European Eel (*Anguilla anguilla*) stock size, anthropogenic mortality and silver eel escapement in the Netherlands 2006-2023.

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Samenvatting

Sinds de jaren 1980 zijn de glasaalintrek en de aalpopulatie zeer sterk teruggelopen. 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') verplichtte 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 en een aantal kanalen.
• 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 tot en met 2018 voor elke drie jaar, en daarna elke zes jaar. Echter, aangezien de huidige stand van de aalpopulatie nog steeds zorgwekkend is (ICES, 2023), hebben de lidstaten afgesproken om voorlopig elke drie jaar te blijven rapporteren aan de EC. Dit heeft ertoe geleid dat Nederland, samen met de meeste, echter niet alle, lidstaten ook in 2021 een rapportage heeft geleverd (Van der Hammen et al., 2021).

De onderhavige rapportage heeft een aantal updates ondergaan in vergelijking met de vorige rapportage (Van der Hammen et al., 2021). De verschillende onderliggende modellen zijn verbeterd. Als gevolg van een andere manier van opwerken en keuzes van de aannames zijn de schattingen voor eerdere driejaarlijkse periodes opnieuw berekend waarbij gebruik is gemaakt van de aanpassingen, zodat de jaren onderling goed vergelijkbaar blijven.

In deze rapportage wordt het aalbeheerplan geëvalueerd in het licht van het bovenstaande beheerdoel uit de Aalverordening (*Artikel 2.4*). 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 zes driejaarlijkse periodes (2006-2008, 2009-2011, 2012-2014, 2015-2017, 2018-2020 en 2021-2023). De bestandsindicatoren zijn B_0, $B_{current}$ en ΣA. B_0 is de biomassaschatting van uittrekkende schieraal in een pristine situatie. $B_{current}$ is de schatting van de

daadwerkelijke schieraalbiomassa die uittrekt naar zee en ΣA geeft de totale antropogene (visserij en barriere) sterfte aan. Voor Nederland wordt alleen gekeken naar de binnenwateren. Voor de kustwateren en de visserijzone is er nauwelijks monitoring en is het relatieve belang als aal opgroei gebied laag in vergelijking tot de binnenwateren. De doelstelling voor de (ongespecificeerde) lange termijn (*artikel 2.4* in de aalverordening*)* is dat de verhouding tussen $B_{current}$ en B_0 hoger is dan 0.40 (40%).

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 ($B_{current}$) gedurende de hele periode met een kleine dip in 2018-2020 (*Tabel 2*). Ook is er een teruggang in antropogene sterfte (ΣA) gedurende deze periode. 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, wat een direct gevolg was van de drie maanden sluiting van de visserij in september, oktober en november naar aanleiding van de implementatie van het aalbeheerplan en de sluiting van de recreatieve visserij. In de meest recente periode (2021-2023) is de antropogene sterfte weer afgenomen en de biomassa uittrekkende schieraal ($B_{current}$) in vergelijking met de periodes ervoor weer gestegen. Deze stijging is het gevolg van de toegenomen biomassa schatting van de aanwezige rode aal en schieraal, wat een resultaat van de toegenomen vangstsuccessen in de gebruikte monitoringen, met name in de rijkswateren (grote rivieren en IJsselmeer/Markermeer). In de kleinere wateren (Kader Richtlijn Water wateren) is het vangstsucces van de monitoringen juist weer wat gedaald.

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; ΣA: antropogene sterfte, ΣH: barriere sterfte, ΣF: visserijsterfte.*

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020	2021-2023
Biomassa	B_0^*	10,400	10,400	10,400	10,400	10,400	10,400
	$B_{current}$ (tonnes)	555	724	830	1,022	952	1,269
	$100^* B_{current}/B_0$	5.3	7.0	8.0	9.8	9.1	12.2
Sterfte	ΣA (anthropogenic mortality rate)	1.80	0.83	0.67	0.55	0.72	0.60
	ΣH (barrier mortality rate)	0.20	0.20	0.19	0.17	0.15	0.11
	ΣF (fisheries mortality rate	1.61	0.63	0.48	0.38	0.57	0.50

* Zonder kustwateren (2,600 ton)

Door aanpassingen aan de infrastructuur bij migratieknelpunten, alsmede de verhouding biomassa tussen verschillende gebieden in Nederland, is de barrière sterfte van schieraal afgenomen (van ΣH = 0.20 in 2006-2008 naar ΣH = 0.11 in 2021-2023). Deze afname is voornamelijk veroorzaakt doordat er in recentere perioden naar verhouding meer aal in de rijkswateren zit ten opzichte van de polders, boezems en kleine meren (KRW wateren). In de rijkswateren is er nauwelijks/geen barriéresterfte apart van de vier grote kunstwerken (WKC Linne en Lith in de Maas, WKC Maurik in de Nederrijn en het sluizencomplex bij IJmuiden). Door de strengere maatregelen om de schieraalsterfte onder de 5% te houden wanneer de schieralen door de WKCs trekken, is ook deze sterfte in de laatste periode (2021-2023) gedaald.

Wanneer de huidige schatting van het aalbestand vergeleken wordt met de schatting van de pristine waarde (*B0*), blijft de status van het aalbestand in Nederland in 2021-2023 laag. De huidige biomassa van uittrekkende schieraal (12.2%) ligt sinds de implementatie van het aalbeheerplan onder de

doelstelling van minimaal 40% van de pristine biomassa. De huidige sterfte door menselijk handelen ligt onder A_{lim}. Er wordt echter door ICES in toenemende mate erkend dat de schattingen van B_0 van de verschillende lidstaten (of 'aal beheer units') niet te vergelijken zijn met de latere biomassa schattingen omdat deze een zeer hoge onzekerheid hebben en veelal op verschillende manieren berekend worden. Daarnaast heeft glasaaluitzet een zeer grote invloed op een lokale aalstand, terwijl het niet zeker is of uitgezette glasaal de paaigronden in de Sargassozee kan bereiken (ICES 2016, 2023).
Aangezien de aalpopulatie een panmixtische populatie is met een natuurlijke verspreiding van Noorwegen tot noord Afrika, zegt de toestand van de aal in Nederland niet veel over de toestand van de gehele populatie. Herstel in de totale aalstand is daardoor de gezamenlijke verantwoordelijkheid van alle landen waar aal van nature voorkomt.
Een eventuele groei in de aalpopulatie zal naar verwachting langzaam verlopen 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.

Uit de analyses die voor deze rapportage zijn uitgevoerd is wederom gebleken dat er grote aannames gemaakt moeten worden om tot een biomassaschatting van schieralen te komen, welke van invloed zijn op de resultaten. De omvang van de opwerking (schieraalbiomassa 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 1990s. 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.
• Ban on fisheries in the large rivers and some canals.
• Release of eel caught (a) at sea and (b) at inland waters by anglers.
• Ban on recreational fishing using professional gear in coastal areas.
• Closed fishing 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 snigglers (*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 Member States to report to the EC on the effectiveness of the eel management plans. This obligation applied every three years until 2018, and every six years thereafter. However, as the current state of the eel population is still worrying (ICES, 2023), Member States have agreed to continue reporting to the EC every three years for the time being. This has led to the Netherlands, together with most, but not all, Member States, also providing a report in 2021 (Van der Hammen et al., 2021).

The current report has undergone a number of updates compared to the previous report (Van der Hammen et al., 2021). In addition to calculating estimates for the most recent years, previously published estimates have been updated based on the new improved methods.

In this report, the eel management plan is evaluated in the light of the above management objective from the Eel Regulation (Article 2.4). The evaluation was carried out using models, catch data, field observations and statistical analyses, which are described in the report. All of this effort results in estimates of a number of stock indicators requested by the EC for six three-year periods (2006-2008, 2009-2011, 2012-2014, 2015-2017, 2018-2020 and 2021-2023). The indicators are B_0, $B_{current}$ and ΣA.

B_0 is the biomass estimate of migrating silver eel in a pristine situation. $B_{current}$ is the estimate of the actual silver eel biomass escaping to sea and ΣA indicates the total anthropogenic (fishing and barrier) mortality. For the Netherlands, only inland waters are considered. There is hardly any monitoring for the coastal waters and the relative importance as eel habitat is low compared to inland waters. The (unspecified) long-term objective (Article 2.4 in the Eel Regulation) is that the ratio between $B_{current}$ and B_0 is higher than 0.40 (40%).

Effects of the Dutch eel management plan on the Dutch eel population

The estimates of the stock indicators show that the measures in the Dutch eel management plan have led to an increase in migrating silver eel biomass ($B_{current}$) during the entire period with a small dip in 2018-2020 (Table 0.2). During the same period, the anthropogenic mortality (ΣA) declined. In particular, the first period after the implementation of the measures from the Dutch eel management plan (2009-2011) resulted in a decrease in anthropogenic mortality, which was a direct consequence of the three months closure of the fishery following the implementation of the eel management plan and the closure of recreational fishing. In the most recent period (2021-2023), anthropogenic mortality has decreased again and the biomass of escaping silver eel ($B_{current}$) increased compared to the previous periods. This increase is the result of the higher biomass estimate of the yellow eel and silver eel, which is an outcome of the higher catch successes in the monitoring, especially in national waters (large rivers and IJsselmeer/Markermeer). In contrast, in the smaller waters (Water Framework Directive waters) the catch success in the monitoring has decreased somewhat.

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; ΣA anthropogenic mortality rate; ΣH barrier mortality rate, ΣF fisheries mortality rate.*

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020	2021-2023
Biomassa	B_0^*	10,400	10,400	10,400	10,400	10,400	10,400
	$B_{current}$ (tonnes)	555	724	830	1,022	952	1,269
	$100^* B_{current}/B_0$	5.3	7.0	8.0	9.8	9.1	12.2
Mortality	ΣA (anthropogenic mortality rate)	1.80	0.83	0.67	0.55	0.72	0.60
	ΣH (barrier mortality rate)	0.20	0.20	0.19	0.17	0.15	0.11
	ΣF (fisheries mortality rate	1.61	0.63	0.48	0.38	0.57	0.50

* Excluding coastal waters (2,600 tonnes)

Due to adjustments to the infrastructure at migration bottlenecks, as well as the biomass ratio between different areas in the Netherlands, the barrier mortality of silver eel has decreased substantially from ΣH = 0.20 in 2006-2008 to ΣH = 0.11 in 2021-2023. This decrease is mainly caused by the fact that there are relatively more eels in the national waters compared to polder waters, and small lakes (WFD waters). In national waters, there is little or no barrier mortality apart from the four major barriers (HPS Linne and Lith in the Meuse, HPS Maurik in the 'Nederrijn' and the sluices/pumping complex at IJmuiden). Due to the stricter measures to keep silvereel mortality below 5% in the HPSs, this mortality has also fallen considerably in the last period.

When the current estimate of the eel stock is compared with the estimate of the pristine value B_0, the status of the eel stock in the Netherlands remains low in 2021-2023. The current biomass of migrating silver eel (12.2%) is still far below the target of 40% of the pristine biomass. However, the current mortality due to human actions (ΣA) is below A_{lim}, since the implementation of the Dutch Eel Management plan (2009). However, it is increasingly recognized by ICES that the estimates of B_0 from the various Member States (or 'eel management units') cannot be compared with the later biomass estimates because they have very high uncertainty and are calculated in different ways. In addition, restocking of glass eel and elver has a major influence on local eel stocks, while it is not certain whether released glass eels can reach the spawning grounds in the Sargasso Sea (ICES 2016,2023).

Since the eel population is a panmictic population with a natural distribution from Norway to North Africa, the status of the eel in the Netherlands does not necessarily reflect the status of the entire population. Recovery is therefore the joint responsibility of all countries in the distribution range of the eel.
Because the different methods to calculate the indicators by the Member States and the high uncertainty in the estimates, the results cannot be properly compared with each other.
Any growth in the eel population is expected to be slow because eel is a long-lived species. It takes an estimated 1-3 years for glass eels to arrive at the Dutch coast and migrate into inland waters. It then takes 3-20 years before these eels become “silver eels” and return to the sea.

The analyses carried out for this report have once again shown that large assumptions have to be made to arrive at a biomass estimate of silver eels. The extent of the reprocessing (eel biomass in all Dutch waters) and the available (historical) data do not lend themselves to very accurate calculations. The stock indicator estimates should therefore be interpreted with caution due to the significant degree of uncertainty surrounding these estimates.

1 Introduction

1.1 EU regulation and the Dutch eel management plan

In response to the decline in the European eel (*Anguilla anguilla*) stock since the 1980's the EU Regulation for the Recovery of the Eel Stock (EC 1100/2007) was adopted in 2007. This so called 'Eel Regulation' required each Member State (MS) within the natural distribution area of the eel to set up Eel Management Plan's (EMP's) 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', Figure 1-1), which, in turn covers a specific eel habitat (for example a river basin). Because the Netherlands is located in the joint delta of four major rivers and the rivers are intertwined and confluent, there are no sharp boundaries between river basins. Therefore the Netherlands is defined as a single EMU and a single EMP was drawn up covering the whole country. 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[1]. In 2012, 2015, 2018 and 2021, progress reports were sent to the EC each showing that the status of eel in Dutch waters remained below the target of 40% of the estimated pristine situation. However, the progress reports also show that implementation of the EMP resulted in an initial increase in biomass and a decrease in anthropogenic mortality (Bierman et al., 2012; van de Wolfshaar et al., 2015 & 2018, Van der Hammen et al. 2021).

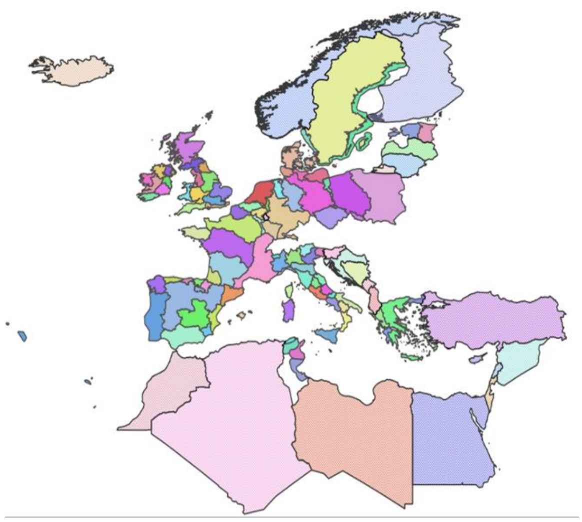

Figure 1-1 Eel Management Units (EMU's).

[1] A quota system in the province of Friesland replacing the three months fishing closure.

Table 1.1 Overview of the implemented measures described in the Dutch Eel Management Plan (Ministry of Agriculture, Fisheries, Food quality and Nature (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]
• Fisheries ban in the main large rivers and some canals.	Unforeseen	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	2009	Early 2010 - ongoing
• 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 but cannot fish more than a fixed 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. An explanation of each stock indicator is briefly described below. 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 difficult. Because the reference points for the ICES approach (B_{lim}, F_{lim}, *Table 1.2*) had not been established for eel, alternative biomass and mortality reference points were developed (ICES, 2014; *Table 1.2*). 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}.

B_0 is the pristine silver eel biomass escapement to sea. It is an estimated equilibrium value of the amount of silver eel (in biomass) that would be able to migrate to the spawning grounds 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 long-term target in 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 9.2*).

$B_{current}$ is an estimate of the current amount of silver eel (in biomass) that is able to reach the open sea to migrate to the spawning grounds. It is an indication of 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, the current biomass also highly depends on the number of recruits (glass eels) arriving at the Dutch coast, which, in turn, depends on the status of the total eel stock in its entire distribution area. The inflow of recruits remains currently at a low level compared to historic values (ICES 2023).

B_{best} is an estimate of the best possible silver eel escapement under recent recruitment conditions. It is an estimate of the current amount of silver eel (in biomass) escaping to the spawning grounds if there would be no current anthropogenic influences.

ΣA, ΣF and ΣH are anthropogenic mortality rates. *ΣA* is the total anthropogenic mortality rate, *ΣF* is the fishing mortality rate and *ΣH* is the anthropogenic mortality rate excluding fishing mortality. For the current evaluation, *ΣH* includes only the silver eel barrier mortality rate. Mortality rates due to, for example, pollution, parasites or illegal removals, are not taken into account.

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). The rationale is that at that level there is no recruitment impairment. 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 pristine spawning-stock biomass, to account for uncertainty (B_{pa}), 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 including a 20% precautionary buffer (50%).

A_{lim}: Eel experience relative high levels of non-fishing anthropogenic mortality compared to other commercially exploited stocks. Therefore, the mortality reference point (A_{lim}) includes all anthropogenic mortality and not only the fishing mortality. A_{lim} is derived from B_{lim} as follows: $\Sigma A = -ln(0.4) = 0.92$ (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 should be lower than A_{lim} to reinforce the tendency for the stock to rebuild (ICES, 2018).

Table 1.2 Overview of the reference points and 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.

Reference point	**Definition**	**Value**
B_{lim}	Biomass limit below which a stock is considered to have reduced reproductive capacity.	0.40 * B_0
A_{lim}	Mortality rate limit above which a stock decline is expected.	0.92
Stock indicator		
B_0	Silver eel escapement (biomass) in the absence of any anthropogenic impact and at historic recruitment levels (inland waters).	10,400 t
$B_{current}$	Silver eel biomass that currently (assessment year) escapes to the sea to spawn.	variable
B_{best}	Silver eel biomass estimate without anthropogenic influences on the current stock, i.e., the best biomass possible under current recruitment levels.	variable
B_{start}	Silver eel biomass before current silver eel mortalities (migration and fisheries) have occurred.	variable
ΣA	Life time anthropogenic mortality; the fishing mortality and the mortality outside of fisheries (HPS's, pumping stations etc.). The sum of anthropogenic mortalities, i.e., $\Sigma A = \Sigma F + \Sigma H$.	variable
ΣH	Anthropogenic mortality rate other than fishing mortality (e.g., barrier mortality).	variable
ΣF	Fishing mortality rate (commercial and recreational).	variable

1.3 Description of the models

To estimate silver eel escapement and anthropogenic mortality, the following calculations were carried out: 1. Yellow and silver eel standing stock, with either a demographic model or a spatial model; 2. Mortality of migrating silver eel and 3. Yellow eel fishing mortality. Each step is briefly described below. In the following chapters, the methods are described in more detail.

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 spatial model is explained in detail in *Chapter 4*.

Demographic model: For the IJsselmeer, Markermeer, the method of the 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, assumptions as made for the spatial model can be extremely influential. Instead, for the lakes IJsselmeer and Markermeer, the fishing mortality rates were estimated by fitting a 'Demographic Model' to electric trawl survey time series. 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 5 and Appendix B1*.

Migration model: to estimate the silver eel mortality during migration from inland water bodies to the sea, due to barriers such as pumping stations and turbines in Hydropower stations (HSP's) a barrier model is used. The model assumes that, depending on their 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 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 (paragraph 1.2). The estimation of the stock indicators is described in *Chapter 7 and 8*.

1.3.1 Structure of the report

The stock assessment consists of several steps. Below the content of each Chapter is summarized:

- *Chapter 2:* Description of the available data
- *Chapter 3*: Description of the biological keys (maturity-at-length, weight-at-length, and sex-ratio-at-length) that are used in the demographic model and the spatial model.
- *Chapter 4*: Description of the spatial model, which is used for the estimation eel biomass in the regionally (Water Framework Directive) and nationally managed (Large rivers) water bodies.
- *Chapter 5*: Description of the demographic model. The model is used for estimating the silver and yellow eel biomass in the large lakes IJsselmeer, Markermeer, Randmeren and Grevelingen.

Chapter 6: Description of the migration model for the estimation of silver eel mortality due to barriers.
Chapter 7: Overview of the total standing stock biomass. In this chapter the results from chapters 2-6 are combined.
Chapter 8: Overview and discussion of the final key stock indicators.
Chapter 9: The report concludes with a general discussion and recommendations for improvements to the stock assessment methodology.

Since the latest report (Van der Hammen et al, 2021) several improvements and updates were made:

1. Addition of the most recent data (up till 2023). This results in the following time periods to report on: 2006-2008, 2009-2011, 2012-2014, 2015-2017, 2018-2020 and 2021-2023.
2. The biological keys are updated with the newest data. This results in an adaptation of the biological keys, including the new data.
3. The demographic model was updated (see *Chapter 5* 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 4*).
5. For the large rivers, now the Water Framework Directive regions are used, instead of previously defined regions and a new GIS map to define the watersurface area is used (*Chapter 4*).

2 Available data

The main available data sets are described below.

2.1 Water Framework Directive monitoring

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[2]. Fishing is done with a fine-meshed electric dipping net whereby the net functions as anode. This method is most suitable to monitor shallow shorelines of all kind of habitats (rip-rap, reed etc.). A minimum/standard transect distance of 250m is used, and a reach of 3m is assumed, so the standard swept area is 0.0375 hectares. 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. More than 13,000 samples by electric dipping nets were available between 2006 and 2022, covering most of the combination of water boards and water body types.

2.2 Silver eel in fyke net monitoring

A survey program started in 2012 to monitor the abundance of migrating silver eel on seven locations during autumn. In 2015 another 4 locations were included, which are sampled once every three years. The program is a collaboration between WMR, Rijkswaterstaat and commercial fishermen. The months September, October, November and December were selected for illustrating trends in silver eel abundance at each location. Only the locations where at least 6 years of data exists are presented here. The trends in silver eel catch per unit of effort (number/fykenight) are not consistent between the different locations. Most locations do not show a clear consistent trend (*Figure 2-2*).

Figure 2-1 Locations of the fykenet monitoring.

In addition, at the Waddensea site at Kornwerderzand a fyke net survey program exists for a longer period (since 2001). In this monitoring, a distinction between yellow and silver eel

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[2] Apart from 'Scheldestromen' see paragraph 4.3.2

is not always made and therefore we present yellow and silver eel together in autumn and spring. There is a downwards trend from the beginning of the monitoring (2001) to ~2007, but in autumn, there is a clear increase in eel cpue (number/fykenight) since ~2016 (*Figure 2-2*).

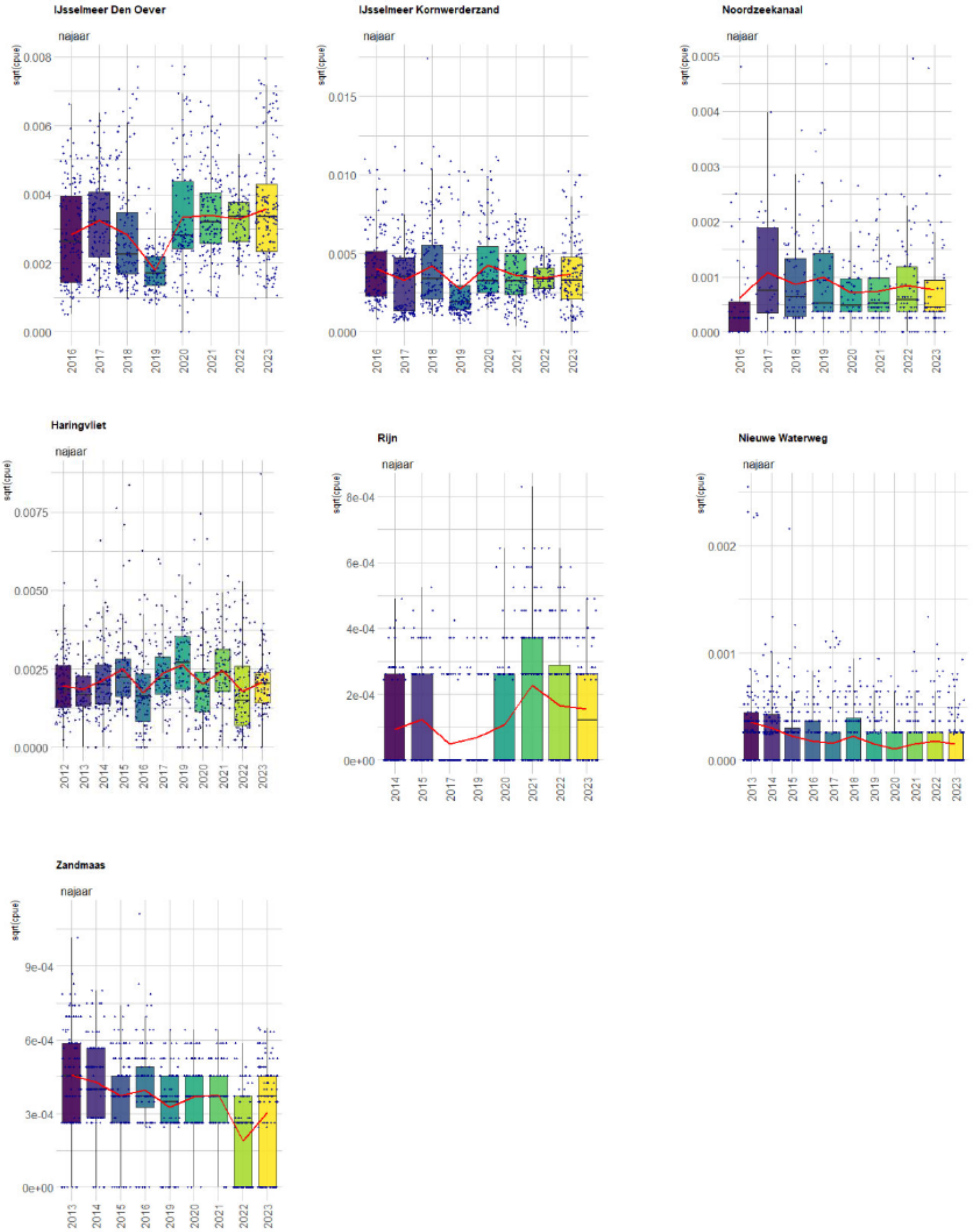

Figure 2-2 Silver eel monitoring in fykenets in autumn. The y-axis presents the square root of the catch per unit of effort (number/fyke night). The box shows the middle 50% of scores (i.e., the range between the 25th and 75th percentile). The red lines are the average and each dot is an individual sample. Note that some years are missing.

