Evaluation of the Dutch Eel Management Plan 2009-2011

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Summary

Evaluatie van het Nederlandse aalbeheerplan: maatregelen hebben geleid tot een substantiële verbetering van de overleving; positieve effecten op de aalpopulatie kunnen pas na vele jaren zichtbaar worden en blijven onzeker, omdat de aal pas na vele jaren terugzwemt naar zee om zich voort te planten en omdat niet goed bekend is welke de oorzaken zijn van de achteruitgang in de aalpopulatie.

Het gaat slecht met de aal in Europa
De aalpopulatie en aalvangsten zijn sterk teruggelopen: De huidige intrek van glasaal is slechts 1-5% van de intrek in de 60-70-er jaren. Deze situatie is zeer zorgwekkend en wordt door de aalwerkgroep van de International Council for the Exploration of the Sea (ICES) als volgt omschreven: "Indications are that the eel stock remains at an historical minimum, continues to de-cline and is outside safe biological limits. Recruitment of both glass eel and young yellow eel continues to decline and shows no sign of recovery. Current levels of anthropogenic mortality, thought to be high on juvenile (glass eel) and older eel (yellow and silver eel), are not sustainable and there is an urgent need to reduce these until there is clear evidence that the stock is increasing." (http://www.ices.dk/workinggroups/ViewWorkingGroup.aspx?ID=75)

Maatregelen voor aalherstel
Om herstel van de aalpopulatie mogelijk te maken heeft De Raad van de Europese Unie in 2007 de "EU Regulation for the Recovery of the Eel Stock (EC 1100/2007)" vastgesteld. Deze verordening verplicht de lidstaten om met een eigen nationaal aalbeheerplan te komen en te implementeren. Het doel van deze aalbeheerplannen is daarbij als volgt omschreven: "Doel van de beheersplannen 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 beheersplannen voor aal worden opgesteld met het oog op het bereiken van die doelstelling op lange termijn."
Lidstaten zijn verplicht om over de voortgang van de nationale aalbeheerplannen te rapporteren aan de Europese Commissie, voor 1 juli 2012. De Europese Commissie zal met deze informatie een verslag opstellen over deze aalbeheerplannen. Dit verslag zal uiterlijk 31 december 2013 door de Europese Commissie worden ingediend bij het Europees Parlement en Raad. Tegen deze achtergrond heeft Nederland een eigen aalbeheerplan opgesteld en geïmplementeerd in juli 2009. 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.

Aalbeheerplan Nederland

Het aalbeheerplan van Nederland omvat de volgende maatregelen:

No	Maatregel	Planperiode	Realisatie
1	Verminderen aalsterfte bij gemalen en andere waterwerken; van de 1800 belangrijkste migratie barrières zullen 900 worden opgelost voor 2015, de andere 900 voor 2027	2015-2027	Uitgesteld vanwege bezuinigingen bij de overheid
2	Verminderen aalsterfte bij waterkrachtcentrales met minstens 35%	2009	november 2011*
3	Instellen van visserij vrije zones in gebieden die van belang zijn voor de trek van aal	2010	1 april 2011
4	Sportvissers zetten gevangen aal levend terug (kustzone en binnenwater)	2009	1 oktober 2009
5	Verbod op recreatieve visserij met beroepstuigen in kustwateren	2011	1 januari 2011
6	Gesloten seizoen: van 1 september tot 1 december	2009	1 oktober 2009
7	Stopzetten van peurvergunningen door de overheid in staatswateren	2009	1 mei 2009
8	Uitzetten van glasaal en kleine aal uit aquacultuur	2009	begin 2010
9	Onderzoek naar kunstmatige voortplanting van aal	lopend	EU-project

* Om technische redenen blijkt een effect van 24% maximaal mogelijk.

Verder zijn per 1 april 2011 grote gebieden gesloten (vooral de grote rivieren) voor de aalvisserij omdat de aldaar gevangen aal niet voldeed aan eisen rond voedselveiligheid door te hoge gehalten aan PCB's en dioxines. Deze maatregel was geen onderdeel van het oorspronkelijke aalbeheerplan, maar is later toegevoegd.

Evaluatie van het aalbeheerplan Nederland

Het aalbeheerplan is geëvalueerd in het licht van de voornoemde "beheersdoelen" uit de Aalverordening.
De methodiek die bij deze evaluatie daarbij is gehanteerd komt voort uit de ICES - "werkgroep aal" (http://www.ices.dk/workinggroups/ViewWorkingGroup.aspx?ID=423), maar is niet beoordeeld door de Advisory Committee van ICES (ICES-ACOM, met vertegenwoordigers van alle 20 ICES landen) die verantwoordelijk is voor alle formele "ICES" adviezen.
Dit betekent dat in deze evaluatie alleen wordt ingegaan op de effectiviteit van maatregelen in relatie tot beheersdoelen opgesteld door de Raad van de Europese Unie. In hoeverre deze beheersdoelen ook in lijn zijn met het voorzorgprincipe of duurzaam beheer volgens ICES-ACOM is niet aan de orde.
De evaluatie is uitgevoerd door middel van modellen, vangstgegevens, veldwaarnemingen en statistische analyses, uitvoerig beschreven in de rapportage. Het geheel van deze inspanning resulteerde in schattingen voor (2008) en na (2011) de implementatie van het Aalbeheerplan van, met name:

- De biomassa uittrekkende schieraal: 440 t in 2008, 480 t in 2011.
- De pristine biomassa aan uittrekkende schieraal: 10.400 t.
- De doelstelling Aalverordening voor Nederland: 4160 ton (40% van de pristine biomassa).
- De uittrek van schieraal tov deze doelstelling: 11% in 2008, 12% in 2011.
- De reductie in antropogene sterfte door de genomen maatregelen: de antropogene sterfte van glasaal naar schieraal is afgenomen van 85% in 2008 naar 67% in 2011.

Deze schattingen zijn ruw, en de daarmee gepaard gaande onzekerheid is in de rapportage omschreven.

Effecten van het Nederlandse aalbeheerplan op de Nederlandse aalpopulatie

De evaluatie laat zien dat de maatregelen uit het Nederlandse beheerplan aal hebben geleid tot een substantiële teruggang in sterfte door menselijk handelen. Deze reductie is voornamelijk het gevolg van beperkingen van de visserij (recreatief en beroep).

De aalpopulatie in Nederland en de uittrek van schieraal zullen pas veel later substantieel verbeteren. De reden dat dit zo lang duurt is dat aal een langlevende soort is. Het duurt meer dan een jaar voordat glasaal na geboorte aankomt voor de Nederlandse kust en de binnenwateren op zwemt. Vervolgens duurt het 5-15 jaar voordat deze aal "schier" wordt, en als schieraal terugtrekt naar zee.

Het blijft 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

1.1 Required stock indicators for evaluation of the Dutch Eel Management Plan

The EU Regulation for the Recovery of the Eel Stock (EC 1100/2007) was adopted in 2007. It required Member States to set up an eel management plan by the end of 2008 (article 2). In The Netherlands the eel management plan (EMP) was implemented in July 2009. An overview of the (un)planned measures is given in Table 1.1.

Table 1.1. Overview of all the (un)foreseen measures described in the Dutch Eel Management Plan to be implement to reach the 40% escapement objective.

No	Foreseen Measure	Planned implementation	Realised implementation
1	Reduction of eel mortality at pumping stations and other water works; of the 1800 most important migration barriers 900 will be solved by 2015 and the remaining 900 by 2027	2015-2027	Due to financial constrains at the Ministry of I&M the scheduled refurbishments have been postponed
2	Reduction of eel mortality at hydro-electric stations with at least 35%	2009	November 2011***
3	The establishment of fishery-free zones in areas that are important for eel migration	2010	1 April 2011**
4	Release of eel caught (a) at sea and (b) at inland waters by anglers	2009	1 October 2009
5	Ban on recreational fishery in coastal areas using professional gear	2011	1 January 2011*
6	Annual closed season from 1 September to 1 December	2009	1 October 2009
7	Stop the issue of licences for eel snigglers by the minister of EL&I I state owned waters	2009	1 May 2009
8	Restocking of glass eel and pre-grown eel from aquaculture	2009	Early 2010
9	Research into the artificial propagation of eel	ongoing	EU-project started
No	**Unforeseen Measure**		
10	Closure eel fishery in contaminated (PCBs, dioxins) areas		1 April 2011

* The use of fykes and long-lines by recreational fishers has been banned in nearly all marine and inland waters waters. The use of gillnets, however, by recreational fishers is still allowed in a few marine waters.
** The vast 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.
*** Due to technical difficulties the maximum achievable reduction in mortality by adjusted turbine management is 24%.

The member states have to report on the progress of the national EMPs to the European Commission (EC) by 30 June 2012 (article 9.1).The EC will present a report with a statistical and scientific evaluation of the outcome of the implementation of the Eel Management Plans to the European Parliament and Council before 31 December 2013.
It was proposed by the ICES WGEEL to include the stock indicators in Table 2 in the progress reports submitted to the Commission by Member States. These indicators needed to be submitted by the 30th of June 2012, in line with Article 9 of the eel regulation (1100/2007). The purpose of the indicators is to render the reports more efficient in demonstrating the progress achieved via the implementation of the

eel management plans. In particular, there needs to be a clear indication as to the achievement of the 40% escapement target is achieved.

Table 1.2. Overview of the additional indicators to be reported on by member states to the EC by the 30th June 2012.

B_0	The amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the stock. B_0 is in the Regulation, as a denominator for the 40%, and in Art 2.5.
$B_{current}$	The amount of silver eel biomass that currently escapes to the sea to spawn. $B_{current}$ is in the Regulation, as the nominator of the proportion of silver eel biomass actually escaping, in Art 9.1.a.
B_{best}	The amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the current stock. B_{best} is not in the Regulation. It could be calculated from $B_{current}$, ΣF and ΣH. In line with the ICES framework this would allow for a cross-check in the interpretation of the quantities above.
ΣF	The fishing mortality rate, summed over the age-groups in the stock, and the reduction effected.
ΣH	The mortality rate outside the fishery, summed over the age-groups in the stock, and the reduction effected. ΣH is in the Regulation, in Art 9.1.c ("level of mortality factors"). ICES considers that ΣA = ΣF+ΣH.
R	The amount of glass eel used for restocking within the country. R is in the regulation, in Art 9.1.d, ("the amount of eel less than 12 cm in length caught and the proportions of this utilised for different purposes"). R is not relevant for The Netherlands as no eel smaller than 12 cm is landed, minimum size in The Netherlands is 28 cm.

The Ministry of Economic Affairs, Agriculture and Innovation (EL&I) has requested IMARES 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 using the modified ICES precautionary diagram.

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

Estimates of both absolute biomass of silver eel escapement and mortality rates are requested by the EC, as listed in Table 1.2. However, for the current evaluation of the EMP, the most important of the required estimates are the anthropogenic mortality rates which, when summed over the age-groups in the stock (ΣA= ΣF+ ΣH), provide an estimate of the **Lifetime Anthropogenic Mortality, or LAM**. The LAM only refers to the continental part of the life cycle of European eels (see Figure 1.1). Management actions were taken to reduce anthropogenic mortalities immediately, and estimated reductions ('before' compared to 'after' the implementation of the EMP) in LAM can therefore be used to evaluate the success of these management actions. Instead, the impact of management actions on the biomass of silver eel escapement also depends on trends in recruitment. Some measures, in particular reduction of yellow eel mortalities and glass eel stocking, will take at least several years to materialise. Estimates of LAM can be used to put current silver eel escapement ($B_{current}$; Table 1.2) into context, by comparing it with the best possible spawner escapement under recent recruitment conditions (B_{best}). The estimated proportion $B_{current}/B_{best}$, referred to as %SPR (current spawner-per-recruit as a percentage of the best possible spawner-to-recruit ratio; ICES 2011a), can be compared with the 40% escapement target of the EU eel regulation. Estimated improvements ('before' compared to 'after') in %SPR are therefore an important means by which the EMP can be evaluated. A certain estimated level of LAM will result in a corresponding %SPR, and these parameters can therefore be used interchangeably. A particular problem in this context is that cohorts that are currently in the stock will all have different lifetime anthropogenic mortality rates, since these rates will have changed over time. However, ICES (2011a) indicated that estimates of either

ΣA or %SPR usually 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 2011a).

To estimate LAM, we consider anthropogenic mortalities during the two main continental life stages of eels (Figure 1.1):
1) fishing mortalities that occur during the yellow eel stage, and;
2) fishing and barrier mortalities that occur during migration as a silver eel.

The reason for considering these two types of mortalities separately is that yellow eel mortalities apply over a sequence of years from transformation as glass eel to yellow eel up until the point of transformation from yellow eel to silver eel. Instead, silver eel mortalities are assumed to apply during a single year in the life cycle of an eel.

To estimate the %SPR for a given level of anthropogenic yellow eel mortality, a population model for yellow eels is introduced in this report (Chapter 2) in which the development of glass eel to silver eels is modelled as a function of growth, maturation and natural and anthropogenic mortality. Estimated silver eel mortality rates are assumed to apply only once in the life cycle of eels and result in a reduction of the %SPR as estimated as a function of the current yellow eel anthropogenic mortality.

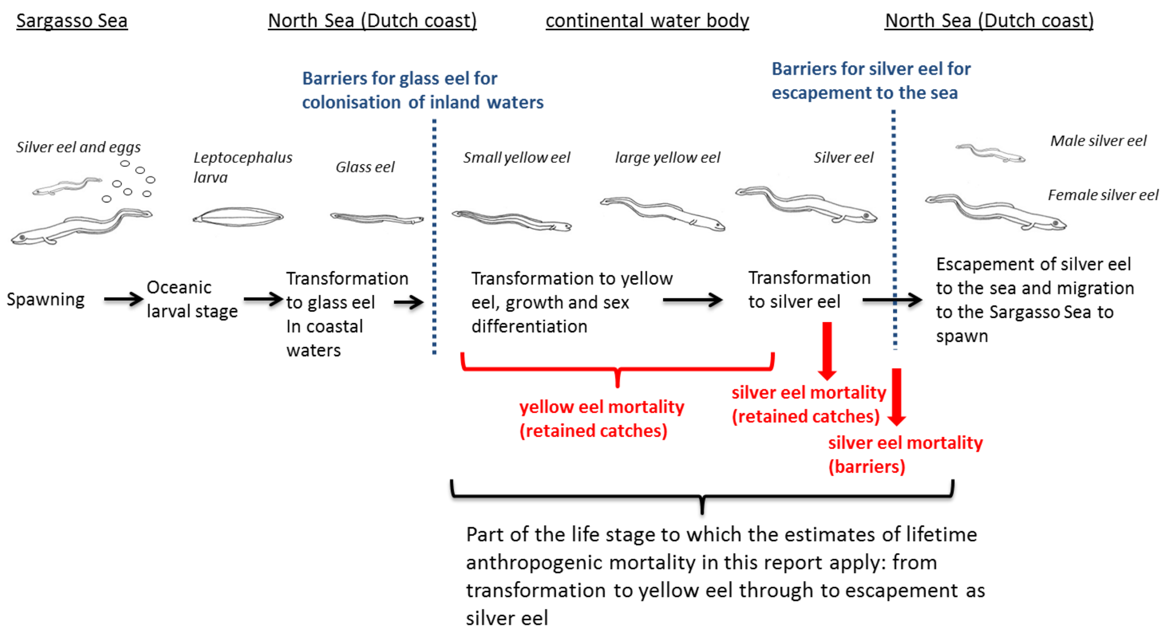

Figure 1.1. The life cycle of a European eel, with the part of the life cycle to which the Dutch Eel Management Plan (EMP) applies and for which 'Lifetime Anthropogenic Mortality' (LAM) rate estimates are made in this report.

<u>Yellow eel mortality rates</u> were estimated as the proportion of the estimated retained yellow eel catches (by both commercial and recreational fisheries) out of the estimated total current standing stock of yellow eels with body lengths above 30 cm. Thus, both types of 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 data from biological market sampling of these catches. 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. The harvested proportion of the yellow eel stock was estimated as the retained catch divided by the estimated standing stock plus the retained catch. Given the estimated current harvested proportion (which can be transformed into an estimate of fishing mortality rate), a %SPR was estimated using the yellow eel population model. This estimate represents the %SPR that a cohort of eels would be expected to produce if they would be exposed throughout their lifetime (from glass eel to silver eel) to the estimated current fishing mortality rate.

Silver eel mortality rates were estimated as the proportion of the retained silver eel catches by commercial fisheries out of the estimated total current production of silver eels, and an additional mortality rate by barriers (pumping stations, hydropower plants, sluices, etc) during silver eel migration. Silver eel retained catches were estimated by splitting the total reported retained catches by commercial fisheries into yellow eel and silver eel, using data from biological market sampling of these catches. The mortality rates induced by migration barriers were estimated using a simple model in which the starting positions of silver eels were split into three hierarchies of water bodies: 1) 1st hierarchy: silver eels that start from 'polder' ('drainage ditches' below sea level) water bodies; 2) 2nd hierarchy: silver eels that start from larger inland regionally managed water bodies with no open connection to the sea (referred to as 'boezem'), and; 3) 3rd hierarchy: silver eels that start from large nationally managed water bodies such as the major lakes and main rivers. Mortality rates of barriers were estimated for each hierarchy. Mortality rates induced by barriers in polder water bodies were estimated by means of a meta-analysis of results 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 from an analysis of the top 34 prioritised migration barriers for eel (Buijse et al., 2009), and a number of telemetry experiments or detailed studies into mortalities induced by these barriers. Barriers were assumed never to be in sequence within a hierarchy, but a proportion of silver eels 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 silver eels that were estimated to be produced in water bodies upstream of the main hydropower plants in the main rivers, for which separate mortality rates were estimated.

As explained above, estimates of the standing stock biomass of silver and yellow eel are necessary to 1) For both silver eel and yellow eel: estimate fishing mortality from retained catches and 2) for silver eel only: split the starting positions of silver eels into the three hierarchies of water bodies ("polder", "boezem" and nationally managed) as explained above. To estimate the standing stock biomass, a spatially explicit approach was taken to unambiguously define the delineations and wetted areas of water bodies included in the assessment. Estimates of absolute stock size were made in two different ways:

1) Static stock survey: 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) For the two main lakes (IJsselmeer and Markermeer) direct estimates of fishing mortality rates were made by fitting the yellow eel population model to a long term stock survey data set (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 these lakes, by multiplying the reported retained yellow eel catches with the inverse of the estimated harvested proportion. The estimates of fishing mortalities were also used for lakes Veluwerandmeren and Grevelingenmeer, since no electrotrawl or electric dipping net data were available for these lakes.

The silver eel production per water body was estimated by splitting the total standing stock into length classes (using the electric dipping net data) and multiplying this with estimates of the proportion of silver eels out of all eels per length class (a 'maturity ogive'). The maturity ogive was estimated using biological market sampling data. For the lakes for which no electric dipping net data were available (and estimates were based on direct estimates of fishing mortalities), an estimate of 15% silver eel production out of the total standing stock biomass was used (the mean of the proportions of silver eel in the other water bodies).

The main data sets which are used for the stock assessments are:

1) **Reported landings** by commercial fishers (by Fish Stock board; 'Visstandbeheerscommissie'(VBC)); these landings are provided by the Ministry of EL&I and are stored at IMARES in the Visstat database of fisheries statistics.
2) **Biological data from market sampling of landings** by commercial fishers. During visits to fishers a representative sample of (usually) 150-200 eels have been selected from the landings, and lengths of individual eels have been measured in order to estimate length-frequency distributions of the landings. Furthermore, a number of eels have been taken from each sample for dissection and the estimation of maturity-at-length, weight-at-length and sex-ratio-at-length. Sex-specific growth curves have been estimated from age readings of 200 eel otoliths.
3) **Surveys** of fish stocks in the **regionally managed water bodies**. Eel sampling within the WFD waters was done following an EU certified protocol. In the assessments presented here, we used only data from electrofishing with electric dipping nets. These data could be used to estimate densities of standing stock on a wetted area basis most straightforwardly. Sampled water bodies are representative for water types defined within The Netherlands based on WFD regulation. Data collection is managed by regional water boards. Electric dipping net data were obtained from two companies; ATKB and Grontmij. A total of 2325 samples by electric dipping net were available between 2006 and 2011, 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** of fish stocks in the **nationally managed water bodies**. The shores of the main rivers (Meuse, Rhine and their downstream counterparts) are 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 WFD water bodies, and were added separately to the spatial model.

1.3 Flow diagram of the stock assessment methodology and structure of this report

As explained in paragraph 1.2, the stock assessment methodology consists of a number of steps before the final overall assessment of %SPR can be made for the whole of The Netherlands. The steps leading up to the final overall assessment are:

1) The development of a **population model for yellow eel** (<u>**Chapter 2**</u>), for the estimation of the best possible spawner production per glass eel, and the reduction in spawner production for a certain level of yellow eel mortality (%SPR).
2) **Estimation of standing stock biomass** using data from electric dipping nets (<u>**Chapter 4**</u>) or by fitting the population model to stock surveys for lakes IJselmeer and Markermeer (<u>**Chapter 3**</u>). <u>Chapter 4</u> 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 in a number of scenarios as a proportion of the density of eel "within-shore" (<1.5 meters from the shore/bank). Estimates of the standing stock biomass of yellow eel and silver eel, which are necessary for the estimation of mortality rates (see paragraph 1.2), are given in <u>paragraph 4.8</u>.
3) Estimation of mortality in the silver eel stage due to barriers (<u>Chapter 6</u>).

A separate chapter is dedicated to comparing the estimates of silver eel production with estimates based on capture-mark-recapture experiments (Chapter 5).

The results from Chapters 2, 3, 4 and 6 are subsequently used for the estimation of the key stock indicators for the whole of The Netherlands (Chapter 7).

The evaluation of the Dutch EMP is presented in Chapter 8, where the stock indicators are presented in the modified precautionary diagram as developed by ICES. The report concludes with a general discussion and recommendations for improvements to the stock assessment methodology (Chapter 9).

The flow diagram below (Figure 1.2) gives a broad overview of the key step in the stock assessment methodology, with reference to the chapters and key paragraphs therein.

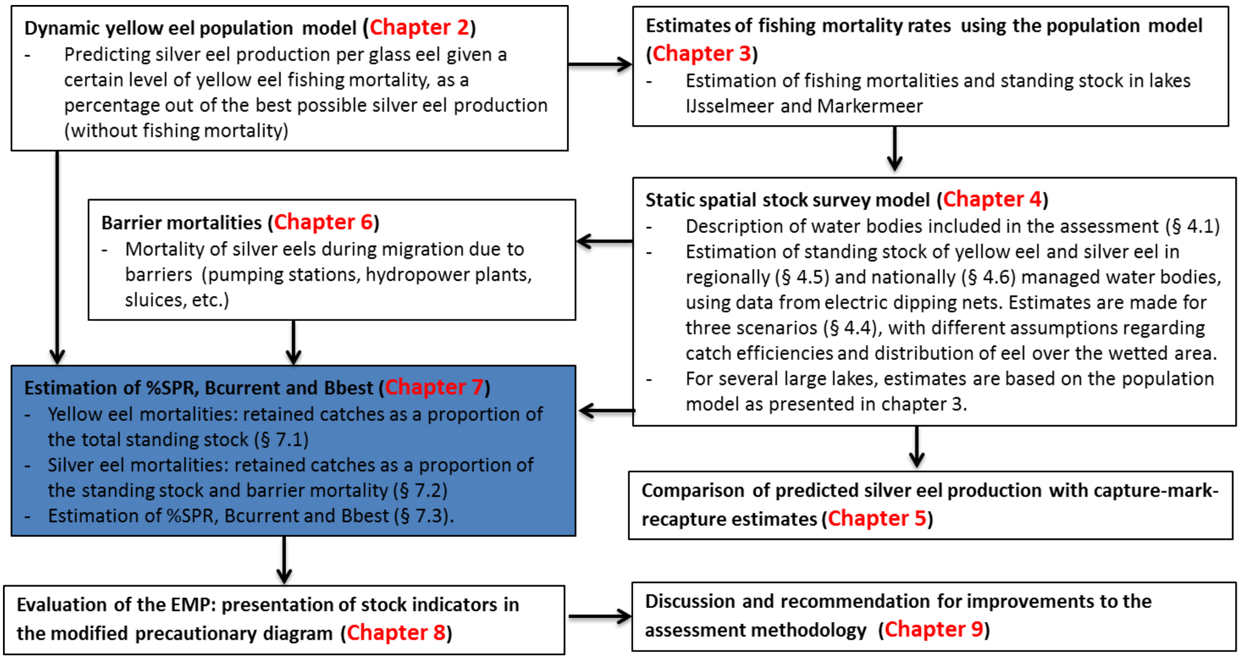

Figure 1.2. A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report (with reference to chapters and key paragraphs therein).

2. A dynamic population model for yellow eel for estimating %SPR

2.1. Introduction

A length based population model for yellow eels is introduced in which the processes of recruitment, growth, natural and fisheries induced mortality, and maturation are used to predict the spawner-to-recruit ratio, given a certain level of yellow eel fishing mortality.

The model is used in Chapter 7 in the overall assessment to compute the reduction in the spawner-to-recruit ratio compared to the best possible spawner-to-recruit ratio in the absence of yellow eel anthropogenic mortality (%SPR). The %SPR is expressed as the percentage silver eel biomass production per glass eel out of the best possible silver eel production, given a certain level of anthropogenic yellow eel mortality (see paragraph 1.2 and Figure 1.1). In Chapter 3, the yellow eel population model is extended so that it can be used to infer fishing mortality rates using stock survey and market sampling data (Figure 2.1)

In this chapter, the basic model is described and estimates of vital parameters (growth rates, maturation-at-length, sex-ratio, natural mortality) are given.

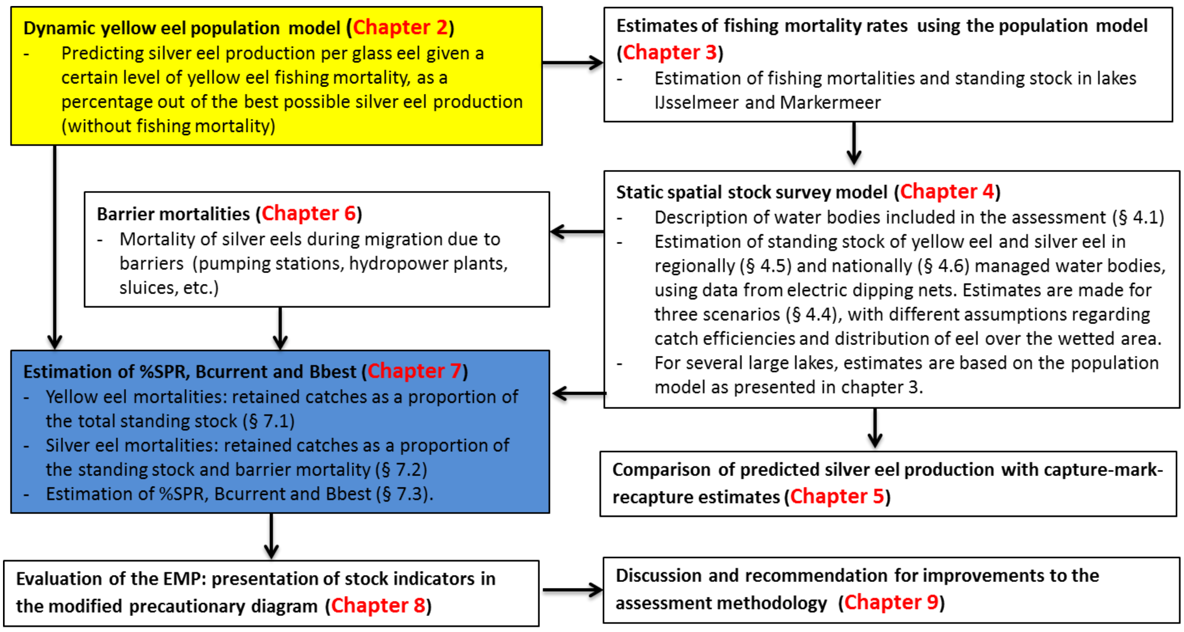

Figure 2.1. A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report (with reference to chapters and key paragraphs therein).

2.2 The population model

The population model describes the fates of cohorts from immigration as glass eels to death by natural causes, capture by fisheries or transformation to silver eel. In global terms, the model can be characterised by:

1) Yearly arrival of glass eel immigrants and transformation to small yellow eels.
2) The yellow eel phase: growth, natural mortality, fishing mortality, and maturation (and subsequent emigration).

The population model works by forward projection of cohorts of yellow eels. A separate growth curve is used for male and female yellow eels. The model tracks the number of eels per sex in discrete length intervals of 5 cm. The progression of the population is modelled as annual transitions between length classes using transition matrices for each sex, according to their growth curve. An annual probability of transformation to the silver eel stage and subsequent emigration ("maturation") is modelled as a function of total body length and sex (Davey & Jellyman, 2005; de Leo & Gatto 1995; Bevacqua & de Leo 2006, Dekker 2000). Both natural and annual fishing mortality (assumed to take place instantaneously) are considered. Natural mortality is assumed to be a constant for yellow eels of all lengths, whereas fishing mortality is modelled as a function of length. The fishing mortality is assumed zero below the legal minimum landing size (28 cm).

Let $N_{a,j,t}$ be the numbers of eels of age *a* (years since transformation to yellow eel), of sex *j* ($j \in \{male, female\}$) in calendar year t. Glass eels are assumed to have age *a*=0 and a length of 7.5 centimetres. For yellow eels (1 year old or more), let *L(a,j)* be the length class (5 cm intervals; starting from 10 cm) that a yellow eels of sex *j* and age *a* falls in: $j \in \{10-15, 15-20, 20-25, \ldots, 100-125\}$. The length class that a yellow reaches at age *a* is a function of the growth curve of sex *j*. The population model is given by:

$$N_{a,j,t} = N_{a-1,j,t-1}\, e^{\left(-M-Fs_{L(a,j)}\right)} \left(1-q_{j,L(a,j)}\right) \qquad \textbf{Eqn 2.1}$$

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*.

The model is used for the prediction of the spawner-to-recruit ratio for a user-specified fishing mortality rate (parameter F in the model) and sex-ratio of a cohort of glass eel. To do this, estimates of the following vital parameters are needed, which are presented in paragraph 2.3:

1. sex-specific growth rates
2. sex-specific maturation-at-length
3. natural mortality
4. selectivity-at-length of the fisheries
5. a length-weight relationship

Table 2.1. Variables and model parameters.

Parameter	**Description**	**Unit**
$N_{a,j,t}$	Estimated population size of glass eels (*a*=0) and yellow eels ($a \geq 1$), of sex *j* in year *t*	Number of eels
j	Indicator for sex ($j \in \{male, female\}$)	
$L(a,j)$	Indicator for length classes for yellow eels $L \in \{10-15, 15-20, 20-25, \ldots, 100-125\}$ as a function of the age and sex	
t	Indicator for calendar year	Years
M	Parameter for natural mortality	Fraction year^{-1}
F	Parameter for Fishing mortality	Fraction year^{-1}
$s_{L(a,j)}$	Selectivity of the fishery for eels of length class *L*	Fraction year^{-1}
$q_{j,L(a,j)}$	Annual probability of maturation for eels of length class *L*	Fraction year^{-1}

2.3 Parameter estimates of vital rates

2.3.1 Maturation-at-length

Maturation-at-length in the population model (Eqn 2.1) is defined as an annual transition rate: the proportion of eels of sex *j* and length class *L* that transform from yellow eel to silver eel.

Eels leave the population after maturation and migrate to the spawning grounds. This means that maturation and mortality in the population model have similar net results on the standing stock. Thus, good estimates of maturation-at-length or maturation-at-age are necessary. Biased estimates of maturation-at-length will inevitably lead to biased estimates of mortalities. This is especially true for males, because the fisheries become selective for individuals with body lengths from 30 cm. This approximately coincides with the lengths above which males start to mature and leave the population.

Obtaining unbiased estimates of maturation-at-length is difficult because the catchabilities of mature eels (silver eels) may be different from immature eels (yellow eels). Silver eels migrate and thus have different behaviour from yellow eels. Migration may take place throughout the year, but is thought to take place mostly in the autumn. The timing of peak migration varies among years and locations and may be related to peaks in water discharge levels (Winter 2011, Winter & Bierman 2010). The differences in catchabilities between yellow eels and silver eels, and the timing of migration in the autumn, have important consequences for the interpretation of biological data on maturation-at-length from market sampling. Retained catches were sampled in recent years mostly during May - August because the fisheries were closed outside this timeframe. The months with peaks in migration are therefore underrepresented in the market sampling data. This may lead to an underrepresentation of silver eels in the catches compared to the true proportions of yellow to silver eel in the underlying population. On the other hand, silver eels may be over-represented if they have higher catchabilities due to their migratory behaviour. At a particular downstream catch location, silver eels may arrive from a large catchment area upstream, leading to high proportions of silver eels in the catches. These high proportions of silver eels cannot be assumed to reflect the proportion of local yellow eels that transform to silver eels. These factors make the interpretation of market sampling data on maturity-at-length difficult.

The maturity-at-length by sex as used in the yellow eel model has been determined as the simple proportion of silver eels out of the total number of eels as found in the biological market sampling data from retained catches of fishers in 2010 (Figure 2.2 and Table 2.2; de Graaf et al., 2011). The observed proportions of silver eels were modelled as a function of length using a logistic model (de Graaf et al., 2011), and the proportions as predicted by this logistic model are taken as annual transition rates of yellow eels to silver eel, given the length class of the yellow eel. For the reasons discussed above, these estimated maturation-at-length curves may be biased.

