalbeheerplan (2009-2011) resulteerde in een afname in antropogene sterfte. De daaropvolgende en de tweede periode na invoering (2012-2014) leidde tot een toename van de uittrekkende schieraalbiomassa (*Tabel 2*). De reductie in antropogene sterfte was voornamelijk het gevolg van een afname van de aanlandingen (visserijdruk) in de commerciële en recreatieve visserij. Echter in de meest recente periode (2018-2020) is de antropogene sterfte weer toegenomen. Dit wordt hoofdzakelijk veroorzaakt door een toename van de commerciële visserij (inspanning en aanlandingen) in het IJsselmeer/Markermeer in de in deze periode (*Table 2-1* & *Appendix A0*). In deze laatste periode (2018-2020) is er ook een afname in de biomassa uittrekkende schieraal ($B_{current}$) in vergelijking met de periodes ervoor. Deze afname is het gevolg van de hogere antropogene sterfte (*LAM*), en de lagere biomassa schatting van de aanwezige rode aal en schieraal. Dit laatste is een resultaat van de lagere vangstsuccessen in de gebruikte monitoringen.

Tabel 2 Schattingen van de belangrijkste bestandsindicatoren. B_0 biomassa schatting voor uittrekkende schieraal in een pristine situatie (tonnen); $B_{current}$ de schatting van de daadwerkelijke schieraalbiomassa uittrek (tonnen); 100 $B_{current}/B_0$ huidige schieraaluittrek als percentage van de pristine uittrek; LAM: Lifetime Antropogene Sterfte; Mbarrier Schieraal barrière sterfte.*

Stock Indicator	B_0^*	$B_{current}$	100* $B_{current}/B_0$	LAM	Mbarrier
2006-2008	10,400	634	6.1%	83%	17%
2009-2011	10,400	837	8.1%	53%	16%
2012-2014	10,400	1,311	12.6%	42%	14%
2015-2017	10,400	1,463	14.1%	40%	11%
2018-2020	10,400	974	9.4%	55%	13%

* Zonder kustwateren (2,600 t)

Door aanpassingen aan de infrastructuur bij migratieknelpunten, alsmede de verhouding biomassa tussen verschillende gebieden in Nederland, is het percentage barrière sterfte van schieraal licht afgenomen (van 17% in 2006-2008 naar 13% in 2018-2020). Van 2015-2017 tot 2018-2020 is de barrière sterfte licht toegenomen van 11% naar 13%. Deze toename is niet veroorzaakt door nieuwe barrières, maar door een verschil in de ruimtelijke verdeling van aal.

De status van het aalbestand in Nederland blijft in 2018-2020 verontrustend met hoge sterfte en lage biomassa. De huidige biomassa van uittrekkende schieraal (9.4%) ligt ver onder de doelstelling van minimaal 40% van de pristine biomassa en de huidige sterfte door menselijk handelen ligt boven de geadviseerde sterfte bij een dergelijke lage biomassa aan uittrekkende schieraal.

Een verbetering in de aalpopulatie in Nederland en in de uittrek van schieraal wordt niet op de korte termijn verwacht omdat aal een langlevende soort is. Het duurt naar schatting 1-3 jaar voordat glasaal aankomt voor de Nederlandse kust en de binnenwateren op zwemt. Vervolgens duurt het 3-20 jaar voordat deze aal “schieraal” wordt, en terugtrekt naar zee. Daarnaast is de aalpopulatie een panmixtische populatie met een natuurlijke verspreiding van Noorwegen tot noord Afrika. Herstel in biomassa is daardoor de gezamenlijke verantwoordelijkheid van alle landen.

Uit de analyses is wederom gebleken dat er grote aannames gemaakt moeten worden om tot een biomassaschatting te komen, welke van invloed kunnen zijn op de resultaten. De omvang van de opwerking (aalbiomassa in álle Nederlandse wateren) en de beschikbare (historische) gegevens lenen zich niet tot zeer nauwkeurige berekeningen. De schattingen van de bestandsindicatoren moeten daarom voorzichtig worden geïnterpreteerd vanwege de aanzienlijke mate van onzekerheid rond deze schattingen.

Summary

Since the 1980s, the arrival of glass eel at the coast and the European eel stock have declined sharply. ICES (the International Council for Exploration of the Sea, www.ices.dk), which provides advice on the status and management of fish stocks at the request of the European Commission (EC), has therefore recommended the implementation of a recovery plan since the 1990's. As a result, in 2007 the EU introduced the 'Council Regulation establishing measures for the recovery of the stock of European eel (EC 1100/2007)'. This regulation (the 'Eel Regulation') requires Member States to develop and implement a national eel management plan. The purpose of these eel management plans is described as follows *(Article 2.4)*:

> *"The objective of each Eel Management Plan shall be to reduce anthropogenic mortalities so as to permit with high probability the escapement to the sea of at least 40% of the silver eel biomass relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock. The Eel Management Plan shall be prepared with the purpose of achieving this objective in the long term."*

The Dutch Eel Management Plan was implemented in July 2009 (*Table 0-1*).

Table 0-1 Overview of the measures in the Dutch Eel Management Plan.

Measure
• Implementation of a program for the improvement of fish migration including eel, which is expected to resolve the issues at 1800 of the most important migration barriers.
• Reduction of eel mortality at hydroelectric stations with at least 35%.
• The establishment of zones where fishing is not allowed in areas that are important for eel migration.   • Closed area to eel fisheries due to high levels of dioxins and PCB's (April 2011)
• Release of eel caught (a) at sea and (b) at inland waters by anglers.
• Ban on recreational fishing using professional gear in coastal areas.
• Annual closed season from 1 September to 1 December.
• Decentralized eel management in the province of Friesland (a quota system).
• Stop the issue of licenses for eel sniggler (*Dutch: 'peur'*) by the minister of LNV in state-owned waters.
• Restocking of glass eel and pre-grown eel from aquaculture
• Research into the artificial propagation of eel

The Eel Regulation also obliges the reporting to the European Commission (EC) on the effectiveness of the eel management plans. This obligation was intended to apply every three years for the first three reports (up to and including 2018), and every six years thereafter. However, as the eel population is still in a worrying state, it was agreed to continue reporting on the status of the eel stock to the EC every three years.

This report has undergone several updates compared to the previous report. First, the three-year reporting period was shifted by 1 year. The most recent reporting period is therefore 2018-2020 (and not 2017-2019), as a result of which the most recent data could be included. In addition, the various models have been improved. The estimates from earlier periods have also been recalculated, taking into account the improvements.

In this report, the eel management plan is evaluated in the light of the management objective from the Eel Regulation. The methodology used in this evaluation is derived from ICES SGIPEE (Study Group on International Post-Evaluation on Eels, 2010a, 2011). The evaluation is carried out using models, catch data, field observations and statistical analyses, which are described in the report. This effort has resulted in estimates of a number of stock indicators for five three-year periods (2006-2008, 2009-2011, 2012-2014, 2015-2017 and 2018-2020). The main indicators are B_0, $B_{current}$ and *LAM*. B_0 is the biomass estimate of escaping silver eel in a pristine situation. The value of B_0 was determined in 2010 (ICES 2010b) and has not changed since. $B_{current}$ is the estimate of the actual silver eel biomass that migrates to the sea. The long-term objective is that the ratio between $B_{current}$ and B_0 will reach and will remain

above 0.40 (40%). *LAM* indicates the level of anthropogenic mortality. This mainly consists of barrier mortality and fishing mortality.

The results show that the measures from the Dutch eel management plan have led to an increase in biomass escaping silver eel ($B_{current}$) and a decrease in anthropogenic mortality (*LAM*) between 2006-2008 and 2015-2017 (*Table 0-2*). In particular, the first period after the introduction of the Dutch eel management plan (2009-2011) led to a decrease in anthropogenic mortality and the second period after implementation (2012-2014) led to an increase in biomass of escaping silver eel (*Table 0-2*). The reduction in anthropogenic mortality was mainly due to decreases in catches from both commercial and recreational fisheries. However, anthropogenic mortality has increased again in the last period (2018-2020). This is caused by an increase in the commercial fisheries (landings and effort), mainly in the lakes IJsselmeer and Markermeer (*Table 2-1* & *Appendix A0*). In this same period there is also a decrease in the biomass of escaping silver eel ($B_{current}$). This is caused by the higher anthropogenic mortality, but also due to the lower biomass estimate of the current standing stock of present yellow and silver eel. This is a direct result of the lower catch success in several monitoring programs.

Due to adjustments to the infrastructure at migration bottlenecks, as well as the biomass ratio between different areas in the Netherlands, the barrier mortality (*Mbarrier*) has decreased slightly between 2006-2008 and 2018-2020 (from 17% in 2006-2008 en 13% in 2018-2020, *Table 0-2*). From 2015-2017 to 2018-2020 the barrier mortality increased slightly from 11% to 13%. This increase was not caused by new barriers, but by a difference in the distribution of eel.

Table 0-2 Estimates of the most important stock indicators. B_0 biomass estimate of escaping silver eel under pristine conditions (tonnes); $B_{current}$ estimate of the current silver eel escapement to the sea (tonnes); 100 $B_{current}$ /B_0 current silver eel escapement as a percentage of the pristine escapement; LAM total (lifetime) anthropogenic mortality rate; Mbarrier: barrier mortality rate.*

Stock Indicator	B_0*	$B_{current}$	100* $B_{current}$/B_0	LAM	Mbarrier
2006-2008	10,400	634	6.1%	83%	17%
2009-2011	10,400	837	8.1%	53%	16%
2012-2014	10,400	1,311	12.6%	42%	14%
2015-2017	10,400	1,463	14.1%	40%	11%
2018-2020	10,400	974	9.4%	55%	13%

* Excluding coastal waters (2,600 tonnes)

The status of eels in the Netherlands remains worrying in 2018-2020 with high mortality and low biomass. The current biomass of silver eel escapement as a percentage of B_0 (9.4%) is far below the target of 40%. In addition, the current mortality due to human activity is remains high and has even increased in the latest period.

An improvement in the eel population in the Netherlands and in the migration of silver eel is not expected in the short term because eel is a long-lived species. It takes an estimated 1-3 years before glass eels arrive at the Dutch coast and enter the inland waters. It then takes 3-20 years for these eels to become silver eels and return to the sea. In addition, the eel population is a panmictic population with a natural distribution from Norway to North Africa. Biomass recovery of the total eel stock is therefore the joint responsibility of all countries within the natural range of the eel population.

The analyses once again show that large assumptions are made in order to arrive at a biomass estimate, which may influence the results. The size of the area for reporting the eel biomass (all Dutch waters) and the (historical) available data do not lend themselves to very accurate calculations. The estimates of the stock indicators used to evaluate the status of the stock need to be interpreted with care due to the significant level of uncertainty surrounding these estimates.

1 Introduction

1.1 EU regulation and the Dutch eel management plan

The decline in the European eel (*Anguilla anguilla*) stock since the 1980's caused the International Council for the Exploration of the Sea (ICES) to recommend the development of a recovery plan for the European eel stock. In response to this advice, the EU Regulation for the Recovery of the Eel Stock (EC 1100/2007) was adopted in 2007. It required each Member State (MS) within the natural distribution area to set up Eel Management Plan's (EMP's) by the end of 2008 with the following aim:

> *"The objective of each Eel Management Plan shall be to reduce anthropogenic mortalities so as to permit with high probability the escapement to the sea of at least 40% of the silver eel biomass relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock. The Eel Management Plan shall be prepared with the purpose of achieving this objective in the long term."*

Each EMP covers an Eel Management Unit (EMU). An EMU covers a specific eel habitat (for example a river basin) and within MS's there can be a set of different EMU's. However, the Netherlands is located in the joint delta of four major rivers and the rivers are intertwined and confluent. Therefore there are no sharp boundaries between river basins in the Netherlands and is was decided that the Netherlands is a single EMU and therefore also a single EMP was drawn up for the Netherlands. The Dutch EMP was approved by the European Commission (EC) in October 2009. After the approval, several measures as described in the EMP to reduce eel mortality were implemented (*Table 1-1*). An adjustment to the EMP was made in 2018, with approval of the European Commission. In 2012, 2015 and 2018, progress reports were sent to the EC showing that the status of eel in Dutch waters remained in a situation regarded as "undesirable", and below the target of 40% of the estimated pristine situation. However, the progress reports also show that implementation of the EMP has resulted in an initial increase in biomass and a decrease in anthropogenic mortality (Bierman et al., 2012; van de Wolfshaar et al., 2015 & 2018).

Table 1-1 Overview of the implemented measures described in the Dutch Eel Management Plan (Ministry of Agriculture, Nature and Food quality (2018).

Measure	Planned implementation	Realized implementation
• Implementation of a program for the improvement of fish migration including eel, which is expected to resolve the issues at 1800 of the most important migration barriers.	2015-2027	2015-2027[a]
• Reduction of eel mortality at hydroelectric stations by at least 35%.	2009	November 2011[b]
• The establishment of zones where fishing is not allowed in areas that are important for eel migration.	2010	1 April 2011[c]
• Closed area to eel fisheries due to high levels of dioxins and PCB's	Unforseen	1 April 2011[c]
• Release of eel caught (a) at sea and (b) at inland waters by anglers.	2009	1 October 2009
• Ban on recreational fishing using professional gear in coastal areas.	2011	1 January 2011
• Annual closed season from 1 September to 1 December.	2009	1 October 2009[d]
• Decentralized eel management in the province of Friesland (a quota system).	-	2018[d]
• Stop the issue of licenses for eel snigglers (*Dutch: 'peur'*) by the minister of LNV in state-owned waters.	2009	1 May 2009
• Restocking of glass eel and pre-grown eel from aquaculture	2009	Early 2010
• Research into the artificial propagation of eel:		
PRO-EEL (EU-project)	2010	2010-2015
EEL- HATCH	2014	2014-2017
EELRIC (Dutch innovation centrum)	2015	2015 - ongoing
Glasaal Volendam (duurzame palingkweek/innovatief broedhuis)	2017	2017 - ongoing

[a] *In agreement with the EC, changes have been made to the original schedule of solving migration barriers.*

[b] *Due to technical difficulties, the maximum achievable reduction in mortality through adjusted turbine management is 24%.*

[c] *There was an (unforeseen) closure of eel fishery in contaminated (PCBs, dioxins) areas (all large rivers). The majority of the contaminated areas that were closed for commercial fisheries on 1/4/2011 include the main rivers. These rivers are the most important migration routes for diadromous species.*

[d] *In 2011 the province of Friesland started a pilot on a quota system. This system was adopted in the eel management plan in 2018. This allows those fishermen fishing in the province of Friesland to fish during the closed season based on a TAC (quota of 36.6 tonnes for all fishermen).*

1.2 Description of stock indicators.

In order to assess the status of the stock, the EC requires each MS to estimate a set of stock indicators (*Table 1-2)*, which are used to evaluate the status of the eel stock in relation to a pristine situation. Estimates of escaping silver eel biomasses and mortality rates of all eel are requested by the EC (*Table 1-2*). An explanation of each stock indicator is briefly described below.

$\boldsymbol{B_0}$ is the pristine silver eel biomass escapement to sea to spawn. It is an estimated value of the biomass that would exist if no anthropogenic mortalities for eel had ever taken place. The B_0 value for the Netherlands is set at 13,000 tonnes, of which 10,400 tonnes in inland waters (ICES 2010b). The target of the EU Regulation is set at 40% of this measure, i.e. 4,160 tonnes in inland waters in the Netherlands. An exact value of B_0 is extremely difficult to assess. Therefore the estimate has a wide uncertainty range and has been subject to discussion (*see Paragraph 9.1.1*).

$\boldsymbol{B_{current}}$ is an estimate of current silver eel biomass escapement to the sea to spawn. It gives an indication on how close a MS is to achieve the long-term objective (40% of B_0) of the EU Regulation for the Recovery of the Eel Stock (EC 1100/2007). However, $B_{current}$ does not depend only on anthropogenic mortality in a single EMU. The current biomass also highly depends on the number of recruits (glass eels) arriving at the Dutch coast. This inflow of recruits is, currently at a very low level (ICES 2020). However, because the European eel is one panmictic population, the inflow of recruits in the Netherlands depends on the silver eel escapement of all countries within the natural distribution area of European eel.

$\boldsymbol{B_{best}}$ is an estimate of the best possible silver eel escapement under recent recruitment conditions. It is an estimate of the current escaping silver eel biomass if there would be no anthropogenic influences.

LAM* and *ΣA, ΣF* and *ΣH are anthropogenic mortality rates for all eel (*Table 1-2*). LAM is the Lifetime Anthropogenic mortality percentage. *ΣA* is the total anthropogenic mortality rate, *ΣF* is the fishing mortality rate and *ΣH* is the anthropogenic mortality rate other than fishing mortality. For this evaluation, *ΣH* includes only the barrier mortality rate. Mortality rates due to, for example, pollution or parasites, are not taken into account, because they are extremely difficult to assess. The mortality rates give an indication whether management measures have resulted in a reduction in anthropogenic mortalities in a single EMU.

Table 1-2 Overview of the main stock indicators to be reported to the EC. The MS's are also obliged to report on the amount of glass eel (eel below 12 cm) harvested for restocking. These are not reported here because this is not relevant for the Netherlands as no glass eel is harvested.

Indicator	Description
B_0	Silver eel escapement (biomass) in the absence of any anthropogenic impact and at historic recruitment levels.
$B_{current}$	Silver eel biomass estimate that currently escapes to the sea to spawn.
B_{best}	Silver eel biomass estimate without anthropogenic influences on the current stock, i.e. the best biomass possible under current recruitment levels.
ΣF	Fishing mortality rate (commercial and recreational).
ΣH	Anthropogenic mortality rate other than fishing mortality (e.g., barrier mortality).
ΣA	The sum of anthropogenic mortalities, i.e., $ΣA = ΣF + ΣH$.

1.3 Structure of the report and flow diagram

To estimate the stock indicators described above (*Table 1-2*) the following calculations were carried out:

1) The biomass of the yellow and silver eel standing stock.
2) The yellow eel fishing mortality.
3) The mortality of migrating silver eel.
4) The total biomass of escaping silver eel.
5) The stock indicators.

Each step is briefly described below. In the following chapters, the methods are described in more detail.

First, the biomass of the yellow eel and silver eel standing stock is estimated with two different models. For most water bodies, but not the large lakes IJsselmeer, Markermeer, Randmeren and Grevelingen, a model where survey density from different surveys is scaled up to the total water surfaces is used. This model is called the 'Static Spatial Model'. For lakes IJsselmeer and Markermeer, a population dynamics model is parameterized to estimate fishing mortality (*F*) in these lakes. This model is called the 'Demographic Model'. The estimated fishing mortality is used in combination with the amount of yearly landings to estimate the standing stock biomass. For the lakes Randmeren and Grevelingen, the eel density as estimated in the lakes IJsselmeer and Markermeer is used as basis. Survey data in the Randmeren was available from 2012, which were used to correct the Randmeren values for differences in survey density.

Static spatial model: Stock estimates were made based on data from electric dipping nets, by scaling data on density (eel biomass per length class per area) to total wetted areas of water bodies. The amount of silver eel was estimated using a maturation key. This method is used for all inland waters, except the large lakes IJsselmeer, Markermeer, Randmeren and Grevelingen. The static spatial model is explained in detail in *Chapter 3*.

Demographic model: For the large lakes the method of the static spatial model is considered unreliable, because the surveys are conducted at the shore and raised to the level of the whole surface of the waterbody. The lakes have a disproportionally large surface area, as compared to the shores and therefore, strong assumptions would have to be made to use this data as is done in the static spatial model. Instead, for the lakes IJsselmeer and Markermeer, the fishing mortality rates were estimated by fitting a 'Demographic Model' to the electric trawl survey time series with the recruitment index at Den Oever as basis for the level of eel recruitment. The estimated fishing mortality rates were used in combination with the landings, to obtain estimates of the total eel standing stock in the lakes. The estimated eel density in the lakes IJsselmeer and Markermeer was also used to estimate the density for the Randmeren and Grevelingen. The number of silver eel was estimated using a maturation key. The demographic model is explained in detail in *Chapter 4*.

Next, a **barrier model** was used to estimate the silver eel mortality during migration from inland water bodies to the sea, due to barriers such as pumping stations and HSP's. The model assumes that, depending on the starting position, silver eels experience a different mortality risk depending on the numbers and types of barriers they encounter during migration to the sea. The estimation of the barrier mortality is described in detail in *Chapter 6*.

By combining the silver eel biomasses resulting from the static spatial model and the demographic model and the mortality of the migrating silver eel, the total biomass of escaping silver eel is estimated. In the final step, the estimated starting biomass, escaping biomass, the landings and the demographic model are combined to calculate the stock indicators (*Table 1-2*). The estimation of the stock indicators is described in detail in *Chapter 7*.

1.3.1 Structure of the report

As explained above, the stock assessment method consists of several steps. Below the content of each *Chapter* is summarized:

Chapter 2: In this chapter the biological keys are presented (maturity-at-length, weight-at-length, and sex-ratio-at-length) that are used in the demographic model and the static spatial model.

Chapter 3: In this chapter the static spatial model is described, which is used for the estimation of yellow eel and silver eel biomass in the regionally and nationally managed water bodies other than the large lakes (lakes IJsselmeer, Markermeer, Randmeren and Grevelingen). For larger, mostly nationally managed water bodies such as the main rivers and for the majority of smaller, mostly regionally managed water bodies, data from surveys using electric dipping nets were available.

Chapter 4: In this chapter the demographic model is described. The model is used for estimating the silver and yellow eel biomass in the large lakes, IJsselmeer, Markermeer, Randmeren and Grevelingen.

Chapter 5: In this chapter the total standing stock biomass is estimated by summing the results from the demographic model for Lake IJsselmeer, Markermeer, Randmeren and Grevelingen (*Chapter 4*) and the results from the standing stock biomass in the spatial spatial model from the other nationally managed waters and regionally managed waters (*Chapter 3*).

Chapter 6: In this chapter the migration model for the estimation of silver eel mortality due to barriers is described.

Chapter 7: In this chapter the results from chapters 2-6 are used for the estimation of the final key stock indicators (*Chapter 7*).

Chapter 8: In this chapter the stock indicators are discussed using the modified precautionary diagram as developed by ICES.

Chapter 9: The report concludes with a general discussion and recommendations for improvements to the stock assessment methodology.

The flow diagram below gives a broad overview of the key steps in the stock assessment methodology, with reference to the chapters.

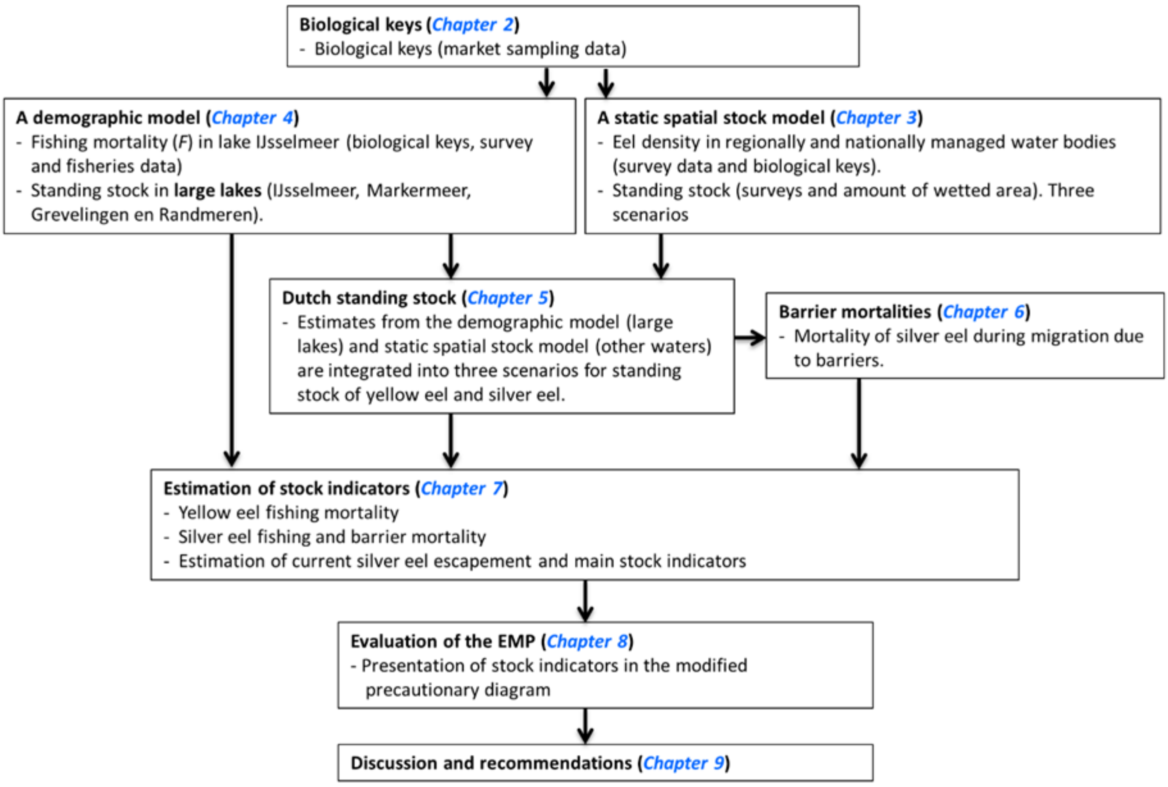

Figure 1-1 Flow diagram representing the key steps in the stock assessment methodology, and the structure of this report.

1.4 Assessment updates

Since the latest report, several improvements were made, which are described below:

1) In order to include the most recent (2020) data, the three-year period was shifted by one year compared to previous reports (Van de Wolfshaar et al., 2015 & 2018), such that the latest period is 2018-2020 (instead of 2017-2019). This results in the following periods to report on: 2006-2008, 2009-2011, 2012-2014, 2015-2017 and 2018-2020.
2) Improvements on the estimation of the biological keys (*Chapter 2*).
    a. Sex ratio at length and maturation at length were estimated with a Generalized additive model (*GAM)*.
    b. Growth was fitted with a von Bertalanffy growth function.
    c. Weight at length is now estimated for males and females separately.
3) The demographic model was updated and improved (see *Chapter 4* for more detail).
4) For the Water Framework Directive (WFD) waters a moving average was estimated, with average values for each six-year period (*Chapter 3*).
5) The Randmeren and Grevelingen have been assessed using eel density instead of fishing mortality from the lakes IJsselmeer and Markermeer, in combination with survey data for the Randmeren (*Chapter 4*).

2 Available data and biological keys

2.1 Short description of main data sets

The main data sets used in the stock assessment are described below. More detailed information is described in van Keeken et al. (2020a):

1) **Retained catches.** Retained catches are defined as the landings from commercial fishers. Since 2010 all freshwater landings are provided by the Ministry of Agriculture Nature and Food Quality (LNV) and are stored in a database ('Visstat'). For the landings in 2006-2008 in the Netherlands, an estimate as given in the EMP (Ministry of Agriculture, Nature and Food quality, 2018) is used. For the period 2009-2011, the average of 2010 and 2011 is used. For lakes IJsselmeer and Markermeer, PO (product board) data is available for the periods before 2010.

2) **Market sampling**. Representative samples are taken from retained catches from commercial fisheries each year and the lengths of the individual eels are measured (van Keeken et al. 2018). Furthermore, several eels per length class were selected from each sample for dissection and measurements of maturity, weight and sex (see van Keeken et al. 2020a for methods). These measurements are used to calculate maturity-at-length, weight-at-length, and sex-ratio-at-length. From a subsample of these eels, age readings of otoliths are conducted, in order to estimate sex-specific growth curves. Data from the market sampling between 2006 and 2020 are used in this assessment. The biological keys (*Chapter 2*) are used in the demographic model (*Chapter 3*), in the static spatial model (*Chapter 4*) and to calculate the reference points (*Chapter 7*).

3) **Surveys in regionally managed water bodies**. Eel sampling within the Water Framework Directive (WFD, 2000/60/EC) waters was executed following an EU certified protocol. In the assessments presented here only data from electrofishing with electric dipping nets were used. Sampled water bodies are representative for water types defined within the Netherlands based on WFD regulation. Data collection is managed and stored by regional water boards. Electric dipping net data for recent years were obtained from ATKB (consultancy for water, soil, and ecology) and several water boards. A total of ~8800 samples by electric dipping nets were available between 2006 and 2019, covering most of the combination of water boards and water body types.

4) **Surveys in nationally managed water bodies**. Within the survey program "Fish Monitoring National Waters," fish species in the main Dutch rivers are monitored yearly (van Keeken et al., 2020a). In the program, the main rivers and water bodies connected to the main rivers are sampled in autumn or in some cases early spring. Depending on the region, sampling started in 1997 or later.

5) **Non - Water Framework Directive waters.** Ditches are underrepresented in the set of WFD water bodies. Therefore, a survey with an electric dipping net is carried out by Wageningen Marine Research (WMR) every year and is added separately to the spatial model. A total of ~350 samples by electric dipping nets were available between 2013 and 2020.

6) **FYMA electric trawl survey in lakes IJsselmeer and Markermeer**. Since 1989, WMR has been conducting an annual (yellow) eel survey in lake IJsselmeer (25 sites) and lake Markermeer (15 sites) with an electrified trawl. The survey takes place in the autumn (October-November). The data is used to tune the demographic model (*Chapter 4,* van Keeken et al., 2020a).

7) **Glass eel survey liftnet Den Oever** Since 1938, recruitment monitoring has been running at Den Oever. The monitoring is conducted with a liftnet (1x1 m) during March-May. Glass eel data are presented as the average number of glass eels per haul in the months of April and May and is used as input for the demographic model (*Chapter 4*).

8) **Recreational landings** Since 2010 a biennial survey has been conducted in the Netherlands to estimate the total eel catches in the recreational fisheries (Van der Hammen, 2017, Van der Hammen in prep.).

9) **Transponder research Meuse** The anthropogenic mortality of migratory silver eels in the Dutch rivers is determined by means of tracking silver eels equipped with a transponder. Within this transponder research, 150 silver eels are provided with a NEDAP transponder once every three years and released in the upper reaches of the Dutch part of the river Meuse. The data is used to estimate silver eel escapement in relation to anthropogenic mortality of silver eel by hydroelectric power stations. Unfortunately, data from the last field season (2019) could not be used due to database crashes at NEDAP and damaged detection stations.

10) **Diadromous fish monitoring programme** A survey programme started in 2012 to monitor the abundance of migrating silver eel on five exit points (Kornwerderzand sluices, Den Oever sluices, North Sea Canal, New Waterway channel, Haringvliet-West inlet) and two entry points for migratory fish (River Rhine and River Meuse) during spring and autumn. The programme is a collaboration between WMR, Rijkswaterstaat and commercial fishermen. The months September, October and November were selected for illustrating trends in silver eel abundance at each location. Because the indices are short (9 years), did not run before the implementation of the EMP and there are missing years, the monitoring is not used in the evaluations, but the trends are reported in Appendix C5.

2.2 Biological Data

Biological keys, such as maturity-at-length, weight-at-length, age-at-length, and sex-ratio-at-length are estimated with the available data from biological market samples. The biological keys are used in the assessment in the static spatial model and in the demographic model to convert lengths to ages or to yellow and silver eel biomass (*Chapter 3, 4* & *7*). The biological keys were based on all sampled eel, which is assumed to result in estimates representative for a national eel population. The biological keys that are presented in this chapter differ from previous years (van de Wolfshaar et al, 2018), because 1) more biological data became available since the previous assessment and 2) some keys were calculated with a different method.

The data used to calculate the biological keys are measurements from eels that were taken from commercial catches (i.e., 'market samples') throughout the Netherlands (van Keeken et al. 2018). In addition, for the estimation of the age-at-length key, otolith readings from the DAK project ('*Duurzaam Aalbeheer door Kennis*') are added to the otolith readings from the market samples. In total > 12,000 individual eels collected from the commercial catches between 2006-2020 were used to assess the biological keys. From 693 individual eels sampled between 2009-2019 (573 from market sampling, 120 from the DAK project) the otoliths were analyzed to assess the interannual growth increments. From these increments, the 'years after arrival' at the coast can be calculated. This differs from age, because the glass eel has already reached an age of 2-3 years before arriving at the coast. In eel research, 'age' usually refers to the age after arrival at the coast. In this report, 'years after arrival' and 'age' both refer to the age after arrival at the coast.

2.3 Sex ratio at length

Males and females have different growth rates and male eel mature and migrate to the sea at smaller lengths and at younger ages compared to females. Consequently, sex ratio is expected to vary with length. In the previous assessments a linear relationship was assumed. Because the real relationship is unknown and unlikely to be linear, in this assessment the length-sex ratio relationship was estimated by fitting a *GAM* (*Figure 2-1*). A *GAM* does not have a fixed shape and can therefore be used to fit a non-linear relationship. Sex ratio as a function of length was assessed for lengths of 28 cm and larger, because determination of the sex is usually not possible at small lengths and there was insufficient data available at smaller sizes because of the minimal landing size of 28 cm.