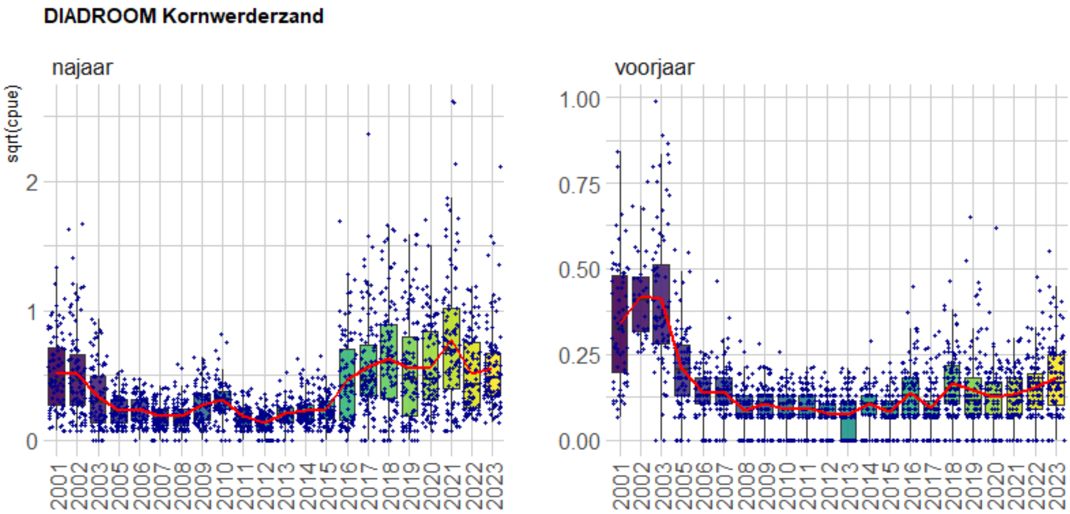

Figure 2-3 Fykenet monitoring at the Waddensea site (discharge sluices) at Kornwerderzand. Silver eel and yellow eel combined. The red lines are the average and each dot is an individual sample. Left: autumn, right: spring.

2.3 Glass eel survey liftnet Den Oever

Since 1938, a recruitment monitoring is running at Den Oever. The monitoring is conducted with a liftnet (1x1 m), each year in March, April and May. Glass eel data are presented as the average number of glass eels per haul in the months of April and May. To show the recent trend, the series is visualized since 1998. From this date no clear trend exists, but there is a large fluctuation in the trend (with better years in 2013 and 2014, but a decrease afterwards, *Figure 2-4*).

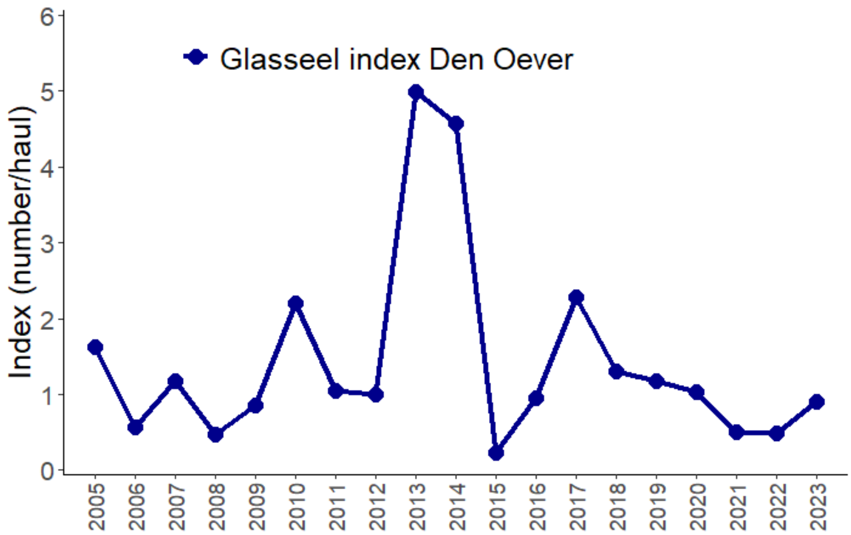

Figure 2-4 Index (number/liftnet haul) in the recent glasseel survey in the sluices at Den Oever. The presented index is the average of the hauls in April and May.

2.4 Ditches and 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) in collaboration with Waardenburg Ecology every year and is added to the spatial model. A total of ~350 samples by electric dipping nets were available between 2013 and 2023.

2.5 Nationally managed water bodies

2.5.1 FYMA electric trawl survey in lakes IJsselmeer and Markermeer

Since 1989, an annual (yellow) eel survey in lake IJsselmeer (25 sites) and lake Markermeer (15 sites) is executed with an electrified trawl. The survey takes place in autumn (October-November). The survey shows a strong decrease in eel numbers and biomass since the beginning of the survey. In recent years, the biomass has increased substantially, mainly due to an increase in average eel length. The data is used to tune the demographic model (*Chapter 4*).

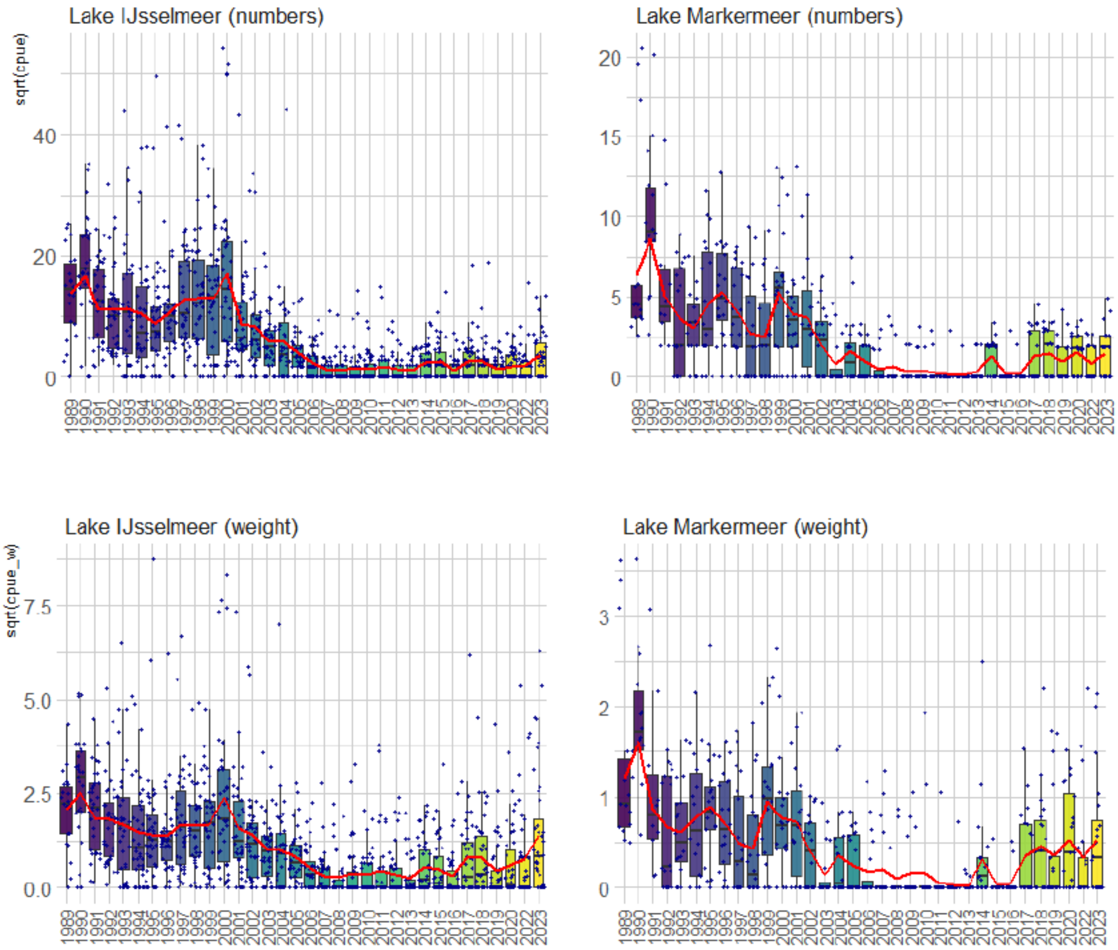

Figure 2-5 FYMA electric trawl survey in lakes IJsselmeer and Markermeer. Top: index (numbers/hectare), bottom: index (weight/hectare). The red line represents the mean. Dots represent individual hauls. Because of an issue with the gear, there is no data in 2021.

2.5.2 FGRA electric dipping net survey in larger rivers and lakes.

Since 1997 an annual, standardized electric dipping net eel survey the large nationally managed rivers started. In later years, more locations were added. The surveys take place in autumn for the downstream water bodies and during spring for the upstream water bodies. Since 2015, the CPUE seems to increase in the downstream rivers (e.g., Nieuwe Merwede, Oude Maas, *Figure 2-6*). These are also the regions where the CPUE is highest compared to other water bodies. The data is used as input for the Dutch eel stock biomass estimation (*Chapter 4*).

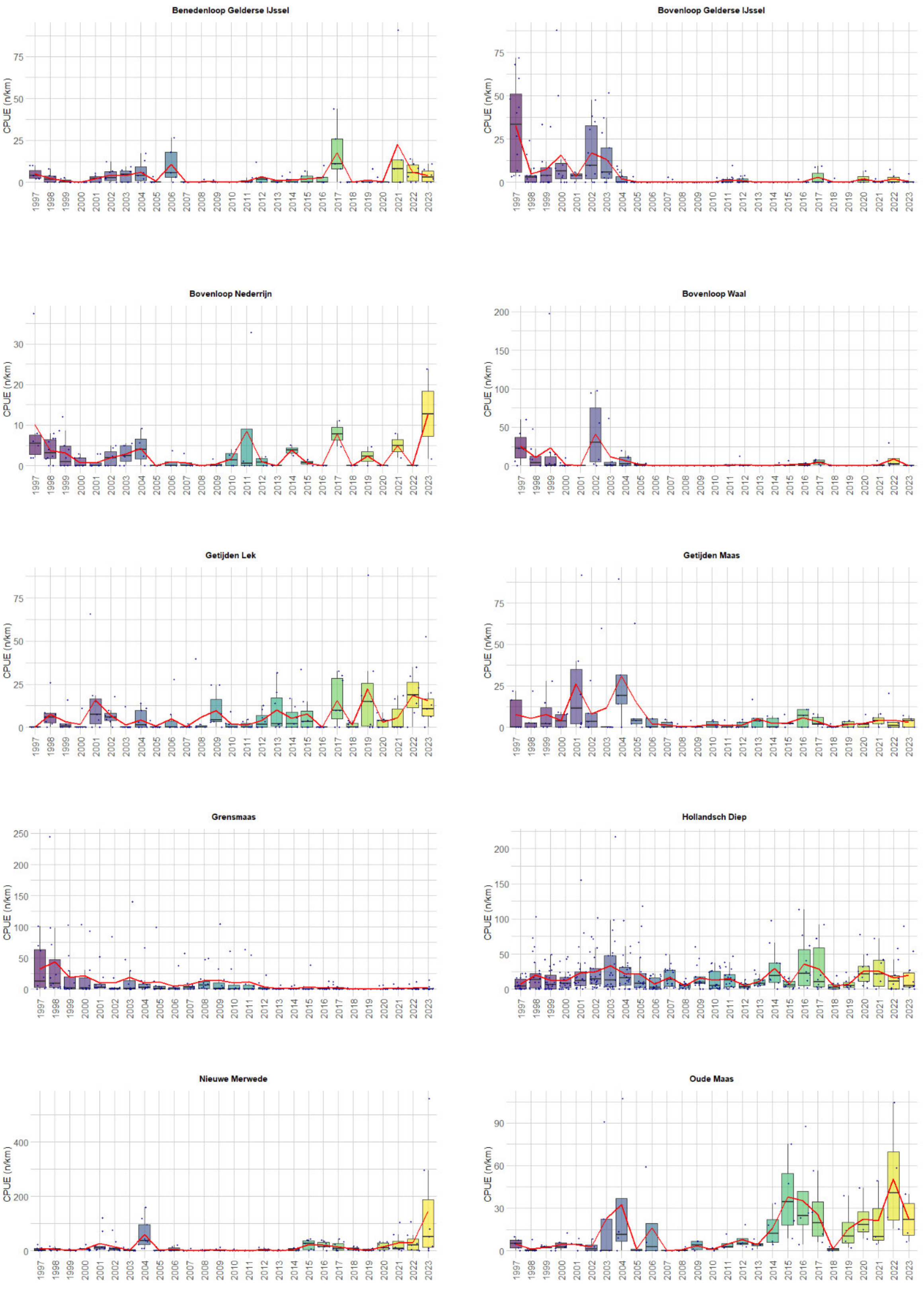

Figure 2-6 FGRA electric dipping net survey in the large rivers (nationally managed water bodies). CPUE (number/km). Note that in some locations, years are missing. Only the locations that have relative long timeseries and with sufficient eel catches are shown.

2.6 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.

2.7 Commercial and recreational landings

2.7.1 Commercial landings

Since 2010 all freshwater fishermen have to report their landings to the Ministry of Agriculture, Fisheries, Food quality and Nature (LVVN, RVO), which are stored in a WMR database ('Visstat'). Since 2010 the landings decreased until 2016, after which they increased. The main increase is due to an increase in the lakes IJsselmeer and Markermeer (*Figure 2-7*).

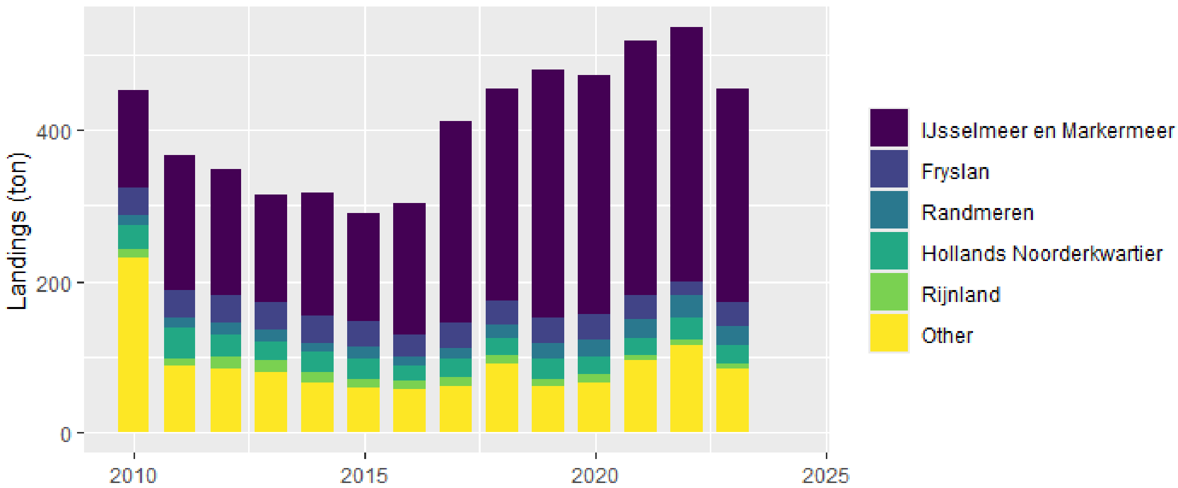

Figure 2-7 Commercial fresh water landings since 2010. Source: RVO

2.7.2 Recreational landings

An estimate of recreationally retained freshwater catches is available biennially starting in 2010 (Van der Hammen & Chen, 2024). Since the implementation of the EMP in 2009 eel recreational fisheries is catch and release only. As such, the retained recreational landings have decreased substantially (*Figure 2-8*). The confidence intervals are quite large, showing the large uncertainty around the estimates. As the recreational survey is limited to anglers, all recreational landings are assumed to be yellow eel (because silver eel are assumed not to eat).

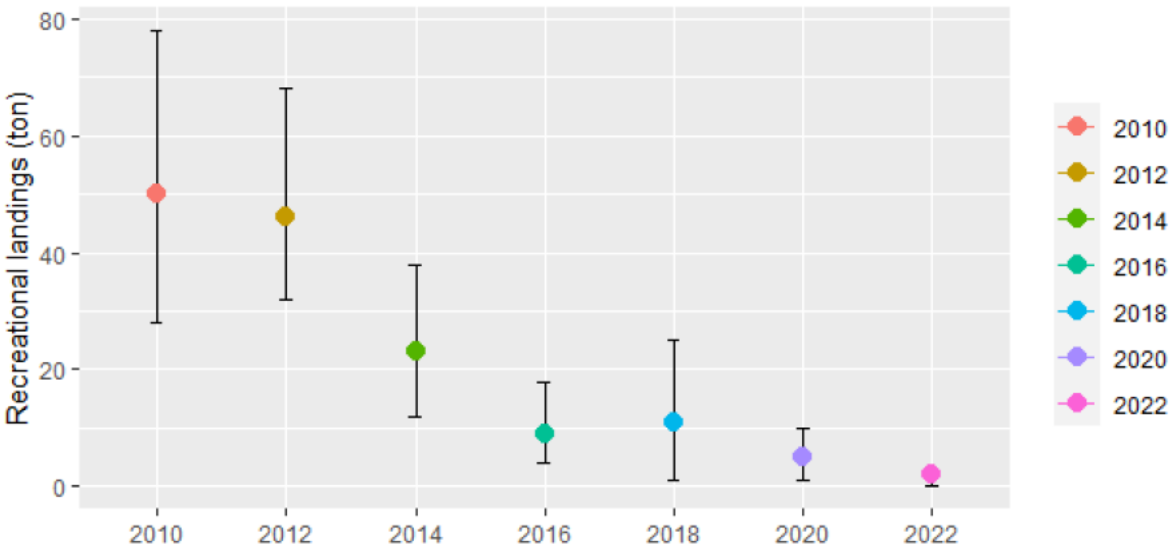

Figure 2-8 Recreational fresh water eel landings since 2010 (in tonnes with 95% confidence intervals). Source: Biennial survey WMR

2.7.3 Landings and recreational fisheries per assessment period

Because reporting of commercial landings became obligatory after the EMP came into place (end of 2009) landings from before 2010 are incomplete (apart from lake IJsselmeer and Markermeer). As part of the EMP, the Ministry of Agriculture, Nature and Food quality estimated the commercial landings in the period before the EMP, resulting in 805 tonnes (525 tonnes yellow eel and 280 tonnes silver eel, LVVN, 2018).

For recreational fisheries, the first estimate based on our regular survey is from 2010. As part of the EMP, the Ministry of *Agriculture, Fisheries, Food quality and Nature* estimated the recreational landings in the period before the EMP, resulting in 200 tonnes yellow eel (LVVN, 2018).

For each 3-year period an average estimate of the commercial and recreational landings was calculated. This resulted in a clear decrease of total catches from the first period (2006-2008) to the second period (2009-2011). After the second period, there was a decrease until 2015-2017, after which total landings increased again to a yearly average of 504 tonnes in 2021-2023 (*Table 2.1*). The proportion of yellow and silver eel is calculated from the proportion of silver eel and yellow eel in the market sampling.

Table 2.1 Overview of average yearly fresh water commercial and recreational landings in tonnes for each period.

	Commercial			Recreational	Total
Period	**Total** (per year)	**Yellow Eel** (per year)	**Silver Eel** (per year)	**Recreational Yellow Eel** (per year)	**Commercial + Recreational** (per year)
2006-2008*	805	525	280	200	1005
2009-2011	410	258	152	62	460
2012-2014	327	206	121	34	362
2015-2017	334	210	124	6	343
2018-2020	469	295	174	8	477
2021-2023	502	316	186	1	504

*Eel management plan p. 16

2.8 Assisted migration

Assisted migration: assisted migration (also called trap and transfer) initiatives, in which silver eel is caught above a barrier and 'lifted' across it, are considered when calculating the overall migration mortality of silver eel. Since 2011, several projects have started at migration barriers (mainly pumping- and hydropower stations) to assist the migration of silver eel. This assisted migration is subtracted from the amount of eel going through the barrier before the mortality is estimated.

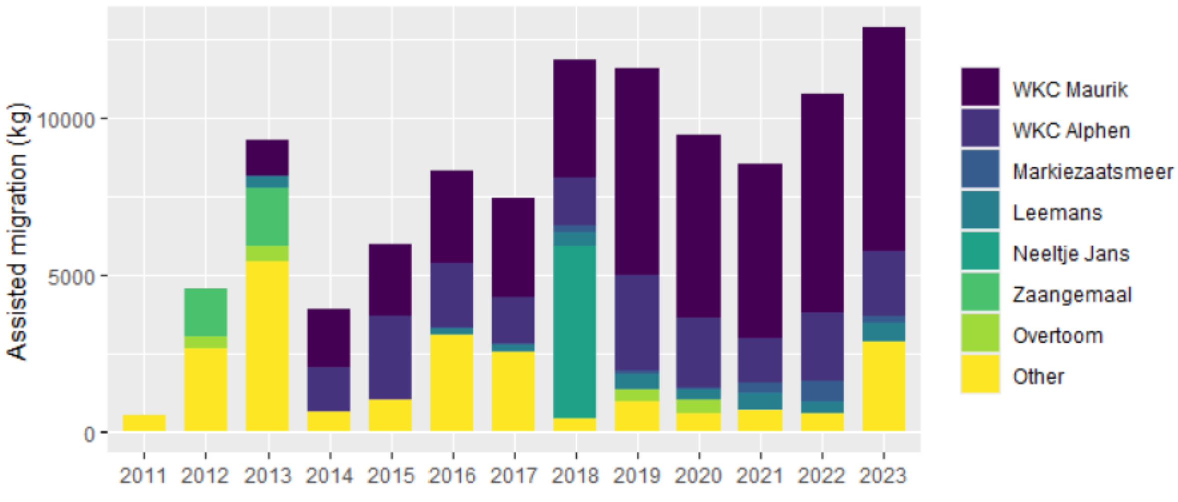

Figure 2-9 Assisted migration (kg per location). The Assisted migration at Neeltje Jans in 2018 consisted of silver eel bought from fishermen at different locations and released at Neeltje Jans. WKC = Hydropower station. Source: DUPAN

3 Biological Keys

3.1 Biological market sampling

Each year samples are taken from retained catches from commercial fisheries and the lengths of the individual eels are measured (van Keeken et al. 2023). Subsequently, several eels per length class are selected for dissection and measurements of maturity, weight and sex (see van Keeken et al. 2023 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, to estimate sex-specific growth curves.

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 are 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 because 1) more biological data became available since the previous assessment and 2) some keys are re-calculated with a different method.

In addition to the market samples, for the estimation of the age-at-length key, otolith readings from the DAK project ('*Duurzaam Aalbeheer door Kennis', N=120*) are added to the otolith readings from the market samples. In total > 12,000 individual eels collected from the commercial catches between 2006-2023 were used to assess the biological keys. From 911 individual eels sampled between 2009-2022 the otoliths were analyzed to assess the 'years after arrival at the coast'; the age after arrival. 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.

3.2 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. Because the real relationship is unknown and unlikely to be linear, the length-sex ratio relationship was estimated by fitting a *GAM* (*Figure 3-1*). 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.