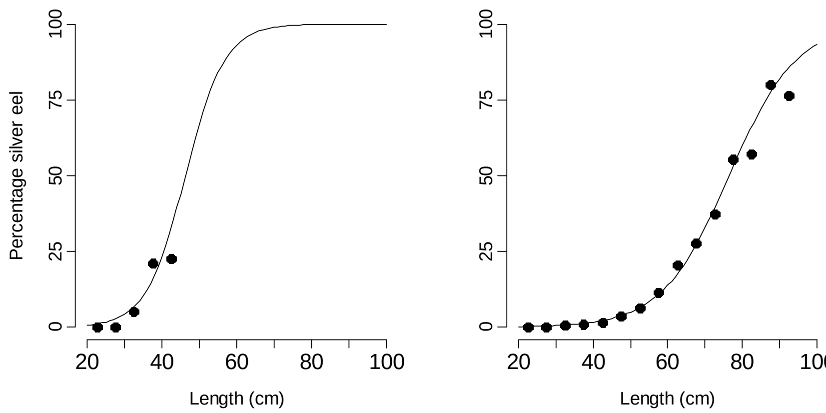

Figure 2.2. Observed (filled circles) and predicted (logistic model) proportions of silver eel out of the total number of eels in the retained catches of fishers, per length class (5 cm intervals). Left panel: males, right panel: females. The proportions as predicted by the logistic model are taken as annual transition rates of yellow eels to silver eel, given the length class of the yellow eel. The proportion silver eel out of the total number of eels per length class in the catches may give biased estimates of these annual transition rates (see text).

Table 2.2. Estimated annual transition rates from yellow eel to silver eel by length class (5 cm intervals). The transition rates are estimated using a logistic model which has been fitted to observed proportions of silver eels in retained catches of commercial fishers (see Figure 2.2).

Sex	**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 90**	**>90**
Males	0	0	0	0.03	0.07	0.16	0.33	-	-	-	-	-	-	-	-	-	
Females	0	0	0	0	0	0	0	0	0.07	0.11	0.17	0.27	0.39	0.53	0.66	0.77	0.9

2.3.2 Growth rates

From the biological market sampling data in 2010, otoliths were collected from a total of 200 eels from several areas in The Netherlands for age reading. 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. The length and weight at age of male and female yellow and silver eels are given in Figure 2.3.

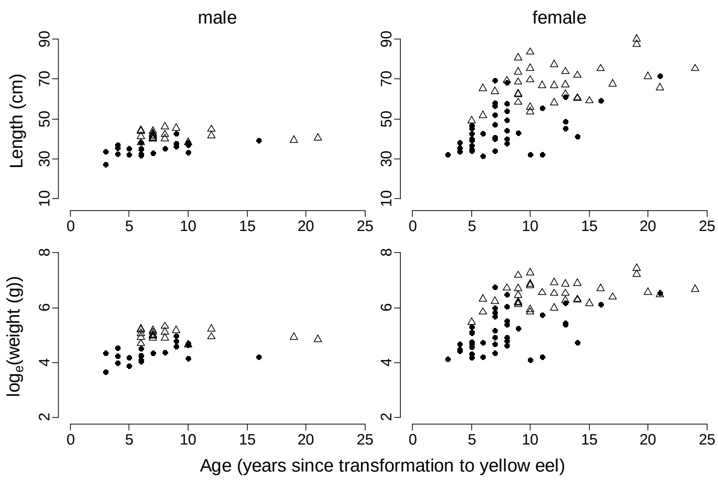

Figure 2.3. Lengths and weights at age of eels selected for age determination in the biological market sampling programme in 2010. Triangles represent silver eels, and filled dots represent yellow eels.

Individual growth curves of the male and female eels are given in Figures 2.4 and 2.5 respectively. The growth curves used in the yellow eel model were estimated using these reconstructed individual growth curves. For females, the reconstructed growth curve is estimated as the average growth per age. For males, the averages of growth rates in the first 5 years since transformation to yellow eel were estimated using only the eels in the sample with ages up to 6 years old. Growth rates of ages 6 and above were estimated using only eels in the sample that were older than 6 years. This is illustrated in Figure 2.5. This was thought to provide a better estimated average growth curve, because males that grow more slowly also become older (mature at older ages).

The average growth curves may be biased for several reasons. The data are from retained catches from commercial fishers (mostly caught using fyke nets), which becomes fully selective for eels at approximately 30 to 35 cm. Thus, slow growing eels will start to appear in the fisheries at older ages than fast growing eels, and this may lead to an underestimation of average growth of the younger eels. On the other hand, slow growing eels may mature at later ages and as such have a longer residence time than fast growing eels, which may mean that they are over-represented in the sample relative to their presence in the population. However, slow-growing eels are also more affected by fishing mortalities before they reach a given length, which may lead to under-representation. Finally, age-reading of otoliths in eels is difficult and uncertain, especially for older eels. Also, variation between individuals in growth curves appears to be large, and using average growth curves in the model may lead to different conclusions as when variability between individuals is allowed for.

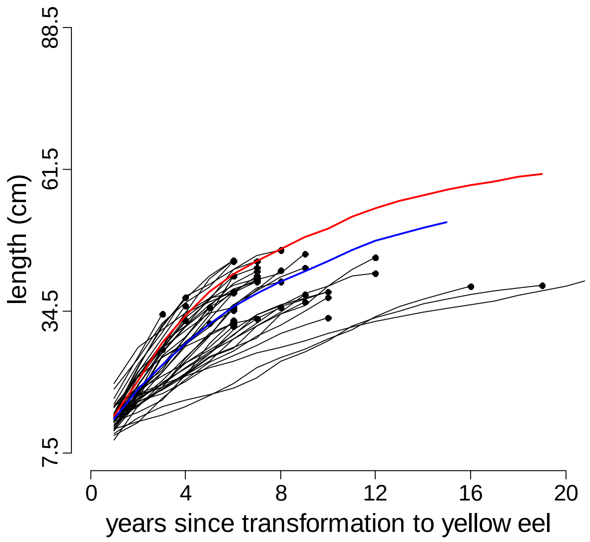

Figure 2.4. *Growth curves of male individuals, estimated by allocating the length of the eel (minus 7.5 cm) to ages in proportion to the relative measured widths of the year-rings in the otoliths. The growth curve given by the red line gives the growth curve resulting from adding the average growth per age of all eels. The growth curve given by the blue line gives the growth curve resulting from adding the average growth for ages 0-5 using eels of ages up to 6 years only, and average growth of ages 6 and above estimated using eels of 6 years and older.*

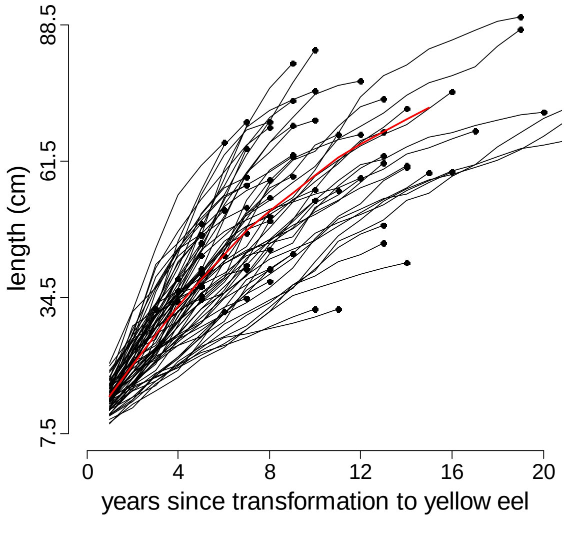

Figure 2.5. *Growth curves of female individuals, estimated by allocating the length of the eel (minus 7.5 cm) to ages in proportion to the relative measured widths of the year-rings in the otoliths. Red line: estimated cumulative growth curve using all eels of all ages.*

The estimated growth curves can be expressed as annual transition rates between length classes of 5 cm intervals, as used in the eel population model. The table of ages which eels spend in the sequential length classes is given in Table 2.5, along with the same table obtained when a constant growth of 3.5 cm per year is assumed such as in Dekker (2000) and van der Meer (2010). The length classes that males and females are predicted to reach for a given age are given in Table 2.3.

Table 2.3. Ages per length class. The length classes $L(a,j)$ as a function of age a and sex j. Indicated per length class are the ages of the eels in which their lengths are predicted to fall within this length class.

Length class *L (a,j)*	**Growth curves as in Figures 2.3, 2.4**		**Growth curve assuming 3.5 cm year^{-1}**	
	Males	**Females**	**Males**	**Females**
	average	**average**	**average**	**average**
Glass eel (7.5 cm)	0	0	0	0
10-15	1	-	1-2	1-2
15-20	2	1	3	3
20-25	3	2	4	4
25-30	4	3	5.6	5-.6
30-35	5-6	4	7	7
35-40	7	5	8-9	8-9
40-45	8-25	6	10-25	10
45-50	-	7	-	11-12
50-55	-	8-9	-	13
55-60	-	10	-	14
60-65	-	11-12	-	15.16
65-70	-	13-14	-	17
70-75	-	15-17	-	18-19
75-80	-	18-19	-	20
80-85	-	20-22	-	21-22
85-90	-	23.24	-	23
90-95	-	25	-	24-25
95-100	-	-		

2.3.3 Length-weight relationship

The following length-weight relationship is used to estimate eel biomass given numbers-at-length (de Graaf et al., 2011; van Keeken et al., 2011):
Weight = exp(-14.45 + 3.217*$\log_e$(length)),
With weight in grams and length in millimeters.

2.3.4 Selectivity of the fisheries

In order to interpret length-frequency distributions of retained catches, and to predict the impact of fisheries mortality on the spawner to recruit ratio, 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 28 cm are returned, and that there is no mortality associated with catch-release. Furthermore, we assume that the fisheries is fully selective at lengths above 30 cm. This assumption could be false for locations where the actual minimum landing size is above 30 cm. However, the proportion of fishers that use a minimum landing size >30 cm is unknown. The selectivity-at-length of the fishery as assumed in the eel population model is given in Table 2.4.

Table 2.4. 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

2.3.5 Natural Mortality

We used an estimate of M=0.138 ($year^{-1}$) for the natural mortality, as in Dekker (2000) and van der Meer (2010).

2.4 Results and discussion

The predicted spawner production (in terms of biomass of silver eel) per recruit (glass eel) for varying sex-ratios and fishing mortalities are given in Table 2.5. The predicted %SPR for various levels of yellow eel fishing mortalities are given in Table 2.6, assuming a sex-ratio of 50% males and females.

Estimates of the "best possible" spawner production, if yellow eel fishing mortality is zero (*F*=0), vary from 41.4 to 69.9 grams per glass eel, depending on the sex-ratio. These predicted values are comparable to the estimates by Dekker (2000) who estimated 40 grams silver eel production per glass eel for a sex ratio of 86% males. Witteveen & Bos (2010c) and van der Meer (2010) estimated 37 grams per recruit, but the sex-ratio which was used in their predictions is unclear.

Different estimates of growth and maturation-at-length were used in the other studies. Witteveen & Bos (2010c), van der Meer (2010) and Dekker (2000) used a constant 3.5 cm per year as a growth curve for both sexes. In contrast, in this report a sex-specific growth curve based on otolith readings is used in which individuals grow faster in the first few years of their lives as yellow eels. With respect to maturation-at-length, eels were assumed to mature at lengths of 45 (males) and 65 cm (females) in van der Meer (2010). and The estimates in Dekker (2000) cannot easily be reproduced because they are presented in graphical form.

It is impossible to decide at this point which of the models gives the best predictions, given the uncertainty and possible biases in the data used to estimate vital parameters in the models such as sex-specific growth rates, weight-at-length, and maturation-at-length (paragraph 2.3). However, we prefer the model presented in this report on the grounds that is parameterised using extensive recent data of the key biological parameters, collected on eels in The Netherlands.

Table 2.5. The predicted number of grams of spawner (silver eel) production per recruit (glass eel) under a range of combinations of trends in sex-ratios and growth models.

Proportion males	F=0	F=0.1	F=0.2	F=0.3
25%	69.9	30.2	14.4	7.6
50%	55.6	25.8	13.4	7.6
75%	41.4	21.4	12.3	7.6

It is important to note that yellow eel mortality greatly affects spawner production becuase eels in The Netherlands take many years to mature, in particular females (Chapter 5). Also, it should be noted that between values of F=0.1 and F=0.2 the spawner production as a percentage of the best possible production (with F=0) drops rapidly from (25.8/55.6)=46.4% at F=0.1 to (13.4/55.6)=24.1%, in the case of a 50%-50% sex-ratio (Table 2.10).

Table 2.6. The predicted %SPR for given values of yellow eel fishing mortalities, for a sex-ratio of 50% males.

Proportion males	**F=0**	**F=0.1**	**F=0.2**	**F=0.3**
50%	100%	46.4%	24.1%	13.7%

3. An extended yellow eel model for estimating mortality rates

3.1 Introduction

A length based population dynamics model for yellow eels was introduced in Chapter 2. In this chapter, we extend the yellow eel population model with the aim to estimate fishing mortalities. We do so by fitting the model to data on catches per unit of effort from stock surveys, or to length-frequency distributions from retained catches (market sampling). To this end, the yellow eel population model is extended by adding a recruitment index for the relative strengths of cohorts and estimates of sex-ratios of cohorts.

Estimates of fishing mortalities derived in this manner are valuable because they can be contrasted with estimates of fishing mortalities based on direct estimates of standing stock biomass (Chapter 4). Also, estimates of fishing mortalities can in turn be used to estimate standing stock biomass from retained catches. This is useful for estimating the standing stock biomass in large lakes because data from electric dipping nets are less representative for such large water bodies (see Chapter 4).

Fishing mortality rates are estimated by comparing ('fitting') model predictions for a range of fishing mortality rates to:

1) observed length-frequency distributions of retained catches from commercial fisheries in a number of regions in The Netherlands. In The Netherlands, a pilot project was started in 2009 and 2010 (in two regions) and in 2011 (in several regions) to sample the retained catches of commercial fisheries to obtain representative length-frequency distributions of the catches. The idea was that these length-frequency distributions could be used to infer fishing mortality rates from the shape of the length-frequency distribution;
2) observed catches per unit of effort per length class from a electro-trawl survey in lakes IJsselmeer and Markermeer. For the purposes of evaluating the dynamic eel population model, the most best-studied (Dekker 2000, 2003) situation is provided by the stocks from lakes IJsselmeer and Markermeer. Both lakes have good recruitment indices, a long-term survey of stock trends, and market sampling of landings by commercial fishers. The stocks from these lakes therefore provides the best opportunity to evaluate the ability of the population model to reproduce stock trends and length-frequency distributions in a given calendar year.

A (mathematical) description of the model is given in paragraph 3.2. Estimates of trends in recruitment and sex-ratios are given in paragraph 3.3. Results are presented in paragraph 3.4. Paragraph 3.4.1 gives the main yellow eel stock trends in lakes IJsselmeer and Markermeer and the main rivers Rhine and Meuse. These long term stock trends are important to take into account when trying to infer mortality rates using survey data or length-frequency distributions. A description of the attempt to estimate fishing mortalities by comparing model predictions to observed length-frequency distributions of retained catches is given in paragraph 3.4.2. No reliable estimates of fishing mortalities could be obtained this way, and the results from this assessment are not further used in this report. Estimates of yellow eel fishing mortality rates, obtained by fitting the population model to the long term electrotrawl fish stock survey data in lakes IJsselmeer and Markermeer are given in paragraph 3.4.3. These estimates were deemed more reliable, and are used in Chapter 4 to estimate the standing stock biomass in lakes IJsselmeer, Markermeer, Grevelingenmeer and Veluwerandmeren. This is illustrated in the flow diagram below (Figure 3.1).

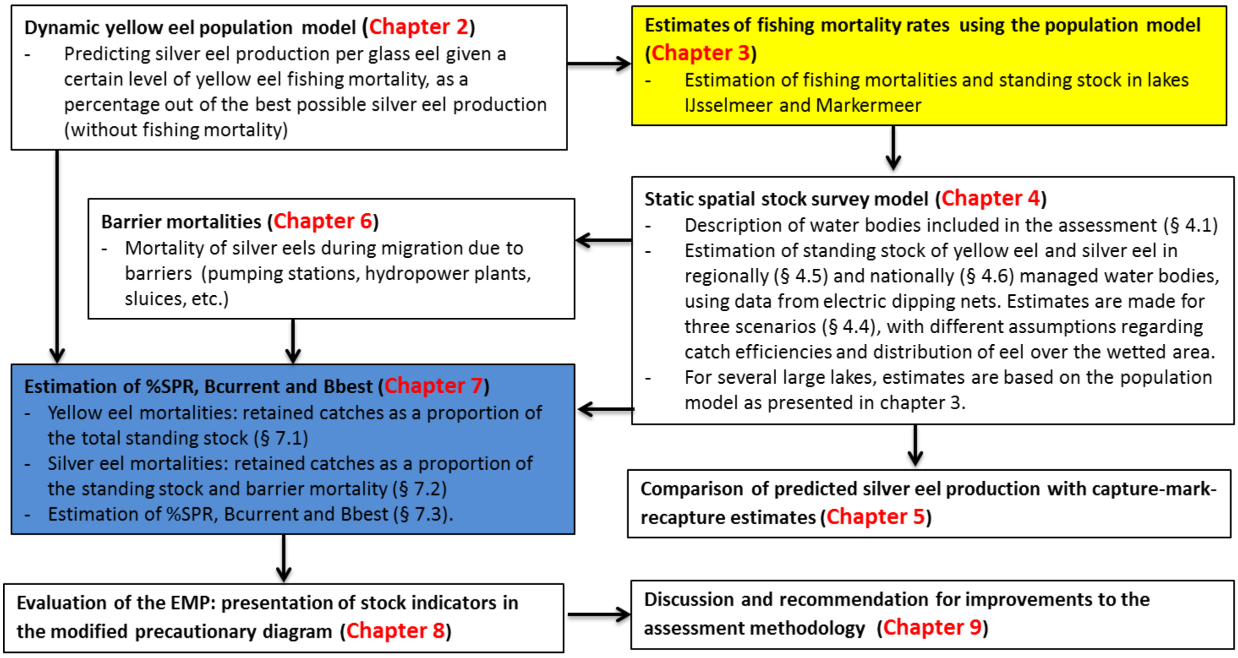

Figure 3.1 A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report. The results from this chapter (yellow) are further used in Chapter 4. Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

3.2 The extended population model

We refer to Chapter 2 for a detailed introduction to the yellow eel population model. The basic model is given by (see Chapter 2 and Table 3.1):

$$N_{a,j,t} = N_{a-1,j,t-1}\, e^{\left(-M-F s_{L(a,j)}\right)} \left(1 - q_{j,L(a,j)}\right) \qquad \textbf{(as Eqn 2.1)}$$

The population model as described in Chapter 2 is used only for the estimation of %SPR for a given sex-ratio and *F*. However, in this chapter we aim to estimate *F* by attempting to predict stock trends and fitting these predictions to:

1) relative length-frequency distributions of retained catches (estimated from data from market sampling of retained catches in several areas in The Netherlands);
2) observations on catches per unit of effort per length class (lakes IJsselmeer and Markermeer).

For the model to predict stock trends, it needs to be extended s with an index of relative recruitment strength and estimates of sex-ratios of the cohorts.
Estimates of growth rates, maturation-at-length, natural mortality, selectivity-at-length of the fisheriesand a length-weight relationship are given in Chapter 2 (paragraph 2.3). A recruitment index and estimates of sex-ratios are given in paragraph 3.3.

3.2.1 Estimating yellow eel fishing mortality rates from relative length frequencies

Because predictions are fitted to relative length-frequency distributions, we need relative levels of recruitment, and the following model for recruitment was used:

$$N_{a=0,j,t} = R_t\, b_{j,t} \qquad \textbf{Eqn 3.1}$$

With R_t an index for relative recruitment strength, and $b_{j,t}$ the sex-ratio of the cohort of glass eels that arrives in year *t*.

If length-frequency distributions of retained catches are used to infer mortalities, it will have to be assumed that these are representative of the length-frequency of the underlying population. Therefore, when comparing frequency distributions with model predictions, we used data from eels over 35 cm only, since the fisheries are assumed to be fully selective from these lengths. For each length-frequency distribution, we have set the frequency of the length class 35-40 cm to 100%. Given different scenarios for sex-ratios and mortalities, we then predicted length-frequency distributions in 2011 (2010 for the downstream parts of the main rivers, because the fisheries there were not sampled in 2011). As recruitment index we used the long term time series of annual average glass eel catches at the Den Oever monitoring station (paragraph 3.3). The other key parameters in the population model; natural mortality, maturation-at-length, and growth rates were as given in paragraph 2.3.
In an initial analysis, observed length-frequency distributions are compared visually to model predictions for a range of fishing mortalities and sex-ratios: F=0 (no fisheries mortality), F=0.1, F=0.2 and F=0.3, for an assumed sex-ratio of 100% females, 50% females and 25% female. In this graphical evaluation, the potential for the use of these frequency distributions in estimating fishing mortalities is assessed. This is done by evaluating if the ratios of the proportion of eels in length classes >40 cm over the proportion in the 35-40 cm length class correspond to ratios as predicted by the model under different assumed fishing mortalities. The outcome of the initial analysis indicated too many problems in the interpretation of the data (see results section of this chapter), and a more formal analysis of the length frequency distributions was not attempted.

3.2.2 Estimating yellow eel fishing mortality rates from stock surveys in lakes IJsselmeer and Markmeer

When model predictions are fitted to data on absolute catches per unit of effort per length class, it is necessary to predict absolute levels of recruitment, and an additional scaling parameter, *k*, needs to be estimated:

$$N_{a=0,j,t} = R_t\, b_{j,t}\, k \qquad \textbf{Eqn 3.2}$$

With $b_{j,t}$ the sex-ratio of the cohort of glass eels that arrives in year *t*, and *k* a scaling parameter to scale relative recruitment strength to observed numbers of eels in the survey.

Two parameters, *F* (annual instantaneous fishing mortality) and *k* are estimated by fitting predicted numbers of eel per length class (summed over ages and sex and growth group, $Z_{L(a,j),t}$, to observed numbers of eels per length class *L*, $X_{L,t}$.

$$Z_{L(a,j),t} = \textstyle\sum_{a,j} N_{a,j,t}\, I_{L,a,j}\, O_{L(a,j)} \qquad \textbf{Eqn 3.3}$$

Where $I_{L,a,j} \in \{0,1\}$ is an indicator function to assign eels of age *a* and sex *j* to a length class ($I_{L,a,j} = 1$ if the growth models predicts the length of eels of age a, and sex *j* to fall within length class *L* and 0 otherwise). Finally, the selectivity of the survey for eels of length class *j* is given by $O_{L(a,j)}$.

For the model fitting, the Poisson likelihood function was used for the probability of observing a count $X_{L,t}$ given a predicted rate $Z_{L(a,j),t}$,:

$$Likelihood(F,k) = \textstyle\prod_{L,t} \frac{Z_{L,t}{}^{X_{L,t}} \mathrm{e}^{-Z_{L.t}}}{Z_{L,t}!} \qquad \textbf{Eqn 3.4}$$

Model predictions were compared to observations in length classes $j \in \{25-30, 30-35, \ldots, 70-75\}$.

Parameters were estimated by profiling the likelihood (Eqn 3.4) using the Metropolis Hastings algorithm (Chib & Greenberg, 1995).

Table 3.1. Variables and model parameters.

Parameter	Description	Unit
$N_{a,j,t}$	Estimated population size of glass eels (*a*=0) and yellow eels ($a \geq 1$), of sex *j* in year *t*	Number of eels
j	Indicator for sex ($j \in \{male, female\}$)	
$L(a,j)$	Indicator for length classes for yellow eels $L \in \{10-15, 15-20, 20-25, \ldots, 100-125\}$ as a function of the age and sex	
t	Indicator for calendar year	Years
M	Parameter for natural mortality	Fraction year^{-1}
F	Parameter for Fishing mortality	Fraction year^{-1}
$s_{L(a,j)}$	Selectivity of the fishery for eels of length class *L*	Fraction year^{-1}
$q_{j,L(a,j)}$	Annual probability of maturation for eels of length class *L*	Fraction year^{-1}
R_t	Glass eel recruitment index in year *t*	Relative index
$b_{j,t}$	Proportion of cohort *t* which develops into sex *j*, where: $b_{male,t} + b_{female,t} = 1$	Fraction
k	Scaling factor to transform the relative recruitment index to absolute observed numbers of eels per m^2.	
$Z_{L(a,j),t}$	Estimated number of eels in length class *L* in calendar year *t*	Number of eels
$O_{L(a,j)}$	Selectivity of the survey for eels in length class *L*	Fraction
$X_{L,t}$	Observed numbers of eels in the survey of length-class *L*, in year *t*	Number of eels

3.3 Estimates of recruitment trends and sex-ratios

3.3.1 Sex-ratio at length

In the eel population model, it is necessary to assign sex-ratios to cohorts. Good quality estimates of sex-ratios are crucial in estimating spawner-per-recruit ratios and mortalities using length-based stock data, given the pronounced differences in maturity-at-length and maturity-at-age between the sexes (see for example Table 16 on page 191 in Tesch, 1977; Melià et al., 2006). 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 considered a likely candidate 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. Lafaille 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). Obtaining unbiased estimates of sex-ratios is difficult given the biology of eels. Males mature and subsequently leave the population at much shorter lengths than females. Therefore, estimates of sex-ratios of cohorts need to be made for eels with length classes below the smallest lengths at which males mature. However, the market sampling data has not focussed on such small eels. As a consequence, eels smaller than 30 cm are rare in the samples because the legal minimum landing size is 28 cm and some fishers use 35 cm as their minimum landing size. Thus, data with which unbiased estimates of sex-ratios of cohorts can be made are scarce. Also, the size or age at which the sex is irreversibly determined is unknown and the sex of eels smaller than 30 cm can often not be determined by dissection and visual inspection.

The sex-ratios as found in the biological market sampling data in 2011 for a number of regions (some of which comprise many inter-linked water bodies with several active fishers) are given in Table 3.2.

Table 3.2. Sex-ratios-at-length, given as the percentage of females per length-class (5 cm intervals), as found in the biological market sampling data in 2011 for a number of regions (some of which comprise many inter-linked water bodies with several active fishers).

Length class		**Region**						
From (cm)	**To (cm)**	**IJssel-meer**	**Marker-meer**	**Noord-Holland**	**Noord-Nederland**	**Rand-meren**	**Zeeland**	**Zuid-Holland**
0	5	-	-	-	-	-	-	-
5	10	-	-	-	-	-	-	-
10	15	-	-	-	-	-	-	-
15	20	0	-	-	-	-	-	-
20	25	0	0	0	0	-	-	-
25	30	18.2	6.2	0	66.7	66.7	33.3	-
30	35	52.4	22.2	71.4	58.3	85.7	25	33.3
35	40	52	75	73.9	65.6	91.7	68.2	57.1
40	45	87	75	66.7	84.2	88.9	100	62.5
45	50	100	100	90.9	90.9	100	94.4	92.3
50	55	100	100	100	100	100	100	100
55	60	100	100	100	100	100	100	100
60	65	100	100	90	100	100	100	100
65	70	100	100	100	100	100	100	100
70	75	100	100	100	100	100	100	100
75	80	100	100	100	100	100	100	100
80	85	100	-	100	100	100	100	100
85	90	100	100	100	100	100	100	100
90	95	-	-	100	100	100	-	100
95	100	-	-	-	100	-	-	100

In lakes IJsselmeer and Markermeer, a market sampling programme has been in operation for several decades. Dekker (2000) found a high proportions of males in the population, and developed a population model for yellow eels in these lakes assuming a sex-ratio of 86% males. However, a problem in the interpretation of sex ratio data from these lakes is that samples were taken at auction. There, eels were pre-sorted in size classes and it is unclear whether the pre-sorting can be assumed to be non-selective for sex. In 2011 (Table 3.3) market samples were taken directly from unsorted catches of fishers. Data on sex-ratios from lakes IJsselmeer and Markermeer combined in the 30-35 cm length class from the biological market sampling programme are given from 2004 to 2011 in Table 3.3. Estimates of sex-ratios in this length class varied from 44% to 83%. A simple mean of the sex-ratios in Table 3.3 gives an average of 65% male. However, given the historically high proportions of males in the population (Dekker, 2000), we used 70% as the proportion of males in the population in Lakes IJsselmeer and Markermeer for the interpretation of stock trends in these lakes. For the visual analysis of length-frequency distributions of retained catches from other region, a range of sex-ratios varying from a minimum of 25% to 100% female has been used.

Table 3.3. Sex-ratios in lakes IJsselmeer & Markermeer.

Year	Females (n)	Males (n)	Unknown (n)	% Male
2004	10	48		82.8
2005	15	66	9	81.5
2006	10	29		74.4
2007	34	27	13	44.3
2008	31	41		56.9
2009	18	20		52.6
2010	26	53		67.1
2011	19	22		53.7

3.3.2 Recruitment index

Recruitment strength of glass eels is monitored in The Netherlands at various stations, but the main index of recruitment is available at the Den Oever monitoring station (Figure 3.2; see Dekker 2002 for a full description), where a large number of hauls are taken every year (de Graaf & Bierman 2011; Table 2.7). Recruitment indices are estimated from glass eel catches per unit of effort as the average number of glass eels per haul in the months April and May. Trends in recruitment at Den Oever and at the other sites are given in Figure 3.3. The relative recruitment strength as used in the population model (Eqn 3.1 and 3.2) is given in Table 3.4.

Figure 3.2. Locations of glass eel monitoring in The Netherlands.

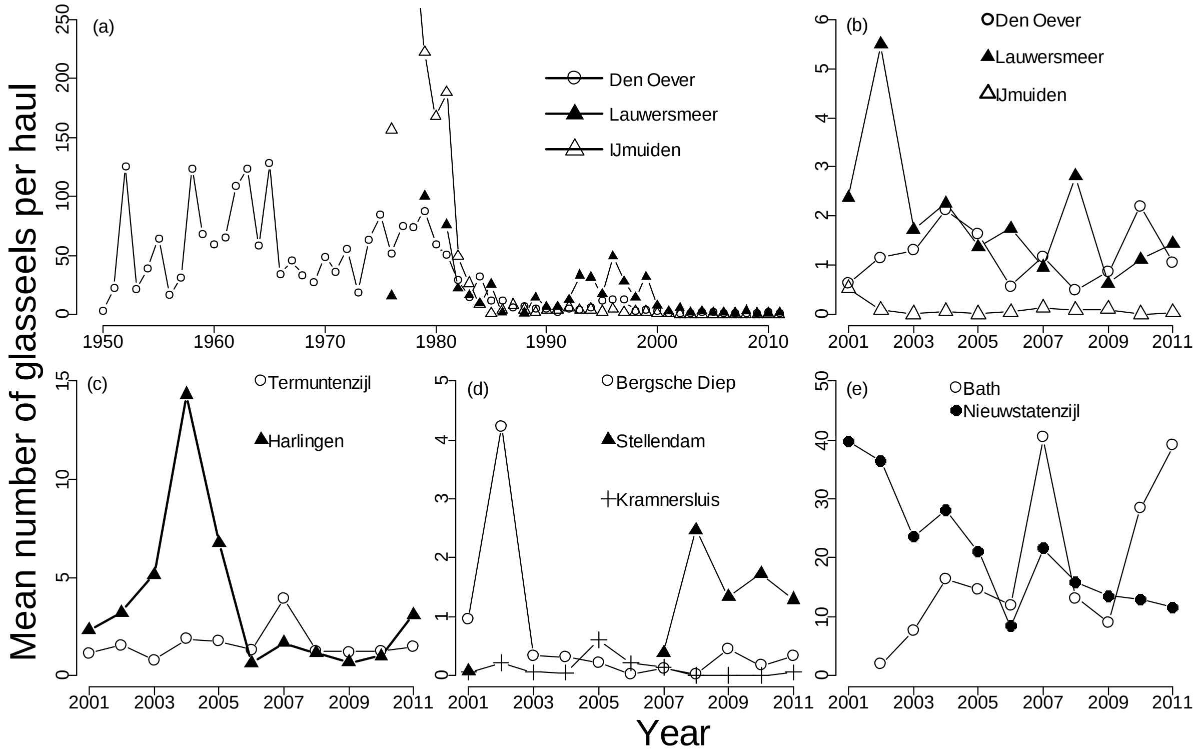

Figure 3.3. Trend indices (mean number per haul in April and May) of glass eel recruitment at different locations along the coast of The Netherlands.

Table 3.4. Average number of glass eel caught per lift net haul at the sluices in Den Oever in the period April-May.

Decade Year	**1930**	**1940**	**1950**	**1960**	**1970**	**1980**	**1990**	**2000**	**2010**
0		22.4	2.7	58.9	48.1	59.0	4.9	2.8	2.2
1		14.3	21.9	65.2	36.1	50.4	1.8	0.6	1.1
2		17.5	125.6	108.9	55.0	29.4	5.2	1.2	
3		13.7	21.1	123.7	18.8	14.7	3.5	1.3	
4		46.1	38.8	58.1	63.0	31.6	5.4	2.1	
5		NA	64.1	128.3	84.3	11.2	11.1	1.6	
6		7.5	16.1	34.0	51.4	11.4	12.5	0.6	
7		7.2	31.3	45.8	75.0	6.2	12.6	1.2	
8	15.3	4.8	124.0	32.9	73.6	7.0	2.4	0.5	
9	71.5	6.6	67.6	27.1	87.7	4.8	3.7	0.9	

3.4 Results

3.4.1 Overview of general trends in the yellow eel stock

Here, we give an overview of stock and recruitment trends in the main rivers and lakes IJsselmeer and Markermeer. The aim of this overview is: (i) to describe the trends in catch per unit of effort and mean lengths of eels, (ii) to describe whether trends in recruitment vary between these waters, and (iii) to describe whether recruitment indices of glass eels are correlated with indices of small yellow eels. The two large lakes IJsselmeer and Markermeer are sampled yearly using an electric beam trawl and an electric dipping net (van Overzee et al., 2011), while the main rivers (Meuse, Rhine and their downstream counterparts) are sampled yearly using a normal beam trawl and an electric dipping net (de Graaf & Bierman 2011).

