Stichting Wageningen Research Centre for Fisheries Research (CVO)

Evaluation of the Dutch Eel Management Plan 2018: Status of the eel population in 2005-2016.

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Nederlandse samenvatting

Aalbeheerplan

Sinds de jaren 1980 zijn de glasaalintrek en de aalpopulatie zeer sterk teruggelopen. Om herstel van de aalpopulatie mogelijk te maken heeft 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)" vastgesteld. Deze verordening verplicht de lidstaten om met een nationaal aalbeheerplan te komen 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."*

In juli 2009 heeft Nederland een aalbeheerplan opgesteld en geïmplementeerd in juli 2009. De belangrijkste maatregelen staan in *Tabel 1*.

Tabel 1 Overzicht van de maatregelen in het Nederlandse aalbeheerplan.

Maatregel aalbeheerplan
Terugzetten van aal (a) op zee en (b) op binnenwater door sportvissers
Verbod op recreatieve visserij, gebruikmakend van professionele vistuigen.
Gesloten aal visseizoen 1 september tot 1 december
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.
Uitzet van glas- en pootaal.
Sluiten van de visserij in de belangrijkste grote rivieren, met als aanleiding dioxineverontreiniging.

Evaluatie van het aalbeheerplan Nederland

Lidstaten zijn verplicht om voor 1 juli 2018 voor de derde maal over de voortgang van de nationale aalbeheerplannen te rapporteren aan de Europese Commissie. De onderliggende rapportage betreft een evaluatie van de effecten van het Nederlandse aalbeheerplan tot op heden, als bijdrage aan de rapportage aan de Europese Commissie.

Het aalbeheerplan is geëvalueerd in het licht van de voornoemde "beheerdoelen" uit de Aalverordening. De methodiek die bij deze evaluatie is gehanteerd komt voort uit de ICES aal werkgroep (WGEEL). Hierdoor wordt in deze evaluatie alleen ingegaan op de effectiviteit van maatregelen in relatie tot beheerdoelen opgesteld door de Raad van de Europese Unie. De evaluatie is uitgevoerd door middel van modellen, vangstgegevens, veldwaarnemingen en statistische analyses, en worden uitvoerig beschreven in de rapportage. Het geheel van deze inspanning resulteerde in schattingen van *voor* (2005-2007) en *na* (2008-2010, 2011-2013 en 2014-2016) de implementatie van het Aalbeheerplan (*Tabel 2*).

Tabel 2 Modeluitkomsten voor evaluatie van het aalbeheerplan.

	2005-2007	2008-2010	2011-2013	2014-2016
Uitrekkende schieraal ($B_{current}$)	1049 (t)	816 (t)	867 (t)	1365 (t)
% Uitrek van schieraal t.o. best mogelijke uitrek.	10%	8%	8%	13%
Antropogene sterfte (*LAM*)	81%	67%	59%	48%
Migratie sterfte schieraal	20%	23%	22%	18%

Effecten van het aalbeheerplan op de Nederlandse aalpopulatie
De evaluatie laat zien dat de maatregelen uit het Nederlandse beheerplan aal hebben geleid tot een teruggang in antropogene sterfte tussen 2005-2007 en 2014-2016. Deze reductie was voornamelijk het gevolg van beperkingen van de visserij. Door aanpassingen aan de infrastructuur bij migratieknelpunten is het percentage sterfte bij schieraal nagenoeg gelijk gebleven (tussen 20% in 2005-2007 en 18% in 2014-2016).

De status van aal in de Nederland blijft in 2015-2016 verontrustend met hoge sterfte en lage biomassa. De huidige biomassa van uittrekkende schieraal (13%) ligt ver onder de doelstelling van minimaal 40% van de pristine biomassa en de huidige sterfte door menselijk handelen (48%) ligt boven de geadviseerde sterfte bij een dergelijke lage biomassa aan uittrekkende schieraal (20%).

Een verbetering in de aalpopulatie in Nederland en in de uittrek van schieraal wordt niet op de korte termijn verwacht omdat aal een langlevende soort is. Het duurt meer dan een jaar voordat glasaal na aankomt voor de Nederlandse kust en de binnenwateren op zwemt. Vervolgens duurt het 5-15 jaar voordat deze aal “schieraal” wordt, en terugtrekt naar zee. Het blijft verder onzeker of de genomen maatregelen op termijn werkelijk zullen leiden tot een duurzaam verbeterde aalstand, omdat niet zeker is welke factoren de achteruitgang in de aalstand hebben veroorzaakt.

1 Introduction

The European eel (*Anguilla anguilla*) stock is declining since the 1980's. The International Council for the Exploration of the Sea (ICES) recommended that a recovery plan be developed for the European eel stock and that exploitation and other human activities affecting the stock be reduced as much as possible. In response to this advice the EU Regulation for the Recovery of the Eel Stock (EC 1100/2007) was adopted in 2007. It required each Member States (MS) to set up an Eel Management Plan (EMP) by the end of 2008:

> *" 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 approved by the European Commission in October 2009. After the approval, several measures as described in the EMP to reduce eel mortality were implemented (*Table 1-1*). Two progress reports were sent to the European Commission (EC) before, in 2012 and 2015. Both evaluations demonstrated that the status of eel in Dutch waters remained in a situation regarded as "undesirable", with high eel mortality and low biomass (Bierman et al., 2012; van de Wolfshaar et al. 2015). In both evaluations, the biomass of escaping silver eel was estimated to be below the target of 40% of the pristine situation and the anthropogenic mortality was above the mortality as the target set in the EMP. In the Netherlands the implementation of the EMP has resulted in a decrease in anthropogenic mortality between 2008 and 2016. The observed reduction in anthropogenic mortality was mainly the result of a decrease in fishing mortality, both commercial and recreational. The remaining measures (reductions in mortality at hydropower plants and pumping stations) had lower impact on a reduction in eel mortality.

The European Commission launched a data call for each member state to submit a set of stock indicators and related data in addition to the present progress report. The stock indicators and data are submitted in tables; the most important indicators are listed in (*Table 1-2*). The purpose of the indicators is to render the reports more efficient in demonstrating the progress achieved via the implementation of the EMP's. In particular, there need to be a clear indication as to the achievement of the 40% escapement target. The Ministry of Agriculture, Nature and Food Quality (LNV) has requested WMR:

1) to provide estimates for the required stock indicators, and
2) to use these indicators to evaluate the impact of the EMP on anthropogenic mortality and biomass of escaping silver eels.

The research in this report is performed within Wettelijke onderzoekstaken (WOT).

Table 1-1 Overview of the implemented measures described in the Dutch Eel Management Plan in order to reach the 40% escapement objective.

Measure	Planned implementation	Realised implementation
• Reduction of eel mortality at pumping stations and other water works; of the 1800 most important migration barriers.	2015-2027	2015-2027[a]
• Reduction of eel mortality at hydro-electric stations with at least 35%	2009	November 2011[b]
• The establishment of fishery-free zones in areas that are important for eel migration	2010	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 fishery in coastal areas using professional gear	2011	1 January 2011
• Annual closed season from 1 September to 1 December	2009	1 October 2009
• Stop the issue of licences for eel snigglers by the minister of EZ in state owned waters	2009	1 May 2009
• Restocking of glass eel and pre-grown eel from aquaculture	2009	Early 2010
• Research into the artificial propagation of eel		
PRO-EEL (EU-project)	2010	2010-2015
EEL- HATCH	2014	2014-2017
EELRIC (Dutch innovation centrum)	2015	2015 - ongoing
• Closure eel fishery in contaminated (PCBs, dioxins) areas	Unforeseen Measure	1 April 2011

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

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

[c] *The majority of the contaminated areas that were closed for commercial fisheries on 1/4/2011 are the main rivers. These rivers are the most important "high ways" for diadromous species like salmon and eel.*

Table 1-2 Overview of the stock indicators and related data to be reported by member states to the EC.

Indicator	Description
B_0	Silver eel biomass without anthropogenic ever having influenced the stock.
$B_{current}$	Silver eel biomass that currently escapes to the sea to spawn.
B_{best}	Silver eel biomass without anthropogenic influences on the current stock.
ΣF	Fishing mortality rate (commercial and recreational).
ΣH	Anthropogenic mortality rate other than fishing mortality.
ΣA	The sum of anthropogenic mortalities, i.e. $\Sigma A = \Sigma F + \Sigma H$.
R	The amount of glass eel (eel < 12 cm) used for restocking within the country. R is not relevant for the Netherlands as no glass eel is harvested.
LAM	Lifetime Anthropogenic mortality
Wetted Area	Wetted area of inland waters, transitional waters and marine waters used in B_0, $B_{current}$ and B_{best}

1.1 General description of the stock assessment methodology and main data sets

The same assessment methodology as in the previous assessments is used (Bierman et al 2012, van de Wolfshaar et al 2015). A short description of the methodology is described here.

Estimates of escaping silver eel and mortality rates of all eel are requested by the European Commission (*Table 1-2*). However, for the current evaluation of the Eel Management Plan, the most important of the required estimates are the estimate of the **Lifetime Anthropogenic Mortality** (*LAM*) and the anthropogenic mortality rate (*ΣA*, *Table 1-2*). Management actions were taken to reduce anthropogenic mortalities (*Table 1-1*), and reductions in *LAM* after the implementation of the EMP can therefore be used to evaluate the success of these management actions. However, the amount of silver eel escaping depends on many factors, which cannot all be controlled by a single member state. For example, silver eel escapement also depends on trends in recruitment, and the eel stock covers several countries. Also, the effect of some measures, such as of the reduction of yellow eel mortality and of glass eel stocking, will take numerous years to materialise.

Estimates of *LAM* are used to put current silver eel escapement ($B_{current}$; *Table 1-2*) into context, by comparing it with the best possible silver eel escapement under recent recruitment conditions (B_{best}). The estimated proportion $B_{current}/B_{best}$, (referred to as *%SPR,* spawner-per-recruit as a percentage of the best possible spawner-to-recruit ratio), can be compared with the 40% escapement target of the EU Regulation for the Recovery of the Eel Stock (EC 1100/2007). Estimated improvements in *%SPR* are therefore important indicators of silver eel recovery by which the EMP can be evaluated. A particular problem in this context is that cohorts will all have a different *LAM*, because the mortality changes over time. However, ICES indicated that estimates of either *LAM* or *%SPR* typically refer to anthropogenic impacts in the most recent year, not to impacts summed over the life history of any individual or cohort in the current stock (ICES 2014 and references therein).

To estimate *LAM*, we consider anthropogenic mortalities during the two main continental life stages of eels (*Figure 1-1*):

1) yellow eel stage: fishing (commercial and recreational) mortalities
2) silver eel stage: fishing (commercial) and barrier mortalities

The reason for considering these stages separately is that yellow eel mortalities apply over a sequence of years from transformation as glass eel to yellow eel until the transformation from yellow eel to silver eel. Instead, silver eel mortalities are assumed to apply only during migration from freshwater towards the sea. Together these mortalities combine to anthropogenic mortality in freshwater.

To estimate the *%SPR* a demographic model for yellow eel is made (*Chapter 3*) in which the development from glass eel to silver eel is modelled as a function of growth, maturation and natural and anthropogenic mortality. Mortality caused by passing of barriers also reduces the *%SPR*. This migration mortality is assumed to occur only during the silver eel stage.

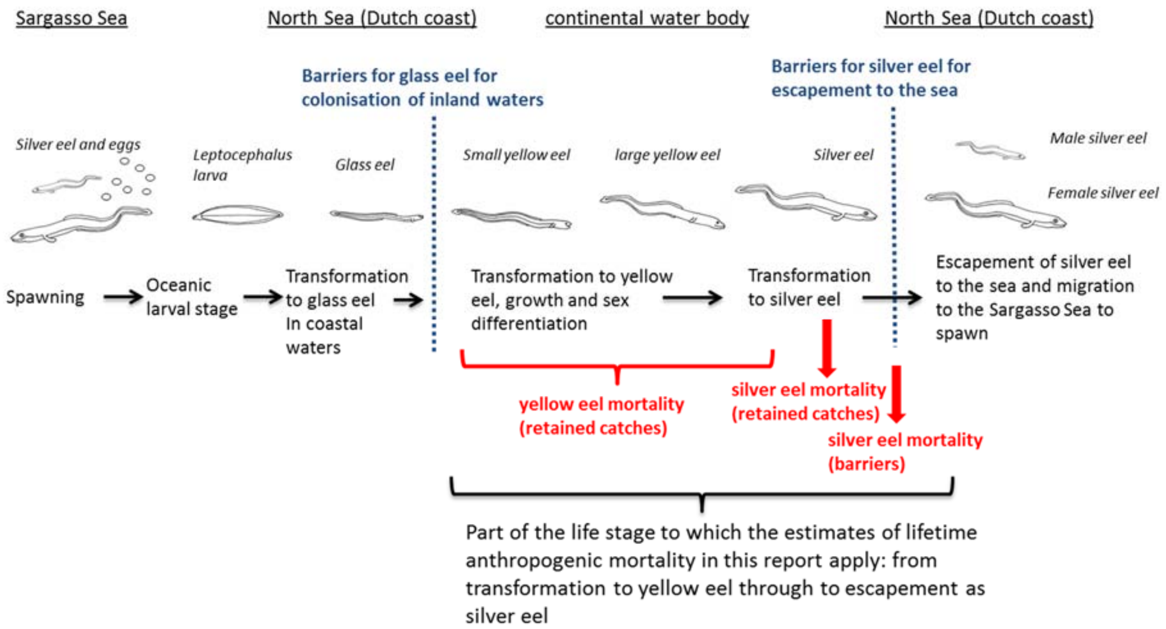

Figure 1-1 The life cycle of European eel, with the part of the life cycle to which the Dutch Eel Management Plan (EMP) applies and for which 'Lifetime Anthropogenic Mortality' (LAM) rate estimates are made in this report.

Yellow eel mortality rates were estimated as the proportion of the retained yellow eel catches (landings) from the total current standing stock of yellow eel with lengths above 30 cm. Commercial and recreational fisheries were assumed to be fully selective for eels above 30 cm in length, and entirely non-selective for eels smaller than 30 cm. Yellow eel retained catches were estimated by splitting the total reported retained catches by the commercial fisheries into yellow eel and silver eel, using a maturity ogive (size at maturation) calculated from market sampling data. Recreational fishing is almost exclusively angling, and therefore retained catches from recreational fisheries were assumed to consist entirely of yellow eel catches as the digestive tract of silver eel degenerates and feeding ceases (and are therefore not caught by angling). The harvested proportion of the yellow eel stock was estimated as the retained catch divided by the sum of the estimated standing stock and the retained catch.

Silver eel mortality rates were calculated as the proportion of the retained silver eel catches out of the total current production of silver eel. Silver eel retained catches were estimated by splitting the total reported retained catches into yellow eel and silver eel, using a maturity ogive (size at maturation) calculated from market sampling data. The mortality rates induced by barriers (pumping stations, hydropower plants, sluices, etc.) were estimated using a model in which the starting positions of silver eel were split into three hierarchies of water bodies:

1st hierarchy: silver eel starting from 'drainage ditches' (referred to the Dutch name: 'polder'), which are below sea level;
2nd hierarchy: silver eel starting from larger inland regionally managed water bodies with no open connection to the sea (referred to the Dutch name: 'boezem');
3rd hierarchy: silver eel that start from large nationally managed water bodies such as the major lakes and main rivers.

Mortality rates by barriers were estimated for each hierarchy. Mortality rates induced by barriers in 'polder' water bodies were estimated using a meta-analysis of studies on a large number of pumping stations. Mortality rates induced by barriers in 'boezem' water bodies and large nationally managed water bodies were estimated using an analysis on the prioritised migration barriers for eel, based on current knowledge from telemetry experiments and detailed studies into mortalities induced by these barriers (Winter et al. 2013a). Water managers were asked for an update on measures taken to improve migration during 2013-2016, which led to reduced mortality estimates for that period. Barriers were assumed never to be in sequence within a hierarchy, but a proportion of silver eel was assumed to be transferred from a lower to a higher hierarchy of water body e.g. from the 1st to the 2nd hierarchy) implying that barrier mortalities could apply sequentially in this manner. An exception was made for eel that matured to silver eel in water bodies upstream of the main hydropower plants in the main rivers, for which separate mortality rates were estimated.

Estimates of the biomass of the standing stock of silver eel and yellow eel are necessary to estimate fishing mortality and to split the starting positions of silver eels into the three hierarchies of water bodies ("polder", "boezem" and "nationally managed"). To estimate the standing stock, a spatially explicit approach was taken to define the delineations and wetted areas of water bodies included in the assessment. Estimates of stock size were made in two different ways:

1) Static spatial model: Stock estimates were made on the basis of data from electric dipping nets, by scaling data on density (eel biomass per length class per swept area) to total wetted areas of water bodies.
2) Demographic model: For Lake IJsselmeer estimates of fishing mortality rates were made by fitting the yellow eel demographic model to a survey time series (electrotrawl survey) of catches per unit of effort per length class. The estimated fishing mortality rates were used to obtain estimates of the total standing stock of eel in the lake, by multiplying the reported retained yellow eel catches with the inverse of the estimated harvested proportion. The estimates of fishing mortalities in Lake IJsselmeer were also used for Lake Markermeer, Randmeren and Grevelingen.

The silver eel production per water body was estimated by splitting the total standing stock into length classes and multiplying the standing stock in each length class with the proportion of silver eel per length class. The maturity per length class was estimated using a maturity ogive (size at maturation) calculated from market sampling data. For the lakes for which no electric dipping net data were available, an estimate of 44% silver eel out of the total standing stock biomass was used based on the market sampling data.

The main data sets used in the stock assessment are:
1) **Retained catches** (landings) from commercial fishers. Since 2010 these landings are provided by the Ministry of Agriculture Nature and Food Quality (LNV) and are stored at WMR in a database ('Visstat').
2) **Market sampling**. Representative samples (usually 150-200 eels) are taken from retained catches from commercial fishers each year. Lengths of individual eels were measured in order to estimate length-frequency distributions of the landings. Furthermore, a number of eels were selected from each sample for dissection and the estimation of the following biological keys: maturity-at-length, weight-at-length and sex-ratio-at-length. Sex-specific growth curves have been estimated from age readings of ~350 eel otoliths, originating from eel from different areas in the Netherlands, including the large lakes and main rivers. The biological keys (*Chapter 2*) are used in the demographic model (*Chapter 3*) and in the static spatial model (*Chapter 4*).
3) **Surveys in regionally managed water bodies**. Eel sampling within the Water Framework Directive (WFD) waters was done following an EU certified protocol. In the assessments presented here only data from electrofishing with electric dipping nets was used. Sampled water bodies are representative for water types defined within the Netherlands based on WFD regulation. Data collection is managed and

stored by regional water boards. Electric dipping net data for recent years were obtained from ATKB (consultancy for water, soil and ecology) and several water boards. A total of ~4600 samples by electric dipping nets were available between 2006 and 2016, covering most of the combination of water boards and water body types. However, data from some regional water boards were missing in the analyses.

4) **Surveys in nationally managed water bodies**. The shores of the main rivers (Meuse, Rhine and their downstream counterparts) were sampled yearly using an electric dipping net.
5) **All water bodies which are included in the Dutch Water Framework Directive (WFD)** have been included in the assessments presented in this report, with the exception of coastal water bodies. The WFD (2000/60/EC (WFD)) has been established by the European Union as a legal framework for the protection and restoration of the aquatic environment across Europe by 2015. A total of 3402 water bodies form the main basis for the stock assessment. Drainage ditches are underrepresented in the set of WFD water bodies, and were added separately to the spatial model.

1.2 Structure of the report and flow diagram

As explained in the previous paragraph, the stock assessment methodology consists of a number of steps needed to make the final overall assessment. Summarizing, the steps leading up to the overall assessment are:

Chapter 2: In this chapter the biological keys (maturity-at-length, weight-at-length and sex-ratio-at-length) are estimated.

Chapter 3: In this chapter the demographic model for yellow eel for Lake Ijsselmeer is developed. The model is used for estimating the best possible spawner (escaped silver eel) production per glass eel, and the reduction in spawner production for a certain level of yellow eel mortality (*%SPR*).

Chapter 4: In this chapter the static spatial model is described. The chapter starts with a description of the water bodies and their main attributes (such as total wetted area) which are included in the assessment. For the larger, mostly nationally managed water bodies such as the main rivers and for the majority of smaller, mostly regionally managed, water bodies, data from surveys using electric dipping nets were available. Fishing operations using electric dipping nets usually take place only close (~1.5m) to shores of water bodies. A separate estimate is therefore needed for the standing stock in the wetted area further than 1.5 meters from the shore/bank, which is estimated as a proportion of the density of eel "within-shore" (<1.5 meters from the shore/bank). Because of uncertainties of the values for density proportion shore-offshore and electric dipping net efficiency three scenarios were chosen with different values of these two parameters. Also the stock assessment is done is for four time periods reflecting differences in fishing effort.

Chapter 5: In this chapter the standing stock biomass is estimated based the demographic model for Lake IJsselmeer (*Chapter 3*) and on estimates of the standing stock biomass of yellow eel and silver eel.

Chapter 6: In this chapter the Estimation of silver eel mortality due to barriers is described.

Chapter 7: The results from chapters 2-6 are subsequently used for the estimation of the final Dutch key stock indicators (*Chapter 7*).

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

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

The flow diagram below (*Figure 1-2*) gives a broad overview of the key steps in the stock assessment methodology, with reference to the chapters and key paragraphs therein.

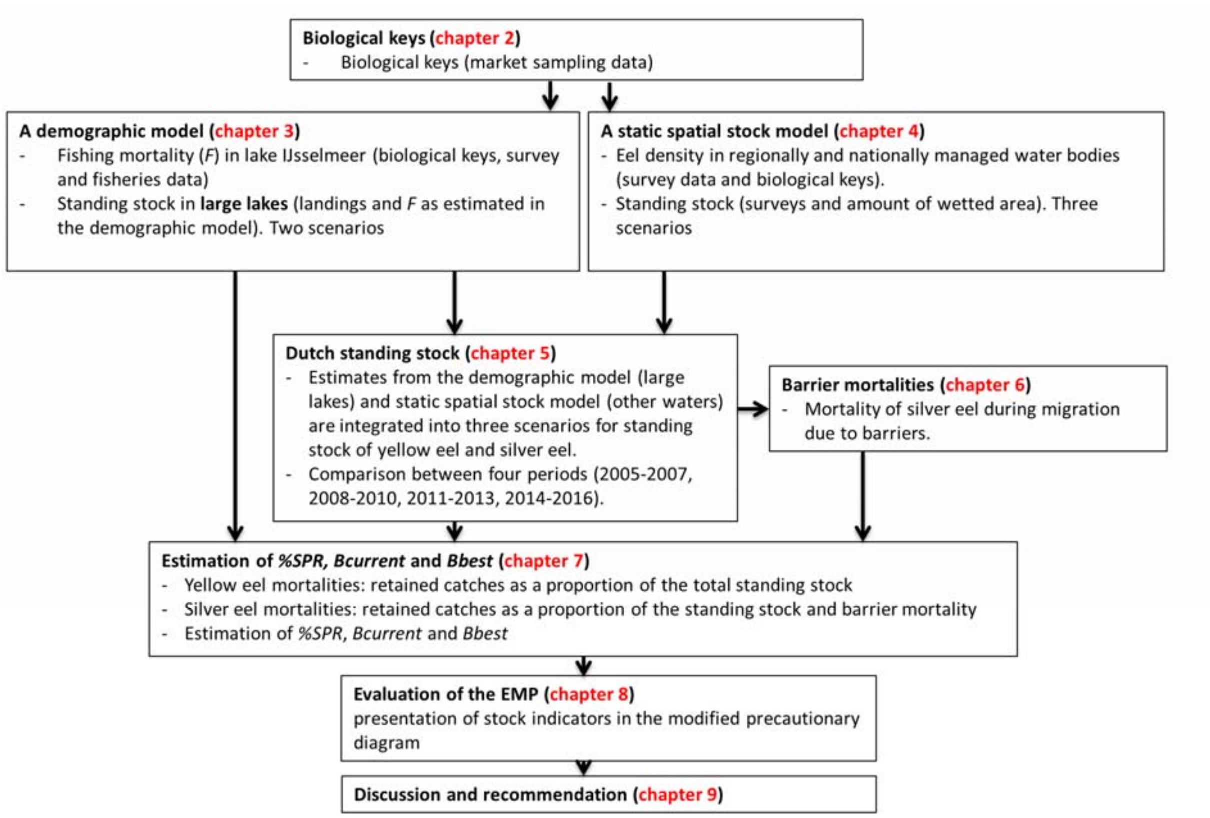

Figure 1-2 Flow diagram representing the key steps in the stock assessment methodology, and the structure of this report (with reference to chapters and key paragraphs therein).

2 Biological keys

In this chapter we present an update for the biological keys, as more biological data on eels were available since the previous assessment (van de Wolfshaar et al, 2015). Eel samples were taken from commercial catches (i.e., 'market samples') from different regions in the Netherlands (van Keeken et al. 2010, 2011). The number of market samples per region was taken proportionally to the catch size of the region. In total, 11067 individuals collected from 2009-2016 were used to assess the biological keys: sex, length, weight and maturity. Of 348 individuals the otoliths were analysed to assess inter annual growth increments. The biological keys are used in the assessment in the static spatial model to convert lengths to biomass and yellow/silver eel (*Chapter 4*) and in the demographic model (*Chapter 3*). Parameter estimates were based on all individuals, disregarding region, thereby using the largest possible data set, resulting in estimates representative for a national eel population (Bierman et al. 2012).

2.1 Sex ratio at length

The processes determining sex in eels are not well understood. Sex differentiation in eels is thought to be not, or only partly, genetically determined. Instead, environmental characteristics are thought to play an important role. Densities, either of recruits or adults, are likely candidates, with high densities leading to more males (Roncarati et al*.*, 1997; Davey and Jellyman, 2005; Huertas and Cerda, 2006; Han and Tzeng, 2006; Bark et al*.*, 2007). Sex-ratios in catchments can change over time (e.g. Laffaille et al*.*, 2006) and can differ markedly between local eel populations in different (parts of) water bodies (e.g. Oliveira et al*.*, 2001; Bark et al*.*, 2007). The sex ratio for the size class 30-35cm of Lake IJsselmeer and Lake Markermeer is shown in *Figure 2-1* and is illustrating a decrease in the proportion of males in recent years. This decrease in the fraction of males is taken into account in the demographic model (*Table 3-1*). The mean proportion of males in Lake IJsselmeer is 59% (2009-2016), and the mean proportion for Lake Markermeer is 41% (2009-2016).