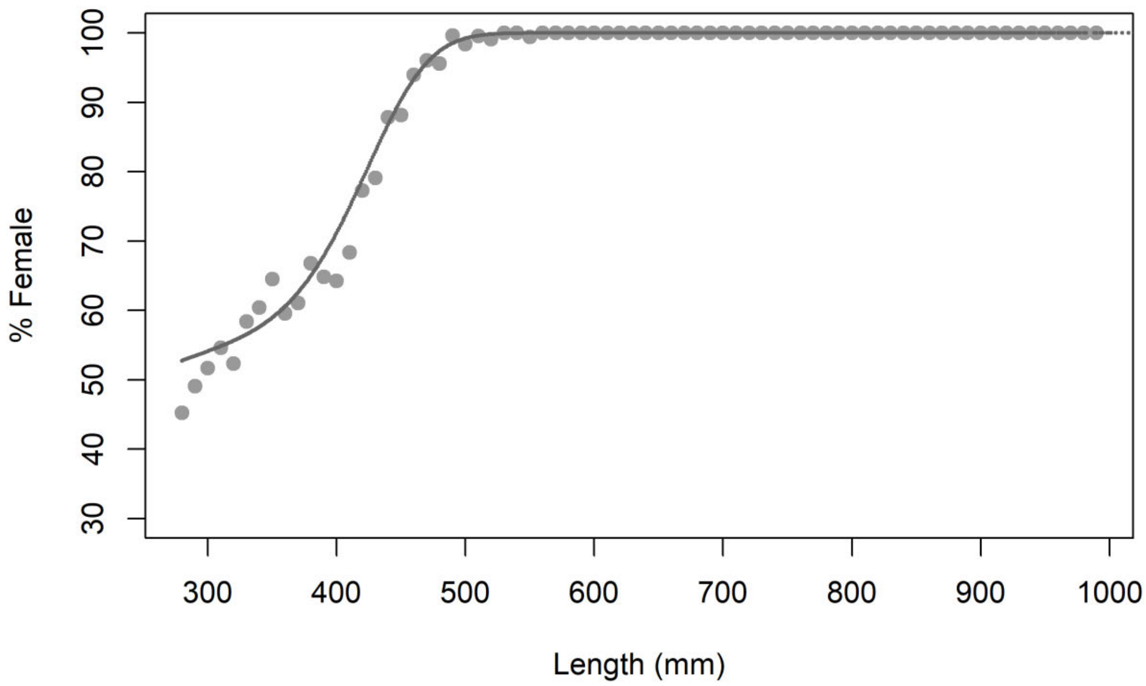

Figure 3-1 Percentage of females per 10mm length class (dots) and GAM fit based on all samples (≥28cm) from the market sampling program 2006-2023 per 10mm length class. N (males) = 2,786, N (females) = 11,682.The GAM fit was done on the raw data (e.g. males = 0 and females = 1), but for visualization the % is plotted.

3.3 Maturation at length

Males mature to silver eel at smaller sizes than females. In contrast to non-migratory fish, most mature eel (silver eel) in the catches represent the eel that became silver eel only recently because most eel start migrating to the sea directly after silvering. 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[3], 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 considered 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 3-2*). The

[3] 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.

analyses show that males start to silver at smaller lengths (~ 33 cm) compared to females (~ 50 cm). The *GAM* analysis (*Figure 3-2*) 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. The fit of the maturity key can have a large impact on the estimation of the proportion of eel becoming mature, especially in waters where eel are relatively large.

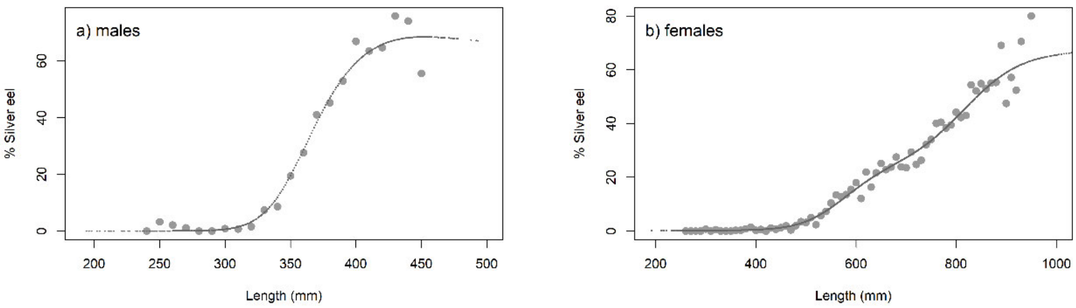

Figure 3-2 Observations (circles, average per 10mm class) and predicted GAM fit (lines) of the percentage of silver eel per length class (10mm). Data source: market sampling program (2006-2023). a) males (N = 2,786), b) females (N= 11,682).

3.4 Weight at length

A length-weight (LW) relationship is used to estimate eel biomass given numbers-at-length. The length - weight relationship does not differ for males and females and is calculated using individual length and weight measurements from yellow eel in market samples (*Figure 3-3*).

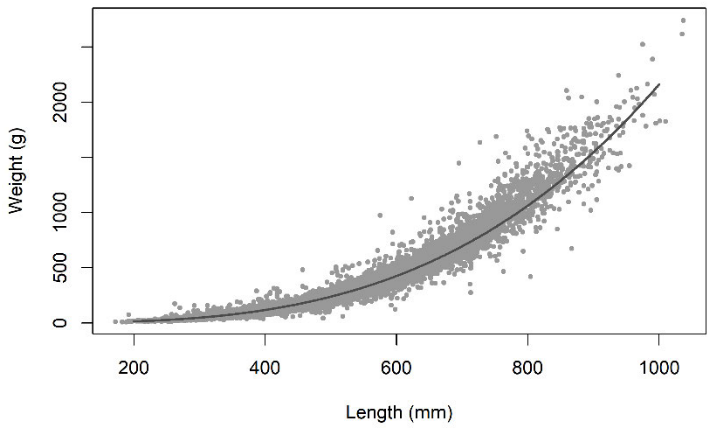

*Figure 3-3 Length-weight relationship for eel based on market sampling data (2006-2023). N = 19,478). Estimated relationship: weight = exp (-14.33 + 3.18*log(L)), with weight in grams and length (L) in millimeters.*

3.5 Growth

Because of the dimorphic differences between males and females, growth is assessed for males and females separately. Otolith readings from eels collected between 2009-2022 were used to age the eels. For age 0, the mean length of glass eels arriving in Den Oever was used (7.3cm). The sex specific growth curve was constructed using a von Bertalanffy Growth fit (*VBLG*). The estimated growth curves are used in the demographic model.

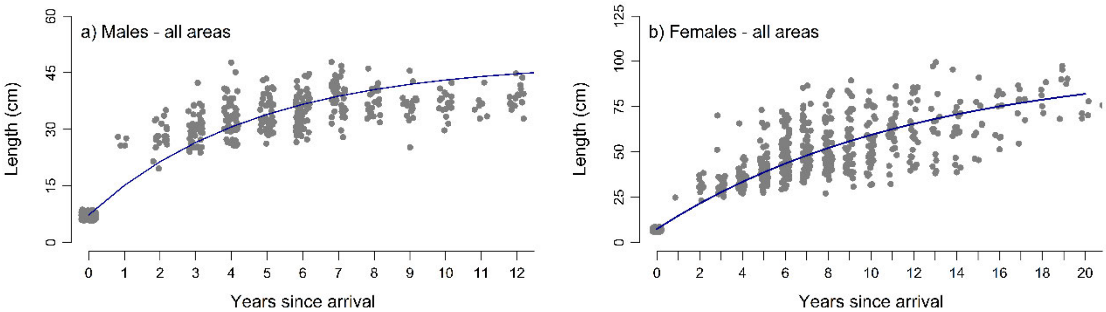

Figure 3-4 Eel growth. Dots are individual age (otolith) readings (age after arrival at the Dutch coast) at the time the eel was caught. Blue lines: estimated growth using a von Bertalanffy fit. a) Males (N=374), b) females (N=537).

3.6 Natural Mortality

Natural mortality is a difficult parameter to assess. It depends on many factors, such as predation, water temperature and food availability. Also mortality due to parasites and diseases can be classified under natural mortality. 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. In addition, it is unlikely that the natural mortality is the same over all age classes. In general, recruitment (glasseel, small yellow eel) is more vulnerable to predation and suffers higher natural mortality compared to older (larger) eel.

4 Spatial model

4.1 Introduction to the model

Only the main rivers (Rhine, Waal, Meuse and IJssel) and the large lakes (IJsselmeer, Markermeer, Grevelingen and Randmeren) and some canals are managed at a national level (*Figure 4-2*). All other water bodies are managed regionally by the water boards.

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 5*).

Except for these nationally managed large lakes, the standing stock of both regionally and nationally managed waters was estimated by a swept area estimate. This is a simple method where 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 from eel of at least 30 cm. Smaller eel are not considered, because the model has been developed to estimate the quantity of silver eels from the standing stock of yellow eels. Small eel (e.g. <30 cm) will not become silver eel in the year of the survey (*Figure 3-2*). The survey density was subsequently split 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*). It is not possible to directly measure the silver eel, because 1) maturity stage is often not recorded and 2) silver eel migrate to the sea, making it difficult to know if it grew up at the sampled location. In this chapter, the estimations of survey density for the regionally managed waters and for the nationally managed waters and subsequently silver eel biomass estimates will be presented. These estimations are used as input for the Dutch eel stock biomass estimation (*Chapter 7*).

4.2 Catch efficiency and habitat preference

Monitoring with the electric dipping net (see also paragraph 2.1) results in a monitoring result, also called catch success or catch per unit of effort (CPUE). This means that only a proportion of the eel present is caught with the monitoring, depending on the catch efficiency of the gear. With high catch efficiency, a high proportion is caught, with low catch efficiency a low proportion of the standing stock is caught with a specific gear.
The monitoring with the electric dipping net is at the shores (banks) only. Therefore, there is no information on the eel density off-shore, away from the banks in the open water. To translate the CPUE of the shores to total number in the waterbody (shore and off-shore), an EU certified protocol (STOWA Handboek Visstandbemonstering 2003) is used for the values of the catch efficiency of the survey gear

and for the differences in density of eel in the offshore area compared to the inshore area. According to the STOWA protocol, the catch efficiency is assumed to be 20%. The density of eel in the offshore area (> 1.5 m from the shore) compared to the inshore area is assumed to be 50% (see *Figure 4-1* for a schematic overview). However, there is no strong basis of these assumptions and a small amount of available literature shows that there is much variation for both assumptions (see next two paragraphs).

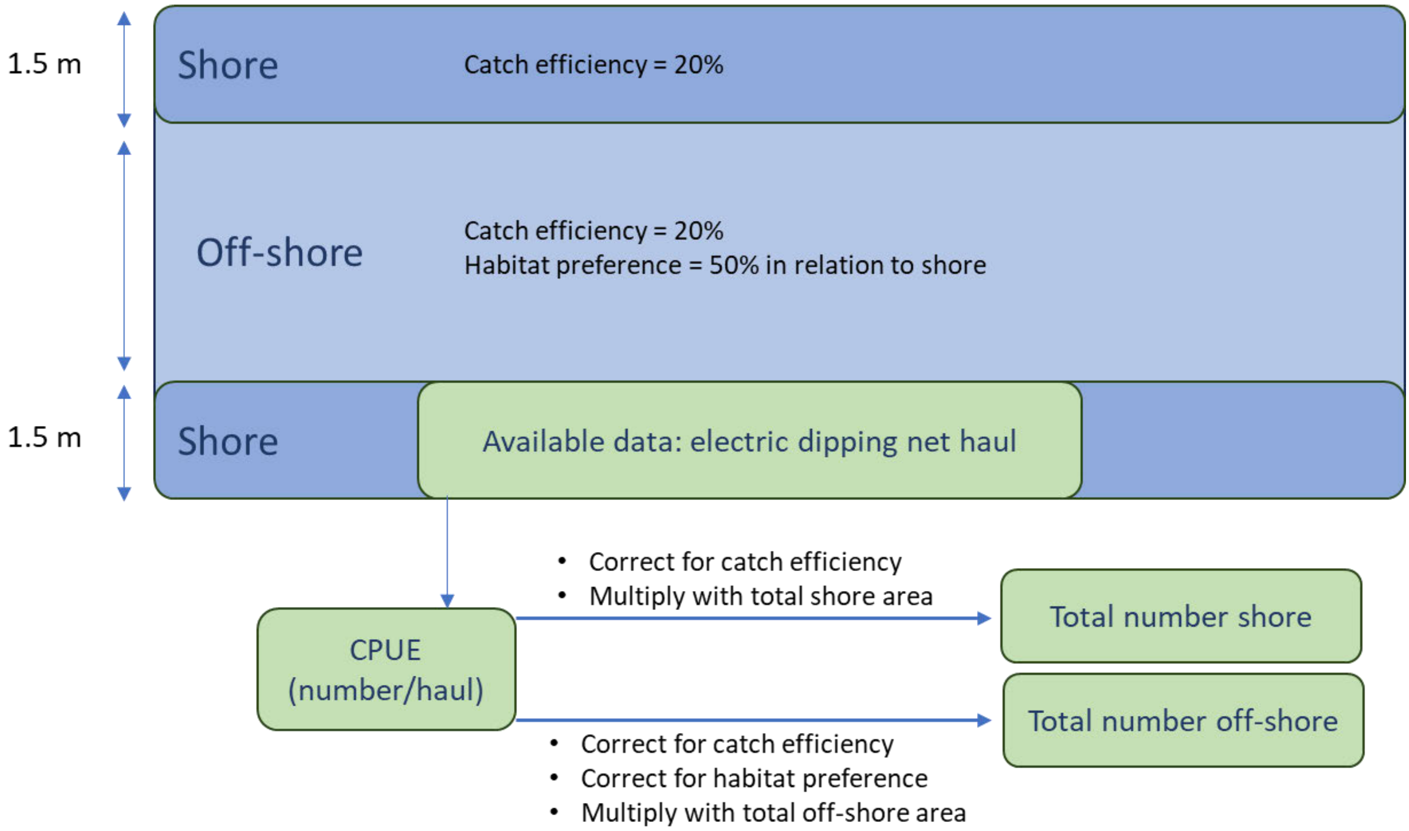

Figure 4-1 Schematic overview of the calculation from data to total numbers.

4.2.1 Catch efficiency – available information

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. 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%.

4.2.2 Habitat preference – available information

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 conversion factor 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. Different 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 differing 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. In the Netherlands the estimated eel densities in habitats that resemble lakes and rivers tend to be higher near shore compared to 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, but the further offshore, the lower the eel density (Stevens et al., 2013; Belpaire et al., 2018). In France, no difference is made between inshore and offshore areas in rivers (Briand et al., 2018).

4.3 Regionally managed water bodies

4.3.1 GIS data

The eel biomass in the regionally managed water bodies was assessed in the same way as presented in previous reports, based on the Water Framework Directive (WFD) fish monitoring program and detailed GIS maps (Bierman et al, 2012; van de Wolfshaar et al, 2015 & 2018; Van der Hammen et al, 2021).

In the Netherlands, the management of WFD waters is executed by 21 so-called water boards (*Figure 4-2*). All WFD surface waters are assigned to a waterbody type, ranging from small ditches to large lakes (*Table 4.1* & Appendix A2). 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 4-3*), which make it possible to calculate the surface area of each WFD water type (van Puijenbroek & Clement, 2010).

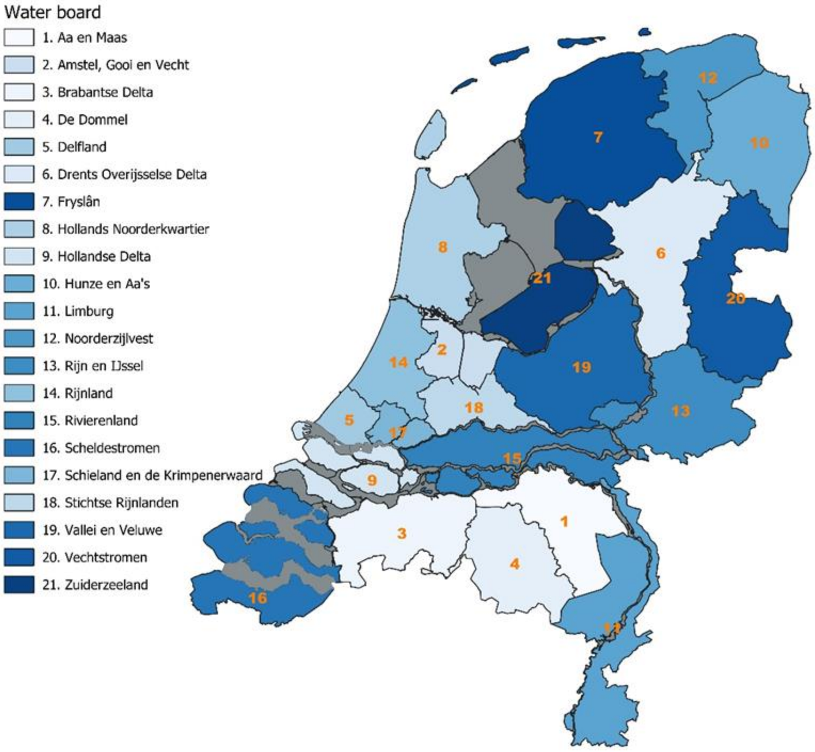

Figure 4-2 The 21 water boards in the Netherlands.

4.3.2 Data availability

WFD waters

Eel monitoring within the regionally managed WFD waters is executed with an electric dipping net, following an EU certified protocol (STOWA Handboek Visstandbemonstering 2003). Water boards (*Figure 4-2*) are obliged to sample their WFD waters at least once every 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 within a water board is thus not expected on a yearly basis. To reduce the variation due to unbalanced sampling, a moving average of six-year periods was used to assess the biomass of eel for each three-year assessment period. 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 for each three-year period. Because data from before 2006 was not available, for the first assessment period (2006-2008) data from 2006-2011 was used. Similarly, 2023 and 2024 data were not yet available from any water board due to the timing of the data request (spring-summer 2023). Therefore, for the last assessment period (2021-2023), a six-year period starting in 2017 (2017-2022) was used. Some WFD 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 their monitoring was predominantly executed with fykes, as electrofishing is difficult in brackish or salt water. To include data from water board Scheldestromen, pre-processed data based on fyke monitoring was used to at least include data from this water board instead of having an area (water board) with missing data. Fyke effort was converted to swept area by taking a 100 m buffer of either side of the fyke and multiply this with the width of the waterbody. For water bodies having a larger width than 20 m, the width was set to a maximum of 20 m. Fyke effort was standardized to two nights of fishing whereby a catch efficiency of 60% is assumed.

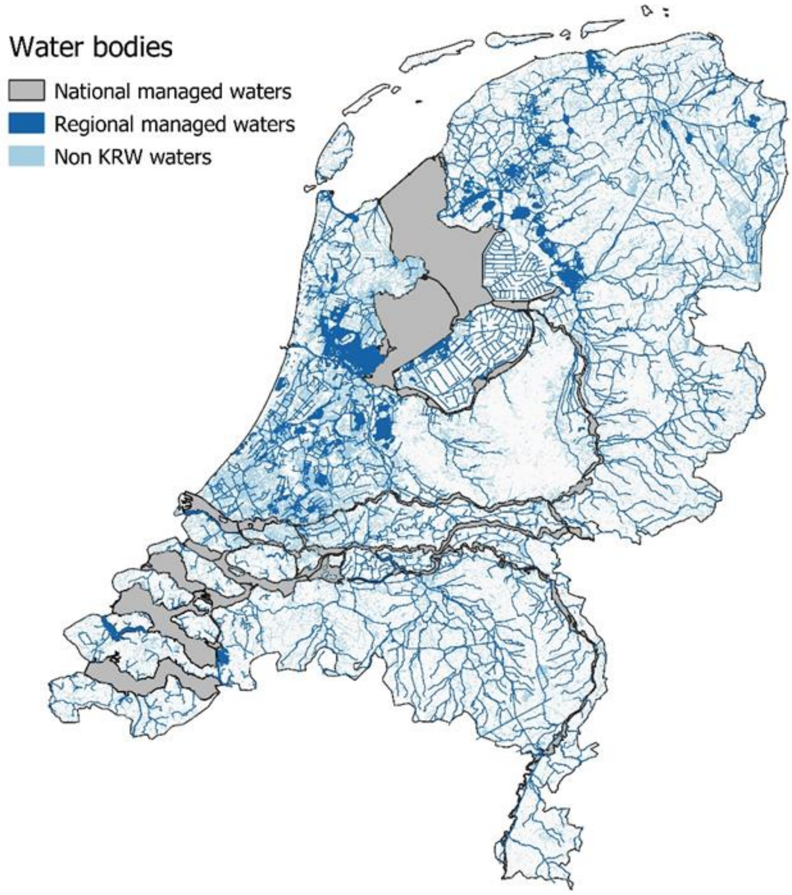

Figure 4-3 The regionally managed WFD waterbodies (dark blue), Non-WFD waters (light blue) and the nationally managed water (grey).

To link the geographic coordinates of the electrofishing sampling locations to the WFD waterbodies in the available GIS map, the coordinates which fell into a polygon (waterbody) were assigned to that polygon. The fishing events which could not be assigned to a polygon, were assigned to the nearest polygon within a margin of 50 meters from the sampling location. For the remaining sampling locations without a match, based on the above-mentioned statements, the waterbody identification code[4] 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 are in use for a single water body, and they sometimes change over time (e.g. after a fusion between waterboards). The remaining fishing events after this last step were excluded from the analyses as information on which waterbody they represent is lacking. Finally, only the fishing events for which the effort (swept area) was known, were used in the analysis. In total, the selection method resulted in the inclusion of 13,307 electrofishing events in 650 WFD water bodies (out of 701 defined based on the available GIS map) in the eel assessment. Note that sampling intensity was lower in earlier years than in the more recent years (Appendix A1). For example, in the first period (2006-2011) only 379 WFD water bodies were sampled, while in the last period (2017-2022) 608 WFD water bodies were sampled out of a total of 701 different WFD water bodies assigned within this assessment.

The variability in the total swept area is large between the different water types (*Table 4.1*, see *Appendix A2* for a description of each water type). The two water types with the largest surface area (shallow, relatively large lakes, M14 and M27, with 58% of the total surface area) have a relatively low sampling

[4]Unique name or code which is applied in the GIS map to each waterbody.

intensity. The relative highest sampling intensity (buffered canals, M3, with 15% of the sampling effort and slow flowing lower streams, R5, with 25%) occurred in water types with a relatively small surface area (4% and 2%, respectively). Nevertheless, most of the small ditches (M1a and M2) are not assigned as a WFD water body and were therefore not included in the WFD sampling program. Those non-WFD ditches cover a large area of > 59,000 hectares (*Table 4.1*, '*Non-WFD ditches'*) and can altogether contribute significantly to the standing stock of eel. To include these non-WFD ditches, an additional fish (eel) sampling ("*Polderbemonstering*") was carried out in these water bodies to estimate the total biomass of eel in non-WFD ditches (van Keeken, 2014a & 2014b; Volwater et al, 2022).

4.3.3 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 water board. Most small ditches can be found in the lower parts of the Netherlands ("Polders"). Therefore, some waterboards with very few ditches are not sampled, resulting in that 15 out of the 21 water boards were included within this additional sampling program. In total, 424 electrofishing events were executed, whereby an area of 13.9 hectares was sampled in non-WFD ditches 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.

4.3.4 Standing stock estimation

To convert eel numbers to biomass, eel lengths were converted to weights with a length-weight relationship (*Figure 4-4*). Subsequently the catch per unit of effort in biomass (CPUE, kg/ha) was calculated. Mean CPUE per waterboard per assessment period are shown in *Figure 4-4* to indicate possible trends, for mean CPUE per WFD water type see *appendix A4*. Thereafter, CPUE was corrected for the assumed catch efficiency of the electric dipping net (20%). 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 50% of that in the littoral zone. Subsequently, CPUE is converted to absolute biomass (kg) for the riverbank and open water surface areas separately.

For upscaling to the total biomass in regional waters, the surface area of each water body 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 WFD water body. Based on 1) the female:male ratio at length (*Figure 3-1*) and 2) the maturity at length for both males and females (*Figure 3-2*), the density and biomass of silver eel was estimated. For water bodies that were not sampled in a six-year period biomass of eel and silver eel was estimated following different steps for these waters without data; 1) density averaged over the same water type within that waterboard or 2) density averaged over that water board. For the additional sampling in ditches, production and total biomass of eel and silver eel within non-WFD waters was estimated per water board. For water boards with no sampling effort within non-WFD waters, the mean density of eel and silver eel within these waters of neighbouring water boards was taken.