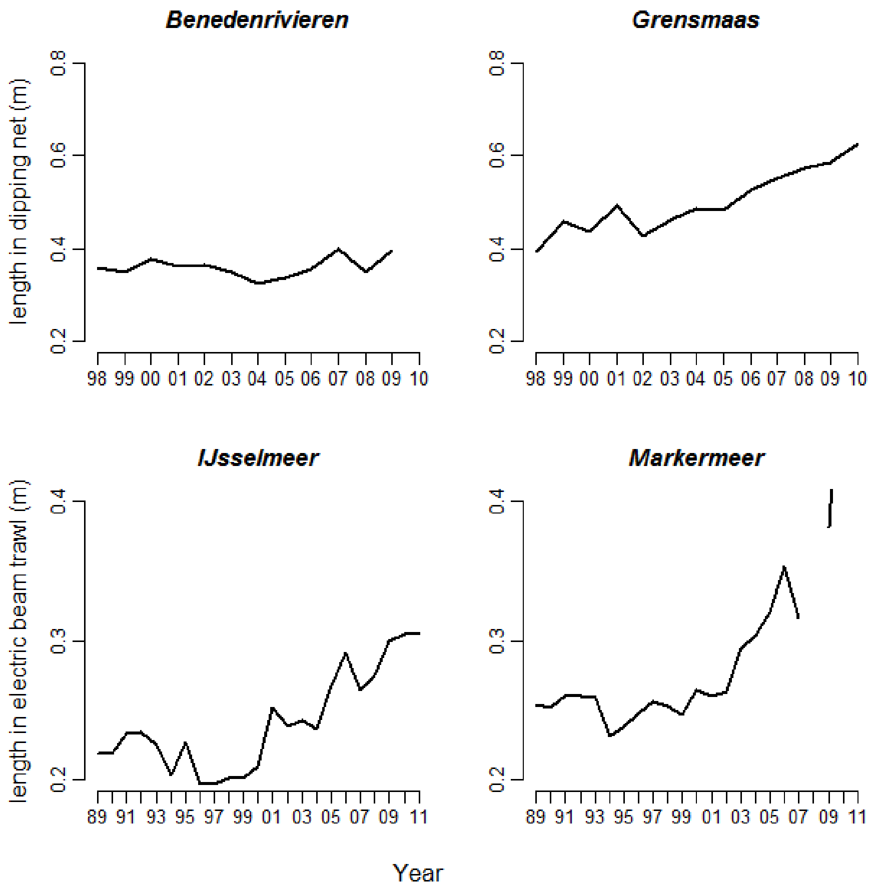

Figure 3.4. Year trends in eel body length (m) caught with an electric dipping net (Benedenrivieren and Grensmaas) and an electric beam trawl (IJsselmeer, Markermeer).

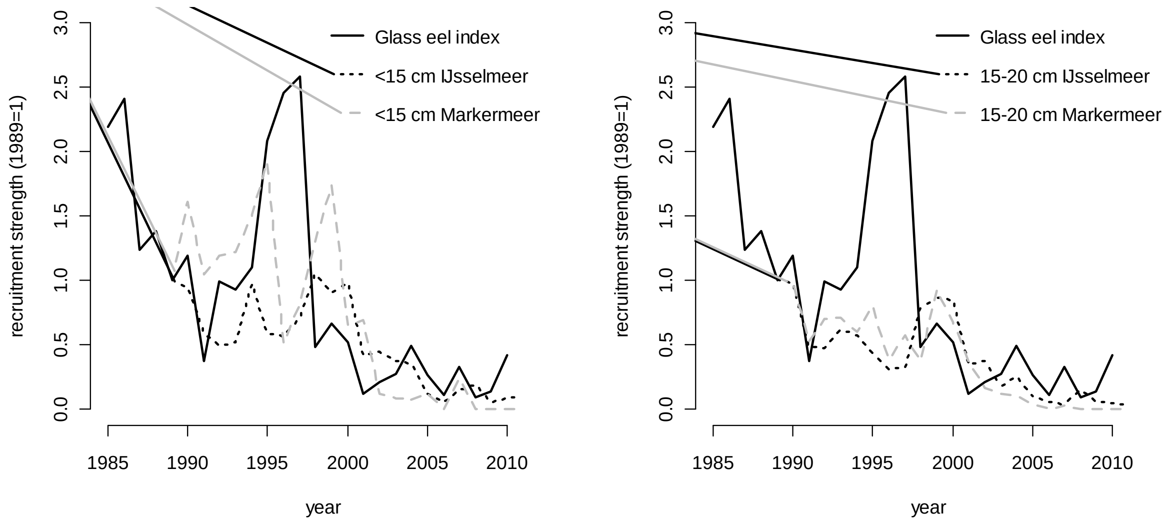

Figure 3.5. Recruitment signal in Lakes IJsselmeer and Markermeer; both glass eel recruitment index (monitoring at the sluice at Den Oever using a dipnet) and small yellow eel recruitment (monitoring using the electrotrawl survey in lakes IJsselmeer and Markermeer). Left panel: comparison of the glass eel recruitment index with the index of recruitment of yellow eels <15 cm in length in Markermeer (grey dashed line) and IJsselmeer (black dotted line). Right panel: comparison of the glass eel recruitment index with the index of recruitment of yellow eels between 20-25 cm in length in Markermeer (grey dashed line) and IJsselmeer (black dotted line).

There is considerable variation in stock trends between different water bodies (Figures 3.4 and 3.5). In lakes IJsselmeer and Markermeer, both recruitment of glass eels and small yellow eel cpue show an overall declining trend since the 1985, but with a pronounced peak in recruitment in 1995-1998. In these lakes, the recruitment signal of glass eels is positively correlated with the cpue of yellow eels up to 40 cm in length, lagged by one or more years (Tables 3.5 and 3.6). The sudden drop in correlation between recruitment signal and eel cpue at lengths above 35 cm coincides with the onset of fisheries mortality as well as the maturation and emigration of males. No clear cohort strength signal is apparent in Tables 3.4 and 3.5, but rather a positive correlation of yellow eel and glass eel cpues at lags from 0 to >15 years. This indicates that both surveys give broadly the same trends over the time series.

The increase in recruitment 1995-1997 in the Den Oever index and lakes IJsselmeer and Markermeer is not apparent in surveys in the main rivers and to a lesser extent at other glass eel monitoring stations. This may indicate spatial variation in glass eel recruitment, but may also be caused by high sampling noise. In recent years, glass eel numbers per tow have been low in the glass eel monitoring programme, and at some locations in some years no glass eels are caught at all. With these low numbers, the ability of the glass eel program to detect small year-on-year changes in absolute abundances of glass eels may have decreased. However, small changes in absolute abundances will result in large year-on-year changes in relative abundance, and these relative differences between years are necessary to interpret length-frequencies. In the upstream sections of the main rivers and lakes IJsselmeer and Markermeer, mean lengths have been increasing since approximately the year 2000.In contrast, in downstream sections of the main rivers, mean lengths have remained relatively constant as well as mean cpues.

The trends in mean lengths and cpues indicate that the eel stock in The Netherlands cannot be assumed to be in equilibrium. Accordingly, vital rates such as recruitment or sex-ratios (see paragraph 2.3) cannot be assumed to be constant over time. Given the instigation of the eel management plan and other changes in the fisheries over the past decades, fishing mortalities can also not be assumed constant over time. The contrasting trends in mean length and cpue between downstream and upstream sections of the main rivers may have been caused by the decline in recruitment. Colonisation of the river system is thought to be partly density-dependent with more upstream migration by elvers or young undifferentiated eels if densities downstream are high (Ibbotson et al*.*, 2002; Edeline et al., 2009). Under falling recruitment, downstream locations are therefore more likely to remain more constant in terms of the length composition and sex-ratio compared to upstream locations.

The shift in mean length in upstream sections of the main rivers and lakes IJsselmeer and Markermeer may have several underlying causes: a decrease in natural and/or anthropogenic mortalities, and/or an increase in the proportion of recruits developing into females. However, it is also possible that the steady increase in mean lengths has been (partly or wholly) caused by the several years of relatively high recruitment from 1995 - 1997. However, the stock surveys in lakes IJsselmeer and Markermeer show an increase in the absolute cpues of eels over 50 cm. This indicates thatthe increase in mean length is not caused just by a strong decrease in recruitment of small yellow eels.

The spatiotemporal trends in the eel stock indicate that it will be difficult to interpret length-frequency distributions in order to estimate mortalities. A combination of trends in vital rates may be underlying the stock trends, and given the spatiotemporal variation, it cannot be assumed that vital rates are similar between water bodies. Apart from lakes IJsselmeer and Markermeer, historical data on trends in vital rates are lacking. A particular problem is the lack of good spatiotemporal recruitment indices, which will be necessary to interpret stock trends. The interpretation of stock trends in lakes IJsselmeer and Markermeer may be more straightforward given that the recruitment index appears to be giving a good indication of cohort strength, and because of the availability of data from a standardised stock survey.

Table 3.5. Lake IJsselmeer. Correlation of cpues of eels in the electrotrawl survey in length classes (5 cm intervals) with the Den Oever glass eel index, lagged by 0, 1, 2 or more years. Positive correlations are given in bold face. Note the sudden drop in correlation between recruitment signal and eel cpue at lengths above 35 cm; this coincides with the onset of fisheries mortality as well as the maturation of males. No clear cohort strength signal is apparent in Tables 3.4 and 3.5, but rather a positive correlation of yellow eel and glass eel cpues at lags from 0 to >15 years. This indicates that both surveys give broadly the same trends over the time series.

Lag Year	**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**
0	0.4	0.2	0.3	0.2	0.1	-0.3	-0.4	-0.5	-0.4	-0.3
1	0.6	0.4	0.3	0.3	0.1	-0.5	-0.5	-0.5	-0.4	-0.3
2	0.7	0.6	0.5	0.4	0.3	-0.4	-0.5	-0.6	-0.5	-0.3
3	0.8	0.9	0.8	0.6	0.3	-0.5	-0.4	-0.5	-0.4	-0.3
4	0.7	0.8	0.8	0.8	0.7	-0.1	-0.4	-0.5	-0.4	-0.3
5	0.6	0.7	0.7	0.7	0.6	0	-0.3	-0.4	-0.4	-0.2
6	0.4	0.6	0.7	0.8	0.8	0.1	-0.3	-0.5	-0.4	-0.3
7	0.4	0.5	0.5	0.6	0.5	-0.1	-0.2	-0.3	-0.2	-0.4
8	0.4	0.6	0.6	0.6	0.5	0	-0.3	-0.3	-0.3	-0.3
9	0.4	0.6	0.7	0.7	0.5	0	-0.2	-0.3	-0.3	-0.3
10	0.5	0.6	0.6	0.6	0.4	-0.2	-0.2	-0.2	-0.3	-0.3
11	0.4	0.6	0.7	0.7	0.6	0	-0.1	-0.3	-0.2	-0.2
12	0.4	0.5	0.6	0.7	0.5	-0.2	-0.2	-0.3	-0.2	-0.2
13	0.4	0.5	0.6	0.7	0.6	0	-0.3	-0.4	-0.3	-0.2
14	0.4	0.5	0.6	0.6	0.5	-0.2	-0.3	-0.4	-0.3	-0.3
15	0.5	0.5	0.6	0.6	0.6	-0.1	-0.4	-0.5	-0.4	-0.3

Table 3.6. Lake IJsselmeer. Correlation of cpues of eels in the electrotrawl survey in length classes (5 cm intervals) with the Den Oever glass eel index, lagged by 0, 1, 2 or more years. Positive correlations are given in bold face. Note the sudden drop in correlation between recruitment signal and eel cpue at lengths above 35 cm; this coincides with the onset of fisheries mortality as well as the maturation of males. No clear cohort strength signal is apparent in Tables 3.4 and 3.5, but rather a positive correlation of yellow eel and glass eel cpues at lags from 0 to >15 years. This indicates that both surveys give broadly the same trends over the time series.

Lag year	**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**
0	0.5	0.5	0.4	0.3	0.4	0.3	0.1	0	0.1	-0.2
1	0.5	0.4	0.4	0.3	0.3	0.2	-0.1	-0.1	-0.1	-0.3
2	0.7	0.6	0.5	0.5	0.5	0.4	0.1	-0.2	-0.1	-0.3
3	0.6	0.7	0.7	0.7	0.6	0.4	-0.1	0	-0.1	-0.3
4	0.6	0.7	0.8	0.8	0.7	0.4	-0.2	0.2	-0.2	-0.2
5	0.4	0.6	0.8	0.8	0.7	0.1	-0.4	0.1	-0.1	-0.2
6	0.4	0.5	0.6	0.7	0.7	0.4	-0.1	-0.1	0.1	-0.1
7	0.3	0.5	0.6	0.7	0.7	0.3	-0.1	-0.1	-0.1	-0.1
8	0.3	0.6	0.7	0.8	0.7	0.4	-0.1	0	0	-0.2
9	0.4	0.6	0.8	0.8	0.7	0.4	0.1	0.1	0	-0.2
10	0.4	0.6	0.8	0.8	0.7	0.3	0	0.1	0	-0.2
11	0.5	0.7	0.8	0.8	0.8	0.5	0.2	0.1	0.1	-0.2
12	0.4	0.6	0.7	0.7	0.8	0.5	0.1	0.1	0	-0.2
13	0.6	0.6	0.6	0.6	0.8	0.7	0.2	0	0.1	-0.2
14	0.6	0.6	0.6	0.6	0.7	0.6	0	0.1	0	-0.2
15	0.7	0.7	0.6	0.6	0.8	0.6	0.1	0.2	0	-0.2

3.4.2 Assessment of relative length-frequencies

In 2011, retained catches of commercial fishers in various months and areas were sampled to obtain length-frequency distributions. The sampling allows exploring the potential use of these frequency distributions in estimating fishing mortalities in different areas.

An overview of mean lengths of eels in the retained catches for different combinations of months, gears and areas is given in Table 3.7. Estimates of relative length-frequency distributions were made based on the samples by assuming that these were directly representative of the retained catches.

Table 3.7. Mean lengths (cm) of eels in retained catches of the commercial fisheries in 2011, for combinations of gears and areas.

	June	**July**	**August**	**September**
IJsselmeer eel box	-	43.6	42.4	-
IJsselmeer Longline	-	-	40	-
IJsselmeer fyke	-	42.7	32.3	-
Markermeer fyke	-	-	41.2	-
Noord-Holland fyke	50.5	45.7	52.4	-
Noord-Nederland fyke	45.9	47.1	51.4	49.5
Randmeren fyke	66.2	-	59.6	-
Zeeland fyke	47	51.8	49.7	-
Zuid-Holland fyke	51.4	51.5	53.8	-

The yellow eel population model introduced in this chapter predicts length-frequency distributions of yellow eels by calendar year. We have therefore compared model predictions under different scenarios for mortalities, with length-frequency distributions of yellow eels only (silver eels and yellow eels were distinguished in the field). An overview of the estimated yellow eel length-frequency distributions is given in Figure 3.6.

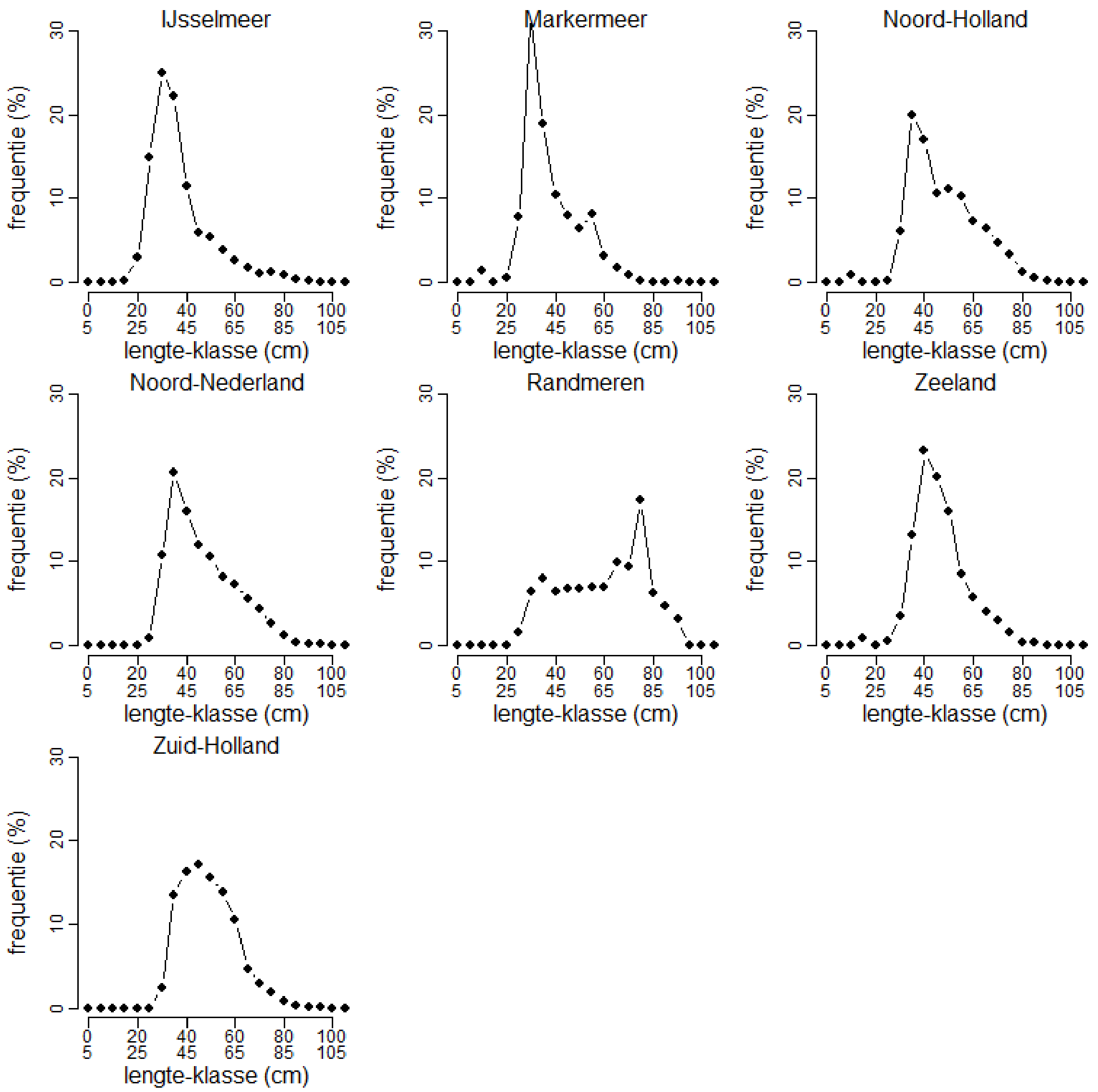

Figure 3.6. Observed length-frequency distributions of the retained catches of eel by commercial fisheries. Given are the proportion of eels per length class (5 cm intervals) out of the total number of eels in the samples, per area.

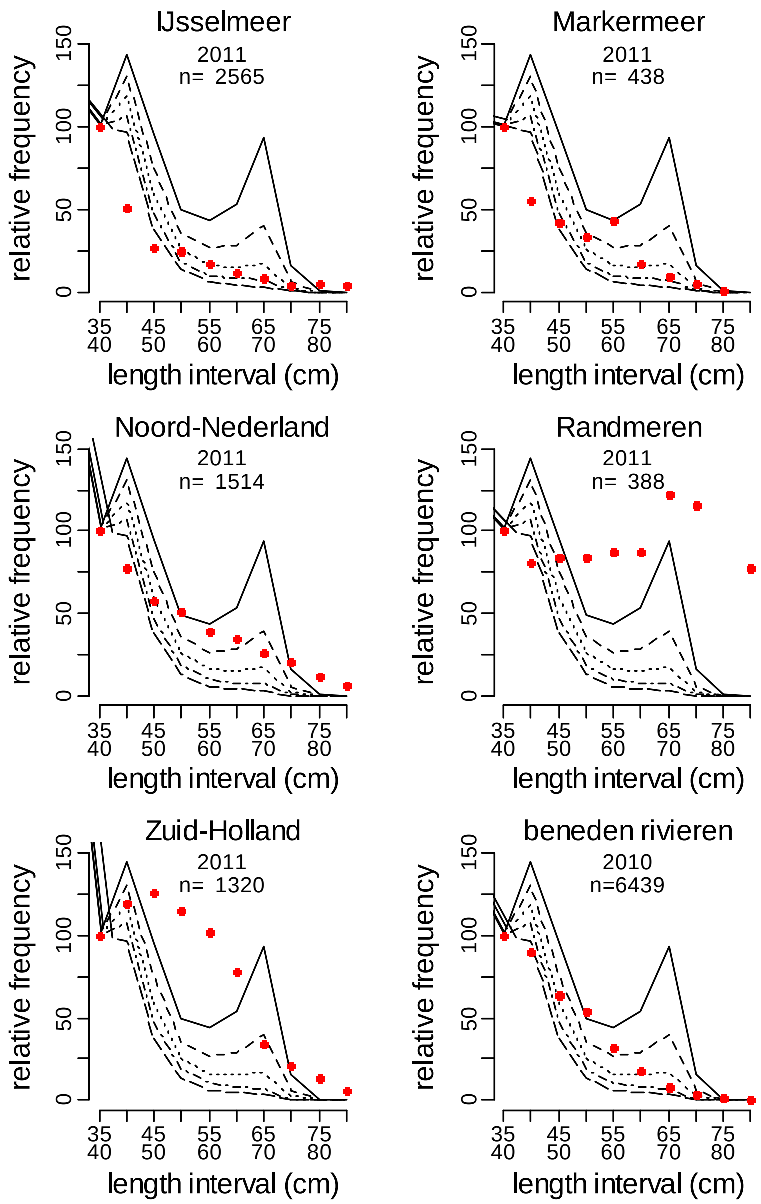

Figure 3.7. Assumed sex-ratio of 100% female (0% male). Observed yellow eel length-frequency distributions of retained catches (red dots; scaled to have the frequency in the 35-40 cm class set to 100%), compared to predicted length-frequency distributions using the yellow eel population model, assuming that 100% of eels in cohorts develop into females, using the Den Oever glass eel recruitment index. Model predictions were made using a minimum fisheries mortality of F=0 (top line: solid), a maximum of F=0.4 (bottom line: dashed) and F=0.1 (first line from the top: dashed), F=0.2 (second line from the top: dotted) and F=0.3 (third line from the top: dash-dot).

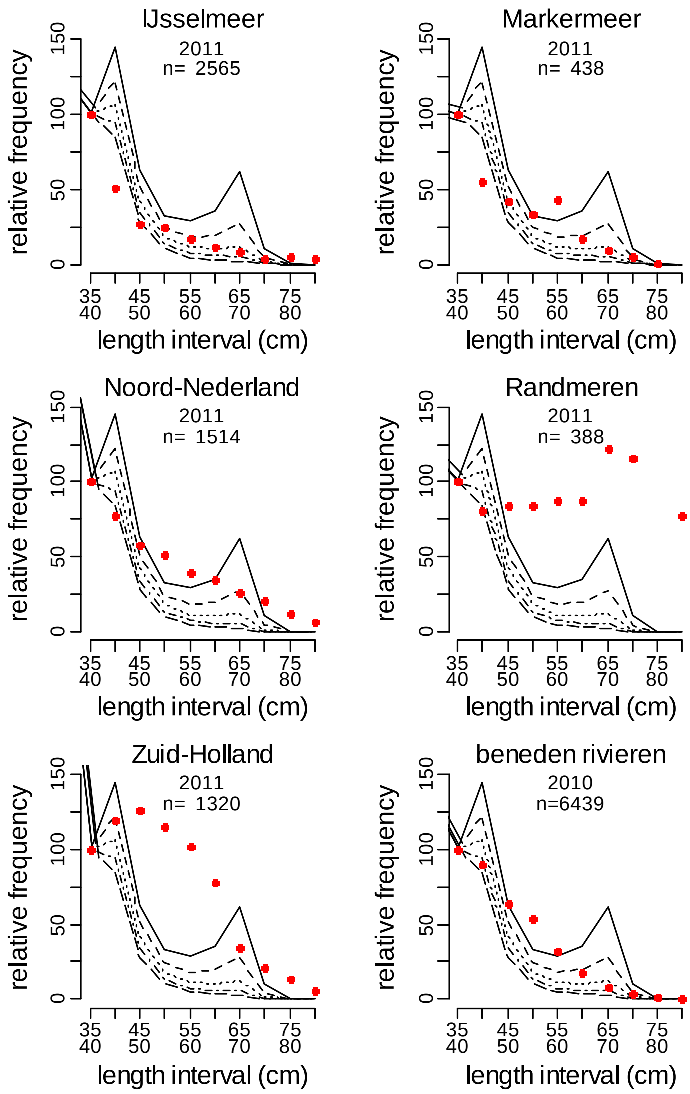

Figure 3.8. Assumed sex-ratio of 50% male. Observed yellow eel length-frequency distributions of retained catches (red dots; scaled to have the frequency in the 35-40 cm class set to 100%), compared to predicted length-frequency distributions using the yellow eel population model, assuming that 50% of eels in cohorts develop into males, using the Den Oever glass eel recruitment index. Model predictions were made using a minimum fisheries mortality of F=0 (top line: solid), a maximum of F=0.4 (bottom line: dashed) and F=0.1 (first line from the top: dashed), F=0.2 (second line from the top: dotted) and F=0.3 (third line from the top: dash-dot).

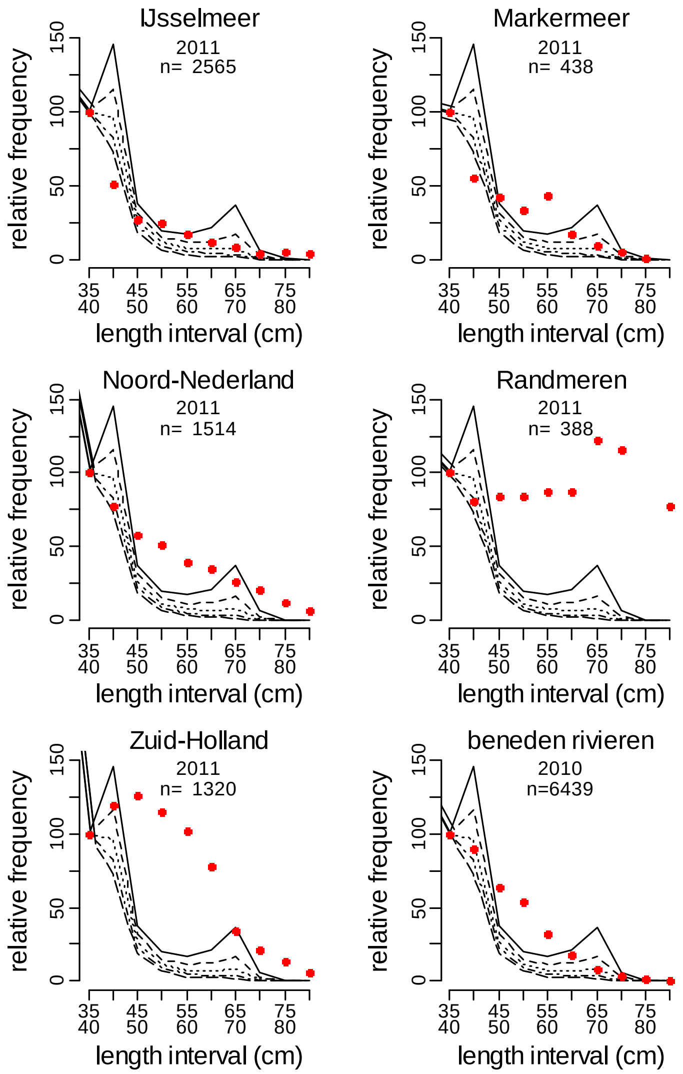

Figure 3.9. Assumed sex-ratio of 25% female (75% male). Observed yellow eel length-frequency distributions of retained catches (red dots; scaled to have the frequency in the 35-40 cm class set to 100%), compared to predicted length-frequency distributions using the yellow eel population model, assuming that 25% (75%) of eels in cohorts develop into females (males), using the Den Oever glass eel recruitment index. Model predictions were made using a minimum fisheries mortality of F=0 (top line: solid), a maximum of F=0.4 (bottom line: dashed) and F=0.1 (first line from the top: dashed), F=0.2 (second line from the top: dotted) and F=0.3 (third line from the top: dash-dot).

The graphical evaluation of length-frequency distributions shows that these are very difficult to interpret. Observations of length-frequencies that fall above the line F=0 indicate that one or more of the model assumptions may be incorrect, such as the assumed growth curves, recruitment index, sex-ratio, natural mortality. Additionally, it is possible that the precision of the length-frequency distributions is insufficient. The assumption of stationarity in these vital rates may also be incorrect. Dekker (2000), in an evaluation of eel length-frequency distributions in lake IJsselmeer also concluded that a catch curve analysis on these data did not work because the stock was not in equilibrium. Observed length-frequencies did not consistently fall within the range F=0 to F=0.4 (which is a very high fishing mortality for yellow eel) for any of the assumed sex-ratios.

In general, it is difficult to obtain estimates of (either natural, fishing or total) mortalities by evaluating length-frequency distributions (Hillborn & Walters, 2001; pages 410-433), especially if a population cannot be assumed to be in equilibrium (with recruitment and mortalitiesbeing constant over a longer period of time). However, for eel, the difficulties in the interpretation of length-frequency distributions (LF-distributions) are likely to be much larger for the following reasons:

1) **Growth rate:** Individual eels show large variability in growth rates and length-at-maturity, which causes cohorts to 'blend' together in LF-distributions such that no clear modes are apparent that represent individual cohorts. Determining age of eels, particularly older individuals can only be done with great uncertainty, precluding the formation of useful age-length keys. Thus, a clear intuitive analysis of the rate of decrease of cohorts over years for estimating mortalities is not possible.
2) **Recruitment:** The eel stock is in a long term decline and given the long stay in inland waters, a trend in recruitment many (>15) years back is needed to interpret present-day LF-distributions. However, for many areas, such a recruitment index is not available. Assuming constant recruitment will likely lead to severely biased estimates.
3) **Natural mortality and migration:** Given the strong decrease in standing stock densities and recruitment it is unlikely that vital rates such as natural mortality and immigration or emigration rates have remained constant. In the absence of historical data on such rates, it is not possible to interpret present-day length-frequency distributions. The eel stock in The Netherlands is spatially structured, and there is an unknown degree of immigration and emigration between linked water bodies. In a stock assessment, immigration and emigration are events which are equivalent to recruitment and death respectively. Ignoring these rates when interpreting length-frequency distributions may lead to severely biased estimates, whereas information on spatial movements of eels is lacking to build more complex spatially explicit models.
3) **Sex-ratio:** Because of the pronounced differences between the sexes in length-at-age and maturity-at-length, detailed long-term information on sex-ratios is necessary for the interpretation of LF-distributions. However, such trends are not available. Obtaining unbiased estimates of sex-ratios is difficult given the differences in maturation-at-length and maturation-at-age between the sexes, and practical difficulties and possible biases in determining the sex of small eels.
4) **Maturation-at-length:** Eels leave the population after maturation, which in a stock assessment is an event with a similar net result on the standing stock as death. Thus, very good estimates of maturation-at-length or maturation-at-age are necessary, and getting these wrong will inevitably lead to biased estimate of mortalities. This is especially true for males because the fisheries become selective for individuals with body lengths from approximately 30 centimeters onwards. This coincides with the lengths above which males start to mature and leave the population. This is reflected in the cohort signal in the survey in Lakes IJsselmeer and Markermeer; some degree of cohort strength is apparent for eels with lengths up to approximately 30-35 centimeters, but disappears abruptly at lengths above 35 cm (IJsselmeer) and 40 centimeters (Markermeer) at which point the cpues drop sharply also. Separating the impact of maturation or fisheries mortality is only possible if one has great confidence in trends in sex-ratios and maturation-at-length curves. This is not the case for the stock assessments in these lakes or indeed anywhere in The Netherlands.

The analysis of such single length-distributions has been called 'fundamentally hopeless' (Hillborn & Walters, 2001; page 432), and we conclude that using the present means of analysis (the use of the yellow eel model), no useful information on fishing mortalities can be obtained using these data (see also the conclusions in Dekker 2000 regarding the catch curve analysis of eel data from lake IJsselmeer). In order to robustly interpret such length-frequency distributions, good quality information of all the critical parameters listed in paragraphs 2.3 and 3.3 are needed, but in particular estimates over a minimum of 15 years preceding the calendar year in which the assessment is done on sex-ratios and recruitment. Furthermore, the chances of getting robust estimates of fishing mortalities will be improved if effort data from commercial fisheries are available, such that relative frequency distributions can be scaled to estimates of catch per unit of effort per length class.

3.4.3 Stock assessment lakes IJsselmeer and Markermeer

The fisheries in Lakes IJsselmeer and Markermeer have been affected at various time points by sharp reductions in the number of fishing gears. Different fishing mortalities were allowed for these different periods in the model (Figure 3.10). The periods for which different F's were allowed were: 1970 up until 1989, 1990 - 1999, 2000 - 2005, 2006 - 2010.