The sex ratio at length was used in the static spatial model to split the survey data into males and females. Both the data on which the sex ratio at length was based as the survey data used in the spatial static model are from recent years (2009-2016). For the demographic model, which encompasses a much longer time series, this sex ratio at length was not used because the sex ratio has changed over time and data for a sex ratio at length are not available from before 2009. Hence for the demographic model a coarse assumption is made on the change in sex ratio with a decrease in the fraction males (*Table 3-1*).

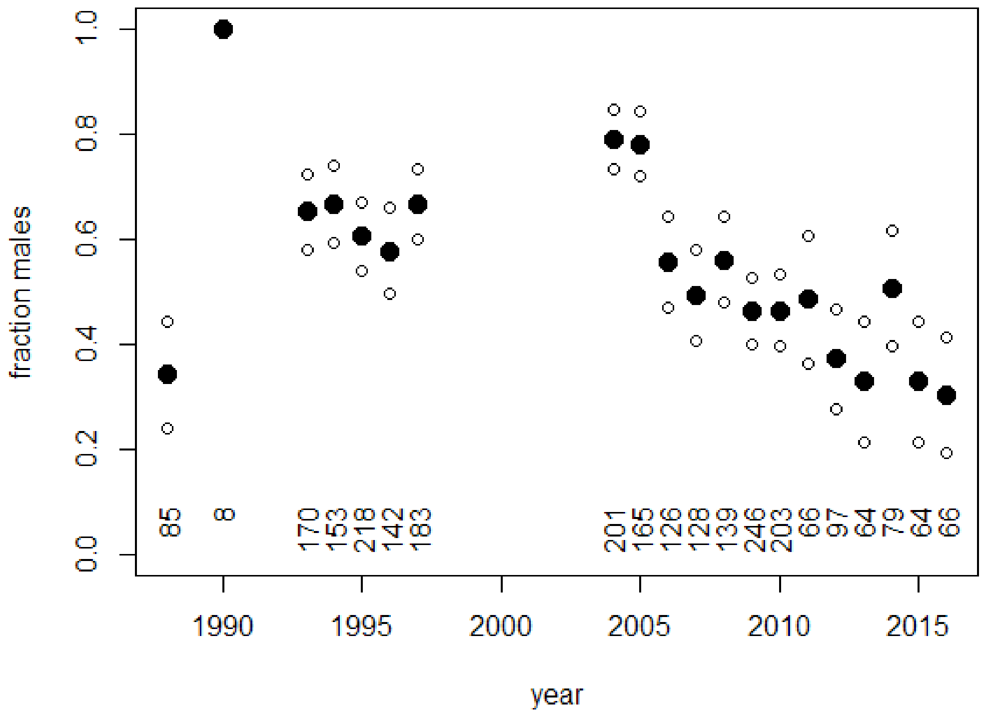

Figure 2-1 Fraction of males in the size class 30-35cm from the Lake IJsselmeer and Lake Markermeer market sampling and the 90% confidence interval. Numbers in the graph denote the number of individuals on which the fraction is based.

Sex ratio as a function of length could only be assessed for lengths larger than 20 cm, because determination of the sex is unreliable at smaller lengths. In addition, catches prior to 2011 were taken from sorted landings and therefore include only eel of 28 cm and larger. Samples from 2011 onwards were collected before sorting of the catch and included undersized individuals. Length classes (1 cm) with 5 or more individuals (of which the sex was determined) were taken into account, resulting in a dataset of 7958 individuals.

Per 1 cm length class, the percentage of females (*%F*) was determined (*Figure 2-2*). Subsequently, the length-sex ratio relationship was estimated for the length classes between 200 mm and 500 mm by linear regression. From 500 mm onwards, the percentage females was set at 100.
The percentage females is dependent on length (*L*, in mm) by:

$$200 < L < 500 \quad \%F = 100*(-0.54 + 0.003*L)$$
$$L >= 500 \quad \%F = 100$$

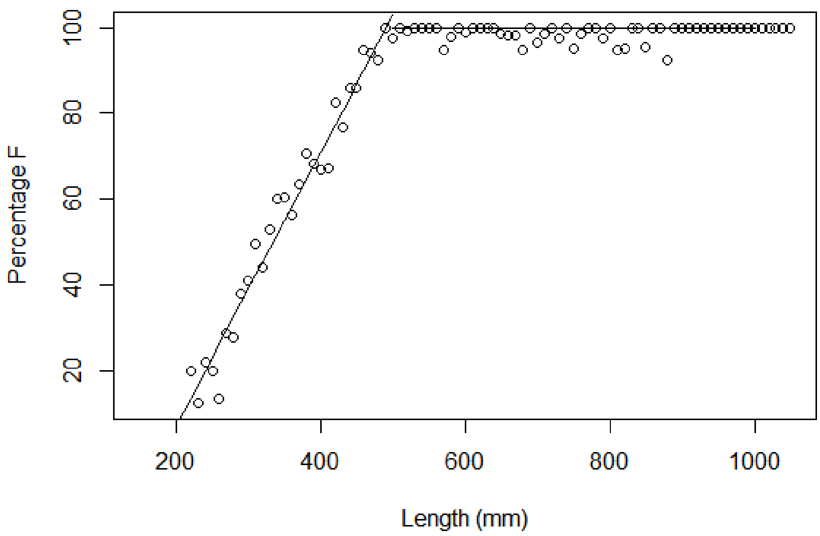

Figure 2-2 Percentage females based upon all samples from the market catches 2009-2016 (circles, in 1 cm classes). The linear fit (solid line) is highly significant for lengths between 200 to 500 mm ($R2 = 0.9721$, $p<0.0001$). Individuals of 500 mm and above are considered female.

2.2 Maturation at length

Because of the closure of the fishery during the silver eel migration since 2009, the sampling of the commercial catches could result in an underestimate of the proportion silver eel in the stock. However, sampling catches during the migration season could overestimate the proportion silver eel as these probably have a higher catchability 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 of the proportion silver eel upstream. These factors need to be considered when interpreting the proportion silver eel in the market sampling.

The maturity (yellow eel vs silver eel) at length was fitted for each sex separately with logistic regression (*Figure 2-3*).

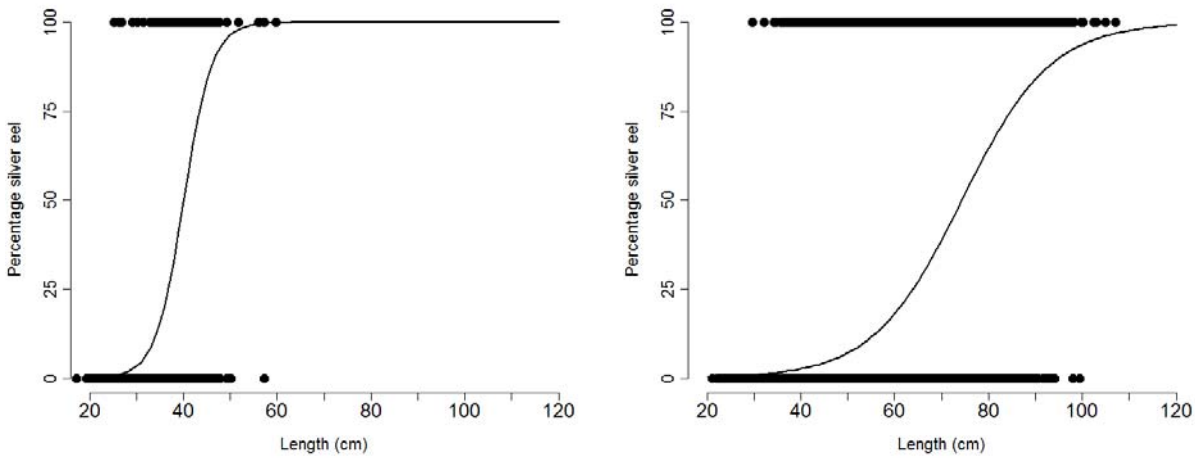

Figure 2-3 Observations (circles) and predicted percentages (lines) of silver eel in the retained catches, per length class (5 cm classes). Left: males; Right: females.

2.3 Weight at length

A length-weight relationship is calculated (market samples, 2009-2016):

$$Weight = exp(-14.29 + 3.178*log(L))$$

With weight in grams and length (L) in millimetres (*Figure 2-4*). The length weight relationship is used to estimate eel biomass given numbers-at-length.

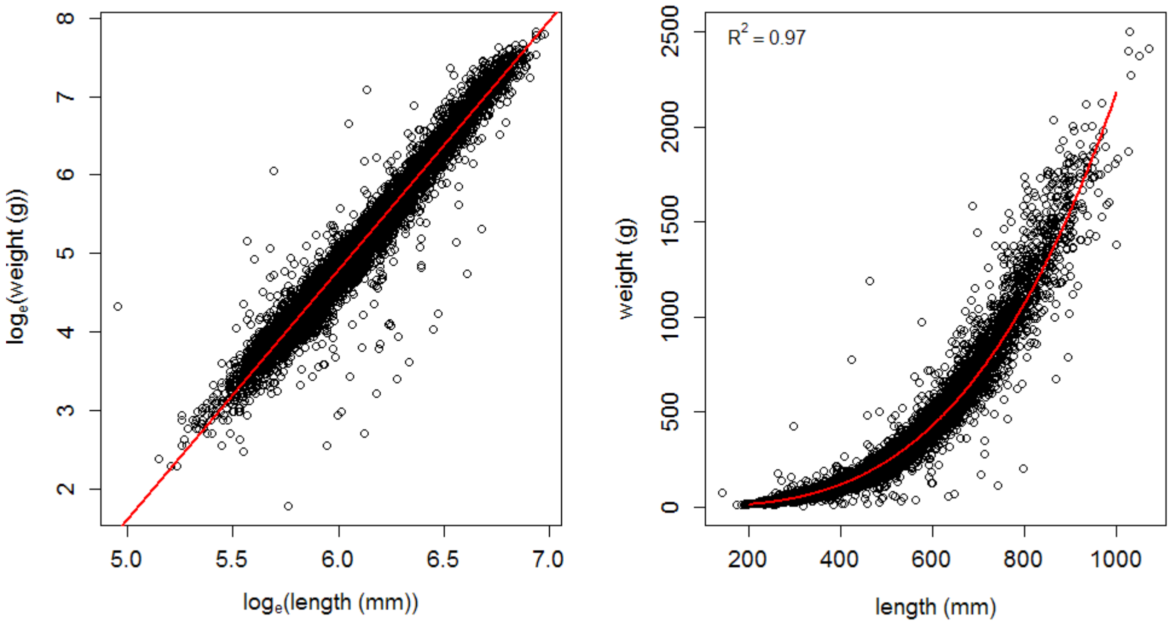

Figure 2-4 Length-weight relationship (red line) for eel based on market sampling data (2009-2016). The left figure is the same relationship as the figure on the right yet with a log-log scale.

2.4 Growth rate

Growth rates are analysed for the sexes separately. Growth increments were based on otolith reading from eels collected in 2009-2016 (348 eels, *Figure 2-5*). Individual growth curves were constructed using the relative distances between annual growth rings and scaling these to the total length of the eel (van Keeken et al., 2011). For the determination of growth curves and ages, the protocols set by the ICES workshop in Age Reading of European and American Eel 2009 (WKAREA) were used. It was assumed that glass eel enter the freshwater system at a length of 7.5 cm. The sex specific growth curve is defined as the cumulative average increment at age. This means that the average increase in length per age class is added to the average increase in the age classes before that. This is different from a curve based on the average length-at-age, where the average is taken per age (without taking prior years into account). Using average length-at-age might be biased for several reasons, among which size-selectivity of the gear used for the commercial catches. Using the cumulative growth increment results in a smoother growth curve and prevents 'shrinking' of individuals at older ages which does occur when using the average length-at-age in the demographic model (*Figure 2-5*). Because the cumulative growth increment is used, the growth curve can be higher than the observed growth patterns of especially slow growing, old, individuals (see for example *Figure 2.4* left panel).

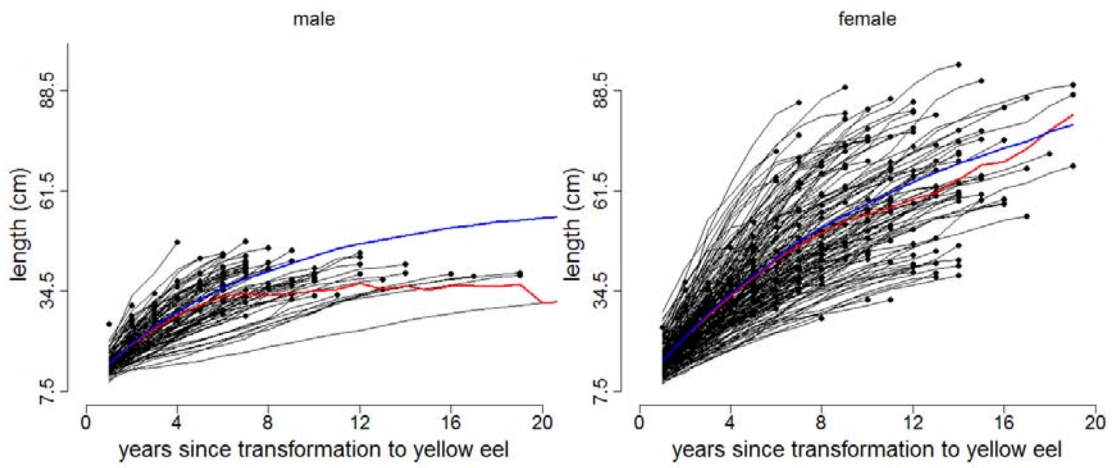

Figure 2-5 Growth curves of individuals (black lines), estimated by allocating the length of the eel (minus 7.5 cm of length assumed for glass eel entering the freshwater system) to ages in proportion to the relative measured width of the year-rings in the otoliths. The estimated cumulative growth curve is given in blue (used in the model). The average length-at-age curve is given in red. Curves are based on 114 males and 234 females.

The estimated growth curves are used in the demographic model as annual transition rates between length classes. The growth rate has an effect on the fishing mortality estimate of the demographic model. Growth influences maturation, with a decrease in growth rate, fish do not reach the same size per age class, and thus maturation occurs at older ages (since maturation is dependent on length, not age). Hence, individuals are present in the yellow eel population for a longer period of time and longer prone to fishing mortality (*F*). As a result of this, the value of *F* in the demographic model is influenced by the growth rate. However, whether slower growth leads to a higher *F* or to a lower *F*, depends on the glass eel index and field data. When survey catches are low, slower growth is more likely to result in a lower estimate of *F*. When survey catches are high, slower growth is more likely to result in a higher estimate of *F*.

2.5 Selectivity of the fisheries a length

In order to interpret length-frequency distributions of retained catches, and to predict the impact of fisheries mortality on the spawner-to-recruit ratio (% SPR), it is necessary to define the selectivity of the fisheries at length. Most of the commercial fisheries on eel takes place using fyke nets, with a legal minimum landing size of 28 cm. We assume that most catches of eel below 30 cm are returned because most fishermen return them into the water to grow, and that there is no mortality associated with catch and release. Furthermore, we assume that the fishery is fully selective at lengths above 30 cm. The selectivity-at-length of the fishery as assumed in the eel stock model is given in *Table 2-1*.

Table 2-1 Assumed selectivity of the fisheries at length.

Length class (5 cm intervals)															
10-15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	50-55	55-60	60-65	65-70	70-75	75-80	80-85	>85
0	0	0	0.5	1	1	1	1	1	1	1	1	1	1	1	1

2.6 Natural Mortality

For natural mortality an estimate is used of *M = 0.138* (per year) for all ages and lengths (Dekker 2000, van der Meer 2009).

2.7 Discussion

As more individuals are sampled the biological keys (sex, length, weight, growth) become more reliable. Compared to the previous stock assessment, for example, the estimate of the length-weight relationship did not change much compared to the current estimate. However, in order to estimate male growth, the average of all male growth increments were used for the current assessment. For the previous assessment only the average growth increment of the males younger than six years old were used because of the small sample size of older individuals and the difficulties of the otolith readings of older males. Now, the samples size has increased and all increments were used. This resulted in lower growth rates for males in the current assessment, which affects the estimates of *F* in Lake IJsselmeer (the demographic model). Maturation curves barely differed from those used previously, and are not likely to have affected the estimate of *F*.

3 A demographic model for yellow eel

A demographic population model was developed to estimate fishing mortality rate (*F*) for Lake IJsselmeer (Bierman et al. 2012). The fishing mortality rate and the registered landings are used to estimate the standing stock in the lakes. The results are used as input to estimate the Dutch eel stock and the *%SPR* (*Figure 1-2*).

3.1 Demographic model

The demographic model follows eel cohorts on an annual basis. The model is based on the assumption that the lake is a closed system (Oeberst and Fladung, 2012; Ciccotti et al., 2012). Glass eel entering the lake are assumed to stay in the lake and silver eel are assumed to be produced locally. This is a strong pragmatic simplification. However, strong assumptions would also have to be made if influxes from other areas would be included in the model.

Each new cohort in the demographic model is based on the glass eel index monitored at Den Oever (*Figure 3-1*). The index is based on numbers per haul and needs to be converted to numbers, which is done by fitting a constant conversion factor in the model. In previous stock assessments this was a single constant, used for the whole period. For the current stock assessment this procedure led to a very bad fit of the model with unusually high fishing mortality. The fit was reasonable for small individuals, but was poor for the intermediate size classes. It was decided to fit two conversion factors, with a break at 2000, based on changes in the length frequency distribution of eel as observed in the survey. The fitted conversion factor for the period > 2000 is over three times lower than the fitted conversion factor for the period before 2000. This means that of each unit of glass eel (from the index) 1/3 less are available to start a new cohort in the model. With this adjustment the model performs better in terms of the confidence interval, the goodness of fit and produces estimates of fishing mortality in line with those of previous assessments.

In the demographic model, each year individual eels grow, mature and die based on length- and sex-specific biological keys (*Chapter 2*). All eels suffer from natural mortality and eels larger than 30cm also from a length-based fishing mortality (*Chapter 2, Table 2-1*). Eels that reach the silver stage are excluded by the model. The cohorts are followed through time, resulting in an annual length-frequency distribution.

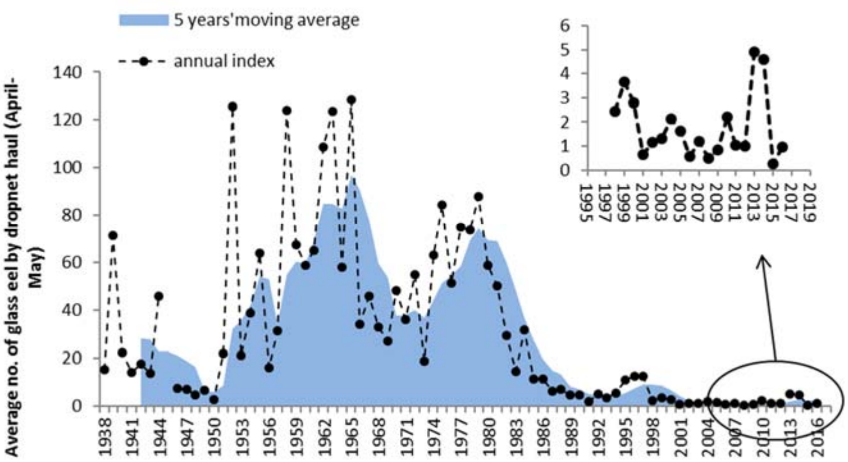

Figure 3-1 Glass eel numbers per haul presented as annual index (black circles) and as a 5-year moving average (blue shade), monitored at Den Oever, The Netherlands.

Let $N_{a,j,t}$ be the number of eel per hectare of age *a* (years since transformation to yellow eel), of sex *j* in year *t*. When entering the model, glass eels are assumed to have age *a*=0 and a length of 7.5 cm. For yellow eel (1 year old or more), let *L(a,j)* be the length class (5 cm intervals; starting from 10 cm) of an eel of sex *j* and age *a*. The length class at age *a* is a function of the growth curve of sex *j*. The population model is given by:

$$N_{a,j,t} = N_{a-1,j,t-1}\, \mathrm{e}^{\left(-M-Fs_{L(a,j)}\right)} \left(1 - q_{j,L(a,j)}\right)$$

With *M* and *F* parameters for the natural and fisheries-induced mortality respectively, $s_{L(a,j)}$ the selectivity-at-length of the fisheries, and $q_{j,L(a,j)}$ the probability of maturation for eels of sex *j* in length class *L*.

This type of model is called 'demographic' model, and is used regularly to describe eel populations (e.g. Dekker, 2000; Oeberst and Fladung, 2012; Ciccotti et al., 2012). Here, the model is used to estimate the fishing mortality, given the glass eel entering the population and the biological parameters. The estimates are done by fitting the predicted length frequency distribution from the model to the length frequency distribution observed in a fisheries independent survey in Lake IJsselmeer. The survey takes place in October and November each year, which is in the silver eel migration season. Therefore, the length-frequency distribution is split into a yellow eel and a silver eel fraction based on the proportion mature eel per length class (*Chapter 2*). The model fit is based on the yellow eel fraction only because of the assumption that mature eel instantaneously leave the model population. The model is also used to predict the spawner-to-recruit ratio (%SPR) for a given stock and fishing mortality (*Chapter 7*).

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

3.2 Varying sex ratio and fishing effort

For Lake IJsselmeer fishing mortality was estimated using a change in sex ratio over time for each new year class, with a decreasing proportion of males (*Table 3-1*). This is based on the decrease in time in the proportion males as observed in the data (*Figure 2.1*). Changes in sex ratio could depend on density dependent processes (e.g. Roncarati et al. 1997, Davey & Jellyman 2005, Bark et al. 2007).

Table 3-1 Proportion of males for different periods.

Period	Proportion Males
<1990	0.7
1991-2005	0.6
2006-2010	0.5
2011-2016	0.4

The model was run for different periods because of sharp changes in 'potential fishing effort' through time (*Figure 3-2*): (*i*) the number of fishing permits has decreased over time in a stepwise manner, with three major changes in total number of permits (Bierman et al. 2012), and (*ii*) since 2009 fishing is prohibited during the silver eel migration period (three months, September-November), thereby reducing the fishing season by one-third. The reason this is called 'potential fishing effort' is because the number of permits is known, but the realized effort is not known. This leads to five periods with potentially distinctly different fishing effort; four time periods based on reduced permits and 1 time period based on shortening of the fishing season (period 1: 1955-1989; period 2: 1990-1999; period 3: 2000-2005; period 4: 2006-2008; period 5: 2009-2016). For each of these fishing periods the fishing mortality is estimated.

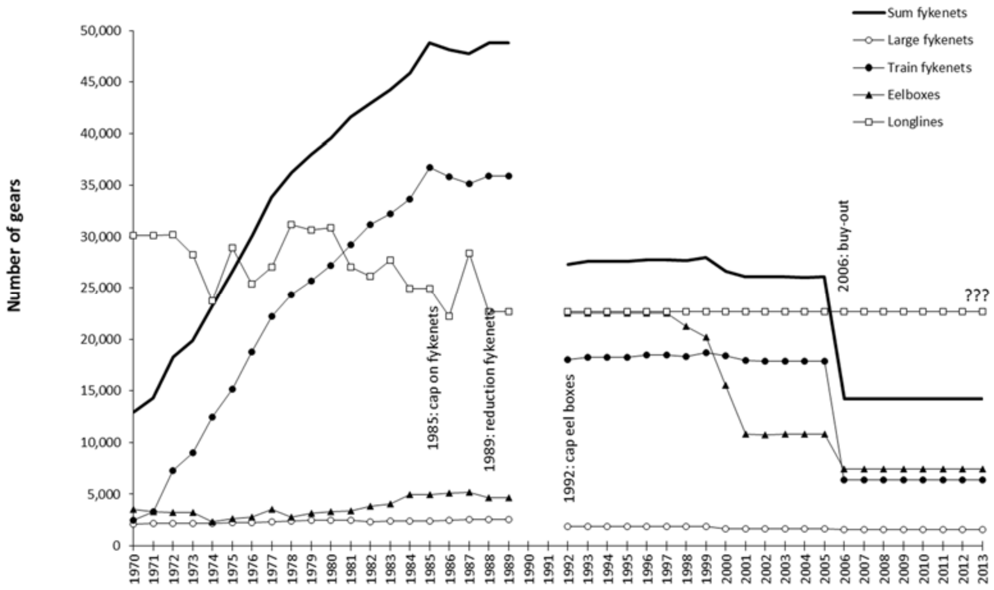

Figure 3-2 Changes over time actual (before 1991) and nominal (after 1991) fishing gears in Lake IJsselmeer and Lake Markermeer. The nominal number of fish gears is the number of legal, registered fishing gears that could potentially be used, it is however, unclear how many of these nominal gears are actually used in the fishery. Note further that the number of longlines are not registered, only the number of long line licenses, hence the uncertainty ("?") about the number of longlines in the fishery.

In order to estimate the fishing mortality, the model length-frequency distribution is fitted to the length-frequency distribution observed in the survey data of Lake IJsselmeer. Since the survey takes place in October-November, the biological keys were used to separate yellow eel and silver eel. Only the yellow eel portion of the survey data was used in the fit. Fishing mortality estimates were based upon time series of length classes of both the model result and the survey.

3.3 Model fit and estimated fishing mortality

The model predictions (number per square km per size class) are presented in *Figure 3-3*. For the smallest size classes (15-20cm and 20-25cm), the prediction captures changes in numbers at a coarse level and the high level of variance is not reflected in the prediction (*Figure 3-3*). For the intermediate size classes (25-30cm and 30-35cm) the prediction is more accurate. The fit for the larger size classes (40-45cm and >45 cm) are less accurate. For size class 40-45cm, the prediction is overestimating the observed numbers until the beginning of the 2000's. For size class >45cm, the prediction after 2005 is less accurate, and the observation of increased numbers is not picked up by the model. This is also reflected in *Figure 3-4*, illustrating that the initial increase in mean length during the early 2000's is captured by the model, but the further increase in recent years as observed from the data is not captured. Interestingly, in the size class 40-45cm such an increase is not observed, only in the larger size classes (>45 cm). Because this increase is not observed in the smaller size classes, but only in the larger size classes, this could indicate that (some of) the larger eels do not originate from Lake IJsselmeer, but come from the rivers. This increase in mean length partly corresponds with the period of seasonal closure of the fishery in 2009.