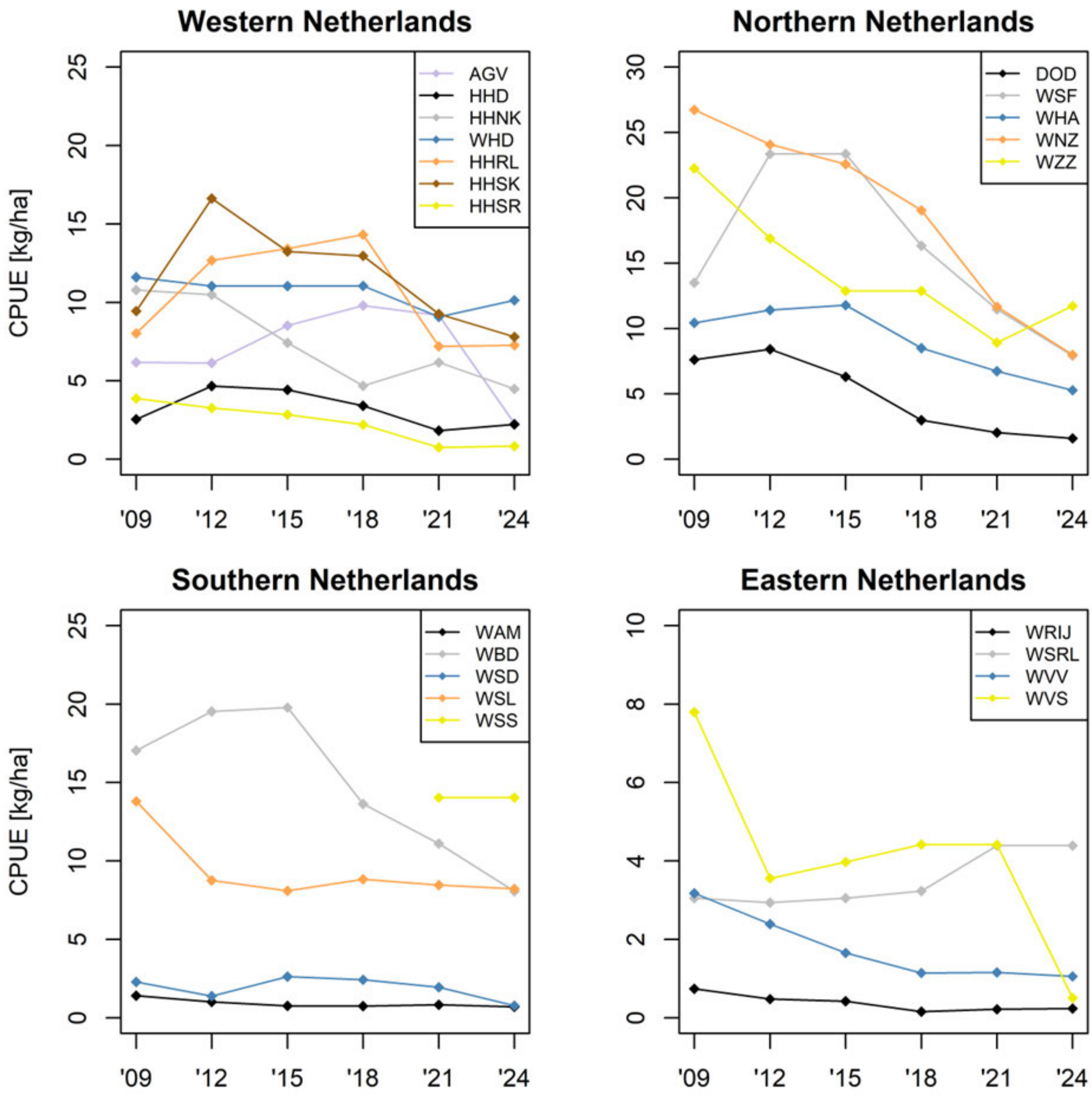

Figure 4-4 Catch per unit of effort (CPUE) per waterboard that have been sampled each period. Waterboard are grouped into four categories based on cardinal directions: West, North, South and East. The year on the x-axis indicate each assessment period and represents data of six years; "09": data from 2006-2011; "12" data from 2009:2014; "15": data from 2011:2016; "18": data from 2013-2018; "21": data from 2015:2020; "24": data from 2017:2022. Note that the y-axis differs for the plots of the four categories. ***West****; AGV = Amstel, Gooi & Vecht, HHD = Delfland, HHNK = Hollands Noorderkwartier, WHD = Hollandse Delta, HHRL = Rijnland, HHSK = Schieland en de Krimpenerwaard, HHSR = Stichtse Rijnlanden.* ***North****; DOD = Drents Overijsselse Delta, WSF = Fryslân, WHA = Hunze en Aa's, WNZ = Noorderzijlvest, WZZ = Zuiderzeeland.* ***South****; WAM = Aa en Maas, WBD = Brabantse Delta, WSD = De Dommel, WSL = Limburg, WSS = Scheldestromen.* ***East****; WRIJ = Rijn en IJssel, WSRL = Rivierenland, WVV = Vallei en Veluwe, WVS = Vechtstromen.*

4.3.5 Standing stock per WFD water type

The density and biomass of eel and silver eel per water type was estimated for all defined assessment periods of six-year, so that each estimate covers a full sampling cycle of six years. The result of the latest six-year period (2017-2022) is presented in *Table 4.1*. A total biomass of 1,471 tonnes of eel (≥ 30 cm) was estimated, of which 261 tonnes silver eel, in regionally managed WFD waters. The highest survey CPUE's were estimated for fast flowing small rivers (R15) (38.6 kg/ha) and large shallow peatland lakes (M27) and fast flowing lower stream (R14) (both 13.4 kg/ha). The surface area of R14 and R15 waters is very small and the contribution to the total biomass is thereby limited. Contribution of large lakes (M14, M27 and M20) 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 4.1*). The estimated biomass of eel in WFD waters (1,471 tonnes) combined with an estimated biomass of 828 tonnes (survey CPUE 3.2 kg/ha) for eel in non-WFD waters resulted in a total estimated biomass of 2,297 tonnes of eel in regionally managed waters for the period 2017–2022. The total biomass estimate of silver eel in regionally managed waters for the period 2017–2022 was 409 tonnes (*Table 4.1*).

4.3.6 Biomass per period

There are large differences in the biomass estimates between the assessment periods (*Table 4.2*). The biomass estimates for the two most recent assessment periods were lowest, while the highest estimates were seen for the periods 2009–2011, 2012-2014 and 2015-2017. Although the estimates are made for a six-year period, there can still be large differences in the circumstances during the sampling, which can influence the monitoring result. For example sampling can occur on a different location within the same water body, by a different person or the water level or temperature may be different, which will cause variation. In addition, although six years of data (2006-2011) was used for the estimate of the first period (2006-2008), the sampling effort compared to the sampling effort in the later periods was still much lower. This was not the case in the latest periods, where the six-year periods had nearly an equal sampling effort compared to the preceding periods (Appendix A1).

4.3.7 Discussion

To estimate the biomass in each regional managed water, sampling in each waterbody with sufficient spatial and temporal coverage is needed, which is not always the case. Firstly, water boards are obliged to sample their WFD waters only once within a time frame of six years. To cover this low temporal coverage, in each 3-year period, the nearest 6 years of data is included. Secondly, especially in the first period, there was less sampling effort, which will have influenced the results. Thirdly, the sampling intensity is not well-balanced between water types. Water types with the highest surface areas have relatively low sampling effort with the electronic dipping net, while the highest sampling effort was performed in water types with relatively (very) small surface areas. Within the WFD fish monitoring program, monitoring is also conducted with other types of "active" fishing gears, like trawls or (beach) seines. Yet, none of these fishing gears is assumed to monitor eel in a reliable manner. In addition, trawling and seining is only conducted in large water bodies (lakes) and thus not conducted for example in flowing streams/rivers WFD waters. Passive fishing gears like fykes, are also not very suitable to monitor eel, because they rely on the activity of the target species and therefore it limits density and biomass estimates for a surface area. Despite, water board Scheldestromen (the only one out of 21) includes fykes to estimate density estimates of eel since electro fishing is restricted by brackish waters in their management area. In general, comparing different fishing methods rises complications due to gear specific selectivities (e.g. length, maturity, activity).

Table 4.1 Estimation of the eel mean CPUE and biomass per WFD water type for the period 2017 – 2022. CPUE and biomass estimates were done for yellow and silver eel combined (≥ 30 cm) and for silver eel only. For a full description of the WFD-water types, see Appendix A2. Catch per unit of effort (CPUE) represents the 'catch success' in the survey and is shown as the total mean per water type.

				All eel (≥ 30 cm)		Silver eel (≥ 30 cm)	
WFD water	**Description of WFD water**	**Area [ha]**	**Swept area [ha]**	**CPUE [kg/ha]**	**Biomass [ton]**	**CPUE [kg/ha]**	**Biomass [ton]**
M1a	Buffered ditches	319	20.4	1.4	0.9	0.3	0.2
M1b	Brackish buffered ditches	36	3.5	9.5	1.0	1.5	0.2
M2	Weakly buffered ditches	27	0.8	4.3	0.4	2.2	0.2
M3	Buffered canals	4,541	76.5	2.8	74.2	0.6	15.5
M6a	Large shallow canals (shipping)	695	9.4	4.1	11.0	0.9	2.6
M6b	Large shallow canals	3,600	15.0	5.1	52.0	1.3	10.7
M7b	Large deep canals	5,235	20.5	9.6	161.1	1.9	27
M8	Buffered peatland ditches	349	18.9	0.3	0.6	0.1	0.1
M10	Peatland canals	4,545	44.6	1.3	22.3	0.3	4.9
M12	Shallow, small, buffered lakes	26.0	0.0	8.1*	0.6	1.7	0.1
M14	Shallow, large buffered lakes	17,762	42.2	7.6	451.6	1.5	72.1
M20	Deep, large, buffered lakes	5,094	6.0	11.5	167	2.1	31.2
M23	Shallow, large calcium rich lakes	399.0	1.4	0.0	0.0	0.0	0.0
M27	Shallow, large, peatland lakes	10,691	19.8	13.4	135.6	4.0	30.7
M30	Weakly brackish waters [0.3-3 g Cl/l]	9,298	15.1	10.6	250.2	2.0	33.5
M31	Small brackish	625.0	0.2	14.0	9.7	3.0	2.0
R4	Slow flowing, upper stream on sand	315.0	23.1	1.3	1.4	0.4	0.4
R5	Slow flowing, lower stream on sand	2,130	129.3	0.9	8.6	0.2	1.8
R6	Slow flowing small river on sand/clay	2,147	50.4	7.9	66.9	1.7	16
R7	Slow flowing side stream on sand/clay	1,141	7.6	8.2	19.2	1.3	2.9
R8	Fresh tidal waters on sand/clay	4,097	5.6	11.6	26.4	2.6	5.9
R12	Slow flowing lower stream on peat	100	7.3	1.5	0.3	0.3	0.1
R13	Fast flowing upper stream on sand	12.0	1.1	0.0	0.0	0.0	0.0
R14	Fast flowing lower stream on sand	18.0	2.3	13.4	1.6	3.3	0.4
R15	Fast flowing small river (siliceous)	56.0	0.4	38.6	6.1	8.2	1.3
R17	Fast flowing upper stream (calcium)	83.0	1.5	0.0	0.0	0.0	0.0
R18	Fast flowing lower stream (calcium)	83.0	2.7	6.4	2.3	1.7	0.7
Regionally managed WFD waters		73,422	525.6		1,471		261.5
Non-WFD waters (M1a)		59,441	13.9	3.2	827.6	0.6	148.8
Total		132,863	539.5		2,298.6		409.3

* *For those water types where no survey was conducted, the average survey CPUE of the water boards in which these water bodies are located is shown.*

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 because some water bodies are not included in the GIS map.

As in previous reports, a catch efficiency of 20% of the electric dipping net and spatial distribution of eel within a habitat (50%, "offshore" compared to "inshore") was used to calculate standing stock biomass in regionally managed waters. These assumptions are quite arbitrary and cause a high level of uncertainty in the absolute biomass estimate. Especially the offshore density assumption has a high contribution to the absolute biomass estimate in large water bodies (i.e. lakes: M14, M30 & canals: M7b, *Table 4.1* & *Appendix A3*). Finally, variation in the biomass estimates may also be a result of stocking activities, which cause large variation in eel biomass in the waterbodies were eel is stocked in unequal numbers in time.

In the non-WFD waters (ditches), the sampling scheme was standardized for sampling method, but not for sampling location (*Paragraph 5.3.3*). Each year, two (or three) different waterboards were selected and within a given waterboard only a very small subsample of all ditches was monitored. As a result, only one single estimate of the non-WFD waters per water board 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.

Table 4.2 Estimates of standing stock of eel in tonnes in the regionally managed waters (WFD water bodies) and non-WFD ditches; all eel (yellow and silver ≥30cm) and silver eel (≥30cm) biomass estimates for six periods (and all years combined for the Non-WFD waters).

	Non-WFD water	**WFD water bodies**					
	All years	**2006-2008***	**2009-2011***	**2012-2014***	**2015-2017***	**2018-2020***	**2021-2023***
All eel	828	2,066	2,572	2,568	2,551	1,807	1,471
Silver eel	149	319	438	527	506	320	261

** 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 2009:2014; Period "2012-2014": data from 2011:2016; Period "2015-2017": data from 2013-2018; Period "2018-2020": data from 2015:2020; Period "2021-2023": data from 2017:2022.*

4.4 *Nationally managed water bodies: Large rivers*

4.4.1 Data availability

Within the survey program "Fish Monitoring National Waters", fish species in the large Dutch rivers are monitored yearly with an electric dipping net at the riverbanks (van Keeken et al., 2023). Depending on the sampling location, the monitoring takes place in spring (March and/or April) or autumn (October and/or November). An exception is the sampling location 'Noordwaard', which is monitored at the end of August/beginning of September (*Table 4.3*). Most sampling locations are monitored yearly, but not all (e.g. 'Volkerak'). All sampling locations are measured at least once in each 3-yearly period (*Table 4.3*). There are also sampling locations where monitoring started after the first period considered here (2006-2008, *Table 4.4*), and only data from more recent periods is available (e.g. monitoring on sampling location 'Afgedamde Maas' started from the period 2009-2011). For these sampling locations, the oldest available data is used for previous periods without data (e.g. data from the period 2009-2011 of sampling location 'Afgedamde Maas' is used for the period before, 2006-2008). See *Figure 4-5* for the classification of the main rivers and *Table 4.3* and *Table 4.4* for an overview of survey details per water body.

Table 4.3 Survey information per WFD water body, for the years 2021, 2022 and 2023. Sampled years = the years in which a region has been sampled, where all = 2021+2022+2023. Survey density is based on data collected using an electric dipping net at the riverbanks. For this table, no correction for the catch efficiency of the gear is made.

WFD water body	Sampled years	Sample period	Survey density (≥30 cm all eel) in riverbank (kg/ha)	Survey density (≥30 cm silver eel) in riverbank (kg/ha)
Bedijkte Maas	all	Spring	1.34	0.49
Beneden Maas	all	Autumn	1.41	0.38
Bergsche Maas	all	Autumn	1.10	0.29
Biesbosch (Brabantse, Noordwaard)	2021, 2022	August	0.89	0.15
Biesbosch (Dortsche) en Nieuwe Merwede	all	Autumn	28.92	6.44
Bovenmaas	all	Spring	0.90	0.29
Grensmaas	all	Spring	0.90	0.29
Hollands Diep	all	Autumn	5.71	1.23
IJssel	all	Spring	2.07	0.50
Merwede	all	Both*	11.15	2.44
Nederrijn/Lek	all	Spring	1.83	0.52
Oude Maas	all	Autumn	6.81	1.41
Volkerak	2022	Autumn	2.17	0.4
Waal, Boven-Rijn	all	Spring	3.73	1.04
Zandmaas	2021, 2022	Spring	1.36	0.36
Zoommeer/Eendracht	2022	Autumn	4.30	0.79

* The survey density of eel in the waterbody Merwede is calculated by the average of four sampling locations, three of them were sampled in autumn, and one in spring.

In previous evaluation reports regions were used that consisted of several WFD waters and some WFD waters were part of two different regions (Van der Hammen et al., 2021). For this report the biomasses in each separate WFD water (here called a water body) are calculated. In most water bodies there is at least one sampling location within the water body, but some water bodies are not sampled (e.g. 'Boven Maas'). For these water bodies, data from sampling locations in adjacent water bodies is used (e.g. for water body 'Boven Maas', the adjacent sampling location 'Grensmaas' is used). The survey density (number of eel caught/hectare) in each water body and period is calculated by first calculating the average survey density at each sampling location and thereafter the average of the sampling locations in each WFD water body.

Table 4.4 Number of hauls per year per sampling location.

Sampling location	WFD Water body	2006-2008	2009-2011	2012-2014	2015-2017	2018-2020	2021-2023
Afgedamde Maas	Beneden Maas	0	13	15	14	13	13
	Merwede						
	Bergsche Maas						
Bedijkte Maas	Bedijkte Maas	7	27	9	18	18	26
Benedenloop Gelderse IJssel	IJssel	21	19	20	21	21	23
Benedenloop Nederrijn	Nederijn/Lek	0	0	8	12	12	12
Benedenloop Waal	Merwede	0	0	14	25	26	27
	Bovenrijn Waal						
Bovenloop Gelderse IJssel	IJssel	44	61	39	31	27	31
Bovenloop Nederrijn	Nederrijn/Lek	24	23	16	12	12	12
Bovenloop Waal	Bovenrijn Waal	34	41	28	21	20	21
Getijden Lek	Oude Maas	29	30	31	30	30	30
Getijden Maas	Beneden Maas	36	36	36	35	36	36
	Bergsche Maas						
Grensmaas	Boven Maas	35	36	35	36	35	36
	Grensmaas						
Heusdensch kanaal	Beneden Maas	0	2	3	3	3	3
Hollandsch diep	Hollandsch diep, Haringvliet oost	60	59	30	30	30	29
Nieuwe Merwede	Merwede	21	21	21	21	21	21
	Dortsche Biesbosch en Nieuwe Merwede						
Noordwaard	Brabantse Biesbosch	0	0	0	24	36	24
Oude Maas	Merwede	15	15	15	15	15	14
	Oude Maas						
Rijn	Bovenrijn Waal	19	22	16	12	8	12
Volkerak	Volkerak	9	9	9	9	9	9
Zandmaas	Zandmaas	11	33	12	16	24	17
Zoommeer	Zoommeer/Eendracht	0	0	0	12	18	6

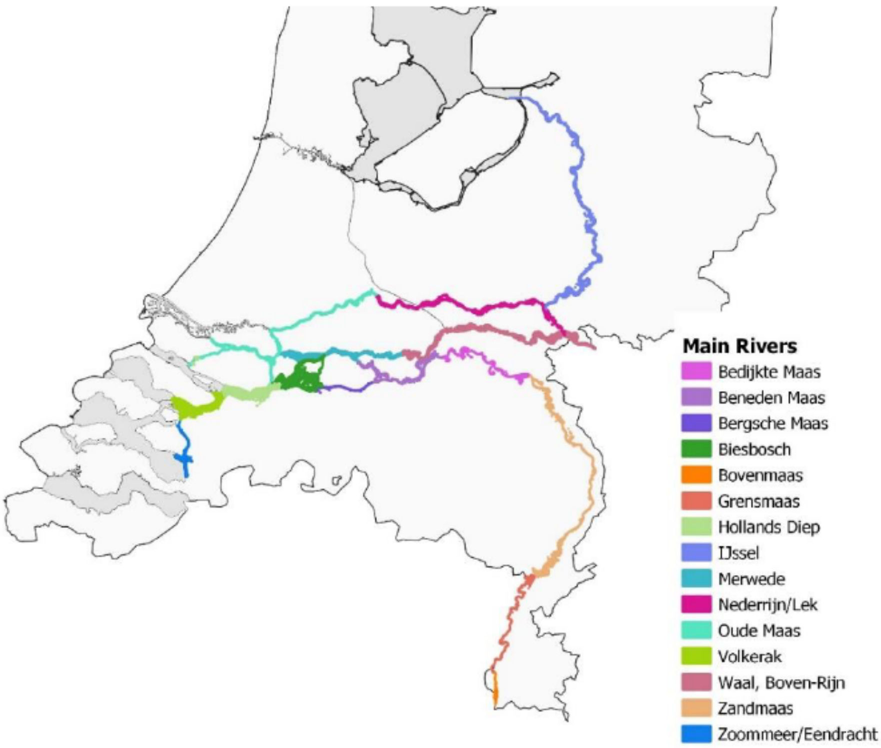

Figure 4-5 Classification of the WFD water bodies in the large (main) rivers. WDF water bodies are represented by different colors.

4.4.2 GIS data

Two types of geographical information were collected: surface area and riverbank length. For each water body, the surface area (ha) and the riverbank length were calculated from shapefiles in QGIS (Puijenbroek & Clement, 2010, *Table 4.5*). While in previous reports different areas were used, this report used WFD waterbodies, as more detailed information is now available for these water bodies (e.g. groins are included in the shapefiles and therefore also in the shore length).

Table 4.5 Surface area and bank length per water region.

WFD Water body	Surface area (ha)	Riverbank length (hm)
Bedijkte Maas	1,294	1,811
Beneden Maas	1,657	2,349
Bergsche Maas	515	607
Biesbosch	3,487	5,993
Bovenmaas	344	448
Grensmaas	873	1,581
Hollands Diep	4,316	1,766
IJssel	2,834	6,725
Merwede	2,585	4,368
Nederrijn/Lek	2,508	5,341
Oude Maas	3,288	4,621
Volkerak	4,705	2,554
Waal, Boven-Rijn	4,698	6,525
Zandmaas	2,734	4,265
Zoommeer/Eendracht	1,279	1,105

4.4.3 Biomass estimate

Biomass in each WFD water body was calculated from survey densities by first correcting for the assumed catch efficiency of the electric dipping net (20%, see paragraph 4.2.2). Next, the water surface area was divided into two areas: the riverbank (littoral zone, 1.5m, the reach of the electric dipping net*bank length) and open water. The open water surface area is the total surface area minus the surface area of the shore. Eel density outside the shore is assumed to be half (50%) of that of the shore (see paragraph 4.2.2). Subsequently, density is converted to absolute biomass for the shore and off-shore surface areas separately. For the Grensmaas, no correction for habitat preference is made and density off-shore is assumed to be equal to that in the shore because sampling with the dipping net takes place in both the open water in this (shallow water) region and along the shore and is thus representative for the total water body. The estimated biomass of all eels (≥ 30 cm), yellow eel (≥ 30 cm) and silver eel (≥ 30 cm) of the latest period (2021-2023) can be found in *Table 4.6*.

For the latest period, most of the biomass can be found in the waterbodies 'Dortsche Biesbosch en Nieuwe Merwede' (726 tons) 'Merwede' (495 tons), 'Hollandsch Diep, Haringvliet oost' (421 tons) and Oude Maas (384 tons). The estimated biomass of eel ≥ 30 cm (yellow and silver) in the period 2021-2023 is also compared to earlier periods. Comparing the latest period to the period 2018-2020, the biomass increased for all waterbodies, except for Volkerak (*Table 4.7*). Compared to the period 2015-2017, in some waterbodies the biomass was higher, while in other waterbodies the biomass was lower (*Table 4.7)*. Due to the more detailed shapefiles we used for the current report, resulting in a smaller water surface area, the total estimated biomass is lower than estimates in previous reports.

Table 4.6 Biomass (tons) of all eel ≥ 30 cm, yellow eel (≥ 30 cm) and silver eel (≥ 30 cm) per waterbody for 2021-2023.

	Biomass (tons)		
WFD water body	**All eel (≥ 30 cm)**	**Yellow eel (≥ 30 cm)**	**Silver eel (≥ 30 cm)**
Bedijkte Maas	29.5	29.4	10.9
Beneden Maas	39.9	39.8	10.7
Bergsche Maas	9.6	9.6	2.5
Biesbosch (Brabantse Noordwaard)	31.2	30.7	5.1
Biesbosch (Dortsche) en Nieuwe Merwede	725.7	723.7	161.0
Bovenmaas	5.3	5.3	1.7
Grensmaas	26.4	26.3	8.5
Hollandsch Diep, Haringvliet oost	420.7	412.9	89.0
IJssel	101.4	101.1	24.5
Merwede	494.5	492.6	107.8
Nederrijn/Lek	79.0	78.8	22.2
Oude Maas	384.1	380.9	78.9
Volkerak	177.9	171.8	31.7
Waal, Boven-Rijn	298.5	298.4	83.1
Zandmaas	63.5	63.3	16.6
Zoommeer/Eendracht	97.7	92.8	17.0

Table 4.7 Biomass of eel ≥ 30 cm (yellow and silver) in tons per waterbody, for each 3-year period.