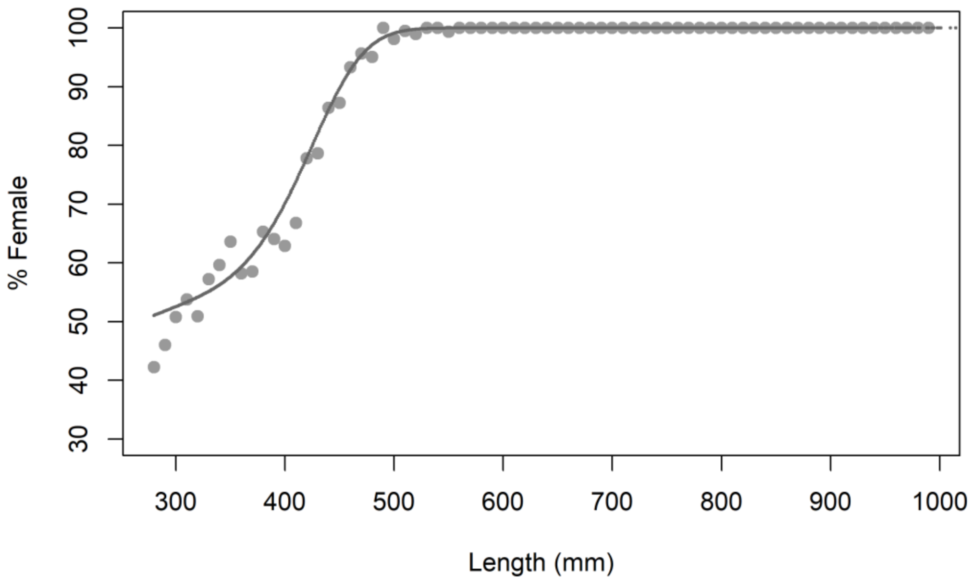

Figure 2-1 Percentage of females (dots) and GAM fit based on all samples (≥28cm) from the market sampling program 2006-2020 per 10mm length class. N (males) = 2,409, N (females) = 10,172.

2.4 Maturation at length

Males become silver eel at smaller sizes than females. Because most eel start their migration to the sea directly after silvering, most of the silver eel that are seen in the catches represent the eel that became silver eel only recently. It is also difficult to assess if the catches are representative for the proportion of silver eel in the stock, because of the timing of the main fisheries. In general, larger numbers of silver eel are caught during the silver eel migration period. Because of the three months closure of the fishery during the silver eel migration (September-November) since 2009[1], the sampling of the commercial catches could result in an underestimate of the proportion of silver eel in the stock at the start of the migration season for the years after 2009. However, market sampling during the migration season could result in an overestimate of the proportion of silver eel as they have higher catchability (in the passive gears) due to increased mobility. In addition, at downstream locations, the silver eel in the catch may originate from upstream locations, which could cause an overestimate of the proportion silver eel downstream and an underestimate upstream. These factors cause uncertainty of the maturity key, which is not taken into account for this report. Because the shape of the relationship between silvering and length is unknown, it was fitted with a *GAM* for both males and females (*Figure 2-2*). The analyses show that males start to silver at smaller lengths (~ 33 cm) compared to females (~ 50 cm). The *GAM*

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[1] A pilot with decentralized, local eel management was conducted in the province of Friesland starting in 2011 and was fully implemented in the EMP in 2018, allowing fishermen in Friesland to fish during the closed season with a quota based on catches in 2010.

analysis (*Figure 2-2*) should be interpreted as the probability of becoming a silver eel at a certain length once that length has been achieved. For example, the ~60% for females at 100cm length does not mean that ~60% of the original number of females have already become a silver eel. Instead, an eel of 100cm length that has not yet matured has a ~60% chance of becoming a silver eel in the present year. In the previous reports (Bierman 2012, van de Wolfshaar et al., 2015 & 2018) the maturity key was fitted with logistic regression. The logistic regression fit should be interpreted as the proportion of eel that has become a silver eel at a certain length. However, because eel migrate from the system after becoming mature, this interpretation does not fit the data very well. The fit of the maturity key can have a large impact on the estimation of the proportion of eel becoming mature, especially in waters were eel are relatively large.

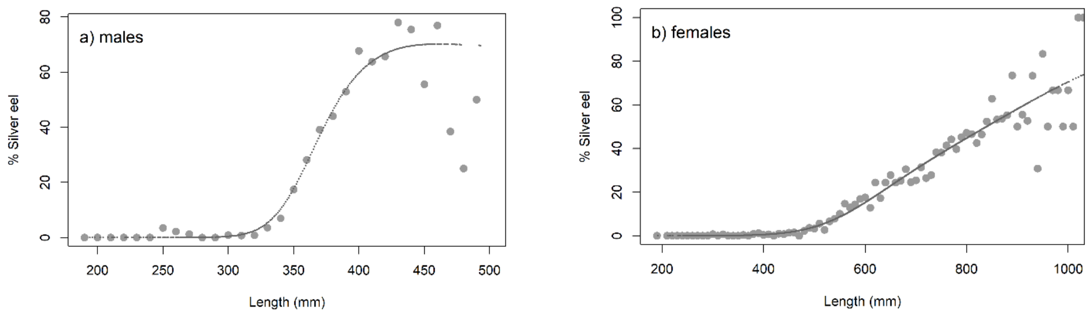

Figure 2-2 Observations (circles, average per 10mm class) and predicted GAM fit (lines) of the percentage of silver eel per length class (10mm). Data source: market sampling program (2006-2020). a) males (N = 2,648), b) females (N= 10,311).

2.5 Weight at length

A length-weight (LW) relationship is used to estimate eel biomass given numbers-at-length. The length - weight relationship is calculated for females and males separately, using individual length and weight measurements from market samples (*Figure 2-3*) and the standard LW relationship (*weight = exp (-a+ b*log(Length)).* In previous reports (Bierman et al. 2012 and van de Wolfshaar et al. 2015 & 2018) no distinction was made between males and females. For consistency with the other keys, the distinction was made for this report. However, for lengths <50 cm (which is more or less the maximum length of males), differences between males and females are very small and are not expected to make much of a difference.

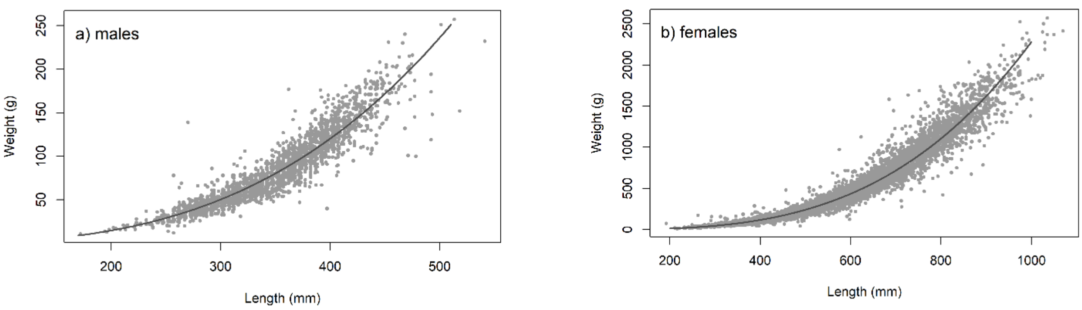

*Figure 2-3 Length-weight relationship for eel based on market sampling data (2006-2020). N (males) = 2,649), N (females) = 10,310. Estimated relationship for males: weight = exp (-13.53 + 3.057*log(L)), for females: weight = exp (-14.71+ 3.247*log(L)). With weight in grams and length (L) in millimeters.*

2.6 Growth

As in previous assessments, growth is also analyzed for males and females separately. Growth increments were based on otolith readings from eels collected between 2009-2019 (*Figure 2-4*). Individual growth curves were constructed using the relative distances between annual growth rings and scaling these to the total length of the eel (van Keeken et al., 2011). For the determination of growth curves and ages, the protocols set by the ICES workshop in Age Reading of European and American Eel (ICES WKAREA, 2009) were used. For age 0, the mean length of glass eels arriving in Den Oever was used (7.1cm). The sex of glass eels is not determined yet and therefore no distinction could be made between glasseel that would become males or females. The sex specific growth curve was constructed using a von Bertalanffy Growth fit (*VBLG*, *Figure 2-4*). The estimated growth curves are used in the demographic model as the annual transition rates between length classes. The VBLG fit differed from the previous used growth curve, where a cumulative growth fit was used (Bierman et al. 2012 and van de Wolfshaar et al. 2015 & 2018). The VBLG was chosen because it reflected the data better compared to the cumulative growth.

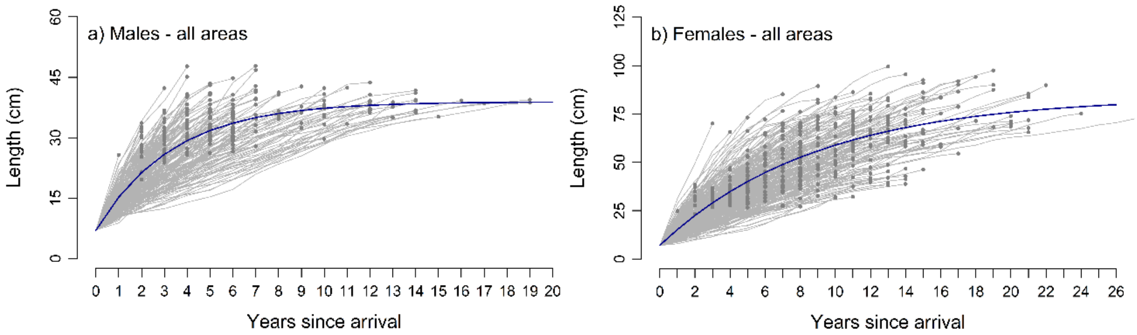

Figure 2-4 Eel growth. Grey lines indicate growth trajectories of individuals based on increments, dark grey dots are final length and age estimates (age after arrival at the Dutch coast) at the time the eel was caught. Blue lines: estimated growth using a von Bertalanffy fit. a) Males (N=288), b) females (N=187).

2.7 Natural Mortality

Natural mortality is a difficult parameter to assess. It depends on many factors, such as predation, water temperature and food availability. The natural mortality used in the demographic model (*Chapter 3*) is set to μ= 0.138 (per year) for all ages and lengths. This estimate is based on Dekker (2000), who made a best guess based on literature and is also used in other eel models (van der Meer, 2009). However, the above mentioned factors cause the value of natural mortality to be highly uncertain.

2.8 Landings per stage and period

Reporting of landings only became obligatory after the EMP came into place (end of 2009). Therefore, for the first period (2006-2008) a reconstructed estimate made by the ministry (Ministry of Agriculture, Nature and Food quality (LNV), 2009, 2018; *Table 2-1*) was used, 805 tonnes (525 tonnes yellow eel and 280 tonnes silver eel). For the second period (2009-2011) the average amount of reported commercial catches for 2010 and 2011 were used (410 tonnes, Ministry of LNV), because the data for 2009 were incomplete. For the other periods (2012-2014, 2015-2017, 2018-2020), an average estimate of the 3 years within the respective periods was calculated from the reported landings (*Table 2-1*). The catch was split into yellow and silver eel based on the length frequency distribution, the sex ratio, maturation and the length-weight relationship (see paragraphs above). This resulted in an estimate of 56% yellow eel in biomass of the total amount of retained catches in the Netherlands. Recreationally retained freshwater catches were available biennially starting in 2010 (Van der Hammen, 2017, & van der Hammen in prep). Therefore, for the first period (2006-2008), the estimate that was also used in the EMP was used (200 tonnes, Ministry of Agriculture, Nature and Food quality, 2009). For the other periods, the estimates from the biennial survey were used. For 2012-2014, two estimates were available, from which the average was taken. It is assumed that the recreational catches consist only of yellow eel[2]. The sum of retained commercial and recreational catches decreased from 2006-2008 to 2015-2017, but increased from 2015-2017 to 2018-2020 (*Table 2-1*). This increase is mainly caused by an increase in landings in lakes IJsselmeer and Markermeer, while most other areas did not show such an increase (Appendix A0).

Table 2-1 Overview of average yearly fresh water commercial and recreational retained catches (landings) in tonnes for each period.

	Commercial			Recreational	Total
Period	**Total**	**Yellow eel**	**Silver eel**	**Yellow eel**	**Commercial + Recreational**
2006-2008	805	525	280	200	1005
2009-2011	410	234	175	75	485
2012-2014	327	187	140	36	363
2015-2017	334	191	143	10	344
2018-2020	469	268	201	10	479

2 Recreational fisheries consist of > 95% of angling. As silver eel do not feed, likely that anglers catch mostly yellow eel.

3 A static spatial model for yellow and silver eel

3.1 Introduction to the model

Only the main rivers (Rhine, Waal, Meuse and IJssel) and the large lakes (IJsselmeer, Markermeer, Grevelingen and Randmeren) are managed at a national level (*Figure 3-1*). All other water bodies are managed regionally by the water boards. A consequence of this management system is that the monitoring of nationally and regionally managed water bodies differs significantly.

The regionally managed water bodies make up around 65% of the total freshwater surface area in the Netherlands (PBL, 2010). These waters are surveyed in a standardized manner since the implementation of the European Water Framework Directive (WFD) in 2000 (2000/60/EC). The nationally managed rivers have been monitored in a standardized manner since 1997. Both the regionally managed water bodies and the nationally managed rivers are monitored with an electric dipping net in the riverbanks.

For the (nationally managed) large lakes (IJsselmeer, Markermeer, Grevelingen and Randmeren) good quality survey data were either not available (Grevelingen) or considered unsuitable for the methods as used for the regionally managed waters or nationally managed rivers. Therefore, stock estimates for the large lakes were based on a different estimation method (a demographic model, see *Chapter 4*).

With the exception of the large lakes, the standing stock of both regionally and nationally managed waters was estimated by a swept area estimate. This is a simple method were eel density is multiplied with the water surface area. To calculate eel density, three estimations are needed: (1) the survey density (or catch success) of yellow and silver eel in a survey, (2) the catch efficiency of the survey gear and (3) the habitat distribution of eel at the survey locations (% eel in the shore versus % eel in the open water). The survey density (catch success) is estimated based on the catches (number/ha) in the survey per length class. This was subsequently translated into silver eel and yellow eel based on a maturity-at-length key, a weight-at-length key and a sex ratio key (*Chapter 2*). Subsequently, the standing stock was estimated for three scenarios, with different assumptions on the catch efficiency of the survey gear and the spatial distribution of eel in the water body. In this chapter, these scenarios will first be described. Then, the estimations of survey density for the regionally managed waters and for the nationally managed waters will be presented and subsequently biomass estimates for these three scenarios will be presented. These estimations are used as input for the Dutch eel stock biomass estimation (*Chapter 5*).

3.2 Three scenarios for the static spatial model

Standing stock estimates for three scenarios that differ in catch efficiency of the electric dipping net and habitat preferences were calculated to account for the major uncertainty.

3.2.1 <u>Catch efficiency</u>

The catch efficiency of survey gear is difficult to assess. Also, the catch efficiency of the electric dipping net depends on the type of water body, the substrate, the time of day, the settings of the gear and the experience of the staff operating the gear (Beaumont et al., 2002). Estimates of catch efficiencies of eel by electrofishing gear are scarce in the scientific literature. Naismith & Knights (1990) assumed a catch efficiency for eel using electrofishing gear of 27% in a river, whereas Baldwin & Aprahamian (2012) estimated efficiencies of approximately 60% in small rivers. Aprahamian (1986) showed size-selective effects of electrofishing, with mean probabilities of capture from 36% for the smallest eels to 59% for the largest. Carrs et al. (1999) reported estimated capture probabilities of 71.5% and 75.1% for lakes and streams, respectively. Belpaire et al. (2018) in an evaluation of the Belgian eel management plan assumed catch efficiencies of 66%.

3.2.2 Habitat preference

Monitoring with an electric dipping net in rivers is usually done near the shore. However, the distribution of eel is not equal between the shore and the open water. This habitat preference is important to consider when scaling biomass at the borders of a water body to the biomass for an entire water body. Eel may prefer the littoral ('inshore') over the open water ('offshore', e.g. Jellyman & Chisnall, 1999;, Schulze et al., 2004). Therefore, a correction was used to account for differences in eel density between the littoral zone and the open water.

The distribution of eels in lakes and rivers is generally thought to depend on the physical and biological characteristics of each water body. Literature on how eel is distributed over a water body is scarce and focuses on the relation between eel density and the distance to the shore, mainly in lakes. Contradicting results were found for lakes; Chisnall & West (1996) found that eel densities offshore in New Zealand lakes were on average 40% of those inshore; Schulze et al. (2004) found a decrease in number with water depth for a reservoir, but did not take the distance to shore into account; Jellyman & Chisnall (1999) and Yokouchi et al. (2009) found a positive relationship between catch per unit effort (*CPUE*) and proximity to the shore. Several others, more-recent studies have found contradicting results for the depth and distance to shore occupation of eels in lakes and estuarine environments (Walker et al., 2014;, Barry et al., 2016;, Bašić et al., 2019). Matsushige et al. (2020) found four different rivers habitat preferences of *Anguilla japonica* that suggested diversification of habitat with growth and that differences in the preferred substrate type depended on body size at the channel scale within these river systems. Despite the contradicting results, the estimated eel densities in habitats that resemble lakes and rivers in the Netherlands tend to be higher near shore compared offshore. Therefore, this is also assumed to be the most likely scenario for the Dutch national waters.

In the EMP's of some of the countries neighboring the Netherlands, different assumptions were made. In Belgium, the density of eels is also assumed to be highest near the shores. To estimate the offshore density, they multiply the inshore density with the outcome of a cumulative Gaussian distribution of the difference between half of the river width and half of the transect width (Stevens et al., 2013; Belpaire et al., 2018). In France, no difference is made between inshore and offshore areas in rivers given the lack of evidence otherwise (Briand et al., 2018).

3.2.3 Three scenarios

Estimates of eel standing stock were computed using three different scenarios that differ in catch efficiency of the electric dipping net and in the habitat preference (*Table 3-1*).

In scenario 1 a high catch efficiency (66%) and low proportion of eel in the offshore area compared to the inshore area (33%) is used (*Table 3-1*). As a consequence, this scenario will lead to the lowest estimated standing stock of eel. In scenario 2 the catch efficiency of the survey gear is assumed to be 20% (following an EU certified protocol, STOWA Handboek Visstandbemonstering 2003) and the proportion of eel in the offshore area compared to the inshore area is assumed to be 50% (*Table 3-1*). This scenario leads to an intermediate estimate of the eel standing stock. In scenario 3, the same estimate for catch efficiency is used as in scenario 2 (20%), but the proportion of eel in the offshore area compared to the inshore area is higher (66%). Scenario 3 will therefore lead to the highest estimate of the standing stock. Scenario 2 is the best guess estimate. All final calculations will be made with scenario 2, unless stated otherwise (*Table 3-1*).

Table 3-1 The three main scenarios used in the approach to stock assessment in which survey data are scaled to wetted areas. A best guess of 20% for catch efficiencies was used with an upper limit of 66%. Densities in areas of water bodies outside 1.5 meters of the shore/bank ("offshore area") were assumed to be either 33%, 50% or 66% of densities within 1.5 meters of the shore/bank ("inshore area").

	Density "offshore" compared to "inshore"		
Catch efficiency	**33%**	**50%**	**66%**
66%	Scenario 1		
20%		Scenario 2	Scenario 3

3.3 Regionally managed water bodies

3.3.1 <u>GIS data</u>

The eel biomass in the regionally managed water bodies was assessed in the same way as presented in previous reports (Bierman et al, 2012; van de Wolfshaar et al, 2015 & 2018). It is based on the WFD fish monitoring program and detailed GIS maps. The management of WFD waters is executed by 21 so-called water boards (*Figure 3-1*). In the Netherlands, all WFD surface waters are assigned to a waterbody type, ranging from small ditches to large lakes (*Table 3-3*). Detailed information per waterbody is obtained from a publicly available GIS map with polygons and line elements of all WFD surface waters in the Netherlands (*Figure 3-1*). The spatial information of the WFD waterbodies in these GIS maps makes it possible to calculate the area of each type of surface water (van Puijenbroek & Clement, 2010).

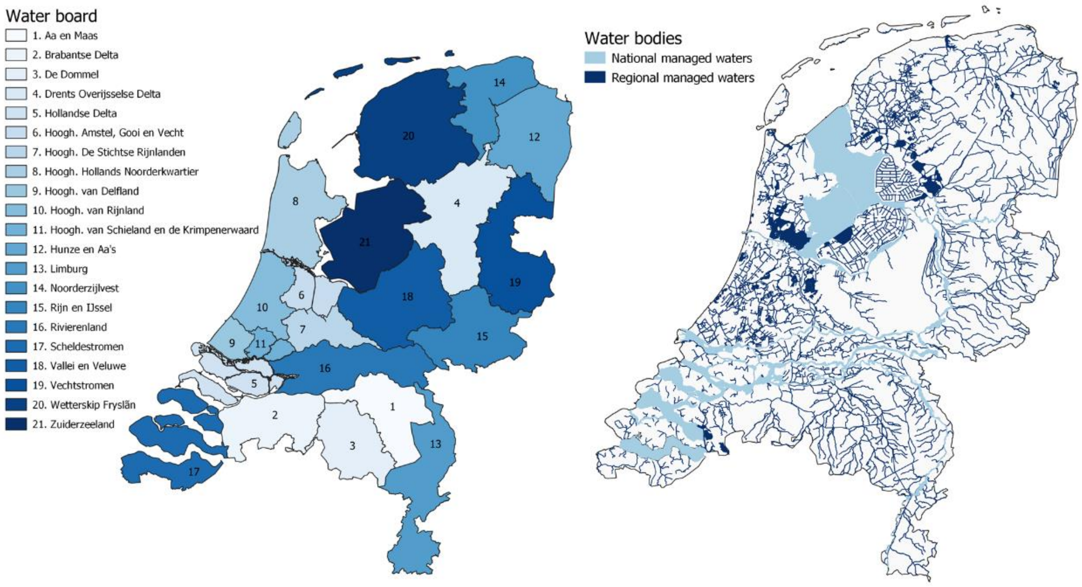

Figure 3-1 Left: the 21 water boards that are in charge of the regional management of WFD waters in the Netherlands (left). Right: The WFD waterbodies (dark blue) and the nationally managed waters (light blue).

3.3.2 <u>Data availability</u>

<u>WFD waters</u>

Eel monitoring within the regionally managed WFD waters was executed with an electrofishing gear, following an EU certified protocol (STOWA Handboek Visstandbemonstering 2003). Sampled waterbodies are expected to be representative for water types as defined in WFD regulation. Water boards are obliged to sample their WFD waters within a time frame of six years, resulting in a different sampling scheme for each waterboard. Most water boards sample a part of their area every year, while others sample a large area within one year, but do not sample every year. Data availability on a yearly basis is thus not necessarily expected. In this report, the following approach was used to select the periods. To reduce the variation due to unbalanced sampling, a moving average of six-year periods was chosen to assess the biomass of eel for the different three-year periods. For the three-year periods in this report, a six-year period starting two years before and one year after the corresponding three-year period was chosen, so that in total six years of data were used. Because data from before 2006 was not available, for the first period (2006-2008) data from 2006-2011 was used. Similarly, 2020 and 2021 data were not yet available from any water board due to the timing of the data request (spring-summer 2020). This early timing is necessary due to the time needed by some of the water boards to deliver the data and to process the data into the right format. Therefore, for the last three-year period (2018-2020), a six-year period starting in 2014 (2014-2019) was used. Because a six-year moving average was used, only the waterbodies that had at least one fishing event in each six-year cycle since 2009 were selected in the analysis, in order to keep the data balanced between the three-year periods. However, the years before 2009 were not used to select the waterbodies included in the analysis, because in these years sampling intensity was much lower and this would result in too much data loss over all years. This results in that the first period (2005-2008) is based on fewer waterbodies than the other periods, whereas the other periods have the same waterbodies included in the analysis. In addition, some WDF data could not be used in the analysis for different reasons. For example, many of the regional waters of water board "Scheldestromen" are brackish, and therefore there was hardly any electrofishing data in this waterboard. Data for water board "Hollandse Delta" is missing for the period 2014–2020 and could therefore also not be used for the other years due to the selection criteria (*Table 3-2*).

Table 3-2 Data per water board per year which were used for the analysis of the standing stock of eel in WFD water bodies (white numbers); white boxes represent missing/not available/incomplete data; blue boxes with a zero represent that data was available, but could not be used due to the selection criteria (see text).

Water board	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Aa en Maas		2		55			66	3		27	1	48	32	27
Brabantse Delta	13	7	52		54	65	23	51			74	30	29	71
De Dommel	27	12		51	16	34	35	25	27	66	3	40	34	56
Drents Overijsselse Delta	19	50	5	5	5	44		8	41	35		63	6	39
Hollandse Delta	0							0						
Hoogh. Amstel, Gooi en Vecht	176	10	16	21	56		140	184		31	71	65	35	102
Hoogh. De Stichtse Rijnlanden	17			19	5		19	22		20	5		19	24
Hoogh. Hollands Noorderkwartier			9	38	14	77			11	20	35	8	59	70
Hoogh. van Delfland			4			54	26		32	29		35	29	
Hoogh. van Rijnland	4			7	18	27	11	9	71	4	5	73		43
Hoogh. van Schieland en de Krimpenerwaard	10	16	11	52	14	25	41	13	59	50	14	52	55	17
Hunze en Aa's		33	23			46	43	75	18	32	38	47	29	55
Limburg		5	24		20	8	17	18	33	35	8	41	31	17
Noorderzijlvest				48	22	60	23	34	31	23	39	50	25	50
Rijn en IJssel	65	18	29	2	26	34	23	32	25	19	19	27	26	35
Rivierenland		46		152	116		99		165				239	
Scheldestromen												0	0	0
Vechtstromen		77	13		74	32	17	54	29	45	84	1	5	60
Vallei en Veluwe			11	68	16	33	48	38	59	49	52	73	59	77
Wetterskip Fryslân	17			49			55			63			57	28
Zuiderzeeland					0			6						6
Number of fishing events	348	276	197	567	456	543	686	572	601	548	448	653	770	777

Sampling locations were included if they were located within WFD waterbodies (polygons) or 50 meters from a line element as defined in the available GIS map. A margin of 50 meters from a line element was assumed to be reasonable since waterbodies having a width of about 100 meters were defined as line elements in the GIS map. To link the electrofishing sampling locations to the GIS map, the geographic coordinates of the electrofishing events were used. Firstly, coordinates which fell into a polygon were assigned to that polygon. Secondly, for the fishing events which could not be assigned to a polygon, each was assigned to the nearest line segment if this was within the margin of 50 meters from the sampling location. Thirdly, for all remaining fishing events without a match, based on the above-mentioned statements, the waterbody identification code was used to find a match. However, this last attempt to link a fishing event with a waterbody resulted in only a few matches since different identification codes/names are in use for a single water body, and might change over time (e.g. after a fusion between waterboards). The remaining fishing events after this last step were excluded from the analyses as a result of lacking information. Finally, only the selected fishing events, in which the sampled area was known, were used for the analysis. In total, the selection method resulted in 7,442 electrofishing events in 533 water bodies being included in the eel assessment for regionally managed WFD waters (*Table 3-2*; *Figure 3-2*).

The variability in sampled area is large between the different water types (*Table 3-3*, see *Appendix A1* for a description of each water type). The two water types with the largest surface area (M14 and M27, both shallow, relatively large lakes, with 28% and 30% of the total surface area, respectively) have a relatively low sampling intensity. The highest sampling intensity (M3 with 17% of the sampling effort and R5 with 22%) has been applied to water types with a relatively small surface area (4% and 2%, respectively). Nevertheless, most of the small ditches (M1a and M2) in the Netherlands are not even assigned as a WFD waterbody and were thus not included in the WFD sampling program. Those non-WFD ditches cover a large area of > 59,000 hectares in total (*Table 3-3*, '*Non-WFD ditches'*) and can subsequently contribute significantly to the standing stock of eel. In order to include these ditches, information of an additional fish sampling ("*Polderbemonstering*") within these non-WFD water bodies was incorporated to estimate the total biomass of eel in ditches (van Keeken, 2014a & 2014b).

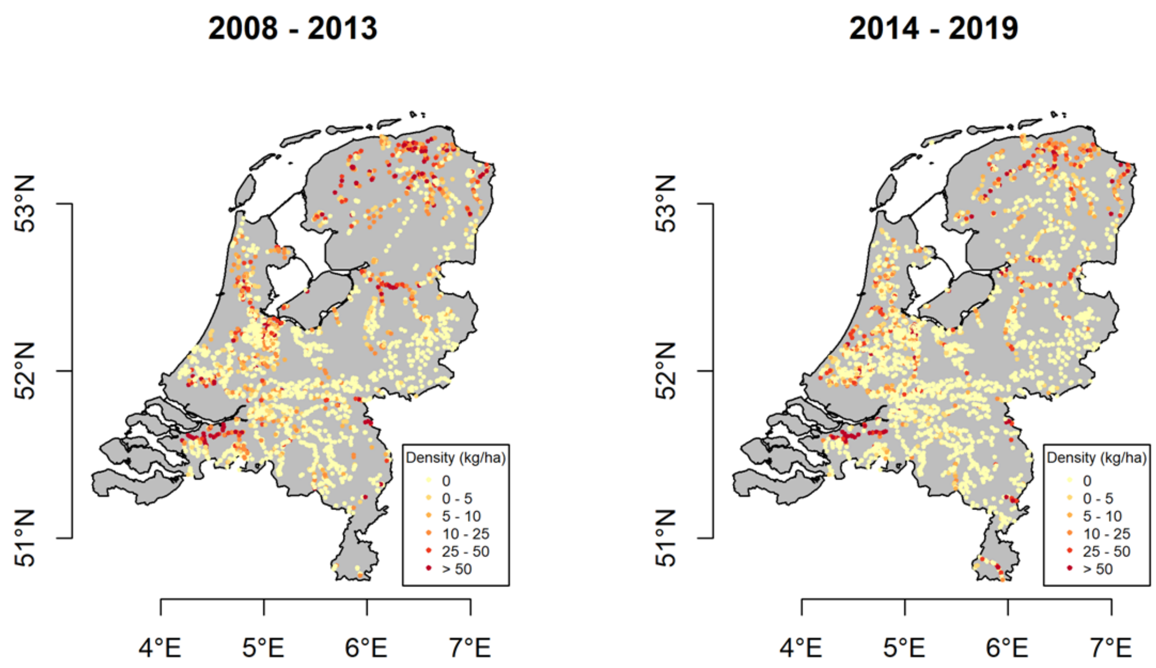

Figure 3-2 Geographical location of fishing events in WFD waters included in the analysis for two different periods (six year cycles), 2008-2013 and 2014-2019. The density (kg/ha) is indicated by the different colors, ranging from 0 (yellow) to >50 kg/ha of eel (red).

3.3.3 Non-WFD waters (ditches)

Eel monitoring of non-WFD ditches was also executed with an electronic dipping net, following the same protocol as the WFD sampling program. Each year, from 2013 onwards, several ditches within a selection of water boards were sampled in a way which would be representative for each waterboard. Most small ditches can be found in the lower parts of the Netherlands ("Polders"). Therefore only 14 out of the 21 waterboards were included within this additional sampling program. In total, 350 electrofishing events were executed, whereby an area of 12.4 hectares was sampled in non-WFD ditches (*Figure 3-3*) and included in the eel assessment for regionally managed waters. Except for the first two years (2013-2014) of the program, in which the sampling was conducted in (early) summer, the ditches were sampled in September.

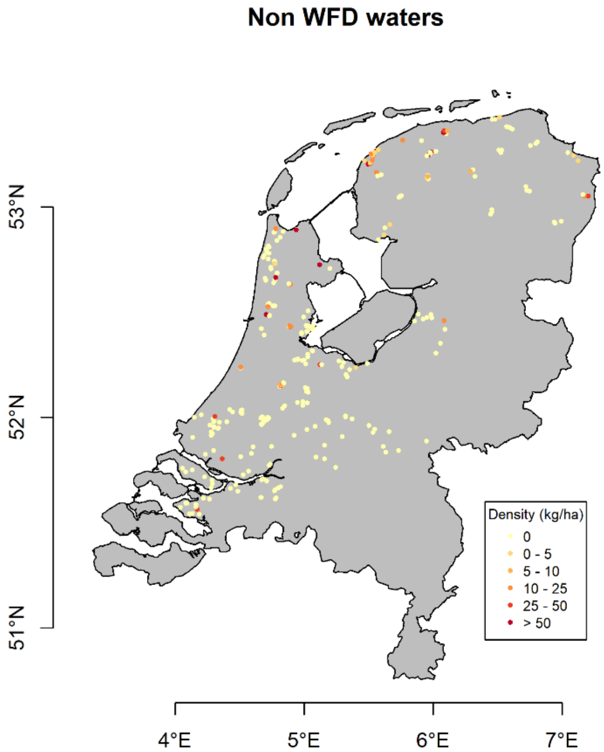

Figure 3-3 Geographical location of fishing events in non-WFD waters included in the analysis for all years within 2013-2020. The density (kg/ha) is indicated by the different colors, ranging from 0 (yellow) to >50 kg/ha of eel (red). Water boards included in this additional program are: Brabantse Delta; Hollandse Delta; Hoogh. Amstel, Gooi en Vecht; Hoogh. De Stichtse Rijnlanden; Hoogh. Hollands Noorderkwartier; Hoogh. Van Delfland; Hoogh. Van Rijnland; Hoogh. Van Schieland en de Krimpenerwaard; Hunze en Aa's; Noorderzijlvest; Rivierenland; Scheldestromen; Vallei en Veluwe; Wetterskip Fryslân.

3.3.4 Standing stock estimation

For each fishing event, the number of eel per length were converted to weight by making use of a length-weight relationship (*Figure 2-3*), from which the survey density of eel (in kg/ha) was calculated. Densities were corrected for the assumed catch efficiency of the electric dipping net (see *Paragraph 3.2.3*). Water surface area was divided into two areas: littoral zone (inshore) and open water (offshore). The width of the littoral zone was set equal to the reach of the dipping net (1.5 meters) and its surface area is the width times the bank length. The open water surface area is the total surface area minus the surface area of the littoral zone. Eel density outside the littoral zone is assumed to be a fraction of that in the littoral zone (50% for scenario 2). Subsequently, density is converted to absolute biomass (kg) for the riverbank and open water surface areas separately. Biomass of silver eel and of all eel (≥ *30 cm)* is estimated according to scenario 2 (*Table 3-8*).