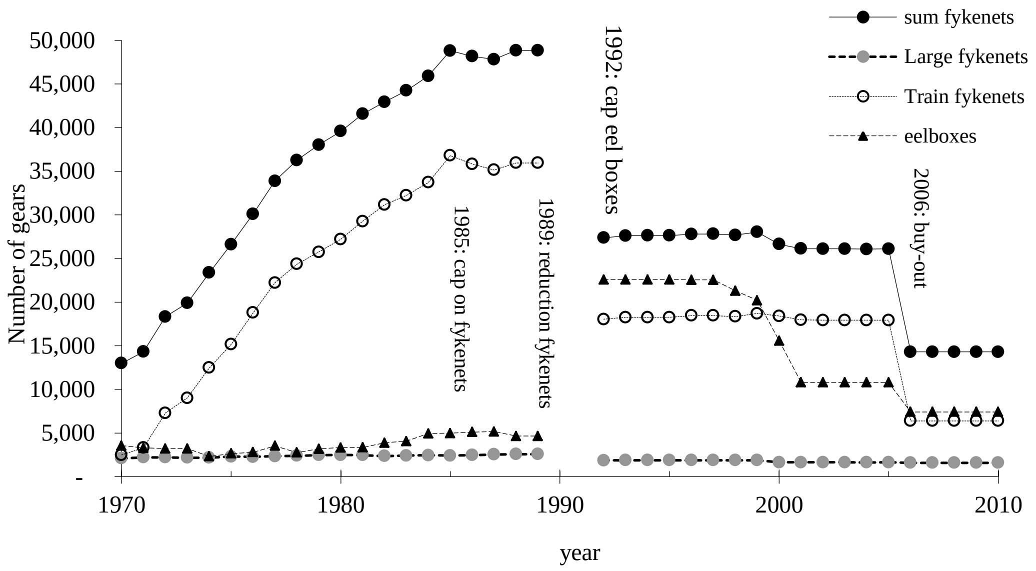

Figure 3.10. Changes over time in the nominal fishing gears in lake IJsselmeer and Markermeer.

The model was fitted to the survey data by estimating the scaling parameter *k* (Eqn 3.2)and the fishing mortalities in the different periods, by profiling the likelihood (Eqn 3.4).

The model was able to reproduce the observed eel stock trends in the lakes over the years reasonably well with a sharp decline in recruitment mirrored in a sharp decline of small yellow eels (<30 cm). The predicted versus observed stock trends, for a selection of length classes and mean length are given in Figures 3.11 and 3.12 (IJsselmeer) and 3.13 and 3.14 (Markermeer).While the cpue of eels <35 cm decreased over time, cpues of large yellow eels (>50 cm) increased, resulting in shifting length-frequency distributions towards on average larger eels (Figures 3.5 and 3.11). This is remarkable since eels over 50 cm in length in these lakes had historically been rare, which has been hypothesised to have been caused by a combination of high fishing mortalities and a high proportions of males in the population (Dekker 2000, 2003). The increase in the dominance of large individuals in the survey was accommodated for in the model by a decrease in fishing mortality (see Table 3.7). Fishing mortality was the only vital rate which was allowed to vary over time in the model (other vital parameters such as sex-ratios, natural mortality, immigration, emigration or maturation-at-length were assumed to be stationary). While it is possible that indeed decreases in fishing mortalities have fully or partly caused the observed increase in large individuals, there are other factors that provide alternative explanations for this trend. Emigration of glass eel or elvers from lake IJsselmeer to Lake Markermeer may have decreased as a function of decreasing recruitment. As a function of the decreased recruitment, natural mortality, growth or sex-ratios may have changed over time. Changes in sex-ratio in particular can greatly affect model predictions, given the pronounced differences in life-history characteristics between the sexes. This may be the case in these lakes, with estimates of 86% male in Dekker (2000) but nearer 50-60% in more recent years (paragraph 3.3). Finally, immigration of large yellow eels or silver eels may also explain the increase of large eels in the stock survey. The stock survey in the lakes is done in autumn, when silver eel migration is thought to take place, and no distinction is made between yellow eels and silver eels. It is possible therefore, that the increase in large eels is due to an increase production of silver eels upstream of lake IJsselmeer, such as the river IJssel or the river Rhine. However, large yellow eels are found in the landings by commercial fishers in lake IJsselmeer (figure 3.7), suggesting that local conditions allow production of large yellow eels.

Table 3.7. Model estimated Fishing mortalities (F) in Lakes IJsselmeer and Markermeer for a number of periods with different fishing efforts.

	IJsselmeer	**Markermeer**
<1989	0.92	0.55
1990-1999	0.67	0.33
2000-2005	0.43	0.47
2006-2010	0.10	0.30

As in Lake IJsselmeer, the stock trends in Lake Markermeer are characterised by a falling recruitment and by an absolute increase in the catch per unit of effort of larger eels (above approximately 50 cm). However, the recruitment in Lake Markermeer fell more sharply compared to Lake IJsselmeer, and was also present in the 30-35 cm length class. A decrease in fishing mortality from the 2000-2005 to the 2006-2010 period was estimated (Table 3.7), but the overall increase in large eels could not be explained by the population model.

The estimates of fishing mortalities in lakes IJsselmeer and Markermeer are highly sensitive to the choice of input parameters. For example, Dekker (2000) reported that growth rates and mortality estimates are positively correlated. Thus, the faster the eels are assumed to grow, the higher estimates of mortality will be (if all other vital rates are the same). In a sensitivity analysis (not reported here) we found the same relationship between growth rates and mortality estimates. In the absence of relevant information, it is necessary to assume that vital rates such as immigration, emigration, natural mortality, growth and sex-ratio have remained constant over long periods of time. This is likely to be unrealistic, especially given the fall in recruitment which is likely to have led to changes in rates of immigration and emigration, sex-ratios, natural mortality and growth rates (Edeline et al., 2009; de Leo & Gatto 1996; Davey & Jellyman 2005; Oliveira et al., 2001). Different assumptions on trends in these vital parameters will inevitably lead to different interpretations of length-frequency distributions and stock trends and, therefore, different estimates of fishing mortalities. Some eel population dynamics models exist which explicitly incorporate density dependent processes, such as density-dependent survival of new recruits (glass eels; the DEMCAM model; Bevacqua & de Leo 2006) or density-dependent sex-ratios (the SMEP model; Walker 2006). However, while these models may be more realistic, good quality parameter estimates for these density-dependent processes are not available and it is likely that including more complex processes in the model will only lead to more possible interpretations of the stock trends. Therefore, estimates of fishing mortalities using the population model need to be interpreted with great care, are uncertain.

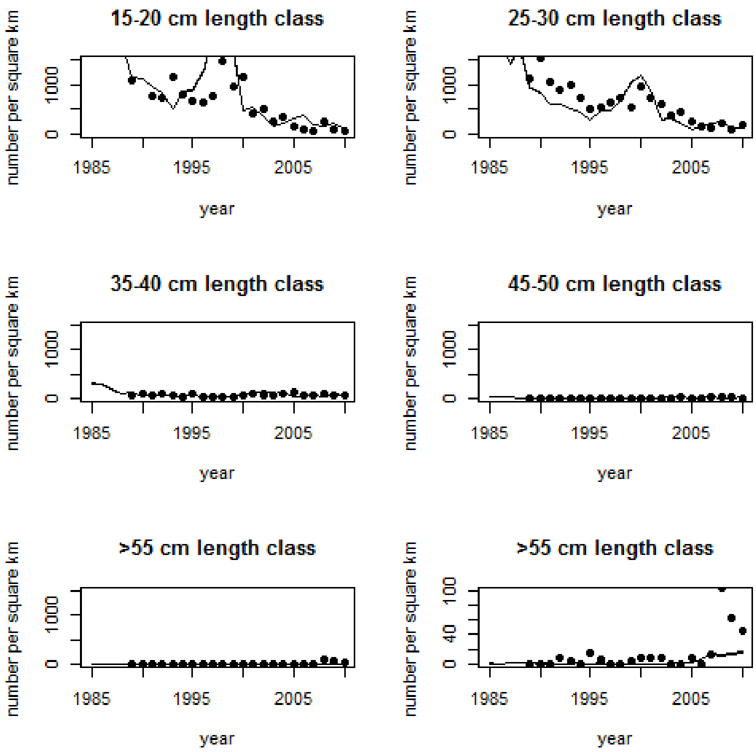

Figure 3.11. Lake IJsselmeer: Observed versus fitted population trends in various length classes.

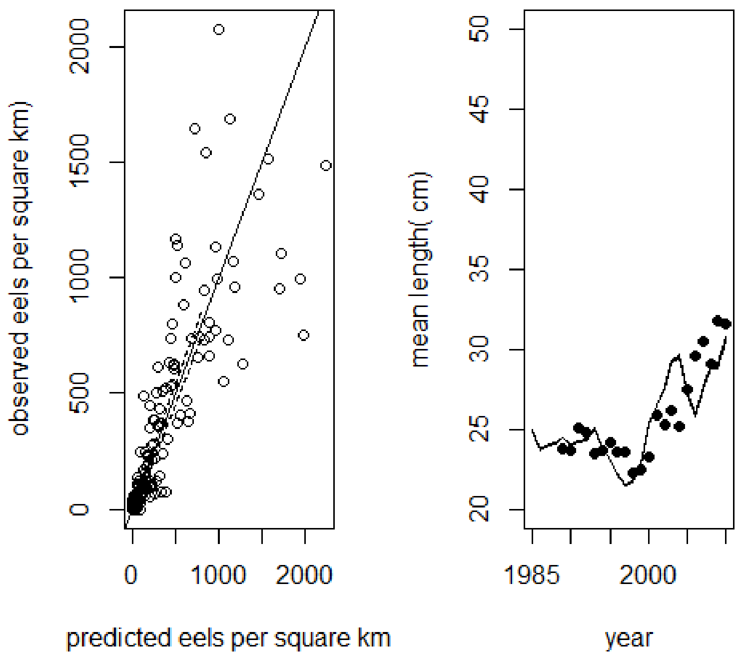

Figure 3.12. Lake IJsselmeer: Observed versus fitted number per square kilometre (all years and all length classes combined) and observed versus predicted trend in mean length.

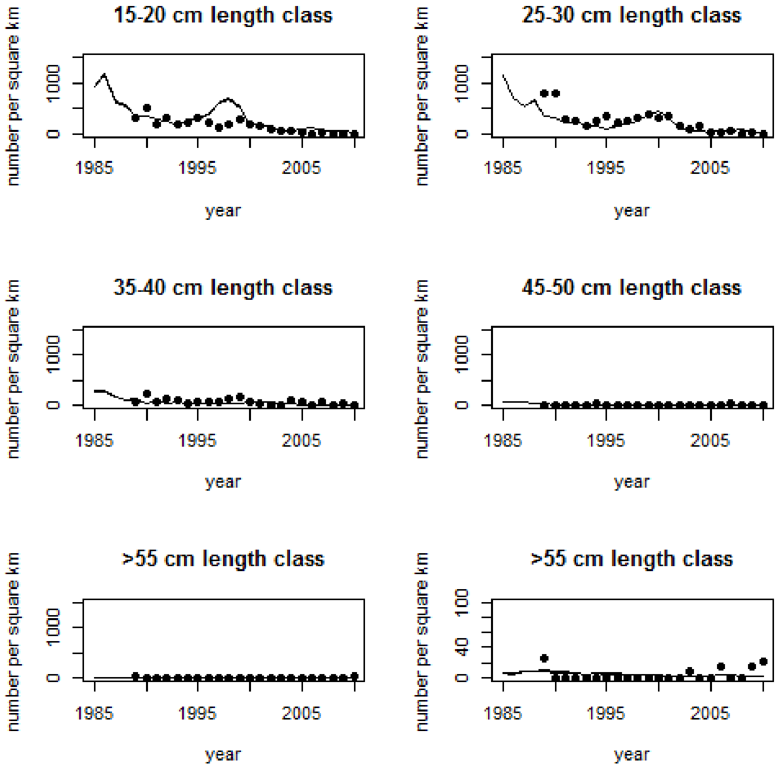

Figure 3.13. Lake Markermeer: Observed versus fitted population trends in various length classes.

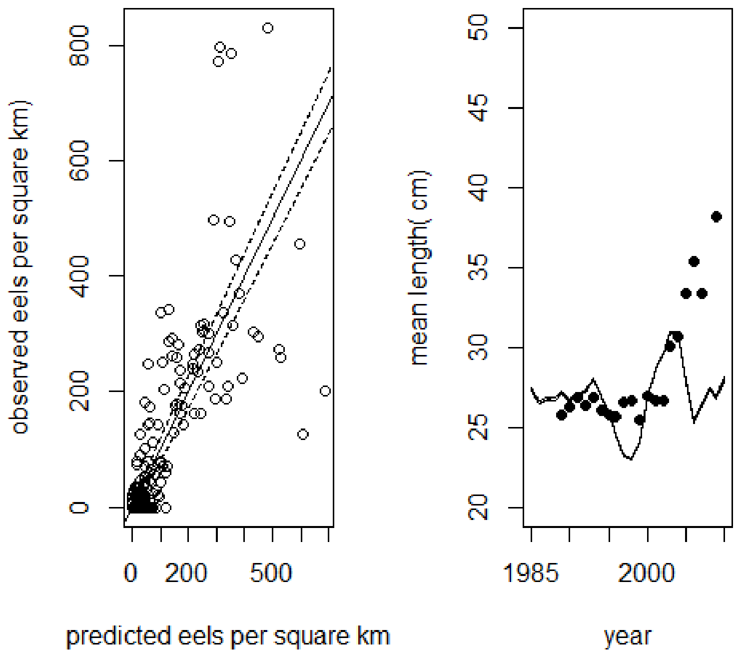

Figure 3.14. Lake Markermeer: Observed versus fitted number per square kilometre (all years and all length classes combined) and observed versus predicted trend in mean length.

4. A static spatial model for yellow and silver eel

4.1 Stock assessment based on survey data

A spatial model was developed in order to estimate the standing stock biomass of yellow and silver eel in The Netherlands. Estimates of standing stock biomass are necessary for estimating the biomass of silver eel escapement and mortalities (see Chapter 1). In Chapter 7, yellow eel and silver eel fishing mortalities for The Netherlands are estimated as the proportion of the retained yellow eel catches out of the standing stock biomass of yellow eel from 30cm in length.

The model incorporates geographic data of Dutch fresh water bodies with spatially-structured eel density data. The eel density data is derived from two national fish surveys and the fisheries model (Chapter 1). For the nationally managed waters (the 'rijkswateren') eel density data from a survey managed by Rijkswaterstaat was used; the "Actieve Monitoring van de Zoete Rijkswateren" (part of the MWTL program). For the regionally managed waters (the 'niet-rijkswateren'), density data collected in the sampling program for the Water Framework Directive (WFD) was used. The latter waters concern the smaller water bodies, such as small rivers and lakes but underrepresents ditches. For some of the larger fresh water lakes, no suitable survey data are available: IJsselmeer, Markermeer, Veluwerandmeren and Grevelingen. For these regions densities estimated with the dynamic eel population model are used.

First, the definition of the various water bodies is given (4.2) and the assumptions of the spatial model are explained (4.3). The model has explicit assumptions concerning the length-weight relation, the sex ratio, the ratio between yellow and silver eel per length class, the catch efficiency of a gear and the habitat preference of eel in a water body. In spatial models such as used here, in which density data from collected surveys are scaled to biomass per water body area, the main sources of uncertainty are the unknown catch efficiencies of the fishing gears and the habitat preference of eel. Both catch efficiency and habitat preference are not well known. Therefore, three scenarios are used to estimate biomass estimates (paragraph 4.4). The survey, GIS data and synthesis per water body are discussed for the three types of waters (regionally managed, nationally managed and the large lakes) separately below (4.5, 4.6 and 4.7, respectively). Finally, combining these data leads to biomass estimates for the three scenarios (4.8).

The results from this chapter are further used in Chapters 5, 6 and 7 (Figure 4.1). Estimates of yellow eel fishing mortality rates in lakes IJsselmeer and Markermeer are used to infer standing stock biomass for a number of large lakes as indicated by the flow diagram below. In Chapter 5, a comparison is made of silver eel production as estimated by the static spatial model with estimates based on capture-mark-recapture experiments. In Chapter 6, barrier mortalities are estimated, based on spatially explicit estimates of silver eel production. In Chapter 7, the estimates of standing stock biomass are used to compute the key stock indicators.

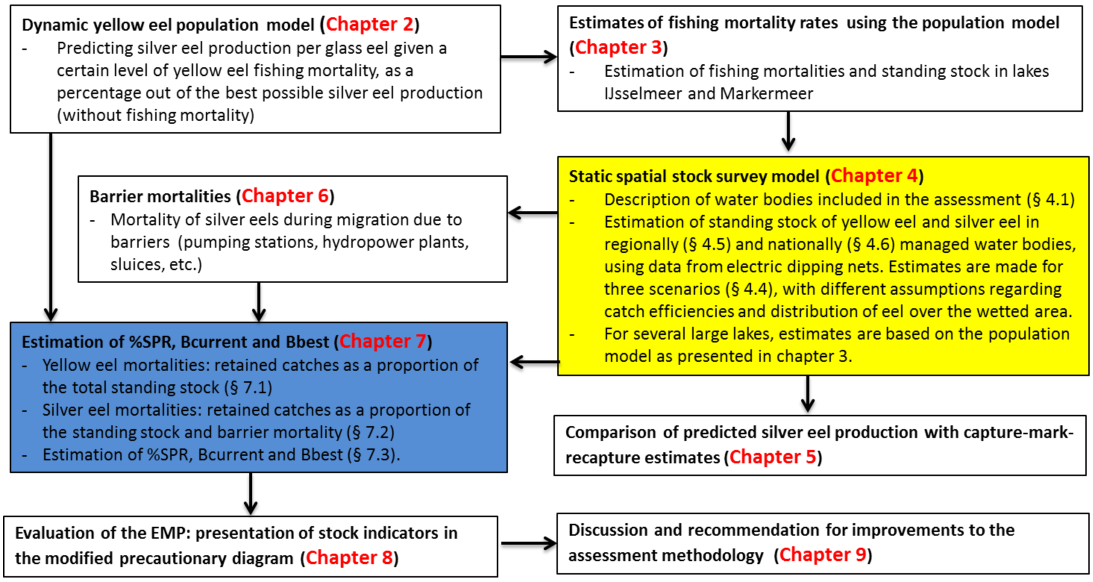

Figure 4.1 A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report. The results from this chapter (yellow) are used in Chapters 5, 6 and 7. Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

4.2 Water bodies

A key part of the methodology for the stock assessment is the definition of the water bodies and their attributes. These attribute include the type of water body (such as lake, river, stream, ditch, etc.), wetted surface area, length in the case of canals and ditches, etc., or length of shoreline (lake) or riverbank (rivers). An explicitly spatial approach is necessary to link data from various monitoring programs to individual water bodies.

In the assessments presented in this report, all water bodies in the Dutch Water Framework Directive (WFD) have been included, 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. These water bodies vary greatly in size from small sections of streams or canals withsurface areas smaller than 0.01 km^2, to large water bodies such as lakes or sections of the main rivers with more than 100 km^2 of wetted area. The set of water bodies are formally described in a number of geographic information system (GIS) computer files. Two different files have been used as a basis for the assessment (http://www.krwportaal.nl):

1) The WFD shapefile of polygons of water bodies (commonly referred to as the 'OWAGV' shapefile; vlakvormige oppervlakte wateren) containing polygons delineating medium-sized and large water bodies.
2) The WFD shapefile in which water bodies are represented as line segments (commonly referred to as the 'OWAGL' shapefile; lijnvormige oppervlakte wateren) containing mostly the smaller water bodies such as streams, canals, and ditches.

The above two shapefiles have considerable overlap (a water body may be represented in both), and therefore a single data base of unique water bodies was created by eliminating water bodies from the OWAGL shapefile which were present in the OWAGV shapefile.

The advantage of using the WFD files is that much of the monitoring data used in the assessment are collected as part of the WFD sampling requirements for assessment of ecological status. Also, given the importance of the WFD for assessing the environmental status of aquatic environments in The Netherlands, information such as typologies of water bodies are readily available. A disadvantage is that not all water bodies are classified as WFD water bodies. In particular fens, small streams, and drainage ditches are missing (PBL 2010; Table 4.1). The fens and part of the smaller streams represent only a very small part of the total surface area of water in The Netherlands. The combined surface areas sum to 218 ha, which is only 0.06% of the Dutch total water surface area, and were therefore not corrected for. However, the large number of drainage ditches -with a considerable length- in the low-lying parts of The Netherlands (the 'polders') are an important omission (PBL 2010; Table 4.1). Hence, in addition to the two shapefiles, we used the estimate of 330000 km of drainage ditch length provided by PBL (2010; Table 4.1). We applied an estimated average width of 1 meter, to obtain a total surface area of 330 km^2 of drainage ditch area.

Table 4.1. An overview of wetted areas in The Netherlands, and the percentage of these areas which are included in the subset of WFD water bodies. This table has been reproduced from: CBS, PBL, Wageningen UR (2010). Oppervlaktewater in Nederland (indicator 1401, versie 01, 6 juli 2010).ww.compendiumvoordeleefomgeving.nl. CBS, Den Haag; Planbureau voor de Leefomgeving, Den Haag/Bilthoven en Wageningen UR, Wageningen.

	Surface area (km^2)	**Length (km)**	**% in set of WFD Water bodies**
Salty waters	62000		20%
Brackish and transitional waters	800		95%
Main rivers	330	650	100%
Canals and larger waterways		6500	90%
Lakes (>50 ha)	2500		100%
Smaller streaming waters (e.g. stream)		6200	70%
Drainage ditches		330000	0.5%
Fenns	2.4		<1%

Estimates of wetted area and shore lengths are computed directly from the WFD shapefile OWAGV, with the exception of the main rivers. For the main rivers estimates of wetted areas are computed using the 'Ecotopenkaart', because the polygons in the OWAGV shapefile included areas which only flood periodically or occasionally. In total, 3582.5 km2 of wetted area is represented in the assessment. An overview of the wetted areas in the model is given in Table 4.2. More information on the OWAGV and OWAGL shapefiles is given in section 4.5.

Table 4.2. Wetted areas of some of the two main lakes in The Netherlands (Lake IJsselmeer and Markermeer), the areas which are closed for fisheries (which comprise the main rivers (Rhine and Meuse and downstream areas) and some larger canals), three other large lakes (the Volkerak-Zoommeer, Lake Grevelingen and the Veluwerand lakes), other larger WFD water bodies as given in the OWAGV shapefile and other smaller WFD water bodies as given in the OWAGL shapefile (see main text). The largest water bodies are managed nationally and typically have long-term surveys.. The smaller water bodies (although there are still some large lakes in this category) are regionally managed by water boards. Since 2006, data on fish stocks are collected in these regionally managed water bodies under the WFD.

Water bodies	**Wetted area (km^2)**	**Management**
Lake IJsselmeer	1137.4	National
Lake Markermeer	700.6	National
Areas closed for commercial eel fisheries, among which the main rivers*	367.8	National
Volkerak-Zoommeer	48.1	National
Lake Grevelingen	139.0	National
Lakes Veluwerandmeren	147.9	Regional
Other water bodies described as polygons in OWAGV	621.8	Regional
Line-shaped water bodies described in OWAGL	89.9	Regional
Drainage ditches	330.0	Regional
Total	3582.5	

* (a) Wetted areas and lengths of river banks for the main rivers and larger canals which are part of the areas which are closed for fisheries have been calculated using the Ecotopenkaart shapefiles instead of the WFD shapefiles of water bodies. (b) These areas are the nationally managed water bodies, together with lakes IJsselmeer, Markermeer, Volkerak-Zoommeer and Grevelingen.

4.3 Assumptions of the spatial model

4.3.1 Length-weight relation

The sampling data are presented as number of individuals and their length, measured in cm. Using a length-weight conversion function the total weight is obtained per haul and per size-class. The length-weight relationship was used as given in paragraph 2.3.

4.3.2 Sex ratio and yellow-silver eel ratio

Standing stock Biomass estimates are generated for yellow eel and silver eel separately. In the surveys, no data on the eel maturity is collected and the fraction on silver eel is calculated here using the key given in Table 4.3. Per length class, individuals are divided into males and females and per gender the fraction silver eel is estimated (see paragraph 2.3 for the data on which these estimates were based).

Table 4.3. Sex ratio and maturity fractions per length class (Van Keeken et al., 2010).

Length class (cm)	Male fraction	Male silver eel fraction	Female silver eel fraction
30-40	0.3182	0.1297	0.0106
40-50	0.0636	0.4489	0.0313
50-60	0		0.0902
60-70	0		0.2172
70-80	0		0.4614
>80	0		0.7747

4.3.3 Catch efficiency

None of the surveys used in this report are done specifically for eel. Various gears are used in the surveys: electric dipping nets, beam trawls, electric beam trawls, fykes and seines. All gears have a catch efficiency of less than 100%. Catch efficiencies of electric dipping are likely to 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). Here, for the electric dipping net, a best guess for catch efficiency of 20% is used, 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 using electric dipping nets 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 & Aphramian (2012) estimated efficiencies of approximately 60% in small rivers. Aprahamian (1986) showed size-selective effects of electro-fishing, with estimated mean probabilities of capture from 0.36 for the smallest (youngest) eels to 0.59 for the largest (oldest). 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%. Hence, a catch efficiency of 20% as set by the research platform for the Dutch regional water boards appears to be rather low.

For the beam trawls and seine, no information on the catch efficiency is available, other than that they probably have a lower catch efficiency than the electric dipping net, where the electric trawl probably has a higher efficiency than the normal trawl. The catch efficiency of the seine is probably the lowest of all gears. Since no research seems to have been done on the catch efficiencies of these gears, it was decided to exclude data collected with these gears from the analyses.

4.3.4 Habitat preference

The habitat preference is an important factor determining how to scale estimates from biomass in samples to estimates of the biomass for an entire water body. The simplest case to scale from samples to water body level is when catches in biomass per hectare are scaled linearly to water body surface area. This method assumes eels have no habitat preference. However, eels might prefer the littoral over the open water, and almost all 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 scaling was done using a correction fraction to account for differences in eel denstiy between the littoral zone and the open water.

How is eel distributed over a water body? Literature on this subject is scarce and focusses on the relation between eel density and distance to 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) both found a positive relationship between catch per unit effort and distance to shore. A recent, still 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 (Oleary personal communication). 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 (pers. comm. C. Briand).

4.4 Three scenarios

Taking into account the uncertainty regarding the catch efficiency of the electric dipping net and the habitat preference of eel, estimates of standing stock sizes were computed using three different scenarios (Table 4.4). In these scenarios, the catch efficiencies and habitat preferences are varied according to results from the literature (see 4.3). 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 assumed the effective wetted area to be 33%, 50% or 66% of the densities within 1.5 meters of shores/banks.

In scenario 1 the highest catch efficiencies (66%) and lowest proportion of eel in the offshore area compared to the inshore area are used, and this scenario will therefore lead to the lowest estimated standing stock of eel. In scenarios 2 and 3 the best guess estimates for catch efficiencies are used (20%), with the proportion of eel in the offshore area compared to the inshore area of 50% and 66% respectively. Scenario 3 will therefore lead to the highest estimates of standing stock.

Table 4.4. 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 areas") were assumed to be either 30%, 50% or 66% of densities within 1.5 meters of the shore/bank ("inshore areas").

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

4.5 Regionally managed water bodies

4.5.1 GIS data

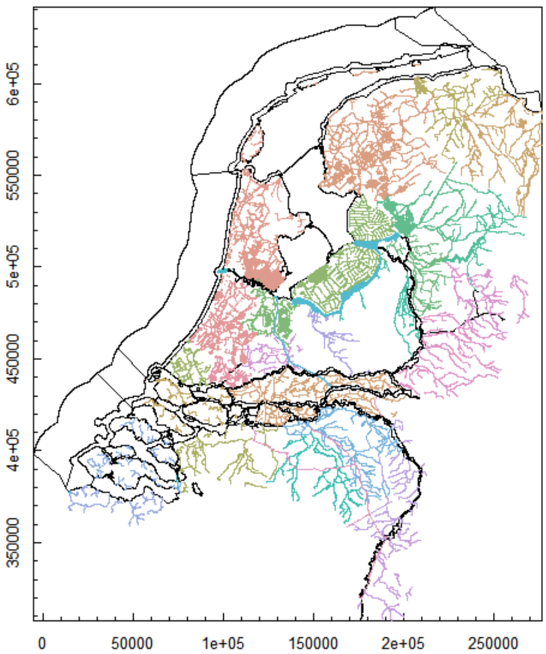

Figure 4.2. Water bodies included in the model, the colors represent different management regions, in black are the nationally managed waters

Of all water bodies the surface area is needed to scale the samples (presented in numbers/ha) to total surface area biomass. When available the surface area indicated by the Polygon map was used. For those water bodies only present on the Line map the average width per water body type was used to estimate the surface area (Table 4.5). This average width was obtained from the descriptors for each water body type provided for the WFD (STOWA 2007a, b). The surface area of the water bodies using the Polygon map is 917.6 ha, and the surface area of the water bodies using the Line map is 89.9 ha. The latter only contains a very small percentage of 0.5% of the total of ditches present in The Netherlands, thereby neglecting 33000 hectares of ditches (Table 4.2). An estimate of the eel biomass present in ditches will therefore be done based on the production estimates for that water type present in the WFD sampling (M1a and M2). For channels 10% and for streams 30% is not represented within the WFD. These two not-represented surface areas together consist of 218 ha, which is only 0.6 promille of the Dutch total water surface area and are therefore not corrected for.

Table 4.5. Water body types defined within the Water Framework Directive in The Netherlands and taken into account in this study of regionally managed waters ('niet-rijkswateren'). For those water types accounted for by the Line Map the average width used in the model to calculate the surface area is presented (STOWA 2007a,b). For those water bodies of a certain water type that are on the Polygon map the surface area from the map is used and hence no average width is needed. 'R' types are natural and flowing, while 'M' types are non-natural lakes, channels or ditches (STOWA 2007a, b).

Code water type	**Description**	**Average width (m)**
M1a/b	Buffered ditches	4
M2	Weakly buffered ditches	4
M3	Buffered regional canals	11.5
M6a/b	Large, shallow canals with/without shipping	15
M7a/b	Large deep canals with/without shipping	15
M8	Buffered fen ditches	4
M10	Fen canals	1.5
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	1.5
R5	Permanent, slow flowing, middle and lower reach, sand	6.5
R6	Slow flowing small river, sand-clay	16.5
R7	Slow flowing river, side channel, sand or clay	15
R12	Slow flowing middle and lower reach, bog	6.5
R13	Fast flowing upper reach, sand	1.5
R14	Fast flowing middle and lower reach, sand	6.5
R15	Fast flowing small river, pebble	16.5
R17	Fast flowing upper reach, calcium rich	1.5
R18	Fast flowing middle and lower reach, calcium rich	6.5

4.5.2 Survey data

Eel sampling within the regionally managed WFD waters was done following an EU certified protocol (STOWA Handboek Visstandbemonstering 2003) using electrofishing. Sampled water bodies are representative for water types defined within The Netherlands based on WFD regulation. Sample data were obtained from two of the companies hired by water boards to conduct WFD fish sampling. However, data of some regional water boards is missing in this analysis, for which averages of the other management areas were used instead. Similarly, some water types were not sampled, for which the average of all other water types was used instead. In addition, not all water types defined in the WFD per regionally managed area were sampled. These waters were included in the model by using estimates of sampled water types of other management areas.

Sampling occasions need to be 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 the water body names given at the time of the data collection were used. For regional waters, this results in 2325 electrofishing events that were used for the eel assessment. These cover the period 2006-2011 and their locations are presented in Figure 4.3.

The sampling data were presented as number of individuals and their length, measured in centimetre. Using the length-weight conversion function (see 4.3.1) the total weight was obtained per haul and per size-class.

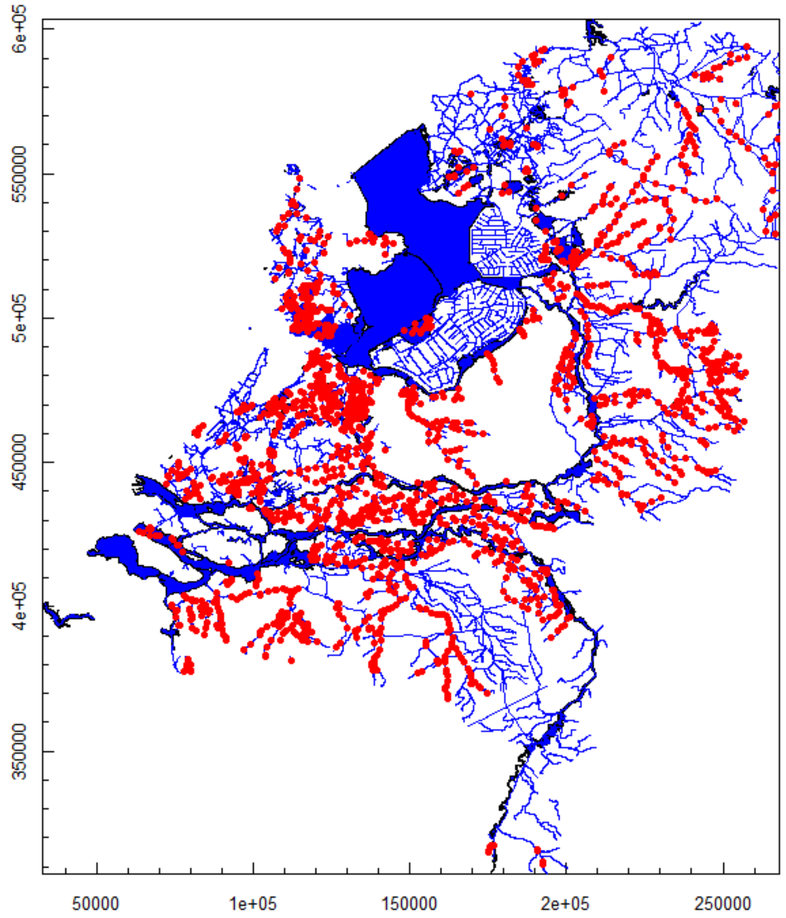

Figure 4.3. Locations of electrofishing sampling occasions used in the model.

4.5.3 Standing stock estimation

Standing stock density was calculated by dividing summed biomass by summed swept area on a per water type and regional water manager level. Mean densities per water type were multiplied with the surface area of that particular water type. This estimate of biomass was then corrected for catch efficiency (three scenarios). Eel production and biomass estimates are done based per water body type and regional water manager. By aggregating at the level of regional water manager is a way to include some spatial extent. In total 24 regional water boards were included in the sampling data out of 29 present in the WFD GIS files.