WFD water body	**2006-2008**	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**	**2021-2023**
Bedijkte Maas	43.0	54.2	9.1	53.2	7.0	29.4
Beneden Maas	5.8	3.9	31.5	64.7	10.2	39.8
Bergsche Maas	2.7	1.8	7.7	26.1	3.0	9.6
Biesbosch (Brabantse Noordwaard)	6.2	6.2	6.2	6.2	9.5	30.7
Biesbosch (Dortsche en Nieuwe Merwede)	22.8	12.2	34.7	216.4	92.6	723.7
Bovenmaas	17.7	32.8	5.3	7.8	1.9	5.3
Grensmaas	88.2	163.0	26.3	38.8	9.4	26.3
Hollandsch Diep, Haringvliet oost	66.7	142.7	305.4	359.7	264.2	412.9
IJssel	23.4	5.1	8.6	44.3	9.2	101.1
Merwede	37.1	28.5	58.4	271.7	75.8	492.6
Nederrijn/Lek	4.0	15.7	5.1	48.0	3.7	78.8
Oude Maas	32.7	41.5	92	220.6	121.9	380.9
Volkerak	130.5	871.2	378.2	98.4	409.5	171.8
Waal, Bovenrijn	40.1	50.8	56.1	176.6	0.0	298.4
Zandmaas	39.6	78.4	36.3	251.0	55.6	63.3
Zoommeer/Eendracht	51.8	51.8	51.8	51.8	15.2	92.8
Total	**612.3**	**1559.8**	**1112.7**	**1935.3**	**1088.7**	**2957.4**

When combining all the water bodies, the estimated biomass of eel ≥ 30 cm (yellow eel & silver eel) for the period of 2021-2023 is the highest (2957 tons), followed by the period 2015-2017 (1935 tons) and 2009-2011 (1560 tons) (*Table 4.8*).

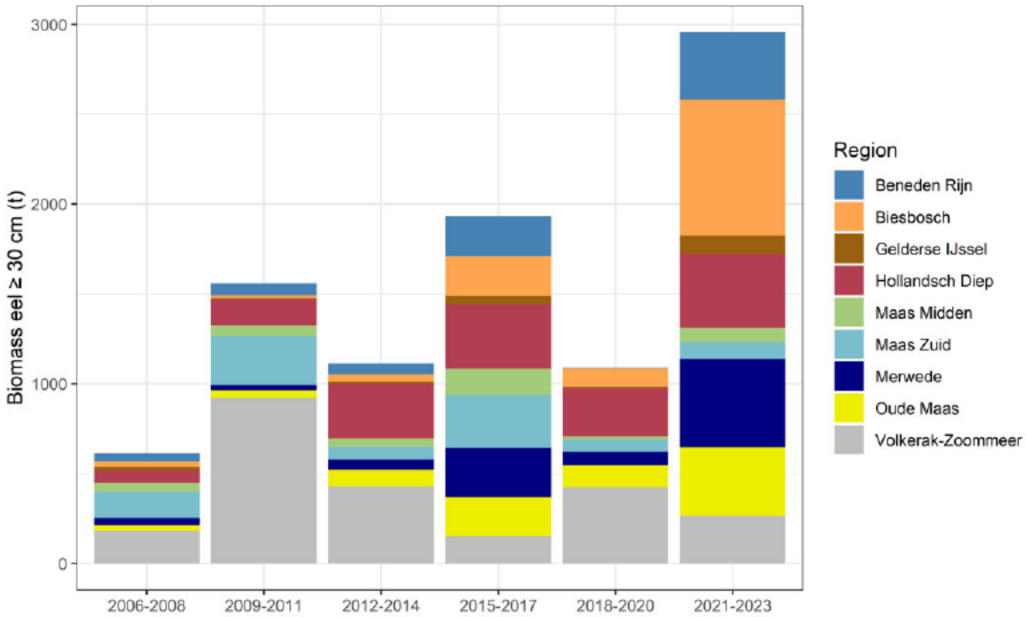

Figure 4-6 Biomass of eel (yellow and silver, ≥ 30 cm) in tonnes per river region, for each 3-year period. The regions consist of the following waterbodies: Beneden Rijn = 'Nederrijn/Lek' & 'Bovenrijn Waal', Biesbosch = 'Brabantse Biesbosch Noordwaard' & 'Dortsche Biesbosch Nieuwe Merwede', Gelderse IJssel = 'IJssel', Hollandsch Diep = 'Hollandsch Diep, Haringvliet oost', Maas midden = 'Bedijkte Maas' & 'Beneden Maas' & 'Bergsche Maas', Maas Zuid = 'Bovenmaas' & 'Grensmaas' & 'Zandmaas', Merwede = 'Merwede', Oude Maas = 'Oude Maas', Volkerak-Zoomeer = 'Volkerak' & 'Zoommeer/Eendracht'.

Table 4.8 Biomass of all eel, eel ≥ 30 cm (yellow and silver eel) and silver eel (≥30 cm) in tonnes for each 3-year period.

	National water bodies					
	2006 – 2008	**2009 – 2011**	**2012 – 2014**	**2015 – 2017**	**2018 – 2020**	**2021 – 2023**
All eel	629	1,635	1,125	1,961	1,097	2,985
Eel (yellow & silver, ≥ 30 cm)	612	1,560	1,113	1,935	1,088	2,957
Silver eel (≥ 30 cm)	114	214	187	403	249	671

4.5 Discussion regionally and nationally managed waters

Concerning the silver eel biomass estimate of the nationally and regionally managed waters, there are some uncertainties of which the most important are:

- The catch efficiency of the gear used for the monitoring is unknown and might also differ per sampling location. Catch efficiency is a crucial parameter for the translation of a relative catch success to an absolute biomass estimate. Any difference between the true and assumed catch efficiency will cause an error of the same magnitude in the biomass estimates.
- Little is known about the eel distribution between the shore area's (riverbanks, littoral zone) and open water. Eel density is known to be lower in open, deeper water, but the exact proportion is unknown and probably differs per location. As electric dipping net monitoring is usually along the riverbank, the translation to the density in the open water originates on the STOWA (2003) protocol, which is based on assumptions (paragraph 4.4.3). This has particularly strong effects on larger water bodies, where relatively large parts of the waterbodies consist of open water.
- The monitoring is not always consistent in time per region: some regions are not sampled on a yearly basis and some regions are sampled in different months. This causes additional variation in the outcome.

Regarding the habitat distribution of eel, there is some data available on this matter based on the Dutch monitoring. In addition to the electric dipping net monitoring, the nationally managed water bodies are also monitored using a beam trawl, at the same time the electric dipping net monitoring occurs. Assuming an equal catch efficiency, which is a very large and uncertain assumption, the CPUE of the beam trawl is about ~10-fold smaller than for the electric dipping net in the large rivers of the Netherlands. Using the same assumption, the CPUE of the beam trawl for lakes IJsselmeer and Markermeer combined is ~38-fold smaller than that of the electric dipping net. These numbers seem to indicate that eels have a much higher density inshore than in the open water than assumed in this report (ratio 1:2). We therefore strongly recommend that further research into this matter as detailed information on the habitat distribution will increase the accuracy of the biomass estimates.

5 Demographic model

5.1 Demographic model

A different method is used for the nationally managed larger lakes (lakes IJsselmeer, Markermeer, Grevelingen and Randmeren) than for the smaller waterbodies (*Chapter 5*). Sampling along the shore is conducted with an electric dipping net, 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 shore and FYMA (used to sample the offshore waters of the lakes) surveys, is unknown. Instead of using the survey data to compute a direct estimate of stock biomass, a demographic model was applied to estimate the eel biomass in the larger lakes IJsselmeer, Markermeer, Randmeren and Grevelingen. The demographic model was used to estimate fishing mortality in the lakes IJsselmeer and Markermeer by fitting the model to relative changes in abundances per age class observed in the survey (see overview *Paragraph 1.2)*. Subsequently, the estimated fishing mortality was used to calculate the biomasses based on the eel landings in the lakes (see *Paragraph 5.6*). The results were used as input to estimate the total Dutch eel stock biomass (*Chapter 7*). In addition, the demographic model was also used in the calculation of one of the stock indicators (*Chapter 7*). In that case, the demographic model was not parameterized for lakes IJsselmeer and Markermeer (*Chapter 7*).

The demographic model assumes a closed system for the freshwater phase, similar to other models 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 demographic model tracks annual eel cohorts through time, for eel from 1989 until 2023. 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, van der Hammen et al. 2021). The changes that were made compared to the last assessment (Van der Hammen et al., 2021) are described in *Appendix B1*. 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.

The estimates of fishing mortality depend 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. Changes towards earlier maturation would lead 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 the fisheries mortality (see also Bierman et al., 2012, van de Wolfshaar et al., 2015 & 2018 and Van der Hammen et al. 2021).

5.2 Model parameters

All parameter values can be found in in *Appendix B1*. Parameterization is based on the biological keys (*Chapter 2*). Recruitment in the model is based the abundance of year class 2. Per age and sex class, the length at the mid-age of the age class is used to derive the probability of maturing and the selectivity of the fishery.

5.3 Model fitting

All details with regards to the model fitting can be found in in *Appendix B1*. No obvious trend is visible in the total number of individuals caught in the FYMA survey from 2006-2023 (*Figure 5-1*). 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 5-2*). Generally, abundance decreases with age. From year 7 after arrival, only a few individuals per age class were observed in the FYMA survey. This introduces large uncertainties in the estimated CPUE. Therefore, the age class 6 was used as the last age class to fit to. The fisheries mortality is estimated for six different periods (2006-2008, 2009-2011, 2012-2014, 2015-2017, 2018-2020, and 2021-2023). The selection of the time periods was based on the period of the assessments.

5.4 Model fit and estimated fishing mortality

The model predictions and the data on the FYMA catches (number per trawled km^2 per age class) are presented in *Figure 5-3*. As expected, the eel abundance decreases with age in the data as well as the model (in the model this is predefined). The residual plot (*Figure 5-4*) shows an underestimation of the model of the abundance in the age classes 5 and 6 years after arrival. In addition, there seems to be a small overestimation of the abundance in the age classes of 3 and 4 years after arrival. The estimated fishing mortalities for lakes IJsselmeer and Markermeer are given in *Table 5.1*, in three-year periods. The fishing mortality for lakes IJsselmeer and Markermeer decreases over time. With an estimate of 0.21 and 0.27 for respectively 2018-2020 and 2021-2023, and estimates of 0.39 and 0.44 for respectively 2012-2014 and 2015-2017, versus a value of 0.97 from 2006-2008, the model predicted a decreasing trend through time.

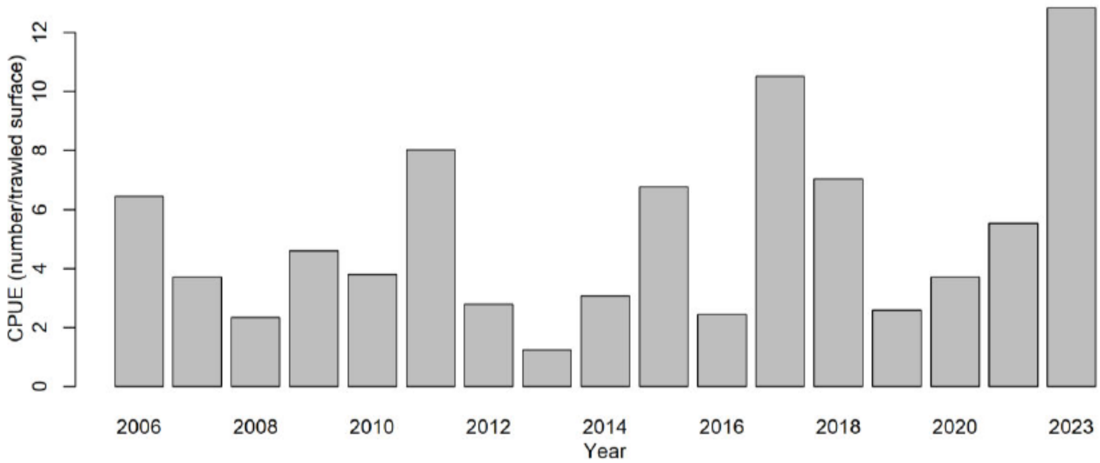

Figure 5-1 Mean CPUE per year in the FYMA electric beam trawl survey for lakes IJsselmeer and Markermeer, together, between 2006-2023.

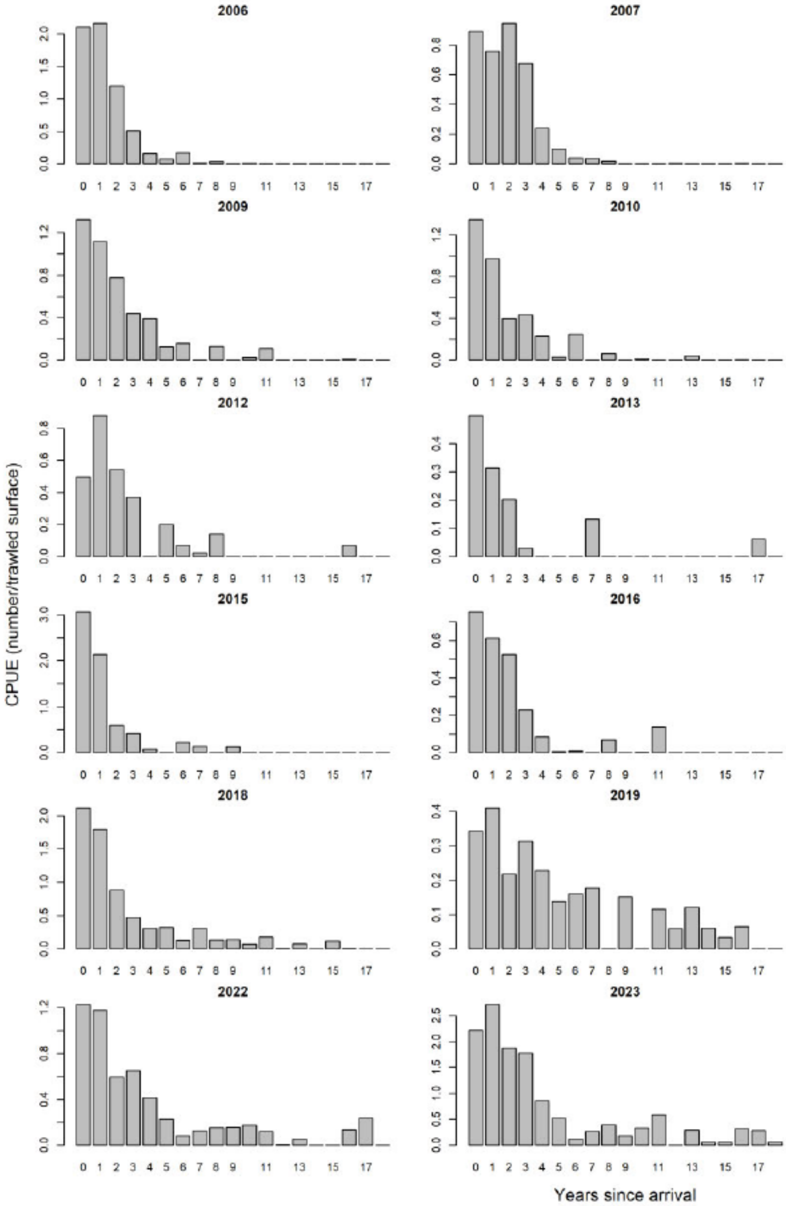

Figure 5-2 Mean CPUE per year per age class (years since arrival) in the FYMA electric beam trawl survey for lakes IJsselmeer and Markermeer, together, between 2006-2018. Note that the scale on the y axis varies per plot.

Table 5.1 Model-estimated mean fishing mortality values (means were taken over 9 year periods) for model fits on lakes IJsselmeer and Markermeer data together. The fishing mortality is estimated for different periods based on 100,000 iterations of the model.

Period	Fishing mortalitylakes IJsselmeer and Markermeer
2006-2008	0.97
2009-2011	0.65
2012-2014	0.37
2015-2017	0.45
2018-2020	0.21
2021-2023	0.27

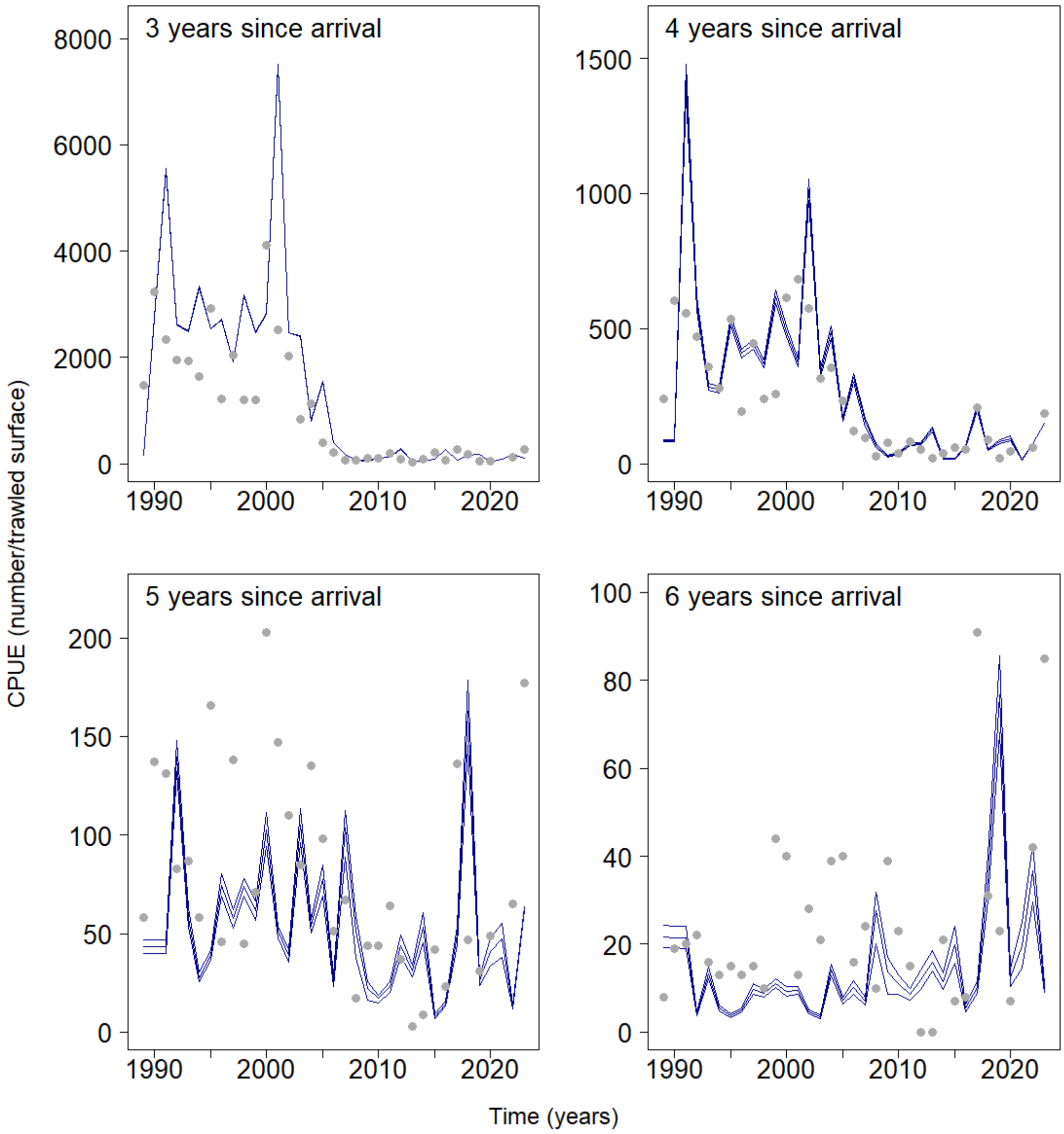

Figure 5-3 CPUE per age class (grey dots) and model predictions (minimum, maximum and mean outcomes of the last 20% of the iterations of the parameters estimated by the model – blue solid lines), both in number per trawled surface area (km^2), for the model fit of the data, together for lakes IJsselmeer and Markermeer.

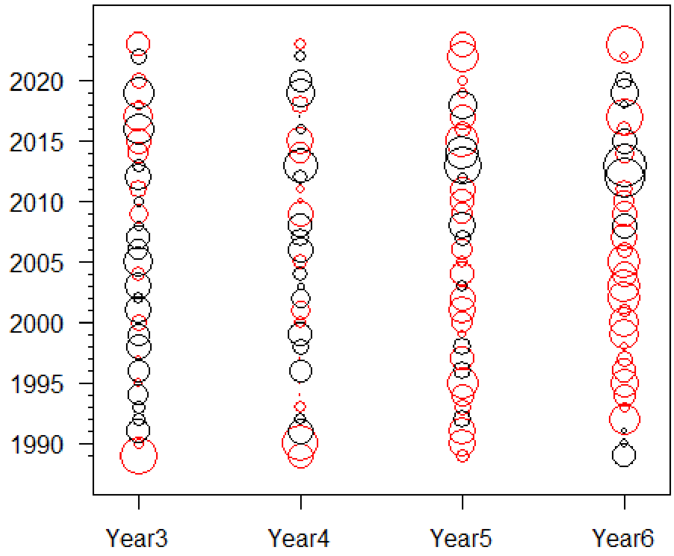

Figure 5-4 Residuals plot of the difference between the observed and predicted eel abundance per age class for the model fit of the data, based on the FYMA survey data for lakes IJsselmeer and Markermeer together, given the mean Fisheries mortality estimates presented in Table 5.1. Both negative (red) and positive (black) deviations are plotted. The size of the circles indicates the value of the residual (with larger being a higher value).

5.5 Discussion

The increase in eel numbers from 7 years after arrival in the lakes and older, as observed in recent years (*Figure 5-2*), 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 or older, may thus actually be relatively fast growing individuals and be younger than that in reality. In addition, growth patterns of eel could have changed over the years due to the large decrease in the density of eels (*Figure 5-3*). 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. Generally, it must be noted that the cohort structure in the FYMA data is not present. This is most likely due to 1) variable growth in eel over the years; 2) lack of sufficient otolith readings to capture the cohorts 3) lack of knowledge on the catchability of the fyma gear and 4) inaccuracy of the otolith readings, which are relatively difficult to read compared to other fish species. 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, especially because growth patterns are highly variable in eel. It is unlikely that the increasing eel numbers of individuals 7 years after arrival and older are migrating silver eel, as silver eel are hardly ever caught in the FYMA survey.

Just like for the previous assessment, the demographic model was fitted to the data of Lakes IJsselmeer and Markermeer combined. For a period of time the numbers of eel in the lake Markermeer survey had 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.

The catchability of the survey is assumed equal for all age classes. This is likely not the case in reality and yet, the assumption on catchability at age is crucial. If the catchability in the FYMA does change with age, the assumption of an age-invariant catchability may lead to a bias in the estimation of fisheries mortality. For example, if catchability decreases with age, the numbers in the FYMA will naturally be lower for older ages than they are in reality. The model may overestimate the fishing mortality to fit the (unrealistic) decrease in numbers with age in the FYMA index.
Compared to the previous stock assessment (Van der Hammen et al., 2021), the estimate of *F* has changed substantially from 2015 onwards. The main causes are: 1) a longer times series is used (until 2023 instead of 2020), which also affects the fit in previous years of the time series, 2) shorter periods were chosen for fitting the *F* values, 3) recruitment was based on the abundance of 2 year old individuals rather than the glass eel index, and, 4) different data were used for all biological keys that were used in the model (see Chapter 3).

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

- Add the retained catches to the model fit to allow an estimate of absolute biomass in the lakes by the model.
- Consider the possibility of the implementation of a varying selectivity of the FYMA survey with length.
- Instead of a Poisson distribution for the calculation of the Likelihood (Appendix B1), a normal distribution could be used in combination with a log transformation of the data. This would be a better fit for abundance data of a continuous nature, rather than count data which are integers.
- 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 extent this process is affected by environmental variables.