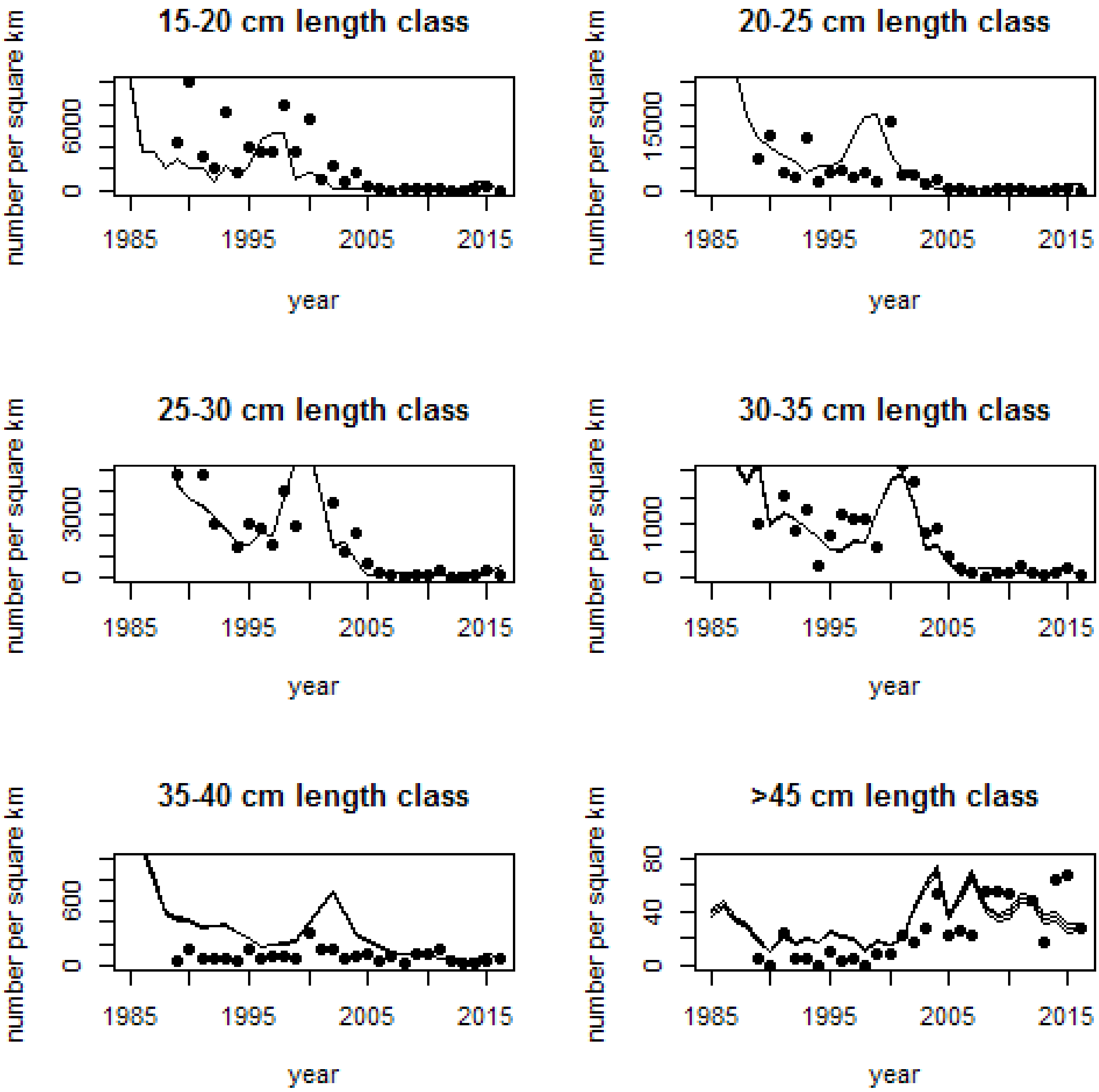

Figure 3-3 Field data and model predictions (number per km^2 per size class) for Lake IJsselmeer, given the F estimates presented in Table 3.2.

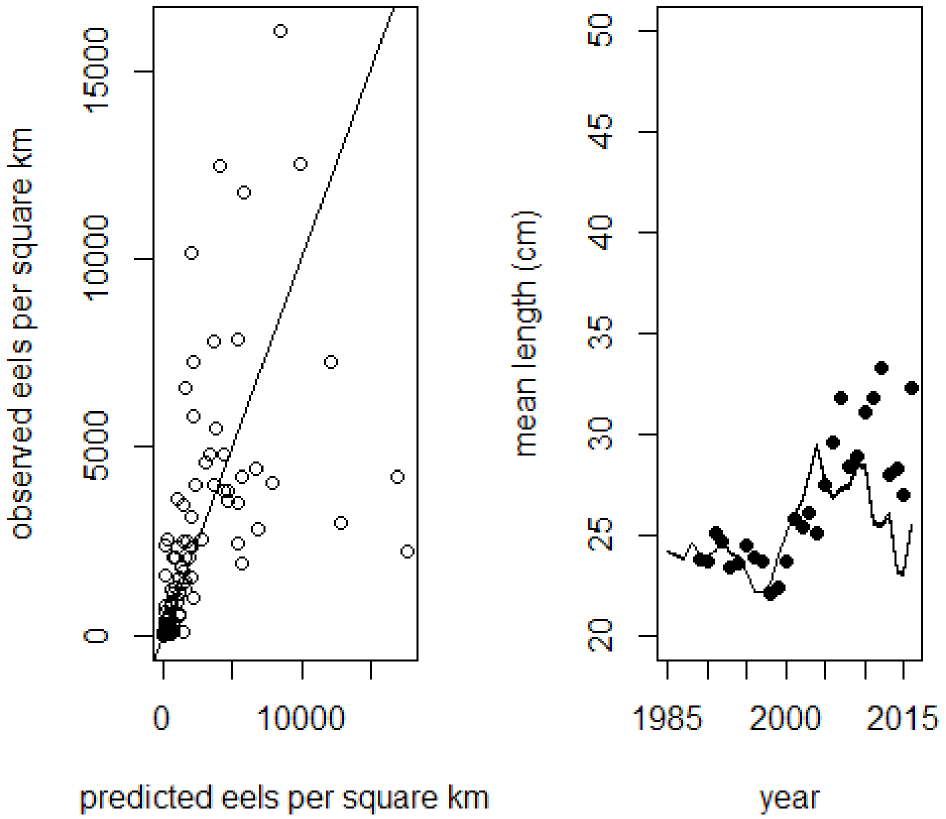

Figure 3-4 Observed (survey data) and predicted (model results) number of individuals per surface area (left) and the observed and predicted annual mean length (right).

The estimated fishing mortality for Lake IJsselmeer is given in *Table 3-2*, for the five periods with different potential fishing effort (see previous paragraph). The fishing mortality (*F*) for Lake IJsselmeer decreases over the first four periods but increases somewhat again for the last period.

Table 3-2 Model estimated median fishing mortality estimates for Lake IJsselmeer for periods varying fishing efforts, and the 90% confidence interval of the likelihood in brackets indicating variance (after 12000 iterations). The smaller the difference between percentiles, the better the fit.

	Fishing mortality (*F*)
<1989	1.29 (1.23-1.30)
1990-1999	0.92 (0.92-0.94)
2000-2005	0.81 (0.80-0.83)
2006-2008	0.29 (0.27-0.31)
2009-2016	0.34 (0.32-0.36)

3.4 Standing stock of large lakes

The four major lakes are Lake IJsselmeer, Lake Markermeer, Randmeren and Grevelingen (*Figure 3-5*). For Randmeren and Grevelingen, survey data for eel were lacking. The standing stock for these lakes was estimated using commercial landings from each lake and fishing mortality from Lake IJsselmeer as estimated in the demographic model.

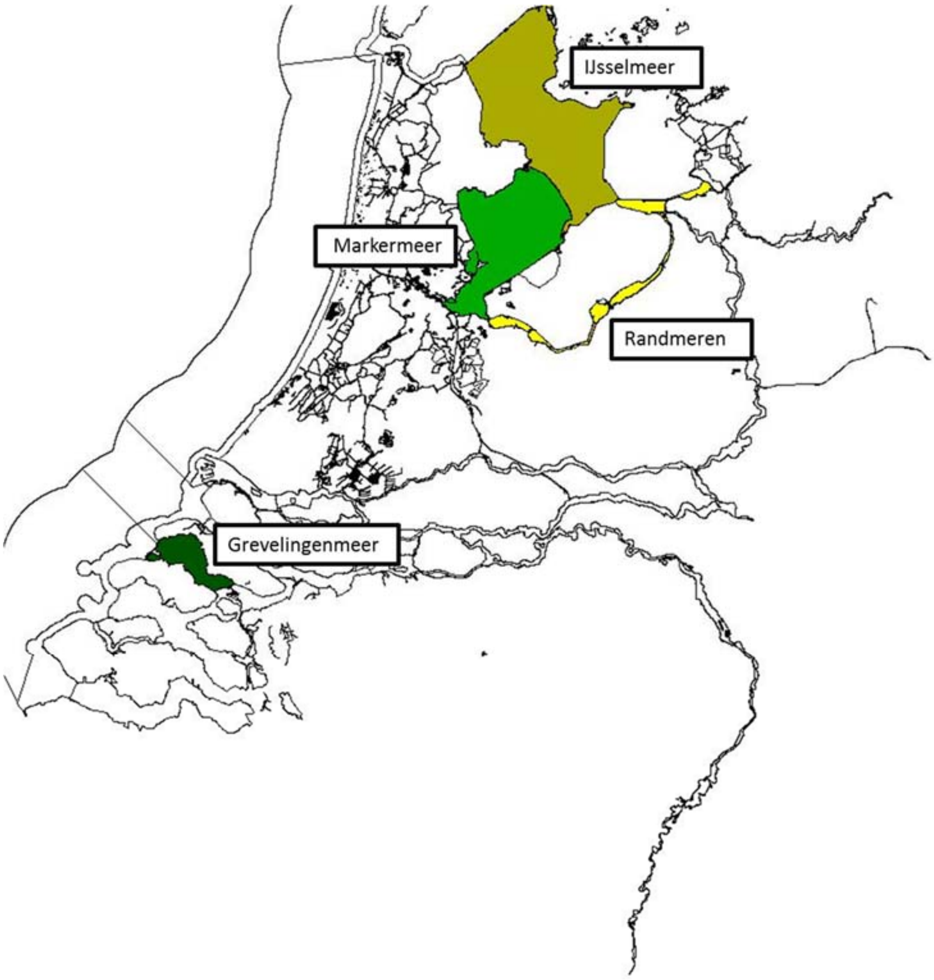

Figure 3-5 The four large Dutch lakes for which standing stock was estimated via landings and estimated fishing mortality.

3.4.1 Commercial landings

Different data sets of commercial landings data exist in the Netherlands, which do not overlap in time, regions or estimated landing size. The data from the Ministry of Agriculture, Nature and Food quality (LNV) is available since 2010 and contains data on four large lakes: Lake IJsselmeer and Lake Markermeer combined, Grevelingen and Randmeren.

Available data were collected from different sources and presented in *Table 3-3*. Catch data from the Product Board (PO) is available from 2005 until 2013 and covers Lake IJsselmeer and Lake Markermeer. However, these landings are lower than those reported to LNV. LNV data are considered more reliable than those of the PO, and hence these were used in further analyses. For those years for which no LNV data are available, the PO data were scaled with the ratio of LNV-PO data for the overlapping years (ratio of 1.48).

For Randmeren only LNV data were available, and hence an estimate could be made from 2010 onwards. For Grevelingen, yellow eel and silver eel landings were available from DUPAN (‘Stichting Duurzame Palingsector Nederland’, made available by Witteveen & Bos), for the years 2002-2012. For the period 2011-2013 data from DUPAN and LNV were combined, assuming for 2013 (LNV data) the same silver eel to yellow eel ratio as reported by DUPAN for 2012. From 2014 onwards only LNV data were available for Grevelingen. The estimated retained catches (landings) for the lakes are shown in *Table 3.3*.

The percentage yellow eel, in the total landings was calculated using the length-frequency distribution in the market sampling of Lake IJsselmeer and Lake Markermeer (2011-2016) and the biological keys (*Chapter 2*). This resulted in an estimation that 60% of the retained catches in biomass consists of yellow eel and 40% consists of silver eel. This percentage was used to convert the reported total retained catches of the large lakes (with exception of Grevelingen 2005-2013 for which separate estimates were available) into yellow eel and silver eel retained catches.

*Table 3-3 Estimated mean yearly retained catches (tonnes) of all eel larger than 30 cm (yellow and silver) in the large lakes, for three time periods, as estimated from LNV/PO-data. * For Lake Grevelingen yellow eel (left inside parenthesis) and silver eel (right inside parenthesis) were available separately.*

All eel > 30 cm	Period	Mean yearly landings (tonnes)
IJssel-/Markermeer	2005-2007	321
	2008-2010	160
	2011-2013	164
	2014-2016	158
Grevelingen	2005-2007	63 (22/41) *
	2008-2010	19 (11/8) *
	2011-2013	9.5 (9/0.5)*
	2014-2016	1.1
Randmeren	2010-2013	11
	2014-2016	11

3.4.2 *Standing stock*

Estimates of the standing stock of yellow eel and silver eel (*Table 3-4*) were subsequently calculated by dividing the landings by the log of estimated fishing mortality (*Table 3.2*). These data are integrated into an estimate of the total Dutch standing stock (*Chapter 5)*. The values have changed since the last assessment (van de Wolfshaar et al 2015) because of 1) Lake Markermeer is not assessed; instead, the *F* value for lake IJsselmeer is used to calculate the standing stock for Lake Markermeer; 2) a longer times series is used; 3) more data was available on growth; 4) the demographic model was adjusted to include two conversion factors for the glass eel index and 5) the method of calculating the length frequency distribution was improved. These changes affect the model fit and therefore also the values in the previous assessments are changed (van de Wolfshaar 2015); these changes are substantial and reflect the variability in the model fit between periods.

Table 3-4 Estimated mean yearly standing stock of yellow eel larger than 30 cm and silver eel in the large lakes, per time period.

	Period	Mean yearly standing stock (tonnes)	
		Yellow eel (> 30 cm)	Silver eel
IJssel-/Markermeer	2005-2007	561	375
	2008-2010	232	154
	2011-2013	236	158
	2014-2016	228	152
Grevelingen	2005-2007	65	120
	2008-2010	26	19
	2011-2013	21	1
	2014-2016	2	1
Randmeren	2010-2013	16	11
	2014-2016	16	10

3.5 Discussion of the demographic model

The increase in eel numbers >45cm in Lake IJsselmeer in recent years is not captured by the model. A logical explanation for the increase in abundance of large eel in the survey data would be the closure of the fishing season during silver eel migration effective from 2009 and the migration of larger eel from upstream of the rivers which, due to high levels of toxicants, are closed to eel fisheries since 2011. Alternatively, these large eel could originate from Germany where eels are stocked.
In the previous assessment, the demographic model was also fitted to the data of Lake Markermeer. However, in recent years the numbers of eel in the survey have decreased to such low numbers that the length frequency distribution was not good enough to fit the model. As a result, the fit of the demographic model for the Lake Markermeer is very poor and the estimate is therefore not used.

The demographic model was adjusted to include two conversion factors for the glass eel index, resulting in improved model performance and more reliable *F* estimate. That raises the question if in Lake IJsselmeer there have been changes in either the inflow from Den Oever into the lake, changes in glass eel mortality (e.g. predation, disease, pollution) or changes in the migration of glass eel to other water bodies.

Compared to the previous stock assessment, the estimate of *F* has changed for each period. The main causes are; 1) a longer times series is used (until 2016 instead of 2013) which also affects the fit in previous years of the time series; 2) more data on growth resulted in slower growth of males 3) the demographic model was adjusted to include two conversion factors for the glass eel index and 4) the method of calculating the length frequency distribution was improved. The slower growth (point 2) results in males taking longer to turn into silver eel and consequently are longer subject to fishing mortality. The model was checked for the effect of changes in the biological keys, by running it with the biological keys of the previous assessment (2015). This resulted in a lower *F* in 2006-2008.
The estimates of *F* for the last period is much higher compared to the estimate of the previous stock assessment. This is due to a further decrease in abundance in the size classes < 45 cm of the survey data in the most recent years (2014-2016). The new estimates of *F* also result in new standing stock estimates for all assessed periods.

4 A static spatial model for yellow and silver eel

Given the complexity of the Dutch water system with many small catchments and regional-level management a GIS approach was used for the regionally managed waters and the nationally managed rivers and some smaller lakes (see also Bierman et al. 2012 and van de Wolfshaar et al. 2015).

Only the main rivers (Rhine, Waal, Meuse and IJssel) and the large lakes (Lake IJsselmeer, Lake Markermeer, Grevelingen and Randmeren) are managed at a national level in the Netherlands. All other water bodies are managed regionally by the water boards. The monitoring of these water bodies is also markedly different, which led to the necessity to different methods of standing stock estimation for nationally and regionally managed water bodies.

The regionally managed water bodies make up 65% of the total freshwater surface area (PBL, 2010). These waters are surveyed in a regular and standardised manner since the implementation of the European Water Framework Directive (WFD) in 2000 (2000/60/EC). The nationally managed rivers are monitored in a standardized manner since 1990. For the four large lakes good quality survey data were not available (Grevelingen and Randmeren) or deemed unsuitable for a swept area estimate (Lake IJsselmeer and Lake Markermeer). Therefore, stock estimates for the large lakes were based on commercial catch data and fishing mortality estimates which are estimated in the demographic model (*Chapter 3*).

For both regional and national waters, the estimation of the standing stock had two distinct elements: the density of eel (kg per hectare) and the amount of water surface area. The density was divided into silver eel and yellow eel based on the maturity ogive (*Chapter 2*). The standing stock was estimated for three scenario's, with different assumptions on the catch efficiency of the survey gear and the spatial distribution of eel in the waterbody. In this chapter these scenarios will first be described. Second, the estimations for the regionally managed waters and for the nationally managed waters will be presented. These estimations are used as input for the Dutch eel stock (*Chapter 5*).

4.1 Three scenario's for the static spatial model

Three scenarios were run with different catch efficiencies of the electric dipping net and different habitat preferences.

4.1.1 Catch efficiency

The catch efficiency of survey gear is difficult to assess. Catch efficiencies of electric dipping nets depend 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). In this study, a catch efficiency of 20% is used for the electric dipping net, as set by the Dutch "Stichting Toegepast Onderzoek Waterbeheer", the research platform for the Dutch regional water managers (Handboek Visbemonstering, STOWA 2003). Estimates of catch efficiencies of eel are scarce in the scientific literature and may be specific to the type of water body, habitat, and gear. Naismith & Knights (1990) assumed a catch efficiency for eel using electrofishing gear of 27% in a river, whereas Baldwin & Aprahamian (2012) estimated efficiencies of approximately 60% in small rivers. Aprahamian (1986) showed size-selective effects of electro-fishing, with mean probabilities of capture from 0.36 for the smallest eels to 0.59 for the largest. Carrs et al. (1999) reported estimated capture probabilities of 0.715 and 0.751 for lochs and streams respectively. Stevens et al. (2009) in an evaluation of the Belgian eel management plan assumed catch efficiencies of 66%.

4.1.2 Habitat preference

Habitat preference is an important factor when scaling biomass in the borders of a water body, to the biomass for an entire water body. In the simplest case survey samples are scaled linearly to water body surface area. This method assumes eels have no habitat preference. However, eels may prefer the littoral over the open water. Most survey samples were obtained during fishing operations with electric dipping nets near the shores of lakes or banks of rivers, streams or canals. The electric dipping net data is therefore taken as representative for eel densities near the shores or banks, whereas eel densities further from shores or banks are likely to be lower (Jellyman & Chisnall 1999, Schulze et al., 2004). Therefore, a correction was used to account for differences in eel density between the littoral zone ('inshore') and the open water ('offshore').
Literature on how eel is distributed over a water body is scarce and focusses on the relation between eel density and the distance to the shore, mainly in lakes. Contradicting results were found for lakes; Chisnall & West (1996) found that eel densities off shore in New Zealand lakes were on average 40% of those inshore; Schulze et al. (2004) found a decrease in number with depth for a reservoir, but did not take distance to shore into account; Jellyman & Chisnall (1999) and Yokouchi et al. (2009) found a positive relationship between catch per unit effort and distance to shore. An unpublished, report of the Inland Fisheries Ireland Eel Monitoring Programme on 13 Irish lakes found differences between lakes and overall no relationship between density and distance to shore (Oleary, unpublished). In the national eel management plans, different relations are used. In Belgium, the biomass near the shore is set to be a fraction (up to roughly 33%) of the total biomass in a water body (Stevens et al., 2009). In France no difference is made between shore and non-shore areas in rivers given the lack of evidence otherwise (Briand, unpublished).

4.1.3 Three scenarios

Estimates of standing stock were computed using three different scenarios (*Table 4-1*) that differ in catch efficiency of the electric dipping net and in the habitat preference of eel. In these scenarios, catch efficiencies and habitat preferences vary according to results from the literature (see above). For the catch efficiency, while we use 20% as our best guess estimate, we also compute estimates in a scenario in which catch efficiencies were assumed to be 66%. For the habitat preference, we assume the density in the offshore area to be 33%, 50% or 66% of the densities in the inshore area (i.e., within 1.5 meters of shores/banks).

In scenario 1 the highest catch efficiency (66%) and lowest proportion of eel in the offshore area compared to the inshore area (33%) is used. This scenario will lead to the lowest estimated standing stock of eel. In scenarios 2 and 3 the best guess estimates for catch efficiency are used (20%), with the proportion of eel in the offshore area compared to the inshore area of 50% and 66% for scenario 2 and 3 respectively. Scenario 3 will therefore lead to the highest estimates of standing stock. Scenario 2 is the best guess estimate. All final calculations will be made with scenario 2, unless stated otherwise.

Table 4-1 The three main scenarios used in the approach to stock assessment in which survey data are scaled to wetted areas. A best guess of 20% for catch efficiencies was used with an upper limit of 66%. Densities in areas of water bodies outside 1.5 meters of the shore/bank ("offshore area") were assumed to be either 30%, 50% or 66% of densities within 1.5 meters of the shore/bank ("inshore area").

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

4.2 Regionally managed water bodies

4.2.1 GIS data

The eel biomass in the regionally managed water bodies was assessed in the same way as presented in Bierman et al. (2012) and van de Wolfshaar et al. (2015). It is based on GIS information of Water Framework Directive (WFD) water bodies and the WFD fish sampling. The regional management of waters is executed by so called water boards (*Figure 4-1*).

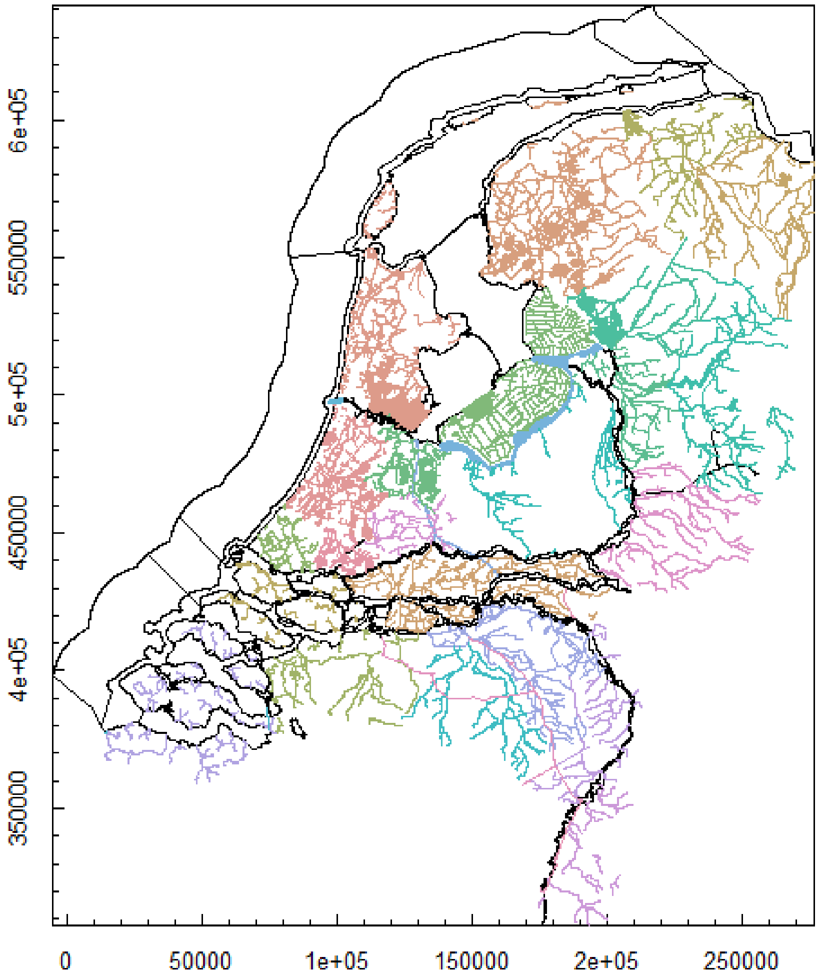

Figure 4-1 Water bodies included in the model, the colors represent different management regions, in black are the nationally managed waters

In the Netherlands, all WFD water bodies are assigned to a water body type. Data were assimilated per water body type (see also Bierman et al. 2012 and van de Wolfshaar et al. 2015). One of the water body types is 'ditches' (M1a and M2), of which only 0.5% are surveyed within the WFD survey program and therefore additional information was needed to calculate the total length and surface of this category. The width of the ditches is included based on the TOP10 map (RWS, Winter et al. 2013a, *Table 4-2*). The TOP10 map gives detailed information of all waters of the Netherlands. Similar to the WFD maps, the TOP10 maps include a map with polygons and a map with line elements. In the line element map the lines with a width between 0.5m and 6.0m were considered as ditch. In the polygon map the category 'waterway' (width more than 6m) was considered to be a ditch. If water bodies were already accounted for in the WFD map, they were discarded in the TOP10 map to prevent double entries.

Table 4-2 Overview of the length (in km) and surface area (in ha) for the three types of ditch, after removal of WFD water bodies. For the category 'waterway, >6m width' the surface area provided by the polygon map was used. For the line categories width 0.5-3 m and width 3-6 m an average width of 1.75 and 4.5 m was used, respectively.

Category	Length (km)	Surface area (ha)
0.5-3.0 m	148116	25920
3.0-6.0 m	24090	10841
> 6.0 m	39627	22681
Total	211834	59441

4.2.2 Survey data

Eel sampling within the regionally managed WFD waters was done with an electrofishing gear, following an EU certified protocol (STOWA Handboek Visstandbemonstering 2003). Sampled water bodies are representative for water types as defined in WFD regulation.

Dutch regionally managed water bodies fall within one of the 22 water boards of the Netherlands. Sample data were obtained from the companies hired by the water boards to conduct WFD fish sampling and/or from the water boards directly. While many water boards provided data timely and ready to use, some water boards provided data that needed processing; data lacking required information or did not provide data at all. For different reasons some data could unfortunately not be used in the analysis (*Table 4-4).* However, not all water boards sample every year, and data on a yearly basis is not necessarily expected.