Within the WFD only 0.5% of the ditches in The Netherlands are represented, leaving an area of 330 km^2 unaccounted for (see Table 4.1; PBL 2010). To make an estimate of the eel stock in these ditches the production estimate of those ditches (type M1a and M2) that are present within the WFD was used to upscale to the total area.

From the yellow eel sampling individual length and weight were used to obtain the biomass per length class. In a similar manner used to assess yellow eel biomass, the biomass for the length classes from 30 cm and larger was estimated using the catch efficiency and the scaling of density with distance to shore. Using the length dependent sex ratio and silver eel fraction for males and females obtained from a market sampling campaign in Dutch waters the total biomass silver eel was calculated.

The eel stock was estimated for the smaller rivers and other water bodies in The Netherlands based on the WFD sampling protocol. Production was estimated per water body type and regional water manager (Table 4.6). 20 of the 25 water types were present in the sampling data. For those water types and regions data where missing, the average productivity was used for the eel biomass estimates and they can be found at the end of Table 4.7.

Table 4.6. Cross table eel standing stock biomass(kg/ha) based on the WFD sampling for each WFD type and regional water manager sampled. Note that these biomass values are not corrected yet for catch efficiency, nor distance of off-shore area. '-' indicates that these water types are not present in the regional managed areas. Only sampled water types and management areas are presented in this table.

	M10	**M14**	**M1a**	**M2**	**M20**	**M23**	**M27**	**M3**	**M6a**	**M6b**
Wetterskip Frysland	16.7	18.0	-	-	-	-	2.2	14.9	-	25.9
Groot Salland	-	-	0.9	-	-	-	-	4.6	-	-
Regge en Dinkel	-	-	0.6	-	-	-	-	1.3	-	-
Rijn en IJssel	-	-	-	-	-	-	-	3.7	-	-
Veluwe	-	18.2	-	-	-	-	-	1.2	6.7	-
Rivierenland	3.1	-	0.0	5.3	-	-	29.8	1.1	0.7	-
Vallei & Eem	-	-	-	-	-	-	-	0.0	-	-
Amstel, Gooi en Vecht	0.2	-	-	-	9.8	-	11.5	-	13.1	4.6
Hollands Noorderkwartier	7.5	-	-	-	30.4	-	-	11.6	-	5.2
Rijnland	0.0	0.0	-	-	20.4	0.0	18.4	0.0	-	13.1
De Stichtse Rijnlanden	0.0	-	5.1	-	-	-	-	1.0	-	13.7
Delftland	0.0	-	-	-	-	-	-	6.7	-	-
Rijkswater_19	-	-	-	-	-	-	-	0.2	23.3	-
Brabantse Delta	-	3.4	-	-	-	-	-	-	-	131.9
De Dommel	-	-	-	-	0.0	-	-	1.4	-	0.0
Hunze en Aa's	-	5.8	-	-	-	-	-	-	-	-
Reest en Wieden	-	-	-	-	-	-	4.3	6.9	-	-
Velt en Vecht	-	-	-	-	-	-	-	2.9	-	-
Zuiderzeeland	-	24.2	-	-	-	-	-	-	-	-
Aa en Maas	-	-	3.0	-	-	-	-	1.1	-	-
Schieland en de Krimpenerwaard	2.2	28.4	0.7	-	4.2	-	6.9	-	-	-
Peel en Maasvallei	-	-	-	-	-	-	-	-	-	-
Roer en Overmaas	-	-	-	-	-	-	-	-	-	-
Rijkswater_92	-	0.0	-	-	-	-	-	-	-	-

Table 4.6. (continued) *Cross table eel standing stock biomass(kg/ha) based on the WFD sampling for each WFD type and regional water manager sampled. Note that these biomass values are not corrected yet for catch efficiency, nor distance of off-shore area. '-' indicates that these water types are not present in the regional managed areas. Only sampled water types and management areas are presented in this table.*

	M7b	**M8**	**R12**	**R14**	**R18**	**R4**	**R5**	**R6**	**R7**	**R8**
Wetterskip Frysland	11.4	-	-	-	-	6.8	17.7	21.3	-	-
Groot Salland	-	6.3	-	-	-	-	-	19.7	40.4	-
Regge en Dinkel	-	-	-	-	-	-	1.7	8.3	-	-
Rijn en IJssel	-	-	-	-	-	-	1.6	2.9	-	-
Veluwe	-	-	-	-	-	-	4.9	-	-	-
Rivierenland	3.6	-	-	-	-	-	4.3	14.6	0.0	3.9
Vallei & Eem	6.7	-	-	-	-	4.7	1.3	6.9	4.7	-
Amstel, Gooi en Vecht	6.1	1.2	-	-	-	-	-	-	-	-
Hollands Noorderkwartier	11.0	-	-	-	-	-	-	-	-	-
Rijnland	-	0.0	-	-	-	-	-	-	-	-
De Stichtse Rijnlanden	2.5	-	-	-	-	-	-	-	-	-
Delftland	-	-	-	-	-	-	-	-	-	-
Rijkswater_19	-	-	-	-	-	-	-	-	-	-
Brabantse Delta	-	-	-	-	-	5.1	2.3	56.5	-	-
De Dommel	-	-	-	-	-	1.8	3.3	1.7	-	-
Hunze en Aa's	-	-	-	-	-	-	8.6	-	-	-
Reest en Wieden	-	-	5.0	-	-	-	3.6	9.3	-	-
Velt en Vecht	-	-	-	-	-	-	-	3.0	-	-
Zuiderzeeland	-	-	-	-	-	-	-	-	-	-
Aa en Maas	-	-	-	0.0	-	0.0	2.7	3.6	-	-
Schieland en de Krimpenerwaard	-	0.0	-	-	-	-	-	-	-	-
Peel en Maasvallei	-	-	-	-	-	-	7.6	-	-	-
Roer en Overmaas	-	-	-	-	8.7	-	-	-	-	-
Rijkswater_92	-	-	-	-	-	-	-	-	-	-

We present the eel biomass summed per water body type (Table 4.7). The eel biomass is estimated in the WFD waters using a catch efficiency of 0.2 and a factor of 0.5 to scale density in the non-shore areas is 2100 metric tonnes. Production per water type varies, with a minimum production of zero for R14, defined as middle and lower reaches of rivers that are fast flowing and with a sandy riverbed, and a maximum of 39.3 kg/ha for R7, defined as slowly flowing sandy rivers or side channel (Eem and Vecht-Zwartewater). The average production of all water types is 7.1 kg/ha. In the overall assessment (Chapter 7), retained catches are contrasted with standing stock biomass of eel with length over 30 cm (Chapter 1). When taking only eel larger than 30cm into account a biomass of 2027.4 tonnes is obtained.

The biomass in ditches not represented in WFD was estimated using production estimates of drainage ditches that were sampled (M1a and M2). Scaling biomasses with swept area, by dividing the sum of biomass with the sum of swept area, yields an average production of 2 kg/ha for ditches. Correcting for the 0.2 catch efficiency then results in 10 kg/ha or a biomass of 330 tonnes of eel in ditches. Summing the biomass in WFD waters and the biomass in the ditches not represented in WFD results in 2431 metric tonnes of yellow eel and silver eel combined.

Table 4.7. Eel standing stock biomass, total effective surface area, biomass and biomass corrected for catch efficiency presented per water body type. Biomasses are provided in metric tonnes using scenario 2. For those water types that were not sampled the overall average production of 7.1 kg/ha was used, presented at the end of the table.

Water Type	**Biomass (kg/ha)**	**Total Area (ha)**	**Biomass (tonnes)**	**Biomass, efficiency corrected (tonnes)**
M10	6.9	979.1	6.76	33.80
M14	10.2	18848.2	193.04	965.19
M1a	1.6	132.3	0.21	1.06
M2	5.3	8.8	0.05	0.23
M20	11.9	2255.1	26.78	133.89
M23	0.0	48.9	0.00	0.00
M27	7.3	11444.9	83.16	415.81
M3	4.8	2089.3	9.99	49.97
M6a	5.3	357.8	1.89	9.43
M6b	11.8	1037.0	12.26	61.32
M7b	7.0	1866.4	13.02	65.10
M8	0.9	647.9	0.58	2.89
R12	3.0	47.2	0.14	0.70
R14	0.0	11.5	0.00	0.00
R18	8.7	38.0	0.33	1.66
R4	2.0	73.0	0.15	0.74
R5	3.9	892.2	3.45	17.24
R6	7.9	1804.3	14.32	71.60
R7	39.3	1151.7	45.28	226.40
R8	3.9	12.2	0.05	0.24
M1b	7.1	0.1	0.00	0.00
M30	7.1	1188.5	8.42	42.09
M7a	7.1	7.7	0.05	0.27
R13	7.1	4.4	0.03	0.16
R15	7.1	22.0	0.16	0.78
R17	7.1	7.3	0.05	0.26
Subtotal		44975.9		2100.82
Ditches	2.0	33000	66	330
TOTAL		77975.9		2430.82

Silver eel estimates were based using biomass per length class as registered in the sampling campaign. Production of silver eel and biomass estimates were done identically as the production and biomass estimates for the all eel, as presented above. The total silver eel biomass in the regional waters is estimated at 343 metric tonnes (Table 4.8).

Table 4.8. Silver eel standing stock biomass, total effective surface area, biomass and biomass corrected for catch efficiency presented per water body type. Biomasses are provided in metric tonnes, using scenario 2. For those water types that were not sampled the overall average production of 1.3 kg/ha was used, presented at the end of the table.

Water Type	**Biomass (kg/ha)**	**Total Area (ha)**	**Biomass (tonnes)**	**Biomass, efficiency Corrected (tonnes)**
M10	1.1	979.1	1.09	5.44
M14	1.4	18848.2	26.38	131.90
M1a	0.5	132.3	0.07	0.35
M2	1.2	8.8	0.01	0.05
M20	2.1	2255.1	4.81	24.06
M23	0.0	48.9	0.00	0.00
M27	1.2	11444.9	13.19	65.95
M3	1.1	2089.3	2.20	11.01
M6a	1.1	357.8	0.39	1.93
M6b	1.2	1037.0	1.22	6.12
M7b	0.8	1866.4	1.46	7.32
M8	0.4	647.9	0.24	1.22
R12	0.7	47.2	0.03	0.17
R14	0.0	11.5	0.00	0.00
R18	2.4	38.0	0.09	0.46
R4	0.5	73.0	0.03	0.17
R5	0.8	892.2	0.73	3.67
R6	1.2	1804.3	2.22	11.11
R7	7.6	1151.7	8.77	43.83
R8	1.2	12.2	0.01	0.07
M1b	1.3	0.1	0.00	0.00
M30	1.3	1188.5	1.57	7.85
M7a	1.3	7.7	0.01	0.05
R13	1.3	4.4	0.01	0.03
R15	1.3	22.0	0.03	0.15
R17	1.3	7.3	0.01	0.05
Subtotal		44975.9		322.96
Ditches		33000		49.5
TOTAL		77975.9		342.76

Direct estimates of silver eel biomass for drainage ditches (M1a and M2) were very low and since these were based on a low sampling intensity (and thus uncertain length-frequencies), we used an estimate of 15% silver eel out of the total eel biomass (330 tonnes in ditches), resulting in a total silver eel biomass of 49.5 tonnes.

The biomass estimates can also be presented per regional water manager (Table 4.9). Large difference in eel biomass estimate exist between regional water boards which is due to the differences in density and surface area of the water types present within the water boards. Following the difference in total eel biomass also the silver eel biomass is not evenly distributed among regional managers. Due to averaging difference the total biomass estimates are slightly different when summed per water manager (2105 tonnes) or per water type (2100 tonnes).

Table 4.9. Eel standing stock biomass, total effective surface area, total eel biomass and silver eel biomass corrected for catch efficiency presented per regional water manager. Biomasses are provided in metric tonnes, using scenario 2. For the regional waters that were not sampled the overall average production of 8.5 kg/ha was used. Ditches other than accounted for with M1a and M2 are not accounted for.

Water board	**Biomass (kg/ha)**	**Total Area (ha)**	**Eel biomass**	**Silver eel biomass**
Vallei & Eem	3.7	109.1	2.03	0.47
Amstel, Gooi en Vecht	9.5	4255.9	201.99	44.45
Hollands Noorderkwartier	11.4	2227.4	126.59	15.89
Rijnland	13.7	2279.0	155.84	23.00
De Stichtse Rijnlanden	6.0	141.5	4.25	0.47
Delfland	3.6	183.0	3.32	0.95
Rijkswater_19	2.4	424.7	5.08	0.60
Wetterskip Frysland	14.5	8338.1	603.51	75.63
Brabantse Delta	35.3	215.7	38.09	6.58
De Dommel	2.0	252.3	2.56	0.61
Hunze en Aa's	6.0	1240.3	36.95	3.30
Reest en Wieden	4.4	5744.8	127.38	14.27
Velt en Vecht	2.9	1318.4	19.12	1.69
Zuider-zeeland	19.5	4222.5	411.33	66.60
Aa en Maas	2.0	325.3	3.25	0.60
Schieland en de Krimpenerwaard	5.5	620.2	17.10	3.34
Groot Salland	35.4	1277.3	226.16	43.83
Regge en Dinkel	3.6	189.6	3.38	0.71
Peel en Maasvallei	4.6	124.2	2.87	0.43
Roer en Overmaas	6.8	82.4	2.81	0.77
Rijn en IJssel	2.3	341.0	4.00	0.93
Veluwe	4.0	154.2	3.12	1.18
Rivierenland	4.2	501.7	10.61	2.59
Rijkswater_92	0.0	8196.4	0.00	0.00
Noorderzijlvest	8.5	1572.5	66.64	11.77
Rijkswater_86	8.5	428.8	18.17	3.21
Rijkswater_89	8.5	2.7	0.12	0.02
Rijkswater_90	8.5	153.6	6.51	1.15
Rijkswater_91	8.5	47.0	1.99	0.35
Zeeuwse Eilanden	8.5	6.2	0.26	0.05

A comparison of eel biomass estimates in the regionally managed waters for the period 2006-2008 and 2009-2011 was made as in 2008 the eel management plan became effective. Estimates of 1683 tonnes (2006-2008) and 2265 (2009-2011) were obtained using scenario 2. These estimates imply an increase in eel biomass during that period. However, differences in the spatial coverage of the data points available per period might be underlying this large difference. The number of samples differs, with 910 vs 1415 sampling events before and after 2008. In addition, more water types were sampled in the period 2009-2011 (7 unsampled types) than in the period 2006-2008 (11 unsampled types). These differences in sampling effort makes a comparison between these periods unreliable.

4.5.4 Discussion

In this biomass assessment choices and assumptions were made based on data availability, time constraints and practicalities. For example, not all water bodies were sampled within the WFD sampling program at this point, and not all regional water boards provided data. In addition, in the obtained data set nearly 30% of the samples could not be linked to a water body, and these were excluded from analysis. This mismatch might be due to measurement error in GPS equipment or errors in data entry.

A large uncertainty in catch efficiency and the factor to scale density with distance to shore exists, which was dealt with by using different scenarios to get an idea on the effect of the uncertainties on the biomass estimates.

For future studies it is recommended that data of all regional water boards are included. These data that are stored in a data-scheme called Piscaria, includes also information on habitat. The WFD eel sampling data were not corrected for habitat in the current assessment but could be done in the following evaluation in 2015.

4.6 Nationally managed water bodies

4.6.1 Survey in the main rivers

Within the governmental survey program "Biologische Monitoring Zoete Rijkswateren", fish species in the main Dutch rivers are monitored yearly. Among others, rivers are sampled using research vessels (the "Actieve Monitoring van de Zoete Rijkswateren" survey, e.g., van Kessel et al., 2010). 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. Sampling takes place in autumn and early spring. There are six focus regions that have been sampled since 1992. A region is always sampled in the same months, but different regions are sampled in different months. There are also extra regions for which data is only available for one of the three years that are considered here. See Figure 4.4 for the classification of regions and Table 4.10 for an overview of survey details per region. The large lakes IJsselmeer and 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 Veluwerandmeren were only sampled with other types of gears (normal beam trawl, seine and fyke), the efficiency of which are even lower and less certain. Thus, the density estimate for these lake are also derived from the dynamic population model (see Chapter 3).

Density per haul is determined (kg/ha), using eel length and a length-weight conversion factor (see paragraph 2.3). 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 4.10 for the density estimates per region. Note that catch efficiency has not been corrected for yet in this table.

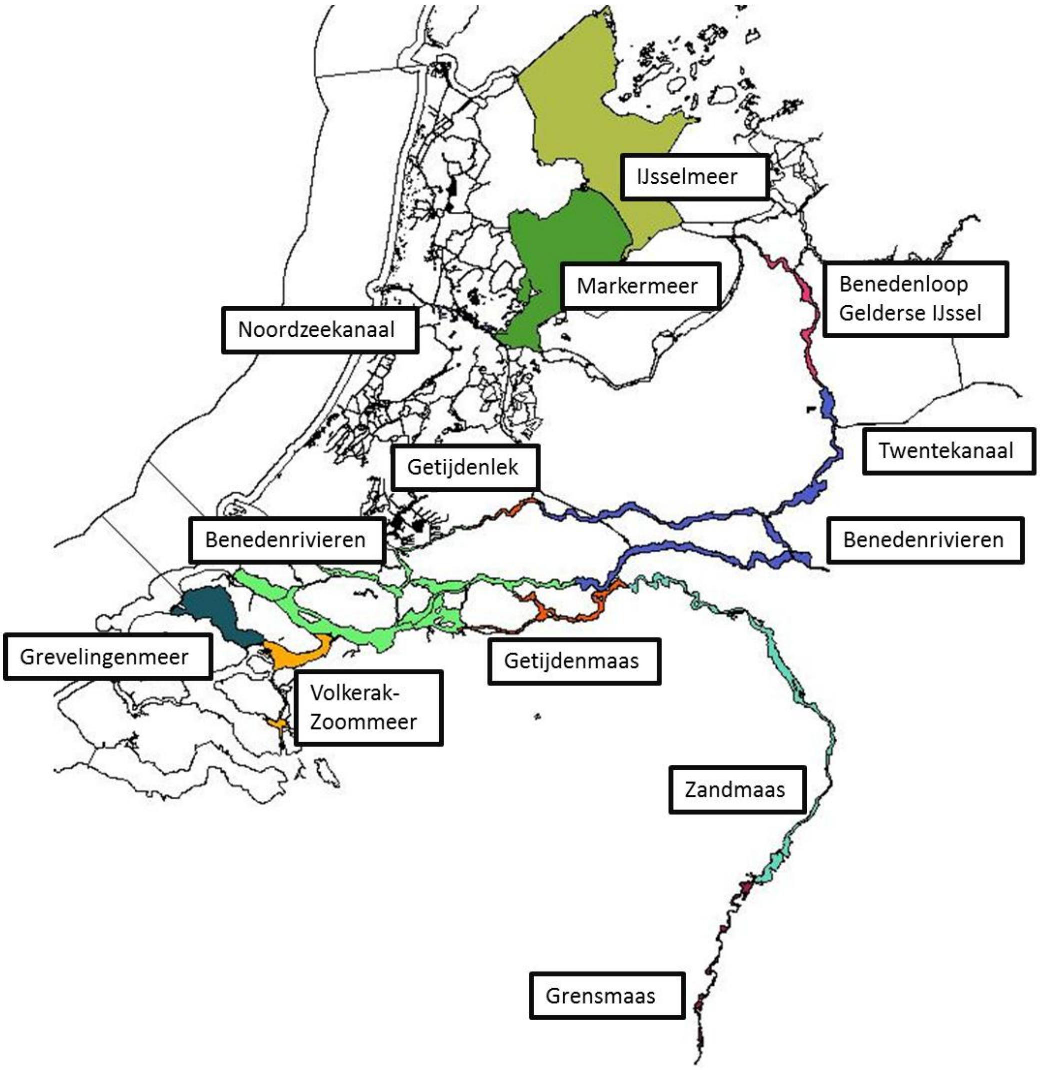

Figure 4.4 Classification of the main rivers and large lakes. Regions are represented by different colours.

Table 4.10. Densities per river region and type of water (main waterway or connected water body). Sampled years = the years in which a region has been sampled, where all = 2008+2009+2010. Sampled months = the months in which a region is sampled. No. samples = the total number of samples collected in the sampled years. Density is based on data collected using an electric dipping net in the riverbanks. No correction for catch efficiency of the gears is made yet. NA = not available.

Region	**Water**	**Sampled years**	**Sampled months**	**No. samples**	**Density riverbank (kg/ha)**
Benedenloop Gelderse IJssel	main	all	2, 3	15	0.39
	connected			4	5.37
Benedenrivieren	main	all	9, 10, 11	91	7.08
	connected			4	2.90
Gelderse Poort	main	all	3, 4	97	0.28
	connected			43	0.11
Getijdenlek	main	all	10, 11	25	3.39
	connected			5	0.96
Getijdenmaas	main	all	11	25	2.10
	connected			11	5.55
Grensmaas	main	all	5	33	35.46
	connected			3	22.11
Twentekanaal	main	2009	2	2	0
Zandmaas	main	all		39	6.57
	connected			18	21.65
Volkerak-Zoom		2008	3	9	11.49

4.6.2 GIS data

Three types of geographical information are collected. The surface area (ha) and bank length (km) of the rivers and lakes are calculated (Table 4.11) using GIS-data (the Ecotopenkaart of Rijkswaterstaat). For the rivers, extra information on bank length was collected (Table 4.11). In some parts of the rivers, bank length is significantly larger than river length because of groynes ("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.

Table 4.11. Surface area, river/lake length and bank length per river region and water type (main waterway or connected waterbody). 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/lake 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
Noordzeekanaal	main	2160	235		235
Twentekanaal	main	396	135		135
Zandmaas	main	2043	305		305
	connected	1413	160		160
Volkerak-Zoom		4814	171		171

4.6.3 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 and Noordzeekanaal. 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. For the lake Grevelingen no electric dipping net data are available and no biomass estimates can be made. For the Noordzeekanaal electric dipping net data is not available in the national survey program. However, electric dipping net data is available from another survey program (the Waterframework Directive survey program, see section 4.6 for more details). Further calculations for this regions are similar to the other river regions.

Biomass of silver eel and of all eel larger than 30 cm is estimated according to scenario 2 (Table 4.12). No information on the ratio yellow eel - silver eel in the surveys is available and a conversion key (see Table 4.3) is used to determine the biomass of silver eel.

Table 4.12. Total biomass, biomass of silver eel and biomass of eel larger than 30 cm tonne kg) per river region, estimated according to scenario 3.

Region	**Total**	**Silver eel**	**>30cm**
Benedenloop Gelderse IJssel	4.39	0.65	4.26
Benedenrivieren	339.93	31.92	315.94
Gelderse Poort	4.18	0.37	4.06
Getijdenlek	4.50	0.26	3.88
Getijdenmaas	17.45	4.23	17.19
Grensmaas	121.15	26.84	119.35
Noordzeekanaal	45.30	6.80	45.30
Twentekanaal	0.00	0.00	0.00
Zandmaas	112.13	30.81	111.95
Volkerak-Zoom	139.10	37.15	131.63

For scenario 2, biomass of the period 2008-2010 is also compared to the period 2005 - 2007 (Table 4.13). For the regions that have been sampled in both years, the total biomass has on average reduced by 66%. This reduction is caused mostly by a strong decline in the 'Benedenrivieren' (downstream areas of the main rivers), which in turn is mostly caused by relatively high densities in 2005.

Table 4.13. Total biomass (tonne kg) per river region, for the current period (2008-2010) and the previous period (2005-2007). Biomass estimated with scenario 2 (see main text for methodology).

Region	**Total biomass 2005-2007**	**Total biomass 2008-2010**
Benedenloop Gelderse IJssel	14.72	4.39
Benedenrivieren	702.74	339.93
Gelderse Poort	25.42	4.18
Getijdenlek	4.75	4.50
Getijdenmaas	75.21	17.45
Grensmaas	36.42	121.15
Noordzeekanaal	NA	45.30
Twentekanaal	NA	0.00
Zandmaas	NA	112.13
Volkerak-Zoom	NA	139.10
Grevelingenmeer	NA	NA
IJsselmeer	356.67	325.00
Markermeer	100.07	57.73

NA = Not available.

4.6.4 Discussion

There are some shortcomings and uncertainties in the methodology used.

Different river 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 densities caught.

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.

The applied division of the rivers into regions is not always ecologically the most logical choice. Averaging over such a scale might lead to under/over-estimating the true densities.

Similar to the regionally managed waters the eel densities are not corrected for habitat in the current assessment

4.7 The large lakes

For Lakes IJsselmeer and Markermeer, independent estimates of fishing mortalities obtained by fitting the dynamic eel population model to stock survey data are available. These can be combined with reported landings eels to estimate standing stocks. Estimates of fishing mortalities in these lakes in recent years varied from F=0.1 (Lake IJsselmeer) to F=0.3 (Lake Markermeer) (see Chapter 3). Here, we have computed stock sizes using F=0.1 and F=0.2 as tentative estimates of fishing mortalities. The same fishing mortality values were used for lake Grevelingenmeer and the lakes Veluwerandmeren. Estimates of fishing mortalities refer to yellow eel mortalities only. Therefore,the reported landings of yellow eel were split intoyellow eel and silver eel biomass, using market sampling data. The split was achieved by combining length-frequencies of the landings with estimated proportions of silver eel per length-class in the landings. Estimated biomass of yellow eel (Table 4.14) and silver eel (Table 4.15) are presented for the fishing mortalities F=0.1 and F=0.2. The silver eel stock in these lakes is assumed to be 15% of the estimated yellow eel stock.

Table 4.14. Estimated standing stock of yellow eel with body lengths over 30 cm, for lakes IJsselmeer, Markermeer, Grevelingen and Veluwerandmeren. The standing stock is estimated by dividing the estimated landings of yellow eel by the assumed fishing mortality: Landings/-log(1-F). Landings of yellow eel were estimated by splitting the total retained catch into yellow eel and silver eel, using biological market sampling data.

	Landings (tonnes)	**Estimated standing stock (tonnes), assuming**	
		F=0.1	**F=0.2**
Lake IJsselmeer	86	814	384
Lake Markermeer	62	586	277
Lake Grevelingen	82	77	37
Lake Veluwerandmeren	9	88	41

Table 4.15. Estimated silver eel biomass for lakes IJsselmeer, Markermeer, Grevelingen and Veluwerandmeren. The silver eel stock in these lakes is assumed to be 15% of the estimated yellow eel stock. (Table 4.14).

	Estimated production (tonnes), assuming	
	F=0.1	**F=0.2**
Lake IJsselmeer	122	57
Lake Markermeer	88	42
Lake Grevelingen	12	5
Lake Veluwerandmeren	13	6

As a comparison to these estimates for the lakes IJsselmeer and Markermeer, biomass estimates based on the survey data were also calculated. As mentioned in section 4.6, an electric beam trawl is the main gear used in the survey in these two lakes, and the catch efficiency of this type of gear is not known. Here, we set the catch efficiency at 20% (similar to scenario 2 for the electric dipping net) and calculated the biomass of these two lakes. Electric beam trawl data were used for the density estimates for the open water. In the shore, samples were habitat stratified (sand, reed and rocks). The sand stratum is sampled with a seine, and the reed and rocks strata are sampled with an electric dipping net. Average density estimates were scaled according to strata surface. The methodology otherwise was similar to that for the other nationally managed waters. For the IJsselmeer, the estimated amount of yellow eel above 30cm was 223 tonnes and for silver eel 36 tonnes. For the Markermeer, the estimated amount of yellow eel above 30cm was 42 tonnes and for silver eel 13 tonnes. These estimates are lower than those derived from the dynamic eel population model with both F=0.1 and F=0.2.

4.8 Estimates of standing stock: 3 scenarios

Here, we report the estimates of the total standing stock of yellow eels with body lengths over 30 cm. This is done because the standing stock estimates will be compared to estimated landings of yellow eel (Chapter 7). To estimate fishing mortalities, it will be assumed that the fisheries on eel is fully selective from 30 cm onwards, whereas eels with body lengths under 30 cm are assumed to have a fishing mortality of 0 (the legal minimum landing size of eel in The Netherlands is 28 cm).

Estimates were made for the three scenarios with different assumed catch efficiencies and distributions of eel (see Table 4.4). For the large lakes with independent estimates of fishing mortalities (IJsselmeer and Markermeer) and the lakes without survey data (Grevelingen and Veluwerandmeren), we used the estimates presented in Tables 4.14 and 4.15. For scenarios 1 and 2, estimates of fishing mortalities of F=0.2 were used, and for scenario 3 an estimate of F=0.1. Estimates of yellow eel standing stock in all water bodies except lakes IJsselmeer, Markermeer, Grevelingenmeer and Veluwerandmeren, were made by subtracting the estimated silver eel production from the total estimated standing stock of eel over 30 cm in length. For the aforementioned large lakes, silver eel production was not deducted from the total biomass, because only yellow eel retained catches combined with an estimate of yellow eel fishing mortalities were used to estimate yellow eel standing stock (>30 cm) directly (Table 4.14). The final estimates, as used in the assessment of lifetime anthropogenic mortalities (presented in Chapter 5), are given in Tables 4.16 (yellow eel) and 4.17 (silver eel).

Table 4.16. Estimated standing stock of eel with body lengths over 30 cm, for three scenarios which differ in assumed catch efficiencies of fishing gears and fishing mortalities. The estimates were made by scaling densities of eels over 30 cm in length, as measured in fisheries-independent surveys, to total wetted areas of water bodies. To estimate the standing stock of yellow eels, the estimated production of silver eel was subtracted from the total biomass. Estimates are provided in metric tonnes. See main text for the specifications of the three scenarios.

	Scenario		
	1	**2**	**3**
Lake IJsselmeer	384*	384*	814+
Lake Markermeer	277*	277*	586+
Other nationally managed water bodies	151	614	724
Lake Grevelingen	37*	37*	77+
Veluwerandmeren	41*	41*	88+
Drainage ditches	85	280.5	280.5
Regionally managed water bodies	351	1697	2204
Total	1326	3331	4774

* Estimate made assuming F=0.2. + Estimate was made assuming F=0.1.

Table 4.17. Estimated production of silver eel, for three scenarios which differ in assumed catch efficiencies of fishing gears and fishing mortalities. Estimates are provided in metric tonnes. See main text for the specifications of the three scenarios.

	Scenario		
	1	**2**	**3**
Lake IJsselmeer	57*	57*	122+
Lake Markermeer	42*	42*	88+
Other nationally managed water bodies	34	139	164
Lake Grevelingen	5*	5*	12+
Lake Veluwerandmeren	6*	6*	13+
Drainage ditches	15	50	50
Regionally managed water bodies	66	321	417
Total	225	620	866

* Estimate made assuming F=0.2. + Estimate was made assuming F=0.1.

5. Comparison of static spatial model and capture-mark-recapture estimates of silver eel biomass

In this chapter, the estimates of the biomass of silver eel based on the static spatial model (Chapter 4) are compared to estimates of silver eel biomass resulting from capture-mark-recapture experiments. The capture-mark-recapture experiments provide independent estimates of silver eel biomass, based on a different methodology and data set. If the estimates from the two different methods are similar, this would give confidence in the obtained estimates of silver eel biomass. The comparisons are not used in the EMP evaluation but are only used as an independent check on the order of magnitude of the estimates obtained from the static spatial model.
Capture-mark-recapture estimates typically apply to a specific location with a potentially large hinterland of wetted area from which migrating silver eels originate. In order to achieve the estimated biomass of silver eel which will migrate per year through the given location based on the static spatial model, three estimates need to be made: (a) the water bodies that are deemed connected to the given location, (b) the proportion of silver eels per water body that will migrate through the given location, (c) the mortality due to barrieres during migration to the given point. The estimates of both (b) and (c) are educated guesses. Mortality estimates were made based on results from Chapter 6. The proportion migration was based on an educated guess. More precise estimates for the latter could be made based on information on flowrates, but this was not available for the present report.

Below, we make four comparisons, for silver eel migration through:
1) The North Sea canal
2) The waterboard Wetterskip Fryslan
3) The river Meuse; just upstream of hydropower plant Linne
4) The river Meuse; just upstream of hydropower plant Alphen

The results from this comparison are used only as an independent check on the quality of the estimates, but are not further used in the stock assessment (Figure 5.1).