5.6 Eel biomass estimation in large lakes

In four large lakes (IJsselmeer, Markermeer, Randmeren and Grevelingen, *Figure 5-5*) eel biomass was estimated in a different way compared to other water bodies. The standing stock for the lakes IJsselmeer and Markermeer was estimated using fishing mortality in these lakes as estimated by the demographic model (*Table 5.1*) 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 for the Randmeren and Grevelingen, we therefore assumed that the eel density is the same as in the lakes IJsselmeer and Markermeer. To estimate biomass, the density is multiplied with the surface area of the lakes. 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)*.

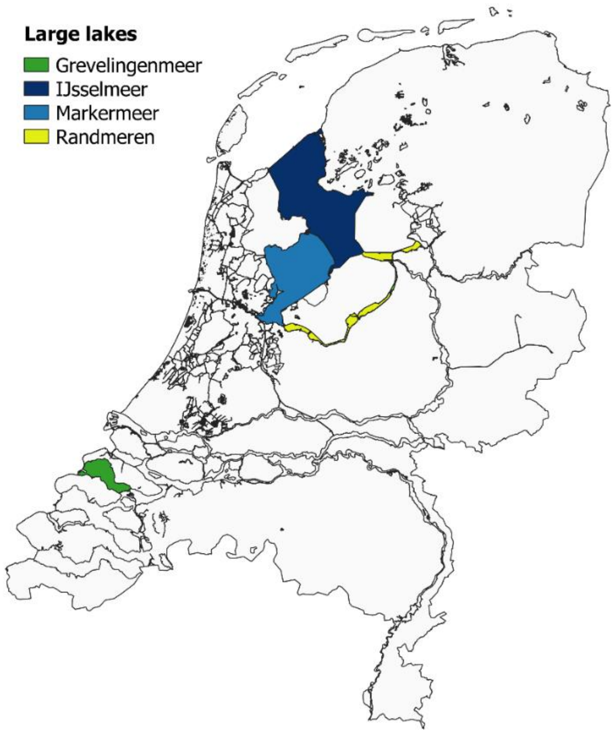

Figure 5-5 The four large Dutch lakes for which the density of lakes IJsselmeer and Markermeer was used as basis for the biomass estimate.

5.6.1 Standing stock lakes IJsselmeer and Markermeer

Estimates of the standing stock were calculated by combining the landings in lakes IJsselmeer and Markermeer (*Table 5.2*). The percentage yellow eel in the total landings was estimated using the length frequency data in the market sampling (*Paragraph 2.1*), sampled in lakes IJsselmeer and Markermeer. In total, 81% of the total retained catches in biomass was estimated to be yellow eel, which 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 (*Table 5.2*): *biomass = landings/(1-exp(-F))*. This resulted in an estimated standing stock of 1,346 tonnes (1,093 tonnes yellow eel and 253 tonnes silver eel, *Table 5.2*) in 2021-2023.

Table 5.2 Estimated mean yearly 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	222	0.97	357
	2009-2011	125	0.65	261
	2012-2014	129	0.37	417
	2015-2017	157	0.45	433
	2018-2020	250	0.21	1,320
	2021-2023	259	0.27	1,093
Silver eel (≥30cm)	2006-2008	51	0.97	83
	2009-2011	29	0.65	60
	2012-2014	30	0.37	96
	2015-2017	36	0.45	100
	2018-2020	58	0.21	306
	2021-2023	60	0.27	253

5.6.2 Standing stock Lakes Randmeren and Grevelingen

For the Randmeren and for Grevelingen, the eel density as estimated in the lakes IJsselmeer and Markermeer is used as basis. The randmeren are adjacent to lakes IJsselmeer and Markermeer and are connected, and therefore eel density are expected to influence to be quite similar. For Grevelingen no good data was available. This methodology results in an estimate of 22 tonnes of silver eel in the Randmeren and in a value of 19 tonnes of silver eel in Grevelingen in the latest period (*Table 5.3*).

Table 5.3 Density (kg/ha), surface area (ha) and yellow eel and silver eel standing stock (tonnes) per lake. Randmeren includes Randmeren-Oost, Randmeren-Zuid, Zwarte Meer en Ketel & Vossemeer.

		2006-2008	**2009-2011**	**2012-2014**	**2015-2017**	**2018-2020**	**2021-2023**
IJsselmeer/Markermeer	Density yellow eel	1.9	1.4	2.3	2.4	7.2	5.9
	Density silver eel	0.5	0.3	0.5	0.6	1.7	1.4
Randmeren	Surface area	16,338	16,338	16,338	16,338	16,338	16,338
	Yellow eel biomass	32	23	37	38	117	97
	Silver eel biomass	7	5	9	9	27	22
Grevelingen	Surface area	13,902	13,902	13,902	13,902	13,902	13,902
	Yellow eel biomass	27	20	32	33	100	83
	Silver eel biomass	6	5	7	8	23	19

6 Migration mortality

Silver eel suffer mortality during downstream migration when passing through pumps. This chapter describes the methodology and data used to estimate this pump mortality during silver eel migration.

6.1 Barrier types

The main types of barriers in the Netherlands are pumping stations, ship locks and discharge sluices. There are also three large hydroelectric power station's (HPS's) in the large rivers Maas and Nederrijn.

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 HPS uses flowing water to set a turbine in motion. 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)*

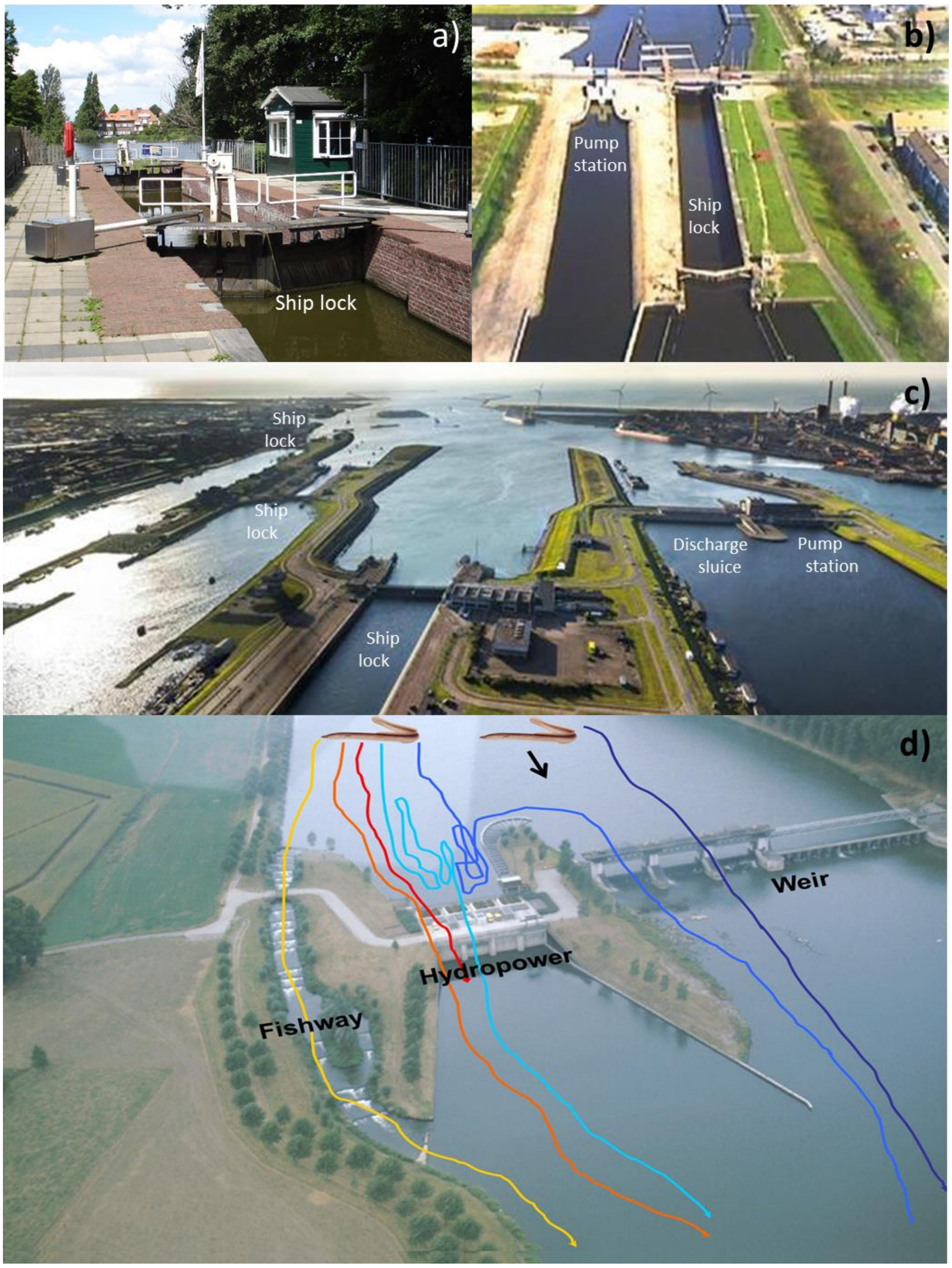

Figure 6-1 An illustration of different types of barriers. Barriers range from simple, e.g. single ship lock (a), to combinations, e.g. pump station and ship lock (b), to very complex sites consisting of a combination of pumping stations, ship locks, sluices or other alternative routes for migration, 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).

6.2 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 and HPSs) 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), which was updated in 2023 (Van der Hammen et al. 2023) to calculate mortality at the most important barriers. 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. See also *paragraph 2.8* and *Figure 2-9.*

6.3 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. In addition to three large HPSs there are many pumping stations and ship locks in the Netherlands (Kroes et al., 2018; Belletti et al., 2020). To estimate silver eel mortality caused by these barriers, knowledge on the following processes is necessary:

1) Silver eel migration routes
2) Mortality rates during passage of barriers
3) Local estimates of silver eel biomass

6.3.1 <u>Silver eel migration routes</u>

For mortality estimate during silver eel migration, migration routes are simplified and based on three hierarchies of water bodies (*Figure 6-2*). The three types are:

1) 1st hierarchy ('polder' water bodies): water bodies which are below sea level. Water levels are controlled by small pumping stations. Most pumping stations discharge water into a 'boezem' water body. Only a few coastal polders have pumping stations that discharge water directly to the sea.
2) 2nd hierarchy ('boezem' water bodies): water bodies such as canals, small inland lakes and smaller streams and rivers. Here, 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 larger boezem waters a combination of different man-made structures (barriers) is usually present at one location (*Figure 6-1*).
3) 3rd hierarchy ('national' water bodies, 'Rijkswateren'): large nationally managed water bodies such as the main rivers Rhine and Meuse and the freshwater lakes IJsselmeer and Markermeer, Randmeren and Grevelingen. In the River Meuse and the Rhine river branch Nederrijn, there are three large HPS's with turbines causing mortality. In IJmuiden there is a large sluice complex including a pumping station. Other national water bodies are connected to sea mainly by discharge sluices (e.g. IJsselmeer, Lauwersmeer, Haringvliet). These locations usually have alternative routes, e.g. ship locks or have an open connection (e.g. Nieuwe Waterweg).

Each hierarchy is connected to the higher hierarchy or to the sea. For each connection the proportion of eel migrating that route and the proportion that will not survive a passage is estimated (*Figure 6-2*). The model 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 not expected to have a large impact on the outcome of the model.

6.3.2 Mortality per hierarchy

Polders (1st hierarchy, pumping stations)

Silver eel migrating from the polder to the boezem or directly to the sea will encounter at least one pumping station, where a fraction will suffer mortality when passing this pumping station. Direct mortality is caused by the pumping station damaging the eel. Pumping stations are divided into three groups with estimated percentage of occurrence given for each type (Kunst et al., 2008); 1. pumps (Propeller pumps: 54%, Centrifugal pumps: 14% and propeller-centrifugal pumps: ~ 5%); 2. Archimedes' screws (27%) and 3. Water wheels (<1%). For each pump type, an overview of available studies (Van der Hammen et al., 2023*)* on the eel mortality at different pumping stations was made and the weighted average mortality per pump type was estimated (*Table 6.1*). In addition to direct mortality resulting from passing through the pump, 50% of the damaged eels are estimated to suffer from delayed mortality due to internal injuries (Kruitwagen & Klinge, 2008). The overall 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*).

It is assumed that silver eel migrate through a single pumping station in order to leave a polder. 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. In the model, the area of polder waters are represented by the wetted area of non-WFD waters (ditches, see *Paragraph 4.3.3).*

Table 6.1 Calculation of the average mortality after passing a pumping station (see also van der Hammen et al., 2023).

Pump type	Occurence (%)	Average mortality* (%)	Weighted Mortality (%)
Propeller pump	54	53.8	29.1
Archimedes' screw	27	12.0	3.24
Centrifugal pump	14	12.4	1.7
Propeller-centrifugal pump	5	13.8	0.7
Water wheel	0.2	0.0	0.0
Average Pump Mortality			**34.7**

* Mortality is % dead + 50% of damaged eel.

Boezem (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). 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% and for passage to the sea the estimated mortality is 5%. These estimates are assumed to be the same for all three-year periods.

National waters (3^{rd} hierarchy)

The 3^{rd} hierarchy consist of national waters. Within the national waters there are four main barriers causing silver eel mortality: three HPSs (located in Maurik, Linne and Lith), and there is a large sluices-pumping complex in the North Sea Channel in IJmuiden. Apart from those there are mainly discharge sluices, which do not cause mortality. The mortality in these four places is estimated separately, mortality from other national water bodies is assumed to be neglectable. This differs from previous reports where mortality in the IJmuiden complex was not calculated separately, but instead an overall mortality of 2% over all national waters was estimated.

HPS Maurik (Amerongen)

A HPS was constructed in the Lower Rhine near Maurik in 1988. The HPS consists of four horizontal Kaplan turbines, each with a capacity of 100 m^3/s. Mortality occurs when the silver eels migrate through one of the turbines. There are also alternative routes to migrate along the complex: there is a fish passage, a ship lock and a weir. The Driel complex is located upstream of the Maurik complex. Between Driel and Maurik the water level is kept at +6.00 m NAP for most of the year. The Nederrijn river branch receives only a small part of the Rhine debit (most water goes to the river branch 'Waal'). As a result, the turbines of the Maurik HPS are often shut down. There are two studies in which the mortality rate of silver eels was determined during migration through the turbines resulting in an average mortality of 12.5% (van der Veen & Kemper 2021a, Kemper & de Bruijn 2013).

To reduce mortality, measures have been in place at the HPS Maurik during the silver eel migration period (August 1 - January 31). These measures are:

1. A turbine may only run if the debit is at least 50 m^3/s (since ~2015).
2. Trap and Transfer of silver eels is carried out in front of the HPS from mid-August to mid-November (since 2013)
3. The river flow is utilized with as few turbines operating simultaneously as possible, i.e. with increasing discharge only when maximum capacity of an operating turbine is reached an additional turbine will be set in operation (since ~2012)
4. The turbine is shut down for 48 hours after the river flow exceeds 200 m^3/s (since 2022).

The measures reduce the mortality estimate to 2.8% (on average over 2020 and 2021, pers. comm L. Jans, RWS).

HPS Linne

HPS Linne is located in the Meuse southwest of Roermond. Mortality occurs when silver eels migrate through the turbines. There are also alternative routes to migrate along the complex: there is a fish passage, a shipping lock and a weir. Without measures, most silver eels migrate through the HPS (Griffioen et al, 2020). The most recent turbine mortality of silver eels was investigated by Van der Veen & Kemper (2021b). They found a mortality of eels passing through the HPS of 24.1% with a turbine flow rate of 50m^3/s and 13.3% with a turbine flow rate of 100m^3/s. 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, relative to the proportion of silver eel that pass through safer routes (weir, ship lock, fishway, see *Figure 6-1*). When corrected for the proportion that migrates through the station this is reduced to 17% at HPS Linne for the total flux of silver eel at these sites for the periods 2006-2008 and 2009-2011. Data on the proportion of eels distributed 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 et al. 2009) was implemented that resulted in a reduction of mortality for the HPS 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 14% for HPS Linne for the periods 2012-2014, 2015-2017, 2018-2020. In the most recent period, the policy is aimed at keeping mortality at the HPS below 5%. Strict measures are taken each year to assure that the mortality is below 5% and therefore it is assumed that the mortality rate will be around the policy target of 5% in the period 2021-2023. The measures taken to keep silver eel mortality below 5% are:

1. The turbines are not allowed to run between 5 p.m. and 7 a.m. from October 1 to December 31. The silver eels can then pass freely over the weir. Since the vast majority of silver eels migrate at night, this leads to a significant decrease in silver eel mortality.
2. From August 1 to January 31, the minimum flow rate at which the turbine may operate is 50 m^3/s and as few turbines as possible are used to utilized the river discharge.
3. From mid-August to mid-November, silver eels are captured in front of the weir complex with traps and released downstream (trap and transfer). Pilot done in 2022.

HPS Lith

The HPS Lith is located further downstream in the Meuse. There are alternative routes to migrate along the complex: there is a fish passage, a shipping lock and a weir. The most recent turbine mortality of silver eels at similar pumps was investigated by Van der Veen & Kemper (2021b) at HPS Linne (previous paragraph). They found a mortality of eels passing through the HPS of 24.1% with a turbine flow rate of 50m^3/s and 13.3% with a turbine flow rate of 100m^3/s. 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, relative to the proportion of silver eel that pass through safer routes (weir, ship lock, fishway, see *Figure 6-1*). When corrected for the proportion that migrates through the station this is reduced to 15% at HPS Lith for the total flux of silver eel at these sites for the periods 2006-2008 and 2009-2011. Data on proportion of eels distributed 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 et al. 2009) was implemented that resulted in a reduction of mortality for the HPS 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 Lith for the periods 2012-2014, 2015-2017, 2018-2020.

Just like at HPS Linne, the recent policy now states that no more than 5% mortality of migrating silver eels may occur at HPS Lith. In 2022, several measures are therefore in force that will reduce silver eel mortality. For this report it is therefore assumed that the mortality rate will now be around the policy target of 5%. The measures taken to keep silver eel mortality below 5% are:

1. The turbines are shut down at night from 4:00 PM to 8:00 AM from August 1 to December 31. Since the vast majority of silver eels migrate at night, this leads to a significant decrease in silver eel mortality.
2. The minimum flow rate at which the turbine may operate is 50 m^3/s and as few turbines as possible are used to drain the river water. This applies to the entire migration season from August 1 to January 31.
3. From mid-August to mid-November, silver eels are captured in front of the dam complex with traps and released downstream (trap and transfer). The amount that can be captured for the complex depends on the silver eel stock, the course of the river flow, the degree of shutdown of the HPS and the effort (number of fykenets) and is on average 2.1 (2015-2017), 2.2 (2018-2020) and 1.9 (2021-2023) tons per year (source: DUPAN).

IJmuiden complex (pumping station – shiplock -discharge sluices) in the North Sea Channel

At the IJmuiden complex (a construction with a pumping station, shiplock and discharge sluices, see *Figure 6-1*), research was carried out into the distribution of eel across the complex (Winter et al 2019, 2020). The research showed that 42% of the silver eels that arrive at the complex migrate to sea via the North shiplock sluice, 27% migrate via the pumping station, 13% via the discharge sluice and 17% travel via 3 smaller shiplocks towards the sea (Winter et al 2019, 2020; van Keeken et al. 2023). It should be noted that these were two years with relatively high use of the pumping station (see Winter et al., 2020).

In a recent study Griffioen & Van der Hammen (2024) estimated 59% mortality of eel going through the pumping station as a best estimate. Assuming that 27% of all silver eels migrate via the pumping station, this means an overall mortality rate of 16% (0.59*0.27) of eel migrating through the entire sluice complex barrier at IJmuiden.

During autumn 2023 for the first time trap and transfer of silver eel was carried out at the freshwater side of the complex, resulting in a total number of 1,666 silver eels being transferred to the seaside of the complex. This is estimated to reduce the mortality of eels migrating through the turbines with ~4% (2.4%-5.1%) to 55% (Griffioen & Van der Hammen, 2024). This reduces the total overall mortality over all silver eel migrating through the entire complex at IJmuiden to 15% (0.55*0.27).

6.3.3 Mortality estimates per hierarchy

Based on the migration routes and mortality estimates reported above, a model scheme was filled with the estimated proportions and mortalities in the latest period (*Figure 6-2*).

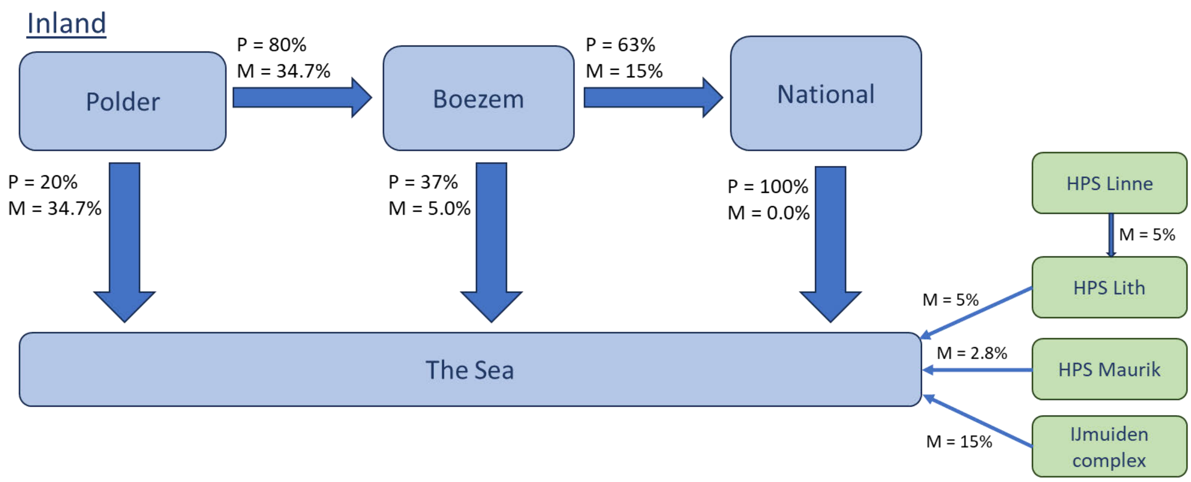

Figure 6-2 A conceptual model for estimating mortality during silver eel migration due to barriers; for 'polder' (1st hierarchy), 'boezem' (2nd hierarchy) and national waters (3rd hierarchy , see text). HPS: Hydropowerstation in river sections. The mortality in the four main barriers in the large rivers include all measures taken by the managers, including trap and transfer. In the three hierarchies (polder, boezem and national), corrections for trap and transfer are made later in the process. P= Proportion going to that hierarchy (%) ; M = mortality (%).The mortality rates of the polders and boezems are the same for each time period. The mortality rates of the four main barriers in the large rivers differ per time period.

6.3.4 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 calculated. The migration routes and mortality rates as described above are combined with the starting migrating silver eel biomass estimates per three-year period and location. This results in an estimate of the total percentage of barrier mortality of the starting migrating silver eel ($M_{barrier}$). The percentage of barrier mortality showed a decrease from the first period (18% 2006-2008, *Table 6.2*) to 8% in the latest period (2021-2023, *Table 6.2*). Most of the decrease is the result from a shift of a larger share of the silver eel abundance from the first and second hierarchies (polder and boezem waters) to the third hierarchy (national waters, e.g. large rivers and lakes). In the large rivers the mortality is relatively low, especially in the most recent period because several substantial measures to reduce eel mortality at the HPSs were carried out. The estimates are used in *Chapter 7* for the estimation of the key stock indicators.

Table 6.2 Total silver eel barrier mortality percentage for all hierarchies combined.

Period	Barrier mortality
2006-2008	18%
2009-2011	17%
2012-2014	15%
2015-2017	13%
2018-2020	11%
2021-2023	8%

6.4 Discussion

The large number of barriers, ranging from small pumping stations to large HPS's, make it difficult to calculate the barrier mortality at each site. In addition, the quality of existing research on barrier mortality is highly variable and often still incomplete. Few sites are well studied, e.g. the sites with HPS's in the River Meuse and Nederrijn (Winter et al., 2006 & 2007; Jansen et al., 2007; Griffioen et al., 2020; Van der Veen & Kemper, 2021b), 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 or executed with very small sample sizes. Also, the relative proportion of silver eels that pass through the turbines over other safer pathways (weirs, sluices, lateral canals), will be affected by changes in discharge patterns and recent management measures. Last, the barrier-mortality approach as used here for the second and third hierarchy waters can be further developed to enable a full site-specific and data driven approach including the first 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[5], the National Fish Migration Route Map RWS/'Nationale Visroutekaart RWS').