Sampling locations were included if they were located within WFD water bodies as defined in the Polygon and Line maps. This was checked using the geographic coordinates of the electro fishing sampling event. Firstly, coordinates which fell into a polygon were assigned to that polygon. Secondly, for the sampling events which could not be assigned to a polygon, the distance to line segments was computed, and the sampling event was assigned to the nearest line segments as long as this was within 25 meters of the sampling occasion. Thirdly, for all remaining sampling events without a match based on GIS the water body names given at the time of the data collection were used. For regional waters, this results in 4612 electrofishing events that were used for the eel assessment (*Table 4-3*). These cover the period 2006-2016 and their locations are presented in *Figure 4-2*.

Table 4-3 Number of electro fishing events per year available in the regional data set.

2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
373	319	218	538	424	566	623	528	359	411	252

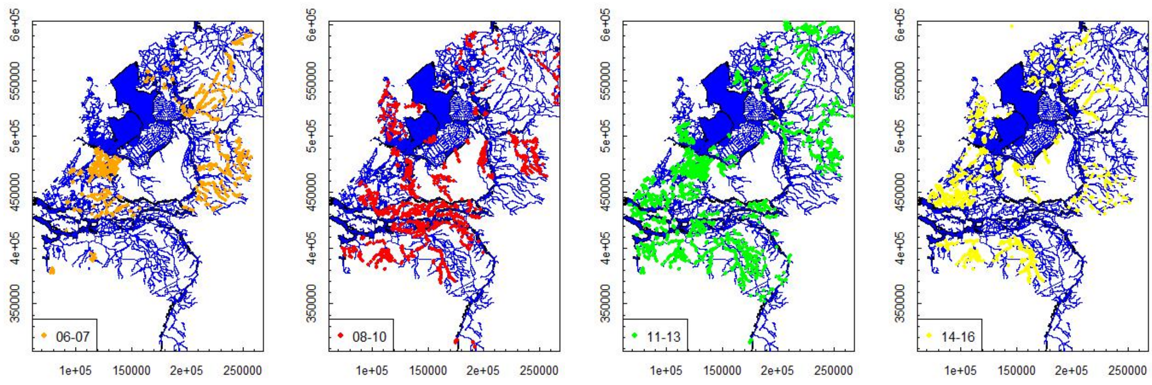

Figure 4-2 Geographical location of available sample points across the Netherlands in Water Framework Directive (WFD) -waters for the different periods, 2006-2007, 2008-2010, 2011-2013 and 2014-2016. Some samples did not have coordinates. They are not shown in the map, but have been included in the analysis when successfully linked to a water body.

The sampling effort differs between periods and regions (*Figure 4-2*). Data from some regions is lacking or could not be used, while from other regions data are available for all periods. Also, the regional coverage is highly variable (*Table 4-4*).

Table 4-4 Data per water board and per year that were used in the analysis (green colour). Note that not all water boards sample each year. For some water boards data was available, but could not be used for different reasons (see text).

Water board	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Aa en Maas											
Brabantse Delta											
De Dommel											
Groot Salland											
Hollandse Delta											
Hoogh. Amstel, Gooi en Vecht											
Hoogh. De Stichtse Rijnlanden											
Hoogh.Hollands Noorderkwartier											
Hoogh. van Delfland											
Hoogh. van Rijnland											
Hoogh. van Schieland en de Krimpenerwaard											
Hunze en Aa's											
Peel en Maasvallei											
Reest en Wieden											
Rijn en IJssel											
Rivierenland											
Roer en Overmaas											
Vechtstromen											
Waterschap Vallei & Veluwe											
Wetterskip Fryslân											
Zuiderzeeland											
Noorderzijlvest											

The variability in the area sampled is large between water types (*Table 4-5*, see *Appendix A* for a description of the types). Sampling effort varies greatly between water types (*Table 4-5*). The two water types with the largest surface area (M14 with 28.1% of the surface area and M27 with 30.2%) have undergone relatively little sampling (8.7% and 7.3%, respectively). The highest sampling intensity (M3 with 14.2% of the sampling and R5 with 21.0%) has been applied to water types with a relatively small surface area (5.5% and 2.4%, respectively) Sampling intensity can also differ greatly between periods.

Table 4-5 The distribution of the sampling effort over the WFD water types. Total area (ha) = surface area of the water type, in hectares. Area sampled (ha) = the amount of area sampled in all sampled years, in hectares. % area of total area = Percentage that the surface area of a water type represents of the sum of all areas. % sampling of total sampling = Percentage that the sampling in a water type represents of the sum of all sampling, given the period. See Appendix A for a description of the types.

	Total area (ha)	Area sampled (ha)	% area of total area	% area sampled	% sampling of total sampling				
		All years	All years	All years	All years	05-07	08-10	11-13	14-16
M10	979.1	44.6	2.6	4.6	11.1	8.1	5.9	14.5	18.1
M14	10651.7	34.8	28.1	0.3	8.7	7.5	8.7	7.3	14.1
M1a	132.3	17.4	0.4	13.2	4.3	2.2	7.4	3.6	1.7
M2	8.8	1.1	0.02	12.5	0.3	0.0	0.8	0.0	0.0
M20	2255.1	5.7	6.0	0.3	1.4	1.5	0.9	1.8	1.5
M23	48.9	0.3	0.1	0.6	0.1	0.0	0.0	0.2	0.0
M27	11444.9	29.2	30.2	0.3	7.3	11.7	3.6	5.3	16.4
M3	2089.3	57.1	5.5	2.7	14.2	8.0	21.0	16.1	0.0
M30	2286.4	1.0	6.0	0.04	0.3	0.0	0.0	0.7	0.0
M6a	357.8	12.7	0.9	3.5	3.2	3.6	2.4	4.3	1.3
M6b	1037.0	16.5	2.7	1.6	4.1	9.5	2.1	4.8	0.0
M7a	7.7	0.2	0.02	2.6	0.1	0.0	0.0	0.1	0.0
M7b	1866.4	10.4	4.9	0.6	2.6	2.1	3.3	1.5	4.9
M8	647.9	22.4	1.7	3.5	5.6	2.5	2.2	7.1	13.5
R12	47.2	1.5	0.1	3.2	0.4	1.4	0.0	0.3	0.0
R13	4.4	0.0	0.01	0.0	0.0	0.0	0.0	0.0	0.0
R14	11.5	0.4	0.03	3.5	0.1	0.0	0.1	0.1	0.0
R15	22.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
R17	7.3	0.0	0.02	0.0	0.0	0.0	0.0	0.0	0.0
R18	38.0	4.4	0.1	11.6	1.1	0.0	2.7	0.6	0.0
R4	73.0	13.2	0.2	18.1	3.3	0.8	3.6	5.2	0.0
R5	892.2	84.3	2.4	9.4	21.0	35.8	24.2	18.9	0.0
R6	1804.3	41.6	4.8	2.3	10.4	5.5	10.6	6.6	27.8
R7	1151.7	1.9	3.0	0.2	0.5	0.0	0.1	1.0	0.6
R8	12.2	0.7	0.03	5.7	0.2	0.0	0.6	0.0	0.0
Total	37837	401	100	1.1	100	100	100	100	100

Based on 1) the regional monitoring data; 2) the length-weight relationship and 3) the sampled surface areas, the biomass production of eel (>= 30 cm) was calculated for each water type and each scenario (*Table 4-1*). Some water types were not sampled in any year (R13, R15, and R17). For these water types the production averaged over all other water types was used. In addition, the sampling data were aggregated over each water board. As for some water boards information was unavailable, the production averaged over all other water boards was used for those without information.

In addition to the WFD sampling program, an additional sampling of ditches was done from 2013 onwards, because most ditches are not assigned as WFD water body (van Keeken 2014a, 2014b). Thus, most ditches are not part of the WFD sampling program. In order to estimate the standing stock in ditches the data from the WFD sampling (types 'M1a and M2) was combined with the data from the ditch sampling survey. To assess the silver eel biomass in the ditches, the average percentage of silver eel based on the regional data was used (23%).

4.2.3 Results; density and standing stock

The data were converted from number per length to weight using the length-weight conversion function (*Chapter 2*). The biomass of eel was calculated per water type and per water board for all years combined using the identical approach as used in previous assessments (Bierman et al. 2012, van de Wolfshaar et al. 2015). The eel biomass in ditches was done separately, based on the ditch sampling program (van Keeken et al. 2014a, 2014b). Summed over the water types, this resulted in a total of 2348 tonnes eel (>30 cm) estimated for the regional waters (*Table 4-6*).

Based on specialized ditch sampling campaigns (van Keeken 2013, 2014) an estimate of 4.7 kg/ha was calculated. By adding the biomass density estimates for the type M1a and M2 and weighing by sampled surface area an estimate of 5.4 kg/ha is calculated as caught in surveys in ditches. Following scenario 2, an estimate of 1426 tonnes is made for eel (yellow and silver) in ditches for which the surface area was not accounted for in the WFD (*Table 4-6*).

*Table 4-6 Eel biomass estimates per water type (2005-2016, scenario 2). Density (kg/ha) = density as caught in the surveys. Biomass (tonnes) = density in survey x total area (corrected for gear efficiency and for ratio inshore-offshore), divided by 1000. Stock assessment, estimate of total biomass of eel > 30 cm (yellow and silver eel) per water type following scenario 2 (see Table 4-1). * based on 23% silver eel.*

Water type	Total eel (>30cm)		Silver eel (>30cm)	
	Density (kg/ha)	Biomass (tonnes)	Density (kg/ha)	Biomass (tonnes)
M10	6.0	29.2	1.2	5.8
M14	20.5	1093.7	4.6	247.1
M1a	1.5	1.0	0.6	0.4
M2	5.1	0.2	1.5	0.1
M20	12.8	143.9	2.4	27.4
M23	0.0	0.0	0.0	0.0
M27	8.2	469.7	1.7	99.5
M3	3.7	38.7	1.1	11.2
M30	5.6	63.5	1.8	20.3
M6a	9.2	16.5	3.8	6.8
M6b	6.0	31.1	1.2	6.5
M7a	3.0	0.1	0.3	0.0
M7b	13.3	124.5	3.1	29.2
M8	0.8	2.5	0.5	1.6
R12	13.3	3.1	2.2	0.5
R14	1.5	0.1	0.2	0.0
R18	6.8	1.3	2.0	0.4
R4	2.1	0.8	0.6	0.2
R5	3.0	13.5	0.8	3.7
R6	10.4	94.1	2.6	23.6
R7	38.0	218.9	9.5	54.9
R8	3.8	0.2	2.0	0.1
M1b	7.9	0.0	2.0	0.0
R13	7.9	0.2	2.0	0.0
R15	7.9	0.9	2.0	0.2
R17	7.9	0.3	2.0	0.1
Total		2348		540
Non WFD ditches	5.4	1426		328*
TOTAL		3774		860

In addition to an estimate based on water type, an estimate was made for all eel (> 30 cm) per water board, as the water boards manage the regional waters (*Table 4-7*). The estimate based on water type will be used in further calculations.

Table 4-7 Biomass of eel > 30 cm based on all sampling data (no periods considered) assessed per water board (non-WFD ditches are not included), following scenario 2. Note that non-WFD ditches are not included.

Water board	Density (kg/ha)	Biomass (efficiency and inshore-offshore corrected) (tonnes)
Aa en Maas	1.4	2.2
Brabantse Delta	23.2	25.0
De Dommel	0.5	0.7
Groot Salland	34.2	218.4
Hollandse Delta	9.8	20.8
Hoogheemraadschap Amstel, Gooi en Vecht	5.8	123.0
Hoogheemraadschap De Stichtse Rijnlanden	5.3	3.7
Hoogheemraadschap Hollands Noorderkwartier	10.8	120.3
Hoogheemraadschap van Delfland	7.1	6.5
Hoogheemraadschap van Rijnland	16.3	185.3
Hoogheemraadschap van Schieland en de Krimpenerwaard	9.7	30.1
Hunze en Aa's	5.7	35.3
Kanalen	5.0	42.8
Noorderzijlvest	8.7	68.2
Peel en Maasvallei	11.4	7.1
Reest en Wieden	4.0	113.7
Rijn en IJssel	1.3	2.2
Rivierenland	4.2	10.5
Roer en Overmaas	5.3	2.2
Vechtstromen	7.5	56.6
Waterschap Vallei & Veluwe	2.9	3.9
Wetterskip Fryslân	20.1	838.3
Zuiderzeeland	20.4	430.6
Scheldestromen	9.6	0.3
TOTAL		2348

4.2.4 Standing stock in regionally managed waters

Different scenarios are used to estimate the eel biomass (*Table 4-1*), based on different values of catch efficiency and different ratios between eel densities in shores and open water. Eel biomass estimates vary between scenarios, with scenario 1 providing the lowest and scenario 3 the highest estimate for eel biomass (*Table 4-8*). In addition to the different scenarios, estimates were made for different periods and for all data combined. The variation in biomass estimate between periods is high. Because of the large variation in sampling effort per water type and/or period, the variation in biomass estimate is most likely driven by the variation in sampling, rather than a change in the eel population (see also *Table 4-3*, *Table 4-4*, *Table 4-5* and *Figure 4-2*). Because of this uncertainty the estimate of all years combined will be used in further analysis. Note that these values differ from the previous report because the biological keys were updated.

*Table 4-8 Estimates of standing stock of eel in the regionally managed waters; all eel larger than 30cm eel and silver eel (>30cm) biomass estimates (tonnes) for the three periods and the three scenario's. The most left column shows the estimates for eel in non-WFD ditches (all years). Other estimates concern the WFD water bodies. *based on 23% silver eel from the other water types (Table 4-6).*

	Non WFD-ditch	WFD water bodies				
	All years	All years	05-07	08-10	11-13	14-16
> 30 cm eel						
Scen 1	412	485	316	291	714	308
Scen 2	1426	2348	1537	1409	3500	1476
Scen 3	1489	3053	2001	1831	4578	1908
> 30 cm silver eel						
Scen 1	95*	112	66	56	163	63
Scen 2	328*	540	317	271	801	299
Scen 3	342*	701	412	351	1047	385

4.3 Nationally managed water bodies

4.3.1 Survey in the main rivers

Within the governmental survey program "Biologische Monitoring Zoete Rijkswateren", fish species in the main Dutch rivers are monitored yearly (*Figure 4.4*). Among others, rivers are sampled using research vessels (the "Actieve Monitoring van de Zoete Rijkswateren" survey, e.g., van der Sluis et al., 2014). Sampling in the open water takes place using a beam trawl and in the riverbanks using an electric dipping net. However, the beam trawl is not very suitable for sampling eel and only data collected with the electric dipping net are used here. Both the main rivers and water bodies connected to the main rivers are sampled (*Table 4-9*). Sampling takes place in autumn and early spring. There are six regions that have been sampled consistently and yearly since 1992. A region is usually sampled in the same months, but different regions are sampled in different months. There are also regions which have been sampled from a later year onwards and for which data is only available for some of the three years that are considered here. Volkerak-Zoommeer is an extreme example. It has not been sampled in 2014 and 2015 and the estimate is based solely on samples from 2016. Likewise, the Zandmaas region has only been sampled in 2015. See *Figure 4.4* for the classification of regions and *Table 4-9* for an overview of survey details per region. The large lakes Lake IJsselmeer and Lake Markermeer are sampled in another national survey program, using an electric beam trawl. However, because no information on the catch efficiency of this gear is available, density estimates from the fisheries model are used instead (see *Chapter 3*). The large lakes Grevelingen and Randmeren were only sampled with other types of gears (normal beam trawl, seine and fyke), the efficiency of which are even lower for eel, and less certain. Thus, the density estimate for these lakes are also derived from the demographic model (see *Chapter 3*).

Density per haul is determined (kg/ha), using eel length and the length-weight conversion factor (see *Chapter 2*). These densities are averaged per region and per type of water (main waterway and connected water body), over all samples of the three focus years. See *Table 4-9* for the density estimates per region for all caught eel. Note that catch efficiency has not been corrected for in *Table 4-9*.

No changes have been made in the methodology since the last assessment (van de Wolfshaar et al. 2015). However, data availability has changed since the last report because some of the data from 2013 were not yet available at the time of analysis of the report in 2015, which were available at the time of writing the current report. This causes a discrepancy between the estimate in the previous and the current report of the estimated biomasses in the period 2011-2013 (especially in the 'Getijden Maas' region). Also, new estimations of biological keys (*Chapter 2*) resulted in altered estimates for previous assessments.

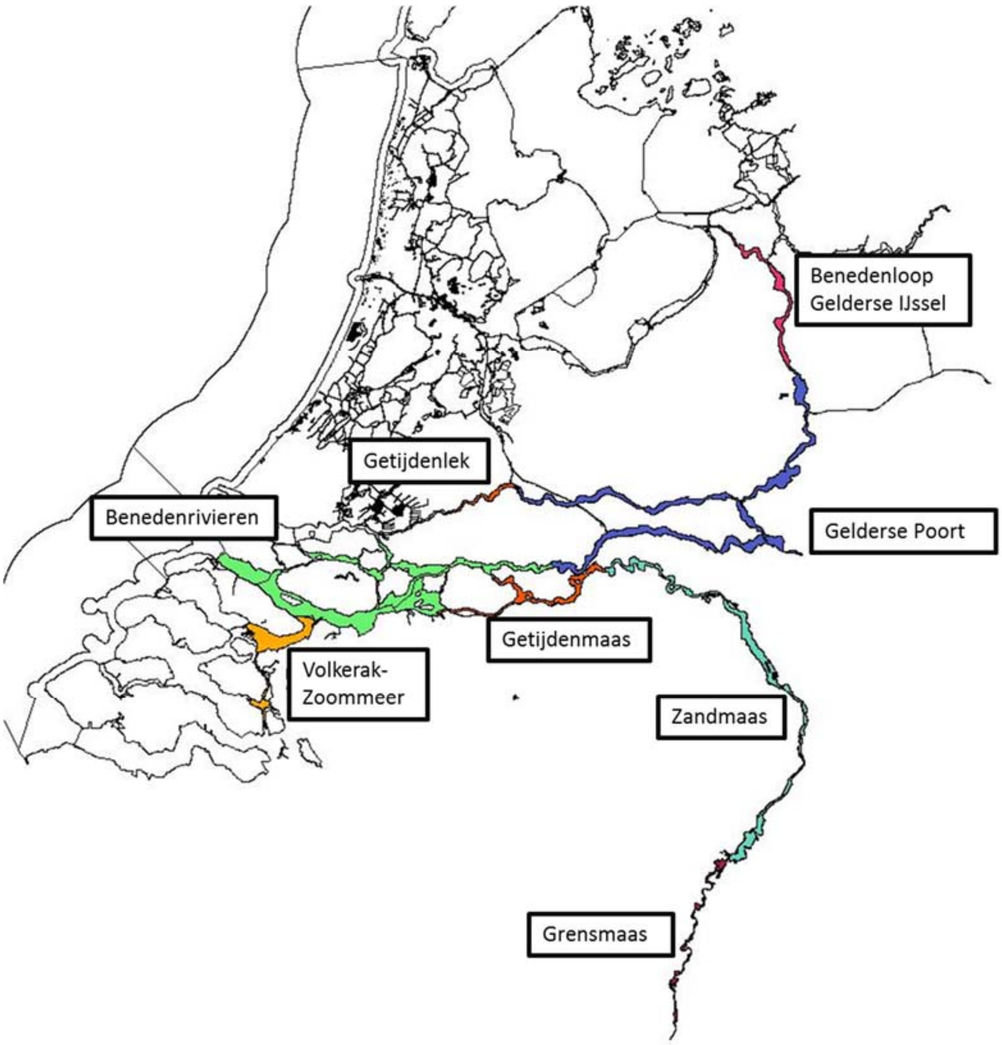

Figure 4.4 Classification of the main rivers. Regions are represented by different colours.

Table 4-9 Survey information per river region and type of water (main waterway or connected water body), for the years 2014, 2015 and 2016. Sampled years = the years in which a region has been sampled, where all = 2014+2015+2016. Survey density in riverbank = density for all eel larger than 30 cm (yellow and silver). Survey density is based on data collected using an electric dipping net at the riverbanks. No correction for catch efficiency of the gear is made yet.

Region	Water	Sampled years	Survey density (>30 cm) in riverbank (kg/ha)
Benedenloop Gelderse IJssel	main	all	3.40
	connected		0.00
Benedenrivieren	main	all	41.26
	connected		61.02
Gelderse Poort	main	all	1.25
	connected		0.81
Getijdenlek	main	all	9.69
	connected		3.38
Getijdenmaas	main	all	19.40
	connected		9.45
Grensmaas	main	all	8.29
	connected		69.23
Volkerak-Zoommeer	main	2016	8.45
Zandmaas	main	2015	12.39
	connected		6.17

4.3.2 GIS data

Three types of geographical information are collected, surface area, bank length and groin length. The surface area (ha) and bank length (km) of the rivers and lakes are calculated (*Table 4-10*) using GIS-data (the 'Ecotopenkaart' of Rijkswaterstaat). For the rivers, extra information on bank length was collected (*Table 4-10*). In some parts of the rivers, bank length is significantly larger than river length because of groynes (Dutch: "kribben") placed perpendicular to the riverbank. These groynes are approximately 90 meters long and placed 200 meters apart (www.rws.nl). In the parts of the rivers with groynes, bank length is thus approximately 1.9 times the river length. By visually scanning satellite images of Google Earth, a rough estimate of the percentage of riverbank with groynes is made: 60% of the Gelderse Poort is estimated to have groynes, and 50% of the Getijdenmaas. The other regions are assumed to have no groynes. The estimates used are the same as in the previous assessment (van de Wolfshaar et al. 2015)

Table 4-10 Surface area, river length and bank length per river region. Groynes = the percentage of a region that has groynes. Bank length is river length with groyne length (1.9 times the river length) included.

Region	Waterbody	Surface area (ha)	River length (km)	Groynes	Bank length (km)
Benedenloop Gelderse IJssel	main	675	118		118
	connected	271	42		42
Benedenrivieren	main	18377	703		703
	connected	1670	498		498
Gelderse Poort	main	5201	557	60%	858
	connected	1468	191		191
Getijdenlek	main	500	52		52
	connected	78	19		19
Getijdenmaas	main	1265	155	50%	224
	connected	753	82		82
Grensmaas	main	426	135		135
	connected	436	49		49
Volkerak-Zoom	main	4814	171		171
Zandmaas	main	2043	305		305
	connected	1413	160		160

4.3.3 *Synthesis*

Densities are corrected for the catch efficiency of the electric dipping net (20% for scenario 2). Water surface area is divided into two areas: littoral zone and open water. The width of the littoral zone is set equal to the reach of the dipping net (1.5 meters) and its surface area is the width times the bank length. The open water surface area is the total surface area minus the surface area of the littoral zone. Eel density outside the littoral zone is assumed to be a fraction of that in the littoral zone (50% for scenario 2). Subsequently, density is converted to absolute biomass (kg) for the riverbank and open water surface areas separately. Alterations are made for the Grensmaas. For Grensmaas no correction for habitat preference is made and density in the open water is assumed to be equal to that in the littoral zone, because sampling with the dipping net takes place in the open water in this (shallow water) region and is thus representative for the open water densities.

Biomass of silver eel and of all eel larger than 30 cm is estimated according to scenario 2 (*Table 4-11*). No information on the ratio yellow eel - silver eel in the surveys is available and the conversion keys (as calculated in *Chapter 2*) are used to determine the biomass of silver eel.

Table 4-11 Biomass of all eel, eel larger than 30 cm and silver eel (tonnes) per river region, estimated according to scenario 2, for 2014-2016.

Region	all eel (t)	eel >30cm (t)	silver eel > 30cm (t)
Benedenloop Gelderse IJssel	5.89	5.89	0.80
Benedenrivieren	2199.84	2172.76	641.87
Gelderse Poort	19.73	19.65	6.45
Getijdenlek	13.04	12.99	2.80
Getijdenmaas	81.74	81.08	30.32
Grensmaas	168.58	168.58	29.06
Volkerak-Zoom	105.70	102.21	17.29
Zandmaas	86.87	86.87	34.79

For scenario 2, estimated biomass of eel larger than 30 cm in the period 2014-2016 is also compared to the periods 2005 - 2007, 2008 -2010 and 2011-2013 (*Table 4-12*). No consistent trend is found in the estimated biomass through time. The biomass estimation for 'Benedenrivieren' is much higher in 2014-2016 than in the periods before. Because 'Benedenrivieren' is a large region, this will have a large effect on the change in biomass of all regions taken together from previous periods compared to 2014-2016.

*Table 4-12 Biomass of eel larger than 30 cm (yellow and silver) in tonnes per river region, for 2005-2007, 2008-2010, 2011-2013 and 2014-2016, following scenario 2. * For these periods no data was available and instead the estimate of 2008-2010 period is used.*

Region	2005-2007	2008-2010	2011-2013	2014-2016
Benedenloop Gelderse IJssel	12.10	4.14	3.12	5.89
Benedenrivieren	401.93	350.37	415.39	2172.76
Gelderse Poort	17.84	4.09	44.04	19.65
Getijdenlek	3.32	4.52	14.22	12.99
Getijdenmaas	35.22	13.75	28.79	81.08
Grensmaas	36.28	117.11	58.50	168.58
Volkerak-Zoom	502.78*	502.78	383.23	102.21
Zandmaas	109.05*	109.05	92.08	86.87

4.4 Discussion

Concerning both the nationally and regionally managed waters, there are some uncertainties. Different regions are surveyed in different months. This implies different mean water temperatures, different eel behaviour, and different silver eel migration activity, all of which can influence the catches in the survey.

A central assumption underlying the stock estimation is that the eels caught in a certain area represent the inhabitants of that area, using it to realise their growth until seaward migration. For the main passage way of silver eel to the sea (i.e., almost all nationally managed rivers and lakes), this assumption entails much uncertainty. On the one hand, eels surveyed during the migration season (i.e., in many of the rivers; see *Table 4-9*) may partly consist of migrating silver eels. These eels were perhaps surveyed in their original habitats too (since areas are surveyed in different time periods), or these eels may have migrated from other countries after maturation to silver eel. This would lead to an overestimation of the silver eel stock of the Netherlands. This could explain the high stock estimates of 'Benedenrivieren', which is the area closest to the coast where silver eel might concentrate just before and during the migration season. This does not explain the very high estimate in specifically the years 2014-2016 of the 'Benedenrivieren', for which the reason remains speculative. Likely the increase in these years is an effect of the prohibition to fish on eel in the main rivers since 2011. Future monitoring will show if the high biomass estimates will persist.

Surveying during or directly after the migration period may also lead to an underestimation of the silver eel stock, because part of the silver eels might have migrated away or might be in water body parts that are not surveyed with the dipping net (e.g. the open water). Thus, because the main surveys in the nationally managed waters (in the IJssel-/Markermeer and the main rivers) take place predominantly during the migration period, there is additionally uncertainty. The same reasoning goes for the regionally managed waters surveyed during or following the migration period. However, with constant survey periods, this does necessarily affect trends in biomass estimates.