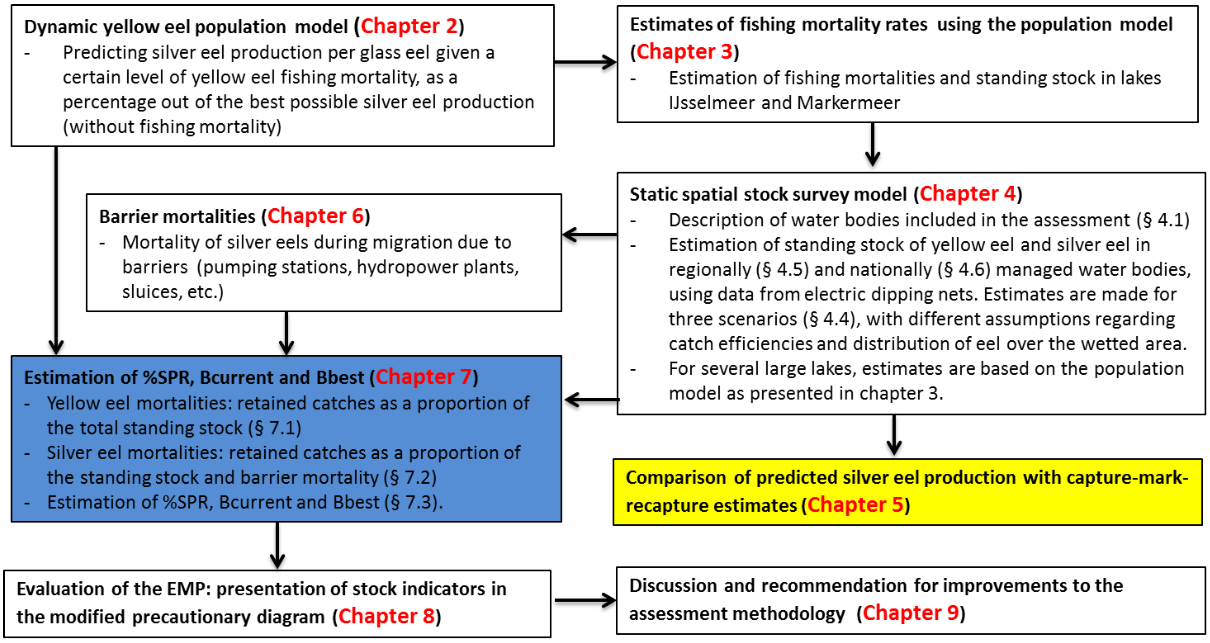

Figure 5.1 *A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report (with reference to chapters and key paragraphs therein). Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.*

5.1 Silver eel migration through the North Sea Canal

Winter (2011) compiled a meta-analysis of a number of studies for estimating silver eel escapement using both tagging and fyke net captures behind sluices and pumping stations near the town of Velsen in The North Sea canal. Winter (2011) estimated 70.000-100.000 individuals migrating through the North Sea canal, with an estimated biomass of 60-85 tonne (mean weight per migrating eel of 850 g; the vast majority of silver eels were found to be female). It is thought that silver eel from the following management areas and water bodies eventually have to pass through the North Sea canal where a large pumping station is in operation in combination with ships locks and sluices. The exact routes that silver eels who end up in the North Sea canal may have followed are unknown, but an educated guess yields the following: approximately 25% of all the silver eel from water board Hollands Noorderkwartier and from all ditches, 50% from Zuiderzeeland, 75% from the water boards Rijnland, Stichtse Rijnlanden, Vallei & Eem, Veluwerandmeren and Lake Markermeer, and 100% from Amstel, Gooi en Vecht and the North Sea Canal itself. Estimates of the biomass of silver eel ending up in the North Sea canal using this migration model are given in Table 5.1 and range from 58 to 208 tonnes for the three scenarios of the yellow eel spatial model.

Table 5.1. Estimated biomass of silver eel passing through the North Sea canal, for three scenarios which differ in assumed catch efficiencies of fishing gears and fishing mortalities (see Chapter 3). Silver eels from various water boards and from Lakes IJsselmeer and Veluwerandmeren are believed to have migration routes that eventually pass though the North Sea Canal. The proportion of the silver eel stock out of the total from a water board or lake with migration routes through the North Sea canal is given by the column 'p'. The mortality due to barriers during migration before arrival in the North Sea canal is given by column 'M'. The estimated production (kg/ha) of silver eel in the water boards and Lakes are given in columns 1, 2 and 3, for the three scenarios. The resulting estimated biomass (tonnes) of silver passing through the North Sea canal for the three scenarios are given in columns $\hat{1}$, $\hat{2}$ and $\hat{3}$.

				Scenario			**Scenario**	
	p	**M**	**1**	**2**	**3**	$\hat{1}$	$\hat{2}$	$\hat{3}$
Hollands Noorderkwartier	0.25	0.1	3.3	15.9	20.6	0.7	3.6	4.6
Rijnland	0.75	0.1	4.8	23.0	29.9	3.2	15.5	20.2
Amstel Gooi en Vecht	1	0.1	9.2	44.5	57.7	8.3	40.0	52.0
Zuiderzeeland	0.5	0.1	13.7	66.0	85.7	6.2	29.7	38.6
Stichtse Rijnlanden	0.75	0.1	0.1	0.5	0.6	0.1	0.3	0.4
Vallei & Eem	0.75	0.1	0.1	0.5	0.6	0.1	0.3	0.4
Markemeer	0.75	0	42.0	42.0	88.0	31.5	31.5	66.0
Veluwerandmeren	0.75	0	6.0	6.0	13.0	4.5	4.5	9.8
Ditches	0.25	0.41	15	49.5	49.5	2.2	7.3	7.3
North Sea Canal	1	0	1.4	6.8	8.8	1.4	6.8	8.8
Total						58.2	139.6	208.0

5.2 Silver eel migration through Waterboard Wetterskip Fryslan

In a pilot project for decentralised eel management (Witteveen & Bos et al., 2010c), eel fishers in the Water board Wetterskip Fryslan have tentatively estimated eel escapement in October/November 2010 using tagging studies with recaptures before exit points (pumping stations) for escapement to the sea. These data were extrapolated to number of escapements per year: 142251 (126212 - 164290) females and 115321 (81353-149291) males (pers. comm. Tim Vriese, with a note that the uncertainty range does not take the uncertainty of the extrapolation into account). If we combine these estimates with estimated mean weight of female and male silver eels of 750 and 150 grams respectively, we obtain estimates of biomass of 121 (105-142) tonnes of silver eel. An educated guess is that the silver eel arriving at these exit points originate almost exclusively from water bodies in this same water board. Estimates of the biomass of silver eel arriving at the exit points in this water board, using the model as presented in this report are given in Table 5.2, and range from 17 to 105 tonnes.

Table 5.2. Estimated biomass of silver eel passing arriving at exit points in the water board Wetterskip Fryslan, for three scenarios which differ in assumed catch efficiencies of fishing gears and fishing mortalities (see Chapter 3). All of the silver eels which are estimated to be produced in water bodies in this water board are thought to arrive at one of the exit points. The proportion of the silver eel stock out of the total from the water bodies with migration routes through the water board is given by the column 'p'. The mortality due to barriers during migration before arrival at the exit points is given by column 'M'. The estimated production (kg/ha) of silver eel in the water bodies are given in columns 1, 2 and 3, for the three scenarios. The resulting estimated biomass (tonnes) of silver arriving at the exit points in the water board for the three scenarios are given in columns $\hat{1}$, $\hat{2}$ and $\hat{3}$.

				Scenario			**Scenario**	
	p	**M**	**1**	**2**	**3**	$\hat{1}$	$\hat{2}$	$\hat{3}$
Wetterskip Fryslan	1	0	15.6	75.6	98.2	15.6	75.6	98.2
Ditches	0.25	0.41	15	49.5	49.5	2.2	7.3	7.3
Total						17.8	82.9	105.5

5.3 Silver eel migration through hydropower plant Linne (River Meuse)

Winter et al. (2007) provide estimates from a capture-mark-recapture experiment using transponder are available for a recapture-location at the Linne hydropower plant which is situated upstream on the river Meuse (near the Belgian border). Estimated numbers of silver eels (found to be predominantly female) range from 30000 - 120000 with a best estimate of 55000. We estimate the biomass (0.75 kg per individual; most individuals were found to be female) at between 22.5 and 90 metric tonnes, with a best estimate of 41.3 tonnes. Estimates of silver eel production in the water bodies upstream of the hydropower plant Linne from the yellow eel model presented in this report are given in Table 5.3 and range from a total of 6.3 to 31.7 tonnes. It has to be noted that the estimates in this report are valid for the period 2008-2011, whereas the estimates based on capture-mark-recapture are valid for 2002, and include silver eel from Belgian and German parts of the river Meuse. The amount of silver eel migrating downstream from abroad is estimated at 16 tonnes from Belgium and 0.5 tonnes from Germany (Vlietinck et al., 2007).

Table 5.3. Estimated biomass of silver eel passing arriving hydropower plant Linne in the river Meuse, under three scenarios (see Chapter 3). In the eel assessment model presented in this report there are two water bodies in The Netherlands in the river Meuse upstream of the power plant. The proportion of the silver eel stock out of the these two water bodies with migration routes through to the hydropower plant is given by the column 'p'. The mortality due to barriers during migration before arrival at the hydropower plant is given by column 'M'. The estimated production (kg/ha) of silver eel in the water bodies are given in columns 1, 2 and 3. The resulting estimated biomass (tonnes) of silver arriving at the hydropower plant for the three scenarios are given in columns $\hat{1}$, $\hat{2}$ and $\hat{3}$. The water body identifier codes are the identifier codes for water bodies in the OWAGV shapefile of polygons of Water Framework Directive water bodies.

Water body identifier			**Scenario**			**Scenario**		
	p	**M**	**1**	**2**	**3**	$\hat{1}$	$\hat{2}$	$\hat{3}$
NL91dlbMM1-2	1	0	2.1	8.7	10.3	2.1	8.7	10.3
NL91dlbMM3-4	1	0	4.4	18.1	21.4	4.4	18.1	21.4
Total						6.3	26.8	31.7

5.4 Silver eel migration through hydropower plant Alphen (River Meuse)

Winter et al. (2007) provide estimates from a capture-mark-recapture experiment using transponder are available for the recapture-location at the Alphen hydropower plant which is situated downstream from the hydropower plant Linne (discussed above) on the river Meuse. Estimated numbers of silver eels (found to be predominantly female) ranged from 80000 - 300000 with a best estimate of 145000. We estimate the biomass (0.75 kg per individual; most individuals were found to be female) at between 60 and 225 metric tonnes, with a best estimate of 109 tonnes. Estimates of silver eel production in the water bodies upstream of the hydropower plant Linne (which pass through with an assumed mortality of 25%) and downstream of Linne but upstream of Alphen from the yellow eel model presented in this report are given in Table 5.4 and range from a total of 13.5 to 64.8 tonnes. It is thought that silver eel from the following management areas eventually have to pass through the hydropower plant at Alphen: approximately half of all the silver eel from water board Aa and Maas and all silver eel from the water boards Peel and Maas, and Roer and Overmaas. It has to be noted that the estimates in this report are valid for the period 2008-2011, whereas the estimates based on capture-mark-recapture are valid for 2002 and include silver eel from Belgian and German parts of the river Meuse. The amount of silver eel migrating downstream from abroad is estimated at 16 tonnes from Belgium and 0.5 tonnes from Germany (Vlietinck et al., 2007).

Table 5.4. Estimated biomass of silver eel passing arriving hydropower plant Linne in the river Meuse, under three scenarios (see Chapter 3). In the eel assessment model presented in this report there are two water bodies in The Netherlands in the river Meuse upstream of the power plant. The proportion of the silver eel stock out of these two water bodies with migration routes through to the hydropower plant is given by the column 'p'. The mortality due to barriers during migration before arrival at the hydropower plant is given by column 'M'. The estimated production (kg/ha) of silver eel in the water bodies are given in columns 1, 2 and 3. The resulting estimated biomass (tonnes) of silver arriving at the hydropower plant for the three scenarios are given in columns $\hat{1}$, $\hat{2}$ and $\hat{3}$.

Water body identifier				**Scenario**			**Scenario**	
	p	**M**	**1**	**2**	**3**	$\hat{1}$	$\hat{2}$	$\hat{3}$
NL91dlbMM1-2	1	0.15	2.1	8.7	10.3	1.8	7.4	8.7
NL91dlbMM3-4	1	0.15	4.4	18.1	21.4	3.8	15.4	18.2
NL91dlbWR1-6NM1	1	0	5.6	22.8	27.0	5.6	22.8	27.0
NL91dlbNM1-2	1	0	1.9	7.9	9.3	1.9	7.9	9.3
Peel and Maas	1	0.1	0.1	0.8	0.5	0.1	0.7	0.5
Roer and Overmaas	1	0.1	0.2	0.4	0.9	0.2	0.4	0.8
Aa and Maas	0.5	0.1	0.2	0.6	0.7	0.1	0.3	0.3
Total						13.5	54.9	64.8

5.5 Discussion

The estimates for the biomass of silver eel based on the static spatial model are compared to independent estimates of silver eel biomass resulting from capture-mark-recapture experiments. In order to compare these data, various assumptions needed to be made (see section 5.1).
The experimental data from the Wetterskip Fryslan water board and North Sea canal are relatively recent and the contribution of silver eel from Belgium and Germany at these locations can be considered to be small to absent, compared to the total production of eel from The Netherlands. Both the capture-mark-recapture (CMR) estimates from the Fryslan water board and The North sea canal overlap with the yellow eel survey model estimates (in metric tonnes): for the North Sea Canal, CMR = 60-85 and model = 58-208, for the Fryslan water board, CMR = 105 - 142 and model = 18-106. The overlap for the Fryslan water board is marginal, but as noted above, the uncertainty of the extrapolation of numbers for October-November to year-round in the CMR study has not been taken into account in this estimate and will lead to a larger range of uncertainty. The experiments at the hydropower plants Linne and Alphen in the river Meuse are done in a different time period than that used for the 2011 model estimates and the eel passing through will include eel from Belgium and Germany. When taking estimates of 'foreign' eel into account, also these two locations deliver estimates that overlap with the estimates based on model data: for Linne, CMR = 23-90 and model = 23-48, for Alphen, CMR = 60-225 and model = 30-82.

The estimates derived from both the CMR experiments and the spatial model have uncertainty ranges that are too large for a detailed comparison (i.e., should we use scenario 1, 2 or 3?). However, the overlap of the ranges of biomass estimates does give confidence that the spatial model produces biomass estimates of at least the right order of magnitude.

6. Mortality during silver eel migration due to barriers

In this chapter we describe the methodology and data on which estimates of barrier mortality are based. These estimates are based upon the spatially explicit estimates of silver eel production as presented in Chapter 5. 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 in Figure 6.1).

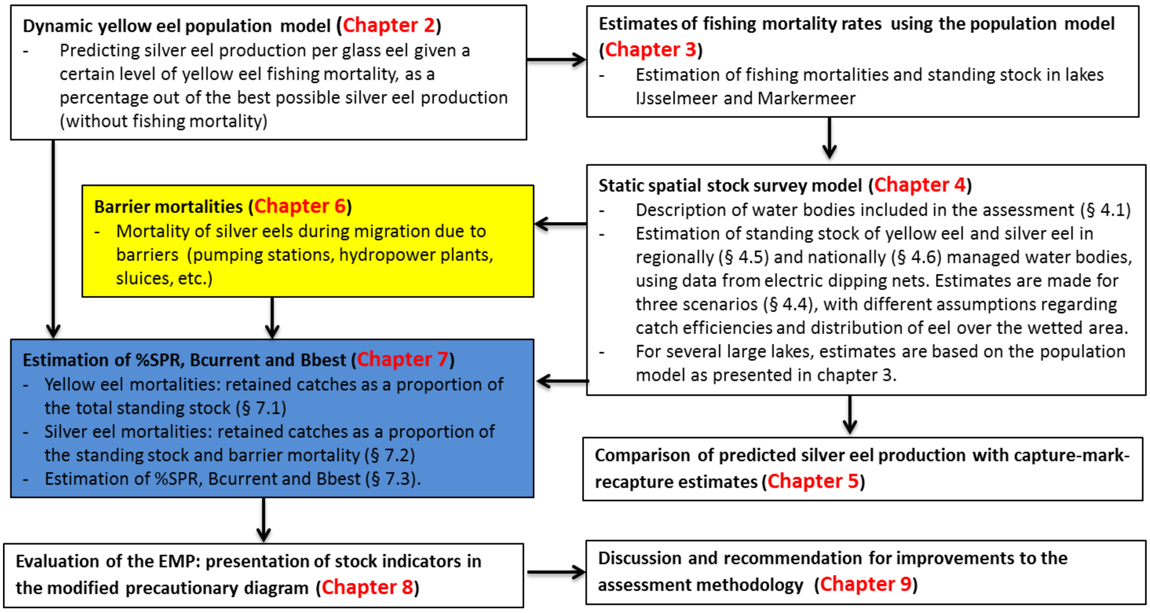

Figure 6.1. A flow diagram representing the key steps in the stock assessment methodology, and the structure of this report (with reference to chapters and key paragraphs therein). Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

6.1 Model for estimating barrier mortality

Estimating the mortality of silver eels during their migration from inland water bodies to the sea, due to barriers such as pumping stations and hydropower plants is a challenging task. The Netherlands consists of a complex network of interlinked large and small water bodies, most of which contain eel. There are thousands of pumping stations and even more other potential barriers that could cause mortalities for silver eels during their migration to the sea.

To construct a model for estimating the mortalities of silver eels caused by these barriers knowledge on the following key processes is necessary:

1) Mortalities that eels experience during passage of barriers
2) Thebiomass of silver eels produced in different inland water bodies, that give the starting positions of silver eels.
3) Routes of silver eels from inland water bodies to the sea, and the barriers eels have to pass on these routes

An attempt was made to compute migration routes of silver eels from individual water bodies to the sea via a number of escapement points, while keeping track of the barriers that an eel passes, based on the shortest distance between linked water bodies in the WFD shapefiles (Chapter 4). Using the shortest distance as the main rule for choosing migration routes made it practically possible to construct computer algorithms to compute such routes. However, the computed shortest routes turned out to be unrealistic in many cases, and these results are not further used. A full description of the attempt to construct the silver eel migration routes, as well as possibilities for constructing algorithms that could be used to estimate more realistic routes, is given in Appendix A.

Instead, a conceptual model for silver eel migration was constructed, based on a hierarchy of water bodies, which may provide a reasonable description of silver eel migration in The Netherlands. In this conceptual model, silver eels are split into three main groups, according to water body type. Three main types of water body are distinguished, corresponding to the three main hierarchies 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 typically high levels of mortality during passage. In the model, each polder is serviced by a single pumping station (there are no pumping stations in sequence). Pumping stations of coastal polders can pump water directly into the sea, causing silver eels to 'escape' if they survive the passage. 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 330 km^2 of wetted area of drainage ditches (Chapter 4; Table 4.2).
2. 2nd hierarchicy (termed 'boezem' water bodies): water bodies such as canals, inland lakes (such as the Frysian lakes), but also smaller streams 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 WFD water bodies (Chapter 4) except the larger lakes Veluwerandmeren and Lauwersmeer. In total, the 'boezem' water bodies represent 89.9 km^2 (linear water bodies in the OWAGL shapefile) and 621.8 km^2 (polygons of water bodies in the OWAGV shapefile) of wetted area in the model.
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), the lakes IJsselmeer and Markermeer, Veluwerandmeren, Lauwersmeer and Grevelingenmeer. Silver eels have been found to experience low levels of mortalities during passage of barriers (if any) from these large water bodies, with the exception of the passage of hydropower plants for eels which start their migration from upstream sections of the main rivers.

A visual representation of the model for migration routes is given in Figure 6.2. The hierarchies of inland water bodies (and sections of rivers upstream of hydropower plants) are connected with each other and with the sea as represented by arrows. The model will have to be completed by:

1) transition rates between types of water bodies: for example the percentage of silver eel moving from polder waters to the sea, or to boezem waters.
2) mortality rates during passage.

Given the starting biomasses of silver eels in the different hierarchies of water bodies, and given estimates of transition and mortality rates, the model can be followed to its logical conclusion to obtain a prediction of the total silver eel mortality during migration to the sea.

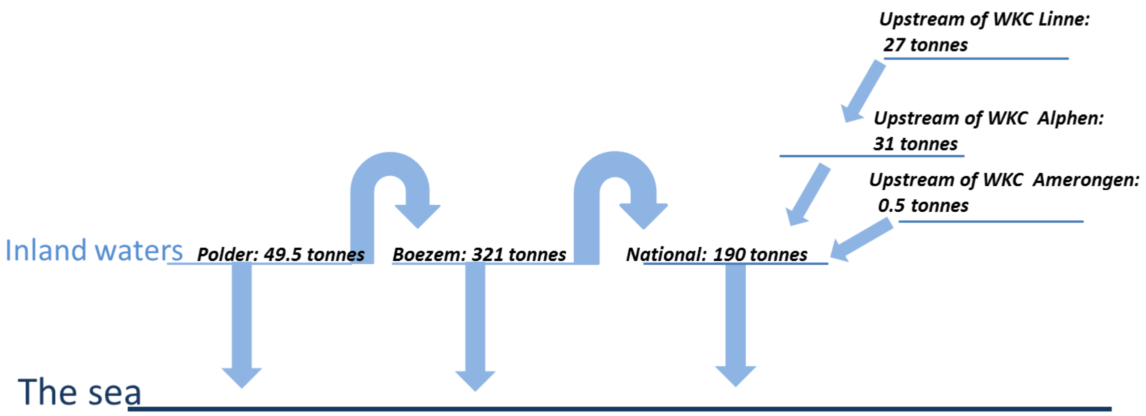

Figure 6.2. Conceptual model for estimating mortality during silver eel migration due to barriers (see text).

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 if silver eels are transferred from one hierarchical class to another, for example from polder to boezem. This approximation 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), such as in the polder 'Berkel' (http://www.hhdelfland.nl/projecten/bergboezem-berkel/aanleiding-en-aanpak).

We note that, given 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 allocation of silver eel biomass over the different hierarchies of water bodies. Instead, the biomass of silver eels that are estimated to die during migration will be dependent on the absolute starting biomass. In the overall assessment presented in Chapter 7, only an estimate of the percentage mortality is necessary as a parameter in the estimation of lifetime anthropogenic mortality, or %SPR. Therefore, in this chapter we only illustrate the model with biomass estimates from scenario 2. The differences in the proportional allocation of silver eels between 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 given in Table 6.1. These estimates are based on the static spatial population model as presented in Chapter 4.

Table 6.1. Estimates of silver eel biomass in the three hierarchical types of water bodies that are distinguished in the conceptual migration model: 'Polder water' (300 km^2 of ditches) 'Boezem waters' (all the regionally managed water bodies without Veluwerandmeren and Lauwersmeer), and 'national waters' (large nationally managed water bodies such as the main rivers and lakes IJsselmeer and Markermeer). See main text for a description of these three types of water bodies. Estimates are provided as metric tonnes.

		Scenario	
	1	**2**	**3**
Polder	15	49.5	49.5
Boezem	66	321	417
National	144	249	399
Total	225	619.5	865.5

6.2 A general note on the estimation of mortalities due to barriers

We note that in the parameterisation of the barrier mortality model we use "net mortality rates" for barriers: the proportion of silver eels that ends up in that barrier multiplied by the proportion that dies during passage. In case an alternative route for migration is available, such as a ship lock or sluice in combination with a pumping station, estimates of net mortality rates are typically lower than the proportion of silver eels that die attempting to pass the barrier. This is illustrated in Figure 6.3, for silver eel mortality during passage of a 'complex' barrier with a pumping station, ship lock and discharge sluice. The width of the grey bar in Figure 6.3 is proportional to the number of silver eels migrating, with the direction of migration indicated by the arrows. A proportion of the eels may use the route to the ship lock. The remaining silver eels will swim in the direction of the pumping station/discharge sluice, and a proportion of these individuals may choose the ship lock. Out of the remaining silver eels that continue to swim towards the pumping station, a proportion may choose to turn around because of cues caused by the pumping station. This may eventually lead to only a small proportion of eels actually passing the pumping station. This small proportions is multiplied by the mortality rate of the pumping station to obtain an estimate of its net mortality rate.

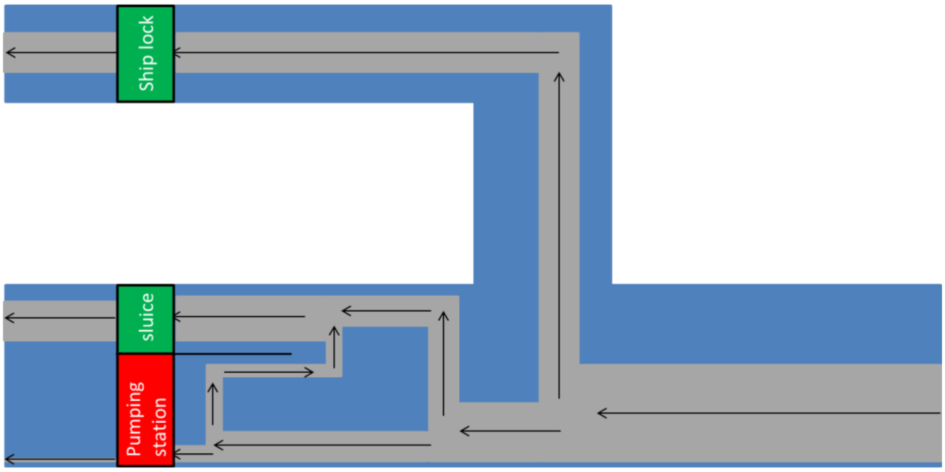

Figure 6.3. An illustration of how net mortality rates are expressed in the barrier mortality model. Barriers may be complex and consist of a combination of pumping stations, ship locks, sluices or other alternative routes for migration (see text). The width of the grey bar in Figure 6.3 is proportional to the number of silver eels migrating, with the direction of migration indicated by the arrows.

In paragraph 6.3 below, an estimate of pumping station mortalities s is derived based on a meta-analysis of a large number of studies. These estimates are mortality rates given that eels attempt to pass these pumping stations and thus assume that there is no alternative route for migration. These estimates are therefore used in the barrier mortality model for the mortality rates during transition from polder waters to boezem waters.

In paragraph 6.4, estimates of barrier mortalities are made for the prioritised list of 34 (often complex) barriers for eel by Buijse et al. (2009). The top 34 prioritised barriers consist of large barriers such as large pumping stations, hydropower plants and sluices (or a combination of these) with relatively large amounts of wetted area upstream. The estimates of barrier mortality rates presented in paragraph 6.4 are net mortality rates which account for the proportion of silver eels that attempt to pass the barriers.

6.3 Mortalities caused during passage of pumping stations

There are direct and indirect effects of pumping stations.. 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 (de Graaf et al., unpublished results). Thirdly, pumping stations will increase the predation risk of fish. Damaged and confused fish will be easier to prey on by piscivorous fish or birds. But also the risk of being captured by commercial or recreational fishermen is higher around pumping station when migratory fish aggregate while searching for an opportunity to pass. In the Yellow Eel Model, however, we will only focus on the impact of pumping stations on the survival of migrating eel when they actually pass through a pumping station.
Pumping stations can roughly be divided in 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 6.4 provides an overview of the distribution of different types of pumping station in The Netherlands (based on a sample of 2813 pumping stations).

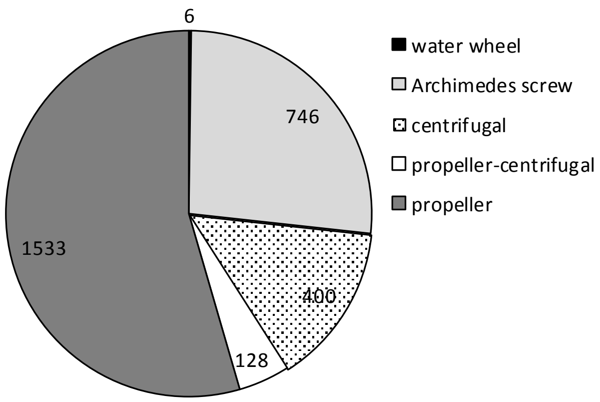

Figure 6.3. Distribution of different types of pumping station in The Netherlands (redrawn from Kunst et al., 2008).

Table 6.2 provides an overview of studies conducted mainly in The Netherlands and Belgium on the impact of different types of pumping stations on the survival of eel. These studies clearly demonstrated that in general propeller pumps with axial or axial/radial water flow caused the highest mortality rates when eel passes 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”.

Table 6.2. Overview of eel damage and mortality by different types of pumping station.

Country	Pump type	Location	Capacity (m^3/min)	rpm	Head (m)	# eel	% damaged	% dead (direct)	Delayed mortality	Reference
NL	Water wheel	Spaarndam	1920	6	0.3	?	0	0	?	Kruitwagen & Klinge 2008
BE	Archimedes	Sint-Karels-molen	30	39	2.9	?	10	4	?	Denayer & Belpaire 1992
BE	Archimedes	De Seine	35	37	3.6	?	37	0	?	Germonpré et al., 1994
BE	Archimedes	Isabella	100	25	?	48		8-19	?	INBO
BE	Archimedes	Isabella	200	21	?	131		13-16	?	INBO
NL	Archimedes	Overwaard	500	17	2.2	43		2	?	Vriese et al., 2010
NL	"de Wit" Archimedes	Halfweg	660	22	0.3	?	0	0	?	Kruitwagen & Klinge 2008a
UK	Archimedes		?	23-31	?	160	0.6	0	?	Kibel 2008
GE	Turbine-Archimedes	Bielefeld	?	?	?	?	0	0	?	Spah 2001
BE	Centrifugal	Elektriek-Zuid	60	49	5	287	1.4	1.4	?	Germonpré et al., 1994
NL	Centrifugal	Schoute	505	143	-1-3.8	?	0	0	?	Kruitwagen & Klinge 2008c
NL	Centrifugal	Gouda	690	70	-0.1-4.2	?	0	0	?	Kruitwagen & Klinge 2008a
NL	Centrifugal	Katwijk	1080	59	-0.1-4.2	?	0	0	?	Kruitwagen & Klinge 2007
NL	Centrifugal	Boreel	400	204	0.9	49		48	?	Vriese et al., 2010
?	Hidrostalpump	?	18	460-600	?	?	0	0	?	Helfrich et al., 2002
?	Hidrostalpump	?	?	890-1200	10	2300	<3	0	?	Patrick & McKinley 1987
NL	Propeller-centrifugal	Lijnden	525	200	5.4	?		50	?	Manshanden 2008
NL	Propeller-centrifugal	Tonnekreek	170	?	1.52	34		0	?	Vriese et al., 2010
NL	Propeller-centrifugal	Schilthuis	350	115	2.8	27		22	?	Vriese et al., 2010
NL	Propeller	Meerweg	37.5	735	1.15	?	100	100	NA	Kruitwagen et al., 2006

* Underestimation as physically undamaged eels did reveal internal damage after dissection which will result in delayed mortality.

Table 6.2. Overview of eel damage and mortality by different types of pumping station. (continued)

Country	Pump type	Location	Capacity (m^3/min)	rpm	Head (m)	# eel	% damaged	% dead (direct)	Delayed mortality	Reference
BE	Propeller	Woumen	60	500	2.7	?	100	100	NA	Germonpré et al., 1994
BE	Propeller	Avrijevaart/ Burggraven stroom	100	480	?	39	98 %	98	?	INBO
NL	Propeller	IJmuiden	15600	64	0.1-2.3	35	71	52	Yes	Kruitwagen & Klinge 2008b
NL	Propeller	IJmuiden	15600	Vari-able	0.1-2.3	114	36	36*	Yes	Witteveen & Bos 2010a
NL	Propeller	IJmuiden	15600	Vari-able	0.1-2.3	251	40.6	41*	Yes	Witteveen & Bos 2010a
NL	Propeller	Hoogland	1500	50	?	77	5	5	Yes	Witteveen & Bos 2010b
NL	Propeller	Lijnden	255	360	5.4	?		100	?	Kruitwagen et al., 2006
NL	Propeller	A.F. Stroink	2500	80	0.6	10	30	0	NO	Kroes et al., 2005
NL	Propeller	Stenen-sluisvaart	60	500	2.7	?	100	100	NA	Germonpré et al., 1995
NL	Propeller	Den Deel	200	165	0.6	?	30	8	?	Riemersma & Wintermans 2005
NL	Propeller (closed)	Kortenhoeve	60	355	0.8	118		32	?	Vriese et al., 2010
NL	Propeller (closed)	Thabor	24	?	1	21		38	?	Vriese et al., 2010

* Underestimation as physically undamaged eels did reveal internal damage after dissection which will result in delayed mortality.

One study dissected physically undamaged eels and concluded that many of these eels had internal injuries which would result in delayed mortality (Witteveen & Bos 2010a). In Table 6.3 we defined mortality as the % dead plus % damaged. 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 6.3).

Table 6.3. Calculation of the average pumping station mortality used to estimate silver eel mortality during migration.

Pump type	**Proportion (Figure 5.1)**	**Average mortality* (%) (Table 5.3)**	**Weighted Mortality (%)**
Water wheel	0.002	0	0
Archimedes screw	0.27	11	2.8
Centrifugal pump	0.14	11	1.5
Propeller-centrifugal pump	0.05	11	0.5
Propeller pump	0.55	67	36.5
PUMP MORTALITY YELLOW EEL MODEL			~41%

* Mortality is % dead + % damaged.

6.4 Estimated mortality rates associated with the top 34 prioritised barriers for eel

Kroes et al. (2008) created a prioritised list of a large number of barriers which in some way may hinder fish migration in The Netherlands. From this list, a smaller prioritised list was made of 34 barriers for eel by Buijse et al. (2009). The top 34 prioritised barriers consist of large pumping stations, hydropower plants and sluices (or a combination of these) with relatively large amounts of wetted area upstream. Here, we present estimates of silver eel mortalities induced by the top 34 prioritised barriers, based on detailed data collected at each of these barriers.
For the top 34 prioritised barriers, it was assessed which part of the silver eel reaching the barrier suffered mortality during passage or experienced blockage by being unable or unwilling to pass the barrier. Almost all barriers are complexes consisting of several types of barriers and therefore offer different possible passage pathways. Another complicating factor is that most of the larger water systems in The Netherlands (i.e. 'boezem' and 'rijkswater' systems) have several outlets. Thus the division and possible redistribution after blockage of silver eels over several outlets determines the overall mortality rate that silver eel experience when leaving a water system. The top 34 will discussed subdivided in different categories:

Pumping stations without alternative routes. For these, the research on the mortality rates in the pumping stations are the best guess for the overall mortality rates of seaward migrating silver eel. This is under the assumption that eventually all eel that approach the pumping station will pass, with or without hesitation and associated delay. Telemetry results in Friesland near four pumping stations suggest this assumption is correct (unpublished results). The mortality rates for the different pumping stations in this category were taken from existing studies; A.F. Stroink (Kroes et al., 2006), Halfweg (Kruitwagen & Klinge 2008a), Katwijk (Kruitwagen & Klinge 2007), Kadoelen (van Wijk 2011), de Waker (van Wijk 2011).