For upscaling to the total biomass in regional waters, the surface area of each water type was used to estimate the total biomass (in tonnes) of eel (≥30 cm), yellow- and silver eel combined) and silver eel (≥30 cm) for each water type. Based on 1) the female:male ratio (*Figure 2-1*) and 2) the maturity at length for both males and females (*Figure 2-2*), the density and biomass of silver eel was estimated. For water types that were not sampled in a six year-period, the density averaged over all water types was used to estimate biomass of eel and silver eel for these waters without data. For the additional sampling in ditches, the same methodology was used to estimate production and total biomass of eel and silver eel within these waters.

3.3.5 Standing stock per WFD water type

The density and biomass of eel and silver eel per water type was estimated for all defined six-year periods, so that each estimate covers a full sampling cycle of six years. The result of the latest six-year period (2014-2019) is presented in *Table 3-3*. Following scenario 2 (*Table 3-1*), a total biomass of 1,791 tonnes of eel (≥ 30 cm) was estimated, of which 399 tonnes silver eel, in regionally managed WFD water bodies. The highest survey densities were estimated for the R14 (22.1 kg/ha) and R18 (19.5 kg/ha) water types. However, the surface area of these waters is very small and the contribution to the total biomass is limited (*Table 3-3*). Contribution of M14, M27 and M20 waters to the total biomass of eel and silver eel was estimated to be the highest, mainly because these water types have a large surface area (*Table 3-3*). The estimated biomass of eel in WFD waters (1,791 tonnes) combined with an estimated biomass of 981 tonnes (survey density 3.8 kg/ha) for eel in non-WFD waters resulted in a total estimated biomass of 2,773 tonnes of eel in regionally managed waters for the period 2014–2019 (*Table 3-3*). The total biomass estimate of silver eel in regionally managed waters for the period 2014–2019 was 584 tonnes (*Table 3-3*). Similar density and biomass estimates were done for the other periods (*Appendix A3*).

Table 3-3 Estimation of the eel survey density and biomass per WFD water type for the period 2014 – 2019 following scenario 2. Production and biomass estimates were done for yellow and silver eel combined (≥ 30 cm) and for silver eel (≥ 30 cm) only. For a full description of the WFD-water types, see Appendix A1 and for the biomass of eel (≥ 30 cm) per period see Appendix A2. Note that survey density is not the density in the lake, but the 'catch success' in the survey. This value is corrected for selectivity of the gear and the ratio between the inshore and offshore area to calculate the total biomass.

				All eel (≥ 30 cm)		**Silver eel (≥ 30 cm)**	
WFD water	**Description of WFD water types**	**Total area (ha)**	**Swept area (ha)**	**Survey Density (kg/ha)**	**Biomass (tonnes)**	**Survey Density (kg/ha)**	**Biomass (tonnes)**
M1a	Buffered ditches	156	11.09	0.4	0.3	0.1	0.0
M2	Weakly buffered ditches	10	1.48	3.9	0.2	1.9	0.1
M3	Buffered canals	3,324	67.51	1.3	13.3	0.4	3.8
M6a	Large shallow canals (shipping)	603	14.32	7.6	13.7	2.4	4.3
M6b	Large shallow canals	1,780	7.7	4.8	25.0	0.8	3.9
M7a	Large deep canals (shipping)	13	0*	7.0	0.3	1.7	0.1
M7b	Large deep canals	3,435	11.77	9.8	91.6	2.3	21.0
M8	Buffered peatland ditches	1,148	11.06	0.1	0.2	0.0	0.1
M10	Peatland canals	1,362	32.66	1.9	9.2	0.3	1.6
M14	Shallow, large, buffered lakes	20,902	31.58	17.6	936.3	4.5	240.3
M20	Deep, large, buffered lakes	4,444	3.25	11.1	125.5	2.3	26.4
M23	Shallow, large, calcium rich lakes	90	0*	7.0	1.7	1.7	0.4
M27	Shallow, large, peatland lakes	22,738	22.89	6.5	372.7	1.0	56.6
M30	Weakly brackish waters (0.3 – 3 g Cl/l)	8,182	6.43	1.3	27.0	0.3	6.1
R4	Slow flowing, upper stream on sand	73	14.42	0.4	0.1	0.1	0.0
R5	Slow flowing, lower stream on sand	1,221	81.11	1.7	7.5	0.3	1.6
R6	Slow flowing small river on sand/clay	3,414	34.37	12.7	114.1	2.4	21.2
R7	Slow flowing side stream on sand/clay	2,272	3.16	7.9	45.3	1.6	9.4
R8	Fresh tidal waters on sand/clay	20	1.05	5.6	0.3	2.4	0.1
R12	Slow flowing lower stream on peat	65	3.70	3.5	0.8	0.9	0.2
R13	Fast flowing upper stream on sand	4	0*	7.0	0.2	1.7	0.0
R14	Fast flowing lower stream on sand	16	0.93	22.1	1.3	5.5	0.3
R15	Fast flowing small river (siliceous)	37	0*	7.0	0.8	1.7	0.2
R17	Fast flowing upper stream (calcium)	7	0*	7.0	0.3	1.7	0.1
R18	Fast flowing lower stream (calcium)	52	1.44	19.5	3.7	4.6	0.9
Total		75,368	361.9		1,791.4		398.7
Non-WFD ditches		59,441	12.4	3.8	981.2	0.7	185.3
TOTAL		**134,809**	374.3		**2,773**		**584**

* For those water types were no survey was conducted, the average survey density of all water types was assumed.

3.3.6 Biomass per period and scenario

Three different scenarios (*Table 3-1*) were used to estimate the eel biomass, based on different values of catch efficiency and different ratios between eel densities at the inshore and offshore areas. Eel biomass estimates vary between scenarios, with scenario 1 providing the lowest and scenario 3 the highest estimate of eel biomass (*Table 3-4*). In addition to the different scenarios, estimates were made for the different periods and for all data combined. The difference in the biomass estimates between the periods is large. Biomass estimates for the first and the last period were the lowest, while the highest estimates were seen for the periods 2009–2011 and 2012-2014. Although the estimates are made for a six-year period and only from waters that are sampled at least once in each such period, with an exception of the first period, the differences can still reflect (to some degree) an unbalanced sampling. If the same waterbody is sampled at least once in each period, it can still be sampled on a different location within the waterbody or by a different person, which can cause some variation. In addition, although six years of data (2006-2011) was used for the estimate of the first period (2006-2008), it still had lower sampling effort compared to the sampling effort in the later periods. This was not the case in the latest periods, where a decline is observed. The last two periods had nearly an equal sampling effort and the same waterbodies were included in the analysis. Therefore, the difference in estimated biomass of eel in these last two periods (2015-2017 and 2018-2020) compared to the period before (2012-2014), reflects a lower estimate in the standing stock of eel in regionally managed waters.

Table 3-4 Estimates of standing stock of eel in tonnes in the regionally managed waters (WFD water bodies) and non-WFD water bodies; all eel (yellow and silver ≥ 30 cm) and silver eel (≥ 30 cm) biomass estimates for three periods (and all years combined for the Non-WFD waters) for the three scenarios.

		Non-WFD waters	**WFD water bodies**				
	Scenario	**All years**	**2006 – 2008***	**2009 – 2011***	**2012 – 2014***	**2015 – 2017***	**2018 – 2020***
Eel ≥ 30 cm	1	283	406	664	776	464	369
	2	981	1,947	3,236	3,793	2,265	1,791
	3	1,024	2,519	4,219	4,953	2,955	2,331
Silver eel ≥ 30 cm	1	54	69	109	200	148	82
	2	185	328	529	984	727	399
	3	193	424	689	1,289	952	519

* these are the three-year periods. Each estimate is based on the nearest six-years of data. Period “2006-2008”: data from 2006-2011; Period “2009-2011”: data from 2007:2012; Period “2012-2014”: data from 2010:2015; Period “2015-2017”: data from 2013-2018;Period “2018-2020”: data from 2014:2019.

3.3.7 Discussion

There are some limitations in the data availability concerning the regionally managed waters. The first issue is that not all water boards sample at least once every three years. Water boards are obliged to sample their WFD waters within a time frame of six years, resulting in a different sampling scheme for each water board. In addition, due to the timing of the data request, the data of 2020 was not yet available from any water board. A second issue concerning the WFD sampling program is that the sampling intensity was not well-balanced. Water types with the highest surface areas have relatively low sampling effort, while the highest sampling effort was performed in water types with relatively (very) low surface areas. A six-year moving average was chosen here, such that a trend can be estimated and changes in eel abundance over time are accounted for. By calculating a six-year moving average, a trend is estimated that takes a great part of the unbalanced sampling into account. For the last two periods, nearly all water types were sampled with sufficient sampling effort. However, especially in the first period, there was less sampling effort, which will have influenced the result. Another issue is that not every fishing event could be linked to a water body and these events had to be excluded from the analysis. This mismatch might be due to measurement errors with GPS equipment, errors during data entry or the selection method of sampling locations used in this analysis. This resulted 3,704 of the fishing events out of 12,518 having to be removed. As in previous reports, three scenarios were used for catch efficiency and spatial distribution of eel within a habitat, which pose issues that remain problematic and cause a high level of uncertainty in the absolute biomass estimate (*Table 3-4*). Variation in the biomass estimates in some waters, may also be a result of stocking activities. This is not a problem for the biomass estimation in this evaluation, because stocked eel are included in the biomass estimates. However stocking may cause large variation in eel biomass in the waterbodies were eel is stocked over the years. Finally, the usage of a new set of biological keys influenced the estimates of yellow and silver eel in the regionally managed waters (see *Chapter 2*).

In the non-WFD waters (ditches), the sampling scheme was standardized for sampling method, but not for sampling location (*Paragraph 3.3.3*). Each year, two different waterboards were selected and within a given waterboard and only a very small subsample of all ditches was monitored. As a result, only one single estimate of the non-WFD waters could be conducted over the whole time period. Any variation in time can therefore not be detected, because the variation between locations is assumed to be higher than between years within the same location.

3.4 Nationally managed water bodies

3.4.1 Data availability

Within the survey program "Fish Monitoring National Waters", fish species in the main Dutch rivers are monitored yearly (van Keeken et al., 2020a). In the program, the main rivers and the water bodies connected to them are sampled in autumn or early spring (*Table 3-6*). Six of the twelve regions have been sampled consistently and yearly since 1997 (*Table 3-6*). Within a region, sampling is usually consistently undertaken in the same month(s) of each year, but different regions are sampled in different months. There are also regions which started to sample later than the first period considered here (2006-2008), which have some missing years or which do not sample yearly. Consequently, in some waters, data is not available for every year within the three-year periods that are considered in this report. For example, Volkerak-Zoommeer has not been sampled in 2018 and 2020 and the latest estimate in this water is based solely on samples from 2019 (*Table 3-5*).

Table 3-5 Number of hauls per year per region per habitat (main or connected water).

Region	Habitat	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Benedenloop	Main	5	5	5	4	5	5	5	4	5	5	5	5	5	5	5
Gelderse IJssel	Connected	2	2	2	2	1	2	2	2	2	2	2	2	2	2	2
Benedenrivieren	Main	31	31	31	30	31	24	21	21	21	21	21	21	21	21	21
	Connected	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Gelderse poort	Main	22	26	25	27	28	29	28	14	14	14	14	15	14	14	10
	Connected	15	16	17	22	21	20	22	10	11	11	11	11	11	10	8
Getijdenlek	Main	7	7	7	7	7	7	8	7	7	7	7	7	7	7	7
	Connected	2	3	3	3	3	3	3	3	3	3	3	3	3	3	3
Getijdenmaas	Main	5	5	5	5	10	11	11	8	8	10	8	8	8	8	9
	Connected	7	7	7	7	9	9	9	9	9	9	8	9	9	9	9
Grensmaas	Main	11	11	11	11	11	11	11	10	11	11	11	11	11	10	11
	Connected	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1
Volkerak-Zoommeer	Main	0	0	9	0	9	0	0	9	0	0	9	0	0	9	0
Zandmaas	Main	0	0	4	4	4	5	6	0	0	4	0	3	4	4	0
	Connected	0	0	7	7	6	7	6	0	0	7	0	2	8	8	0

Due to Covid-19 restrictions, some (parts of) regions were not sampled in 2020. These include some stations in the upper reach of the River Gelderse IJssel, all stations in the River Rijn (both part of the Gelderse Poort region) and all stations in the Zandmaas (North and South). See *Figure 3-3* for the classification of regions and *Table 3-6* for an overview of survey details per region.

In the large lakes (lakes IJsselmeer and Markermeer, Randmeren and Grevelingen) the relationship between the eel density inshore compared to offshore is even more uncertain because of the large area offshore. Therefore, density estimates in lakes IJsselmeer and Markermeer from the demographic model are used as basis for these lakes instead (*Chapter 4*).

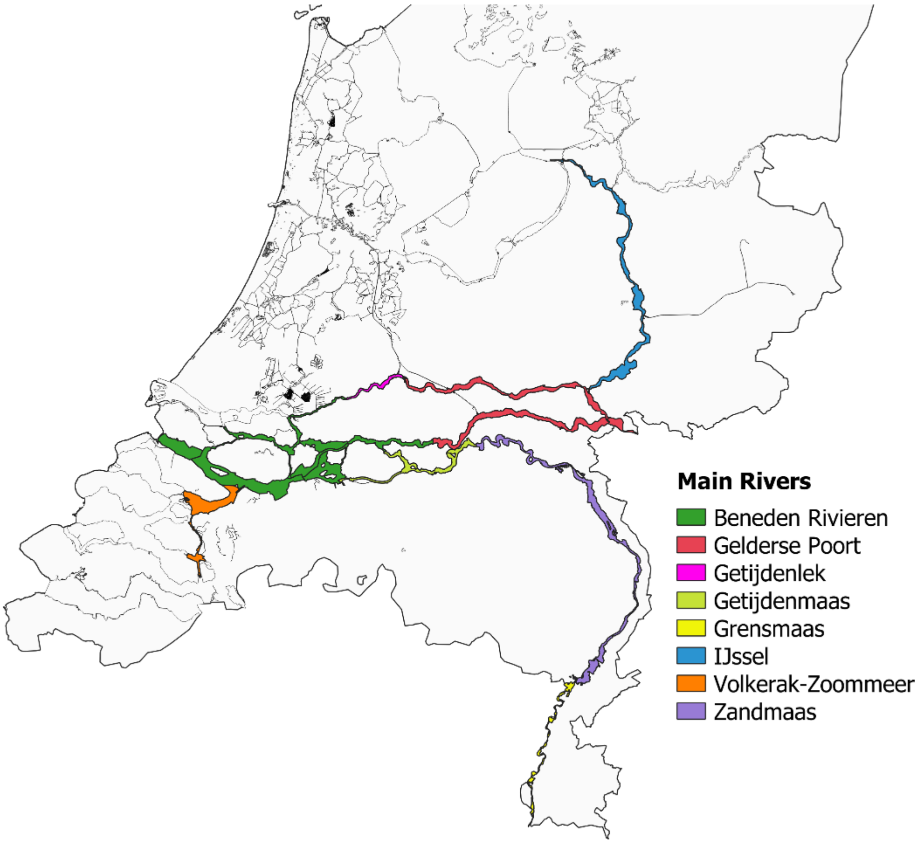

Figure 3-4 Classification of the main rivers. Regions are represented by different colors.

Table 3-6 Survey information per river region and type of water (main waterway or connected water body), for the years 2018, 2019 and 2020. Sampled years = the years in which a region has been sampled, where all = 2018+2019+2020. Survey density in riverbank = density for all eel ≥30 cm (yellow and silver). Survey density is based on data collected using an electric dipping net at the riverbanks. For this table, no correction for catch efficiency of the gear is made.

Region	**Habitat**	**Sampled years**	**Sample period**	**Survey density (≥30 cm) in riverbank (kg/ha)**
Benedenloop Gelderse IJssel	main	all	Spring	1.38
	connected			0.00
Benedenrivieren	main	all	Autumn	24.71
	connected			0.00
Gelderse Poort*	main	all	Spring	1.46
	connected			0.00
Getijdenlek	main	all	Autumn	11.86
	connected			6.05
Getijdenmaas	main	all	Autumn	3.66
	connected			1.25
Grensmaas	main	all	Spring	2.39
	connected			0.00
Volkerak-Zoommeer	main	2019	Autumn	35.05
Zandmaas*	main	2018, 2019	Spring	1.80
	connected			11.22

*Due to Covid-19 restrictions, some (parts of) regions were not sampled in 2020: some stations in the upper reach of the river Gelderse IJssel, all stations in the river Rijn (both part of the Gelderse Poort region) and all stations in the Zandmaas (North and South).

3.4.2 GIS data

Three types of geographical information were collected; surface area, bank length and groin length. The surface area (ha) and bank length (km) of the rivers and lakes were calculated (*Table 3-7*) using GIS data (the 'Ecotopenkaart' of Rijkswaterstaat[3]). For the rivers, additional information on bank length was collected (*Table 3-7*). In some parts of the rivers, the bank length is significantly larger than the river length because of groins (Dutch: 'kribben') which are placed perpendicular to the riverbank. These groins are approximately 90 meters long and placed 200 meters apart (www.rws.nl). In the parts of the rivers with groins, bank length is thus approximately 1.9 times the river length. By visually scanning satellite images of Google Earth, a rough estimate of the percentage of riverbanks with groins was made: 60% of the Gelderse Poort is estimated to have groins, and 50% of the Getijdenmaas. The other regions are assumed to have no groins. The estimates used are the same as in the previous assessments (Bierman et al. 2012; van de Wolfshaar et al., 2015 & 2018).

Table 3-7 Surface area, river length and bank length per river region. Groins = the percentage of a region that has groins. Bank length is river length with groins length (1.9 times the river length) included.

Region	Habitat	Surface area (ha)	River length (km)	Groins	Bank length (km)
Benedenloop Gelderse IJssel	main	675	118		118
	connected	271	42		42
Benedenrivieren	main	18,377	703		703
	connected	1,670	498		498
Gelderse Poort	main	5,201	557	60%	858
	connected	1,468	191		191
Getijdenlek	main	500	52		52
	connected	78	19		19
Getijdenmaas	main	1,265	155	50%	224
	connected	753	82		82
Grensmaas	main	426	135		135
	connected	436	49		49
Volkerak-Zoommeer	main	4,814	171		171
Zandmaas	main	2,043	305		305
	connected	1,413	160		160

3.4.3 Biomass estimate

Densities were corrected for the assumed catch efficiency of the electric dipping net (20% for scenario 2). Water surface area was divided into two areas: littoral zone (inshore) and open water (offshore). The width of the littoral zone was set equal to the reach of the dipping net (1.5 meters) and its surface area is the width times the bank length. The open water surface area is the total surface area minus the surface area of the littoral zone. Eel density outside the littoral zone is assumed to be a fraction of that in the littoral zone (50% for scenario 2). Subsequently, density is converted to absolute biomass (kg) for the riverbank and open water surface areas separately. For the Grensmaas, no correction for habitat preference is made and density in the open water is assumed to be equal to that in the littoral zone, because sampling with the dipping net takes place in the open water in this (shallow water) region and is thus representative for the open water density. Biomass of silver eel and of all eel (≥ *30 cm)* is estimated according to scenario 2 (*Table 3-8*).

[3] https://maps.rijkswaterstaat.nl/dataregister/srv/dut/catalog.search#/metadata/8a2sa797-915t-mn3s-pwnr-va1luhr81fos

Table 3-8 Biomass (tonnes) of eel ≥ 30 cm, yellow eel and silver eel per river region, estimated according to scenario 2, for 2018-2020.

	Biomass (tonnes)		
Region	**All eel (≥ 30 cm)**	**Yellow eel (≥ 30 cm)**	**Silver eel (≥ 30 cm)**
Benedenloop Gelderse IJssel	2.4	1.6	0.8
Benedenrivieren	1,141.9	896.8	245.1
Gelderse Poort	19.5	14.6	4.9
Getijdenlek	16.3	13.8	2.5
Getijdenmaas	14.3	10.8	3.5
Grensmaas	5.1	3.3	1.8
Volkerak-Zoom	424.1	315.7	108.4
Zandmaas	49.7	24.6	25.1

For scenario 2, estimated biomass of eel in the period 2018-2020 is also compared to the earlier periods (*Figure 3-5*;*Table 3-9*), showing that there does not seem to be a general biomass trend in all of the river regions. Most of the biomass can be found in the Benedenrivieren. After the steep increase of estimated biomass in 2015-2017 in most river regions, biomass estimates have decreased again to levels similar to those of 2012-2014 or lower. Exceptions are the Volkerak-Zoommeer, Getijdenlek and the Grensmaas regions. For the Volkerak-Zoommeer region, the biomass estimate from 2015-2017 was considerably lower than in 2012-2014 but has increased again in 2018-2020 to levels similar to those of 2012-2014. For the Grensmaas River region, there seems to be a steep decline since the period 2009-2011. For the Getijdenlek region, biomass estimates have increased from 2006-2008 to 2012-2014 and have remained relatively stable since then (*Figure 3-5*).

Table 3-9 Biomass of eel ≥ 30 cm (yellow and silver) in tonnes per river region, for five 3-year periods, following scenario 2.

Region	**2006-2008**	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
Benedenloop Gelderse IJssel	13.8	1.4	4.2	19.0	2.4
Benedenrivieren	311.0	412.8	908.0	2,147.0	1,141.9
Gelderse Poort	8.8	29.7	26.0	93.4	19.5
Getijdenlek	2.8	6.4	15.5	17.2	16.3
Getijdenmaas	15.0	8.1	42.1	90.7	14.3
Grensmaas	100.5	100.5	19.5	27.7	5.1
Volkerak-Zoommeer	131.3	874.1	381.3	101.8	424.1
Zandmaas	61.3	105.9	49.5	345.6	49.7
Total	**645**	**1,539**	**1,446**	**2,842**	**1,673**

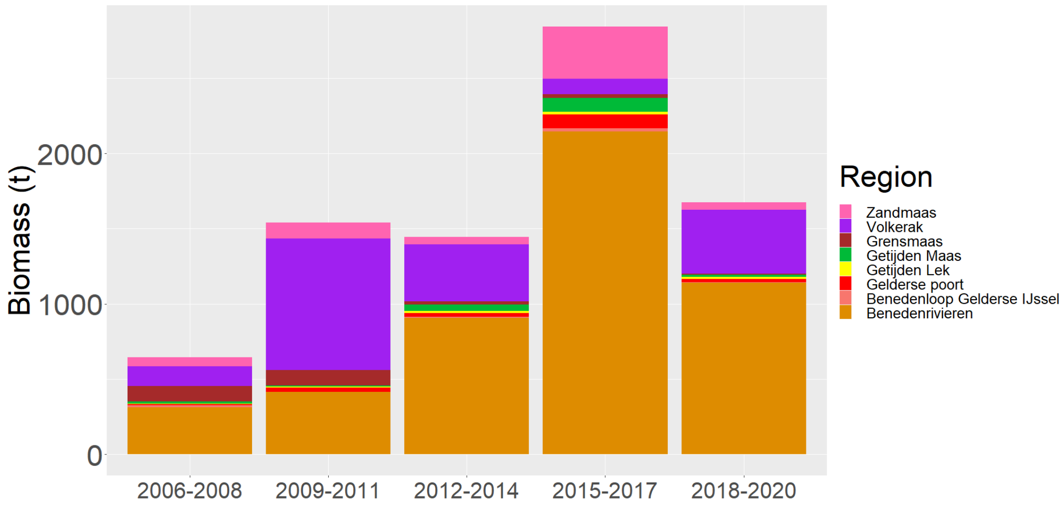

Figure 3-5 Biomass of eel ≥ 30 cm (yellow and silver) in tonnes per river region, for five 3-year periods, following scenario 2.

When combining all the river regions, the estimated biomass for the period of 2015-2017 is the highest, followed by the most recent biomass estimate from 2018-2020. The biomass estimate of this latter period ranges from 337 tonnes for scenario 1 to 2,200 tonnes for scenario 3. The best guess scenario 2 biomass estimate for all the river regions combined is 1,673 tonnes (*Table 3-10*).

Table 3-10 Biomass of all eel and of silver eel (≥30 cm) in tonnes per river region, for five 3-year periods, following each scenario.

		National water bodies				
	Scenario	**2006 – 2008**	**2009 – 2011**	**2012 – 2014**	**2015 – 2017**	**2018 – 2020**
Eel ≥ 30 cm	1	141	320	293	577	337
Eel ≥ 30 cm	2	645	1,539	1,446	2,842	1,673
Eel ≥ 30 cm	3	815	1,993	1,896	3,725	2,200
Silver eel ≥ 30 cm	1	29	43	54	133	79
Silver eel ≥ 30 cm	2	134	200	266	654	392
Silver eel ≥ 30 cm	3	168	255	348	857	515

3.4.4 Discussion

There are some shortcomings and uncertainties in the methodology used for the nationally managed waters. Various regions are not sampled every year or regions are sampled in different months, which adds uncertainty to the estimates. However, because each region is sampled within each three-year reporting period, uncertainty within each three-year is diminished. However, despite the good comparability of these periods, abiotic factors and sampling deviations could still have large effects on the catch efficiency and the biomass estimates.

Another factor of uncertainty that can influence the biomass estimates is the lack of detailed information on the number and distribution of groins in the rivers. Here, we used a very coarse method to estimate that number per region. But probably the highest level of uncertainty due to insufficient knowledge of two crucial factors: the catch efficiency of the survey gear and habitat preference of eel. These factors cause a large variation in the biomass estimate.

3.5 Discussion regionally and nationally managed waters

Concerning both the nationally and regionally managed waters, there are some uncertainties. As mentioned before, the most important uncertainty is because the selectivity of the electric dipping net and the habitat preference are highly uncertain, the biomass estimates of the WFD and non-WFD waters are also highly uncertain.

A central assumption underlying the stock estimation is that the eels caught in a certain area represent the inhabitants of that area and that the eels do not move away from this habitat until they are silver eel. For the main passageway of silver eel to the sea (i.e., mostly through nationally managed rivers and lakes), this assumption entails much uncertainty. On the one hand, eels surveyed during the migration season in autumn (i.e., in many of the rivers; see *Table 3-3*) may partly consist of migrating silver eels. These eels were perhaps already surveyed in the habitats where they grew up (since areas are surveyed in different time periods), or these eels may have migrated from other countries after maturation to silver eel. This would lead to an overestimation of the silver eel stock in the Netherlands. Possibly, this could explain the high density estimates of the Benedenrivieren, which is the area closest to the coast where silver eel might concentrate just before and during the migration season.

In contrast, monitoring during or directly after the migration period may lead to an underestimation of the silver eel stock, because some of the silver eel might have migrated away or might be in the parts of the water body not surveyed with the dipping net (e.g. the open water). Thus, because the main surveys in the nationally managed waters take place during the migration period, there is additional uncertainty. The same reasoning goes for the regionally managed waters surveyed during or following the migration period. However, with consistent survey periods, this is not expected to affect the trends in biomass estimates.

4 A demographic model for yellow eel

4.1 Introduction to the demographic model

A different method as used in *Chapter 3* is used for the nationally managed larger lakes (lakes IJsselmeer, Markermeer, Grevelingen and Randmeren). The sampling with the electric dipping net occurs along the shore, and the assumed inshore:offshore density ratio used for the smaller water bodies is not suitable for the lakes because the lakes have a disproportionate amount of surface area compared to the shores. Moreover, the catchability of the FYMA survey, used to sample the offshore waters of the lakes, is unknown. Instead of an extrapolation of the survey data to the surface density, another method was therefore applied to estimate the eel biomass in the larger lakes IJsselmeer, Markermeer, Randmeren and Grevelingen. First, a demographic model was developed to estimate fishing mortalities in the lakes IJsselmeer and Markermeer, by fitting the model to relative changes in abundances observed in the survey. Subsequently, the estimated fishing mortalities were used to calculate the biomasses, based on the eel landings in the lakes (see *Paragraph 4.10*).

In addition, the demographic model was also used in the calculation of one of the stock indicators, the % Spawner-Per-Recruit as a percentage of the best possible spawner-to-recruit ratio (%SPR, *Chapter 7*). The *%SPR,* is needed to calculate the total anthropogenic mortality rate (*ΣA),* which can be compared with the 40% escapement target of the Eel regulation. In that case, the demographic model was not parameterized for lakes IJsselmeer and Markermeer (*Chapter 7*).

For lakes IJsselmeer and Markermeer, a demographic population model was used to estimate fishing mortality rates (F) in four periods since 1989. The model estimates F values and a glass eel to recruits conversion factor (K) based on a fit of the abundance per age class between the model output and FYMA survey data (see overview *Paragraph 2.1*). The model was fitted to the relative abundances of the FYMA survey. Also, recruitment in the model is based on a relative measure, the glass eel abundance index. Subsequently, the estimated F values and the registered landings were used to estimate the standing stock biomass in the lakes. The results were used as input to estimate the total Dutch eel stock biomass (*Chapter 5*).

The demographic model tracks annual eel cohorts through time, for eel from 1989 until 2020. The demographic model has been improved compared to the model that was used in previous assessments (Bierman et al. 2012, van de Wolfshaar et al., 2015 &, 2018). The changes that were made compared to the last assessment (van de Wolfshaar et al., 2018) are described below. In the demographic model, each year individual eels grow, mature and die based on length and sex specific biological keys (*Chapter 2*). Eels that reach the silver stage migrate away from the lakes and are excluded from the model. The cohorts are followed through time, resulting in an annual age-frequency distribution. Annual recruitment is independent from the local freshwater population and is based on the glass eel index.

The estimates of F depend heavily on the field data and on the biological parameters used in the model. For example, maturation is considered a loss of eel in the system, because silver eels are assumed to migrate to sea directly. Early maturation leads to a decrease of the fishing mortality of the stock. Likewise changes in sex-ratio and in growth rate affect the migration of silver eel from the modelled population, and hence the fishing mortality estimate. Uncertainty in the biological parameters increases the uncertainty in the estimates of F (see also Bierman et al., 2012 and van de Wolfshaar et al., 2015 & 2018).

4.2 Model update

Several improvements were made to the demographic model. All changes in the model are described in *Appendix B1*. The main changes to the demographic model compared to the model used in the previous eel assessment (van de Wolfshaar et al., 2018) are:

- The model was fitted to survey data of lakes IJsselmeer and Markermeer together, due to scarcity of data in lake Markermeer in some years.
- Different periods for which a single F estimate was calculated were changed such that the periods better represented the changes in eel fisheries management (*Paragraph 4.7*)
- The length-class based fit between model and data that was previously used, has been changed to an age-class based fit to allow for a better comparison between model and data. Since male and female eel display different growth patterns, it is not straightforward to choose sizes of length classes that result in an even distribution of the age classes over the size classes.
- The moment of comparison of the model with the survey data was moved from April to October, to better fit the ages of the individuals in the survey data. Previously, the model output at the start of the year (April) was compared with survey data that was collected from September- November.
- The derivation of some of the biological keys changed (*Chapter 2*).
- Small changes were made in the selectivity of the commercial fisheries to better match the minimum landing size legislation for eel (*Paragraph 4.5*).

4.3 Demographic model

The demographic model assumes a closed system for the freshwater phase, similar to models previously described for eel (see Oeberst and Fladung, 2012; Ciccotti et al., 2012). The glass eels that enter the lakes are assumed to stay there until they mature to silver eel and begin their migration to the sea. For lakes IJsselmeer and Markermeer, this is a pragmatic simplification, because these are not entirely closed.

The eel population in lakes IJsselmeer and Markermeer was modeled using a discrete time, Leslie matrix population model (Caswell, 2001). The model tracks the eels from when they enter the lakes until they become mature and start their migration to the ocean. We use a "reproductive subsidy" model (Hughes & Tanner, 2000) for a population that depends on external recruitment. Population projections with annual, externally driven recruitment follow:

$$\mathbf{x}(t+1) = \mathbf{A} \cdot \mathbf{x}(t) + \mathbf{r}(t).$$

The vector with the number of individuals in each age class $\mathbf{x}$, changes through time t, depending on the annual projection matrix $\mathbf{A}$ and the time dependent recruitment vector $\mathbf{r}(t)$.
The model distinguishes between males and females, as eels display sexual disparity in growth and maturation. The different cohorts, or age classes in the model, are represented by i and the sex classes by g. The transition probabilities between age classes are defined as P_{gi}. We use two separate matrix models for the two sexes, for females $\mathbf{A}_f$:

$$\mathbf{A}_f = \begin{pmatrix} 0 & 0 & 0 & \dots \\ P_{fi} & 0 & 0 & \dots \\ 0 & P_{fi} & 0 & \dots \\ \vdots & \vdots & \vdots & \ddots \end{pmatrix}$$

And for males $\mathbf{A}_m$:

$$\mathbf{A}_m = \begin{pmatrix} 0 & 0 & 0 & \dots \\ P_{mi} & 0 & 0 & \dots \\ 0 & P_{mi} & 0 & \dots \\ \vdots & \vdots & \vdots & \ddots \end{pmatrix}$$

The transition probability P_{gi} depends on the survival probability $e^{-F(t)\,z_{gi}-\mu}$ and the probability of maturing M_{gi}:

$$P_{gi} = e^{-F(t)\,z_{gi}-\mu}\left(1 - M_{gi}\right).$$

The survival probability depends on the natural mortality μ, fisheries mortality $F(t)$ and fisheries selectivity z_{gi}.