4.4.1 Regionally managed waters

There are some shortcomings and uncertainties in the data availability concerning the regionally managed waters. Like in previous years, the first issue is that not all collected data was available for the analysis presented here. A second issue is, given the six year cycle of the WFD monitoring and the three year cycle for the eel stock assessment, that not all water types are equally represented for the periods used in the eel stock assessment. This means that some types are over- and some types are under-represented if using the data at the three year interval as intended for the eel assessment. Thus, biomass estimates were not calculated per interval and only one estimate for all years was presented. In addition, not every sampling occasion could be linked to a water body and these were excluded from the analysis. This mismatch might be due to measurement error in GPS equipment or errors in data entry. As in 2012 and 2015, scenarios were used for catch efficiency and habitat use, issues that remain uncertain. The new set of biological keys, based on more individuals, had an effect on the estimates of yellow and silver eel of the regionally managed waters.

Concerning the WFD sampling program in general, the sampling intensity is not well balanced: water types with the highest surface area have relatively low sampling intensities while the highest sampling intensities are performed on water types with relatively very low surface area. A more balanced sampling program is therefore recommended.

4.4.2 Nationally managed waters

There are some shortcomings and uncertainties in the methodology used for the nationally managed waters. Various regions are not sampled every year, which makes the estimate per period less reliable. For two regions (Volkerak-Zoommeer and Zandmaas), information was not available for every period and densities were assumed to have stayed equal compared to other (sampled) periods. The most extreme example is Volkerak-Zoommeer, which was (a) only sampled in two of the nine years (of 2005-2013), (b) only sampled in one period (2008-2010) and (c) sampled in different months in the two years (September in 2010 and May in 2008). However, the uncertainty following from this unbalanced monitoring scheme is much less than that following the unbalanced monitoring scheme of the regional waters.

Detailed information on the amount and distribution of groynes in the rivers is lacking. Here, we used a very coarse method to estimate the amount of groynes per region.

Furthermore, sufficient knowledge of two important factors is lacking: the catch efficiency of the survey gear and habitat preference of eel. These factors cause a large variation in the biomass estimate.

5 Stock estimate for the Dutch standing stock

In this chapter the total Dutch stock is estimated, based upon the information for all water bodies as described in the previous chapters. The results are used for *Chapter 6* and *Chapter 7*.

5.1 Three scenarios and four time periods

5.1.1 Three scenarios

To estimate the standing stock in the large lakes (i.e. Lake IJsselmeer, Lake Markermeer, Randmeren and Grevelingen, see *Chapter 3*), fishing mortality in Lake IJsselmeer was used (*Table 3.2)*. To estimate the standing stock in the other waters (regionally managed waters and the nationally managed rivers), three scenarios were run with different catch efficiency and habitat preference (*Table 4-1*). These scenarios are integrated into three overall scenarios with varying catch efficiency and habitat preference (*Table 5-1*). Of these three scenarios, the standing stock estimate of scenario 2 is taken as the best guess estimate. Scenario 1 is the minimum estimate and scenario 3 as the maximum estimate.

Table 5-1 The three scenarios used in the stock assessment, for both the large lakes (via landings and fishing mortality, Chapter 3) and for all other water bodies (via survey data scaled to wetted areas, Chapter 4). Scenario 2 is the best guess estimate.

Scenario	Large lakes		Other waters	
	Fishing mortality 2005-2007	Fishing mortality 2008-2016	Catch efficiency	Density offshore compared to inshore
1	0.29	0.34	66%	33%
2	0.29	0.34	20%	50%
3	0.29	0.34	20%	66%

In addition to stock estimates in 2014-2016, estimates for previous time periods are provided, based on the latest parameterization of the biological keys. As mentioned in *Chapter 4* (section 4.2 and 4.3), extrapolation between periods is necessary because not enough data was available for all regions in all periods (*Table 5-2*). Extrapolation is needed for the Volkerak-Zoommeer, Zandmaas, the Randmeren, ditches and the other regionally managed waters (*Table 5.2*). The method of extrapolation for these five types of water bodies is discussed in the next section.

*Table 5-2 Eel biomass estimate availability ('data') per period and per water body.'-' indicates that there was not sufficient data available. * The regionally managed waters data are not used per time interval, but are grouped in the analysis into one year.*

		2005-2007	2008-2010	2011-2013	2014-2016
Large lakes	Randmeren	-	-	data	data
	Other	data	data	data	data
Regionally managed waters	Ditches	-	-	data	data
	Other*	data*	data*	data*	data*
Nationally managed waters	Volkerak-	-	data	-	data
	Zandmaas	-	data	data	data
	Other	data	data	data	data

5.1.2 Extrapolation between periods

In case of missing information, data was extrapolated from one period (with data available) to another (for which no data were available), based on known estimates and then used for other water bodies.

For the Randmeren, landings data was available for the periods 2011-2013 and 2014-2016. Because these lakes have an open connection to Lake IJsselmeer and Lake Markermeer, the extrapolation to the other periods was based on the ratio of biomass estimates between periods of these two neighbouring lakes (*Table 5-3*).

For the ditches survey data was available for 2011-2016. The average estimate over 2011-2016 was used for all periods. For the other regionally managed water bodies survey data were available for the years 2006 to 2016, yet the spatial-temporal resolution was such that no estimate for separate periods could be done. The density estimate for all years combined was used as estimate of the standing stock for all three periods.

In the nationally managed waters no survey data were available for Zandmaas in the period 2005-2007 and hence the estimate of 2008-2010 was used for this prior period. Volkerak-Zoommeer was only sampled in the period 2008-2010 and 2014-2016. The estimates for 2008-2010 were therefore used for 2005-2007 and the average of 2008-2010 and 2014-2016 was used for 2011-2013. For the other nationally managed water bodies estimates for the periods are available and no extrapolation is needed.

The biomass estimates for all eel larger than 30 cm (scenario 2), including extrapolation, is given in *Table 5-3*. For the regions with data for all periods (Lake Ijsselmeer, Lake Markermeer, Grevelingen and other nationally managed water bodies) a change in stock biomass through time is examined. While the large lakes show a decrease in eel biomass, the nationally managed waters show a biomass increase for the most recent period. Overall, there is no trend in eel biomass over the four periods for all eel > 30cm. The period 2011- 2013 has the lowest biomass and the period 2014-2016 has the highest biomass (*Table 5-3*).

Table 5-3 Extrapolation of estimates for biomass of all eel >30cm (yellow and silver eel combined, in tonnes) in scenario 2.

		2005-2007	2008-2010	2011-2013	2014-2016
Large lakes	IJssel-/Markermeer	936	386	394	380
	Grevelingen	185	54	24	3
	Randmeren	65	27	27	26
Regionally managed waters	Ditches	1426	1426	1426	1426
	Other	2348	2348	2348	2348
Nationally managed water bodies	All combined	1119	1106	1039	2650
	Total	6079	5347	5258	6833

The different values of fishing mortality in the large lakes compared to the previous assessment, have a profound effect on the biomass estimates. Because *F* is smaller for 2005-2008 compared to the previous assessment, the biomass estimate is higher. Likewise, the higher *F* for the other periods leads to a lower biomass estimate compared to the previous assessment. The biomass estimates for the ditches and the regionally managed waters as well as the nationally managed waters are in the same order of magnitude compared to the previous assessment (van de Wolfshaar et al. 2015). The estimate for 2014-2016 for the nationally managed waters is more than double than that of the other periods, which is due to the estimate in one river section (see also section 4.4).

5.2 National stock biomass overview

For each scenario, for eel >30 cm and for silver eel, the available data were extrapolated in order to obtain values for all periods. By subtracting silver eel biomass from the biomass estimate for all eel (> 30 cm), the yellow eel biomass (> 30 cm) is calculated. The yellow eel biomass and the silver eel biomass (B_{start}) are used for the estimates of key stock indicators (*Chapter 7*). An overview of the total national stock biomass estimates for the different scenarios and time periods is given in *Table 5-4*.

Table 5-4 Total stock biomass (tonnes) estimates for yellow eel and silver eel (> 30cm, B_{start}) for each period and each scenario. Scenarios 1-3 differ in catch efficiency and habitat preference (Chapter 3).

	2005-2007	2008-2010	2011-2013	2014-2016
Yellow eel > 30cm				
Scenario 1	2277	1887	1865	2063
Scenario 2	4494	4083	3999	5038
Scenario 3	5381	4948	4852	6194
Silver eel > 30cm				
Scenario 1	983	651	639	744
Scenario 2	1585	1255	1258	1795
Scenario 3	1811	1474	1486	2190

6 Mortality during silver eel migration due to barriers

The data of the report of 2015 is updated with regard to the major changes and measures that have been made to improve silver eel migration to the sea. For this update, an inventory was held among water boards to renew the information on barrier specifics conserving migration (Kroes et al. 2018). In addition, an update of the list of pump types that were replaced by others was made for those barriers that are in the Water Framework Directive (WFD) waters.

In this chapter we describe the methodology and data on which estimates of barrier mortality are based. The mortality during silver eel migration is one of the sources of mortalities which are used in the overall assessment as presented in *Chapter 7* (see the flow diagram of the stock assessment *Figure 1-2*).

6.1 Model for estimating barrier mortality

Estimating the mortality of silver eels during their migration from inland water bodies to the sea is, due to barriers such as pumping stations and hydropower plants, a challenging task. The Netherlands consists of a complex network of interlinked large and small water bodies, most of which contain eel, either by natural immigration or stocking. There are thousands of pumping stations and even more other potential barriers that could cause silver eel mortality during their migration to the sea (Kroes et al. 2018). To construct a model on silver eel mortality caused by these barriers, knowledge on the following key processes is necessary:

1) Routes of silver eels migrating from inland water bodies to the sea, and the barriers that these eels have to pass on these routes
2) The biomass of silver eels at the start of their seaward migration in the different types of water bodies.
3) Mortalities during passage of barriers

A conceptual model for silver eel migration was built, based on a hierarchy of water bodies, which may provide a reasonable description of silver eel migration in The Netherlands (*Figure 6.1*). In this conceptual model, silver eels are split into three groups of starting origin, according to water body type. These three main water body types correspond to the three main hierarchy levels of water bodies in The Netherlands:

1) 1st hierarchy (termed **'polder' water bodies**): water bodies which are below sea level and serviced by a large number of small pumping stations with often high levels of mortality during passage. In the model, each polder is serviced by a single pumping station (i.e. no multiple pumping stations in sequence). Pumping stations of coastal polders can pump water directly into the sea, in which case the silver eels that survive the passage of these sites are directly contributing to the silver eel 'escapement' out of the Netherlands. However, for most polders, pumping stations would discharge water into a water body of the 2nd hierarchy in our model ('boezem' water bodies). In the model, polder waters are represented by the wetted area of drainage ditches (see *Chapter 4)*;
2) 2nd hierarchy (termed **'boezem' water bodies**): water bodies such as canals, small inland lakes (such as the Frysian lakes), but also smaller streams and rivers which are either connected directly to the sea or to large nationally managed water bodies (the 3rd hierarchy of water body in the model; see below) via larger pumping stations and/or sluices. In the model, boezem waters are represented by all regionally managed Water Framework Directive (WFD) water bodies (see *Chapter 4)*;
3) 3rd hierarchy (termed **'national' water bodies**): large nationally managed water bodies such as sections of the main rivers Rhine and Meuse (including downstream parts, *Chapter 4*), the lakes IJsselmeer and Markermeer, Veluwerandmeren and Grevelingenmeer (*Chapter 3*). Silver

eels have been found to experience low levels of mortalities during passage of most of the barriers (because these are mainly discharge sluice complexes) in these large water bodies. The exception are the passages of hydropower plants by eels that start their migration from upstream sections of the main rivers Rhine and Meuse. Both these sections hold a substantial biomass of silver eel (van de Wolfshaar et al. 2015).

The framework of the model for migration routes is illustrated in *Figure 6.1*. The hierarchies of inland water bodies (and sections of rivers upstream of hydropower plants) are connected with each other and with the sea as presented by the arrows. The following will be estimated within this model approach:

1) Transition rates between types of water bodies.
2) Mortality rates during passage from one type of water bodies to the next.

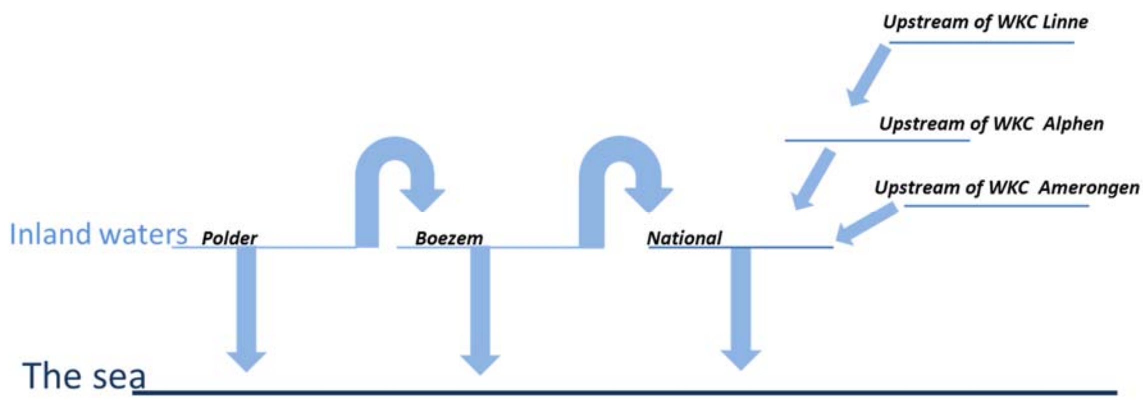

Figure 6.1 A conceptual model for estimating mortality during silver eel migration due to barriers; for 'polder' (1st hierarchy), 'boezem' (2nd hierarchy) and national waters (3rd hierarchy of waters see text). WKCs are hydropower stations in river sections of the national waters.

Given the starting biomasses, transition estimates and mortality rates of silver eels in the different hierarchies of water bodies, the model will yield a prediction on the overall silver eel mortality during migration from inland waters to the sea.

A key assumption in this model is that barriers within a hierarchical class, for example within polder waters, are never in sequence. Instead, sequential barrier mortality only occurs when silver eels are transferred from one hierarchical class to another, for example from polder to boezem. This approach is thought to hold true in the majority of cases. However, there are some polder waters with two boezem layers, in which polder waters are pumped into an 'inner boezem' and subsequently pumped into an 'outer boezem' (which would be the second hierarchy in the model presented here).

Given the assessed mortality and transition rates, the percentage of silver eels (out of the total starting biomass) that is estimated to die during migration is dependent only on the proportional distribution of silver eel biomass over the different hierarchies of water bodies. Instead, the biomass of silver eels that is estimated to die during migration will be dependent on the absolute biomass of all starting silver eel. In the overall assessment presented in Chapter 7, only an estimate of the percentage mortality is necessary as a parameter for the estimation of lifetime anthropogenic mortality (*LAM*). In this chapter, we only illustrate the model with the best guess biomass estimates (scenario 2). The differences in the proportional distribution of silver eels over the hierarchies of water bodies are small between the scenarios. The estimated total production of silver eel in the three hierarchical types of water bodies are based on the static spatial population model (*Chapter 4*).

6.1.1 General approach to assess mortality rates at barriers

For the parameterisation of the barrier mortality model we use "net mortality rates" for barriers: the proportion of silver eels that ends up in front of that barrier multiplied by the proportion that dies during passage. If there is only one route available in passing a barrier, the mortality rate of this barrier can be multiplied by the number of silver eels that end up in front of the barrier. In our approach we consider blockage (i.e. silver eels that end up at barriers but are not passing), the same as mortality, since in both cases these silver eels do not contribute to the 'escapement' of silver eel to sea. In case an alternative route for migration trough a pumping station or hydropower plant is available, such as a ship lock, sluice of fish pass, estimates of net mortality rates are typically lower than the proportion of silver eels that suffer mortality attempting to pass the pumping station or hydropower plant (*Figure 6.2*).

Silver eel is distributed over different potential routes, e.g. part of the eels pass via a ship lock and part via a pumping station (*Figure 6.2b*). In *Figure 6.2d* an example is given for a river site in the Meuse (Linne) where silver eels can pass via the hydropower station (the most dangerous route where part of the eels suffer mortality) or via the weir or fishway. The migration behaviour and distribution of silver eel over the different routes is dependent on, among others, the discharge distribution, and can therefore vary in time (Jansen et al. 2007). For some sites, the distribution of silver eels in some years is available from, for instance, telemetry studies such as the weir-hydropower complexes at Linne and Lith in the Meuse (Winter et al. 2006, 2007, Jansen et al. 2007, de Boer, in prep), or the sluice pumping station complex at IJmuiden (Winter 2011).

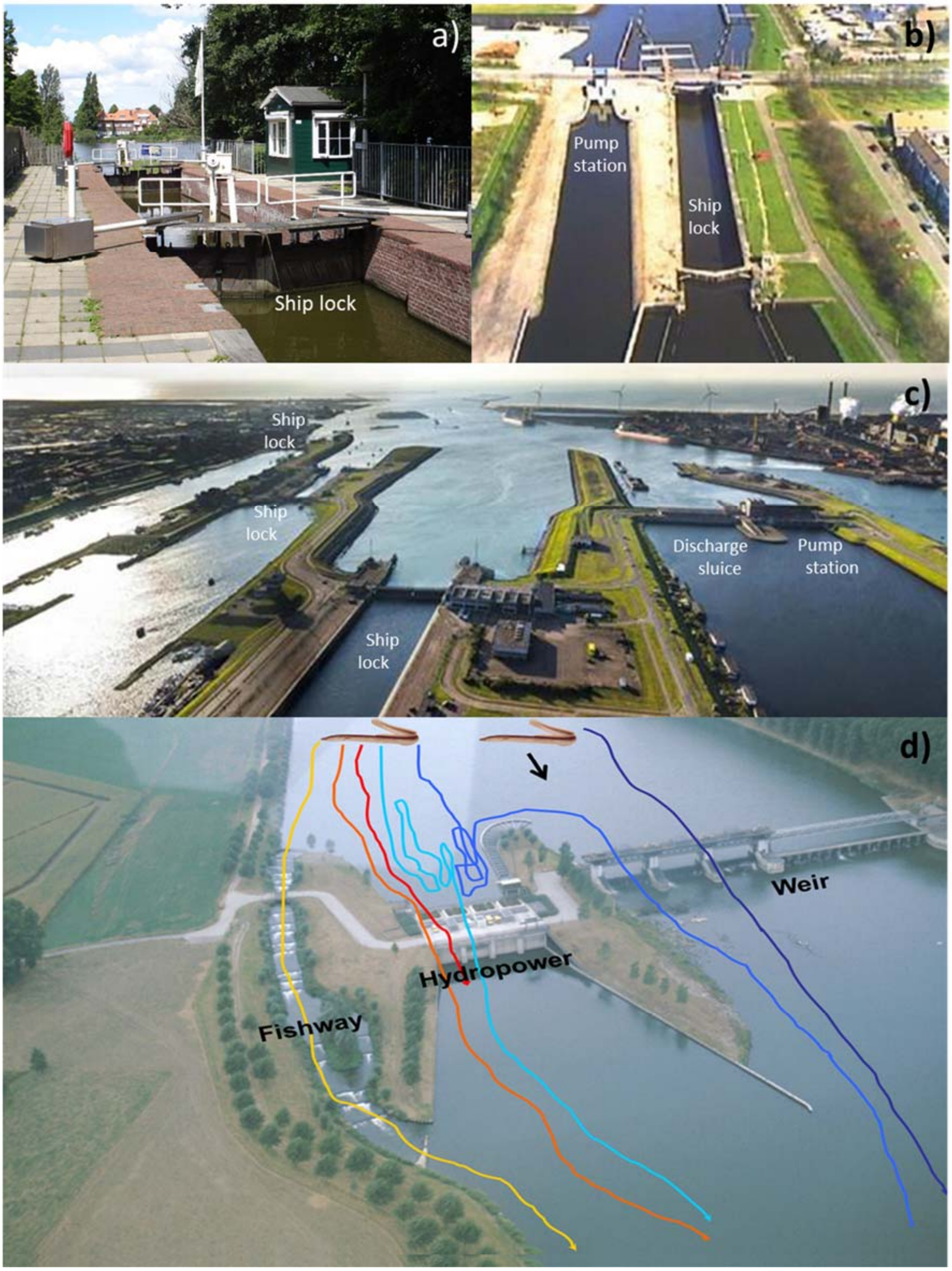

Figure 6.2 An illustration of different types of barriers. Barriers range from simple, e.g. single ship lock (a), to combinations, e.g. pump station and ship lock (b), to very complex sites consisting of a combination of pumping stations, ship locks, sluices or other alternative routes for migration (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 hydropower plant, and therefore the distribution of eel passing via the different routes per site determines the overall mortality rate for the entire site (d).

6.1.2 Mortality rates and transition from polder to boezem or the sea.

Silver eel migrating from the polders to the boezem waters will encounter pumping stations. There are direct and indirect effects of pumping stations on silver eel migration. In the first place, pumping stations can cause damage and direct or delayed mortality in eel when passing through a pump. Secondly, a pumping station may function as a barrier for eel, both during upstream and downstream migration. However, a recent study demonstrated that for migrating silver eel, pumping stations delayed migration but did not function as a permanent barrier as long as the pumping stations are running at some point in time (van Keeken et al., 2013). Thirdly, pumping stations will increase the predation risk of fish. Damaged and confused fish will have a higher chance of being predated by piscivorous fish or birds. But also the risk of being captured by fishermen is higher around pumping stations when migrating silver eel aggregate while searching for an opportunity to pass (e.g. Winter, 2011). Here, however, we will focus on the impact of pumping stations on the survival of migrating silver eel when they actually pass through a pumping station.

Pumping stations can roughly be divided into three groups: 1) water wheels, 2) Archimedes screws, and 3) pumps [centrifugal pumps (radial water flow); propeller-centrifugal pumps (radial/axial water flow), propeller pumps (axial water flow), Figure 6.3].

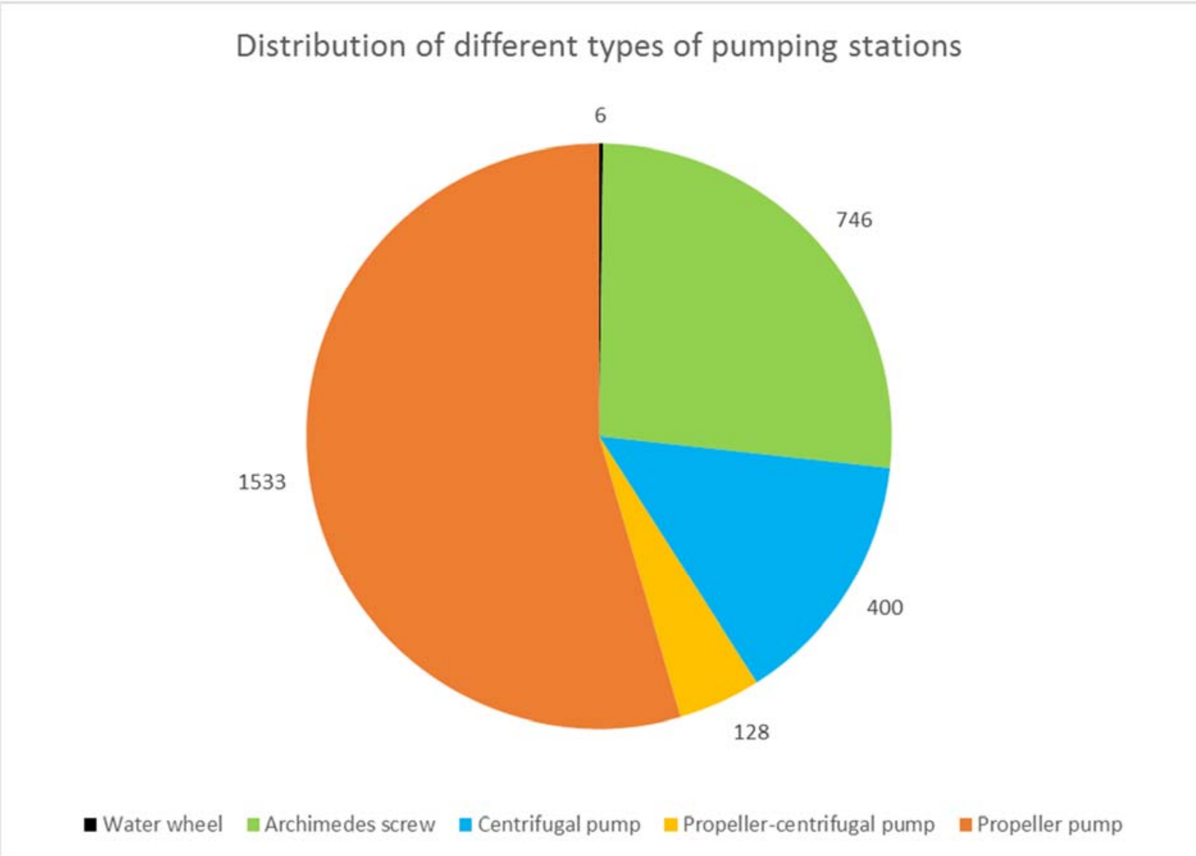

Figure 6.3 Distribution of different types of pumping stations in the Netherlands (redrawn from Kunst et al., 2008). Water boards are increasingly replacing 'fish unfriendly' pumping stations for 'fish friendly' pumping stations, but for this evaluation a complete update was not yet incorporated.

Appendix B provides an overview of studies, conducted mainly in the Netherlands and Belgium, on the impact of different pumping station types on the survival of eel. These studies clearly demonstrate that, in general, propeller pumps with axial or axial/radial water flow caused the highest mortality rates when eel pass through these types of pumps. Unfortunately, at least in the Netherlands, these type of propeller pumps are the most common type used to regulate water levels. On a "fish friendliness" scale, propeller pumps are in general regarded as "unfriendly", while water wheels and Archimedes screws are relatively "friendly". Water boards are increasingly replacing 'fish unfriendly' pumping stations for 'fish friendly' pumping stations, but for this evaluation this was not incorporated yet. Also, increasing numbers of pumping stations are facilitated with fish passage facilities by the different water boards, but efficiencies of these in diverting silver eels from pumping stations are not known in most cases. The fish passages are almost always designed to facilitate fish migration into polders, where it is the only available route to pass the pumping station. For fish migrating out of the polders, passage through the pumping station and fish pass thus provide two potential routes where the discharge through the pumping station is always much larger than the small discharges that go through the fish passes. This generally results in mostly low efficiencies of diverting fish from the flow through pumping stations. Additional measures or systems to divert eels to fish passes are scarce or often less effective (Kroes et al. 2013). Finally, since Kunst et al. (2008), more studies on mortality rates at pumping stations were carried out that could be used to update the mortality assessment per pumping station type.