Ship locks. Fish migration through ship locks in The Netherlands is poorly studied and the degree of blockage for silver eel migrating seaward at locations with ship locks is largely unknown. It is assumed that direct mortality is negligible and only blockage might occur. For this assessment we assumed low blockage (arbitrarely set to 20 %) intermediate blockage (50%) and high blockage (80 %) depending on dimensions (high when large), frequency of ship passage (high when numerous), transition between fresh and salt water (high in transition situations, low in freshwater-freshwater or saltwater-saltwater), and discharge rate (high when discharging much water). This was done for the ship lock complexes at Krammersluizen, Bergsche Diep Sluis, sluizencomplex Terneuzen, Brouwerssluis, Volkeraksluizen and Oranjesluizen.

Discharge sluices with ship locks. These type of barriers are often situated near the sea, and discharge excess freshwater via discharge sluices during low tide. Most of these discharge sluices are accompanied by one or more ship locks. Typically these barriers provide discontinuous migration windows lasting up to several hours. The frequency of these windows depend on the amount of excess water within the water system that it drains. As such, migratory windows for migrating silver eels occur frequently and no extra mortality is expected in general. Only during 'dry' periods, some delay in migration may occur and an increase in 'indirect' natural (predation risk) or fishery mortality might occur. In the downstream section of the Rivers Meuse and Rhine there are three main discharge sluice complexes: Haringvliet (Meuse-Rhine delta), Afsluitdijk Den Oever and Afsluitdijk Kornwerderzand (Lake IJsselmeer). Telemetry studies in the lower Meuse-Rhine delta showed that the Haringvliet sluices hardly hampered silver eel migration. During periods with fewer migration windows, silver eeleither waited until the first high discharge event to pass the sluices or migrate via an alternative open accessible route to the Nieuwe Waterweg outlet ("Spui"; Winter & Bierman 2010). This is also expected to be the case in other locations with discharge sluices, i.e. Afsluitdijk Den Oever and Kornwerderzand, Lauwerssluizen, Schermersluis, Spuisluis Oost-oever. Only for the Houtribdijk Krabbegatsluizen and Houtribsluizen the frequency of migration windows is less than for the other locations. However, the silver eel from lake Markermeer can also migrate to sea through the Noordzeekanaal, i.e. Oranjesluizen, Gemaal IJmuiden/Sluizen complex. The ship locks in all these locations may add additional migration windows for silver eel, but it thought that these play a minor role in facilitating the seaward migration compared to the adjacent discharge sluices.

Pumping stations with discharge sluices and ship locks. At several locations excess water is discharged through different parallel situated pumping stations, discharge sluices and ship locks offering different migratory routes in time and space to silver eels. The pumping station route is potentially hazardous for the silver eel. Thus the fraction of silver eel using the pumping station as the seaward route combined with the percentage of eel that gets mortality wounded while passing this station determines overall mortality rates. The best studied location with a pumping station, discharge sluices and ship locks is Gemaal IJmuiden/Sluizen complex (several monitoring, DIDSON and telemetry studies reviewed in Winter 2011). Despite the high mortality percentage >42 % at the propeller pumping station, overall mortality rate of all silver eel passing complex IJmuiden was estimated to be only 1.0-2.5 %. This is because the majority of silver eels were found to use alternative routes to the sea, e.g. the adjacent discharge sluices. A large fraction of the silver eel approaching the pumping station in operation hesitated to enter the pumping stations and showed return and search behaviour. As discussed in the above paragraph (5.2.1), pumping stations greatly vary in mortality rates and degree of blockage, i.e. fraction of the silver eel that show hesitation to enter the pumping station. For the other locations few studies are available (mostly concerning the mortality rate in the pumping station), and the fractions of silver eels that use the different parallel migration routes are unknown. For Gemaal de Helsdeur mortality rate in the pumping station was assessed to be only 2 %, and therefore overall mortality rate is even less.

Pumping stations with ship locks. For these locations the ship locks offers an alternative migration route to the pumping station for silver eel. However, little is known on the passage of silver eel through ship locks and the factors determining success rate. We assessed overall mortality based on available studies for the locations at pumping stations Gemaal Overtoom (van Wijk 2011), Zaangemaal (van Wijk 2011), J.L. Hooglandgemaal and Johan Frisosluizen Stavoren (Witteveen & Bos 2010b), De Ruiter (Kampen 2011). For the pumping station Wouda Gemaal at Lemmer, blockage was estimated to be high given that this pumping station only operates 5 times per year on average. For this location, however, other outlets in the Frisian boezem-water system are also available for seaward migrating silver eel.

Hydropower plants with weirs and fishways. Silver eel can choose different alternative routes when approaching these hydropower, weir, fishway complexes. The discharge distribution over different pathways and the behavioural response of silver eels determine the overall mortality caused by the hydropower stations. For the Meuse this is well-studied with telemetry and monitoring research (Winter et al., 2006, Jansen et al., 2007, Winter et al., 2007, unpublished results for 2010-2011). For the hydropower plant Amerongen in the Rhine branch lek-Nederrijn, the overall mortality is unknown and the average of the mortality rate in the Meuse was taken as a preliminary estimate.

An overview of the estimated blockage and mortality rates for the top 34 barriers is given in Table 6.4. For most types and combinations of structures at barriers, we ignored blockage effects, because: (i) alternative routes were available or (ii) eel were found to have mainly experienced delay rather than blockage e.g. at hydropower stations or pumping stations. Only at barriers consisting of ship locks without alternative routes we used blockage rate as a precautionary best guess for 'indirect' mortality, although knowledge on eel migration through the various ship locks is still very limited.

Table 6.4. Estimated mortality and blockage rates of the top 34 prioritised barriers for eel (Buijse et al., 2009).

Barrier name	**Mortality rate**			**Blockage rate**		
(B: Boezem water; R: Rijkswater, i.e. large Nationally managed water body)	**min**	**max**	**best guess**	**min**	**max**	**best guess**
Wouda Gemaal, Lemmer (B)	0	50	25	50	100	98*
J.L. Hooglandgemaal and Frisosluizen, Stavoren (B)	3	16	6	0	10	0+
A.F. Stroink (B)	10	100	30	0	10	0
Lauwerssluizen (R)	0	0	0	0	10	0
sluizencomplex Nieuwe Statenzijl (B)	0	0	0	0	20	0
Haringvlietsluizen (R)	0	0	0	0	10	0
Flakkeese spuisluis (R)	0	0	0	0	10	0
Krammersluizen (R)	0	0	0	10	90	20
Bergsche Diep sluis (R)	0	0	0	10	90	20
sluizencomplex Terneuzen (R)	0	0	0	10	90	20
Brouwerssluis (R)	0	0	0	10	90	80*
Volkeraksluizen (R)	0	0	0	10	90	50*
Hydropower plant Amerongen (R)	6	21	16	0	10	0
Gemaal IJmuiden/sluizen complex (R)	1	2.5	2	0	30	0
Oranjesluizen (R)	0	0	0	10	90	20
Hydropower plant Alphen (R)	6	18	15	0	10	0
Hydropower plant Linne (R)	9	21	17	0	10	0
Afsluitdijk Stevinsluis Den Oever (R)	0	0	0	0	8	0
Afsluitdijk Kornwerderzand (R)	0	0	0	0	11	0
Houtribdijk Krabbersgatsluizen (R)	0	0	0	50	86	75*
Houtribdijk Houtribsluizen (R)	0	0	0	50	79	70*
gemaal Spaarndam (B)	0	2	0	0	30	0
gemaal Katwijk (B)	0	1	0	0	30	0
gemaal Halfweg (B)	0	2	0	0	30	0
Gemaal De Helsdeur (B)	0	20	2	0	10	0
Schermersluis (B)	0	0	0	0	10	0
Zaangemaal (B)	0	30	10	0	10	0
Gemaal Overtoom (B)	0	10	4	0	30	0
Gemaal Kadoelen (B)	8	8	8	0	30	0
Gemaal de Waker (B)	2	2	2	0	30	0
Spuisluis Oost-oever (B)	0	0	0	0	10	0
Mijndense sluis/gemaal (B)	0	50	10	0	30	10
Gemaal de Ruiter (B)	20	100	70	0	10	0

* These barriers mainly have a blockage effect, but alternative escape routes are available which results in better silver eel escapement than suggested by these percentages.
\+ These are in fact two barriers in the top 34 prioritised list which are part of the same complex and are therefore combined in this analysis.

6.5 Estimation of barrier mortality

The following estimates of mortalities were used in the model:

1. From polder waters to boezem waters or to the sea or: a best guess estimate of **41%** mortality (minimum of 25% and maximum of 66% average mortality on the silver eels in polder waters), based on a meta-analysis of estimates from a large number of studies, as presented in paragraph 6.2.
2. From Boezem waters to large nationally managed waters or to the sea: a best guess estimate of **7.5%** mortality (minimum of 5% and a maximum of 15% on the silver eels in boezem waters), based on the estimates of mortalities of barriers in the top 34 prioritised eel barriers in boezem waters. Some large pumping stations servicing boezem waters have been found to have low associated mortalities during passage. Whilst some pumping stations have larger or unknown mortality rates, in most cases there are alternative routes for migration available, such as through discharge sluices and ship locks, and at least part of the silver eels have been found to use these alternatives (paragraph 6.4).
3. From large nationally managed water bodies to the sea: a best guess estimate of **2.5%** mortality (minimum of 1% and a maximum of 5% on average for silver eels in these water bodies), based on a number of studies which have shown that: The North Sea canal is the only major route with a pumping station, but silver eels have been found to overwhelmingly avoid the pumping station and 'escape' through the different sluices or potentially ship locks that are available. In a review of the studies done in the North Sea canal, Winter (2011) estimated only approximately between 1% and 2.5% mortality due to the pumping station. Downstream parts of the main rivers are thought to have no major barriers. Winter & Bierman (2010) found no effect of the Haringvliet dam on escape rates: when the sluices in the dam are closed due to low water levels eels were found to escape to the sea via alternative available routes. Eels from Lake IJsselmeer are thought to experience few problems in leaving the lake via the available sluices which offer regular opportunities for migration (paragraph 6.4). Silver eels from Lake Markermeer and the Veluwerandmeren can choose two routes to escape to the sea: the North Sea canal via sluices which are thought to cause no mortality during passage, and via lake IJsselmeer in which case a dam has to be passed through infrequently opened discharge sluices (paragraph 6.4).
4. An exception to the low mortalities experienced by silver eels during migration from nationally managed waters to the sea, are the sections of the main rivers upstream of the three hydropower plants in the main rivers (two in the river Meuse and one in the river Rhine). These sections of river are represented explicitly in the model. Estimates of mortalities in the hydropower plants are: a best guess estimate of 15% (minimum of 10% and a maximum of 25%), based on a number of studies as reviewed in paragraph 6.4.

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 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 they are further exposed to mortalities. We do not known the proportion of boezem water bodies that are connected directly to the sea, but estimate this to be small at 20%, whereas most water bodies are thought to be connected eventually to the North sea canal, lakes IJselmeer or Markermeer or one of the main rivers. The large regionally managed water bodies are all thought to be connected to the sea with a transition rate of 100%. Silver eels in the sections of river upstream of the hydropower plants are estimated to all (100%) pass through these barriers during their downstream migration to the sea. The silver eel barrier mortality model is visualised in Figure 6.5.

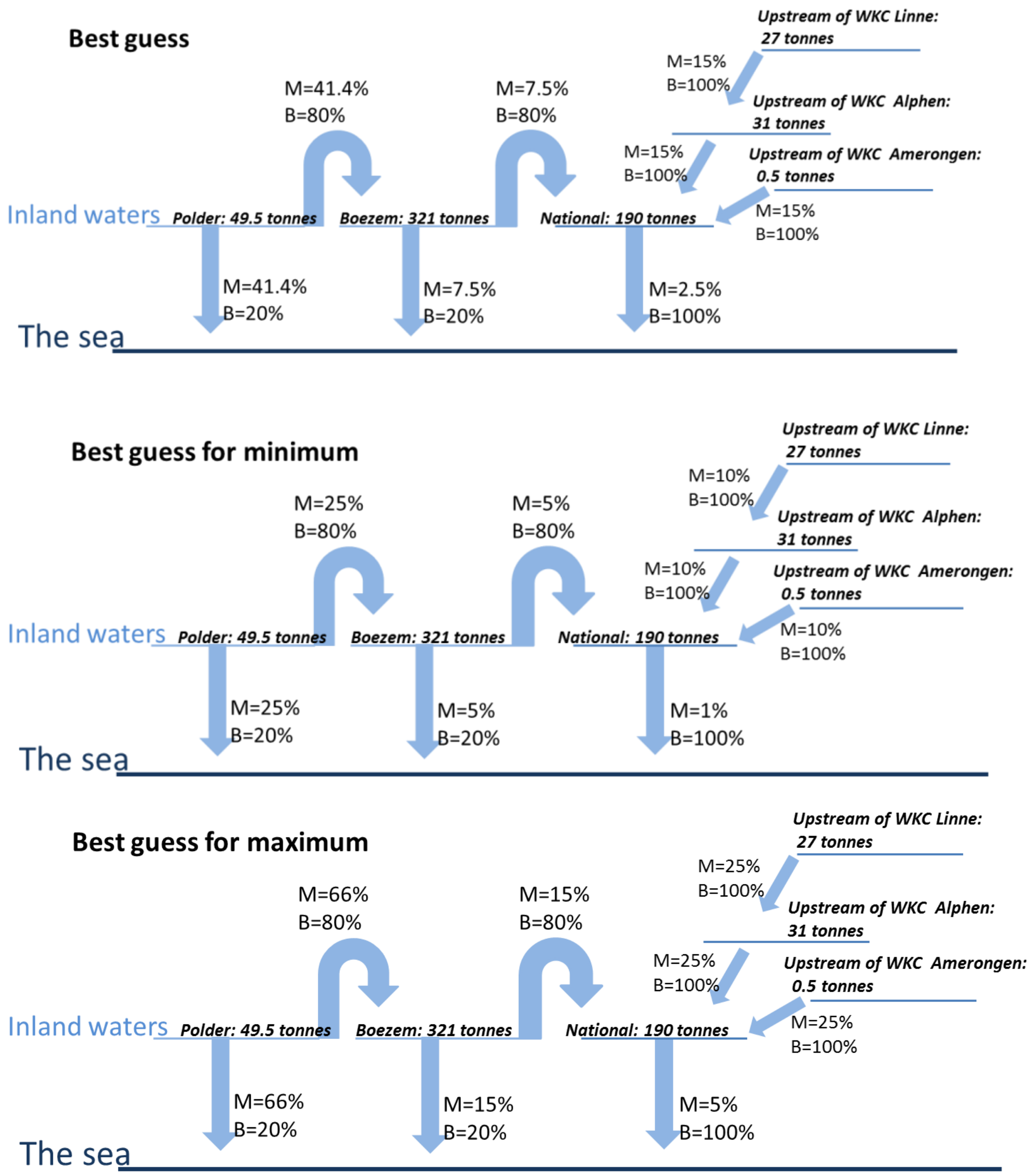

Figure 6.5. Schematic overview of the model to estimate silver eel mortality due to barriers sea (pumping stations, hydropower plants, (discharge) sluices, ship locks, etc.) during migration from inland waters to the sea. Given are three scenarios which differ in estimated mean barrier mortalities during transitions from one hierarchical type of water body to another (see text for estimation of barrier mortalities).

The estimated mortalities of silver eel are given in Table 6.5. In these estimates, the reduction in mortality due to trap-and-transfer initiatives, in which silver eel is caught above a barrier and 'lifted' across it, is estimated by deducted the total tonnage (1 tonne) of silver eel which was reported to have been transferred in this manner.

Table 6.5. Estimated mortalities of silver eel in hydropower plants, pumping stations, sluices and other barriers. Estimates are given as metric tonnes (absolute values are valid only for estimates produced under scenario 2), and as percentages of the total starting population of silver eel (valid for all scenarios). See Figure 6.5 for mortality rates which have been used for the best guess, best guess for minimum, and best guess for maximum.

	Best guess for minimum	**Best guess**	**Best guess for maximum**
Leaving polder waters	12.4	20.5	32.7
Leaving boezem waters	17.5	25.8	50.2
Hydropower plants	8.2	12.1	19.5
Major national waters (other than hydropower plants)	5.1	12.3	22.8
Trap-and-transfer	-1	-1	-1
Total	42.2	69.7	124.2
% mortality from total silver eel biomass in inland waters	6.8%	11.3%	20.1%

We note that the mortality estimates in hydropower plants in Table 6.5 are likely to be underestimates, because these include only silver eels produced in Dutch sections of the main rivers. However, an estimated 149 tonnes of silver eel migrate downstream on the river Rhine from Germany (pers. comm. Claus Wysujak), and 16.5 tonnes of silver eel migrate downstream on the river Meuse from Belgium and Germany. 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). Using this estimate, the mortality on these eels migrating from Germany is estimate at 149*0.06*0.15 = 1.3 tonnes (15% mortality during passage; Figure 6.5). The additional 16.5 tonnes of silver eel in the Meuse river is expected to pass the hydropower plants at Linne and Alphen, causing an additional 16.5*0.15 + (0.85)*16.5*0.15 = 4.6 tonnes of silver eel mortality.
The silver eel mortalities on these 'foreign' eels migrating from Germany and Belgium are given in Table 6.6, but we note that these mortalities are not taken into account in the evaluation of the Dutch EMP.

Table 6.6. Estimated mortalities of silver eel in hydropower plants and pumping stations, including estimated tonnages of silver eel migrating downstream from Germany and Belgium on the rivers Rhine and Meuse (see main text). Estimates are given as metric tonnes and as a percentage of the total starting population of silver eel. See Figure 6.5 for mortality rates which have been used for the best guess, best guess for minimum, and best guess for maximum.

	Best guess for minimum	**Best guess**	**Best guess for maximum**
Leaving polder waters	12.4	20.5	32.7
Leaving boezem waters	17.5	25.8	50.2
Hydropower plants silver eel from Netherlands	8.2	12.1	19.5
Hydropower plants silver eel from Germany and Belgium	3.9	5.9	9.9
Major national waters (other than hydropower plants)	5.1	12.3	22.8
Trap-and-transfer	-1	-1	-1
Total	46.1	75.6	134.1

The silver eel mortality and injury level due to hydropower stations has been monitored in the river Meuse. Telemetry studies of migrating silver eel in 2002 and 2004 indicated a mortality range of 16-34% in two hydropower stations in the Meuse that are set in series, almost half of the total mortality (Winter & Jansen, 2006). Preliminary results of telemetry studies in 2010 showed little change in mortality by the hydropower stations in the Meuse in the period 2009-2011 (Table 6.7). This is no surprise because adjustments to achieve a 35% reduction in direct mortality to the turbines did not start till November 2011 at these hydropower stations. In both hydropower stations in the Meuse adjusted turbine management will continue till 31 January 2012. Furthermore, it is foreseen that due to technical difficulties a reduction in mortality of 24% will be realised instead of the 35% which was originally intended.

Adjusted turbine management was also implemented on 17 November 2012 at Amerongen hydropower station in the Lower Rhine. No data is currently available about the effect on mortality but it is likely that the impact is similar to the results found in the River Meuse.

Table 6.7. Observed and estimated mortality ratios of silver eels which have been caught and released upstream in the river Meuse (2002, 2004 from Winter & Jansen, 2006; 2010 from Winter & Griffioen, unpublished data).

	2002 (n=121)		**2004 (n=105)**		**2010 (n=121)**	
	Obs (%)	**Est (%)**	**Obs (%)**	**Est (%)**	**Obs (%)**	**Est (%)**
successful passage to sea*	37	>37	31	>31	33	>33
commercial fisheries	15	21-15	13	19-22	0	0
recreational fisheries	1	1	2	3	0	unknown
hydropower plants (total)		16-26		25-34		22-26
“unknown”	38	11-25	35	10-22		41-45

* Exclusive possible mortality between the last detection station and the sea in the Nieuwe Waterweg (~20km).

6.6 Discussion

The model as presented here to estimate mortality of silver eels during migration needs to be further evaluated and developed. Several maps and lists of barriers are available (e.g. Kroes et al., 2008; Buijse et al., 2009). However, this is to our knowledge the first formal model to estimate mortalities during passage of barriers which takes account of variation in starting positions and migration routes of silver eels. 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. In particular, the characterisation of water bodies as polder or boezem waters need to 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 model could be based upon the methodology described in Appendix A to estimate migration routes. Such a route analysis will provide the best basis for models to compute barrier mortality rates. Recommendations are made in Appendix A to further improve this spatially explicit route analysis.

7. Estimates of key stock indicators

In this chapter, the key stock indicators (see Chapter 1) are estimated, based on the results from other chapters.

As explained in detail in Chapter 1 (paragraph 1.2: Figure 1.1), anthropogenic mortalities are split explicitly into two groups:

1. fishing mortalities that occur during the yellow eel stage
2. fishing and barrier mortalities that occur during migration as a silver eel

The reason for considering these two types of mortalities separately is that yellow eel mortalities apply over a sequence of years; from transformation as glass eel to yellow eel to transformation from yellow eel to silver eel. Instead, silver eel mortalities are assumed to apply during a single year in the life cycle of an eel.

Yellow eel fishing mortality is estimated (paragraph 7.1) as a function of the proportion of retained catches out of the total estimated yellow eel standing stock biomass (as estimated in Chapter 4). Silver eel mortality is estimated (paragraph 7.2) as a function of barrier mortality (as estimated in Chapter 6) and the proportion of retained catches out of the total silver eel standing stock biomass (as estimated in Chapter 4).

The yellow and silver eel mortalities are combined in a single estimate of Lifetime Anthropogenic Mortality (LAM) which can also be expressed as %SPR (paragraph 1.2) using the yellow eel population model from Chapter 2.

The dependence of the estimates made in this chapter on other chapters is visualised in the flow diagram in Figure 7.1.

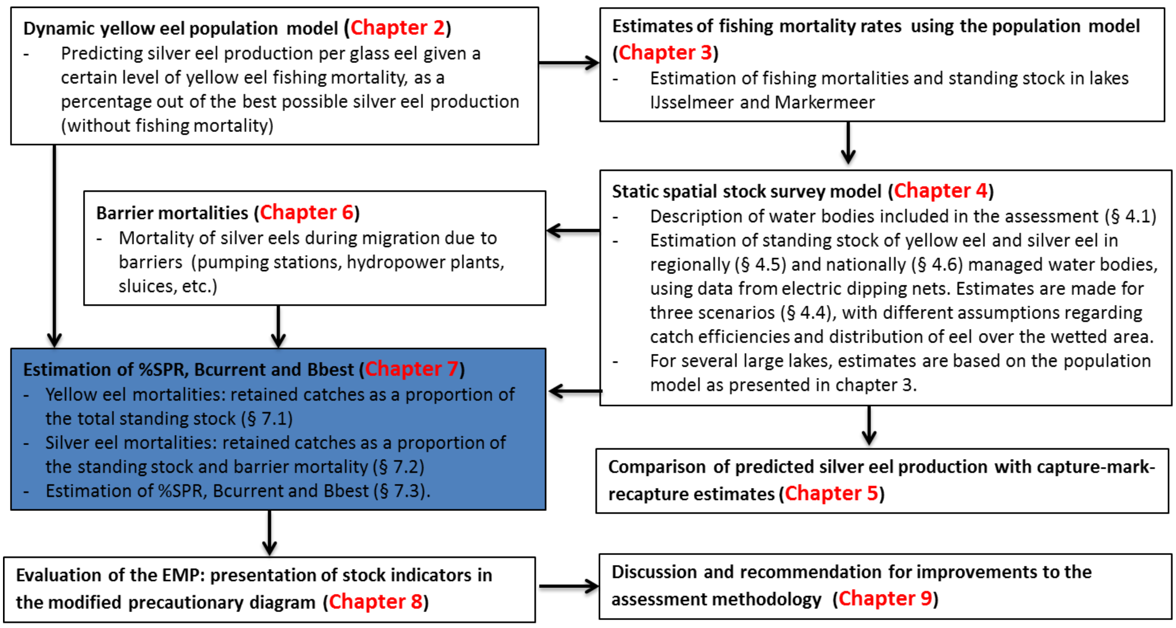

Figure 7.1. A flow diagram representing the key steps in the stock assessment methodology. Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

7.1 Yellow Eel anthropogenic mortality

The following causes of anthropogenic mortalities in the yellow eel stage are considered:

1. Mortality due to retained catches by commercial fisheries. The total landings of yellow eel are estimated at 290 tonnes (see Table 5.1).
2. Mortality due to retained catches by recreational fisheries. The total landings of yellow eel are estimated at 100 tonnes (van der Hammen & de Graaf 2011).

Not considered are:

1. Mortalities of yellow eel after release recreational fisheries. Mortality after release is unknown but may be substantial.
2. Mortalities of yellow eel induced by non-reported catches, such as poaching
3. Mortalities of yellow eel in barriers, such as pumping stations

Estimates of yellow eel anthropogenic mortalities are a function of the proportion of the retained catches out of the estimated total standing stock of yellow eel with body lengths over 30 cm. Thus, for these estimates, the fisheries are assumed to be fully selective for all eels with body lengths over 30 cm.
Fishing mortality of yellow eel is assumed to take place before the surveys are conducted. Therefore, estimated total retained catches of yellow eel are added to the estimates of standing stock derived from the surveys. The fisheries on eel start approximately in May and stop in August because of the seasonal fisheries closure that is part of the Dutch EMP. The seasonal closure was not put into place in the water board Wetterskip Fryslan' where a pilot project for decentralised eel management was done and fisheries continued until December. Most of the electric dipping net surveys in regionally managed waters were done in August - October. Hence, the assumption that fishing mortalities take place before the survey seems reasonable for these water bodies. In the main rivers Rhine and Meuse, some surveys took place in the spring, whereas others took place in the autumn. However, no fisheries took place in the main rivers since these were closed for the fisheries in 2011. Surveys in lakes IJsselmeer and Markermeer took place in the autumn.

An overview of estimated retained catches by recreational and commercial fishers in 2008 and 2011 is given in Table 7.1.
In 2011, the total retained catches by commercial fishers (reported to the Ministry of Economic affairs, Agriculture and Innovation) amounted to 367 tonnes. The retained catches have been split into yellow eel and silver eel catches. The proportions of silver eel in the landings have been estimated using the biological market sampling data (proportions of silver eel in the landings per length class). It was estimated that yellow eel retained catches amounted to 290 tonnes and silver eel retained catches amounted to 77 tonnes.
Also in 2011, retained catches of yellow eel by recreational fishers have been estimated at 100 tonnes (van der Hammen & de Graaf 2011). We have assumed that these retained catches consist entirely of yellow eels.
In 2008 (the period before the eel management plan), retained catches of yellow eel by commercial and recreational fishers were estimated at 640 and 200 tonnes by commercial and recreational fishers respectively. In addition, 280 tonnes of silver eel was caught by commercial fishers (The Ministry of Agriculture, Nature and Food quality, 2009; Tables 2.3.1 and 2.3.3).

Table 7.1. An overview of estimated retained catches by recreational and commercial fishers in 2008 and 2011. Estimates are provided in metric tonnes.

	2008		**2011**	
	Silver eel	**Yellow eel**	**Silver eel**	**Yellow eel**
Recreational	0	200	0	100
Commercial	280	640	77	290
Total	280	840	77	390

An estimate for yellow eel fishing mortality $\hat{F}$, for use in the dynamic eel population model in Eqn 2.1 (Chapter 2) is then:

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

Where RC is the retained catch of yellow eel (Table 7.1).

We note that with this estimate of fishing mortality, it is assumed that this mortality was instantaneous. Therefore, the effects of natural mortality on stock trends within a year is ignored. However, this approximation seems reasonable for eel because it is such a long lived species and year-on-year trends in biomass are relatively small.
Estimates of yellow eel fishing mortalities are given in Table 7.2.

Table 7.2. Estimated yellow eel fishing mortalities in 2011. Fishing mortalities ($\hat{F}$) are estimated as $\hat{F}$=-$\log_e$(1 - (RC/(biomass+RC))). The yellow eel fishing mortality is used to predict %SPR using the yellow eel population model presented in Chapter 2. Yellow eel standing stock biomass was estimated using the static spatial model using three scenarios (see Chapter 4).

		Scenario	
	1	**2**	**3**
Total biomass yellow eel >30 cm	1326	3331	4774
Total retained catch commercial fishers	290	290	290
Total retained catch recreational fishers	100	100	100
$\hat{F}$	0.258	0.111	0.079

The yellow eel fishing mortalities as presented in Table 7.2 are estimated average mortalities for the total eel stock in The Netherlands. Since some large water bodies have been closed for commercial fisheries, fishing mortalities will be lower (with only recreational fisheries) in these areas and on average higher in the water bodies in other parts of The Netherlands.

7.2 Silver Eel anthropogenic mortality

The following causes of anthropogenic mortalities in the silver eel stage are considered:

1) Mortality due to retained catches by commercial fisheries. The total landings of silver eel are estimated at 77 tonnes (see Table 7.1).
2) Mortality due to barriers (pumping stations, hydropower plants).

Not considered are the following possible causes of anthropogenic mortalities:

- Unreported catches, e.g. poaching

The mortality due to silver eel catches is computed by comparing the estimated 77 tonnes of silver eel retained catches with estimated silver eel production estimates (as presented in Chapter 4). The best estimates of mortality during migration of silver eel is 11.3% as presented in Chapter 6.

The contribution of silver eel mortality (barriers and retained catches) to LAM was computed by first taking the retained silver eel catches away from the estimated silver eel production in 2011, and subsequently (from the remaining tonnage of silver eel) take the percentage away that is estimated to die during migration due to barriers (Chapter 6; Table 6.5). Let the percentage of silver eels that are estimated to die during migration to the sea be represented by the parameter $\boldsymbol{\alpha}$:

$$\frac{\boldsymbol{\alpha}}{\mathbf{100}} = \mathbf{1} - \left({(\boldsymbol{B_{start}} - \boldsymbol{RC})(\mathbf{1} - \boldsymbol{M_{barrier}})}\Big/{\boldsymbol{B_{start}}} \right) \qquad \textbf{Eqn. 7.1}$$

Where B_{start} is the biomass of silver eel production, 'starting' migration from inland water bodies (before silver eel mortalities have taken place), RC is the estimated retained silver eel catch (77 tonnes in 2011), and $M_{barrier}$ is the percentage barrier mortality (11.3%; Chapter 6, Table 6.5).

For example, the total starting biomass of silver eel under scenario 2 (Chapter 4) is estimated to be 614 tonnes. For this scenario, the estimate of the proportion of silver eels that die is:

$$\frac{\boldsymbol{\alpha}}{\mathbf{100}} = 1 - \left({(619.5 - 77)(1 - 0.113)}\Big/{619.5} \right) = 0.223$$

Estimates of the proportion of silver eels that survive fishing and barrier mortalities are given in Table 7.3.

Table 7.3. Estimated silver eel mortalities in 2011.The percentage of silver eels that are estimated to die is given by the parameter $\boldsymbol{\alpha}$ (Eqn. 7.1).

	Scenario		
	1	**2**	**3**
Total biomass silver eel	225	619.5	865.5
Total retained catch commercial fishers	77	77	77
Mortality Barriers (best estimate)	11.3%	11.3%	11.3%
$\boldsymbol{\alpha}$	41.7%	22.3%	19.2%

7.3 Estimated %SPR, $B_{current}$ and B_{best}

The estimated yellow eel (paragraph 7.1) and silver eel (paragraph 7.2) mortalities can be used to estimate total Lifetime Anthropogenic Mortalities (LAM) before (2008) and after (2011) the eel management plan was put into place. As explained in Chapters 1 and 2, LAM can be expressed as %SPR.

The overall estimates of %SPR were made in the following steps:

1. The impact of the estimated yellow eel fishing mortality ($\hat{F}$; Table 7.1) on the production of silver eel is estimated using the eel population model from Chapter 2, and expressed as the % silver eel production out of the best possible production (see Chapters 1 and 2). Let the estimate of the proportion silver eel production, resulting from yellow eel fishing mortality, be represented by the parameter β.
2. The contribution of silver eel mortality (barriers and retained catches) to LAM is given by the estimates of the parameter $\boldsymbol{\alpha}$ as given in paragraph 7.2. Let the remaining 'surviving' tonnage of silver eel be '$B_{current}$'.

The current escapement of silver eel as a percentage of the best possible escapement (if all anthropogenic mortalities were mitigated), %SPR, is estimated as:

$$\frac{\%SPR}{100} = \boldsymbol{\beta}\left(1 - \alpha/_{100}\right) \qquad \textbf{Eqn. 7.2}$$

An estimate of lifetime anthropogenic mortality is then given by:

LAM = 100 - %SPR **Eqn. 7.3**

An estimate of the best possible escapement of silver eel (if all anthropogenic mortalities are reduced to zero), B_{best}, has been made as (again expressed as a percentage):

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

The stock surveys indicated conflicting trends in the period before and after the eel management plan between the regionally managed waters and the nationally managed waters. In the regionally managed waters, the total standing stock (all eel of all length classes) increased from 1683 tonnes in the period 2006-2008 to 2265 tonnes in 2009-2011.. In contrast, in the nationally managed waters a decrease of almost 66% over the period 2005-2007 compared to the period 2008-2010 was estimated (Chapter 3). However, this decrease was due mostly to the inclusion the year 2005 in the "before" period, which happened to be a peak year. Here, we estimate the standing stock of eel in the year 2008 to be 25% larger than the standing stock of eel in the year 2011, which is approximately the same as an annual decrease of 10% over a period of 3 years from 2008 to 2011 ($1 - 0.9^3 \approx 0.25$).

For the silver eel mortalities due to barriers in 2008, the 2011 estimate has been used, because no evidence was found that mortalities due to barriers had decreased (Chapter 1, Table 1.1; Chapter 6, Table 6.7).