The annual recruitment $\mathbf{r}_g(t)$ is independent from the local yellow eel abundance. The recruitment per sex class depends on the sex ratio of the recruits $\rho(t)$ (female ratio in recruits). For female recruitment $\mathbf{r}_f(t)$:

$$\mathbf{r}_f(t) = \begin{pmatrix} K\rho(t)\, I(t) \\ 0 \\ 0 \\ \vdots \end{pmatrix},$$

Recruitment further depends on the annual glass eel index $I(t)$ and the glass eel to recruits conversion factor K. Male recruitment $\mathbf{r}_m(t)$ follows:

$$\mathbf{r}_m(t) = \begin{pmatrix} K(1-\rho(t))\, I(t) \\ 0 \\ 0 \\ \vdots \end{pmatrix}.$$

Numbers through time thus follow $\mathbf{x}_f(t+1) = \mathbf{A}_f \cdot \mathbf{x}_f(t) + \mathbf{r}_f(t)$ for females and $\mathbf{x}_m(t+1) = \mathbf{A}_m \cdot \mathbf{x}_m(t) + \mathbf{r}_m(t)$ for males.

The model follows eel in the lakes from 0.5 to 21.5 years after arrival in the lakes (*Table B1, Appendix B1*). The reason for starting the model 0.5 years after arrival in the lakes is that the FYMA survey takes place in September-November while the glass eel arrive at the Dutch coast in spring and the glass eel survey takes place from March-May. The census moment of the model, or the time at which model and data are compared with each other, is therefore set to October to match the FYMA survey, half a year after the glass eels enter the Lakes. The age classes of the model thus run from 0.5-1.5 years in age.

4.4 Annual recruitment parameters

Recruitment in the model is based on the glass eel inflow, which is monitored at Den Oever (*Figure 4-1*). The glass eel index $I(t)$ is based on numbers per haul and needs to be converted to numbers of yellow eel/trawled surface as used by the FYMA, which is done by multiplying with the glass eel to yellow eel conversion factor (K). In 2014, 2017 and 2019, stocking with glass eels (all three years) and elvers (2014) have taken place in lake Markermeer. Here, it is assumed that the amount of stocking is neglectable compared to the number of glass eels entering the lakes.

The female ratio of the recruits ($\rho(t)$) is based on market sampling data between 1978 and 2019. Most eel gender is determined after being in fresh water for two years (Beullens et al., 1997) and sex differentiation has been related to factors such as eel density at the time of forming the sexual organs (e.g., Roncarati et al., 1997; Davey & Jellyman, 2005; Bark et al., 2007). The female ratio varies annually, based on the sex of individuals at 2 years after arrival (*Appendix B2*). For the missing years, the average value over the years with data was used.

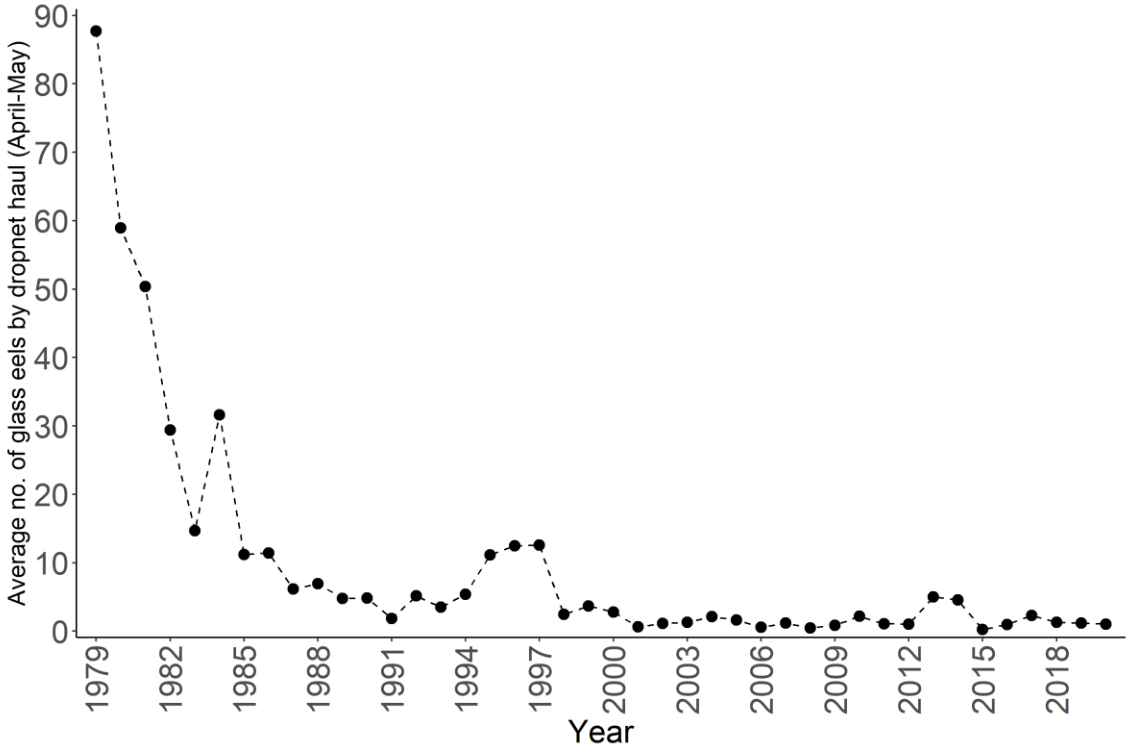

Figure 4-1 Glass eel average numbers per haul presented as an annual index (black circles) from 1979-2020, monitored at Den Oever, the Netherlands.

4.5 Age specific model parameters

All parameter values can be found in *Table B1* in *Appendix B1*. Parameterization is based on the biological keys (*Chapter 2*). Per age and sex class, the length at the mid-age of the age class is used to derive all the length-dependent and age-specific parameters. This means, for example, that for the first age class in the model, that runs from 0.5 – 1.5 years after arrival, the mid-age used for parameterization was 1 year after arrival. The length at the mid-age of the age class was determined from the von Bertalanffy growth curves (*Figure 2-4*). Age- and sex-specific probabilities of maturing (M_{gi}) were derived from length-based estimates of the proportion of mature eel in the market samples (*Figure 2-2*). Fishing selectivity (z_{gi}) is assumed zero for age classes with a length smaller than the minimum landing size, which is 28 cm. Rings in fyke-nets to allow escapement of undersized eel have been mandatory since the 1980s and few undersized eel are caught with this commercial gear. Moreover, eel is a robust species that can easily survive for some time out of the water or while captured in a net. Catch and release mortality of eels below the minimum landing size caught in fykes is assumed to be negligible. About one fifth of the eel catches in the lakes are caught by longlines (Dutch: 'hoekwant'). Catch and survival of undersized eel from the longlines is unknown and are therefore not taken into account in this study. In the model, eels in age classes (see below) that include individuals of 28 cm or more, therefore suffer from fishing mortality (*Appendix B, Table B1*). Fishing selectivity for age classes that are partly fished is equal to the proportion of the time an individual is 28 cm or larger in that age class, according to the standard growth curve *(Chapter 2, Figure 2-4*). Natural mortality is assumed to be independent from age or length and constant through time, $\mu = 0.138$ (Dekker, 2000: see *Paragraph 2.7*). Parameter values for $F(t)$ and K were estimated based on a log-likelihood estimation procedure by comparison between sampling data and model output.

4.6 Model fitting

To allow for a comparison of the (age-structured) model with the (length-based) FYMA survey data, the FYMA data were converted from length to age (*Figure 4-2*). The FYMA survey data were converted from

CPUE per length class (1-cm increments) to CPUE per age class based on the von Bertalanffy growth curves (*Chapter 2*). The von Bertalanffy growth curves were corrected for the time deviation between the glass eel survey (March - May) and the market sampling (May - September). The CPUE per age class was calculated based on the age classes defined for the demographic model (*Table B1, Appendix B*). Since the growth curves are sex-specific, the proportion of males to females per length class was set first (*Chapter 2*). For the length classes below 28 cm, the model assumes a sex-ratio that is equal to the sex ratio of the 28 cm length class because there was not a sufficient number of sexed individuals smaller than 28 cm to determine a length-dependent sex ratio. Parameter values were estimated for a model fit on a combination of lakes IJsselmeer and Markermeer data. The weighted mean between lake IJsselmeer and lake Markermeer was calculated based on the surface area between the lakes, which is 62:38.

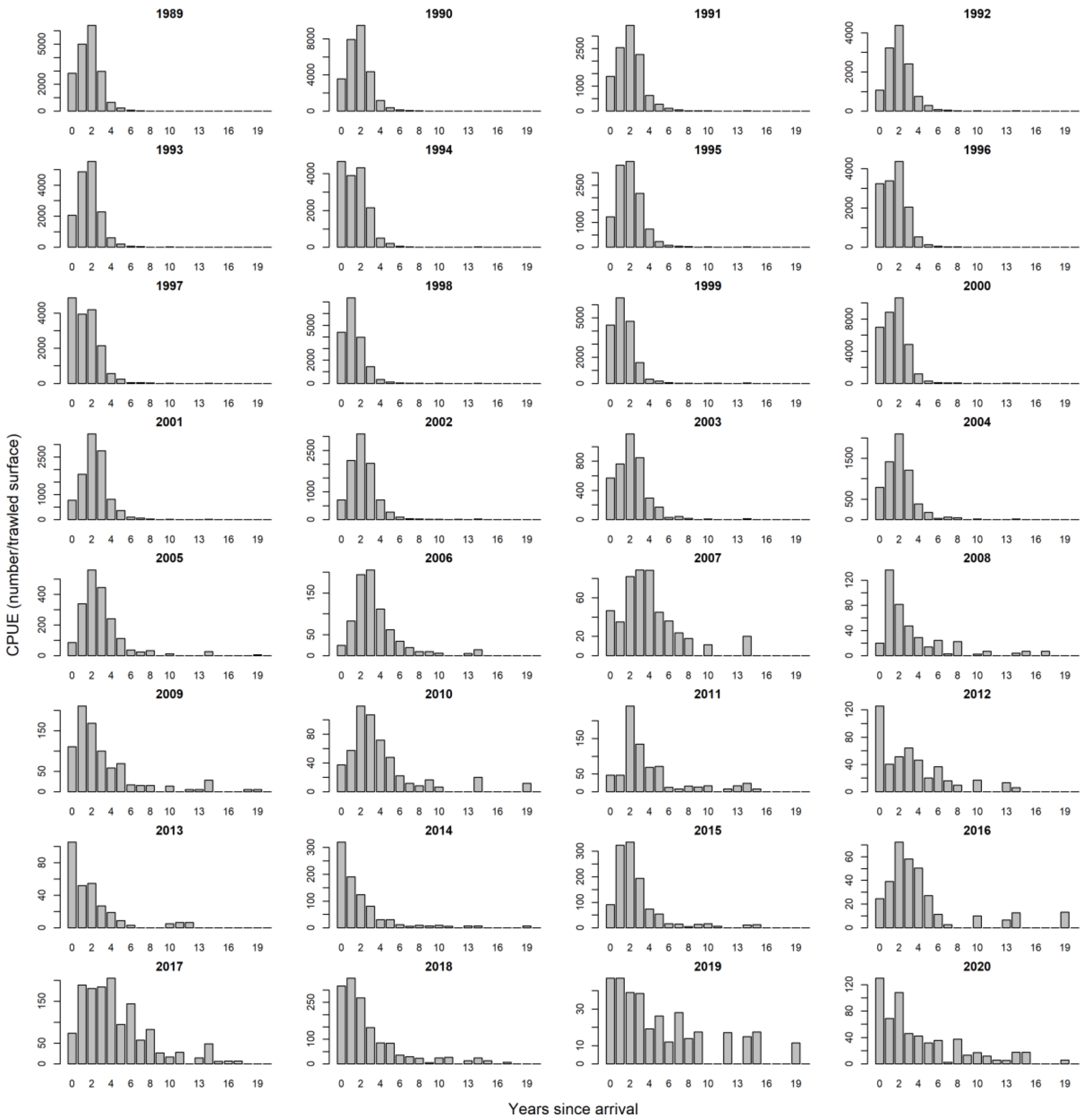

Figure 4-2 Mean CPUE per year and per age (years since arrival) in the FYMA electric beam trawl survey for lakes IJsselmeer and Markermeer, together, between 1989-2020, after application of the length-age key. Note that the scale of the y-axis differs per graph.

There is no information available on the selectivity of the survey gear and therefore the assumption was made that this selectivity is equal for all lengths. However, the model was fit to data starting from eel at the age of two years since arrival at the coast, because it seems as if the number of individuals per age class increases from year 0 to year 1 and in some years from year 1 to 2 after arrival (*Figure 4-2*). This is indicative of a lower catchability of the smallest individuals in the FYMA survey data. As long as an equal selectivity is assumed, the absolute selectivity of the FYMA survey is not expected to affect the F estimates since all estimates are based on relative changes in abundances.

From year 7 after arrival, only a few individuals per age class are observed in the FYMA survey. This introduces large uncertainties in the estimated CPUE. Therefore, the age class 7+ was used as the last age class in the model.
Parameter values for $F(t)$ and K were estimated with a minimum log-likelihood Metropolis-Hastings algorithm following van de Wolfshaar et al. (2015 & 2018) and Bierman et al. (2012). The algorithm computes population projections for different values of $F(t)$ and K and estimates which population age distribution to be the best match for the data. Through stochastic iterations (n = 50,000), the algorithm finds the combination of parameters for which the fit is best. The likelihood was calculated based on a Poisson distribution and the prior likelihood is based on an even distribution. Jump sizes of 1% of the first values were used for the estimated parameters K and $F(t)$. The results below are based on initial values for K of 0.1 and for F of 1.0, but robustness of the results was tested through the use of different initial values. For every outcome, the acceptance rate of the stochastic iterations was checked and a visual check of the convergence and the correlation between the estimated parameter values was performed. An acceptance rate of maximally 30% was maintained. In case the acceptance rate, the convergence or correlations were not satisfactory, the number of iterations was increased. In addition, visual checks were performed on the residual plots of model and data.

4.7 Periods in fishing effort

F is estimated for five different periods (1968-1988, 1989-1999, 2000-2008, 2009-2014 and 2015-2020). The selection of the time periods was based on various motivations. The value of F may change for the consecutive periods because of possible changes in fishing effort through time. These are referred to as *possible* changes in fishing effort because the number of permits is known, but the realized effort is not known before 2010, when the registration of used effort became obligatory. Consequently, the ‘potential’ fishing effort is known, but the used effort is unknown. Moreover, it is also unknown if a reduction in potential effort (number of permits) has led to a reduction in realized effort. The decision on the breaks in the periods was therefore somewhat arbitrarily determined, based on what we consider major changes in potential fishing effort due to changes in management. The first period, starting in 2000, is based on a reduction in permits for eel boxes (Bierman et al., 2012). The second break, between 2008 and 2009, is because in 2009 the EMP came into place, causing fishing to be prohibited during the main silver eel migration period (September-November). The break between 2014 and 2015 was chosen because, since 2015, the fishing (effort) that was allowed on some types of commercial fish was limited (Tien et al., 2015), which may have led to an increase in effort for eel fishing. Not all of the changes in potential fishing effort were included, however. For example, the break due to the 2006 buy-out was excluded, because it would create a very short period (2006-2008) for the fishing effort. In addition, the buy-out did not seem to have a large impact on the trend in landings, which was steadily decreasing between 2000 and 2009. In the report the F values from the period 2000-2008 onwards are reported, as the first period for the overall eel assessment (2006-2008) falls within this period.

4.8 Model fit and estimated fishing mortality

The model predictions and the actual data on the FYMA catches (number per trawled km^2 per age class) are presented in *Figure 4-3*. As expected, the eel abundance decreases with age in the data as well as the model (in the model this is predefined). Quite an obvious decrease in abundance is visible through the years in the age classes of eel from 2-5 years after arrival. In the age class of year 7+ after arrival, no such decline is observed. While the model predictions follow the general downward trend in the true data for the eel age classes between 2-6 years after arrival, there is a large deviation between the model and the data for the 7+ years age class. The residual plot (*Figure 4-4*) shows again the strong underestimation of the model on the abundance in the age classes 7+ years after arrival. In addition, there seems to be a small overestimation on the abundance in the age classes of 2 and 3 years after arrival in the most recent years. In the earlier years of the time series, this was an underestimation.

The estimated fishing mortalities for lakes IJsselmeer and Markermeer are given in *Table 4-1*, for the four periods (see previous paragraph). The fishing mortality (F) for lakes IJsselmeer and Markermeer decreases over the first two periods but increases again in the last period. The estimated K value is 0.06, which is the conversion factor between the relative glass eel index at the Afsluitdijk in Den Oever to the CPUE for yellow eel 0.5 years after arrival in the lakes in the FYMA. It includes the success rate of entering the lakes and survival probability from 0.0 to 0.5 years after arrival in the lakes.

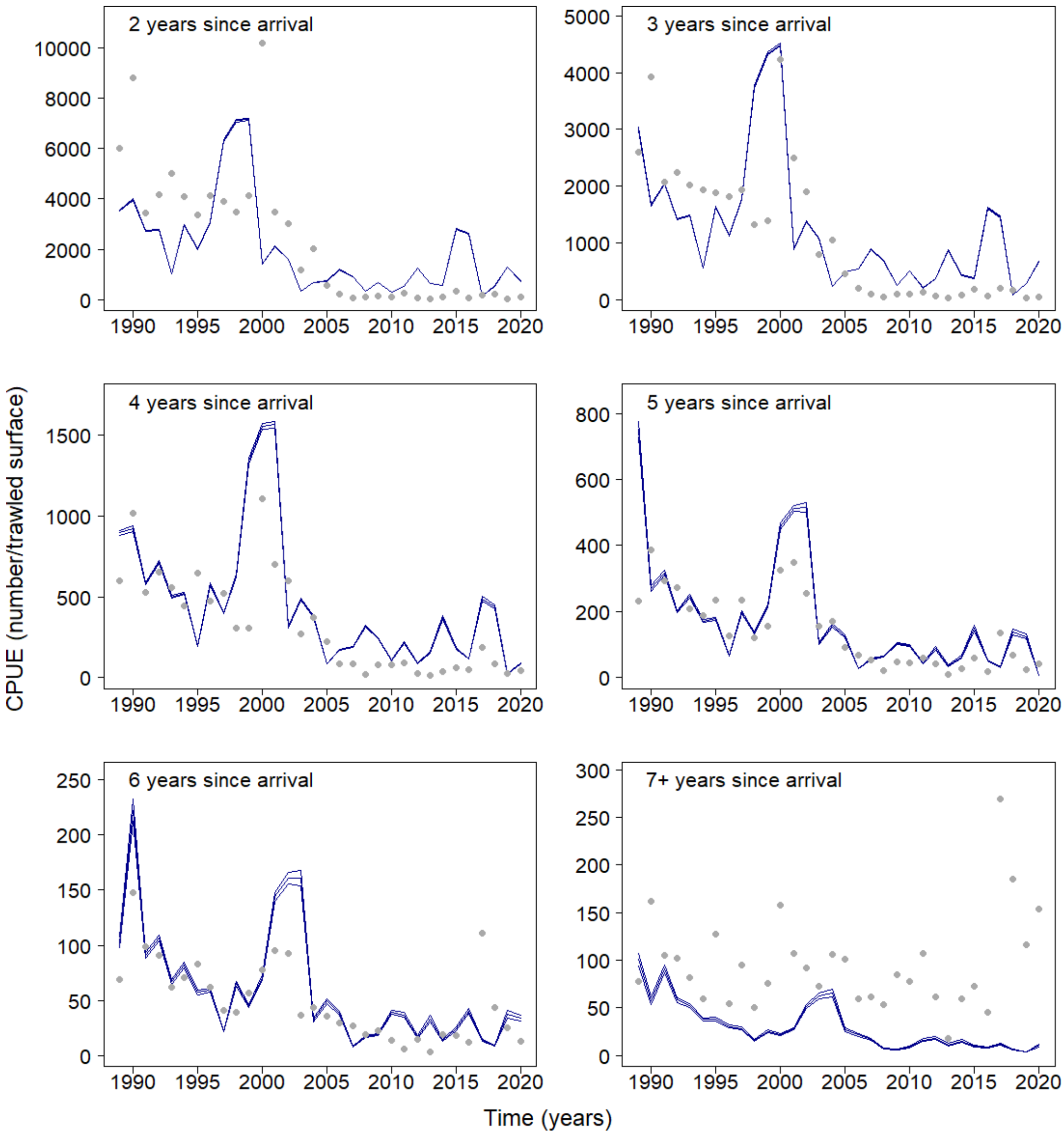

Figure 4-3 CPUE per age class (grey dots) and model predictions (minimum, maximum and mean outcomes of the last 20% of the iterations of the parameters estimated by the model – blue solid lines), both in number per trawled surface, area (km^2), for the model fit of the data, together for lakes IJsselmeer and Markermeer.

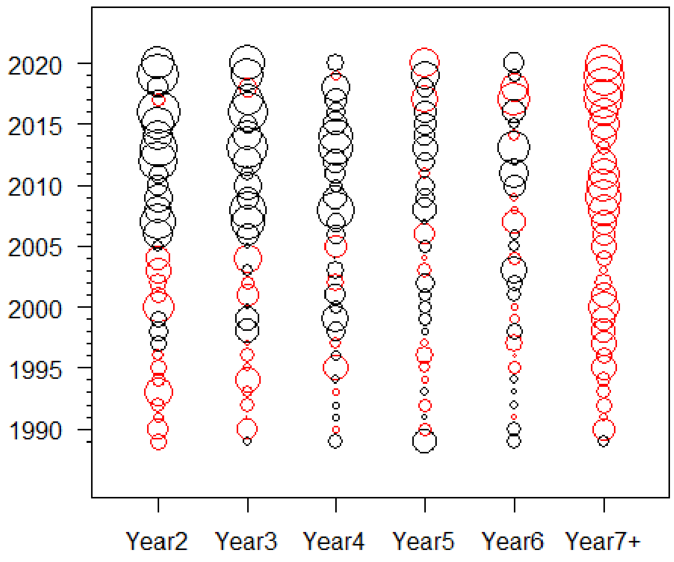

Figure 4-4 Residuals plot of the difference between the observed and predicted eel abundance per age class for the model fit of the data, together for lakes IJsselmeer and Markermeer, given the mean F estimates presented in Table 3.1. Both negative (red) and positive (black) deviations are plotted. The size of the circles indicates the value of the residual (with larger being a higher value).

Table 4-1 Model-estimated mean fishing mortality values (F) for model fits on lakes IJsselmeer and Markermeer data together. F is estimated for different periods, and the 90% confidence interval of the estimated parameter values in brackets indicates the variance in the values estimated by the model (after 50,000 iterations).

	Fishing mortality lakes IJsselmeer and Markermeer
F (2000 - 2008)	1.10 (1.08 – 1.12)
F (2009 - 2014)	0.76 (0.73 - 0.79)
F (2015 - 2020)	1.04 (1.01 – 1.07)

4.9 Discussion of the demographic model

The increase in eel numbers 7+ years after arrival in lake IJsselmeer in recent years is not captured by the model. Generally, the model underestimates the numbers for this age class. Potentially, the underestimation for the older ages stems from the large individual-level variability in eel growth (Panfili et al., 1994). The assumed growth curves do not allow for variability in age with size or for changes in growth over time. Part of the individuals that are estimated at 7+ years after arrival may thus actually be relatively fast growing individuals and be younger than that. In addition, growth patterns of eel could have changed over the years due to the large decrease in the density of eels (*Figure 4-3*). Also the environmental conditions in lakes IJsselmeer and Markermeer have changed substantially over the years (Soudijn & van de Wolfshaar, 2021), which may cause variation in growth.
The growth curves that we currently use in the model are constant through time. Perhaps the best solution would be to use an annual age-length key, but the numbers of eel that are aged each year are not sufficient to support such data analysis. It is unlikely that the increasing eel numbers 7+ years after arrival in lakes IJsselmeer and Markermeer are migrating silver eel, as silver eel are hardly ever caught in the FYMA survey.

In the previous assessment, the demographic model was fitted to the data of lake IJsselmeer and Markermeer separately. However, in recent years the numbers of eel in the lake Markermeer survey have decreased to such low numbers that the length frequency distribution was not good enough to fit the model. As a result, the fit of the demographic model to data of Lake Markermeer alone is very poor and the estimates cannot be used. Therefore, the choice was made to fit on the data of Lakes IJsselmeer and Markermeer combined.

Compared to the previous stock assessment (van de Wolfshaar et al., 2018), the estimate of *F* has changed substantially from 2000 onwards. The main causes are: 1) a longer times series is used (until 2020 instead of 2016), which also affects the fit in previous years of the time series, 2) different periods were chosen for the *F* values, 3) a different growth curve was used, and, 4) the model was improved from a length-based to an age-based fit. In addition to the changes mentioned above, the model was now fitted on the data of lakes IJsselmeer and Markermeer together.

The demographic model has been strongly improved compared to the previous assessment. There are still several possible improvements possible for the model:

- Add the retained catches data to the model fit to allow an estimate of absolute biomass in the lakes by the model.
- Consider the possibility of the use of different length-sex ratio relationships for different time periods in the model. Currently the model is based on a constant length-sex ratio through time while it is likely to vary with changes of *F* and the sex ratio of the recruits.
- Consider the possibility of the implementation of a varying selectivity of the FYMA survey with length.
- Consider the possibility of using different age-length curves for different time periods in the model, or a variable age-length key. It is not totally clear so far whether there are sufficient otolith readings available for such an exercise and how much variability in growth occurs through time.
- Consider the possibility of using different maturity-length curves for different time periods in the model. It is not totally clear how much variability in maturity occurs through time or to what extend this process is affected by environmental variables.
- Consider the possibility of using 0- or 1-year old individuals in the FYMA as a measure for the number of 'recruits' in the model instead of the glass eel index. There have been considerable changes in water management regimes over the years that have likely affected the flow of glass eel from the coast to the lakes between years. It is however impossible to determine how these changes in water management may have affected the ability of glass eel to reach the lakes.

4.10 Eel biomass estimation in large lakes

In the major large lakes (IJsselmeer, Markermeer, Randmeren and Grevelingen, *Figure 4-5*) eel biomass was estimated in a different way compared to other (smaller) water bodies (see *Chapter 3* and Bierman et al., 2012). The major reason for choosing a different method, is that the relationship between the density inshore compared to offshore is highly uncertain (*Chapter 3*) and results in large overestimations of the standing stock in water bodies with a large proportion of offshore water.
The standing stock for the lakes IJsselmeer and Markermeer was estimated using fishing mortality in these lakes as estimated by the demographic model (*Table 4-1*) and the commercial landings. For the biomass in the Randmeren and Grevelingen, no parameterized demographic model is available and the estimated density in the lakes IJsselmeer and Markermeer (standing stock/ha) was used as basis and – where available – corrected by the difference in *CPUE* in the shore as estimated with a dipping net. In the Randmeren, there is electric dipping net survey data available from 2012 onwards. In the saline lake Grevelingen, survey data with an electric dipping net is lacking completely. For the Randmeren, we therefore assumed that the eel density is the same as in the lakes IJsselmeer and Markermeer, corrected for the difference in *CPUE* from the electric dipping net surveys in the lakes. For Grevelingen, we do not have any reliable *CPUE* estimate available for eel. Therefore we use the uncorrected density from the lakes IJsselmeer and Markermeer. This is a strong assumption, and unlikely to be entirely true, but because good quality data is lacking a strong assumption needed to be made. Note that the method used here is different from previous reports. In van de Wolfshaar et al. (2018), it was assumed that the

fishing mortality in lakes IJsselmeer and Markermeer also applied to the other lakes. The assumption of equal fishing mortality, is considered to be an even stronger assumption, because the fishing mortality is not just depending on the density in the lakes, but also largely depending on management rules such as the relative amount of fishery, which differs a lot between the Randmeren, Grevelingen and lakes IJsselmeer and Markermeer.

The estimated biomasses from the lakes IJsselmeer and Markermeer, Randmeren and Grevelingen are integrated into an estimate of the total Dutch standing stock (*Chapter 5)*.

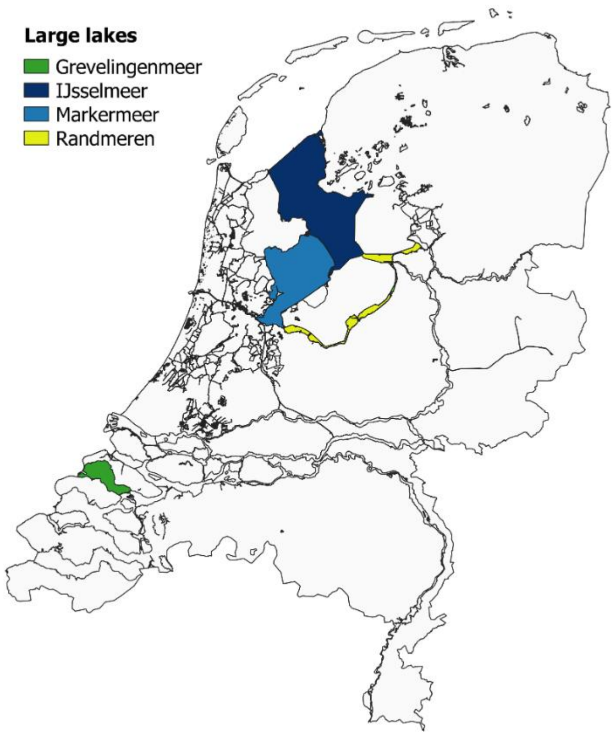

Figure 4-5 The four large Dutch lakes

4.10.1 Standing stock lakes IJsselmeer and Markermeer

Estimates of the standing stock were calculated by combining the landings in lakes IJsselmeer and Markermeer (*Appendix A0* and *Paragraph 2.8*). The percentage yellow eel in the total landings was estimated using the (representative) length data in the market sampling (*Paragraph 2.1*), sampled in lakes IJsselmeer, Markermeer, Randmeren and Grevelingen. In total, 74% of the total retained catches in biomass was estimated to be yellow eel. This percentage was used to convert the reported total retained catches into yellow eel and silver eel retained catches. Estimates of the standing stock of yellow eel and silver eel were subsequently calculated by combining the landings and the estimated fishing mortality as following: *biomass = landings/(1-exp(-F)),* (*Table 4-2*). This resulted in an estimated standing stock of 477 tonnes (355 tonnes yellow eel and 122 tonnes silver eel, *Table 4-2*) in 2018-2020, an increase of 179 tonnes since the previous period (2015-2017). This method assumes that the fishing mortality of silver eel is the same as the mortality of yellow eel. As there are many differences in behavior when silver eel starts to migrate, this assumption is uncertain, because the increased mobility of silver eel might also increase the catchability, which would lead to a higher fishing mortality. However, the absence of fishing in the main migration period of silver eel (closed period in September-November) since 2009, may have caused a lower fishing mortality for silver eel compared to yellow eel. Therefore, it is uncertain whether fishing mortality on silver eel is similar to yellow eel, or if it is an over- or underestimation.

Table 4-2 Estimated mean landings (tonnes), fishing mortality, and yearly standing stock (tonnes) in the lakes IJsselmeer and Markermeer per 3-year period.

		IJsselmeer and Markermeer		
	Period	**Landings**	**Fishing mortality**	**Standing stock (tonnes)**
Yellow eel (≥30cm)	2006-2008	202	1.10	305
	2009-2011	113	0.76	215
	2012-2014	117	0.76	222
	2015-2017	143	1.04	222
	2018-2020	228	1.04	355
Silver eel	2006-2008	71	1.10	104
	2009-2011	40	0.76	74
	2012-2014	41	0.76	76
	2015-2017	50	1.04	76
	2018-2020	80	1.04	122

4.10.2 Standing stock Lakes Randmeren and Grevelingen

For the Randmeren, it is assumed that the eel density as estimated in the lakes IJsselmeer and Markermeer can be used as basis for the density in the Randmeren, corrected for the difference in *CPUE* in the electric dipping nets surveys at the different locations (*Table 4-3*). Because the electric dipping net surveys in the Randmeren took place from 2012 onwards, for the first two periods (2006-2008 and 2009-2011) the *CPUE* from the period 2012-2014 is used to estimate the standing stock. This methodology results in an estimate of 6.3 tonnes of silver eel in the Randmeren for the latest period (*Table 4-4*).

For Grevelingen, the assumption is made that the eel density is the same as in the lakes IJsselmeer and Markermeer. This results in a value of 9.2 tonnes of silver eel in the latest period (*Table 4-4*).

Table 4-3 Mean CPUE per reporting period (tonnes/sampled ha) of the electric dipping net in the inshore surveys of the lakes IJsselmeer and Markermeer and the Randmeren per period.

	CPUE				
Lake	**2006-2008**	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
IJsselmeer/ Markermeer	5.17	4.26	4.98	6.21	6.55
Ketel & Vossemeer	8.85*	8.85*	8.85	6.88	7.47
Randmeren-Oost	0.59*	0.59*	0.59	2.54	3.12
Randmeren-Zuid	0.47*	0.47*	0.47	3.11	2.19
Zwarte Meer	0.27*	0.27*	0.27	0.11	1.52

* as no data was available for the Randmeren from before 2012-2014, for the first two periods (2006-2008 and 2009-2011, the *CPUE* from 2012-2014 was assumed)

Table 4-4 Density (tonnes/ha), surface area (ha), CPUE correction and yellow eel and silver eel standing stock (tonnes) per lake.