In one study, seemingly undamaged eels were dissected and it was concluded that many of these eels had internal injuries which would result in delayed mortality (Kruitwagen & Klinge, 2008a). In *Table 6.1* we defined mortality as the '% dead' plus a fraction of 0.5% as '% damaged eel'. The average silver eel mortality during passage of pumping stations was estimated as the average of the mortalities for each type of pumping station weighted by its occurrence (*Table 6.1*).

Table 6.1 Calculation of the average pumping station mortality used to estimate silver eel mortality during migration (see also Appendix B).

Pump type	Proportion (Figure 6.3)	Average mortality* (%) (Appendix B)	Weighted Mortality (%)
Water wheel	0.002	0	0
Archimedes screw	0.27	12	3.2
Centrifugal pump	0.14	12	1.8
Propeller-centrifugal pump	0.05	9	0.4
Propeller pump	0.55	56	29.3
Pump Mortality (estimate used in Yellow Eel Model)			~35%

* Mortality is % dead + 0.5 % damaged.

Extrapolation to Silver eel migration

Ideally, a bottom up approach would be used to start with a polder specific silver eel biomass that is starting to migrate and assess mortality rates based on the mortality rate of the specific pumping stations per polder. However, data on starting biomasses in all thousands of polders and site specific mortality rates per pumping station in each of these polders are not available. Instead, for the 1^{st} hierarchy 'polder waters', average densities per polder area and an overall estimate of mortality rate based on the national distribution of types of pumping stations and estimated mortality rates were used to provide an overall estimate of escapement from polders to sea, and to the 2^{nd} hierarchy of boezem waters. From polder waters to boezem waters or to the sea: a best guess estimate of 35% mortality was used, based on a meta-analysis of estimates from a large number of studies, as presented in the paragraph above and *Appendix B*. Regionally, the starting biomass and mortality rates will be different than the average, but for the purpose of estimating a national mortality rate, this generic approach for the 1^{st} hierarchy will largely level out.

To complete the model, transition rates between the three hierarchies of water bodies (and the sections of river upstream of the hydropower plants) are needed. The majority of polders (except some coastal polders) are thought to have pumping stations that discharge water into the boezem rather than to the sea. We estimated (best guess) that 20% of the eel in polder waters is transferred directly from polder to sea, whereas the remainder (80%) is transferred to boezem waters where additional mortality due to barrier passage might occur (*Figure 6.4*).

6.1.3 Mortality rates and transition from boezem to national waters or the sea

The mortality estimates for silver eel migrating from boezem to national waters are based on an inventory of the main migration barriers for silver eel migrating from the Netherlands (Winter et al. , 2013a & 2013b), which was updated for changes and input from water boards during 2013-2017 for this evaluation study (*Appendix C*). In other words, an up to date overview was made of possible barriers for silver eel migration such as, ship locks, discharge sluices, hydropower stations, weirs and pumping stations., More often than not, a combination of these man-made structures are present at sites. The list of the main potential barriers is mostly based on the size of the catchment area that discharges via the potential barrier. To prioritize these potential barriers, the potential biomass of silver eel (as estimated on densities and area of the waters in the catchment area that discharges via the barriers) was used in combination with an assessment of the mortality rates at the potential barrier. In the current evaluation study, we used the estimated overall mortality rates per site (often a combination of different types of man-made structures) up till 2017 which are updated from Winter et al. (2013a, 2013b). For each of these barriers, it is known if passage leads directly to the sea or to national waters, hence, the distribution can be calculated. This information allows for mortality and distribution estimates weighted by the silver eel biomass at the barrier as a fraction of total silver eel biomass. Therefore, for the 2nd hierarchy a bottom up approach to determine national mortality rates was feasible.

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 is 6% for passage to the sea and 14% for passage to national waters.

6.1.4 Mortality rates from national water bodies to sea, and hydropower stations

The approach for barrier mortality estimation for national waters is also based on the inventory of Winter et al. (2013a, 2013b) and updated for the period 2013-2017 as described above for the barriers in boezem waters.

For the locations of the two largest hydropower stations, both situated in the River Meuse, several studies for route selection and mortality rates are available (Winter et al. 2006, Jansen et al. 2007, de Boer, in prep). Since mid-November 2011, an adapted turbine management regime was implemented to reduce eel mortality from 24% (Bruijs et al. 2003) to 18% (based on a model study that estimated a reduction of 24-25% of the mortality rate at the hydropower stations, Buijse, 2011). This was implemented on 17 November 2011 at HPS Alphen/Lith and on 21 November 2011 HPS Linne in the Meuse and on 17 November 2011 at hydropower station (WKC) Amerongen in the Lower Rhine. In 2017, the energy companies that exploit the hydropower plants were ordered to take additional measures to reduce hydropower mortality to 5%. It was ordered by court that, as a temporary measure starting in 2017, the hydropower stations were allowed to operate during night-time in August-September. In an ongoing study that runs from 2017-2019, it will be assessed whether using a Migromat warning system at WKC Alphen (see Bruijs et al. 2003) and replacing the turbines at WKC Linne with fish-friendly turbines by Fish Flow Innovations/Pentair-Nijhuis (see Winter et al. 2012) during 2019-2020 can lead to a reduced mortality of < 5%. The effect of these measures on silver eel mortality at the hydropower stations is currently not yet known. It is however, very likely that mortality rates as a result of this are lower than the rates used in the previous evaluation study (van de Wolfshaar et al. 2015). Although there is no current research or data available to determine the current mortality rates at hydropower stations, we incorporated the modelled reduction of mortality rates by 25% as estimated by Buijse

(2011), and used a best guess of 13% net mortality for WKC Linne, 14% HPS Alphen and 10% for WKC Amerongen.

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 from national waters to the sea (excluding hydropower stations) is 0.5%. Most of these waters are connected to the sea by discharge sluice systems which cause no mortality. It is assumed that silver eel that passes the hydropower stations enters the national waters (minus the mortality loss) and suffer from the migration mortality from the national waters to the sea. However, silver eel passing WKC Linne will also pass WKC Alphen as both hydropower stations are in the River Meuse.

6.2 Migration mortality

Based on the distribution and mortality estimates reported above, the model scheme can be filled with a best guess mortality scenario. This scenario is illustrated in *Figure 6.4*.

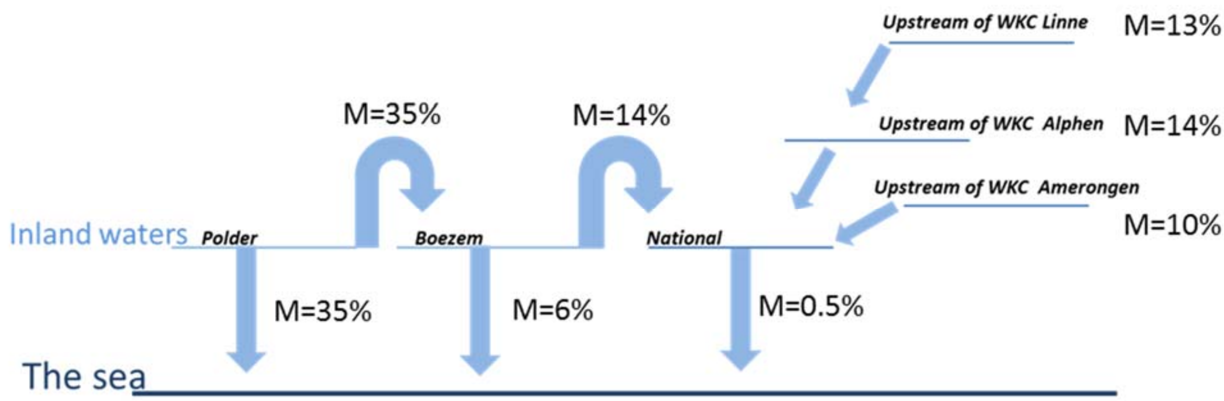

Figure 6.4 Migration mortality scheme, used to estimate overall migration mortality of silver eel. 'WKC' = hydropower station (*Dutch: 'waterkrachtcentrale'*).

Recent 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 (pilot) projects have started at migration barriers (pumping stations) to assist the migration of silver eel. However, the mortality rates of silver eel passing the selected barriers has been assessed to be at a moderate to low level (Bierman *et al.* 2012; Winter *et al.* 2013a). Thus, the net amount of eels saved by the assisted migration is much lower than the amount caught and released. In 2013, the barriers for silver eel were prioritised (Winter *et al.* 2013a) to improve the selection and efficiency of assisted migration initiatives. Applying location-specific mortality rates, the net amount of 'saved' eels was based on the mortality rate of the given site.

The absolute escapement estimates in hydropower plants are likely to be underestimates, because these include only silver eels produced in Dutch sections of the main rivers. However, silver eel migrating downstream on the river Rhine from Germany and silver eel migrating downstream on the river Meuse from Belgium and Germany will attribute to the numbers of silver eel starting in the Netherlands. The proportion of silver eel migrating down the Rhine river from Germany passing the river section of the Amerongen hydropower plant is estimated to be 6% (Klein-Breteler et. al. 2007). The silver eel mortalities on these 'foreign' eels migrating from Germany are not taken into account in the evaluation of the Dutch Eel Management Plan (EMP).

6.3 Discussion

Given the large number of polders and the lack of site-specific data for most of these sites, assessing the polder waters using a 'bottom up' approach is not feasible at this stage and therefore mortality rates are based on overall silver eel production estimates combined with an overall calculated average mortality rate. For the 'boezem' and national waters, in contrast to the 2012 evaluation, the current (and previous, 2015) evaluation, a more bottom up approach could be used. This uses site specific estimates for mortality rates at specific sites and silver eel production estimates from the catchment area that discharges via the barrier site based on an inventory study from 2013 (Winter et al. 2013a, 2013b). This approach yielded more accurate estimates than the more general approach based on averages as used in the 2012 evaluation. The quality of the underlying data that was used in the updated silver eel barrier assessments is however highly variable and often still incomplete. Some sites are very well studied, e.g. the sites with hydropower stations in the River Meuse (Winter et al. 2006, 2007, Jansen et al. 2007, Griffioen et al. 2013, de Boer, in prep.), the discharge sluices complexes in Haringvliet (Winter & Bierman 2010) and at the sluices-pumping station complex at IJmuiden (Winter 2010 and ongoing study in 2017-2019), but for other sites, e.g. ship locks and most of the pumping station sites, data on relative route passage and mortalities per route at a specific site are still largely lacking. The barrier-mortality model as presented here can be further developed to enable a full 'bottom up and site-specific data driven' approach. Several maps and lists of barriers are available (e.g. Kroes et. al. 2018; Buijse et. al. 2009). However, this is to our knowledge the first formal model to estimate mortalities during passage of barriers which takes variation in starting positions and migration routes of silver eels into account. Additionally, we have used net mortality rates for individual barriers which account for possible alternative routes which silver eels may use (*Paragraph 6.2*).

The model presented here can be used as a blue print for further development and refinement, for instance to be carried out in regional case studies. In particular, the characterisation of water bodies as polder or boezem waters can then be further evaluated. Also, the assumption that pumping stations are not positioned in sequence within polder or boezem waters needs to be further evaluated. A more realistic spatially explicit 'bottom up' model as indicated above could be based upon the methodology described in Bierman et al. (2012) to estimate migration routes. Such a route analysis will provide the best basis for models to compute barrier mortality rates.

There is an additional potential anthropogenic mortality factor that has not been taken into account so far; the effect of ship traffic and mortality by ship propeller strike. There are indications of this kind of mortality, e.g. sometimes, substantial numbers of severely damaged silver eels are found at the shores of the Waal (the main branch of the Rhine where heavy shipping traffic occurs). Also, in our telemetry studies, we still have a substantial part of silver eel disappearance (e.g. up to one third in a river Meuse study, Winter et al. 2006) during downstream migration in rivers that cannot be attributed to other mortality causes. Ship traffic impact is a potential candidate factor in these cases. So far, these observations are only anecdotal. There are, however, no research or dedicated studies available on the impact of ship traffic, i.e. propeller strike mortality, for silver eel in the Netherlands. Also in other countries this knowledge is largely lacking.

7 Estimates of key stock indicators

The key stock indicators are based on the results from other chapters as well as additional information on catches. To calculate the Lifetime Anthropogenic Mortality (*LAM*, *Table 1.2*) for each period, mortalities for yellow eel and silver eel were split into a fishing and a barrier component. Mortality by barriers passage is assumed to affect only silver eel. *LAM* can be expressed as the 'spawner per recruit ratio' (*%SPR*, ICES 2014 and references therein). This ratio can be calculated based on the percentage of overall silver eel mortality from fishing and migration (α) and the percentage silver eel production from yellow eel when accounted for the yellow eel fishing mortality (β, *Figure 7.2*). To calculate β, maturation to silver eel is estimated in order to estimate yellow eel fishing mortality (see *Chapter 3*). Maturation is calculated before silver eel mortality from fishing or barriers. Consequently, β is the silver eel production from yellow eel with fishing mortality on yellow eel ($F_{yellow}>0$) divided by the silver eel production from yellow eel without yellow eel fishing mortality ($F_{yellow}=0$). Hence, β is a measure of the current silver eel production (with current $F_{yellow}>0$) relative to the best possible production in absence of yellow eel fishing mortality ($F_{yellow}=0$).

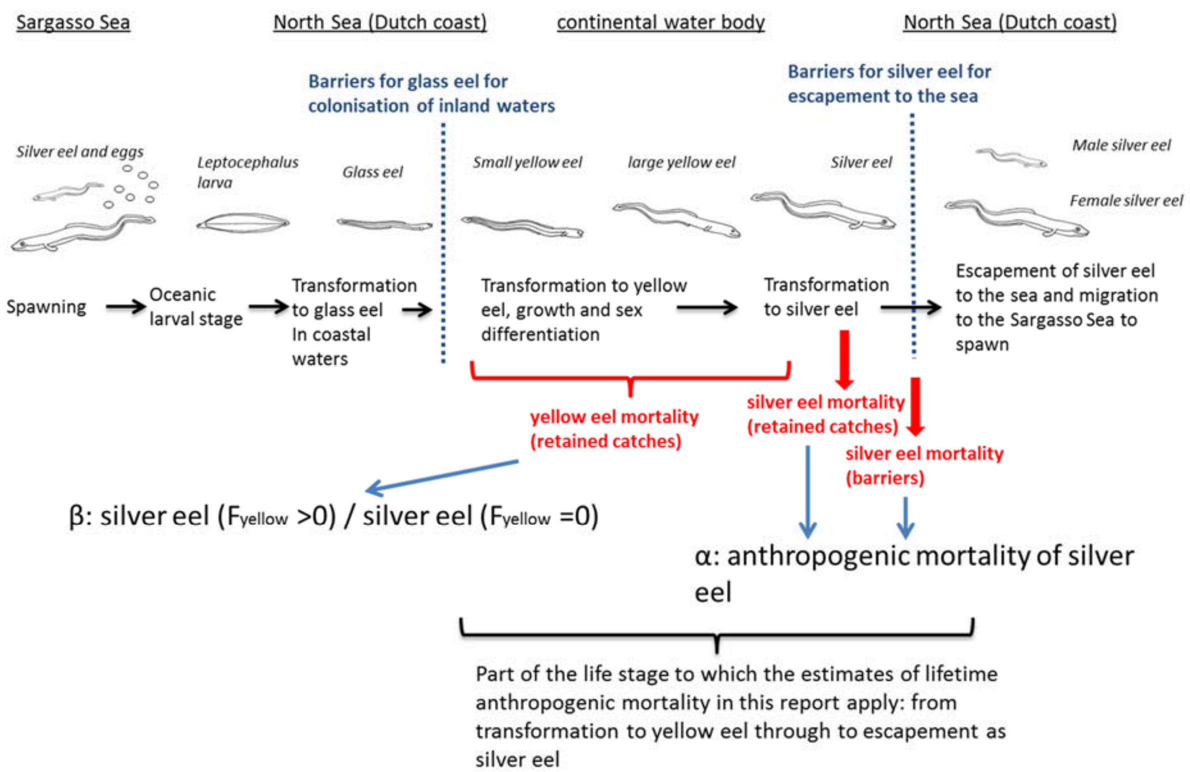

Figure 7.2 The life cycle of a European eel, with the part of the life cycle to which the Dutch Eel Management Plan (EMP) applies and for which 'Lifetime Anthropogenic Mortality' (LAM) rate estimates α and β are made in this report.

7.1 Yellow eel anthropogenic mortality

Yellow eel anthropogenic mortality is defined as fishing mortality from both commercial and recreational catches, including mortality from capture and release. Mortality from non-reported catches such as poaching or mortality caused by barriers is not included here.

The fishing mortality ($\hat{F}$) is a function of the proportion of retained catches and the estimated standing stock, following the equation:

$$\hat{F} = -ln(1 - (RC/(biomass+RC)))$$

where *RC* is the retained catch of yellow eel and biomass the biomass of yellow eel > 30 cm, both in tonnes. An assumption of calculating the fishing mortality in this way is that all mortality occurred at once and effects of natural mortality on stock trends within a year are ignored. However, this assumption seems reasonable because eel is a long lived species and year-on-year trends are relatively small.

In order to calculate the fishing mortality $\widehat{F}$, information on the retained catches is needed from both recreational and commercial fisheries, which includes catches from marine waters. For the period 2005-2007, retained catches of yellow eel were estimated at 640 and 200 tonnes by commercial and recreational fishers respectively (*Table 7.1*). In addition, 280 tonnes of silver eel was landed by commercial fishers (The Ministry of Agriculture, Nature and Food quality, 2009; Tables 2.3.1 and 2.3.3; including Vriese et al. 2007). Released recreational catches were estimated to be 100 tonnes.

For the period 2008-2010 the total amount of commercial catches for 2010 was used, 452 tonnes (Ministry of LNV). For the periods 2011-2013 and 2014-2016 the average of the total commercial catches from 2011-2013 and from 2014-2016 were used, 344 and 303 tonnes respectively (Ministry of LNV). The catch was split into yellow and silver eel based on the length frequency distribution, the sex ratio, maturation and the length-weight relationship (*Chapter 2*). This resulted in an estimate of 56% yellow eel in the retained catches. Recreational retained and returned freshwater catches in 2010-2011, 2012-2013 and 2014-2015 were used (van der Hammen et al. 2013 & 2017). A mean estimate of 12% catch & release mortality was assumed to calculate the losses from returned eel from the recreational fisheries (van der Hammen & de Graaf, 2017). It is assumed that the recreational catches consist only of yellow eel. The amount of retained commercial and recreational catches is decreased every period. Also, the proportion of retained catches versus released catches by recreational fishers has decreased every time period. An overview of estimated retained and released catches by recreational and retained catches by commercial fishers for the three periods is given in *Table 7-1*.

Table 7-1 Overview of fresh water commercial and recreational catches for the periods in tonnes. The released recreational catches are converted into a loss based on catch-&-release mortality of 12%, the total biomass of released eel by recreational fishers is given in parentheses.

	2005-2007		2008-2010		2011-2013		2014-2016	
	Silver eel	Yellow eel	Silver eel	Yellow eel	Silver eel	Yellow eel	Silver eel	Yellow eel
Recreational retained	-	200	-	100	-	59	-	30
Recreational Released loss (total)	-	12 (100)*	-	21 (175)	-	25 (212)		29 (264)
Commercial	280	640	194	248	151	193	133	170
Total	280	852	194	369	151	277	133	229

**rough estimate (extrapolation) based on the trend in the ratio between retained/released eel in 2008-2010 and 2011-2013.*

The total yellow and silver eel biomass, for the different periods and scenarios, is presented in *Table 5-4* (*Chapter 5*). Based on the function above, $\widehat{F}$ is calculated for each period and scenario (*Table 7-2*).

Table 7-2 Fishing mortality ($\widehat{F}$) estimates for yellow and silver eel, for each scenario and period.

	2005-2007	2008-2010	2011-2013	2014-2016
Yellow eel				
Scenario 1	0.32	0.18	0.14	0.11
Scenario 2	0.17	0.09	0.07	0.04
Scenario 3	0.15	0.07	0.06	0.04
Silver eel				
Scenario 1	0.25	0.26	0.21	0.16
Scenario 2	0.16	0.14	0.11	0.07
Scenario 3	0.14	0.12	0.10	0.06

As a logical result from the 3 scenarios with different biomass estimates, for each period there is a decreasing $\widehat{F}$ with increasing biomass estimate.

Parameter *β (Table 7-3)*, used to calculate *%SPR*, is the percentage of silver eel realized relative to a situation without fishing mortality: $\beta = 100 * (B_{current}/(B_{best}*(1\text{-}\alpha))$.

Table 7-3 Estimates of β per period and per scenario.

	2005-2007	2008-2010	2011-2013	2014-2016
Scenario 1	12.3	27.3	35.5	41.5
Scenario 2	28.2	51.3	59.2	67.8
Scenario 3	33.6	57.1	64.6	72.6

7.2 Silver eel anthropogenic mortality estimate (α)

Silver eel anthropogenic mortality (α) consist of mortality during migration from freshwater to the sea (*Chapter 6*) and fishing mortality as presented above (see also *Figure 7.2*). The mortality is calculated as the percentage of losses due to anthropogenic mortality relative to the initial silver eel biomass:

$$\alpha = 100 * (1 - ((B_{start} - RC)(1 - M_{barrier})/B_{start})$$

Where B_{start} is the silver eel biomass before silver eel mortalities have taken place; $\boldsymbol{RC}$ is the retained silver eel catch; and $M_{barrier}$ is the percentage barrier mortality.

For each period and scenario the parameters were calculated: B_{start} (*Chapter 5*), *RC* (*section 7.1*) and $M_{barrier}$ (*Chapter 6*). With these ingredients and the equation above the values of α were calculated for each scenario and period (*Table 7-4*).

Table 7-4 The estimates of α for the scenario's and periods, α being the percentage of silver eel lost through anthropogenic mortality.

	2005-2007	2008-2010	2011-2013	2014-2016
Scenario 1	42.5	46.0	40.2	32.6
Scenario 2	33.8	34.9	31.0	24.0
Scenario 3	32.1	33.2	29.6	22.9

7.3 Estimation of *%SPR*, $B_{current}$ and B_{best}

The yellow and silver eel mortality rates and the subsequent values of α and β can be used to calculate the *%SPR*, the spawner to recruit ratio. Here, *%SPR* is defined as the current escapement of silver eel as a percentage of the best possible escapement (if all anthropogenic mortalities were mitigated). *%SPR* is calculated as following:

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

An estimate of lifetime anthropogenic mortality is then given by:

$$LAM = 100 - \%SPR$$

B_{best}, an estimate of the best possible escapement of silver eel (if all anthropogenic mortalities for yellow and silver eel are zero), is calculated as (expressed as a percentage):

$$B_{best} = (B_{current} * 100)/\%SPR$$

$B_{current}$ is the current escapement of silver eel, the surviving part of the silver eel stock (B_{start}) after all silver eel anthropogenic mortality (1-α). These indicators were calculated for the different periods and scenarios (*Table 7-5*).

Table 7-5 Overview of all stock indicators per period and per scenario. Yellow eel and silver eel stock estimates refer to eel larger than 30 cm. Scenario 2 (grey) is the best guess estimate.

2005-2007			**Scenario**	
		1	**2**	**3**
Yellow eel mortality	Yellow eel stock (tonnes)	2277	4494	5381
	Retained catch (tonnes)	852	852	852
	$\hat{F}$	0.32	0.17	0.15
	β	12.3%	28.2%	33.6%
Silver eel mortality	Silver eel stock (tonnes) (B_{start})	983	1585	1811
	Retained catch (tonnes)	280	280	280
	Mortality Barriers	20%	20%	20%
	α	42.5%	33.8%	32.0%
$B_{current}$	Tonnes	566	1049	1239
B_{best}	Tonnes	8041	5619	5434
%SPR	Spawner per recruit, %	0.07%	18.7%	22.8%
LAM	Lifetime anthropogenic mortality	93.0%	81.3%	77.2%

2008-2010			**Scenario**	
		1	**2**	**3**
Yellow eel mortality	Yellow eel stock (tonnes)	1887	4083	4948
	Retained catch (tonnes)	369	369	369
	$\hat{F}$	0.18	0.09	0.07
	β	27.3%	51.3%	57.1%
Silver eel mortality	Silver eel stock (tonnes) (B_{start})	651	1255	1474
	Retained catch (tonnes)	194	194	94
	Mortality Barriers	23%	23%	23%
	α	46.0%	34.9%	33.1%
$B_{current}$	Tonnes	353	816	993
B_{best}	Tonnes	2390	2445	2601
%SPR	Spawner per recruit, %	14.8%	33.4%	38.2%
LAM	Lifetime anthropogenic mortality	85.2%	66.6%	61.8%

2011-2013			**Scenario**	
		1	**2**	**3**
Yellow eel mortality	Yellow eel stock (tonnes)	1865	3999	4852
	Retained catch (tonnes)	277	277	277
	$\hat{F}$	0.14	0.07	0.06
	β	35.6%	59.2%	64.6%
Silver eel mortality	Silver eel stock (tonnes) (B_{start})	639	1258	1486
	Retained catch (tonnes)	151	151	151
	Mortality Barriers	22%	22%	22%
	$\boldsymbol{\alpha}$	40.1%	31.0%	29.5%
$B_{current}$	Tonnes	383	867	1055
B_{best}	Tonnes	1803	2123	2320
%SPR	Spawner per recruit, %	21.3%	40.9%	45.5%
LAM	Lifetime anthropogenic mortality	78.7%	59.1%	54.5%

2014-2016			**Scenario**	
		1	**2**	**3**
Yellow eel mortality	Yellow eel stock (tonnes)	2063	5038	6192
	Retained catch (tonnes)	229	229	229
	$\hat{F}$	0.11	0.04	0.04
	β	41.5%	67.8%	72.6%
Silver eel mortality	Silver eel stock (tonnes) (B_{start})	744	1795	2190
	Retained catch (tonnes)	133	133	133
	Mortality Barriers	18%	18%	18%
	$\boldsymbol{\alpha}$	32.6%	24.0%	22.8%
$B_{current}$	Tonnes	503	1365	1698
B_{best}	Tonnes	1797	2647	3031
%SPR	Spawner per recruit, %	28.0%	51.5%	56.0%
LAM	Lifetime anthropogenic mortality	72.0%	48.5%	44.0%

8 Evaluation of the EMP

In this chapter the impact of the eel management is evaluated using the modified ICES precautionary diagram.