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

7 Overview national stock biomass

In this chapter the eel biomasses as estimated for the different waterbodies in the Netherlands (*Chapter 4* and *Chapter 5*) are summed. In *Chapter 7*, the summed biomasses are used to calculate the key stock indicators as requested by the EC.

7.1 National stock biomass

In previous chapters, the biomass estimates for yellow eel and silver eel are estimated for all water bodies in the Netherlands and subsequently summed to calculate the total biomass (*Table 7.1*). 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 and from the second to the third period (2012-2014) there was a small decrease. After the third period there was an increase to the fourth period (2015-2017) and again a small decrease to 2018-2020. In the most recent period (2021-2023) there is a large increase (*Table 7.1*). The current estimate is a standing stock of almost 7,000 (6,823) tonnes of eel (≥30 cm) in the Netherlands. The yellow and silver eel biomass estimates for each three-year period show the same long term trend as the total biomass, with initial increasing biomasses, a decline in the previous periods and an increase in the latest period. As most of the surveys are executed in autumn, the standing biomass reflects the situation in the autumn, right after the fishing season.

Table 7.1 Biomass estimates (all eel ≥ 30 cm, in tonnes) for each period per water type.

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020	2021-2023
Regional waters	Ditches	828	828	828	828	828	828
	WFD Waters	2,066	2,572	2,568	2,551	1,807	1,471
National waters	Large lakes (IJsselmeer/Markermeer, Grevelingen, Randmeren)	512	374	598	620	1,894	1,568
	Large rivers	612	1,560	1,113	1,935	1,089	2,957
Total	**Yellow and silver**	**4,017**	**5,334**	**5,106**	**5,934**	**5,617**	**6,823**
	Yellow eel	3,340	4,463	4,131	4,761	4,543	5,448
	Silver eel	677	871	975	1,174	1,073	1,375

7.2 Discussion

The values presented here show an increase in eel biomass (*Table 7.1*). This is due to an increase in the national waters: the large lakes show an almost 3-fold increase in biomass in the last two periods, compared to the periods before. The large rivers have an enormous increase in 2021-2023, compared to the previous period, where a decline was observed. This pattern, however, differs for the regional waters. In contrast to the national waters, the biomass estimates in the WFD-waters have declined in the last two periods (*Table 7.1*). Surveys in eel in ditches do not cover all ditches in space and time, and therefore only a single values has been estimated for all periods and no trend can be followed.

8 Stock indicators

To fulfill the obligations under the eel regulation (EC 1100/2007) MS's have to report on a list of stock indicators (*Table 1.2*). The stock indicators B_0, $B_{current}$, B_{best}, ΣH, ΣF and ΣA are based on the estimated biomasses and silver eel fishing and barrier mortalities and additional information on landings and recreational fisheries. The pristine silver eel biomass (B_0) is a constant value that was set at 10,400 tonnes for the Dutch inland waters (ICES 2010b). The methods to calculate the stock indicators are listed in Appendix B2. A detailed description of the stock indicators can be found in *Chapter 1*.

To assess the stock status, the current lifetime anthropogenic mortality rate (ΣA) and the current silver eel escapement biomass ($B_{current}$) are calculated. Subsequently, the percentage of current escapement biomass in relation to the estimated pristine situation (B_0) is calculated for each three yearly period (*Table 8.1*).

8.1.1 Anthropogenic mortality

The results show that since the first period (2006-2008) the anthropogenic mortality rate ΣA decreased, with a huge decrease between the first and second period and fluctuating afterwards. This decrease is mostly caused by a decrease in fishing mortality (fewer landings and recreational landings). In the most recent period, the anthropogenic mortality has also decreased compared to the period before, to a value of $\Sigma A = 0.60$ (corresponding to 45% mortality). This is due to a higher estimate of the standing stock compared to previous years, despite a small increase in the yearly landings (477 tonnes in 2018-2020 vs. 504 in 2021-2023, *Table 2.1*). The value of the current lifetime anthropogenic mortality ($\Sigma A = 0.60$) lies below A_{lim} (0.92), suggesting that the anthropogenic mortality is at a level that an increase in the stock size would be expected if other EMU's would also have values of ΣA below A_{lim}.

8.1.2 Biomass

The results show that the silver eel escapement biomass ($B_{current}$) has increased from 555 tonnes in 2006-2008 to 1,269 tonnes in 2021-2023. The proportion $B_{current}/B_0$ demonstrates that the current biomass of escaping silver eel is 12.2% of the pristine situation which is below the target of 40%.

Table 8.1 Overview of stock indicators used to evaluate the impact of the EMP on the biomass of escaping silver eel and anthropogenic mortality per period. Biomasses are in tonnes (t). Yellow eel and silver eel stock estimates refer to eel of at least 30 cm.

		2006-2008	2009-2011	2012-2014	2015-2017	2018-2020	2021-2023
Biomass	B_{start} (tonnes)	957	1,023	1,096	1,298	1,248	1,561
	$B_{current}$ (tonnes)	555	724	830	1,022	952	1,269
	B_{best} (tonnes)	3,363	1,656	1,628	1,777	1,957	2,322
	B_0^*	10,400	10,400	10,400	10,400	10,400	10,400
	$100^* B_{current}/B_0$	5.3	7.0	8.0	9.8	9.1	12.2
Mortality	ΣA (anthropogenic mortality rate)	1.80	0.83	0.67	0.55	0.72	0.60
	ΣH (barrier mortality rate)	0.20	0.20	0.19	0.17	0.15	0.11
	ΣF (fisheries mortality rate)	1.61	0.63	0.48	0.38	0.57	0.50

* Excluding coastal waters (2,600 t)

8.2 Discussion: Status of the eel stock in the Netherlands

8.2.1 Anthropogenic mortality

Anthropogenic mortality is the direct result of the measures taken by a MS. Low anthropogenic mortality can be achieved by reducing fishing mortality and barrier mortality. In the Netherlands, the implementation of the EMP has resulted in a reduction in ΣA between the first period (2006-2008) and the last period (2021-2023) from 1.80 to 0.60, and ΣA has been below A_{lim} (A_{lim} = 0.92) since the second period (2009-2011, *Table 8.1*). 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 between the first (2006-2008) and 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 =1.61 in 2006-2008 to ΣF = 0.63 in 2009-2011, *Table 8.1*). However, after that first decrease, the anthropogenic mortality has first gone down to 0.38 in 2015-2017 but has been higher in the latest two periods due to the increase in landings (*Table 8.1*).

Barrier mortality (ΣH) also showed a decrease from 0.20 to 0.11 (*Table 8.1*) from 2006-2008 to 2021-2023. A reason for the reduction is that the biomass in the WFD monitoring (polders and boezems) has decreased, in contrast with the biomass in national waters. Because the mortality of migrating silver eels from polders and boezem's is assumed to be higher than in the national waters, the increase in biomass in the national waters (large rivers and lakes including IJsselmeer and Markermeer), especially in the latest period causes the barrier mortality to decrease. Another reason for the low estimate of the barrier mortality in the latest year is that there have been strict measures at HPSs (new management scheme), to keep the silver eel mortality below 5%. Also the increased trap and transfer activities have had a positive effect in the reduction of eel mortality at barriers.

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 (large rivers) 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.

8.2.2 Biomass escaping silver eel

Between the periods 2006-2008 and 2021-2023, there was an increase in the biomass estimate of escaping silver eel ($B_{current}$) in every period, with the largest increase in the most recent period (2021-2023). However, when the current estimate of the eel stock is compared with the estimate of the pristine value B_0, the status of the eel stock in the Netherlands in 2021-2023 remains below the set target in the eel regulation. The current biomass of migrating silver eel (12.2%) is still far below the target of at least 40% of the pristine biomass. However, it is more and more recognized by ICES (2023) that the estimates of B_0 from the various Member States (or 'eel management units') cannot be compared with the later biomass estimates because they have very high uncertainty and are calculated in different ways.

8.2.3 Total European stock

To validate how the eel stock is doing, an assessment covering the whole distribution area of the European eel from North Africa to Norway should be carried out. However, because this assessment does not exist, the only way to validate if the total spawning stock is increasing is to have a look at the recruitment (glasseel) levels. If the stock is doing better, an increase in the recruitment is expected. Currently, glass eel recruitment at the European level has not significantly increased or decreased after the implementation of the EMP in 2009 (ICES 2023). If one EMU alone, such as the Netherlands, would reduce all anthropogenic mortality to zero, a recovery of the European eel stock is not expected. To maximize the chance of recovery, maximum protection of European eel will have to be accomplished throughout its entire natural range: the responsibility for improvement of eel stock lies with all countries in the natural range of the eel distribution.

9 Conclusions

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_{start} B_0, and ΣA, *Table 8.1*) 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.

The main results of this assessment are that in the most recent period (2021-2023), the current silver eel escapement $B_{current}$ (1,269 tonnes), is still much below the target of 40% of the estimated pristine situation (B_0), but the anthropogenic mortality (ΣA = 0.60) is below A_{lim} (A_{lim} = 0.92). Because of the low $B_{current}$ in relation to B_0, the status of eel in Dutch waters remained in a situation regarded as "undesirable". However, $B_{current}$ is increasing and ΣA has decreased since the latest period.

After implementation of the EMP in 2009 the estimate of $B_{current}$ increased: from 555 tonnes in 2006–2008 to 1,269 tonnes in 2021-2023 (*Table 8.1*). The increase in the latest period is a direct result of the survey findings from the national surveys including the lakes IJsselmeer and Markermeer. The biomass in lakes IJsselmeer and Markermeer showed a strong increasing trend, starting at around 2017, with an increase in the estimate of silver eel escapement from 440 tonnes (2006-2008) to 1,346 tonnes. Also the landings in lakes IJsselmeer and Markermeer have increased (from 273 per year (2006-2008) to 319 tonnes per year (2021-2023), yellow and silver eel combined, *Table 5.2*).

The biomass estimated in the large rivers also increased from 612 tonnes in 2006-2008 to 2,957 tonnes in 2021-2023 (*Table 7.1*). The main increase was observed in the WFD waters Biesbosch, Hollands Diep/Haringvliet oost, Merwede, Oude Maas en Bovenrijn/Waal (*Figure 4-5*, *Table 4.7*). The large rivers are large water bodies and in all large rivers the eel fishery is closed due to pollution. Because these waterbodies comprise such large areas, the influence of the survey outcomes in these area's is large on the total biomass and the assumptions on the ratio of eel distribution between the shore and the open water are strongly influencing the estimated eel biomasses.

In contrast to the large rivers, the total biomass estimate in the regional waters (WFD waters) declined from 2015-2017 to 2021-2023 (*Table 7.1*). Within the regional waters, Wetterskip Fryslân is highly influential because it represents the highest biomass estimates (*Appendix A3*). Since eel fishing in Fryslân is based on a yearly set quota (36.6 tonnes for all fishermen), the lower biomass is unlikely to be related to increased catches in this area. In this region, there is also much restocking of glass eel, which could cause fluctuation in the biomass estimate between periods.

For all components of the standing stock biomass estimates, the accuracy is low. For the static spatial model, main sources causing low accuracy are the catch efficiency of the electric dipping net and the habitat preference. However, apart from the selectivity and the habitat preference there is probably also a high level of sampling variation. Variation in sampling could be caused because 1) 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, water temperature, weather, exact location, time in the year, sampler) and 2) 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 this variation 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 catch efficiency 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.

9.2 Pristine biomass estimate (B_0)

The target of 40% of the pristine biomass (B_0) is the only quantitative target in the EU regulation and therefore there is much focus by governments and stakeholders on the value and estimation of B_0. Most EMUs calculated B_0 from data (mostly landings) before 1980. However, because anthropogenic factors such as fisheries, restocking and other anthropogenic mortalities have existed in most EMUs before those dates, the estimates do not truly refer to a period in which no anthropogenic mortalities existed. Although most EMUs seem to have done the best job possible to estimate B_0 given the available information, there are several issues with the B_0 calculation causing many and large uncertainties around the estimates (ICES 2021). 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. In addition, the estimation method of B_0 differs in many EMUs from the estimation method of $B_{current}$ and therefore can not be compared, especially given the large uncertainties around both estimates. Because of these uncertainties and differences in estimates, ICES is now of the opinion that B_0 cannot be compared with $B_{current}$ (ICES 2023, ICES 2021) and the ICES fisheries advice has never been based on these reference points (ICES 2023).

In the Netherlands, the B_0 value is set at 13,000 tonnes (10,400 tonnes for inland waters). However, the uncertainty of the value is large and is and has been subject to discussion many times, starting right after the first attempt to calculate it. Initially the pristine silver eel biomass 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. If, instead of the 13,000 tonnes the lower or higher bound of the Eijackers et al. estimation was used, $B_{current}/B_0$ in the Netherlands would be between 6% and 20%.

9.3 Biological keys

The maturity-at-length and the sex ratio-at-length were analysed with a *GAM* (*Chapter 2*). *GAM*'s are non-linear and therefore do not have a forced shape. However, in a *GAM* no underlying relationship is assumed, therefore the final shape was based on visual rather than statistical criteria and is consequently partly a result of expert judgement.

The shape of the maturity-at-length key has substantial impact on the final results, because it defines the proportion of eel that grows into silver eel within a year. For the growth-at-length curve (*Paragraph 2.6*) von Bertalanffy growth was assumed. This curve is used only for the demographic model. However, it has a substantial impact there, because the faster the growth, the earlier eel become mature.

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 lengths and is also not expected to be 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.

9.4 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 because these assumptions are extremely difficult to assess. However, it is still needed to at least get some more knowledge of both uncertainties as especially for wider/larger water bodies, assumptions of the distribution of eel over the water body may lead to large under or overestimates.

9.4.1 Regionally managed waters

In the biomass assessment for the regionally managed water bodies WFD fish survey data was used. A problem with this data is that not all water bodies are sampled in the same manner and 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.

9.4.2 Nationally managed waters

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

- The national managed waters are relatively large waterbodies. As a result the relative sampling intensity compared to the surface area is relatively small. Even though three years of data are used for each period, which smooths the values over these years, outliers due to for example timing of the survey, water temperature, water levels, eel behaviour and silver eel migration activity still cause large noise in the estimations, which has relatively large impact on the biomass estimate in each water body. A statistical model correcting for some of these abiotic factors can be used in the future to correct for some of these causes of variation.
- Because in the large water bodies, the open water is relatively large, but no sampling takes place in the open water, the assumption of 50% biomass in the open water compared to the shore is highly influential in these waters.
- In the large rivers, the influence of silver eel migrating from other areas or countries is high, because they are the main migration routes. In Germany many glass eels have been restocked, which migrate to the Dutch waters.

9.5 Demographic model

The main decreasing stock trends since 1989 in lakes IJsselmeer and Markermeer could be explained reasonably well by the demographic model (*Chapter 5*), but only to a certain extent. Several sensitivity analyses showed that the estimated F value by the demographic model is highly 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 unknown. 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.

The assessment outcome of the demographic model in lake IJsselmeer and Markermeer is highly sensitive to the biological keys, especially growth. As the number of otolith readings is limited, growth is currently modelled as a single key, thus assuming constant growth over time. In addition, the natural mortality estimate is only a crude estimate and assumed to be constant over all lengths, ages and periods. An intensive study on growth and more realistic estimate of natural mortality should be investigated.

9.6 Silver eel migration model

Water boards have invested and are still investing substantially in improving migratory opportunities at barriers, but most solutions are targeted to facilitate upstream migration. Potentially, this has improved glass eel immigration into inland 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).

For the silver eel migration model, accurate mortality when passing the barriers is needed. This requires 1) knowledge of the amounts of silver eels that arrive at the barrier; 2) the division of silver eels that end up at a certain barrier site over the different migration routes and 3) the mortality (and injury) rate of the silver eel that migrate through the pump. For some sites, good data on route selection is available, e.g. at the HPSs 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.

9.7 Unquantified sources of anthropogenic mortality

The main sources of mortality of European eel in the Netherlands are 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:

- Impact of (human-induced) viruses, parasites and pollution. Eels that are carrying toxins or diseases may very well grow up to silver eels. However the amount of stress experienced during the journey to the spawning grounds may cause these silver eels to have a (much) reduced chance to contribute to the spawning.
- Poaching (unreported landings or illegal removals).
- 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.
- Yellow eel mortality in HPSs and pumping stations.
- Catch and release mortality in recreational fisheries.

9.8 Restocking

In the Netherlands large numbers of restocking of glass eel and elvers (eels that are grown in culture facilities for some time before being restocked) have existed for decades. After the eel decline, the commercial restocking lessened due to the high glass eel prices. After restocking became one of the management measures in the Dutch EMP, restocking was financed by public resources. As a consequence, numbers of restocked eels increased (from a yearly average of 818 kg in 2006-2008 to 2,035 kg in 2021-2023, ICES 2023b). In the Netherlands, restocking is commissioned by the ministry of *Agriculture, Fisheries, Food quality and Nature* (LVVN) and is executed by the DUPAN foundation (www.DUPAN.nl), a foundation representing Eel processers, fish farmers and eel fishermen. The latest ICES advice (2023) states:

> '*ICES notes that the restocking of eels (the practice of moving eels from one waterbody to another) is intended as a conservation measure in EU Council Regulation (EC) No. 1100/2007 (EU Council, 2007) and is implemented in many eel management plans. Restocking is reliant on a glass eel catch, which is in contradiction with the current advice. The net benefit of restocking to the reproductive potential of the eel stock is unknown. It requires information on e.g. the carrying capacity of glass eel source estuaries, reliable mortality estimates at each step of the restocking process, and the spawning potential of stocked vs. non-stocked eels. While a local increase in eel production may be apparent (ICES, 2016), an assessment of net benefit to the spawning stock was unquantifiable. When constrained by the above-mentioned uncertainties and potential harmful effects, while following the precautionary approach, no catch for restocking should be allowed.*'

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. Because of this, it is unknown how much the current increase in $B_{current}$ is a result of restocking and how much the restocking contributes to the spawning population. It is therefore recommended to mark all stocked eels before release. In that way, otolith analysis will enable an estimate of the proportion of stocked eel in the (silver) eel population.

9.9 Future of the eel advice

As other European countries are using similar spatial models to estimate yellow eel standing stock and silver eel production (Höhne et al. 2024), close international collaboration is needed to enhance the quality and uniformity of these models in the future. In addition, fundamental differences exist among European countries with respect to the calculation of B_0. Standardization of assessment methods is of utmost importance to ensure the recovery of the European eel stock and its sustainable exploitation.

9.9.1 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 (*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 uses an analysis of the level of recruitment compared to levels before the recruitment had dropped as a basis for its advice (ICES, 2023). 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 originally proposed in 2026-2027 (ICES, 2021). Recently an EU (EMFAF) funded project has started ('DIASPARA'). Within this project several partners 'aim at providing tools to enhance the coherence of the scientific assessment process from data collection to assessment, with the final objective of supporting more holistic advice and to better inform a regional management'. This is a first step to a whole stock assessment for the European eel: see DIASPARA: DIAdromous Species: moving towards new PARadigms to achieve holistic scientific Advice - WUR

10 References

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Appendix A1 *Number of monitoring events per water board per year*

*Number of monitoring events per water board per year which were used for the analysis of the standing stock of eel in WFD water bodies; blue boxes represent that data was available and the number indicates the number of monitoring events; white boxes represent missing/not available/incomplete data; blue boxes with no number that data was available, but only pre-processed data (see text). *These are Rijkskanalen (national waters) that are regionally sampled/managed.*

Water board	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Aa en Maas			1	83			107	1		31	1	53	27	29	27	112	108
Amstel, Gooi en Vecht	167	9	16	22	56	5	149	210	5	22	72	62	54	99	38	54	64
Brabantse Delta	7	5	47		48	60	31	60	87	31	73	42	42	76	37	37	64
De Dommel	28	14	36	53	15	33	35	25	62	46	58	41	39	59	81	52	90
Delfland			5			55	27		32	30		36	31	3	41	50	
Drents Overijsselse Delta	19	85	7	5	9	52	7	17	45	30	70	79	69	100	119	63	98
Fryslân	24			51			73			97	10		70	51	62	55	28
Hollands Noorderkwartier			31	38	14	115	12	24	30	95	48	72	66	78	50	45	30
Hollandse Delta	4						1	114							112	18	
Hunze en Aa		33	35	79	68	50	52	76	49	48	53	48	50	73	84	56	45
Limburg	4	55	29	4	30	12	33	44	36	77	48	75	81	60	79	62	5
Noorderzijlvest		12	58	65	22	67	32	40	33	34	42	53	36	57	56	29	55
Rijkskanalen*	2	11	1	6	3	12	15	15	29	18	17	16	33	14	40	8	16
Rijn en IJssel	67	24	36	2	28	44	25	32	30	21	32	29	27	38	44	27	43
Rijnland	20	6		7	53	46	59	14	114	35	8	95		128	105	1	20
Rivierenland		39		149	117		107		156				226				
Scheldestromen																	
Schieland en de Krimpenerwaard	10	16	11	53	14	27	48	15	54	72	52	52	71	41	30	70	44
Stichtse Rijnlanden	24		5	15	6		39	31	6	14	9	11	35	37	31	34	29
Vallei en Veluwe			11	68	14	33	54	39	56	56	53	78	66	78	25	45	36
Vechtstromen	4	84	39	70	70	79	54	54	97	107	78	1	27	56	50	72	39
Zuiderzeeland					27			21						14			12
Number of fishing events	380	393	368	770	594	690	960	832	921	864	724	843	1050	1091	1111	890	826

Appendix A2: Water body types defined within the WFD in the Netherlands.