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020
IJsselmeer/Markermeer	Density yellow eel	0.0017	0.0012	0.0012	0.0012	0.0019
IJsselmeer/Markermeer	Density silver eel	0.00057	0.00040	0.00041	0.00041	0.00066
Ketel & Vossemeer	Surface area	4,067	4,067	4,067	4,067	4,067
	CPUE correction	1.71	2.08	1.78	1.11	1.14
	Yellow eel biomass	11.5	9.9	8.7	5.4	8.9
	Silver eel biomass	4.0	3.4	3.0	1.9	3.1
Randmeren-Oost	Surface area	6,318	6,318	6,318	6,318	6,318
	CPUE correction	0.11	0.14	0.12	0.41	0.48
	Yellow eel biomass	1.2	1.0	0.9	3.1	5.8
	Silver eel biomass	0.4	0.4	0.3	1.1	2.0
Randmeren-Zuid	Surface area	4142	4142	4142	4142	4142
	CPUE correction	0.09	0.11	0.09	0.50	0.33
	Yellow eel biomass	0.6	0.5	0.5	2.5	2.7
	Silver eel biomass	0.2	0.2	0.2	0.9	0.9
Zwarte Meer	Surface area	1,811	1,811	1,811	1,811	1,811
	CPUE correction	0.05	0.06	0.05	0.02	0.23
	Yellow eel biomass	0.2	0.1	0.1	0.0	0.8
	Silver eel biomass	0.1	0.0	0.0	0.0	0.3
Total Randmeren	**Yellow eel biomass**	**27.1**	**19.1**	**10.2**	**11.1**	**18.2**
	Silver eel biomass	**9.3**	**6.5**	**3.5**	**3.8**	**6.3**
Grevelingen	Surface area	13,902	13,902	13,902	13,902	13,902
	Yellow eel biomass	23	16.3	16.8	16.8	26.8
	Silver eel biomass	7.9	5.6	5.8	5.8	9.2

5 Overview national stock biomass

5.1 Overview

In this chapter the total eel stock biomass in the Netherlands is estimated for each period, based upon the biomass estimates of the different water bodies as described in *Chapter 3* and *Chapter 4*. In *Chapter 7*, these biomasses are used to calculate the key stock indicators as requested by the EC.

Stock estimates are provided for the periods 2006-2008, 2009-2011, 2012-2014, 2015-2017 and 2018-2020. In some cases, extrapolation between periods is necessary because insufficient data was available for every region in all periods (*Table 5-1*). Extrapolation is needed for the Volkerak-Zoommeer, Zandmaas, the Randmeren, ditches and the other regionally managed waters (*Table 5-2*). How extrapolation was done for the specific water body with missing data, is explained in *Chapter 3* and *Chapter 4*.

*Table 5-1 Eel biomass estimate availability ('data') per period and per water body.'-' indicates that there was not sufficient data available. * The non-WFD waters (ditches) data are not used per time interval but are grouped in the analysis.*

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020
Large lakes	IJsselmeer/ Markermeer	data	data	data	data	data
	Grevelingen	-	-	-	-	-
	Randmeren	-	-	data	data	data
Regionally managed waters	Ditches*	-	-	data	data	data
	WFD	data	data	data	data	data
Nationally managed waters	Volkerak-Zoommeer	-	data	-	data	data
	Zandmaas	-	data	data	data	data
	Others	data	data	data	data	data

5.2 National stock biomass

In *Chapter 3* and *Chapter 4*, the biomass estimates for all eel (≥30 cm), yellow eel and silver eel, are estimated for all water bodies in the Netherlands. An overview of these estimates per period is given in *Table 5-2*. Subsequently, the total biomass is given by summing up the estimates of all water bodies per period to get a total estimate of the eel standing stock in the Netherlands (*Table 5-2*). This total biomass estimate shows that from the first period (2006-2008) to the second period (2009-2011) there was an initial large increase in biomass with around 2,000 tonnes and from the second to the third period (2012-2014) there was a small increase (~ 500 tonnes). After the third period there was a very small decrease of ~ 100 tonnes to the fourth period (2015-2017) and again a decrease of ~450 tonnes to for the most recent period (2018-2020, *Table 5-2*). The current estimate is a standing stock of almost 5,000 (4,961) tonnes of eel (≥30 cm) in the Netherlands. The yellow and silver eel biomass estimates for each scenario (see *Paragraph 3.2*) and three-year period show the same trend as the total biomass, with initial increasing biomasses, but a decline in the latest two periods for yellow eel. Silver eel biomass increased until 2015-2017 but declined in the most recent period (*Table 5-3*).

Table 5-2 Biomass estimates (all eel ≥ 30 cm, in tonnes) in scenario 2 (see Paragraph 3.2) for each period.

		2006-2008	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
Regionally managed waters	Ditches	981	981	981	981	981
	WFD Waters	1,947	3,236	3,793	2,265	1,791
Nationally managed waters	IJsselmeer/Markermeer	409	289	298	299	476
	Grevelingen	31	22	23	23	36
	Randmeren	36	26	14	15	24
	Others (main rivers)	645	1,539	1,446	2,842	1,673
	Total	4,049	6,093	6,555	6,425	4,981

Table 5-3 Total standing stock biomass (tonnes) estimates for yellow eel and silver eel (≥ 30 cm) for each period and each scenario. Scenarios 1-3 differ in catch efficiency and habitat preference for all water bodies except the large lakes (Chapter 3). The silver eel biomass estimates are used in Chapter 7

	2006-2008	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
Yellow eel					
Scenario 1	1,032	1,312	1,293	1,240	1,174
Scenario 2	**3,280**	**5,093**	**5,034**	**4,773**	**3,869**
Scenario 3	3,926	6,350	6,292	5,953	4,728
Silver eel					
Scenario 1	274	291	394	421	352
Scenario 2	**769**	**999**	**1,521**	**1,652**	**1,113**
Scenario 3	907	1,222	1,915	2,088	1,365

5.3 Discussion

The values presented here show an initial increase in eel biomass, but a decrease in recent years (*Table 5-3*). This pattern is, however, not equal for the different water bodies. The biomass estimates in the WFD-waters had a large influence on the trend in the total biomass. The estimate of the WFD-waters shows very high biomass estimates in the second and third period, and lower values in the last two periods (*Table 5-2*). Other waters do not follow the same trend. For example, the large lakes follow a opposite trend compared to the WFD-waters, with high biomass in the first and last period. However, with a lower biomass estimate, the large lakes are less influential on the total biomass estimate of the stock. Also the other nationally managed waters (excluding the large lakes), follow a different trend. They have a low biomass estimate in the first period and much higher values in the later periods. Especially in the period 2015-2017 the value of the national waters is highly influential on the total biomass estimate, with a much higher value (mainly due to the 'Benedenrivieren', see *Table 3-9*) compared to all other periods (*Table 5-2*).

6 Mortality during silver eel migration due to barriers

6.1 Silver eel barrier mortality

Silver eel suffer barrier mortality during downstream migration when passing through barriers. This barrier mortality is one of the sources of mortalities which are used in the overall assessment as presented in *Chapter 7* (see the flow diagram of the stock assessment, *Figure 1-1*). This chapter describes the methodology and data on which estimates of barrier mortality during silver eel migration are based.

To update the data since the last assessment (Van de Wolfshaar et al., 2018) an inventory was held among water boards to renew the information on barrier specifics concerning migration. In addition, an update of the list of barriers that were replaced by a different type was made for barriers in the WFD waters.

This chapter also deals with assisted migration in which silver eel is caught upstream from a barrier and 'lifted' across it, so called 'trap and transfer' mitigation (*Paragraph 6.3*).

6.2 Barrier types

There are different types of barriers (*Figure 6-1*):

1. **Pumping stations**: pumping stations (Dutch: 'gemaal') are mainly used for the drainage of low-lying land and pump water from a polder into another water. Most pumping stations are situated in the areas in the Netherlands that lay below sea level and refrain the land from flooding. In the Netherlands there are thousands of pumping stations (*Figure 6-1b*).

2. **Ship locks**. Locks are built in places where the level of the water within a waterbody changes. A ship lock allows ships and vessels to travel up or down a water body to a higher or lower water level. The lock controls the depth in the lock, allowing for different levels at each side of the lock (*Figure 6-1*a).

3. **Discharge sluices**. Discharge sluices (Dutch: 'spuisluis') are built to control water levels, and discharge excess water by periodically opening them when the water levels of the receiving water body are lower than the 'upstream' water body (*Figure 6-1c*).

4. **Weirs**. Weirs are built to control water levels in both running waters, i.e. streams and rivers, and smaller polders. They can be lowered or lifted when the upstream water levels are too high (*Figure 6-1d*).

5. **Hydroelectric power station (HPS)**. A hydroelectric power station uses flowing water to set a turbine in motion. These stations are located on rivers and usually are large barriers. In the Netherlands there are three HPS's on two main national rivers. Two in the river Meuse and one in the river Rhine (*Figure 6-1d)*

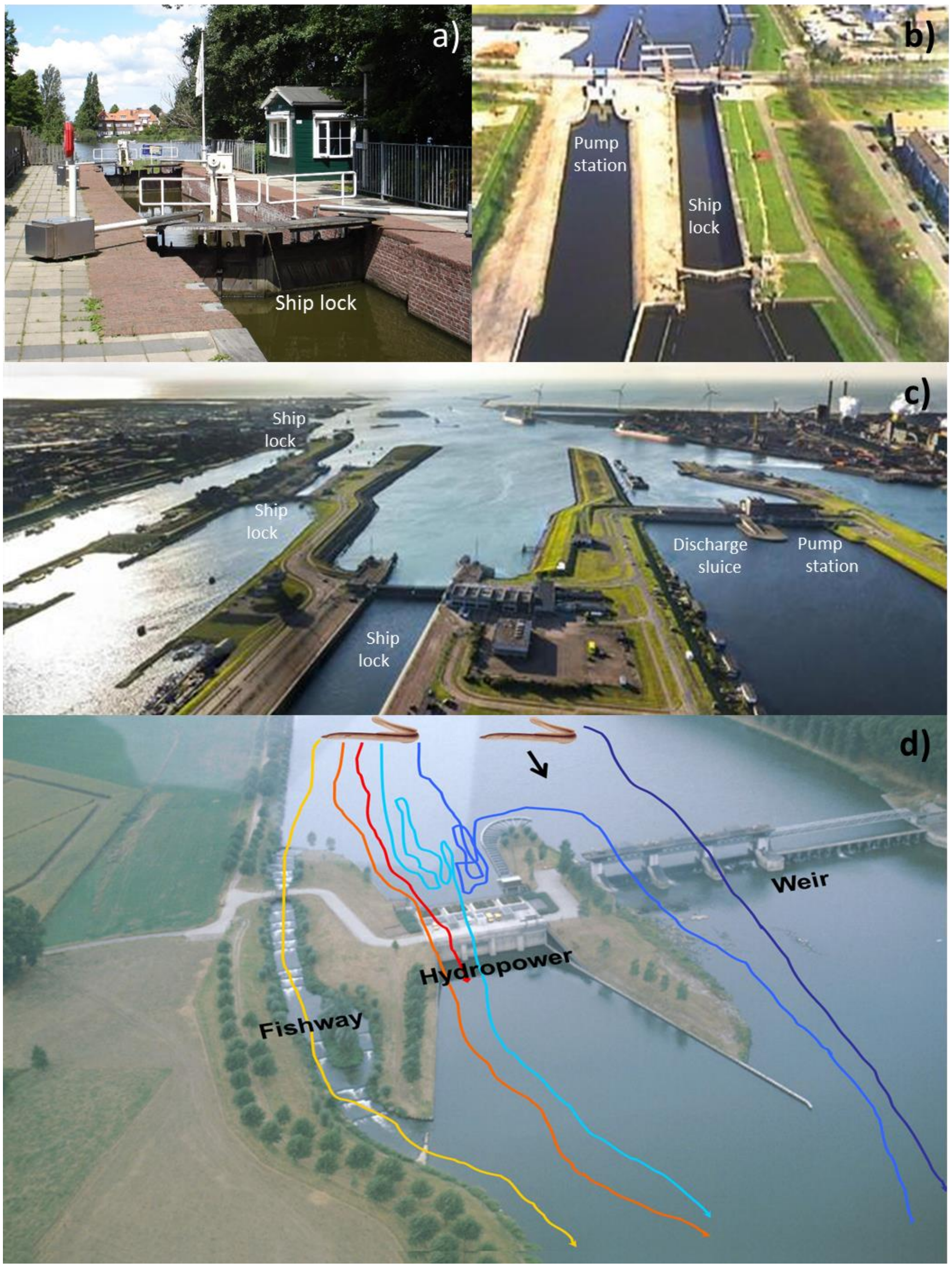

Figure 6-1 An illustration of different types of barriers. Barriers range from simple, e.g. single ship lock (a), to combinations, e.g. pump station and ship lock (b), to very complex sites consisting of a combination of pumping stations, ship locks, sluices or other alternative routes for migration, for example this site at IJmuiden in the North Sea Canal to sea (c). the more complex a site is, the more routes silver eel can follow to pass the obstruction. Mortality rates per route can be different, e.g. through a HPS, and therefore the distribution of eel passing via the different routes per site determines the overall mortality rate for the entire site, for example this site at Linne in the Meuse (d).

6.3 Assisted migration

Recent assisted migration (also called trap and transfer) initiatives, in which silver eel is caught above a barrier and 'lifted' across it, are taken into account when calculating the overall migration mortality for silver eel. Since 2011, several projects have started at migration barriers (mainly pumping stations) to assist the migration of silver eel. Because not all silver eel passing the selected barriers for assisted migration suffer from mortality or injuries, an assessment was done to estimate the absolute amount of saved eel. In 2013, a selection of the main barriers was made (Winter et al., 2013a) to improve the selection and efficiency of assisted migration initiatives. Applying location-specific mortality rates, the overall amount of 'saved' eels was based on the mortality rate of the given site. This value is subtracted from the migration mortality biomass estimate.

6.4 Model for estimating barrier mortality

Assessing the mortality of silver eels during their migration from inland water bodies to the sea is difficult due to the large numbers of barriers. There is a huge amount of pumping stations, many ship locks and three larger HPS's in the Netherlands (Kroes et al., 2018; Belletti et al., 2020). To construct a model on silver eel mortality caused by these barriers, knowledge on the following processes is necessary:

1) Silver eel migration routes, when migrating from inland water bodies to the sea
2) The barriers that the silver eels encounter along these routes
3) Mortality rates during passage of barriers

For this assessment, a silver eel migration model was built, based on a hierarchy of water bodies, providing a reasonable description of silver eel migration in the Netherlands (*Figure 6-2*). In this model, silver eels are split into three 'hierarchy levels'; each hierarchy level representing a water body type where they start the migration route to the sea. The three hierarchy levels are:

1) 1st hierarchy ('polder' water bodies): water bodies which are below sea level and serviced by a large number of small pumping stations. In the model, it is assumed that silver eel migrate through a single pumping station in order to leave a polder (i.e. no multiple pumping stations in sequence). For most polders, pumping stations discharge water into a 'boezem' water body (2nd hierarchy, see below), which will face additional barriers. Pumping stations of polders close to the coast can pump water directly into the sea, in which case the silver eels that survive the passage of these sites do not face another barrier and are able to escape to the sea directly. In the model, polder waters are represented by the wetted area of non-WFD waters (ditches, see *Paragraph 3.2)*;

2) 2nd hierarchy ('boezem' water bodies): water bodies such as canals, small inland lakes and smaller streams and rivers. In the model, boezem waters are represented by all regionally managed WFD water bodies (*Paragraph 3.2)*. Boezem waters are either connected directly to the sea or to large nationally managed water bodies (3rd hierarchy, see below) via larger pumping stations, ship locks, weirs and/or discharge sluices. In case they are connected to national water bodies, they face additional barriers.

3) 3rd hierarchy ('national' water bodies): large nationally managed water bodies such as sections of the main rivers Rhine and Meuse (including downstream parts, *Chapter 3*), the freshwater lakes IJsselmeer and Markermeer, Randmeren and Grevelingen (*Paragraph 4.9*). In the River Meuse and the Rhine river branch Nederrijn/Lek, there are large HPS's. National water bodies are connected to sea mainly by discharge sluices (e.g. IJsselmeer, Lauwersmeer, Haringvliet), and/or by large pumping stations (e.g. IJmuiden), always in combination with ship locks at each of these locations or have an open connection (e.g. Nieuwe Waterweg).

The framework of the model for migration routes is illustrated in *Figure 6-2*. The hierarchies as described above are connected with each other and with the sea as presented by the arrows. For each arrow the proportion of eel choosing that route and the proportion that will not survive a passage is estimated. Barrier mortality occurs when silver eels pass from one hierarchy to another, or from one hierarchy to the sea. The model thus assumes that barriers within the 1^{rst} and 2nd hierarchy are never in sequence: eel cannot experience the barriers that belong to the same hierarchy more than once. There are only a few polder waters with two or even more boezem layers, in which polder waters are pumped into an 'inner boezem' and subsequently pumped into an 'outer boezem'. Because the area of polder water that has multiple pumping stations before reaching a boezem water is small, it is therefore of little influence on the outcome of the model. For the 3rd hierarchy, on the different routes from the 2nd hierarchy to sea via the national water bodies, different subsequent barriers can be passed along each of the potential routes.

For the parameterization of the model, the migration routes as described in Winter et al. (2013a & 2013b) are used. For each possible route and three-year period, an average proportion of silver eel going that route is calculated. Second, the proportion of eel that does not survive passing a barrier is calculated for each three-year period (*Paragraph 1.4*). The proportion of eel that has not arrived at the sea, but has migrated to the next hierarchy will subsequently encounter another barrier, where again a proportion of eel will survive etc. In the next paragraph the estimated migration routes and mortality estimates per hierarchy are described.

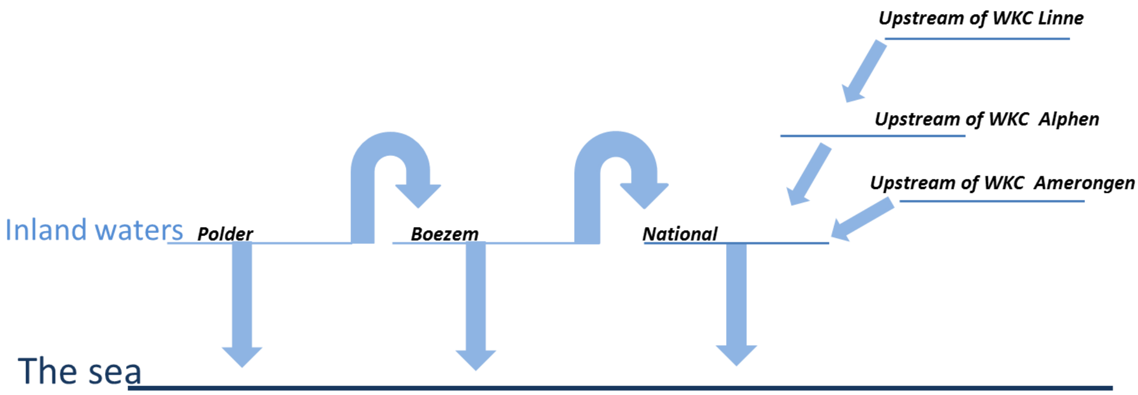

Figure 6-2 A conceptual model for estimating mortality during silver eel migration due to barriers; for 'polder' (1st hierarchy), 'boezem' (2nd hierarchy) and national waters (3rd hierarchy , see text). WKCs are HPS's in river sections of the national waters.

6.4.1 1st hierarchy: from polder to boezem or to the sea.

Migration routes

Most polders (1st hierarchy) have pumping stations that discharge water into the boezem (2nd hierarchy) rather than to the sea. Only some coastal polders have pumping stations that discharge water directly to the sea. In the model it is assumed that 80% of the eel in polder waters migrate to boezem waters where additional mortality due to sequential barrier passage might occur. The remainder (20%) is migrating directly from the polder to the sea, such as all polders in Zeeland and part of the polders in Zuid-Holland, Friesland and Groningen.

Mortality

Silver eel migrating from the polder (1st hierarchy) to the boezem (2nd hierarchy) or directly to the sea will encounter at least one pumping station. A fraction of these migrating silver eel suffer direct and indirect mortality when passing this pumping station. The direct mortality is caused by the pumping station damaging the eel. Indirect mortality can occur at pumping stations because eel that aggregate in front of a barrier have a higher chance of being predated by piscivorous fish or birds. Also the risk of being captured by fishermen is higher around pumping stations when migrating silver eel aggregate while searching for an opportunity to pass (e.g. Winter et al., 2020). A recent study demonstrated that for migrating silver eel, pumping stations delayed migration but did not function as a permanent barrier for most eel (van Keeken et al., 2020b). Here, only the mortality (including additional mortality of injuries) when the eel pass through a pumping station is taken into account, because the indirect mortalities have not been quantified and are expected to be small compared to the direct mortality.

Pumping stations can roughly be divided into three groups (Kunst et al., 2008):

1) **Pumps** (72%):
   a. 54% Propeller pumps,
   b. 14% Centrifugal pumps and
   c. ~5% Propeller-centrifugal pumps
2) **Archimedes' screws** (27%)**:** A pump type that pumps water by turning a screw-shaped surface inside a pipe.
3) **Water wheels** (0.2%)**.**

An overview of studies on the impact of different pumping station was made and is listed in *Appendix C1*. This resulted in that especially the most common Propeller pump (*Table 6-1*) cause the highest mortality when silver eel pass through. Other pumps, water wheels and Archimedes' screws show lower mortality rates compared to propeller pumps (*Table 6-1*, *Appendix C1*).

After passage of a pumping station, eel also suffer from internal injuries which results in delayed mortality, where a fraction of 0.5 of the damaged eels were assessed to suffer delayed mortality (Kruitwagen & Klinge, 2008). Therefore, mortality was calculated as direct mortality (%) plus a fraction of 0.5 of the % damaged eels for each of the pump types. The average silver eel mortality during passage of pumping stations was estimated as the weighted average of the mortalities for each type of pumping station and estimated to be 34.7% (*Table 6-1*). The estimate of mortality at pumping stations is the same for every three-year period.

Table 6-1 Calculation of the average pumping station mortality used to estimate silver eel mortality during migration (see also Appendix C1).

Pump type	**Proportion**	**Average mortality* (%) (*Appendix C1*)**	**Weighted Mortality (%)**
Water wheel	0.002	0.0	0.0
Archimedes' screw	0.27	12.0	3.2
Centrifugal pump	0.14	12.0	1.8
Propeller-centrifugal pump	0.05	9.0	0.4
Propeller pump	0.54	56.0	29.3
Pump Mortality			34.7

* Mortality is % dead + half of the % damaged.

6.4.2 2nd hierarchy: from boezem to national waters or the sea

Migration routes

At larger boezem waters a combination of different man-made structures is usually present (see *Figure 6-1* for examples). An up to date overview was made of the most important barriers for silver eel migration from boezem to the national waters in the larger waters of the Netherlands (*Appendix C2*). The most important barriers for silver eel migration are selected: 1) based on the size of the area that discharges via the potential barrier and 2) based on the biomass distribution of silver eel as estimated in *Chapter 3* and *Chapter 4* (Winter et al., 2013a & 2013b, and updates; *Appendix C2*). For each of these most important barriers, it is known whether a passage leads directly to the sea or to a national water. Combining this information with information on the amount of silver eel per starting location/water body (*Chapter 3* and *Chapter 4*), allows for distribution estimates based on a so called 'bottom up' approach. In this approach the estimates are not based on a pooled average (as used in the 1st hierarchy) but on the real migration routes (see *Appendix C2* for the results of this assessment). For each water board, the biomass of starting eel in the boezem waters (2nd hierarchy) is divided over the different outlets with potential barriers (fluxes along the routes to sea or national waters) according to Winter et al. (2013a, 2013b). Then for each of these fluxes of silver eel biomass per outlets with potential barriers the biomass flux was corrected for barrier mortality for each specific site. In case the flux went into a national water body, then the biomass flux corrected for mortality was added to the starting biomass of silver eel in the receiving national water body. The total amount of silver eel biomass for a national water body was then corrected for the mortality that these eels were expected to suffer on their route to the sea. This approach results in estimates of biomass losses relative to the starting silver eel biomass for each of the main barriers in the 2nd and 3rd hierarchy (*Appendix C2*).

<u>Mortality</u>
Similar to the migration routes, the mortality estimates for silver eel migrating from boezem to national waters are based on an inventory of the most important migration barriers for silver eel (Winter et al., 2013a & 2013b, and updates; *Appendix C2*). Given the mortalities of barriers weighted by the amount of silver eel per barrier relative to the total amount of silver eel, the overall estimated mortality for a passage from a boezem to national waters is 15.2% and for passage to the sea the estimated mortality is 5.0%. These estimates are assumed to be the same for every three-year period.

6.4.3 <u>3rd hierarchy: from national waters to sea, including HPS's</u>

The 3rd hierarchy consist of national waters. Within the national waters there are three hydroelectric power stations (HPS), and apart from those there are mainly discharge sluices.

<u>Hydroelectric Power Stations (HPS)</u>
The overall mortality of silver eel migrating through a river site with a HPS depends on the proportion of the silver eel that go through the HPS station and the mortality they suffer when passing this, relative to the proportion of silver eel that pass through safer routes (weir, ship lock, fishway, see *figure 6-1d*). For the HPS station Linne and Alphen mortality rate was initially 24% for the eels that passed through the HPS station, and when corrected for the proportion that actually migrates through the station, resulting in 15% at HPS Alphen and 17% at HPS Linne for the total flux of silver eel at these sites for the periods 2006-2008 and 2009-2011. Data on proportion of eels divided over the different routes at a site was derived from telemetry studies (Winter et al. 2006, Jansen et al. 2007). In mid-November 2011, an altered turbine management (Buijse 2009) was implemented that resulted is a reduction of mortality rate for the hydropower stations from 24% to 19%. When corrected for the proportion that migrated through the hydropower stations from more recent telemetry studies (Griffioen et al. 2020) this resulted in 13% for HPS Alphen and 14% for HPS Linne for the periods 2012-2014, 2015-2017, 2018-2020. For HPS Amerongen a hydropower mortality of 9.5% was determined (Kemper 2014). This value was taken as a best guestimate for all periods.

<u>Migration routes and mortality 3rd hierarchy other than HPS's</u>
Similar to the boezem waters (2nd hierarchy), the mortality estimates for silver eel migrating from boezem to national waters are based on an inventory of the most important migration barriers for silver eel (Winter et al., 2013a & 2013b, and updates; *Appendix C2*). Apart from the HPS's, most of the national waters are connected to the sea by discharge sluice systems which cause no mortality. This leads to an estimate of the overall mortality rate from national water bodies (apart from mortality at HPS's) to sea of 2.0%. This estimate is higher than in previous assessments (0.5% van de Wolfshaar et al., 2018), mainly because of two new insights. First, recent intensive telemetry studies (Winter et al., 2019; Winter et al., 2020) on silver eel in the North Sea Canal area, including in lake Markermeer, showed that a substantial part (~ 40%) of the silver eel migrate from this lake via the sluice-complex Oranjesluizen to the North Sea Canal and via the pumping station sluice complex at IJmuiden to sea.

Second, the overall mortality rate at the pumping station sluice complex at IJmuiden with recent extensive telemetry studies (2017-2018) was demonstrated to be higher (15-28% per year, Winter et al., 2019; Winter et al., 2020) compared to earlier estimates (2-3%, Winter, 2011). The main reason causing this increased mortality was that a larger proportion of silver eel select a more hazardous route via the pumping area than previously estimated. The two new insights combined, i.e. 40% of Markermeer silver eels go to IJmuiden and the mortality rate at IJmuiden is much higher than earlier assessed, causes that the overall mortality rate from all national water bodies to sea (of which only a small proportion migrates to sea via IJmuiden, and the majority via other routes, e.g. via lake IJsselmeer to the Wadden Sea, via Nieuwe Waterweg and Haringvliet to the North Sea) to be 2.0%, whereas in the previous evaluation in 2018 this was underestimated at 0.5%. In this report these new insights are applied to the calculations of migration mortality of all three-year periods.

6.5 Summary

6.5.1 Model scheme 2018-2020

Based on the migration routes and mortality estimates reported above, the model scheme was filled with the estimated mortalities (*Figure 6-3*).

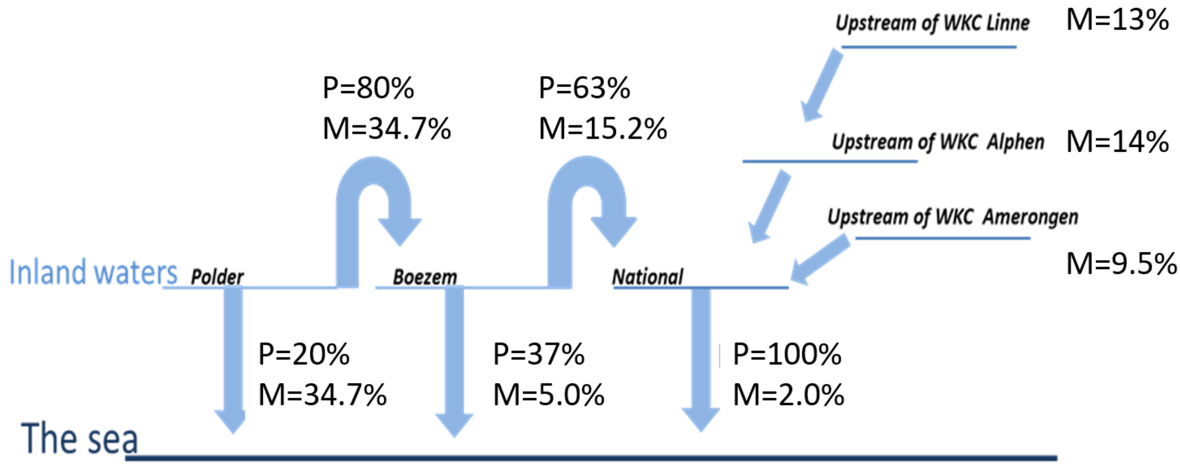

Figure 6-3 Migration scheme, listing the used to estimate overall migration mortality of silver eel. 'WKC' = HPS. P: the percentage within each hierarchy (polder, boezem or national) of silver eel migrating to sea or to the next level. Mortality estimates for the most recent period (2018-2020).

6.5.2 Mortality estimates per period and hierarchy

The final mortality percentages are listed in *Table 6-2*. For the 1st (polder waters) and the 2nd (boezem waters) only one estimate is used for all periods. Changes in barrier mortality over time thus only occur in the HPS's (Linne and Alphen) and in the national waters (*Table 6-2*)

Table 6-2 Silver eel migration barrier mortality rates per period.

	2006 – 2008	2009 – 2011	2012 – 2014	2015 – 2017	2018 – 2020
From polder to boezem	34.7%	34.7%	34.7%	34.7%	34.7%
From polder to sea	34.7%	34.7%	34.7%	34.7%	34.7%
Boezem to national waters	15.2%	15.2%	15.2%	15.2%	15.2%
Boezem to passage to the sea	5.0%	5.0%	5.0%	5.0%	5.0%
HPS Linne	17%	17%	13%	13%	13%
HPS Alphen	15%	15%	14%	14%	14%
HPS Amerongen	9.5%	9.5%	9.5%	9.5%	9.5%
Mortality national waters to sea	2.3%	2.3%	2.1%	1.5%	2.0%

6.5.3 Total mortality rates

For the estimation of the key stock indicators in *Chapter 7*, a single estimate of barrier mortality for migrating silver eel is needed. The migration routes and mortality rates as described above (*Table 6-2*) are combined with the starting biomass estimates per three-year period and location (*Chapter 3* and *Chapter 4*) for each three-year period. This results in an estimate of the total proportion of barrier mortality of migrating silver eel ($M_{barrier}$). The proportion of barrier mortality showed a decrease from the first period (0.17, 2006-2008, *Table 6-3*) to 0.13 in the latest period (2018-2020, *Table 6-3*). These estimates are used in *Chapter 7* for the estimation of the key stock indicators.

Table 6-3 Total silver eel barrier mortality for all hierarchies combined.

Period	Barrier mortality ($M_{barrier}$)
2006-2008	0.17
2009-2011	0.16
2012-2014	0.14
2015-2017	0.11
2018-2020	0.13

6.6 Discussion

Given the large number of small polders (1st hierarchy) and the lack of site-specific data for most of these sites, mortality rates for those waters are based on overall silver eel production estimates combined with an overall calculated average mortality rate. For the boezem (2nd hierarchy) and national waters (3rd hierarchy), a so called 'bottom up' approach was followed, using site specific mortality estimates (Winter et al., 2013a &, 2013b, and updates). This approach yielded more accurate estimates than the more general approach based on averages as used for the 1st hierarchy. However, the quality of the underlying data that was used in the updated silver eel barrier assessments is highly variable and often still incomplete. Some sites are very well studied, e.g. the sites with HPS's in the River Meuse (Winter et al., 2006 & 2007; Jansen et al., 2007; Griffioen et al., 2020), the discharge sluices complexes in Haringvliet (Winter & Bierman, 2010) and at the sluices-pumping station complex at IJmuiden (Winter 2011; Winter et al., 2019 & Winter et al., 2020), but for other sites, e.g. ship locks and most of the pumping stations, data on silver eel mortality at a specific site are usually lacking. However, for some of these sites conditions have changed since these studies were carried out and updated estimations are needed to determine current mortality losses due to pumps or HPS's. Research that determined mortality for eels that passed through HPS's in the River Meuse was carried out in 2002 when average silver eel length in the river Meuse was 65cm (Bruijs et al. 2003). However, the average length of silver eel in the upper parts of the river Meuse increased over the years and now averages 83cm (period 2012-2020, WMR data), which will result in higher mortality when passing the HPS's. In future evaluations the effect of increased average length on mortality should be determined. In addition, the relative proportion of silver eels that pass through the turbines over other safer pathways (weir, sluices, lateral canals), will be affected by changes in discharge patterns (e.g. increasing incidence of dry periods in summer and autumn) and recent management measures (e.g. periodically closing turbines during nights in autumn and additional fish friendly turbine operation adaptations). In future evaluations, additional studies and data need to be incorporated to determine overall HPS's mortality under the recently changed conditions. Last, the barrier-mortality approach as used here for the 2nd and 3rd hierarchy waters can be further developed to enable a full site-specific and data driven approach including the 1rst hierarchy. Several maps and lists of barriers are available (e.g. Kroes et al., 2018; Buijse et al., 2009 and in the Amber barrier atlas[4], the National Fish Migration Route Map RWS/Nationale Visroutekaart RWS).