8.1 ICES Precautionary Diagram

ICES developed a precautionary approach (PA) framework (ICES 2014 and references therein). The PA framework uses limit reference points (LRP; B_{lim} and F_{lim}) reflecting stock states that should be avoided, and precautionary reference points (PRP; B_{pa} and F_{pa}) reflecting the risk of crossing the LRPs. Both LRP and PRP are defined in terms of fishing mortality (*F*) and spawning stock biomass (*B*) (*Table 8-1*). The ICES precautionary approach framework illustrates the division between management and science. While science defines the limit references points, the decisions on acceptable risk levels (i.e. precautionary reference points) should be made by the management.

Reference points reflecting the state of the whole stock are needed to use the ICES precautionary approach. For the eel stock, these reference points have not been established. The eel stock is divided over many waterbodies, in marine and in freshwater, and over many countries. This makes an assessment of the total eel stock and the calculations of reference points extremely difficult. The ICES precautionary diagram can therefore not directly be used for the eel stock. In the next paragraph is explained how the precautionary diagram is modified for the eel case, such that it can be used by each Member State.

Table 8-1 Reference points of the ICES precautionary approach framework (ICES 2014).

Reference point	Definition
B_{lim}	Biomass limit below which a stock is considered to have reduced reproductive capacity.
F_{lim}	Exploitation rate that is expected to be associated with stock 'collapse' if maintained over a longer time.
B_{pa}	Biomass above which the stock is considered to have full reproductive capacity, having accounted for estimation uncertainty (precautionary buffer to avoid that true spawning stock biomass is at B_{lim} when the perceived spawning stock biomass is at B_{pa})
F_{pa}	Exploitation rate below which exploitation is considered to be sustainable, having accounted for estimation uncertainty (precautionary buffer to avoid that true fishing mortality is at F_{lim} when the perceived fishing mortality is at F_{pa}.)

8.2 ICES Precautionary Diagram modified for eel

Over the past years the ICES Study Group on International Post-Evaluation of Eel (SGIPEE) and the ICES Working Group on Eels (WGEEL), have progressively been working on a pragmatic framework for an evaluation of the status of the eel stock and the effect of management measures. This framework for deriving stock indicators is based on four estimates (ICES 2014 and references therein):

Estimate	Definition
B_0	Silver eel biomass without any anthropogenic influences (pristine biomass).
$B_{current}$	Silver eel biomass that currently (assessment year) escapes to the sea to spawn.
B_{best}	Silver eel biomass without any anthropogenic influences on the current (assessment year) stock.
LAM	Life time anthropogenic mortality; the fishing mortality and the mortality outside the fishery (hydropower plants, pumping stations etc.).

In the Dutch Eel Management Plan (Ministry of Agriculture, Nature and Food Quality, 2009), the estimate of pristine biomass in inland waters (B_0 = 10.400 t) was provided. The other estimates ($B_{current}$, B_{best} and *LAM*) are estimated in the previous chapters of this report (see *Table 7-5*).

In the modified ICES precautionary diagram (*Figure 8-1*), the horizontal axis reflects the status of the stock in relation to the estimated pristine situation ("*% escaping silver eel, 100*$B_{current}/B_0$*"). The vertical axis reflects the current lifetime anthropogenic mortality; the ratio of the current silver eel biomass and the biomass without anthropogenic mortality ("*LAM, 100*(1-$B_{current}/B_{best}$*"). The horizontal axis demonstrates to what extent the status of the eel stock is sustainable while the vertical axis illustrates to what extend the current use and management of the stock are sustainable.

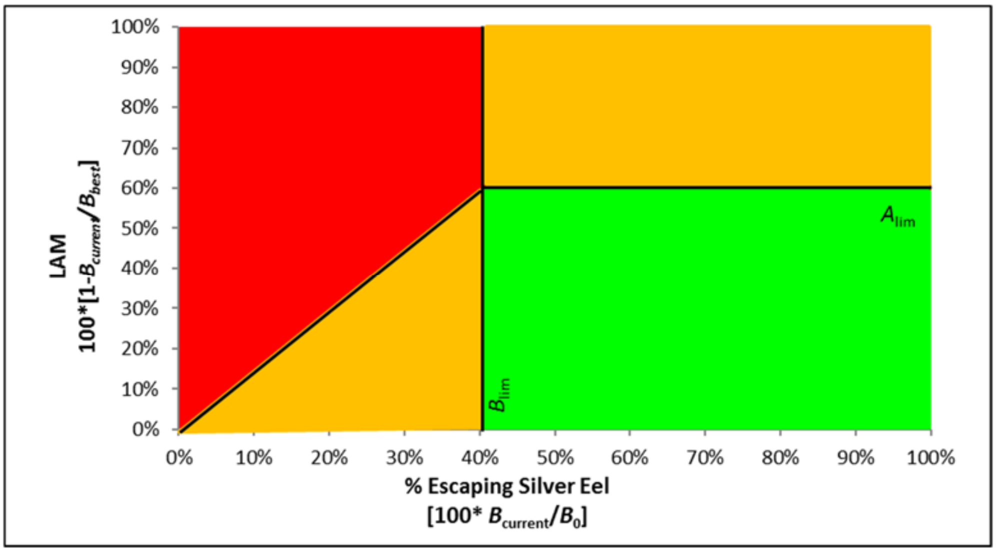

Figure 8-1 ICES modified precautionary diagram. The horizontal axis represents the status of the stock in relation to pristine conditions (silver eel escapement expressed as a percentage of the pristine escapement). The vertical axis represents the impact made by anthropogenic mortality. LAM = Lifetime anthropogenic mortality, a measure for the current silver eel biomass ($B_{current}$) relative to the current best possible biomass (B_{best}, current biomass without anthropogenic mortality).

8.2.1 Reference points ICES modified Precautionary Diagram

Because the reference points for the ICES PA approach (B_{lim}, B_{pa}, F_{lim}, F_{pa}) are not available for eel, alternative biomass (*B*) and mortality reference points had to be developed. In the eel case the mortality reference points will consist not only of *F* (fishing mortality) but of all anthropogenic mortality (*A*).

Biomass reference points:

ICES (2002) discussed a potential reference value for spawning-stock (escaping silver eel) biomass:

"a precautionary reference point for eel must be stricter than universal provisional reference targets. Exploitation, which provides 30% of the virgin (F=0) spawning-stock biomass is generally considered to be such a reasonable provisional reference target. However, for eel a preliminary value could be 50%."

Thus, ICES advised to set the biomass reference point (e.g. B_{lim}) above the universal value (30%), at a value of 50% of the virgin spawning-stock biomass (e.g. B_0). The EU (Council Regulation 1100/2007) decided to set B_{lim} at 40% of B_0, in-between the universal level (B_{lim} =30%) and the level advised by ICES (B_{lim} = 50%).

Mortality reference values:
ICES has not advised on specific values for mortality-based reference points. Currently the ICES advice is a precautionary advice and is not reference point related (ICES 2017):

"ICES advises that when the precautionary approach is applied for European eel, all anthropogenic impacts (e.g. recreational and commercial fishing on all stages, hydropower, pumping stations, and pollution) that decrease production and escapement of silver eels should be reduced to – or kept as close to – zero as possible."

An alternative mortality limit reference point (A_{lim}) was developed as follows. As said above, the Eel Regulation sets a limit for the escapement of silver eel (B_{lim}) at 40% of the pristine escapement (B_0). An eel stock with a biomass of escaping silver eel of 40% of B_0 corresponds to a lifetime anthropogenic mortality limit of A_{lim} = 0.92 (Dekker 2010), which corresponds to *%SPR* = 40% and a *LAM* of 60%. If $B_{current}$ is smaller than B_{lim} a proportional reduction in mortality reference values is applied, i.e. a linear relation between the advised mortality rate (A_{lim}) and biomass (*Figure 8-1*).

The modified ICES precautionary diagram developed by ICES needs to be carefully interpreted. The target biomass (B_{lim} = 40% B_0) has not been scientifically assessed to determine if it can be used as a true precautionary biological limit reference point. There is no guarantee that even if all Member States were at 40% B_0 that the eel stock would be recovered. Likewise, A_{lim} is also not scientifically assesses and is derived from B_{lim}, with a reducing scale of A_{lim} below B_{lim}. Therefore there is also no scientific basis that A_{lim} is a true reference point above which the eel stock will recover.

In conclusion, the ICES modified diagram can be used to demonstrate the status of the eel stock with respect to the management targets/limits as formulated in the EC Eel Regulation, but cannot be used to predict the recovery of the stock. For this reason, the ICES Advisory Committee (ACOM) has been reluctant to advise on the status of the eel stock without scientifically testing the targets/limits developed by ICES to ensure they are precautionary and will lead to a recovery.

8.3 Evaluation of the Dutch Eel Management Plan

The status of the Dutch part of the eel stock in 2005-2016 and hence, the evaluation of the Dutch Eel Management Plan is graphically presented, using the ICES Modified Precautionary Diagram (*Figure 8-1*). The evaluation demonstrates that before and after the implementation of the EMP the status of eel in Dutch waters remains in a situation regarded as "undesirable" with high mortality and low biomass. Current biomass of escaping silver eel is 13% of the pristine situation which is below the target of 40%. Current lifetime anthropogenic mortality (*LAM* = 48%) is above the recommended mortality (*LAM* = 19.5%) at the current estimate of the %of escaping silver eel (13%, *Table 8-2*).

Table 8-2 Stock indicators used to evaluate the impact of the EMP on the biomass of escaping silver eel (horizontal axis modified precautionary diagram) and anthropogenic mortality (vertical axis modified precautionary diagram).

Stock Indicator	2005-2007	2008-2010	2011-2013	2014-2016
B_0*	10400 t	10400 t	10400 t	10400 t
$B_{current}$	1049 t	816 t	867 t	1365 t
B_{best}	5619 t	2445 t	2123 t	2547 t
$100^* B_{current}/B_0$	10%	8%	8%	13%
%SPR	19%	33%	41%	52%
LAM	81%	67%	59%	48%

* Excluding coastal waters (2600 t)

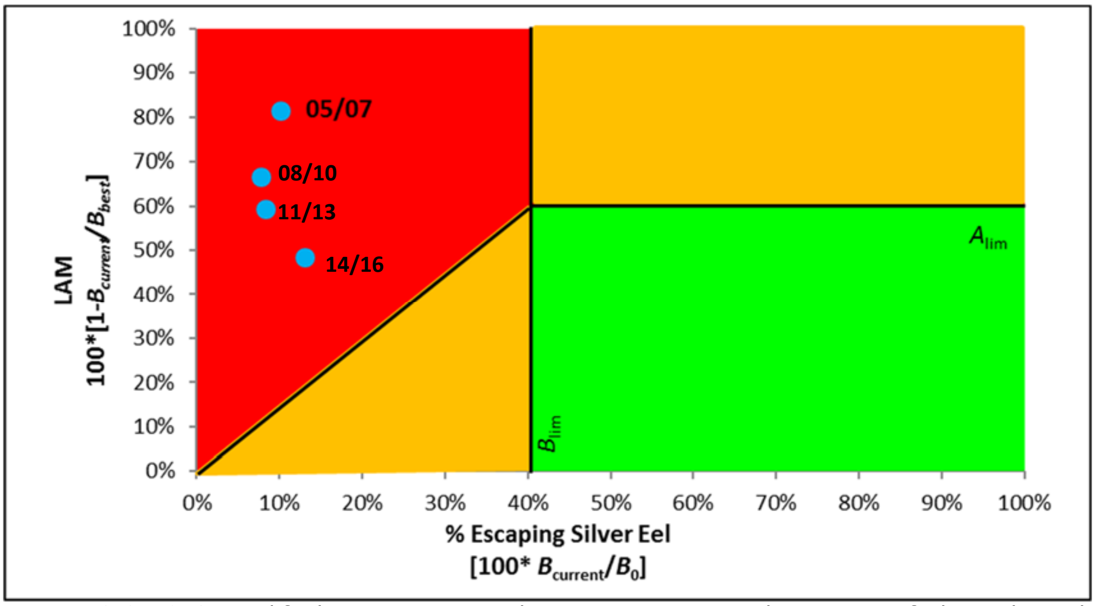

Figure 8-2 ICES modified precautionary diagram presenting the status of the eel stock in the Netherlands in 2005-2007, 2008-2010, 2010-2013 and 2014-2016 with respect to management targets. The horizontal axis represents the status of the stock in relation to pristine conditions. The vertical axis represents the impact made by anthropogenic mortality. LAM = Lifetime anthropogenic mortality.

<u>Anthropogenic mortality (vertical-axis)</u>

A reduction in lifetime anthropogenic mortality (*LAM,* vertical axis) can be achieved by reducing fishing mortality and barrier mortality. A reduction in anthropogenic mortality is therefore the direct result of the measures taken by a Member State. In the Netherlands, the implementation of the EMP has resulted in a reduction in *LAM* between 2005 and 2016 from 81% to 48%. In each 3 year period, a reduction was achieved (*Figure 8-2*). This reduction was mainly the result of a decrease in fishery mortality, both commercial and recreational: retained catches (landings) of both commercial and recreational fishery strongly decreased between 2005-2007 and 2014-2016. Barrier mortality did not show such a strong decrease. It should be noted however, that the barrier mortality in the periods 2005-2007 and 2008-2010 was assessed with a more general crude approach (Bierman et al. 2012) than in the periods 2011-2013 and 2014-2016, when a more bottom up approach was used for the 'boezem' and national waters (see Chapter 6). The new approach will be more accurate and therefore more reliable. When comparing barrier mortality during 2011-2013 to 2014-2016 it has reduced from 22% to 18% (Table 7-5) mainly due to measures at hydropower plants (new management scheme), and replacing some pumping stations with 'fish-friendlier' pumping types. Since it is unlikely that barrier mortality has increased from 2005-2007 to 2011-2013 (there were no new barriers, no hydropower stations or pumping stations were placed during this period), it is more likely that the general approach used to asses barrier mortality in 2005-2007 and 2008-2010 was an underestimation of barrier mortality. Water boards did invest substantially in improving migratory opportunities at migration barriers, but most solutions targeted to facilitate upstream migration. Potentially, this has improved glass eel immigration into inland waters and as a consequence indirectly enhanced the potential silver eel biomass starting to migrate in the different waters. Mitigation mortality in a downstream direction is more difficult since it requires replacing pumping stations or hydropower plants or deflecting silver eel to alternative routes with no mortality, for which effective measures are still largely lacking (Kroes et al. 2013). The reduction in barrier mortality during 2005-2016 appears relatively small (although perhaps somewhat underestimated), when compared to the reduction in fishing mortality and as a result the contribution of barrier mortality increased slightly between 2005-2007 and 2014-2016 (*Figure 8-3*) most likely due to the increase in the number of silver eel surviving the fishery and having to pass barriers during their migration to the sea. The overall mortality of eels passing a barrier did, however, decrease from 20% (2005-2007) to 18% (2014-2016) (*Chapter 7*). The main source of quantified anthropogenic mortality in 2014-2016 remained commercial and recreational fishing mortality, although the relative contribution of fishing mortality has decreased to a great extent (*Figure 8-3*).

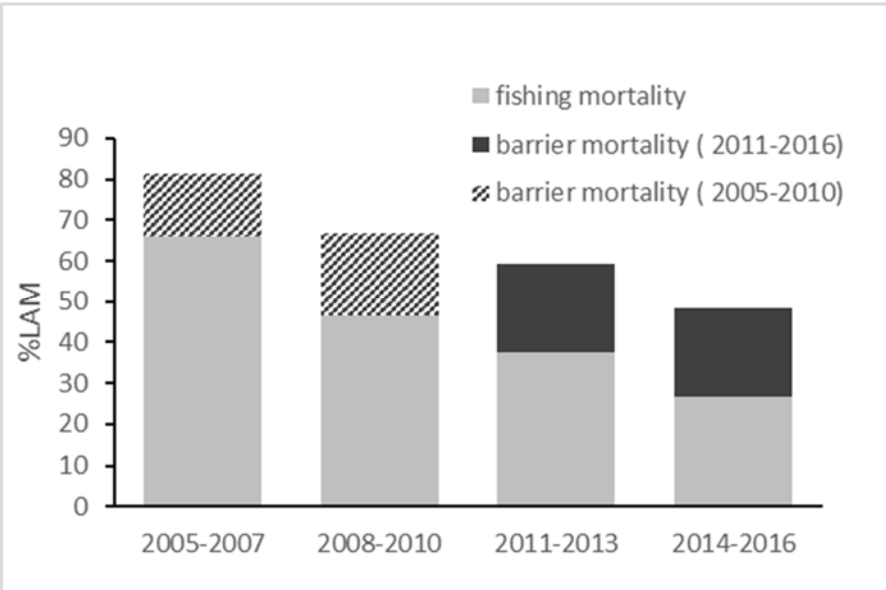

Figure 8-3 Changes in the contribution of F (fishering mortality) and H (barrier mortality) to the life time anthropogenenic mortality of eel in the Netherlands. For the evaluations in 2005-2007 and 2008-2010 a more crude general approach was used to assess barrier motality (shaded bars), whereas for the 2011-2013 and 2014-2016 evaluation a more detailed bottom up approach was used for the boezem and national waters (black bars)

<u>Biomass escaping silver eel (horizontal-axis)</u>

Between 2005-2007 and 2008-2013, there was a modest decrease in the biomass of escaping silver eel while between 2008-2013 and 2014-2016 there was a modest increase (horizontal axis; *Figure 8-2*). Large differences between years in biomass were not expected as current silver eel escapement has largely been determined by processes (recruitment, anthropogenic mortality) that occurred in the previous 5-15 years. Furthermore, an increase in glass eel recruitment will, at the earliest, result in an increase of silver eel after 5-15 years, and glass eel recruitment has not significantly increased after the implementation of the EMP in 2009. Moreover, the total silver eel biomass (x-axis) depends not only on the status of the Dutch part of the eel stock, but also on the stock status in the other Member States.

On the short term, the maximum possible contribution on recovery of the eel stock by the Netherlands is a reduction of anthropogenic mortality to close to zero. However, if the Netherlands would reduce all anthropogenic mortality to zero, a direct recovery of the European eel stock is still unexpected. In order to maximize the chance of recovery, maximum protection of eel will have to be accomplished throughout its range (inside and outside Europe). Even then there is no guarantee for the recovery of the European eel stock because the cause of the decline of the stock remains unknown. The European eel directive is developed to cover the risk that the decline of the European eel population is due to a decline in silver eel escapement due to anthropogenic mortality.

In other words, the Netherlands can be hold accountable for changes in anthropogenic mortality (vertical axis in the modified ICES precautionary diagram). However, the status of the whole eel stock depends on the actual cause(s) of the decrease and on the protective actions undertaken in other countries. The responsibility for improvement of eel stock lies with all countries where European eel is distributed.

Three scenarios

In this study, the reference points and biomasses are estimated for 3 scenarios (*Table 5-1*). These scenarios differ in catch efficiency and in the ratio of offshore and inshore densities (scenario1: catch efficiency = 66% and density offshore compared to inshore = 33%, scenario2: catch efficiency = 20% and density offshore compared to inshore = 50%, scenario3: catch efficiency = 20% and density offshore compared to inshore = 66%, see *Table 5-1*). The results show that these scenarios vary in the predicted *LAM* and in the % escaping silver eel (*Figure 8-4*). Especially the different catch efficiency results in a higher estimation of *LAM* (difference between scenario 1 and 2/3). However, none of the scenarios results in a positive state of the stock for eel (*Figure 8-4*).

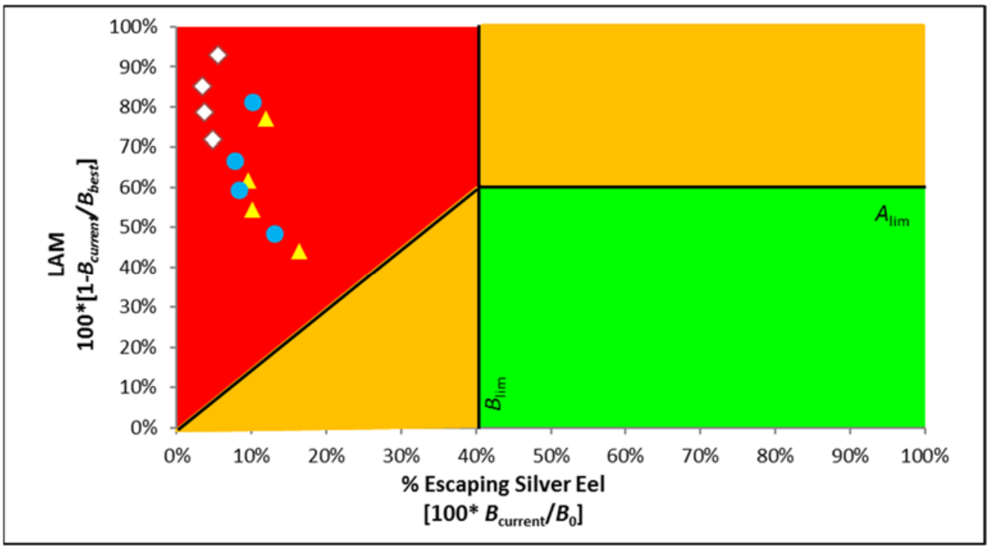

Figure 8-4 ICES modified precautionary diagram presenting the status of the eel stock in the Netherlands in 2005-2007, 2008-2010, 2010-2013 and 2014-2016 with respect to management targets. Different colors and symbols represent different scenarios of catch efficiency of the electric dipping net and of the difference in density between inshore and offshore. White diamonds respresent scenario 1, blue circles respresent scenario 2 and yellow triangles respresent scenario 3 (Table 5-2).

Uncertainty in Pristine silver eel biomass (B_0)

The B_0 value for inland waters in the Netherlands is set at 10400 t. However, the value has a wide range (inland waters B_0 = 10400 t, range 5200-16200 t). In *Figure 8-5* the effect of the variation in B_0 is shown for scenario 2. The results show that these scenarios vary substantially in the % escaping silver eel (*Figure 8-5*). However, none of the scenarios results in a positive state of the stock for eel (*Figure 8-5*).

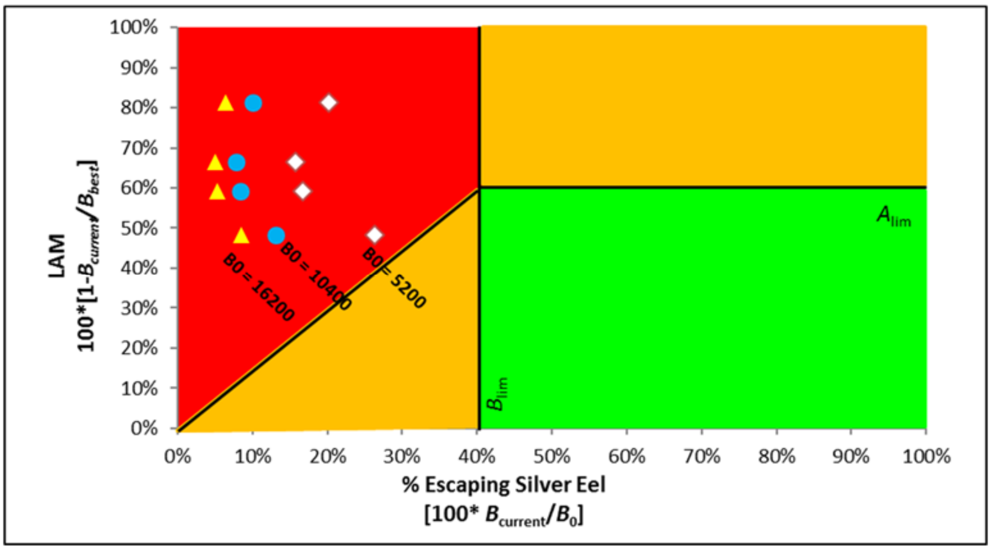

Figure 8-5 ICES modified precautionary diagram presenting the status of the eel stock in the Netherlands in 2005-2007, 2008-2010, 2010-2013 and 2014-2016 with respect to management targets.White diamonds respresent the results using the lower limit of B_0, blue circles respresent the results using B_0 and yellow triangles respresent the results using the upper limit of B_0.

8.4 Uncertainties of the current evaluation

The estimates of the stock indicators (B_{best}, $B_{current}$, B_0 and *LAM*) used to evaluate the status of the stock (*Figure 8-2*) need to be interpreted with care due to the significant level of uncertainty surrounding these estimates. The main uncertainties are described below.

Pristine silver eel biomass (B_0)

The B_0 value for inland waters in the Netherlands is set at 10400 t. However, the value has a wide range (6500-20250 t, inland waters B_0 = 10400 t, range 5200-16200 t). In addition, this range has been subject to discussion. Initially the pristine silver eel biomass (B_0) in the Netherlands, was set at 10.000-15.000 t (Klein Breteler 2008). In a review it was concluded that B_0 was between 6500-20250 t (Eijsackers et al., 2009). However, ICES (review of the national eel management plans, ICES 2010) did not accept all arguments of Eijsackers et al. (2009) and set B_0 at 13000 t. A second review of B_0 values for the Netherlands 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 (Rabbinge et al., 2013).

Anthropogenic mortality (B_{best}, $B_{current}$ and *LAM*)

The estimates for lifetime anthropogenic mortality (*LAM*) is set by the values of B_{best} and $B_{current}$ (*Table 7-5*). These values are uncertain due to the following main assumptions that influence $B_{current}$:

- the efficiency of the electrofishing gear
- distribution of eel over the surface of a water body in the static spatial population model
- assumptions of *F* when estimating eel populations using the demographic population model for some of the larger lakes (*Chapter 3*).

Unquantified sources of anthropogenic mortality

The estimated lifetime anthropogenic mortality is most likely an *underestimate* of the true anthropogenic mortality because some sources of mortality have not been quantified:

- poaching
- yellow eel mortality in hydropower plants and pumping stations
- impact of (human-induced) viruses, parasites and pollution

When interpreting the impact of the eel management plan on the status of the eel stock in the Netherlands, it is highly important to keep these uncertainties in mind.

9 Conclusions and recommendations

The European Commission requested the Member States to evaluate the progress of the Eel Management Plans and the status of the European eel stock. In this report, we have described and discussed 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 *LAM*) for four periods (2005-2007, 2008-2010, 2011-2013 and 2014-2016). In this chapter, we discuss the used methodologies and provide recommendations for further improvements of the models for the next evaluation.

9.1 Demographic model

The eel demographic model plays an important role in the final assessment by estimating fishing mortality in the large lakes and estimating the contribution of anthropogenic mortality during the yellow eel stage to Lifetime Anthropogenic Mortality (*LAM*). The impact of yellow eel mortality on silver eel production was estimated using the demographic model. This model incorporates the key biological processes of eel during the yellow eel stage; sex-specific growth, sex-specific maturation and mortality. Uncertainty and possible biases in the model estimates arise from the uncertainty in these key biological processes:

Sex-ratio: The higher the proportion of females, the higher the expected silver eel biomass per glass eel. Females grow larger, older and, after a certain age, faster than males.