WFD water	Description of WFD water	Category	Examples
M1a	Buffered ditches	Ditches	
M1b	Brackish buffered ditches	Ditches	
M2	Weakly buffered ditches	Ditches	
M3	Buffered canals (regional) ("Tochten")	Canals	Alm, Drentse kanalen, Peelkanalen & Tochten Zuiderzeeland
M6a	Large shallow canals with shipping	Canals	Linge, Haarlemmertrekvaart & Naardertrekvaart
M6b	Large shallow canals without shipping	Canals	Kromme Mijdrecht, Hollandsche IJssel & Zwolse vaart
M7a/b	Large deel canals with/without shipping	Canals	Twentekanalen, Julianakanaal & Prinses Margrietkanaal
M8	Buffered peatland ditches ("laagveensloten")	Ditches	
M10	Peatland canals ("laagveen kanalen/(Polder)vaarten")	Canals	Wiericke's, polder Westzaan & Alblas
M12	Shallow, relatively small, buffered lakes	Lakes	Zwaluwmeer
M14	Shallow, relatively large, buffered lakes	Lakes	Alde Feanen, Oldambtmeer, Leekstermeer, Naardermeer & 't Twiske
M20	Deep, relatively large, buffered lakes	Lakes	Alkmaardermeer, Gaasperplas, Vinkeveense plassen, & 't Joppe
M23	Shallow, large, calcium-rich lakes	Lakes	Amsterdamse waterleidingduinen, Meijendel & Zuid-Kennemerland
M27	Shallow, relatively large, peatland lakes ("laagveenplassen")	Lakes	Beulakerwijde, Westeinderplassen, Nieuwkoopse plassen & Paterswoldsemeer
M30	Weakly brackish waters, 0.3 - 3 g [Cl/l]	Lakes	Lauwersmeer, Markiezaatsmeer, Noordzeekanaal & wateren Scheldestromen
M31	Small brackish waters, > 3 g [Cl/l]	Canals	Wateren Scheldestromen
R4	Permanent, slow flowing, upper part stream on sandy riverbed	Rivers	Chaamse beken, Hooidonkse beek, Virdisch Graaf & Viltische Graaf
R5	Permanent, slow flowing, middle- or lower part stream on sandy riverbed	Rivers	Azelerbeek, Drentse Aa, Groote Beerze & Roggelse beek
R6	Slow flowing small river on sandy/clay riverbed	Rivers	Berkel, Boven Mark, Dinkel, Kromme Rijn & Oude IJssel
R7	Slow flowing river/side stream on sandy/clay riverbed	Rivers	Reitdiep-Kommerzijl, Vecht & Westerwoldsche Aa
R8	Fresh tidal waters on sandy/clay riverbed	Rivers	Oude Maas
R12	Slow flowing, middle- or lower part stream on peat riverbed	Rivers	
R13	Fast flowing, upper part stream on sandy riverbed	Rivers	Rode Beek
R14	Fast flowing, middle- or lower part stream on sandy riverbed	Rivers	Halsche beek en Hooge Raam
R15	Fast flowing small river on siliceous riverbed	Rivers	Roer
R17	Fast flowing, upper part stream on calcium rich riverbed	Rivers	Caumerbeek, Eyserbeek & Gulp
R18	Fast flowing, middle- or lower part stream on calcium rich riverbed	Rivers	Geleenbeek & Geul

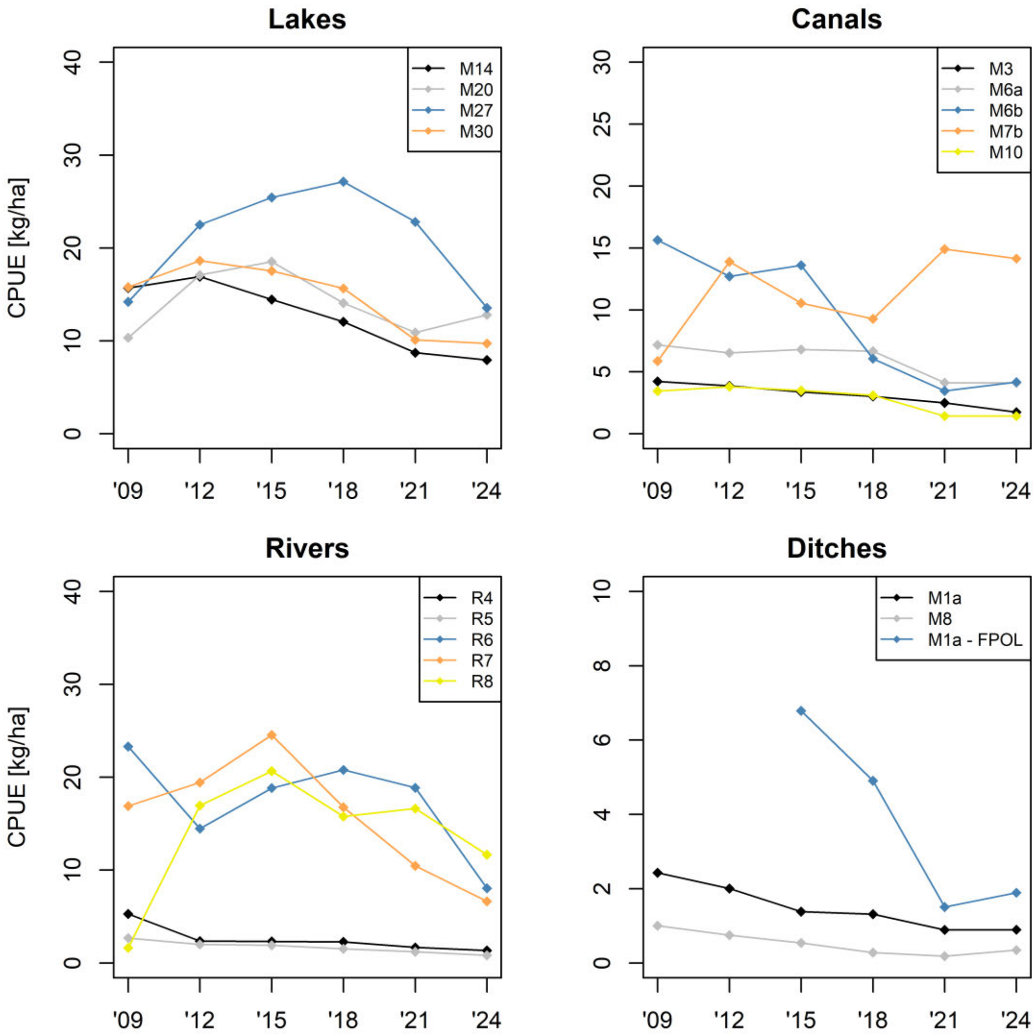
Appendix A3 A top-15 list of regional water bodies

A top-15 list of regional water bodies that have 1) the highest CPUE within the KRW sampling and 2) contribute most in biomass to the standing stock of eel in regionally managed waters.

	Top-15 Regional waters						
	CPUE-based			**Biomass-based**			
	Waterbody	cpue [kg/ha]	*Water board*	**Waterbody**	biomass [ton]	Area [ha]	*Water board*
1	t Joppe	104.1	*Rijnland*	Frieze boezem - overige meren	187	5615	*Fryslan*
2	Hoge Boezem van de Overwaard	90.9	*Rivierland*	Frieze boezem - grote diepe kanalen	108	957	*Fryslan*
3	Niers	87.5	*Limburg*	Markiezaatsmeer	72	1192	*Brabantse Delta*
4	Haven van Stellendam	85.3	*Hollandse delta*	Oostvaardersplassen	71	1695	*Noorderzijlvest*
5	Loohoek	65.2	*Scheldestromen*	Amstelmeer	61	650	*Hollands Noorderkwartier*
6	Kanaal door Voorne	64.2	*Hollandse delta*	Brielse Meer en Bernisse	60	555	*Hollandse Delta*
7	Rottemeren	63.5	*Schieland en de Krimpenerwaard*	Lauwersmeer	57	2360	*Noorderzijlvest*
8	Oosterland	46.3	*Scheldestromen*	Fluessen e.o.	48	3049	*Fryslan*
9	Friese boezem - grote diepe kanalen	42.7	*Fryslan*	Mark en Vliet	43	389	*Brabantse Delta*
10	Brielse Meer en Bernisse	42.3	*Hollandse delta*	Alkmaardermeer	35	601	*Hollands Noorderkwartier*
11	Maasnielderbeek benedenloop	42.0	*Limburg*	Boezem	33	3065	*Drents Overijsselse Delta*
12	Mark en Vliet	41.6	*Brabantse delta*	t Joppe	24	92	*Rijnland*
13	Binnenschelde	41.1	*Brabantse delta*	Alde Feanen	22	588	*Fryslan*
14	Zuiderdiepboezem	40.6	*Hollandse delta*	Hollandsche IJssel	19	288	*Rijkskanalen*
15	Swalm	40.2	*Limburg*	Hoendiep-Aduarderdiep	18	203	*Noorderzijlvest*

Appendix A4: Catch per unit of effort (CPUE) for several WFD water types.

WFD water types are grouped into four categories; lakes, canals, rivers and ditches. The x-axis indicate each assessment period and represents data of six years; "09": data from 2006-2011; "12" data from 2009:2014; "15": data from 2011:2016; "18": data from 2013-2018; "21": data from 2015:2020; "24": data from 2017:2022. Note that the y-axis differs for the plots of the four categories.



Appendix A5: Overview of inland and coastal waters

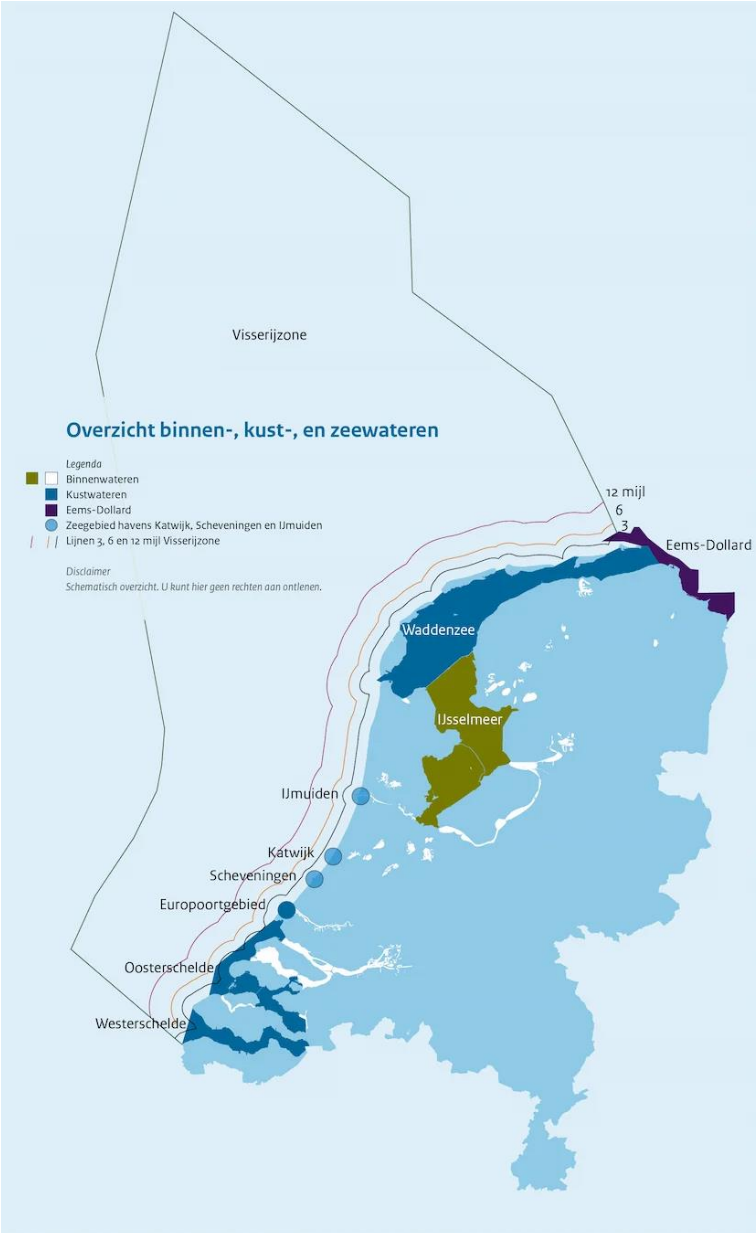


Figure A5 Overview of coastal and inland waters in the Netherlands. Source RVO: [Overzicht zee-, kust- en binnenwateren (rvo.nl)]

Appendix B1: Details of the demographic model

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}:

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

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 $\mathbf{r}_f(t)$:

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

Recruitment further depends on the abundance of individuals in year class 2 $I(t)$. Male recruitment $\mathbf{r}_m(t)$ follows:

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

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 2.5 to 21.5 years after arrival in the lakes (*Table B1, Appendix B1*). The reason for starting the model 2.5 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 2.5-21.5 years in age.

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 5-2*). The FYMA survey data were converted from CPUE per length class (1-cm increments) to CPUE per age class based on the Von Bertallanfy growth curves. 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. 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.

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. 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. Moreover, some densities of eel increase with age in the cohorts (*Figure B1*), or no individuals are detected, especially from age class 7 and older. Therefore, the age class 6 was used as the last age class to fit the model to.

Parameter values for $F(t)$ 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 estimates which population age distribution best matches the data. Through stochastic iterations (n = 100,000), 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 $F(t)$. The results are based on initial values for F of 1.0, 0.5, 1.5 and 2.0, 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.

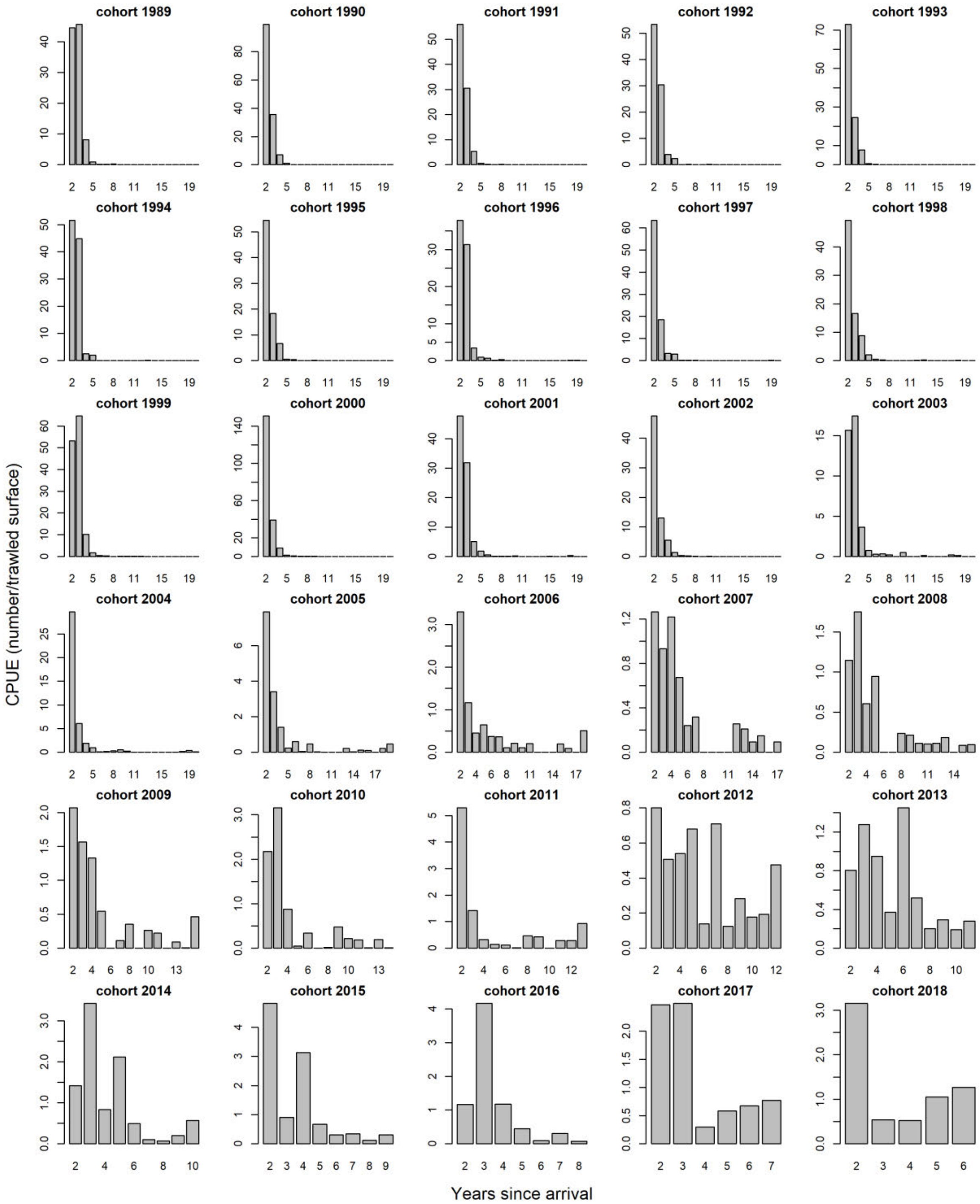


Figure B1 Mean CPUE per year of arrival (cohort) per age class (years since arrival) in the FYMA electric beam trawl survey for lakes IJsselmeer and Markermeer, together, between 2006-2018. Note that the scales on the x and y axis vary per plot.

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 der Hammen et al., 2021) are:

- The biological keys and FYMA survey data were updated with the newest information up to 2023.
- Instead of the glass eel index, the abundance at Year 2 in the FYMA survey was used as the abundance of the recruits.
- The age 7+ year class was no longer used for fitting, the model was fitted on the year classes 3-6.
- The sex ratio-length biological key was adjusted (see *Chapter 2*):
    - The sex ratio- length relationship now varies over time, the key was made per three year period.
- Multiple 3 year periods were used for fitting F values, and a walking mean value was calculated over 9 year periods for each of the assessment periods.

Model parameters

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 missing years, the average value over the years with data was used.

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, which runs from 2.5 – 3.5 years after arrival, the mid-age used for parameterization was 3 years after arrival. The length at the mid-age of the age class was determined from the von Bertalanffy growth curves (paragraph 3.5). 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 3-2*). Fishing selectivity (z_{gi}) is assumed zero for age classes with a length range 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 (*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. Natural mortality is assumed to be independent from age or length and constant through time, $\mu = 0.138$ (Dekker, 2000: see *Paragraph 3.6*).

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 (mm)		Maturation probability		Fisheries selectivity	
	Female	Male	Female (M_{fi})	Male (M_{mi})	Female (z_{fi})	Male (z_{mi})
2.5	256	249	0.002	0	0.11	0
3.5	315	294	0.002	0.002	1	0.83
4.5	370	329	0.003	0.065	1	1
5.5	420	358	0.007	0.287	1	1
6.5	466	381	0.019	0.468	1	1
7.5	508	400	0.043	0.593	1	1
8.5	547	415	0.087	0.658	1	1
9.5	583	427	0.128	0.673	1	1
10.5	616	437	0.183	0.681	1	1
11.5	647	445	0.219	0.684	1	1
12.5	674	451	0.241	0.684	1	1
13.5	700	457	0.273	0.684	1	1
14.5	724	461	0.296	0.684	1	1
15.5	745	464	0.337	0.684	1	1
16.5	765	467	0.368	0.681	1	1
17.5	784	469	0.385	0.681	1	1
18.5	801	471	0.422	0.681	1	1
19.5	816	472	0.46	0.681	1	1
20.5	831	473	0.479	0.681	1	1

Table B2 Annual abundance of Year 2 based on the FYMA survey (survey density/haul) and female ratio of recruits based on the segmented regression model (Appendix X)

Year	Year 2	Female ratio $\rho(t)$	Year	Year 2	Female ratio $\rho(t)$
1968	200	0.67	1996	2496	0.65
1969	200	0.67	1997	4047	0.61
1970	200	0.67	1998	3136	0.56
1971	200	0.67	1999	3543	0.52
1972	200	0.67	2000	9551	0.47
1973	200	0.67	2001	3104	0.43
1974	200	0.67	2002	2993	0.38
1975	200	0.67	2003	984	0.34
1976	200	0.18	2004	1864	0.30
1977	200	0.67	2005	490	0.29
1978	200	0.95	2006	211	0.33
1979	200	0.98	2007	89	0.37
1980	200	0.97	2008	71	0.42
1981	200	0.97	2009	132	0.46
1982	200	0.96	2010	135	0.51
1983	200	0.95	2011	328	0.56
1984	200	0.94	2012	50	0.60
1985	200	0.93	2013	50	0.64
1986	200	0.92	2014	97	0.69
1987	200	0.91	2015	306	0.73
1988	200	0.89	2016	75	0.76
1989	3318	0.87	2017	179	0.79
1990	6697	0.85	2018	211	0.67
1991	3640	0.82	2019	34	0.67
1992	3444	0.79	2020	102	0.67
1993	4560	0.76	2021	200	0.67
1994	3355	0.73	2022	123	0.67
1995	3568	0.69	2023	221	0.67

Appendix B2: Calculation of the stock indicators

To fulfill the obligations under the eel regulation (EC 1100/2007) MS's have to report on a list of stock indicators (*Table 1.2*). The stock indicators B_0, $B_{current}$, B_{best}, ΣH, ΣF and ΣA are based on the estimated biomasses and silver eel fishing and barrier mortalities and additional information on landings and recreational fisheries. The pristine silver eel biomass (B_0) is a constant value that was set at 10,400 tonnes for the Dutch inland waters (ICES 2010b). A detailed description of the stock indicators can be found in *Chapter 1*.

In order to estimate the stock indicators ΣA, $B_{current}$, B_{best} and B_{start} the following calculations are executed:

1. Yellow eel anthropogenic mortality rate

Yellow eel mortalities apply over a sequence of years as the yellow eel stage generally takes between 3-20 years. The yellow eel anthropogenic mortality rate $\widehat{F}$ is defined as the yellow eel mortality caused by commercial and recreational fisheries. Yellow eel migration mortality is not estimated, because it is already accounted for implicitly in the estimate of the yellow eel standing stock. The yellow eel fishing mortality rate ($\widehat{F}$) is estimated as a function of the proportions of retained catches and the estimated biomasses of the standing stock, following the equation:

$$\widehat{F} = -ln(1 - Catch_R/(Biomass + Catch_R))$$

where $Catch_R$ is the yellow eel retained catch and *Biomass* is the yellow eel biomass (≥ 30 cm, tonnes) as estimated in previous chapters. 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 fishery on eel is from May to August, because of the fisheries closure in September, October and November (part of the Dutch EMP[6]). 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 and the biomasses as estimated in previous chapters, $\widehat{F}$ is calculated for each period and scenario (*Table 0.1*).

Table 0.1 Mean yearly ***yellow eel*** *biomasses, mean yearly yellow eel retained catches (commercial and recreational) and mean yearly yellow eel fishing mortality rates ($\widehat{F}$) for each period.*

	Yellow eel		
period	**Standing stock Biomass (tonnes)**	**Retained catches (commercial and recreational, tonnes)**	**Fishing mortality rate ($\widehat{F}$)**
2006-2008	3,340	725	0.20
2009-2011	4,463	320	0.07
2012-2014	4,131	239	0.06
2015-2017	4,761	216	0.04
2018-2020	4,543	303	0.06
2021-2023	5,448	317	0.06

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

2. B_{start}

B_{start} is the Silver eel biomass before current silver eel mortalities (migration and fisheries) have occurred. The fishing mortality of silver eel is assumed to take place before the surveys are conducted. 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 (as estimated in the spatial and demographic model) and the silver eel landings (*Table 0.2*).

3. β

The parameter β represents the proportion silver eel production out of the best possible silver eel production. Parameter β is calculated using the demographic model, but parameterized specifically for the β calculation. This is done by assuming a fictitious population and comparing the outcome of the amount of yellow eel reaching the silver eel stage applying the yellow eel fishing mortality rate $\widehat{F}$ as calculated in step 1 with the outcome if $\widehat{F}$ is zero. As β is the ratio between the two values, the size of the starting population makes no difference (and thus an fictitious population size is appropriate).

4. α

Alpha (α) is the proportion of silver eel anthropogenic mortality and represents the silver eel fishing and barrier mortality during migration from freshwater to the sea. 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} represents the silver eel biomass before silver eel mortalities (migration and fisheries) have occurred; $Catch_R$, represents the retained silver eel catch; and $M_{barrier}$ (as calculated in *Chapter 6*) represents the proportion barrier mortality (*Table 0.2*).

5. Spawner per Recruit (SPR)

The Spawner per recruit is the number of spawners reached per recruit, which is needed to estimate $B_{current}$ in the following step. To estimate *SPR*, the parameters α and β (step 3 and 4) are needed. Subsequently, the *SPR* is estimated as:

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

6. $B_{current}$ and B_{best}

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:

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

7. Lifetime Anthropogenic Mortality (ΣA)

Subsequently, the Lifetime Anthropogenic Mortality rate is calculated as:

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

*Table 0.2 Silver eel biomass standing stock (in autumn), 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).*

Silver eel					
Period	**Standing Stock biomass (tonnes)**	**$Catch_R$ (commercial)**	**B_{start}**	**Barrier mortality proportion ($M_{barrier}$)**	**Anthropogenic mortality proportion (α)**
2006-2008	677	280	957	0.18	0.42
2009-2011	871	152	1023	0.17	0.29
2012-2014	975	121	1096	0.15	0.24
2015-2017	1174	124	1298	0.13	0.21
2018-2020	1073	174	1248	0.11	0.24
2021-2023	1375	186	1561	0.08	0.19

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
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			
						mean	**12.3**	**3**		
						Mortality (dead+0.5*damaged)			**13.8**	

Table C2 : Overview of eel mortality when passing through pumping stations with an Archimedes' screw.

	Pump description	Capacity (m^3/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			Wijk van, 2011
	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			
						mean	**4.0**	**15.9**		
								Mortality (dead+0.5*damaged)	**12.0**	

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

	Pump description	Capacity (m^3/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			
						mean	**11.2**	**1.4**		
						Mortality (dead+0.5*damaged)			**12.4**	

Table C4 : Overview of eel mortality when passing through pumping stations with a propeller-centrifugal pump (axial-radial water flow).

	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			
							40.5	**26.5**		
						Mortality (dead+0.5*damaged)			**53.8**	

Table C5 : Overview of eel mortality when passing through HPS station.

	Pump description	Debit (m^3/s)	Name	Year	N (no. Eel)	Dead (%)	Reference
Hydropower Station	HPS (turbine)	30-126	Lith_Alphen			24.3	Bruijs et al 2003
	HPS (turbine)	30	Lith_Alphen	2019	53	58	Da Graca & Kemper, 2020
	HPS (turbine)	50	Lith_Alphen	2019	65	26	Da Graca & Kemper, 2020
	HPS (turbine)	100	Lith_Alphen	2019	121	26	Da Graca & Kemper, 2020
	HPS (turbine)	80-100	Linne	2011	33	33	Da Graca & Kemper, 2020
	HPS (turbine)	total	Linne	2011		19	Buyse et al 2009
	HPS (turbine)	50	Linne	2020/2021	108	13.3	Van der Veen & Kemper, 2021b
	HPS (turbine)	100	Linne	2020/2021	98	24.1	Van der Veen & Kemper, 2021b
	HPS (turbine)	100	Maurik	2012	92	7	Kemper & de Bruijn, 2013
	HPS (turbine)	50	Maurik	2012	68	22	Kemper & de Bruijn, 2013
	HPS (turbine)	100	Maurik	2021	118	10.2	Van der Veen & Kemper, 2021a
	HPS (turbine)	50	Maurik	2021	101	10.9	Van der Veen & Kemper, 2021a
					Mean	**22.8**	

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