[4] https://amber.international/barrier-atlas/

7 Stock indicators

7.1 Stock indicators

Under the eel regulation (EC 1100/2007) MS's are obliged to report on a list of stock indicators (*Table 7-1*, this is a similar table as *Table 1-2*, but listed here again for readability purposes). The stock indicators are based on the biomasses and silver eel barrier mortalities as estimated in previous chapters (*Chapter 3,4,5 & 6*) and additional information on retained catches. In this chapter, these key stock indicators are estimated. Only B_0 (pristine silver eel biomass, *Table 7-1*) is a constant value that was calculated in 2010 (ICES 2010b). The B_0 value for inland waters in the Netherlands is set at 10,400 tonnes (ICES 2010b). The other stock indicators vary per period.

In this assessment, mortalities during the yellow eel and silver eel stages are split into two groups. The reason is that yellow eel mortalities apply over a sequence of years as the yellow eel stage usually takes between 3-20 years. Silver eel mortality is assumed to apply during a single year in the life cycle of an eel. The yellow and silver eel mortalities are combined in a single mortality rate, the 'Lifetime Anthropogenic Mortality' (*ΣA, Table 7-1*). *ΣA* is the mortality that eel experience throughout their lifetime and it is based on the ratio between the current silver eel biomass escapement ($B_{current}$, *Table 7-1*) and the best possible escapement under current recruitment levels (B_{best}, *Table 7-1*).

Table 7-1 Overview of the main stock indicators to be reported to the EC. The MS's are also obliged to report on the amount of glass eel (eel <12 cm) that are harvested for restocking. These are not reported here because this is not relevant for the Netherlands; no glass eel are harvested.

Indicator	Description
B_0	Pristine silver eel biomass. An estimate of escapement in the absence of any anthropogenic impact and at historic recruitment levels.
$B_{current}$	Silver eel biomass estimate of the current silver eel escapement to the sea.
B_{best}	The best silver eel biomass possible under current recruitment levels, i.e. silver eel biomass estimate without anthropogenic influences on yellow eel and silver eel stock, i.e.
ΣF	Fishing mortality rate (yellow and silver eel, commercial and recreational).
ΣH	Anthropogenic mortality rate from other sources than fishing mortality. This is mainly barrier mortality during downstream migration.
ΣA	Total anthropogenic mortality rate, i.e. the sum of *ΣF* and *ΣH*.

7.2 Yellow eel anthropogenic mortality rate

One of the stock indicators that needs to be reported to the EC is the total anthropogenic mortality rate *ΣA (Table 7-2). ΣA* is defined here as the yellow eel fishing mortality over all ages, the silver eel fishing mortality, and the silver eel barrier mortality (see *Paragraph 7.4*). The yellow eel fishing mortality rate $\hat{F}$ is used as input to estimate the stock indicators (*Paragraph 7.4*)

In this section the yellow eel anthropogenic mortality rate $\hat{F}$ is estimated. It is defined as the yellow eel fishing mortality from both commercial and recreational catches. Yellow eel barrier mortality is not estimated, because it is expected to be very low and because it will not influence the estimation of the yellow eel standing stock, because it is already accounted for implicitly in the estimate of the yellow eel standing stock. However, it might cause for a small underestimation of the anthropogenic mortality.

The yellow eel fishing mortality rate ($\hat{F}$) is estimated as a function of the proportions of retained catches and the estimated biomasses of the standing stock, following the equation:

$$\hat{F} = -log(1 - Catch_R/(Biomass + Catch_R))$$

where $Catch_R$ is the retained catch of yellow eel by commercial and recreational fisheries. Biomass is the biomass of yellow eel (≥ 30 cm, tonnes) as estimated in *Chapters 3* and *4*. This calculation of the fishing mortality is based on the assumption that all mortality during a year occurs at once. All fishing mortality of yellow eel is assumed to take place before the surveys are conducted. The main fisheries on eel is from May to August, because of the fisheries closure from September to 1 December (part of the Dutch EMP[5]). Most of the electric dipping net surveys in regionally managed waters, ditches, and also the FYMA survey in lakes IJsselmeer and Markermeer (*Chapter 4*) take place in the autumn after the period with the main fisheries.

Based on the equation above, $\hat{F}$ is calculated for each period and scenario (*Table 7-2*). The biomasses were estimated in *Chapter 3* and *Chapter 4* and are presented in *Table 5-3* and the yellow eel landings in *Table 2-1*.

Table 7-2 Mean yearly yellow eel biomasses, mean yearly retained catches (landings) and mean yearly fishing mortality rates ($\hat{F}$) for yellow eel for each period (scenario 2). The biomasses are derived from Table 5-3 and the total yellow eel removals (landings) from Table 2-1.

	Yellow eel		
period	**Standing stock Biomass (tonnes)**	**Retained catches (commercial and recreational, tonnes)**	**Fishing mortality rate ($\hat{F}$)**
2006-2008	3,280	725	0.20
2009-2011	5,093	309	0.06
2012-2014	5,034	223	0.04
2015-2017	4,773	201	0.04
2018-2020	3,869	278	0.07

7.3 Silver eel anthropogenic mortality

The silver eel anthropogenic mortality proportion α represents the fishing and barrier mortality during migration from freshwater to the sea (*Chapter 6*). The mortality is calculated as the proportion of losses due to anthropogenic mortality relative to the silver eel biomass at the start of migration:

$$\alpha = 1 - (B_{start} - Catch_R) * (1 - M_{barrier})/B_{start}$$

Where B_{start} (*Table 7-3*) represents the silver eel biomass before silver eel mortalities (migration and fisheries) have occurred; $Catch_R$, (*Table 2-1*) represents the retained silver eel catch; and $M_{barrier}$ (*Chapter 6*) represents the proportion barrier mortality. The parameter α is calculated for each assessment period (*Table 7-3*).

As for yellow eel mortality, the fishing mortality of silver eel is assumed to take place before the surveys are conducted (see *Paragraph 7.1*). Therefore, the silver eel biomass estimate before anthropogenic mortality B_{start} is assumed equal to the sum of the estimated standing stock biomass in autumn and the silver eel landings.

[5] With the exception of the water board Wetterskip Fryslan, where fishing in September - November is allowed due to a quota system.

Table 7-3 Silver eel biomass standing stock (as estimated in Chapter 3 and Chapter 4), total silver eel retained catches ($Catch_R$), the biomass prior to anthropogenic mortalities (B_{start}), the barrier mortality proportion ($M_{barrier}$) and the total anthropogenic mortality proportion during migration from freshwater to the sea (α) for scenario 2.

		Silver eel			
Period	**Standing Stock biomass (Ch3,4 and5)**	***$Catch_R$* (commercial)**	***B_{start}***	**Barrier mortality proportion ($M_{barrier}$)**	**Anthropogenic mortality proportion (α)**
2006-2008	769	280	1,049	0.17	0.37
2009-2011	999	175	1,174	0.16	0.29
2012-2014	1,521	140	1,661	0.14	0.21
2015-2017	1,652	143	1,795	0.11	0.18
2018-2020	1,113	201	1,314	0.13	0.26

7.4 %SPR, ΣA, $B_{current}$ and B_{best}

To calculate the Lifetime Anthropogenic Mortality rate, *ΣA*, yellow eel and silver eel mortality estimates were split into a fishing and a barrier component. Barrier mortality is only estimated for downstream migrating silver eel and is thus assumed not to affect yellow eel. The estimated yellow eel and silver eel mortalities (see previous paragraphs) are used to estimate the total 'Lifetime Anthropogenic Mortalities' (*ΣA*). To estimate *ΣA*, first the *%SPR* (Spawner per Recruit), $B_{current}$ (current silver eel escapement) and B_{best} (best possible current silver eel escapement) are estimated. The basis for the methods used is formulated by ICES (WGSGIPEE, 2010a & 2011).

To estimate *%SPR*, the parameter α (percentage of silver eels that die during migration, *Paragraph* 7.3), and β, the proportion silver eel production out of the best possible silver eel production are used. Parameter α is calculated in *Paragraph 7.2,* and parameter β is calculated using the demographic model (*Chapter 4*). However, the survey data for lakes IJsselmeer and Markermeer are not used in the model. Instead, based on the matrices $\mathbf{A}_f$ and $\mathbf{A}_m$ (*Paragraph 4.2*), the age and sex specific maturation probability (M_{gi}), the lengths at mid-age of the age classes and the length-weight relationships (*Figure 2-4*, *Chapter 2*), a ratio between the maturing biomasses for F = 0 and for fishing mortalities equal to the values as estimated in paragraph 7.2 ($\hat{F}$;*Table 7-2*) was calculated. This ratio of the maturing biomasses is expressed as the proportion silver eel production out of the best possible production (if no mortality had taken place). This proportion is represented by the parameter β. Subsequently, the *%SPR* is estimated as:

$$\%SPR = 100 * \beta * (1 - \alpha)$$

The estimate of the current escapement of silver eel $B_{current}$ is equal to the surviving part of the starting value of silver eel (B_{start}) after removal of all silver eel anthropogenic mortalities and is calculated as:

$$B_{current} = (B_{start} - Catch_R) * (1 - M_{barrier})$$

$B_{current}$ and $\%SPR$ are used for the estimate of B_{best} (the best possible escapement of silver eel, if all anthropogenic mortalities for yellow and silver eel are zero). B_{best} is calculated as:

$$B_{Best} = B_{current}/\%SPR$$

Subsequently, the Lifetime Anthropogenic Mortality rate is calculated as:

$$\Sigma A = -\ln (B_{current}/B_{best})$$

The indicators *%SPR*, $B_{current}$, B_{best} and ΣA were calculated for five different periods (*Table 7-4*). The results show that since the first period (2006-2008) the yellow and the silver eel stock biomass have increased until the latest period. The starting value of silver eel (B_{start}, *Table 7-4*) increased from 1,049 tonnes in 2006-2008 to 1,795 tonnes in 2015-2017). In the latest period (2018-2020), however, it decreased with almost 500 tonnes to an estimate of 1,314 tonnes. The anthropogenic mortality rate ΣA showed a decreasing trend, with a huge decrease between the first and second period, and a slower decrease until 2015-2017. However, in the most recent period, the anthropogenic mortality increased again, to a value of ΣA = 0.79 (corresponding to 55% mortality). This is due to a lower estimate of the standing stock compared to previous years, as well as an increase in the landings (344 tonnes in 2015-2017 vs. 479 tonnes in 2018-2020, *Table 7-4*). The results will be discussed in more detail in *Chapter 8*.

Table 7-4 Overview of all stock indicators per period. Yellow eel and silver eel stock estimates refer to eel (≥30 cm). Values are for scenario 2 (best guess estimate, see Chapter 3). Values for $B_{current}$ (tonnes) and ΣA for the other scenarios are listed in Table 7-5.

		2006-2008	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
Yellow eel	Yellow eel stock (tonnes)	3,280	5,093	5,034	4,773	3,869
	Retained catch (tonnes)	725	309	223	201	278
	$\hat{F}$	0.20	0.06	0.04	0.04	0.07
	β	0.27	0.66	0.73	0.74	0.61
Silver eel	Silver eel stock (tonnes)	769	999	1,521	1,652	1,113
	Retained catch (tonnes)	280	175	140	143	201
	Mortality Barriers (prop)	0.17	0.16	0.14	0.11	0.13
	$\boldsymbol{\alpha}$	0.37	0.29	0.21	0.18	0.26
	B_{start} (tonnes)	1,049	1,174	1,661	1,795	1,314
	$B_{current}$ (tonnes)	634	837	1,311	1,463	974
	B_{best} (tonnes)	3,759	1,791	2,270	2,420	2,153
Lifetime	*%SPR* (spawner per recruit)	16.9%	46.8%	57.8%	60.5%	45.2%
	%LAM (anthropogenic mortality %)	83.1%	53.2%	42.2%	39.5%	54.8%
	ΣA (anthropogenic mortality rate)	1.78	0.76	0.55	0.50	0.79
	ΣH (barrier mortality rate)	0.22	0.19	0.20	0.18	0.18
	ΣF (fisheries mortality rate	1.56	0.57	0.35	0.33	0.61

Table 7-5 $B_{current}$ (tonnes) and ΣA for all 3 scenario's. The scenario's represent the uncertainty of main assumptions in the static spatial model. Scenario 2 is the best guess scenario (Chapter 3).

	Scenario	**2006-2008**	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**
	1	236	249	344	376	317
$\boldsymbol{B_{current}}$	**2 (best guess)**	**634**	**837**	**1,311**	**1,463**	**974**
	3	756	1,034	1,662	1,861	1,204
	1	3.72	2.01	1.51	1.42	1.95
ΣA	**2 (best guess)**	**1.78**	**0.76**	**0.55**	**0.50**	**0.79**
	3	1.55	0.64	0.46	0.42	0.67

8 Evaluation of the EMP

8.1 Precautionary approach and limit reference points

In this chapter the impact of the EMP is evaluated using the methods as developed by ICES (2014). To be able to evaluate the status of the eel stock, ICES (2014) developed a precautionary approach (PA) framework. The PA framework uses limit reference points (B_{lim} and F_{lim}, *Table 8-1*) reflecting stock states that should be avoided. Within the eel framework, the limit reference points are set such that they take the uncertainty of the limit reference points into account, hence the precautionary reference points (B_{pa} and F_{pa}, *Table 8-1*) are set at the same value as the limit reference points (B_{lim} and F_{lim}).

For the eel stock, no reference points reflecting the total eel stock have been established. The eel stock is divided over many water bodies in many countries, also outside the EU. This makes an assessment of the total eel stock and the calculation of reference points extremely difficult. Therefore, a precautionary diagram is developed for the eel case, such that it can be used by each MS separately. Because the reference points for the ICES approach (B_{lim}, F_{lim}, *Table 8-1*) had not been established for eel, alternative biomass and mortality reference points were developed (ICES, 2014; *Table 8-1*).

***B_{lim}*:** A universal provisional biomass reference point (B_{lim}) is a level of exploitation which provides 30% of the pristine (no anthropogenic mortality ever) spawning stock biomass (B_0). In 2002, ICES advised to set the biomass reference point (e.g. B_{lim}) above the universal value, at a value of 50% of the virgin spawning-stock biomass, to account for uncertainty, such that $B_{pa} = B_{lim}$. The EU (Council Regulation 1100/2007), however, decided to set B_{lim} at 40% of B_0, in-between the universal level (30%) and the level advised by ICES (50%).

***A_{lim}*:** Eel experience relative high levels of anthropogenic mortality in addition to fishing mortality compared to other commercially exploited stocks. Therefore, the mortality reference point (A_{lim}) includes all anthropogenic mortality (A) and not only the fishing mortality (F). The EU Eel Regulation (Council Regulation 1100/2007) has set the limit for the escapement of silver eel (B_{lim}) at 40% of the pristine escapement (B_0). A_{lim} is derived from B_{lim} as follows: $\Sigma A=-ln(40\%)=0.92$ (ICES 2018). Thus, an eel stock with a biomass of escaping silver eel of 40% of B_0 is estimated to correspond to a lifetime anthropogenic mortality limit of A_{lim} = 0.92. At low biomass, however, the anthropogenic mortality advised is reduced, to reinforce the tendency for the stock to rebuild (ICES, 2018).

The status of a local eel stock (within an eel management unit) is in an undesirable state if it is below either B_{lim} or A_{lim}.

Table 8-1 Reference points and stock indicators needed for the precautionary approach.

Reference point	Definition	Value
B_{lim}	Biomass limit below which a stock is considered to have reduced reproductive capacity.	40% * B_0
A_{lim}	Mortality rate limit above which a stock decline is expected.	0.92
Stock indicator		
B_0	Silver eel biomass without any anthropogenic influences (pristine biomass).	10,400 t
$B_{current}$	Silver eel biomass that currently (assessment year) escapes to the sea to spawn.	*Table 8-2*
ΣA	Life time anthropogenic mortality; the fishing mortality and the mortality outside of fisheries (HPS's, pumping stations etc.).	*Table 8-2*

8.2 *Status of the eel stock in the Netherlands*

To assess the stock status, first the current silver eel escapement biomass ($B_{current}$) in relation to the estimated pristine situation (B_0) is calculated (*Paragraph 7.3*, *Table 8-2*) and subsequently plotted against the current lifetime anthropogenic mortality rate (*ΣA, Table 7-4, Figure 8-1)*.

The evaluation demonstrates that the status of the eel in the Dutch waters is still in the 'red' area of the precautionary diagram (*Figure 8-1*) and thus remains in a situation regarded as undesirable, with high mortality and low biomass. The current biomass of escaping silver eel is 9.4% of the pristine situation which is below the target of 40%. The value of the current lifetime anthropogenic mortality ($ΣA$ = 0.79) lies below A_{lim} (A_{lim} = 0.92). However, the recommended mortality at the current estimate of the percentage of escaping silver eel is below the current anthropogenic mortality (*Figure 8-1*, 'red area').

Table 8-2 Stock indicators used to evaluate the impact of the EMP on the biomass of escaping silver eel and anthropogenic mortality. Biomasses are in tonnes (t).

Period	B_0*	$B_{current}$	100* $B_{current}/B_0$	ΣA
2006-2008	10,400 t	634 t	6.1%	1.78
2009-2011	10,400 t	837 t	8.1%	0.76
2012-2014	10,400 t	1,311 t	12.6%	0.55
2015-2017	10,400 t	1,463 t	14.1%	0.50
2018-2020	10,400 t	974 t	9.4%	0.79

* Excluding coastal waters (2,600 t)

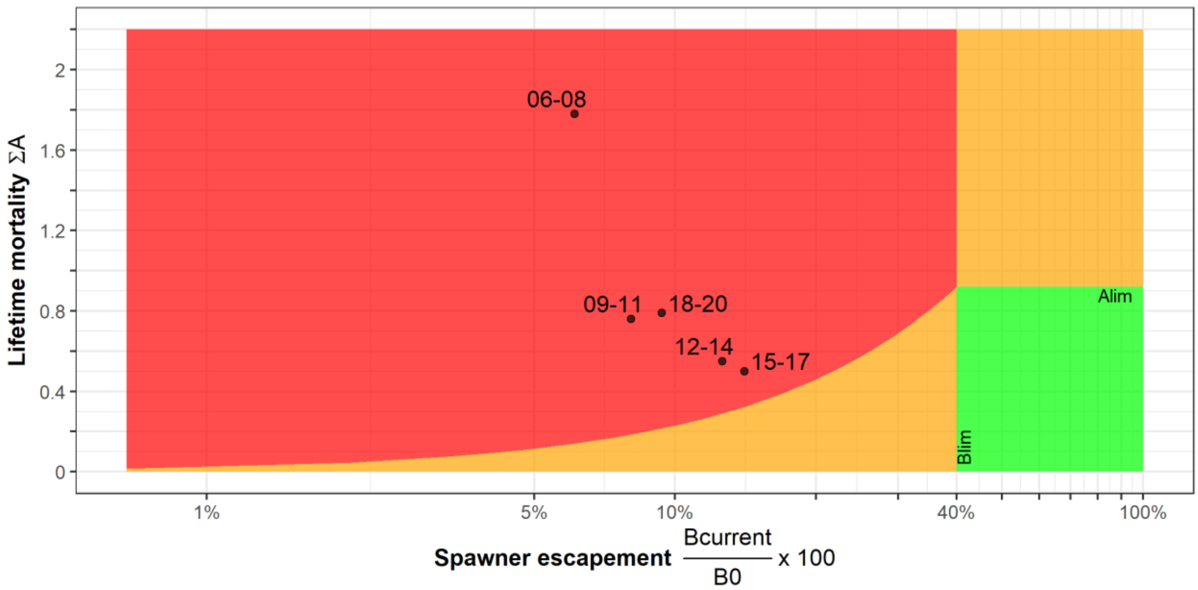

Figure 8-1 ICES modified precautionary diagram presenting the status of the eel stock in the Netherlands in 2006-2008, 2009-2011, 2012-2014, 2015-2017 and 2018-2020 with respect to management targets. The horizontal axis represents the status of the stock in relation to pristine conditions. The vertical axis represents the impact made by anthropogenic mortality. ΣA = Lifetime anthropogenic mortality, presented as a rate. Note that the x-axis is on a logarithmic scale.

8.3 Discussion of the status of the eel stock in the Netherlands

The status of the eel stock in Dutch waters remains in a situation regarded as undesirable with low biomass. In the precautionary diagram (*Figure 8-1*), the horizontal axis demonstrates the current biomass in relation to the best possible biomass, while the vertical axis illustrates the level of anthropogenic mortality on the stock. Below the interpretation of the axes is discussed in more detail.

8.3.1 Anthropogenic mortality (vertical axis in *Figure 8-1*)

A reduction in lifetime anthropogenic mortality (*ΣA, Figure 8-1*) can be achieved by reducing fishing mortality and barrier mortality. A reduction in anthropogenic mortality is therefore the direct result of the measures taken by a MS. In the Netherlands, the implementation of the EMP has resulted in a reduction in *ΣA* between the first period (2006-2008) and the second-last period (2015-2017) from 1.78 to 0.50, corresponding to an increase in the percentage spawner per recruit (*%SPR*) from 17% to 60% (*Table 7-4*). This reduction in *ΣA* was mainly the result of a decrease in fishing mortality, both commercial and recreational: retained catches (landings) of both commercial and recreational fisheries strongly decreased between 2006-2008 and 2015-2017. The greatest reduction in mortality was achieved in the second period (2009-2011), showing the result of the implementation of the eel management plan (2009), as a result of which the fishing mortality has reduced to a large extend (from *ΣF* =1.56 in 2006-2008 to *ΣF* = 0.57 in 2009-2011, *Table 7-4*). However, in the most recent period (2018-2020), the mortality rate has increased from 0.50 to 0.79 (*Table 7-4*, *Figure 8-1*). This is caused by an increase in the commercial fisheries (landings and effort), mainly in the lakes IJsselmeer and Markermeer (*Table 2-1* & *Appendix A0*).

Barrier mortality (*Mbarrier)* showed a decrease from 17% to 13% (*Table 6-3*) from 2006-2008 to 2018-2020. The reduction is caused mainly due to measures at HPS's (new management scheme), replacing some pumping stations with 'fish-friendlier' pumping types and differences in eel distribution. From 2015-2017 to 2018-2020, the barrier mortality increased from 11% to 13% (*Table 6-3*). This increase is not caused by new barriers (no HPS's or pumping stations were placed during this period), but because of a difference in the distribution of eel.

Lifetime anthropogenic mortalities were estimated using the retained catches and barrier mortalities in relation to the standing stock. The current *ΣA* is calculated by taking the sum of the mortalities of all ages. This is not the same as the *ΣA* that new recruits (glass eels) are expected to experience throughout their inland life span. The *ΣA* in a new cohort recruits may differ from the current *ΣA* because of different mortality rates compared to the current rates. This could be a result of effects of the measures taken to reduce mortality, such as closed areas (main rivers and some large canals) and reductions in fishing mortalities. The estimated *ΣA* consist of fisheries mortality over all life stages and barrier mortality of silver eel. The silver eel biomass is a result of the surviving yellow eel after yellow eel mortality occurred. Therefore, silver eel mortality contributes usually less to *ΣA* compared to yellow eel mortality.

8.3.2 Biomass escaping silver eel (horizontal axis)

Between the periods 2006-2008 and 2015-2017, there was an increase in the biomass estimate of escaping silver eel ($B_{current}$) in every period, with the largest increase between 2009-2011 and 2012-2014. However, there was a decrease in the estimate of escaping silver eel biomass between the period 2015-1017 and 2018-2020 (horizontal axis; *Figure 8-1*). Large differences between periods in biomass were not expected as an increase in glass eel recruitment will, at the earliest, result in an increase of silver eel after 3-20 years. However, glass eel recruitment has not significantly increased after the implementation of the EMP in 2009 (ICES 2020). The level of glass eel recruitment, depends only for a small part on the status of the Dutch part of the eel stock. If one EMU alone, such as the Netherlands, would reduce all anthropogenic mortality to zero, a recovery of the European eel stock is still not necessarily expected. In order to maximize the chance of recovery, maximum protection of European eel will have to be accomplished throughout its natural range, within and outside Europe. The responsibility for improvement of eel stock lies with all countries in the natural range of the eel distribution.

9 Conclusions and recommendations

9.1 Biomass estimates

The EC requested the MS's to evaluate the status of the European eel stock. In this report, the data and methods which were used to estimate the stock indicators for the Dutch part of the eel stock (B_{best}, $B_{current}$, B_0 and ΣA) were described. However, the estimates of the stock indicators used to evaluate the status of the stock ($B_{current}$, B_{best} , B_0, and ΣA, *Table 7-4*) need to be interpreted with care due to the significant level of uncertainty surrounding these estimates. In this final chapter, the used methodologies and results are discussed. Furthermore, recommendations are provided for further improvements of the models.

The main results of this assessment are that in the most recent period (2018-2020), the current silver eel escapement $B_{current}$ (974 tonnes), is still very much below the target of 40% of the estimated pristine situation (B_0). The anthropogenic mortality (ΣA = 0.79) is below A_{lim} (A_{lim} = 0.92). However, at the current estimate of $B_{current}$, the anthropogenic mortality was above the mortality as the target set in the EMP (*Figure 8-1*). Therefore, the status of eel in Dutch waters remained in a situation regarded as "undesirable" for both the silver eel escapement and anthropogenic mortality (red region in *Figure 8-1*).

After implementation of the EMP in 2009 the estimate of $B_{current}$ increased in every reporting period until 2015-2017: from 634 tonnes in 2006–2008 to 1,463 tonnes in 2015-2017. However, in the most recent period (2018-2020) the estimate of $B_{current}$ declined again to 974 tonnes. The decline is mainly a result of a lower estimate of the starting silver eel stock (B_{start}), which results from a lower estimate of the yellow eel stock: from 4,773 tonnes in 2015-2017 to 3,869 tonnes in 2018-2020 (*Table 7-4*). This decline in yellow eel biomass estimate is a direct result from the national surveys other than the large lakes and from the regional (WFD) surveys (*Chapter 3*). The biomass estimated in the national surveys other than the large lakes declined from 2842 tonnes in 2015-2017 to 1,673 tonnes in 2018-2020 (*Table 5-2*). The main decline was observed in the 'Benedenrivieren' (*Figure 3-4*) which declined from 2,147 tonnes to 1,142 tonnes between 2015-2017 and 2018-2020 (*Table 3-9*). The 'Benedenrivieren' is one of the larger water bodies and (like all large rivers) the eel fishery is closed. Because the 'Benedenrivieren' comprises such a large area, the influence of the survey outcome in this area is large on the total biomass. The total biomass estimate in the regional waters (WFD waters) also declined from 2,265 tonnes in 2015-2017 to 1,791 tonnes in 2018-2020 (*Table 5-2*). Within the regional waters, Wetterskip Fryslân is highly influential because it represents the highest biomass estimate (*Appendix A3*). In this region, there is also a lot of restocking of glass eel, which could cause fluctuation in the biomass estimate between periods, because also waters with glass eel restocking are monitored. Since eel fishing is based on a yearly set quota (36.6 tonnes for all fishermen), the lower biomass cannot be related to increased catches. The biomass in lakes IJsselmeer and Markermeer showed an opposite trend between 2015-2017 to 2018-2020, with an increase in the standing stock biomass from 299 tonnes to 476 tonnes (*Table 5-2*). However, also the estimated fishing mortality is very high in lakes IJsselmeer and Markermeer ($F = 1.10$) and the landings have also increased (from 193 to 308 tonnes, yellow and silver eel combined, *Table 4-2*).

For all components of the standing stock biomass estimates, the accuracy is low. For the static spatial model (*Chapter 3*), main sources causing low accuracy are the selectivity of the electric dipping net and the habitat preference (*Paragraph 3.2*). However, apart from the selectivity and the habitat preference there is probably also a high level of sampling error. Even though the water bodies have been sampled at least once in every three-year period, and the number of hauls is substantial, the amount of sampling per water body is still small. Variation between years can arise due to the condition during sampling (water level, weather, exact location, time in the year, sampler), which cause additional sampling error. Also, high variation may be caused by sampling in water bodies were restocking occurred in recent years. How much of the changes in eel standing stock biomass is caused by sampling error is impossible to say. Inaccuracy in the demographic model is mainly caused by low accuracy of many components of the input data. The biological keys and natural mortality (*Chapter 2*) are uniform in time and the same keys are assumed as for all water bodies. Also, the relative selectivity by the survey gear per length is unknown. These cause uncertainty in the estimated fishery mortality values, and as a consequence in the biomass estimate. For the large lakes Grevelingen and Randmeren, strong assumptions had to be made, also causing a large amount of uncertainty.

9.1.1 Pristine biomass estimate (B_0)

$B_{current}$ and ΣA are not the only stock indicators that affect whether the state of the eel stock in the Netherlands is in a desired state and reaches the goals set out by the Eel Regulation (*Chapter 1*). The x-axis of the precautionary diagram represents the ratio between $B_{current}$ and B_0 and thus depends highly on the estimate of B_0. The B_0 value for inland waters in the Netherlands is set at 10,400 tonnes. However, the uncertainty of the value is large and has been subject to discussion. Initially the pristine silver eel biomass (B_0) in the Netherlands, was set at 10,000-15,000 tonnes (Klein Breteler, 2008). In a first review (Eijsackers et al., 2009) it was concluded that the range was wider and that B_0 was between 6,500-20,250 tonnes. However, ICES (review of the national eel management plans, ICES 2010b) did not accept all arguments of Eijsackers et al. (2009) and set B_0 at 13,000 tonnes. A second review (Rabbinge et al., 2013) concluded that the method to calculate B_0 was fundamentally of good quality with respect to adhering to the guidelines set by the Eel Regulation. However, the estimation of the value of B_0 is generally acknowledged to be extremely difficult. Due to limitations in data from earlier periods, the variation in numbers per water body, historical restocking levels and uncertainties about density dependent natural mortality, it is effectively impossible to estimate a reliable estimate of B_0 for the Netherlands.

9.2 Biological keys

The maturity-at-length and the sex ratio-at-length were analysed with a *GAM* instead of a binomial *GLM* (*Chapter 2*). *GAM*'s are non-linear and therefore do not have a forced shape. In contrast, a binomial *GLM* has a fixed shape between 0% and 100%. Because of this, the previous use of the binomial *GLM* had strong assumptions, such as a 100% male sex ratio at small lengths in the sex ratio-at-length key and 100% chance of maturation at large lengths in the maturation-at-length key. These assumptions did not fit the data distribution and biology very well. Because *GAM's* do not have fixed shapes of the fitting curve, *GAM* curves fit the data much better compared to the binomial *GLM*. On the other hand, in a *GAM* no underlying relationship is assumed, therefore the final shape has to be chosen by vision and is consequently partly a result of expert judgement.

The new maturity-at-length had substantial impact on the final results, because a smaller proportion of eel was assumed to grow into silver eel within a year compared to the binomial *GLM*. The growth-at-length curve (*Paragraph 2.6*) also changed. As growth in eel is different in that eel can grow to very large lengths, a von Bertalanffy growth fit was not necessarily expected. However, it fitted the data quite well and was therefore used for this assessment.

Natural mortality depends on many factors, such as predation, water temperature, pollution and food conditions, which makes it a difficult parameter to assess. Natural mortality is also unlikely to be the same for all stages and is also not constant through time. The natural mortality used in the demographic model (*Chapter 4*) is based on Dekker (2000), who made a best guess based on literature. The above-mentioned factors cause the used value of natural mortality (M = 0.138) to be highly uncertain.

9.3 Static spatial model

9.3.1 Regionally managed waters

In the biomass assessment for the regional managed water bodies WFD fish survey data was used (*Chapter 3*). A problem with this data is that not all water bodies are sampled in the same manner. Some water boards sample more frequently than others. Also, even though the sampling intensity has increased, the sampling does not cover all water bodies. The choice of the waterbody, but also the location and timing within the water body are important for accurate comparison of the density between water bodies and years. As the sampling has a time frame of six years, a six-year moving average was calculated. For each three-year period, the six years closest to the three-year period were chosen. This allows one to detect changes over time. This differs from previous reports (Bierman et al., 2012; van de Wolfshaar et al., 2015 & 2018), were a single estimate for all years was used. As a consequence, irregular or inconsequent sampling has a higher influence on the final biomass estimate.

9.3.2 Nationally managed waters

The most important causes of uncertainties in the biomass estimates of the nationally managed waters are:

- In the assessment of the nationally managed waters, the biomass estimate of one river section (Benedenrivieren) dominated the overall biomass estimate of the nationally managed waters. The Benedenrivieren have a relative large surface area (*Figure 3-4*), and as a consequence a high biomass estimate, but also the survey density is high (*Table 3-6*). In this area, the influence of silver eel migrating from other areas or countries may be high, because it is the area closest to the coast where silver eel might concentrate before starting migration. However, because the water body is also relatively wide, the assumption of the habitat selection has a large impact compared to other smaller waterbodies.