Growth rates: The higher the growth rate, the higher the expected silver eel biomass per glass eel. Increased growth leads to increased maturation rates and hence faster withdrawal of silver eel from the population. With the current parameterization of growth at age, males grow slower than with previous parametrization (Bierman et al. 2012 and van de Wolfshaar et al. 2015). This decreased growth rate results in a decrease of the estimated fishing mortality.

Size at maturity: If individuals mature at a smaller size, the fishing mortality decreases because silver eel will leave the inland population. The proportion of silver eel in the retained catches was used for estimating maturation-at-length rates. Depending on representativeness of the sampling this may lead to an over- or under-representation of silver eel in the Dutch eel population. However, samples are taken over a five months period, including the period close to the start of migration in order to mitigate over- or underestimation of the proportion of silver eel.

Natural mortality: Natural mortality is a difficult parameter to assess. It depends on many factors, such as predation, water temperature, pollution and food conditions. It is also not expected to be the same for all stages and is also not constant through time. The natural mortality used in the demographic model (*Chapter 3*) 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.

Each year, from an additional number of eel, the length, weight, age and maturation are assessed, improving the estimation of the biological keys each year. This time period (2014-2016), the length-weight relationship and the maturation-at-length did not change significantly compared to the previous assessment, which implies high reliability. However, compared to the previous assessment, we find a decrease in the growth rate of males which led to a decrease in the estimate of *F*.

The utility of the demographic model for estimating fishing mortality using relative length-frequencies is questionable. To interpret present-day data or historical stock trends, a high quality index of recruitment, trends in sex-ratios, sex-specific growth rates, natural mortality, and migration rates are required. Because eel recruitment and eel densities have decreased considerably, it is unlikely that parameters have remained constant. In many water bodies in the Netherlands, length-frequency distributions have shifted towards relatively more large eel; mean lengths in stock surveys have been increasing in Lake IJsselmeer, Lake Markermeer and upper reaches of the main rivers. In addition, the sex ratio has changed towards more females (*Figure 2-1*, *Figure 3-4*).

The best studied waterbody in the Netherlands is Lake IJsselmeer for which recruitment indices and a long-term stock survey are available. The main stock trends in Lake IJsselmeer - a decreasing trend of young eel and an increase in the abundance of larger eels - could be explained reasonably well by the model, but only to a certain extent. For the most recent years, this increase in large individuals was not captured by the model, indicating that they might come from elsewhere. They could have originated from the rivers, where fishing is prohibited since 2011 because of high contaminant levels. Alternatively, they could come from other countries. The model assumes a closed system which is not the case.

For this report, the demographic model was adjusted to include two conversion factors for the glass eel index, resulting in improved model performance. That raises the question if in Lake IJsselmeer there have been changes in either the inflow from Den Oever into the lake, changes in glass eel mortality (e.g. predation, disease, pollution) or changes in the migration of glass eel to other water bodies.

Like in previous assessments, a model fit was also done for Lake Markermeer. However, the model fitted the data very poorly due to very low numbers of eel in the survey. Therefore it was decided not to use the model – and the resulting estimated fishing mortality. The biomass of Lake Markermeer was estimated using the same method as for the Grevelingen and the Randmeren. An additional survey in Lake Markermeer targeting the shores might increase the number of eels in the survey. However a long time series is required for fitting the model.

9.2 Static spatial model

Anthropogenic mortalities of yellow eel were estimated using the proportion of the retained catch, out of the total standing stock of yellow eel. The standing stock of yellow eel is based on fisheries independent surveys (except the four large lakes), where catches per unit of effort were raised to total water body area to estimate the total standing stock.

The main advantages for estimating standing stock using the survey approach are:

- The estimates are based on large amounts of survey data which are collected using standardised protocols.
- The estimates are based on a transparent methodology, which relies mostly on two simple parameters (catch efficiency and eel distribution within 1.5 meters of the shore/bank).
- The estimates can be compared with independent estimates of standing stock such through capture-mark-recapture experiments.
- The estimates are spatially explicit, and can thus be used to obtain estimates of barrier mortality during migration.

The main weaknesses in the methodology is the uncertainty of the catch efficiencies of the surveys and the distribution over the waterbodies. This lack of knowledge results in uncertainty around the estimates of the standing stock. In this study, the sensitivity to catch efficiency is shown by the range in predictions using the three scenarios with differing catch efficiencies and eel distributions. To improve the quality of the estimate for the ditches, additional data was gathered in ditches, which led to a higher estimate of eel densities.

9.2.1 *Regionally managed waters*

In the biomass assessment for the regional managed water bodies, assumptions were made based on data availability. A problem with Water Framework Directive (WFD) fish survey data is that not all regional water boards provided (good quality) data. Several records could not be linked to a water body and these records were excluded from the analysis. Such a mismatch might be due to measurement error in GPS equipment, missing longitude and/or latitude or errors in data entry, such as typing errors in waterbody names. In this assessment, the data from all years were used for a single estimate to be able to cover all waterbodies. In the future, a running average could be investigated, to allow for changes over time.

9.2.2 *Nationally managed waters*

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

- The main rivers in the nationally managed waters include water bodies which are only connected to the river for a short period of time per year. No eel surveys are conducted in these water bodies and they are excluded in the current assessment. However, the surface area of these non-connected water bodies (64 ha) is negligible (<0.01%).

- In the current assessment the biomass estimate of one river section dominated the overall assessment. This is because the area has a large surface area and there was a relative unusual large catch of large eels. It is unknown if these eels were local and future monitoring will show if there is indeed an increase in biomass.

- Eels are not equally distributed over different habitats in the littoral zone. For example, eel densities are expected to be higher in complex habitats like groynes. At present, eel densities in nationally managed waters cannot be corrected for habitat (sand, vegetation, rocks).

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

In the current assessment, the eel stock in the large lakes (Lake IJsselmeer, Lake Markermeer, Randmeren and Grevelingen) was determined using the demographic population model of Lake IJsselmeer due to the lack of survey data in the other lakes. Capture-mark-recapture studies could provide independent estimates of the standing stock in these lakes. A better understanding of eel distribution in the lakes may allow in upscaling eel densities from the littoral zone to lakes as a whole. An alternative could be the development of an electric beamtrawl designed to effectively capture eel >30 cm length in large lakes and main rivers.

9.3 Silver eel migration model

The approach used in the previous evaluation (2011-2013), i.e. an approach based on average mortalities for polders and a bottom up site specific estimation of mortality rates for 'boezem' and national waters was updated in this evaluation.

For the bottom up approach, as used for boezem and national waters (2nd and 3rd hierarchy), the division of silver eels that end up at a certain barrier site over the different migration routes is needed. For some sites, good data on route selection is available, e.g. at the hydropower stations 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.

The current evaluation is based on more data than the previous evaluation. The bottom up approach in the boezem and national waters is therefore more complete. For the polders, the average mortality rate for silver eel was estimated to be ~35%, similar to 2011-2013. For boezem waters to national waters it was 14% in 2014-2016, and from 'boezem' to sea it was 6%, similar as in 2011-2013. Hydropower mortality rates were assessed to have lowered from 15-17% per site in 2011-2013, to 10-14% in 2014-2016. However, this was based on a modelling study on the altered hydropower management and not on field measurements. For the migration from national waters to sea, the mortality rate is somewhat lower than before, i.e. 0.9 % in 2011-2013 and 0.5% in 2014-2016.

9.4 Lifetime Anthropogenic Mortality (*LAM*)

The estimated %*LAM* consist of fisheries mortality over all life stages and barrier mortality of silver eel. The mortality caused by fisheries contribute the most to the *LAM,* although the relative contribution by barriers has increased. Also, individual females contribute more than males because females are on average heavier as they grow larger since they take longer to reach maturity. Mortality of silver eel occurs only on the proportion of eel that could develop from yellow eel to silver eel. This means that silver eel mortality contribute relative little to the estimated *LAM*.

Mortalities of yellow eel and silver eel were estimated using the retained catches and barrier mortalities in relation to the standing stock. The current *LAM* is calculated by taking the sum of the mortalities of all ages. This is not the same as the *LAM* that new recruits (glass eels) are expected to experience throughout their inland life span. The *LAM* of new recruits may differ from the current *LAM* because of different mortality rates compared to the current rates. This could be a result of effects of the measures taken to reduce mortality, such as closed areas (main rivers and some large canals) and reduction in fishing mortalities.

As mentioned in *Chapter 8*, the *LAM* estimation in this report are expected to be *underestimates* of the true *LAM* because not all sources of mortality have been quantified and accounted for, e.g.:

- yellow eel mortality in hydropower plants and pumping stations
- impact of human-induced viruses, parasites and pollution
- poaching

9.5 Evaluation of the EMP using stock indicators

The estimated key stock indicators B_{best}, $B_{current}$, B_0 and *LAM* have been evaluated in relation to management targets/limits as formulated in the EC Eel Regulation, using the modified ICES precautionary diagram (*Chapter 8*). The modified ICES precautionary diagram developed by ICES should be carefully interpreted. The target biomass (40% B_0) is a management target and it has not been scientifically determined whether it could be used as a true precautionary biological limit reference point. In other words, it is unknown that, if all Member States were at 40% B_0, the eel stock would recover.

Therefore, the diagram can be used to demonstrate the status of the eel stock with respect to the management targets/limits as formulated in the EC Eel Regulation. However, the Advisory Committee (ACOM) of ICES is reluctant to advise on the status of the eel stock without scientifically testing the targets/limits developed by ICES (2014 and references therein) to ensure they are precautionary and will lead to a recovery.

When interpreting the impact of the eel management plan on the status of the eel stock in the Netherlands using the modified ICES precautionary diagram, it is important to keep the following three aspects in mind:
- the limits/targets are management limits/targets and do not guarantee a recovery of the stock,
- the uncertainties surrounding the estimated indicators B_{best}, $B_{current}$, B_0 and *LAM* are high, and

- there are unquantified sources of anthropogenic mortality.

9.6 Recommendations

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

Demographic Model

Biological parameters

- *Sex-ratio*: The estimation of the sex ratio in Lake IJsselmeer for eels smaller than 30 cm is based on a relatively low number of eels and therefore the precision of the estimation could be improved by increasing the sample size.
- *Growth rate*: Growth rates could be improved by including eels smaller than 30 cm. Age and growth increments of small eel (<30 cm) are being determined as part of the WOT eel research programme.
- *Size at maturation*: The size at maturation (maturity ogive) for a given area should be improved by using data collected year round. Eels are now collected from fishermen with exception of the migration season.

Anthropogenic mortalities:

- Sources of anthropogenic mortalities that are not included in the assessment should be quantified such as yellow eel mortality pumping stations and hydropower plants and poaching. Experiments have now been conducted in collaboration with German scientists to determine catch-&-release mortality for eel and improve the current estimate. In the next report it will be attempted to quantify the C&R mortality.

Effort

- At present, only the potential effort is known (the number of permits is known, but the realized effort is not known, see *Chapter 3.2*). If the realized effort would be known, together with catch statistics this would give better insight in (trends in) the Landings per unit of Effort (LPUE).

Spatial Model

Regionally managed waterbodies (WFD survey data):

- The accessibility of WFD fish survey data of regionally managed waters should be improved. This could be done by establishing a central data base for the Netherlands, and ensure that the data is properly checked to ensure the quality of data.

Catch efficiency:

- Experiments should be conducted to determine the efficiencies of electrofishing for eel in different WFD water types in both nationally and regionally managed waters.

Habitat preference

- Knowledge should be obtained of the distribution of eel between the borders of a water body and the open water.

Survey data use:

- The survey data should be examined for differences caused by monitoring in different periods. Especially differences in the amount of silver eel between surveys in the migration period and outside the migration period should be studied.

Silver Eel Migration Model

Migration routes:

- Based on a new barrier assessment for migrating silver eel, silver eel mortality estimates were improved by using a weighted importance of individual barriers based on catchment size for the 'boezem' and national waters. The barrier-mortality model as presented here to estimate mortality of silver eels during migration can be further developed to enable a full 'bottom up and site-specific data driven' approach for all types of waters and barriers.

Silver eels migrating downstream from Belgium and Germany:
The mortality caused by hydropower stations on silver eels migrating downstream on the river Meuse from Belgium and the river Rhine from Germany ('foreign' silver eels) have not been taken into account in the estimation of *LAM* in this report. It is still unclear whether these mortalities should be included in the *LAM* of silver eels in the Netherlands or in the country where maturation took place (Germany, Belgium). It is recommended that international agreement is achieved how these mortalities should be accounted for when silver eels pass several Member States during migration.

9.7 International "level playing field" stock indicators

As many other European countries (France, UK, Ireland) are using similar spatial models to estimate yellow eel standing stock and silver eel production, close international co-operation and collaboration will enhance the quality and uniformity of these models in the future. In addition, fundamental differences exist among the Netherlands, Belgium, Germany and the UK with respect to converting fisheries landings to silver eel production, selection of the reference period and correcting for glass eel stocking when calculating B_0. Germany, Belgium and the UK probably underestimated B_0 (ICES 2010). An independent international review of the methods used to estimate the stock indicators is required to create a level playing field, to enhance trust among member states. Furthermore standardization of assessment methods is of utmost importance to ensure the recovery of the European eel stock and its sustainable exploitation. The need for a "level playing field" was acknowledged by the European Commission which intends to request an external scientific review of the methodologies used by Member States to estimate the stock indicators. This request is currently being prepared.

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Appendices

Appendix A: Water types used in the WFD

Table A1: Water body types defined within the Water Framework Directive in the Netherlands that were taken into account in this study of regionally managed waters.

Code water type	Description
M1a/b	Buffered ditches
M2	Weakly buffered ditches
M3	Buffered regional canals
M6a/b	Large, shallow canals with/without shipping
M7a/b	Large deep canals with/without shipping
M8	Buffered fen ditches
M10	Fen canals
M14	Shallow, relatively large, buffered lakes
M20	Relatively large, deep, buffered lakes
M23	Shallow, large, calcium rich lakes
M27	Relatively large, shallow, fen lakes
M30	Weakly brackish waters
R4	Permanent, slow flowing, upper reach, sand
R5	Permanent, slow flowing, middle and lower reach, sand
R6	Slow flowing small river, sand-clay
R7	Slow flowing river, side channel, sand or clay
R12	Slow flowing middle and lower reach, bog
R13	Fast flowing upper reach, sand
R14	Fast flowing middle and lower reach, sand
R15	Fast flowing small river, pebble
R17	Fast flowing upper reach, calcium rich
R18	Fast flowing middle and lower reach, calcium rich

Appendix B: Barriers from polder to boezem

Table B1: Overview eel mortality when passing pump stations with a propeller pump (axial water flow). * Underestimation as seemingly undamaged eels did reveal internal damage after dissection which would result in delayed mortality.

	Pump description	Capacity (m3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
axiaalpump	Gesloten schroefpomp	60	0.8	355	Kortenhoef	118	32			Vriese et al., 2010
	Gesloten schroefpomp FFI	81	1	333	FFI	25	0			Vriese, 2009
	Gesloten schroefpomp	1500		50	J.L. Hoogland	77	5	5		Kruitwagen & Klinge, 2010a
	Gesloten schroefpomp	2500	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	3000	variable	64	IJmuiden	251	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	2400			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**	**53.8**	

Table B2 : Overview eel mortality when passing pump stations with a propeller-centrifugal pump (axial-radial water flow).

	Pump description	Capacity (m3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
semi-axiaal pump	Schroefcentrifugaalpomp	170	1.52		Tonnekreek	34	0			Vriese et al., 2010
	Hidrostal		10	890-1200		2300	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	1000	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			
							7.7	**3**	**9.2**	

Table B3 : Overview eel mortality when passing pump stations with a centrifugal pump (radial water flow).

	Pump description	**Capacity (m3/min)**	**Height (m)**	**Rotation (rpm)**	**Name**	**n**	**dead (%)**	**damaged (%)**		**Reference**
radial pump	Centrifugaalpomp	38	3.5	368	Duifpolder	12	0			Vriese et al., 2010
	Centrifugaalpomp	60	5	49	Elektriek-Zuid	?	1.4	1.4		Germonpré et al., 1994
	Centrifugaalpomp	400	0.9	205	Boreel	49	49			Vriese et al., 2010
	Centrifugaalpomp	1080	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			
							11.2	**1.4**	**12.4**	

Table B4 : Overview eel mortality when passing pump stations with an Archimedes screw.

	Pump description	Capacity (m3/min)	Height (m)	Rotation (rpm)	Name	n	dead (%)	damaged (%)		Reference
Archimedes screw	Turbinevijzels				Vijzel Bielefeld	?	0			Spah, 2001
	Buisvijzel FFI	0.6	1	57	FFI (gedwongen blootstelling)	23	0			Vriese, 2009
	Vijzel	30	2.9	39	Sint-Karelsmolen	?	4	10		Germonpré et al., 1994
	Vijzel	35	3.6	37	De Seine, Vlaanderen	?	0	37		Denayer & Belpaire, 1992
	Spaans Babcock	500	2.2	17	Overwaard	43	2			Vriese et al., 2010
	De Wit vijzel	660	0.3	22	Halfweg (natuurlijke doortrek)	24	0			Kruitwagen & Klinge, 2008c
	Buisvijzel (Landustrie Sneek BV)	40	2.7	39.1	Ennemaborgh	101	8			Vis et al., 2013
	Buisvijzel (Landustrie Sneek BV)	23	2.7	23.8	Ennemaborgh	*112*	3			Vis et al., 2013
	Vijzel	335	0.35		Kolhoorn	16	0			Kroon & van Wijk, 2013
	Vijzel	350	1.14		Kadoelen	59	8			VisserijServiceNederland, 2010
	Vijzel			23-31		160	0	0.6		Kibel, 2008
	Vijzel	100		25	Isabella	48	13.5			INBO
	Vijzel	200		21	Isabella	131	14.5			INBO
	Vijzel	90	0.64		Overtoom	*7*				VisserijServiceNederland, 2010
	Vijzel	43	1.25		Bergermeer	*3*				VisserijServiceNederland, 2010
	Vijzel	660	0.3	22	Halfweg (natuurlijke doortrek)	*5*				Kruitwagen & Klinge, 2008c
	Buisvijzel FFI	32			Hoekpolder	*2*				Wanink et al., 2012
	Vijzel				Schalsum	*2*				Koopmans, 2013
	Vijzel	23	0.73		Sudhoeke	*9*				Vriese et al., 2010
					Pooled studies with n <10	28	3.6			
							4.0	**15.9**	**12**	

Appendix C. Barrier assessment list Boezem and National waters

Table C1. Overview of the most important barriers, their characteristics and their estimated mortalities (based on Winter et al. 2013a, 2013b and updated for 2013-2017)

					Potential silver eel	Mortality	Silver eel losses (ton)		Losses per site (%)*	
Water Board	Barrier site	Type Barries**	from	to	arting biomass (ton)	best guess	min	max	min	max
noorderzijlvest	Spijksterpompen	Gema	B	Z	4.67	0.30	1.40	1.40	30	30
noorderzijlvest	Noordpolderzijl	Gema	B	Z	3.11	0.30	0.93	0.93	30	30
noorderzijlvest	Waterwolf Electra	Gema+Keer	B	R	38.93	0.30	3.27	10.74	8	28
noorderzijlvest	De Drie Delfzijlen	Gema+Spui	B	Z	11.68	0.30	0.98	3.22	8	28
noorderzijlvest	Lauwerssluizen	Spui+Sche	B	Z	197.05	0.00	0.00	0.00	0	0
wetterskip	Roptazijl	Gema	B	Z	10.40	0.50	5.20	5.20	50	50
wetterskip	Zwarte Haan	Gema	B	Z	10.40	0.50	5.20	5.20	50	50
wetterskip	Lemmer (Wouda)	Gema + Sche	B	Z	3.47	0.25	0.48	0.82	14	24
wetterskip	Stavoren	Gema + Sche	B	R	69.33	0.06	2.29	3.95	3	6
wetterskip	Ezumazijl	Gema + Sche	B	R	10.40	0.50	4.26	5.10	41	49
wetterskip	Harlingen	Spui+Sche	B	Z	138.65	0.00	0.00	0.00	0	0
wetterskip	Dokkumer Nieuwe Zijlen	Spui+Sche	B	R	103.99	0.00	0.00	0.00	0	0
hunze en aa	Duurswolde	Gema+Spui	B	Z	7.37	0.50	1.03	3.39	14	46
hunze en aa	Termunterzijl	Gema+Spui+Sche	B	Z	6.63	0.30	1.83	1.99	28	30
hunze en aa	Nieuw Statenzijl	Spui+Sche	B	Z	31.69	0.00	0.00	0.00	0	0
hunze en aa	Delfzijl	Spui+Sche	B	Z	28.01	0.00	0.00	0.00	0	0
reest en wieden	Stroink	Gema	B	R	49.79	0.11	5.48	5.48	11	11
reest en wieden	Zenemuden	Gema+Keer+Sche	B	R	16.60	0.50	4.36	4.56	26	28
velt en vecht	Haandrik	WKC+Stuw+Vist	B	R	23.58	0.17	4.01	4.01	17	17
amstel gooi en vecht	De Ruiter	Gema + Sche	B	R	9.08	0.25	1.86	2.23	21	25
amstel gooi en vecht	Mijndense Sluis	Gema + Sche	B	R	7.38	0.10	0.61	0.72	8	10
amstel gooi en vecht	Spiegelpolder	Gema + Sche	B	R	3.41	0.25	0.70	0.83	21	25
HHNK	Kadoelen	Gema	B	R	4.20	0.08	0.34	0.34	8	8
HHNK	De Waker	Gema	B	R	1.40	0.02	0.03	0.03	2	2
HHNK	Leemans	Gema	B	Z	11.20	0.10	1.12	1.12	10	10
HHNK	Lely	Gema	B	Z	4.20	0.25	1.05	1.05	25	25
HHNK	Vier Koggen	Gema	B	R	9.80	0.10	0.98	0.98	10	10
HHNK	Grootslag	Gema	B	R	7.00	0.02	0.14	0.14	2	2
HHNK	Zaangemaal	Gema + Sche	B	R	16.81	0.01	0.05	0.15	0	1
HHNK	Overtoom	Gema + Sche	B	R	0.14	0.04	0.00	0.01	1	4
HHNK	Helsdeur	Gema+Spui+Sche	B	Z	44.81	0.30	3.76	12.37	8	28
HHNK	Schermersluis	Sche	B	R	1.40	0.00	0.28	0.28	20	20
HHNK	Oostoever	Spui	B	Z	16.81	0.00	0.00	0.00	0	0
rijnland	Katwijk	Gema	B	Z	67.27	0.01	0.67	0.67	1	1
rijnland	Halfweg	Gema	B	R	24.11	0.04	0.96	0.96	4	4
rijnland	Gouda	Gema	B	R	12.69	0.10	1.27	1.27	10	10
rijnland	Leeghwater	Gema	B	R	20.31	0.30	6.09	6.09	30	30
rijnland	Spaarndam	Gema + Sche	B	R	22.85	0.01	0.06	0.21	0	1
Delfland	Schoute	Gema	B	Z	3.34	0.30	1.00	1.00	30	30
Delfland	Zaayer	Gema	B	R	6.26	0.02	0.15	0.15	2	2
Delfland	Westland	Gema	B	R	2.30	0.10	0.23	0.23	10	10
Delfland	Schiegemaal	Gema	B	R	1.46	0.10	0.15	0.15	10	10
Delfland	v.d. Burg	Gema	B	Z	1.88	0.30	0.56	0.56	30	30
Delfland	Parksluizen	Gema + Sche	B	R	5.64	0.25	0.39	1.30	7	23
HHSK	Schilthuis	Gema	B	R	21.51	0.30	6.45	6.45	30	30
HHSK	Verdoold	Gema	B	R	15.58	0.11	1.71	1.71	11	11
HHSK	Johan Veurink	Gema	B	R	7.42	0.50	3.71	3.71	50	50
HHSK	Krimperwaard	Gema	B	Z	5.93	0.30	1.78	1.78	30	30
HHSK	Abraham Kroes	Gema + Sche	B	R	23.00	0.30	1.93	6.35	8	28
rivierenland	J.U. Smit	Gema	B	R	10.34	0.04	0.41	0.41	4	4
rivierenland	Altena	Gema	B	R	7.24	0.50	3.62	3.62	50	50
rivierenland	Hollands-Duits	Gema	B	R	7.24	0.25	1.81	1.81	25	25
zuiderzeeland	Vissering	Gema + Sche	B	R	23.45	0.25	3.22	5.57	14	24
zuiderzeeland	Buma	Gema + Sche	B	R	17.93	0.25	2.47	4.26	14	24
zuiderzeeland	Smeenge	Gema + Sche	B	R	12.42	0.50	3.41	5.90	28	48
zuiderzeeland	Wortman	Gema + Sche	B	R	20.69	0.25	2.85	4.91	14	24
zuiderzeeland	De Blocq van Kuffeler	Gema + Sche	B	R	34.49	0.25	4.74	8.19	14	24
zuiderzeeland	Lovink	Gema + Sche	B	R	12.42	0.25	1.71	2.95	14	24
zuiderzeeland	Colijn	Gema + Sche	B	R	16.55	0.12	1.07	1.85	6	11
R	Sluizen-complex IJmuiden	Gema+Spui+Sche	R	Z	443.66	0.02	6.65	12.87	2	3
R	Krammersluizen	Sche	B	Z	0.06	0.00	0.03	0.03	50	50
R	Bergse Diep Sluis	Sche	R	Z	0.00	0.00	0.00	0.00	50	50
R	Terneuzen	Sche	R	Z		0.00	0.00	0.00	0	0
R	Volkeraksluizen	Sche	R	Z	0.06	0.00	0.03	0.03	50	50
R	Bathse spuisluis	Spui	R	Z	9.74	0.00	0.00	0.00	0	0
R	Krabbersgat-sluizen	Spui+Sche	R	Z		0.00	0.00	0.00	0	0
R	Houtrib-sluizen	Spui+Sche	R	Z		0.00	0.00	0.00	0	0
R	Haringvliet-sluizen	Spui+Sche	R	Z	229.12	0.00	0.00	0.00	0	0
R	Aflsuitdijk Kornwerderzand	Spui+Sche	R	Z	364.24	0.00	0.00	0.00	0	0
R	Afsluitdijk Den Oever	Spui+Sche	R	Z	364.24	0.00	0.00	0.00	0	0
R	Oranjesluizen	Spui+Sche+Vist	R	Z	214.83	0.00	0.00	0.00	0	0
R	Nieuwe Waterweg		R	Z	476.61	0.00	0.00	0.00	0	0

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

** Gemaa=Pumping station; Sche=ship lock; Spui=Discharge sluice; Vist=fishway; Stuw=Weir; Keer=Protection sluice

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

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