- Different river regions are surveyed in different months. As a result, water temperature, eel behaviour and silver eel migration activity may differ because of the sampling period, causing additional noise in the estimations.

- In the current assessment, the eel stock in the large lakes (IJsselmeer, Markermeer, Randmeren and Grevelingen) was determined using the demographic model and the landings. This method is a bit cumbersome. If research was done on a better understanding of eel distribution in the lakes using all available surveys, upscaling eel densities from the littoral zone to lakes as a whole could be carried out to validate the results from the demographic model.

9.4 Demographic model

The main decreasing stock trends since 1989 in lakes IJsselmeer and Markermeer could be explained reasonably well by the demographic model (*Chapter 4*), but only to a certain extent. For example, the increase in eel numbers 7+ years after arrival in Lake IJsselmeer in recent years is not captured by the model. Also, small changes in the data are not captured. For this report, several updates were made to the model. The most important updates of the demographic model were:

- The model is fitted to survey data of lakes Ijsselmeer and Markermeer together, due to scarcity of data in lake Markermeer in some years.

- Different periods for which a single *F* estimate was calculated were changed such that the periods still represented the changes in eel fisheries management, but also such that the number of years for each period does not fluctuate too much.

- The length-class based fit between model and data that was previously used, has been changed to an age-class based fit to allow for a better comparison between the model and the survey data.

- The moment of comparison of the model with the survey data was moved from April to October, to better fit the ages of the individuals in the survey data.

- The updated biological keys were used as input and a different initial sex ratio was assumed.

Several sensitivity analyses showed that the estimated *F* value by the demographic model is sensitive to differences in the biological keys. Although the only parameter that can change over time is the fishing mortality, the model is sensitive to the assumed maturity-at length, growth rate and initial sex-ratio. As eel mature to silver eel they migrate to the ocean. This means that they leave the lakes, which means that they are also 'removed' from the model. If this happens at smaller lengths, eel have left the system at an earlier age, which has consequences for the estimated *F*.

The demographic model also assumes that the selectivity of the survey gear (FYMA, *Chapter 2*) is equal for all length classes at ages from 2 years and older. However, the selectivity of the survey gear is not known. Changes in the assumption of the survey gear at length will influence the outcome. Similarly, a single estimate of *F* is calculated for all eel above the minimum landing size.

Last, the estimate of natural mortality is highly uncertain. It is assumed constant for all ages and sizes, which is unlikely to be true as smaller eel are more prone to, for example, predation. Different assumptions of *M* will result in different estimates of *F*, and thus in a different biomass estimate.

9.5 Barrier mortality

Water boards did invest substantially in improving migratory opportunities at migration barriers, but most solutions targeted to facilitate upstream migration. Potentially, this has improved glass eel immigration into inland waters and as a consequence indirectly enhanced the potential silver eel biomass starting to migrate in the different waters. Mitigation of mortality in a downstream direction is more difficult since it requires replacing pumping stations or HPS's or deflecting silver eel to alternative routes with no mortality, for which effective measures are still largely lacking (Kroes et al., 2013).

Much investment is still being carried out by the water boards to improve upstream migration along barriers into and within inland Dutch waters (measures for to the WFD). This may have led to an increased rate of immigration of glass eel into inland waters and. However, little is known on the immigration of glass eel and distribution of glass eel over inland water, and no quantification of the overall outcome of these migration mitigation measures at barriers can be made at present.

As was demonstrated for a validation with extensive telemetry studies and mark-recapture experiments for the North Sea Canal catchment (see *Appendix C3*), the estimate in the model as described in *Chapter 6* is reasonable.

9.6 Unquantified sources of anthropogenic mortality:

The main sources of mortality of European eel in the Netherlands are certainly the fishing mortality and the mortality caused by barriers. However, there are other sources of mortality that have not been quantified and may be substantial. The main sources are:

- Poaching (unreported landings or illegal removals).
- Yellow eel mortality in HPS's and pumping stations.
- Impact of (human-induced) viruses, parasites and pollution.
- Bycatch mortality of undersized eel. Most landings originate from fykes. Only a small amount of undersized by-catch is expected in this fisheries. However, also (~ 20%) of the catches are caught with a longline (Dutch: 'hoekwant'). Undersized bycatch and its survival of this gear is unknown.
- Catch and release mortality in recreational fisheries.
- Mortality by ship propeller strike. Sometimes, substantial numbers of damaged silver eels are found at the shores of the river Waal where heavy shipping traffic occurs. Also, in our telemetry studies, we still have a substantial part of silver eel disappearance during downstream migration in rivers and canals that cannot all be attributed to other mortality causes. Ship traffic impact is a potential candidate factor in these cases. So far, these observations are only anecdotal. There are, however, no research or dedicated studies available on the impact of ship traffic on silver eel. This can be tackled in meta-analyses of many telemetry studies combined, which is currently attempted by a cooperation of researchers with the European Tracking Network and EU-Cost action.

9.7 Recommendations

In this chapter an overview of (previous) recommendations for further adjustments to improve the quality of the assessment for the next evaluation is given.

9.7.1 Spatial Model

One of the most important sources of uncertainty in the spatial model are the catch efficiency and the habitat preference. The recommendation to study these effects has been made since the first evaluation report (Bierman et al., 2012), but no progress was made. The reason is that these assumptions are extremely difficult to assess. However, it is still needed to at least get some more knowledge of both uncertainties. In addition, especially for wider water bodies, assumptions of the distribution of eel over the water body may lead to unrealistic values.

9.7.2 Demographic Model

The assessment outcome is sensitive to the biological keys. To interpret present-day data or historical stock trends, a good index of recruitment, trends in sex-ratios, sex specific growth rates, natural mortalities and migration rates are required. Because eel recruitment has fallen sharply, it is probably unrealistic to assume that vital parameters have remained constant over time. At present the biological parameters are assumed to be constant over the entire time period. As more biological data is sampled, biological keys that are time or location specific should be analysed further. In addition, the natural mortality estimate is only a crude estimate and assumed to be constant over all lengths, ages and periods. A more realistic estimate of natural mortality should be investigated. Also, the calculation on the large lakes is based assumptions and research should be was done on a better understanding of eel distribution in the lakes using all available surveys.

9.7.3 Silver eel migration model

For the silver eel migration model as used for boezem and national waters (2nd and 3rd hierarchy), the division of silver eels that end up at a certain barrier site over the different migration routes is needed. For some sites, good data on route selection is available, e.g. at the HPS's in the Meuse (Winter et al., 2006 &, 2007; Jansen et al., 2007) and the large ship lock/sluice/pumping station complex at IJmuiden (Winter, 2011). However, on most sites, divisions of silver eel are mainly based on assumptions and extrapolations from research on other sites.

In addition, for some sites conditions have changed and updated estimations are needed to determine current mortality losses due to pumps or HPS's. Mortality depends on the eel length and discharge patterns. In future evaluations, additional studies and data need to be incorporated to determine overall HPS's mortality under the recently changed conditions.

9.7.4 Immigration of glass eel along barriers

To quantify and determine the effects of improved migratory opportunities for glass eel and how this results in increasing local yellow eel production and silver eel escapement, dedicated studies on population estimates of glass eel at barriers can be carried out, as has been done for the North Sea Canal catchment, with an estimated 10 million glass eels entering at IJmuiden in 2018 (Winter et al, 2020). When these approaches are carried out alongside the main immigration routes for glass eel into Dutch waters, such a quantification can be made.

9.7.5 Restocking

In the Netherlands restocking of glass eel and ongrown eel (eels that are grown in culture facilities for some time before being restocked, also called "pre-grown", ICES 2016) exists for decades. After the decline of glass eel availability, this commercial restocking lessened due to the increase of the price of the glass eel. After restocking became one of the management measures in the Dutch eel management plan (EMP), restocking was financed by public money, causing the amount of restocked glass eels and ongrown eel (elvers) to increase (from 818 kg in 2006-2008 to 3024 kg in 2018-2020, ICES 2020b). The restocking is commissioned by the ministry of Agriculture, Nature and Food quality (LNV) and is executed by the DUPAN foundation (www.DUPAN.nl), a foundation representing Eel processers, fish farmers and eel fishermen. Ongrown eels are usually bought from an aquaculture company in the Netherlands. The latest ICES advice (2020a) states:

> '*ICES notes that the restocking of eels, which is considered a management action in the EU regulation and in many eel management plans, is reliant on a glass eel fishery catch. Evidence shows that translocated and stocked eel can contribute to yellow and silver eel production in recipient waters, but information on the contribution to actual spawning is missing because of a general lack of knowledge of eel spawning. Internationally coordinated research is required to determine any net benefit of restocking on the overall population, including carrying capacity estimates of glass eel source estuaries, detailed mortality estimates at each step of the restocking process, and performance estimates of stocked vs. non-stocked eels. Estimation of the prospective net benefit should be carried out prior to any restocking activity, such as increasing silver eel escapement by restocking to attain stock recovery. Restocking should take place only where survival in silver eel escapement is high, and it should not be used as an alternative to reducing anthropogenic mortality. Where eel are translocated and stocked, measures should be implemented to evaluate their fate and their contribution to silver eel escapement. Such measures should include regionally-coordinated mass marking of eels to distinguish stocked eels from natural recruits in future scientific surveys.*'

Because of the stocking practices in the Netherlands, stocked eel are indirectly included in this assessment, because it is not possible to distinguish between eel originating from natural migration and stocked eels. It is therefore recommended to follow the ICES advice and conduct the marking of all stocked eels before release.

9.8 International "level playing field" stock indicators

As many other European countries (France, UK, Ireland) are using similar spatial models to estimate yellow eel standing stock and silver eel production, close international cooperation and collaboration will enhance the quality and uniformity of these models in the future. In addition, fundamental differences exist among the Netherlands, Belgium, Germany and the UK with respect to converting fisheries landings to silver eel production, selection of the reference period and correcting for glass eel stocking when calculating B_0. Germany, Belgium and the UK probably underestimated B_0 (ICES, 2010). Standardization of assessment methods is of utmost importance to ensure the recovery of the European eel stock and its sustainable exploitation.

9.9 Future of the eel advice (ICES WKFEA)

In this report, the estimated key stock indicators have been evaluated in relation to management targets/limits as formulated in the EC Eel Regulation, using the modified ICES precautionary diagram (*Chapter 7* and *Chapter 8*). However, the Advisory Committee (ACOM) of ICES is reluctant to advise on the status of the eel stock using these targets, because they have not been scientifically tested to ensure that they are precautionary and will lead to a recovery of the eel stock. ACOM therefore only analyses the level of recruitment compared to levels before the recruitment had dropped (ICES, 2020a). For this reason, the ICES workshop WKFEA "Future of the Eel Advice" was initiated (February 2020). The objective of WKFEA was to discuss the current advice framework, consider options for future assessments and draft a roadmap towards recommendations for an adapted or completely new advice framework on fishing opportunities and, potentially, other anthropogenic pressures on European eel. This has led to a roadmap describing the (ICES) workshops and (EU) projects aimed at developing a population model that would include the entire stock, which would lead to new management targets in a benchmark now proposed in 2026-2027 (ICES, 2021).

10 References

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Appendix A0 Retained catches and effort per region

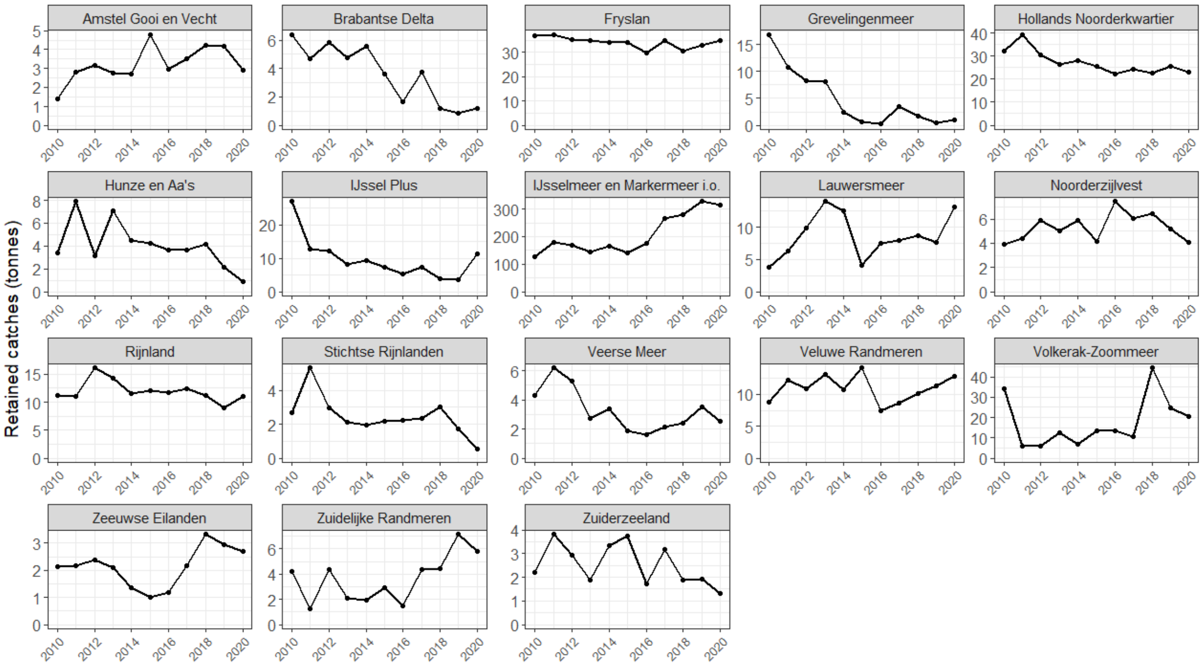

Figure A1. Eel retained catches (tonnes) per region and year (source: RVO).

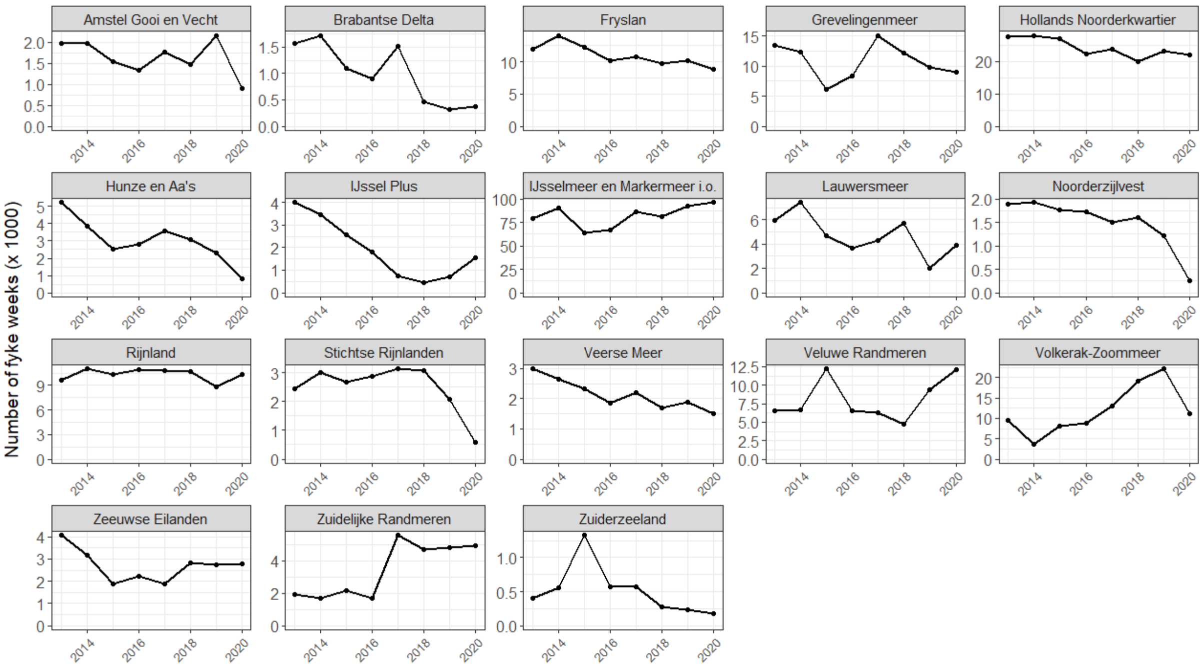

Figure A2. Eel fyke effort (number of fyke weeks, different types of fykes combined) per region and year (source: RVO).). Effort is self reported by the fishermen.

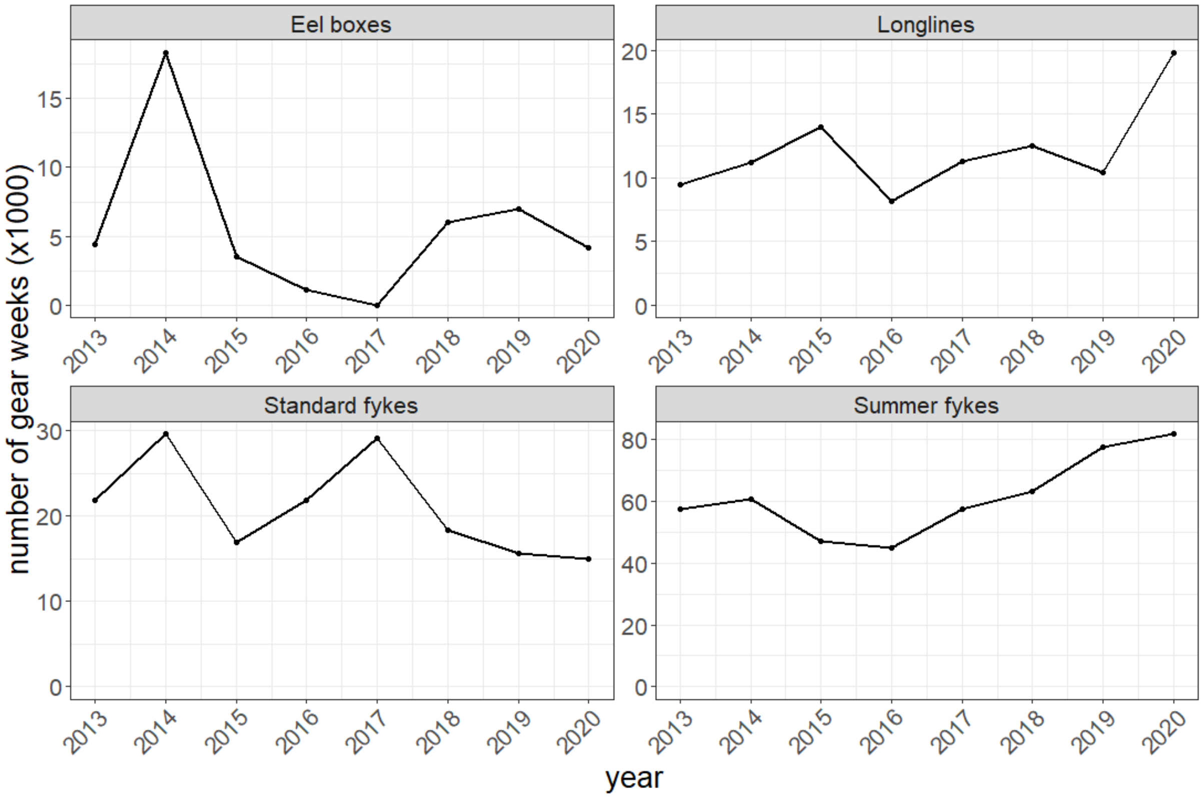

Figure A3. Eel effort per gear type in lakes IJsselmeer and Markermeer per year (source: RVO). Effort is self reported by the fishermen. Some corrections have taken place from previous graphs.

Appendix A1: Water Framework Directive (WFD) water types

Table A1. Water body types defined within the WFD in the Netherlands that were taken into account in this study of standing stocks in regionally managed waters.

Water type code	Description
M1a	Buffered ditches
M2	Weakly buffered ditches
M3	Buffered canals (regional)
M6a/b	Large shallow canals with/without shipping
M7a/b	Large deep canals with/without shipping
M8	Buffered peatland ditches ("laagveensloten")
M10	Peatland canals ("laagveen kanalen")
M14	Shallow, relatively large, buffered lakes
M20	Deep, relatively large, buffered lakes
M23	Shallow, large, calcium-rich lakes
M27	Shallow, relatively large, peatland lakes ("laagveenplassen")
M30	Weakly brackish waters (0.3 - 3 g Cl/l)
R4	Permanent, slow flowing, upper part stream on sandy riverbed
R5	Permanent, slow flowing, middle- or lower part stream on sandy riverbed
R6	Slow flowing small river on sandy/clay riverbed
R7	Slow flowing river/side stream on sandy/clay riverbed
R8	Fresh tidal waters on sandy/clay riverbed
R12	Slow flowing, middle- or lower part stream on peat riverbed
R13	Fast flowing, upper part stream on sandy riverbed
R14	Fast flowing, middle- or lower part stream on sandy riverbed
R15	Fast flowing small river on siliceous riverbed
R17	Fast flowing, upper part stream on calcium rich riverbed
R18	Fast flowing, middle- or lower part stream on calcium rich riverbed

Appendix A2: Eel biomass per water type per period

Table A2. Density and biomass of eel (≥30 cm) based on sampling data of WFD water bodies assessed per water type per period following scenario 2. Note that non-WFD water bodies are not included.

		2006 - 2008		2009 - 2014		2015 - 2020	
WFD- water	**Total area (ha)**	**Density (kg/ha)**	**Biomass (tonnes)**	**Density (kg/ha)**	**Biomass (tonnes)**	**Density (kg/ha)**	**Biomass (tonnes)**
M1a	156	0.0	0.0	1.8	1.2	0.4	0.3
M2	10	3.7	0.2	5.5	0.2	4.3	0.2
M3	3,324	2.1	21.8	3.7	38.4	1.2	12.7
M6a	603	1.4	2.4	5.8	10.3	6.7	12.0
M6b	1,780	3.4	17.9	6.9	35.9	4.8	24.9
M7a	13	3.7	0.1	9.7	0.4	6.9	0.3
M7b	3,435	5.2	48.2	7.2	67.1	8.2	76.2
M8	1,148	0.7	2.4	0.8	2.5	0.1	0.2
M10	1,362	0.1	0.3	8	39.2	1.8	8.9
M14	20,902	10.0	531.1	33.4	1,776.9	17.0	907.2
M20	4,444	7.3	82.2	10.0	112.5	9.7	109.5
M23	90	3.7	0.9	0.0	0.0	6.9	1.7
M27	22,738	4.4	252.6	19.1	1,091.6	6.1	348.6
M30	8,182	3.7	76	5	104.0	1.2	24.6
R4	73	4.8	1.7	2.2	0.8	0.5	0.2
R5	1,221	2.4	10.5	3.4	15.2	1.6	7.3
R6	3,414	8.6	77.5	12.7	114.2	12.7	114.9
R7	2,272	3.7	21.2	37.9	218.4	7.8	45.1
R8	20	3.7	0.2	4.7	0.3	5.3	0.3
R12	65	2.9	0.7	17.4	4.1	2.7	0.6
R13	4	3.7	0.1	9.7	0.2	6.9	0.2
R14	16	3.7	0.2	10.5	0.6	26.9	1.5
R15	37	3.7	0.4	9.7	1.1	6.9	0.8
R17	7	3.7	0.1	9.7	0.4	6.9	0.3
R18	52	2.0	0.4	8.5	1.6	19.4	3.7
Total	**75,368**		1,149.1		3,637.1		1,702.2

Appendix A3: Eel biomass per water board

Density and biomass estimates were done in the same way as was done per water type (paragraph 4.2.3), for each water board in the periods 2006–2008, 2009–2014 and 2015–2020. Both density (kg/ha) and biomass (tonnes) were estimated. Biomass estimations from the most recent period (2015–2020) were for most water boards lower than those of the period 2009–2014. Only a few of the water boards had a higher estimated biomass and the estimations for the remaining were more or less similar.

Table A3. Density and biomass of eel (≥30 cm) based on sampling data of WFD water bodies assessed per water board per period following scenario 2. Note that non-WFD water bodies are not included.

		2006 - 2008		2009 - 2014		2015 - 2020	
Water board	**Total area (ha)**	**Density (kg/ha)**	**Biomass (tonnes)**	**Density (kg/ha)**	**Biomass (tonnes)**	**Density (kg/ha)**	**Biomass (tonnes)**
Aa en Maas	470	4.8	7.2	1.4	2.2	0.6	0.9
Brabantse Delta	2,358	15.7	96.2	3.8	23.2	3.7	22.6
De Dommel	391	1.0	1.3	1.7	2.1	1.4	1.8
Drents Overijsselse Delta	13,810	4.0	141.0	9.1	319.1	4.3	149.5
Hollandse Delta	925	4.8	11.7	10.1	24.6	6.4	15.4
Hoogh. Amstel, Gooi en Vecht	8,623	9.8	214.5	6.2	134.6	8.4	182.7
Hoogh. De Stichtse Rijnlanden	225	8.4	65.9	4.7	3.3	2.5	1.8
Hoogh. Hollands Noorderkwartier	4,714	3.2	43.4	8.1	110.5	4.2	57.3
Hoogh. van Delfland	298	4.8	4.5	9.7	8.9	4.8	4.4
Hoogh. van Rijnland	4,679	4.8	58.4	16.4	199.8	7.5	91.6
Hoogh. van Schieland en de Kr.	1,084	3.7	11.6	9.6	30.4	13.2	41.7
Hunze en Aa's	2,250	4.1	25.3	8	49.8	4.9	30.5
Noorderzijlvest	3,017	4.8	37.7	10.6	83.5	3.2	25.4
Rijn en IJssel	518	1.8	3.0	0.4	0.8	0.3	0.5
Rivierenland	799	6.0	15.0	4.1	10.2	6.7	16.7
Scheldestromen	10	4.8	0.1	10.1	0.3	6.4	0.2
Vechtstromen	2,816	7.1	53.6	9.9	74.5	9.7	73.5
Waterschap Limburg	289	8.3	8.5	9.4	9.7	15.5	16.0
Waterschap Vallei & Veluwe	414	0.8	1.1	2.5	3.3	2.5	3.2
Wetterskip Fryslân	16,193	7.8	323.5	37.3	1553.2	12.3	514.2
Zuiderzeeland	8,153	4.8	101.3	45.2	953.4	21.3	449.2
Total			**1,224.8**		**3,597.4**		**1,699.1**

Appendix B1: Details of the demographic model

Model update

Several improvements were made in the demographic model. The changes to this compared to the model used in the previous eel assessment (van de Wolfshaar et al., 2018) are:

- The biological keys and FYMA survey data were updated with the newest information up to 2020.
- The length-class based fit between model and data that was previously used has been changed to an age-class based fit to allow for a better comparison between the model and survey data. In previous versions of the eel assessment, the abundances per age class predicted by the model were transformed to abundances per length class such that the model predictions could be compared to the abundances per length class observed in the FYMA survey data. In the current assessment, the abundances per length of the FYMA survey data are converted to abundances per age class using sex-specific von Bertalanffy growth curves and a general length – sex ratio relationship.
- Some biological keys have been changed (see *Chapter 2*):
  - Eel growth is now based on a von Bertalanffy growth curve per sex.
  - Maturation at length, sex ratio at length and the sex ratio of the recruits are now based on *GAM* analyses.
- The age-specific model parameters (maturation, fisheries selectivity), were previously based on the initial length of the age class and are now based on the length at the mid-age of the age class. We consider this mid-length more representative for the characteristics of the individuals in the age class then the length at the start of the age class.
- The moment of comparison of the model with the survey data was moved from April to September to better fit the timing of the collection of the survey data.
- The model is fit to survey data of lakes Ijsselmeer and Markermeer together, due to scarcity of data in Lake Markermeer in some of the years.
- Different periods for which a single *F* estimate was calculated were changed such that the periods better represented the changes in eel fisheries management (see *Chapter 4*)
- The selectivity of the survey was previously assumed to differ slightly between length classes. We now assume an equal selectivity of the survey for all age classes, because there is no information on the selectivity of the survey available.
- Small changes were made to the selectivity of the commercial fisheries (see *Chapter 4*).

Model parameters

Table B1 Life history parameters; the length at the start of the age class corresponds to the length in September of each year.

Years in IJsselmeer/ Markermeer	Length per age class				Maturation probability		Fisheries selectivity	
	At start age class (mm)		At mid age class (mm)					
	Female	Male	Female	Male	Female (M_{fi})	Male (M_{mi})	Female (z_{fi})	Male (z_{mi})
0.5	86	86	127	128	0	0	0	0
1.5	165	164	202	196	0	0	0	0
2.5	237	222	270	246	0	0	0.34	0
3.5	301	266	330	283	0.001	0	1	0.59
4.5	358	298	384	310	0.003	0.010	1	1
5.5	409	321	433	331	0.010	0.053	1	1
6.5	455	339	476	346	0.028	0.177	1	1
7.5	496	352	514	357	0.048	0.270	1	1
8.5	532	361	549	365	0.086	0.369	1	1
9.5	565	369	580	371	0.124	0.369	1	1
10.5	594	374	607	376	0.167	0.461	1	1
11.5	620	378	632	379	0.198	0.461	1	1
12.5	643	381	654	382	0.229	0.461	1	1
13.5	664	383	674	384	0.260	0.461	1	1
14.5	683	385	692	385	0.290	0.538	1	1
15.5	700	386	707	386	0.320	0.538	1	1
16.5	715	387	721	387	0.335	0.538	1	1
17.5	728	387	734	388	0.350	0.538	1	1
18.5	740	388	745	388	0.378	0.538	1	1
19.5	751	388	755	388	0.392	0.538	1	1
20.5	760	388	764	388	0.392	0.538	1	1

Table B2 Annual glass eel index based on lift net survey (survey density/haul) and female ratio (see Paragraph 4.4) of recruits based on the segmented regression model (Appendix B2)

Year	Glass eel index	Female ratio $\rho(t)$
1968	32.9	0.67
1969	27.1	0.67
1970	48.1	0.67
1971	36.1	0.67
1972	55.0	0.67
1973	18.8	0.67
1974	63.0	0.67
1975	84.3	0.67
1976	51.4	0.18
1977	75.0	0.67
1978	73.6	0.95
1979	87.7	0.98
1980	59.0	0.97
1981	50.4	0.97
1982	29.4	0.96
1983	14.7	0.95
1984	31.6	0.94
1985	11.2	0.93
1986	11.4	0.92
1987	6.2	0.91
1988	7.0	0.89
1989	4.8	0.87
1990	4.9	0.85
1991	1.8	0.82
1992	5.2	0.79
1993	3.5	0.76
1994	5.4	0.73

Year	Glass eel	Female ratio $\rho(t)$
1995	11.1	0.69
1996	12.5	0.65
1997	12.6	0.61
1998	2.46	0.56
1999	3.7	0.52
2000	2.8	0.47
2001	0.6	0.43
2002	1.2	0.38
2003	1.3	0.34
2004	2.1	0.30
2005	1.6	0.29
2006	0.6	0.33
2007	1.2	0.37
2008	0.5	0.42
2009	0.9	0.46
2010	2.2	0.51
2011	1.1	0.56
2012	1.0	0.60
2013	4.9	0.64
2014	4.6	0.69
2015	0.2	0.73
2016	1.0	0.76
2017	2.3	0.79
2018	1.3	0.67
2019	1.2	0.67
2020	1.0	0.67

Appendix B2: Recruitment sex ratio for lakes IJsselmeer and Markermeer

Whether an eel becomes a female or a male depends on external factors, such as food availability or (intraspecific) competition. This means that the sex ratio can change over time if, for example, the food levels or the eel density change.

To parametrize the demographic model for the lakes IJsselmeer and Markermeer, an estimate of the initial sex ratio of the recruitment is required. For this reason, a statistical analysis was conducted to determine the initial sex ratio of eel in these lakes. The analysis is done with market sampling data (*Chapter 2*), as this was the only available data in the Netherlands where sex is determined. As there is also asymmetric growth between males and females, especially for the larger eel, (fishing) mortality is expected to influence the sex ratio of the larger individuals. Therefore, in order to parameterize the demographic model, the sex ratio needs to be determined for eel as young as possible. However, the sex is difficult to assess for small individuals (or the eel has not determined its own sex yet) and almost no eels from sizes below 28 cm are caught in the market sampling. Therefore, eels at an estimated age of 2 years are selected for the analysis.

Year class estimate

Sex ratio needed to be estimated by year class. To determine the year class, the age samples across years from the market samples (see *Chapter 2*) were used to construct a fixed sex-specific age-length key using only the age readings from eel in lake IJsselmeer and Markermeer, assuming that for each sex, the growth rate does not change across the years.
The first step is to estimate the age for the fish for which we have length measurements. In the analysis, we first applied von Bertalanffy growth model (VBGF) model to the fishes with both age and length measurements. The model was applied to females and males separately. Only the end measurements were used. The von Bertalanffy Growth Function (VBGF) estimates the mean length at a given age. We then applied the inverse transformation of the estimated von Bertalanffy length-age function to get the estimated age, assuming the length measurement is the mean length. Von Bertalanffy growth, from Beverton and Holt (1957):

$$L_t = L_\infty[1 - e^{-K(t-t_0)}]$$

where L_t is the expected or mean length at age t, L_∞ is the asymptotic mean length, K is a measure of the exponential rate at which L_t approaches L_∞ (Schnute and Fournier, 1980) and t_0 is the theoretical age at which L_t would be zero. The estimated parameters were (females: *Linf* = 85.3, *K* = 0.13, *t0* = -1.00, males: *Linf* = 45.7, *K* = 0.16, t0 = -4.02). Inverse VBGF:

$$t = \frac{-1}{K}\ln(1 - \frac{L_t}{L_\infty}) + t_0$$

The age of the individual fish with length measurements was estimated, after applying the inverse VBGF function. As the length samples were assumed to be representative of the population in the lakes, this would give an approximated age distribution which is representative of the population. A young age (i.e. age 2) was selected for further analysis. The age was estimated using the inverse VBGF and translated to year class. To get an integer number, the age was rounded to the nearest integer.

Proportion of female at recruitment

The proportion of females was estimated for each yearclass at age 2 (from 1976-2017).
Four statistical models were applied:

1) *GLM* using year as a continuous predictor. Both *GLM's* were estimated through maximum likelihood estimation.
2) Segmented regression with one breakpoint, using year as a continuous predictor. The model coefficients were estimated through maximum likelihood estimation with an interactive procedure of estimating the best breakpoint.
3) Segmented regression similar to model 2, but with two breakpoints.

4) *GAM* using thin plate regression spline (i.e. smooth function) with year as the predictors. The dimension of basis k is set as 6 (number of basis function=k-1=5). We limit a relatively low k to avoid over-fitting. Smoothing parameters were estimated through generalized Cross Validation (GCV) criterion.

In all 4 models, the response variable is modeled as a Bernoulli distribution, and a logit link function links the linear predicter to the mean of the Bernoulli distribution. Model 3, the segmented regression with two breakpoints, had the lowest AIC (Akaike Information Criterion) and was selected for use in the demographic model (*Chapter 3*). The results show that a large number of males from yearclass 1976-1977 is causing a very low female proportion for these 2 cohorts, and a substantial increase of the proportion of females for the years after. These two first years are a bit strange, and further data analysis is needed to determine if they are outliers due to, for example, poor data collection or if they actually represent the true value. This is also the case for the yearclasses 1978-1980 where a 100% female sex ratio was observed at the estimated age 2. Yearclass 1983, 1985, 2000 and 2001 are missing, due to the missing gender samples in 1985, 1987 and 2002-2003. There are a lot of approximations and pre-processings during the calculation: a time-invariant age length key (ALK) is assumed and it is estimated from very few age samples. However, it is the result of the best available information at present. Additional age readings and data analyses are needed to confirm the result presented here.

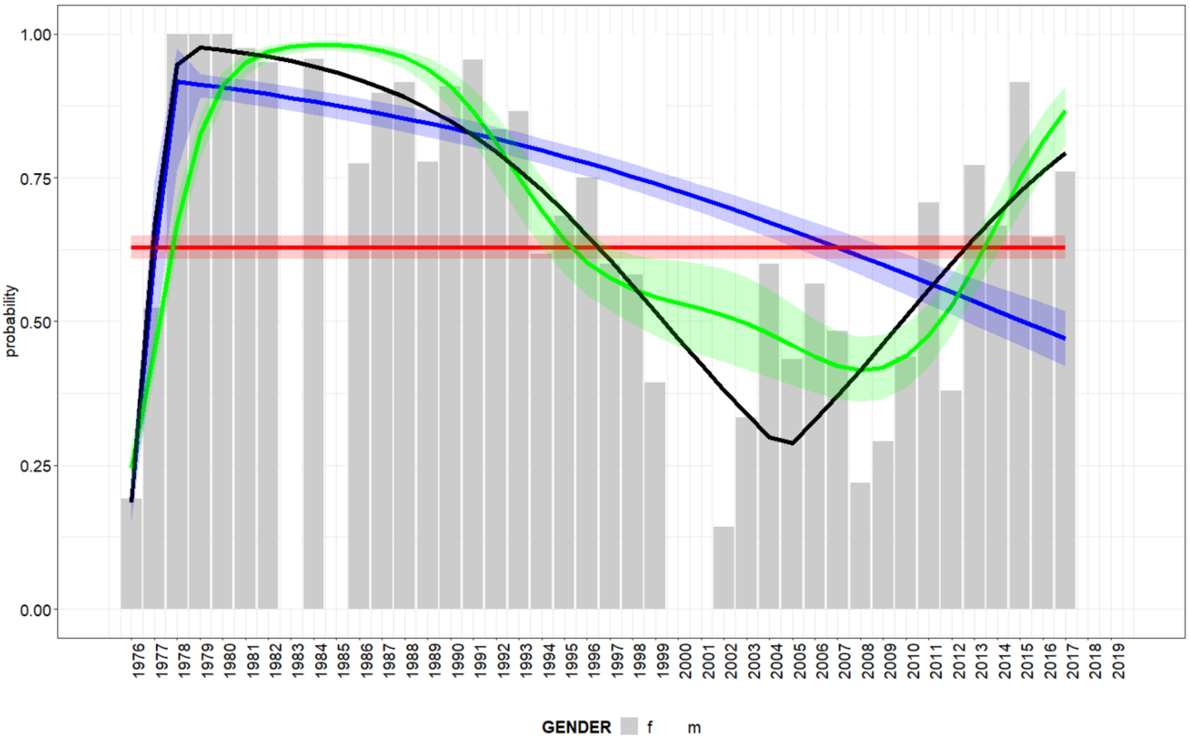

Figure B1. estimated proportion of female by yearclass (1976-2017) at age 2. Black: segmented regression with 2 breakpoints (selected option); Red: one value across all years; blue: segmented linear regression with one breakpoint; green: spline. Raw annual estimates are plotted as grey bars.

Appendix C1: Overview eel mortality pump stations with a propeller pump

Table C1: Overview of eel mortality when passing through pumping stations with a propeller pump (axial water flow). * Underestimation as seemingly undamaged eels did reveal internal damage after dissection which could result in delayed mortality.

	Pump description	Capacity (m^3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
axiaalpump	Gesloten schroefpomp	60	0.8	355	Kortenhoef	11	32			Vriese et al., 2010
	Gesloten schroefpomp FFI	81	1	333	FFI	25	0			Vriese, 2009
	Gesloten schroefpomp	1,500		50	J.L. Hoogland	77	5	5		Kruitwagen & Klinge, 2010a
	Gesloten schroefpomp	2,500	0.6	80	A.F. Stroink	10	0	30		Kroes et al., 2006
	Open schroefpomp	24	0.98		Thabor	21	38			Vriese et al., 2010
	Open schroefpomp	60	2.7	500	Stenensluisvaart	?	100			Germonpré et al., 1994
	Open schroefpomp	76			Offerhaus	10	0			Vriese, 2010
	Open schroefpomp	200	0.6	165	Den Deel	?	8	30		Riemersma & Wintersmans,
	Bulbpomp Nijhuis	3,000	variable	64	IJmuiden	25	41*	41*		Kruitwagen & Klinge, 2008a
	Schroefpomp	30	1.35	900	Kralingseplas	19	100			Kruitwagen & Klinge, 2010b
	Schroefpomp	400	1,34-4,64		Krimpenerwaard	19	100			Kruitwagen & Klinge, 2010b
	Schroefpomp	184	1.05	185	De Waker	69	1.4			VisserijServiceNederland,2010
	Schroefpomp	2,400			Zaangemaal	65	0			VisserijServiceNederland,2010
	Schroefpomp	180	1.07	180	Meldijk	30	33			Kroon & van Wijk, 2012
	propeller	60	2.7	500	Woumen (BE)	?	100			Germonpré et al., 1994
	propeller	100		480	Avrijevaart/Burgraven (BE)	39	98			INBO
	BVOP	255	5.4	360	Lijnden	*2*				
	Gesl. Schroefp. (compact)	90	2.7	364	HZ Polder	*6*				Vriese et al., 2010
	Gesl. Schroefp. (compact)	105	2.2	291	Berkel	*5*				Vriese et al., 2010
	Gesl. Schroefp. (compact)	135	0,5-1	307	Antlia	*6*				Vriese et al., 2010
	Gesloten schroefpomp	26	3.08		Makkemermar	*2*				Vriese et al., 2010
	Gesloten schroefpomp	42	2,4 - 3,1		Aalkeet buitenpolder	*1*				Kruitwagen & Klinge 2010c
	Open schroefpomp	40	1.67	580	Nijverheid	*2*				Vriese et al., 2010
	Open schroefpomp	120	0.1		Tilburg	*9*				Vriese et al., 2010
	Gesloten schroefpomp FFI				Kralingseplas	*3*				Waning et al., 2012
	Open schroefpomp	90			Offerhaus	*2*				Kroes & de Boer, 2013
	schroefpomp	120	340	340	Balgdijk	*5*				Kroon & van Wijk, 2012
					Pooled studies with n <10		32.6			

Table C2 : Overview of eel mortality when passing through pumping stations with a propeller-centrifugal pump (axial-radial water flow).

	Pump description	**Capacity (m3/min)**	**Height (m)**	**Rotation (rpm)**	**Name**	**n**	**dead (%)**	**damaged (%)**		**Reference**
semi-axiaal pump	Schroefcentrifugaalpomp	170	1.52		Tonnekreek	34	0			Vriese et al., 2010
	Hidrostal		10	890-1,200		2,300	0	3		Patrick & McKinley 1987
	Schroefcentrifugaalpomp	350	2.8	115	Schilthuis	27	22			Vriese et al., 2010
	BEVERON	505	2,4	143	Schoute (natuurlijke doortrek)	36	0			Kruitwagen & Klinge, 2008b
	BEVERON	525	5.4	200	Lijnden	*6*				
	Hidrostal	21	3.6	577	Ypenburg	*8*				Vriese et al., 2010
	Hidrostal	42.5	3.5	552	Wogmeer	*8*				Vriese et al., 2010
	Schroefcentrifugaalpomp	300	4.4		Leemans	*4*				Kroon & van Wijk, 2013
	Schroefcentrifugaalpomp	250	2-5,5	165	Abraham Kroes (Ringvaart gemaal)	*8*				Kruitwagen & Klinge, 2010b
	VOPO met schroefomdraaiing	25	0.15	1,000	De Zllk	*2*				Vriese et al., 2010
	Schroefcentrifugaalpomp	85		416	Willem-Alexander	*1*				Vriese et al., 2010
	Schroefcentrifugaalpomp	24	1.15		B.B. Polder	*2*				Vriese et al., 2010
	Schroefcentrifugaalpomp	22	1.15	735	Meerweg	*9*				Klinge, 2008
					Pooled studies with n <10		39.6			

Table C3 : Overview of eel mortality when passing through pumping stations with a centrifugal pump (radial water flow).

	Pump description	Capacity (m3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
radial pump	Centrifugaalpomp	38	3.5	368	Duifpolder	12	0			Vriese et al., 2010
	Centrifugaalpomp	60	5	49	Elektriek-Zuid	?	1.4	1.4		Germonpré et al., 1994
	Centrifugaalpomp	400	0.9	205	Boreel	49	49			Vriese et al., 2010
	Centrifugaalpomp	1,080	1.7	59	Katwijk	56	0			Kruitwagen & Klinge, 2007
	Centrifugaalpomp	325	3.5	168	Grootslag	438	0			Kroon & van Wijk, 2013
	Centrifugaalpomp	160	0.3		JC de Leeuw	*5*				Kroon & van Wijk, 2013
	Centrifugaalpomp	690	1.7	70	Gouda (natuurlijk)	*2*				Kruitwagen & Klinge, 2008c
	Centrifugaalpomp	690	1.7	70	Gouda (gedwongen)	*4*				Kruitwagen & Klinge, 2008c
	Centrifugaalpomp	28	0,55-1,05	320	Hoekpolder	*1*				Kruitwagen & Klinge, 2010c
					Pooled studies with n <10		16.7			

Table C4 : Overview of eel mortality when passing through pumping stations with an Archimedes’ screw.

	Pump description	Capacity (m3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
Archimedes’ screw	Turbinevijzels				Vijzel Bielefeld	?	0			Spah, 2001
	Buisvijzel FFI	0.6	1	57	FFI (gedwongen blootstelling)	23	0			Vriese, 2009
	Vijzel	30	2.9	39	Sint-Karelsmolen	?	4	10		Germonpré et al., 1994
	Vijzel	35	3.6	37	De Seine, Vlaanderen	?	0	37		Denayer & Belpaire, 1992
	Spaans Babcock	500	2.2	17	Overwaard	43	2			Vriese et al., 2010
	De Wit vijzel	660	0.3	22	Halfweg (natuurlijke doortrek)	24	0			Kruitwagen & Klinge, 2008c
	Buisvijzel (Landustrie Sneek BV)	40	2.7	39.1	Ennemaborgh	101	8			Vis et al., 2013
	Buisvijzel (Landustrie Sneek BV)	23	2.7	23.8	Ennemaborgh	*112*	3			Vis et al., 2013
	Vijzel	335	0.35		Kolhoorn	16	0			Kroon & van Wijk, 2013
	Vijzel	350	1.14		Kadoelen	59	8			VisserijServiceNederland, 2010
	Vijzel			23-31		160	0	0.6		Kibel, 2008
	Vijzel	100		25	Isabella	48	13.5			INBO
	Vijzel	200		21	Isabella	131	14.5			INBO
	Vijzel	90	0.64		Overtoom	*7*				VisserijServiceNederland, 2010
	Vijzel	43	1.25		Bergermeer	*3*				VisserijServiceNederland, 2010
	Vijzel	660	0.3	22	Halfweg (natuurlijke doortrek)	*5*				Kruitwagen & Klinge, 2008c
	Buisvijzel FFI	32			Hoekpolder	*2*				Wanink et al., 2012
	Vijzel				Schalsum	*2*				Koopmans, 2013
	Vijzel	23	0.73		Sudhoeke	*9*				Vriese et al., 2010
					Pooled studies with n <10	28	3.6			

Appendix C2: Barrier assessment list Boezem and National waters

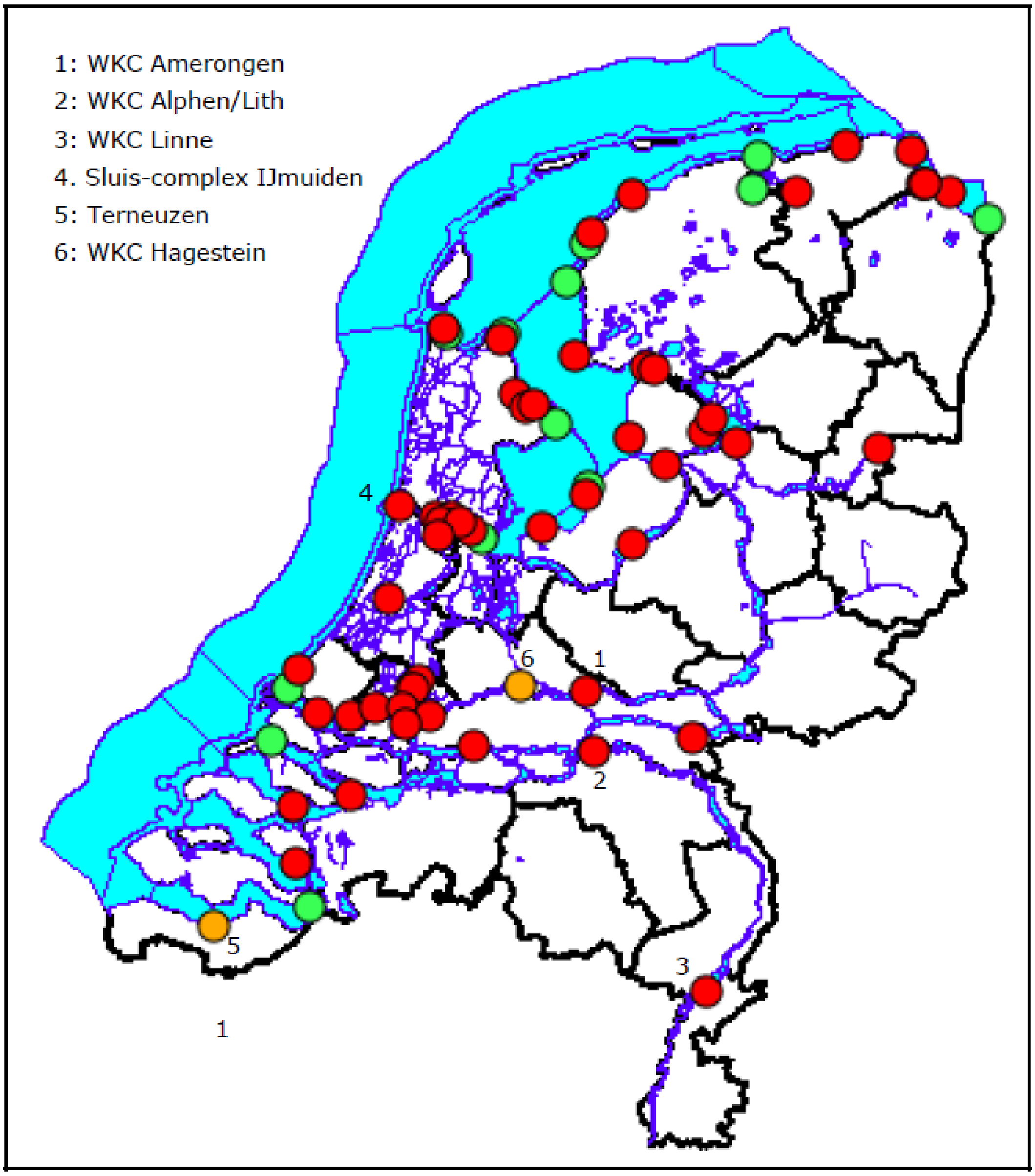

Figure appendix C2: All main 2nd and 3rd hierarchy barriers, where green indicates no mortality and red indicates a mortality estimate; the HPS's in the rivers: 1: WKC Amerongen, 2: WKC Alphen and 3: WKC Linne and 4: the barrier-complexes at IJmuiden are numbered. Orange barriers are not taken into account because 5: the WKC Hagestein has not been in operation since 2005, and no data is availalbe on 6: the canal from Belgium to Terneuzen.

Table C5. Overview of the most important barriers, their characteristics and their estimated mortalities (based on Winter et al. 2013a, 2013b and updated for 2018-2020).

Waterboard	Site (potential barrier)	Barrier type	from	to	Potential silver eel starting biomass (ton)	Mortality (%) best guess	Potential silver eel losses (ton)		Losses per site (%)*	
							min	max	min	max
noorderzijlvest	Spijksterpompen	Gema	B	Z	0.85	0.30	0.25	0.25	30	30
noorderzijlvest	Noordpolderzijl	Gema	B	Z	0.56	0.30	0.17	0.17	30	30
noorderzijlvest	Waterwolf Electra	Gema+Keer	B	R	7.05	0.30	0.59	1.95	8	28
noorderzijlvest	De Drie Delfzijlen	Gema+Spui	B	Z	2.11	0.30	0.18	0.58	8	28
noorderzijlvest	Lauwerssluizen	Spui+Sche	B	Z	83.86	0.00	0.00	0.00	0	0
wetterskip	Roptazijl	Gema	B	Z	5.50	0.50	2.75	2.75	50	50
wetterskip	Zwarte Haan	Gema	B	Z	5.50	0.50	2.75	2.75	50	50
wetterskip	Lemmer (Wouda)	Gema + Sche	B	Z	1.83	0.25	0.25	0.44	14	24
wetterskip	Stavoren	Gema + Sche	B	R	36.64	0.06	1.21	2.09	3	6
wetterskip	Ezumazijl	Gema + Sche	B	R	5.50	0.50	2.25	2.69	41	49
wetterskip	Harlingen	Spui+Sche	B	Z	54.96	0.00	0.00	0.00	0	0
wetterskip	Dokkumer Nieuwe Zijlen	Spui+Sche	B	R	73.29	0.00	0.00	0.00	0	0
hunze en aa	Duurswolde	Gema+Spui	B	Z	1.61	0.50	0.22	0.74	14	46
hunze en aa	Termunterzijl	Gema+Spui+Sche	B	Z	1.45	0.30	0.40	0.43	28	30
hunze en aa	Nieuw Statenzijl	Spui+Sche	B	Z	6.90	0.00	0.00	0.00	0	0
hunze en aa	Delfzijl	Spui+Sche	B	Z	6.10	0.00	0.00	0.00	0	0
reest en wieden	Stroink	Gema	B	R	15.43	0.11	1.70	1.70	11	11
reest en wieden	Zenemuden	Gema+Keer+Sche	B	R	5.14	0.50	1.35	1.41	26	28
velt en vecht	Haandrik	WKC+Stuw+Vist	B	R	23.58	0.17	4.01	4.01	17	17
amstel gooi en vecht	De Ruiter	Gema + Sche	B	R	5.60	0.25	1.15	1.37	21	25
amstel gooi en vecht	Mijndense Sluis	Gema + Sche	B	R	4.55	0.10	0.37	0.45	8	10
amstel gooi en vecht	Spiegelpolder	Gema + Sche	B	R	2.10	0.25	0.43	0.51	21	25
HHNK	Kadoelen	Gema	B	R	0.93	0.08	0.07	0.07	8	8
HHNK	De Waker	Gema	B	R	0.31	0.02	0.01	0.01	2	2
HHNK	Leemans	Gema	B	Z	2.49	0.10	0.25	0.25	10	10
HHNK	Lely	Gema	B	Z	0.93	0.25	0.23	0.23	25	25
HHNK	Vier Koggen	Gema	B	R	2.18	0.10	0.22	0.22	10	10
HHNK	Grootslag	Gema	B	R	1.56	0.02	0.03	0.03	2	2
HHNK	Zaangemaal	Gema + Sche	B	R	3.74	0.01	0.01	0.03	0	1
HHNK	Overtoom	Gema + Sche	B	R	0.03	0.04	0.00	0.00	1	4
HHNK	Helsdeur	Gema+Spui+Sche	B	Z	9.97	0.30	0.84	2.75	8	28
HHNK	Schermersluis	Sche	B	R	0.31	0.00	0.06	0.06	20	20
HHNK	Oostoever	Spui	B	Z	3.74	0.00	0.00	0.00	0	0
rijnland	Katwijk	Gema	B	Z	18.74	0.01	0.19	0.19	1	1
rijnland	Halfweg	Gema	B	R	6.72	0.04	0.27	0.27	4	4
rijnland	Gouda	Gema	B	R	3.54	0.10	0.35	0.35	10	10
rijnland	Leeghwater	Gema	B	R	5.66	0.30	1.70	1.70	30	30
rijnland	Spaarndam	Gema + Sche	B	R	6.36	0.01	0.02	0.06	0	1
Delfland	Schoute	Gema	B	Z	1.16	0.30	0.35	0.35	30	30
Delfland	Zaayer	Gema	B	R	0.09	0.02	0.00	0.00	2	2
Delfland	Westland	Gema	B	R	0.39	0.10	0.04	0.04	10	10
Delfland	Schiegemaal	Gema	B	R	0.39	0.10	0.04	0.04	10	10
Delfland	v.d. Burg	Gema	B	Z	1.16	0.30	0.35	0.35	30	30
Delfland	Parksluizen	Gema + Sche	B	R	0.96	0.25	0.07	0.22	7	23
HHSK	Schilthuis	Gema	B	R	5.07	0.30	1.52	1.52	30	30
HHSK	Verdoold	Gema	B	R	3.67	0.11	0.40	0.40	11	11
HHSK	Johan Veurink	Gema	B	R	1.75	0.50	0.87	0.87	50	50
HHSK	Krimperwaard	Gema	B	Z	1.40	0.30	0.42	0.42	30	30
HHSK	Abraham Kroes	Gema + Sche	B	R	5.42	0.30	0.46	1.50	8	28
rivierenland	J.U. Smit	Gema	B	R	1.58	0.04	0.06	0.06	4	4
rivierenland	Altena	Gema	B	R	1.10	0.50	0.55	0.55	50	50
rivierenland	Hollands-Duits	Gema	B	R	1.10	0.25	0.28	0.28	25	25
zuiderzeeland	Vissering	Gema + Sche	B	R	53.86	0.25	7.41	12.79	14	24
zuiderzeeland	Buma	Gema + Sche	B	R	41.19	0.25	5.66	9.78	14	24
zuiderzeeland	Smeenge	Gema + Sche	B	R	28.51	0.50	7.84	13.54	28	48
zuiderzeeland	Wortman	Gema + Sche	B	R	47.52	0.25	6.53	11.29	14	24
zuiderzeeland	De Blocq van Kuffeler	Gema + Sche	B	R	79.21	0.25	10.89	18.81	14	24
zuiderzeeland	Lovink	Gema + Sche	B	R	28.51	0.25	3.92	6.77	14	24
zuiderzeeland	Colijn	Gema + Sche	B	R	38.02	0.12	2.46	4.25	6	11
R	Sluizen-complex IJmuiden	Gema+Spui+Sche	R	Z	121.90	0.15	18.28	31.69	15	26
R	Krammersluizen	Sche	B	Z	7.30	0.00	3.65	3.65	50	50
R	Bergse Diep Sluis	Sche	R	Z	0.59	0.00	0.29	0.29	50	50
R	Terneuzen	Sche	R	Z		0.00	0.00	0.00	0	0
R	Volkeraksluizen	Sche	R	Z	7.30	0.00	3.65	3.65	50	50
R	Bathse spuisluis	Spui	R	Z	95.67	0.00	0.00	0.00	0	0
R	Krabbersgat-sluizen	Spui+Sche	R	Z		0.00	0.00	0.00	0	0
R	Houtrib-sluizen	Spui+Sche	R	Z		0.00	0.00	0.00	0	0
R	Haringvliet-sluizen	Spui+Sche	R	Z	265.68	0.00	0.00	0.00	0	0
R	Aflsuitdijk Kornwerderzand	Spui+Sche	R	Z	228.80	0.00	0.00	0.00	0	0
R	Afsluitdijk Den Oever	Spui+Sche	R	Z	228.80	0.00	0.00	0.00	0	0
R	Oranjesluizen	Spui+Sche+Vist	R	Z	180.99	0.00	0.00	0.00	0	0
R	Nieuwe Waterweg		R	Z	260.80	0.00	0.00	0.00	0	0

* taking alternative routes and blockage into account (cf. Winter et al. 2013)

**Gemaal=Pumping Station; Sche=Ship Lock; Spui=Discharge Sluice; Vist=Fishway; Stuw=Wier; Keer=Protection Sluice

B=Boezem waters; R=National waters; Z=Sea

Appendix C3: Validation of the assessment method for the North Sea Canal

In 2016-2018, extensive acoustic telemetry studies and mark-recapture studies were carried out for the North Sea Canal region, including lake Markermeer (Winter et al., 2019; Winter et al., 2020). Within these studies, migration routes and losses of eels along the routes could be determined from the acoustic telemetry results, and with mark-recapture studies at IJmuiden with PIT-tags. The accurate population estimates of the number of silver eels that arrive at IJmuiden could be determined. These results can be compared to the outcome of the hierarchical approach used in this evaluation to assess the biomass of silver eel that escapes to sea via IJmuiden.

In the evaluation approach to assess silver eel escapement at IJmuiden, this is assessed to be 122 tonnes per year for 2018-2020, using the new insights showing that 40 % of the silver eel from lake Markermeer also migrate via IJmuiden. The starting of biomass silver eel in this evaluation method for the North Sea Canal (IJmuiden) catchment was assessed at 212 tonnes, i.e. an average loss rate of 43%.

The mark recapture-studies for silver eel migration at IJmuiden yielded 101.347 ± 10.990 silver eels in 2016 and 89.233 ± 9.791 in 2017 (Winter et al., 2019; Winter et al., 2020). Earlier assessments yielded 70,000-100,000 silver eels arriving at IJmuiden in 2007-2008 (Winter 2011), which is in line with the later, more-precise estimates of the number of silver eels at IJmuiden. With an average weight of 850 g per silver eel for the region (Winter, 2011), this yields 86.1 ± 9.3 tonnes in 2016 and 75.8 ± 8.3 tonnes in 2016. The telemetry data suggests that 45-50% of the starting silver eel do not reach the sea, even though the representativeness of the various tagged groups in the hinterland of the North Canal catchment for all starting silver eel within this catchment is not known. This implies that 152-191 tonnes of silver eel would have started as derived from the combination of mark-recapture experiments and telemetry studies.

Given the number of assumptions that are present in the evaluation approach, the outcome of the starting biomass of 212 tonnes vs. 152-191 tonnes in the telemetry studies, the escapement of 122 tonnes, vs. 76-86 tonnes in the telemetry studies and the overall loss rate of 43% vs. 45-50% in the telemetry studies, these are in relatively close range and add confidence in the following assessment method in this evaluation for the other regions. These results suggest that the escapement and starting biomass for this region is slightly overestimated in the evaluation, when compared to the results of the telemetry experiments.

Appendix C4: Overview of the parameters used in the barrier mortality estimation.

Table C6. Overview of the parameter estimations used in the estimation of the barrier mortality.

Name	Value	Period	Description
fracAm	0.219	all	proportion through Amerongen
CboezemCzee	0.370	all	prop. from boezem to sea
CboezemCrijks	0.630	all	prop. from boezem to national waters
CpolderCzee	0.200	all	prop. from polder to sea
CpolderCboezem	0.800	all	prop. from polder to boezem
Spolder	0.347	all	mortality polder to boezem or sea
Sboezemzee	0.050	all	mortality from boezem to sea
Sboezemrijks	0.152	all	mortality from boezem to national waters
Srijks	0.023	2006-2008	mortality from national waters to sea
Srijks	0.023	2009-2011	mortality from national waters to sea
Srijks	0.021	2012-2014	mortality from national waters to sea
Srijks	0.015	2015-2017	mortality from national waters to sea
Srijks	0.020	2018-2020	mortality from national waters to sea
SWKClinne	0.170	2006-2008	mortality HPS Linne
SWKClinne	0.170	2009-2011	mortality HPS Linne
SWKClinne	0.130	2012-2014	mortality HPS Linne
SWKClinne	0.130	2015-2017	mortality HPS Linne
SWKClinne	0.130	2018-2020	mortality HPS Linne
SWKCalph	0.150	2006-2008	mortality HPS Alphen
SWKCalph	0.150	2009-2011	mortality HPS Alphen
SWKCalph	0.140	2012-2014	mortality HPS Alphen
SWKCalph	0.140	2015-2017	mortality HPS Alphen
SWKCalph	0.140	2018-2020	mortality HPS Alphen
SWKCamer	0.095	all	mortality HPS Amerongen
PODD	0.000	2006-2008	assisted migration
PODD	0.134	2009-2011	assisted migration
PODD	1.362	2012-2014	assisted migration
PODD	1.900	2015-2017	assisted migration
PODD	2.290	2018-2020	assisted migration

Appendix C5: Diadromous fish monitoring programme

A survey programme started in 2012 to monitor the abundance of migrating silver eel on five exit points (Kornwerderzand sluices, Den Oever sluices, North Sea Canal, New Waterway channel, Haringvliet-West inlet) and two entry points for migratory fish (River Rhine and River Meuse) during spring and autumn (Figure C3). The programme is a collaboration between WMR, Rijkswaterstaat and commercial fishermen. The months September, October and November were selected for illustrating trends in silver eel abundance at each location. In 2015 and 2018 four extra locations were monitored but not shown in Figure C4. Both eel biomass and numbers fluctuate strongly on a yearly basis at all locations (Figure C4). As the trends index is relatively short (9 years), there is no information before the implementation of the EMP and there are missing years (Figure C4), the data is not used in this report.

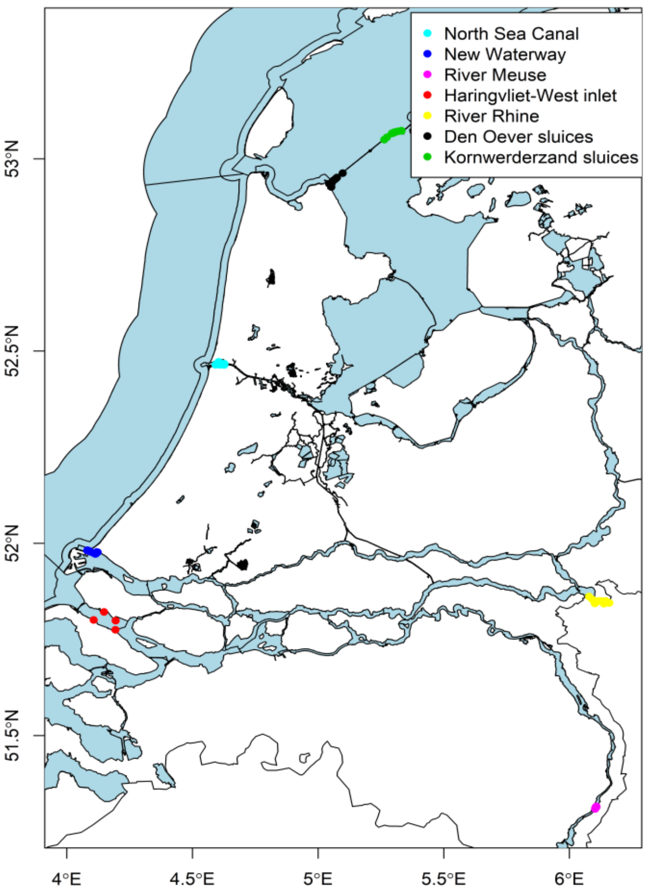

Figure C3. *Fyke Locations in the diadromous fish monitoring programme.*

Figure C4. *CPUE of silver eel caught during the diadromous fish monitoring per catch location. Data is missing or not used because of inconsistency of sampling locations/period for the Haringvliet-West inlet in 2018, for the Den Oever sluices in 2012, 2014 and 2015, for the Kornwerderzand sluices in 2012, 2013 and 2015, the River Meuse in 2017 and 2018, the North Sea Channel in 2015 and for the River Rhine in 2012, 2016 and 2018.*

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Approved by: dr. N. Tien
Researcher Wageningen Marine Research

Signature:

Date: 28/10/2021

Approved by: dr.ir. T.P. Bult
Director Wageningen Marine Research

Signature:

Date: 28/10/2021

Approved by: Ing. S.W. Verver
Head Centre for Fisheries Research

Signature:

Date: 28/10/2021