The resulting estimates of %SPR, B_{best} and $B_{current}$ in 2008 and 2011 (respectively before and after the eel management plan was put into place) are given in Tables 7.4 and 7.5. Estimates of %SPR varied from 10% to 43.8% in 2011, and from 0% to 24.5% in 2008. Thus a substantial reduction in mortality was estimated over the period before and after the EMP, given the same scenario for the estimation of standing stock biomass in both periods. The escapement of silver eel ($B_{current}$) was estimated to be between 131 and 700 tonnes in 2011 and between 0 and 712 tonnes in 2008. Estimates for the best possible escapement ranged from 1310 to 1598 tonnes in 2011.

Table 7.4. Overall assessment for 2011 (after the eel management plan was put into place), with estimated yellow eel and silver mortalities, and the resulting estimated Lifetime Anthropogenic Mortality (LAM) or current escapement Δ which is expressed as the percentage of current escapement of silver eel ($B_{current}$) out of the estimated best possible escapement if all anthropogenic mortalities were mitigated (B_{best}). β: silver eel production per recruit as % from B_{best}, under current estimated yellow eel morality ($\hat{F}$). $\boldsymbol{\alpha}$: estimated mortality in the silver eel stage, expressed as a percentage of the eel that dies during migration to the sea out of the total biomass of silver eel that is produced in inland waters. The %SPR is estimated as: $100\,\beta\left(1 - {}^{\alpha}/_{100}\right)$, and LAM = 100 - %SPR.

		Scenario 1	Scenario 2	Scenario 3
Yellow eel mortality	Yellow eel stock (>30 cm) (tonnes)	1326	3331	4774
	Retained catch (tonnes)	390	390	390
	$\hat{F}$	0.26	0.11	0.08
	β	17.2%	43.0%	54.2%
Silver eel mortality	Silver eel stock (tonnes)	225	620	866
	Retained catch (tonnes)	77	77	77
	Mortality Barriers (best estimate)	11.3%	11.3%	11.3%
	$\boldsymbol{\alpha}$	41.7%	22.3%	19.2%
$B_{current}$	Tonnes	131	482	700
B_{best}	Tonnes	1310	1443	1598
%SPR	% from B_{best} ($B_{current}$/B_{best})	10.0%	33.4%	43.8%
LAM	100-%SPR	90.0%	66.6%	56.2%

Table 7.5. The final overall assessment for 2008 (before the eel management plan was put into place), with estimated yellow eel and silver mortalities, and the resulting estimated Lifetime Anthropogenic Mortality (LAM) or current escapement Δ which is expressed as the percentage of current escapement of silver eel ($B_{current}$) out of the estimated best possible escapement if all anthropogenic mortalities were mitigated (B_{best}). β: silver eel production per recruit as % from B_{best}, under current estimated yellow eel morality ($\hat{F}$). $\boldsymbol{\alpha}$: estimated mortality in the silver eel stage, expressed as a percentage of the eel that dies during migration to the sea out of the total biomass of silver eel that is produced in inland waters. The %SPR is estimated as: $100\,\beta\left(1 - {}^{\alpha}/_{100}\right)$, and LAM = 100 - %SPR.

	Parameter/unit	**Scenario 1**	**Scenario 2**	**Scenario 3**
Yellow eel mortality	Yellow eel stock (>30 cm) (tonnes)	1658	4164	5968
	Retained catch (tonnes)	840	840	840
	$\hat{\boldsymbol{F}}$	0.41	0.18	0.13
	β	8.1%	26.5%	37.2%
Silver eel mortality	Silver eel stock (tonnes)	281	775	1083
	Retained catch (tonnes)	280	280	280
	Mortality Barriers (best estimate)	11.3%	11.3%	11.3%
	$\boldsymbol{\alpha}$	100%	43.3%	34.2%
$B_{current}$	Tonnes	0	439	712
B_{best}	Tonnes	-	2927	2909
%SPR	% from B_{best} ($B_{current}$/B_{best})	0%	15.0%	24.5%
LAM	100-%SPR	100%	85.0%	75.5%

7.4 Discussion

In this chapter, we have presented the available data sets and methodologies which were used to estimate the current escapement of silver eel ($B_{current}$), the best possible escapement (B_{best}) and the Lifetime Anthropogenic Mortality (LAM; expressed as %SPR) of eel before and after the EMP was put into place, as well as the final estimates of these parameters. More detail on the data sets, methodologies and estimates are given in other chapters.

The mortalities during the yellow eel stage caused by commercial and recreational fishers contribute most to the estimated LAM. This can be explained by the fact that eels take many years to mature. Mortality during the yellow eel stage prevents eels from developing into silver eels. Silver eel mortalities caused by barriers and retained catches can only act on the resulting proportion out of the best possible production of silver eel (if all anthropogenic mortalities during the yellow eel stage were mitigated). This means that the present-day estimated contribution of mortality caused by retained catches of silver eel and barrier mortality contribute relative little to the estimated LAM. The estimated decrease in mortality over the period before and after the EMP can be also attributed most to decreases in yellow eel mortality, although estimated retained silver eel catches in 2008 (280 tonnes) also contributed a substantial amount to the estimated LAM in this period.
It is important to note that several sources of potential anthropogenic mortalities have not been taken into account, in particular:

- Unreported retained yellow eel and silver catches caused by e.g. poaching
- Catch-and-release mortality of eel in the recreational fishery
- Mortality of yellow eel by pumping stations and hydropower plants.
- Mortality during silver eel migration on the main rivers which cannot be attributed to the hydropower plants (Winter et al., 2006).
- Mortalities induced by pollutants, viruses and parasites introduced by humans.

The above mentioned sources of anthropogenic mortalities which have not been taken into account in this report may have a substantial effect on the actual LAM, in particular because small increases in yellow eel mortality can have pronounced effects on the estimated LAM (Chapter 2). For downstream migration on the Meuse telemetry experiments indicate that overall escapement success has remained relatively constant in the period 2002 - 2010. Mortalities attributable to hydropower plants have remained constant over the assessed years. Whilst mortality attributed to commercial fisheries has decreased in line with the expectation since the fisheries have been closed, there has been a corresponding increase in unaccounted disappearances due to unknown causes (telemetry experiments: Table 5.9). This may indicate that mortality due to e.g. poaching has increased, but this may also indicate increased natural mortality or an increased proportion of silver eels halting migration. Thus, no empirical evidence has been found as yet of reduced mortalities during silver eel migration on the Meuse river due to the closing of the commercial fisheries (Table 5.9).

Yellow and silver eel mortalities have been estimated using the retained catches and barrier mortalities in recent years (2008-2011). The overall estimated LAM for animals in recent years is not the same as the LAM that glass eels arriving in 2012 are expected to experience throughout their life. In particular, the effect of the closed areas (main rivers and some large canals) has not yet fully materialised; in the near future the eel stock may increase in the closed area relative to the eel stock in other parts of The Netherlands due to decreased mortality. If landings from other parts of The Netherlands stay the same relative to the stock size, then the mean fishing mortality is expected to decrease further in the near future. However, as discussed above, no empirical evidence has been found as yet of reduced mortalities during silver eel migration on the Meuse river due to the closing of the commercial fisheries (Table 5.9).

8. Evaluation of the Dutch EMP

In this chapter we evaluated the impact of the eel management plan using the indicators (B_0, B_{best}, $B_{current}$ and LAM for 2008 and 2011) and modified ICES precautionary diagram.

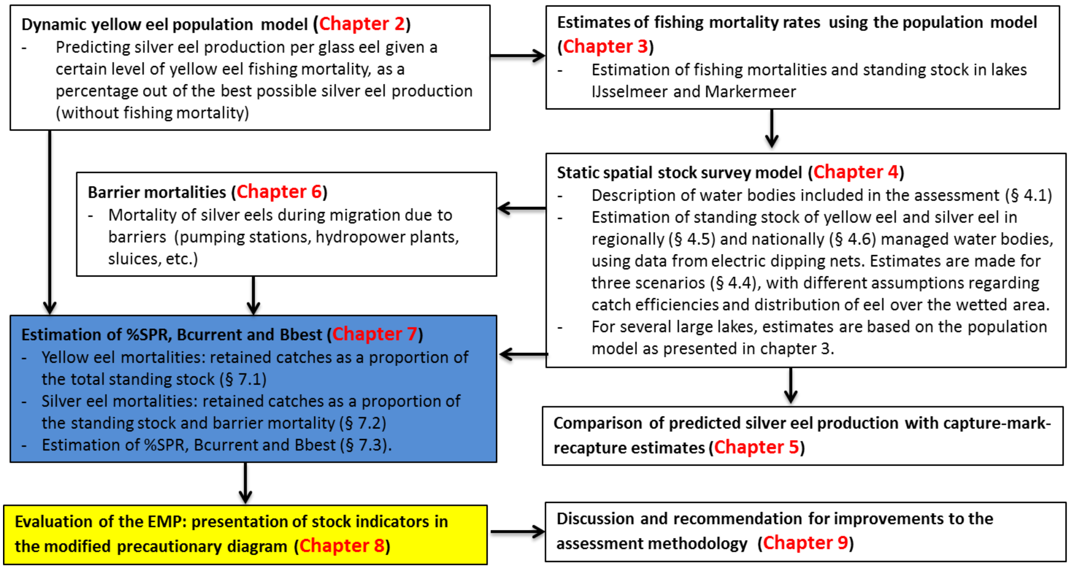

Figure 8.1. A flow diagram representing the key steps in the stock assessment methodology. Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

8.1 ICES Precautionary Diagram

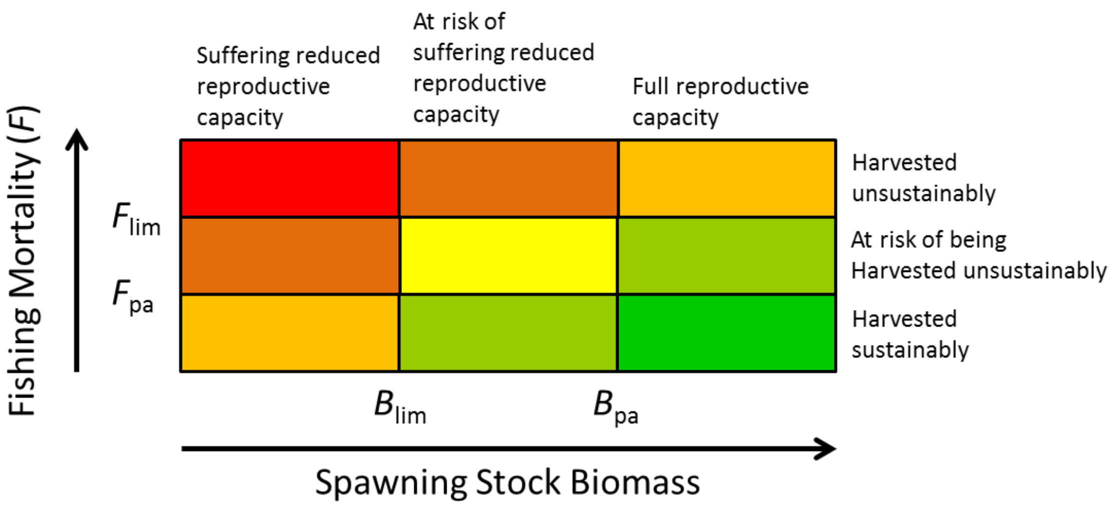

Figure 8.2. Schematic representation of the ICES Precautionary Approach showing the status of a stock in relation to the limit reference points and precautionary reference points (see also Table 1.1).

The conceptualization and handling of uncertainty in advice forms the fundament for successful management of fish stocks such as eel. ICES (2011a) developed a precautionary approach (PA) framework to address the issues of uncertainty in its advice. The PA framework builds on establishing limit reference points (LRP) reflecting stock states that should be avoided, and precautionary reference points (PRP) reflecting the risk of crossing the LRPs. Both reference points are defined in terms of fishing mortality (*F*) and spawning stock biomass (*B*) (Figure 8.2; Table 8.1).

Table 8.1. Reference points of the ICES precautionary approach framework (ICES 2009a; see also Figure 1.1).

	Spawning stock biomass (SSB)	**Fishing mortality (F)**
LIMIT reference point	B_{lim}: minimum biomass. Below this value recruitment is expected to be 'impaired' or the stock dynamics are unknown.	F_{lim}: exploitation rate that is expected to be associated with stock 'collapse' if maintained over a longer time.
PRECAUTIONARY reference point	B_{pa}: precautionary buffer to avoid that true SSB is at B_{lim} when the perceived SSB is at B_{pa}.	F_{pa}: precautionary buffer to avoid that true fishing mortality is at F_{lim} when the perceived fishing mortality is at F_{pa}.
	The buffer safeguards against natural variability and uncertainty in the assessment. The size of the buffer depends upon the accuracy of the projections (of SSB and F) and the risk society accepts that the true SSB is below B_{lim} and the true F is above F_{lim}. The accuracy of the projections depends on the magnitude of the variability in the natural system and of the accuracy of the population estimates.	

The ICES precautionary approach framework also illustrates the appropriate division between management and science. While it is the responsibility of science to define the limit references points, the decisions on precautionary reference points (i.e. defining acceptable risk levels) lies with "society". The two functions of the reference points therefor create transparency in the separation of responsibilities between scientists and managers, and in the communication of uncertainty in the advice.

8.2 ICES Precautionary Diagram modified for eel

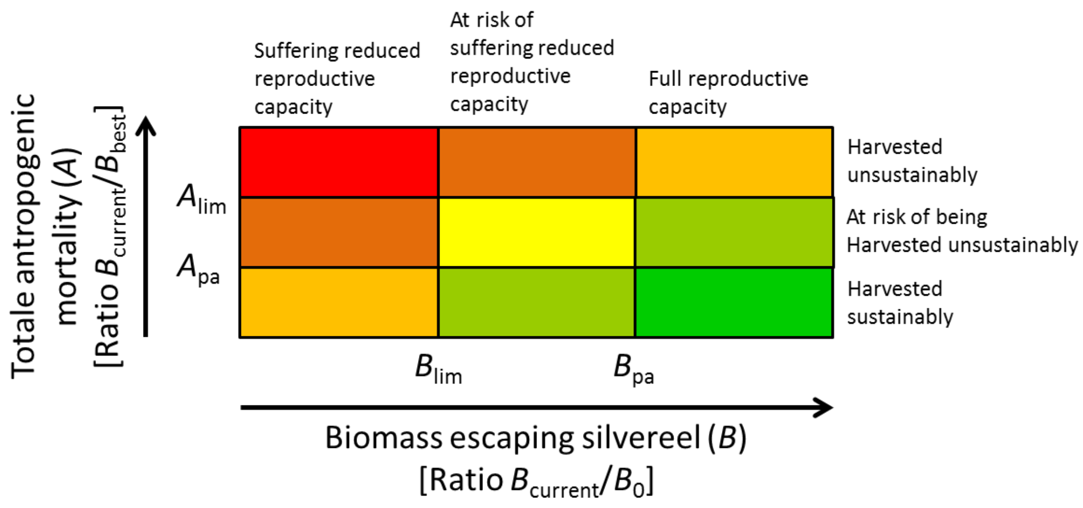

Figure 8.3. The ICES Precautionary Diagram modified for eel. A schematic representation of the status of the eel population (horizontal axis) and the impacts of anthropogenic mortality (vertical axis) in relation to the limit reference points and precautionary reference points.

Over the past two 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 a (inter)national post-evaluation of the status of the eel stock and the effect of management measures (ICES 2010a, 2010b, 2011a).

Firstly ICES (2010a, 2011a) derived a framework for deriving stock indicators, based on four estimates:

B_0 The amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the stock.

$B_{current}$ The amount of silver eel biomass that currently (assessment year) escapes to the sea to spawn.

B_{best} The amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the current stock.

ΣA The life time anthropogenic mortality; the fishing (commercial + recreational) mortality rate, summed over the age-groups in the stock, and the reduction effected + the mortality rate outside the fishery (hydropower plants, pumping stations etc.), summed over the age-groups in the stock, and the reduction effected.

In the Eel Management Plans (Ministry of agriculture, nature and food quality, 2009), The Netherlands has provided estimates of pristine biomass (13.000 t) and of current anthropogenic impacts, and thus has set reference points to which the state of the local stock and efficacy of implemented management actions can be compared.

In the second place ICES (2010a, 2010b, 2011a) adapted the classical ICES precautionary diagram to the eel case (Figure 8.3). On the horizontal axis "spawning stock biomass" was replaced by "biomass escaping silver eel". In the modified ICES precautionary diagram the horizontal axis reflects the status of the stock (biomass escaping silver eel, ratio $B_{current}/B_0$) in relation to the estimated pristine situation. On the vertical axis "fishing mortality" has been replaced with "total anthropogenic mortality", a summation of all (quantified) sources of anthropogenic mortality during the continental phase of eel. The vertical axis indicates to what extend the current population is protected in comparison with a situation where no anthropogenic mortality occurs and the production of silver eel per glass eel is at its maximum (anthorpogenic mortality, ratio $B_{current}/B_{best}$). 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.

8.3 Reference points ICES Precautionary Diagram modified for eel

With the stock indicators (B_o, B_{best}, $B_{current}$, ΣA) and a pragmatic framework (modified precautionary diagram) in place, the missing (and most difficult) piece of the puzzle remains the quantification of the limit reference points and precautionary reference points. The modified precautionary diagram shown in ICES (2010a,b) (erroneously) quantified a number of management reference points, for which, however, no value had been agreed. During ICES WGEEL in 2011 the general ICES framework for setting reference values (ICES 2009a, 2010c) was used to suggest reference values specific for eel (ICES 2011b).

ICES provides fisheries advice that is consistent with the broad international policy norms of the Maximum Sustainable Yield approach, the precautionary approach, and an ecosystem approach while at the same time responding to the specific needs of the management bodies requesting advice. When information for determining reference points is poor or absent, ICES (2009a) advises that provisional reference points are set.

Biomass reference values (horizontal axis)
ICES (2002) discussed a potential reference value for spawning-stock 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%." That is: ICES advised to set B_{lim} above the universal value of 30%, at a value of 50% of B_0. ICES (2007) added: "an intermediate rebuilding target could be the pre-1970s average SSB level which has generated normal recruitments in the past." The EU decided to set B_{lim} at 40% of B_0, in-between the universal level (30%) and the level advised by ICES (50%).

Mortality reference values (vertical axis)
ICES has not advised on specific values for mortality-based reference points, but the wordings "the lowest possible level" and "as close to zero as possible" imply that F_{lim} and therefore A_{lim} should be set close to zero. Over the years, the implied time frame for this advice has changed from "until a plan is agreed upon and implemented", to "until stock recovery is achieved" and "until there is clear evidence that the stock is increasing". The first and third phrases are more interim precautionary mortality advice than clear reference point related to any biomass.
The Eel Regulation (Council Regulation 1100/2007) sets a limit for the escapement of (maturing) silver eels, at 40% of the natural escapement (that is: in the absence of any anthropogenic impacts and at historic recruitment). Thus, for an eel stock with a biomass of escaping silver eel ≥ 40% of the biomass in the pristine situation, this corresponds to a lifetime mortality limit of ΣA lim = 0.92 (unless strong density dependence applies). In other words a minimal escapement of 40% of the currently best achievable escapement B_{best} is taken as a limit on mortality (%SPR = 40). However, the Eel Regulation does not define which (reduction in) mortality should be adhered to if $B_{current}$ is < than B_{lim}.

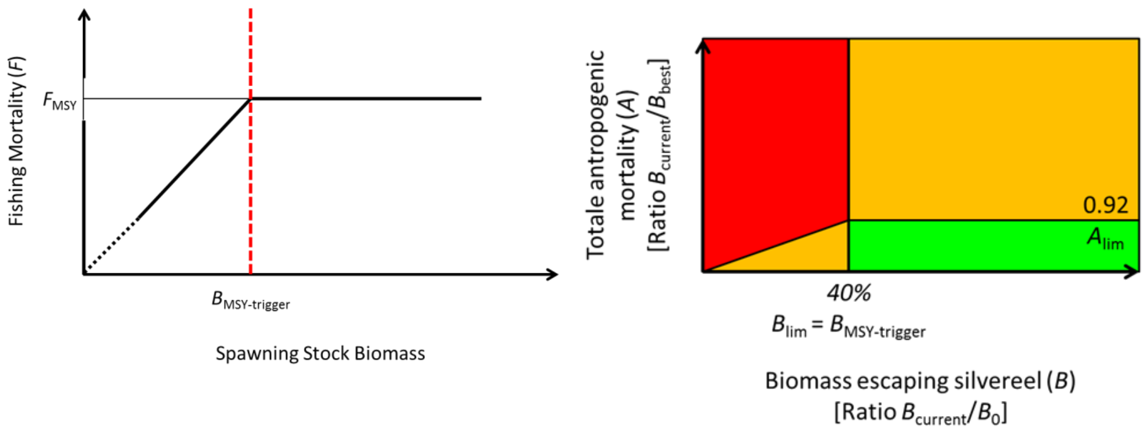

Figure 8.4 Suggested reduction in fishing mortality (F) if $B_{current}$ ≤ $B_{MSY\text{-}trigger}$ (left; ICES 2011c) according to ICES protocol. Suggested references points (A_{lim}, B_{lim}, $B_{MSY\text{-}trigger}$) and reduction in anthropogenic mortality if $B_{current}$ ≤ $B_{MSY\text{-}trigger}$ within the modified precautionary diagram (left; ICES 2011b) with respect to ***management*** *targets.*

As an initial option, ICES (2011b) recommended for eel to set $B_{MSY\text{-}trigger}$ at B_{lim}, and to reduce the mortality target below $B_{MSY\text{-}trigger}$ corresponding to ICES protocol (Figure 6.3). For long-lived stocks with population size estimates, ICES bases its advice on attaining an anthropogenic mortality rate at or below the mortality that corresponds to long-term biomass targets. However, $B_{MSY\text{-}trigger}$ is a biomass level triggering a more cautious response. Below $B_{MSY\text{-}trigger}$, the anthropogenic mortality advised is reduced, to reinforce the tendency for stocks to rebuild. Below $B_{MSY\text{-}trigger}$, ICES applies a proportional reduction in mortality reference values (i.e. a linear relation between the mortality rate advised and biomass; Figure 8.4).

The modified ICES precautionary diagram developed by ICES (2011b) needs to be carefully interpreted. The target biomass (40% B_0) has **not** been scientifically assessed to determine if it can be used as a true precautionary biological limit reference point. In other words, if all Member States were at 40% B_0 would the eel stock be considered to be "recovered"? Furthermore, as ICES (2011b) derived the A_{lim} from the Eel regulation's (management target) B_{lim} (40% B_0), with a reducing scale of A below B_{lim}, again there is no guarantee that if mortality is reduced below that level, the eel stock will recover.

Therefor the diagram is acceptable in principle to demonstrate the status of the eel stock with respect to the **management** targets/limits (40% B_0 and A_{lim} derived from 40% B_0) as formulated in the EC Eel Regulation, but ACOM has until now been reluctant to advise on the status of the eel stock without scientifically testing the targets/limits developed by ICES (2011b) to ensure they are precautionary and will lead to a recovery.

8.4 Evaluation

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).

2008			**2011**		
Stock Indicator	**Estimate**	**Source**	**Stock Indicator**	**Estimate**	**Source**
B_0*	10,400 t	EMP (2009)	B_0	10,400 t	EMP (2009)
B_{2008}	439 t	This report	B_{2011}	482 t	This report
B_{best}	2927 t	This report	B_{best}	1443 t	This report
ΣA	1.89	This report	ΣA	1.10	This report

* Excluding coastal waters (2600 t).

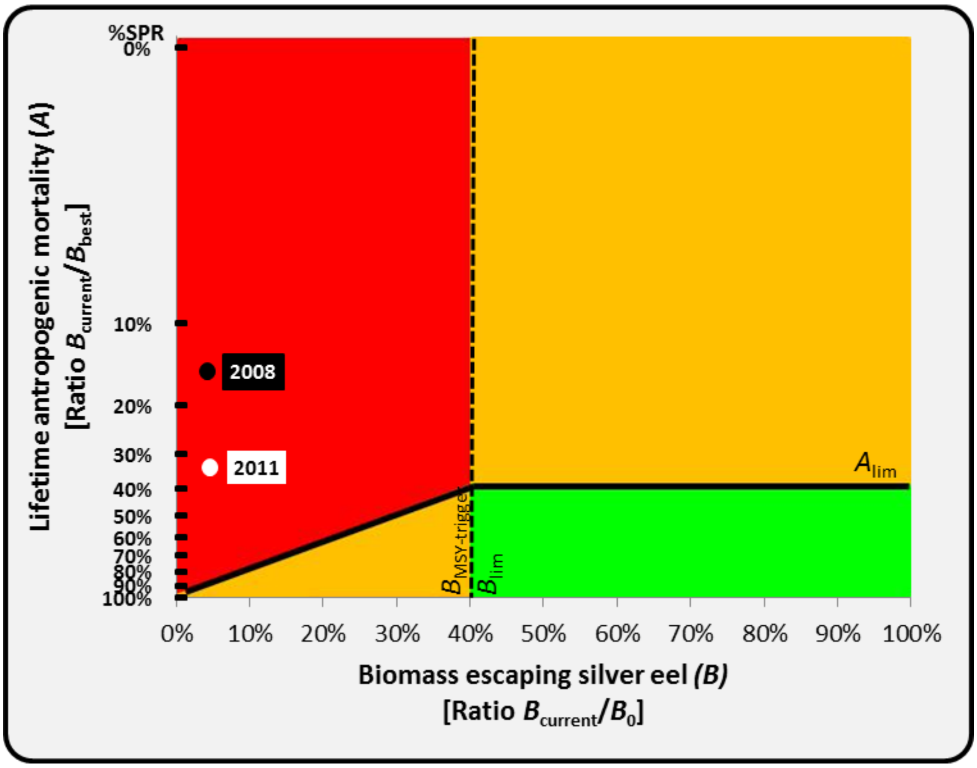

*Figure 8.5. ICES modified precautionary diagram presenting the status of the eel stock in The Netherlands in 2008 and 2011 with respect to **management** targets. The horizontal axis represents the status of the stock in relation to pristine conditions, while the vertical axis represents the impact made by anthropogenic mortality. %SPR = spawner potential ratio, a measure for the survival to silver eel relative to pristine conditions.*

The status of the eel population in 2008 and 2011 and hence, the evaluation of the Dutch Eel Management Plan is graphically presented in Figure 8.5, using the ICES Modified Precautionary Diagram with respect to the management targets from the EC Eel Regulation. The evaluation demonstrated that before and after the implementation of the EMP the status of eel in Dutch waters remained in a situation (high mortality, low biomass) regarded as "undesirable". Current biomass of escaping silver eel is below the target of 40% of the pristine situation and current anthropogenic mortality is above the recommended mortality at such low biomass of escaping silver eel (following the modified precautionary diagram developed by ICES 2011b).

Measures to reduce anthropogenic mortality are relatively quick and easy to implement and will directly result in measurable improvements (vertical axis). A reduction in anthropogenic mortality is therefore a good indicator of the drive and prowess of a member state. In The Netherlands the implementation of the EMP has resulted in a decrease in anthropogenic mortality (Figure 8.5). The observed reduction in anthropogenic mortality is almost solely the result of a decrease in fishery mortality, both commercial and recreational. Landings of both commercial and recreational fishery have been halved since the implementation of the EMP. The remaining measures (hydropower plants, pumping stations etc) have had so far limited measurable impact on a reduction in mortality.

Between 2008 and 2011, no noticeable increase in the biomass of escaping silver eel was observed (horizontal axis; Figure 8.5). An increase was also 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, if it had occurred after 2009, will at the earliest result in an increase of silver eel after 5-15 years (2014-2025).

The maximum that can be achieved by The Netherlands on the short term is a reduction of anthropogenic mortality to as close to zero as possible. If The Netherlands would reduce anthropogenic mortality to zero, there will be no guaranty that the European eel stock will truly recover. In order to achieve a genuine recovery of the eel stock, similar levels of protection of the 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 actual cause of the decline of the eel is still unknown. The European eel directive is only 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 a (lack of) changes in anthropogenic mortality (vertical axis) in the modified ICES precautionary diagram. However, whether in the long term an increase in escaping silver eel as a result of an increase in recruitment will be observed in The Netherlands is depending on the protective actions undertaken in other (non-)European countries. The Netherlands, like other countries share the responsibility for improvement of the biomass of escaping eel, the horizontal axis in the modified ICES precautionary diagram.

8.5 The uncertainties of the current evaluation

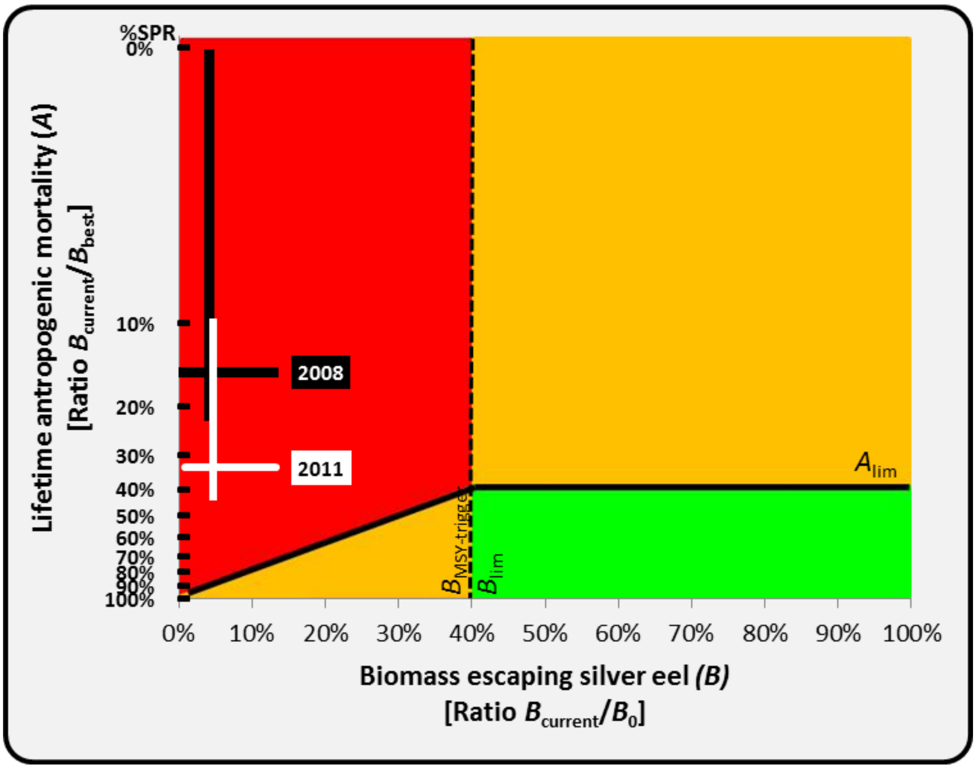

*Figure 8.6. ICES modified precautionary diagram illustrating the uncertainties around the biomass estimates of escaping silver eel (range B_0; Eijsackers 2009) and estimates of anthropogenic morality (scenarios 1-3; catch efficiency, densities eel in open water) in The Netherlands in 2008 and 2011 with respect to **management** targets. The horizontal axis represents the status of the stock in relation to pristine conditions, while the vertical axis represents the impact made by anthropogenic mortality. %SPR = spawner potential ratio, a measure for the survival to silver eel relative to pristine conditions.*

The estimates of the stock indicators B_{best}, $B_{current}$, B_0 and ∑A used to evaluate the status of the stock in Figure 8.5 need to be interpreted with care due to the significant level of uncertainty surrounding these estimates. The range of uncertainty around the stock indicators is visualized in Figure 8.6.

<u>Horizontal axis "biomass":</u> Initially the amount of silver eel biomass (B_0) that would have existed in The Netherlands if no anthropogenic influences had impacted the stock, was set at 10.000-15.000 ton with a target biomass (40% B_0) of 4000-6000 t (Klein Breteler 2008). Eijsackers et al. (2009) reviewed the calculations of Klein Breter (2008) and concluded that a target biomass (40% B_0) of silver eels lies realistically more between 2600-8100 t (or B_0 = 6500-20250 t). During the review of the national eel management plans (ICES 2009b), ICES did not accept all arguments of Eijsackers et al. (2009) and set B_0 at 13000 t with a corresponding escapement target of 5200 t. On the horizontal axis the range around the estimate for the biomass of escaping silver eel is set by the values of B_0 (6500 - 20250 t; Eijsackers et al., 2009; EMP, 2009) and $B_{current}$ (Table 7.3 for 2011 and Table 7.4 for 2008).

<u>Vertical axis "mortality":</u> On the vertical axis the range around the estimates for lifetime anthropogenic mortality set by the values of B_{best} but especially $B_{current}$ (Table 5.10 for 2011 and Table 5.11 for 2008). $B_{current}$ is strongly influenced by assumptions on the efficiency of the electrofishing gear, distribution of eel over the total surface of a water body in the static spatial population model and assumptions of *F* when estimating eel populations using the dynamic population model for some of the larger lakes (scenario's 1-3, see Chapter 4).

Finally the estimated lifetime anthropogenic mortality in this report is most likely an *underestimate* of the true life time anthropogenic mortality as not all sources of mortality have been quantified and accounted for:

- catch-&-release mortality of the recreational fishery
- yellow eel mortality in hydropower plants and pumping stations
- poaching
- the "unknown" mortality observed in the telemetry studies in the Rivier Meuse (Table 5.9)
- 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 using the modified ICES precautionary diagram, it is of utmost importance 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 ∑A, and
- the unquantified sources of anthropogenic mortality.

9. Conclusions and recommendations

In this report, we have described the data and methods which were used to estimate the stock indicators B_{best}, $B_{current}$, B_0 and ∑A, which were requested by the European Commission to evaluate the status of the European eel stock. Here, we highlight the main advantages and disadvantages and uncertainties of the used methodologies and provide recommendations for future improvements of the models.

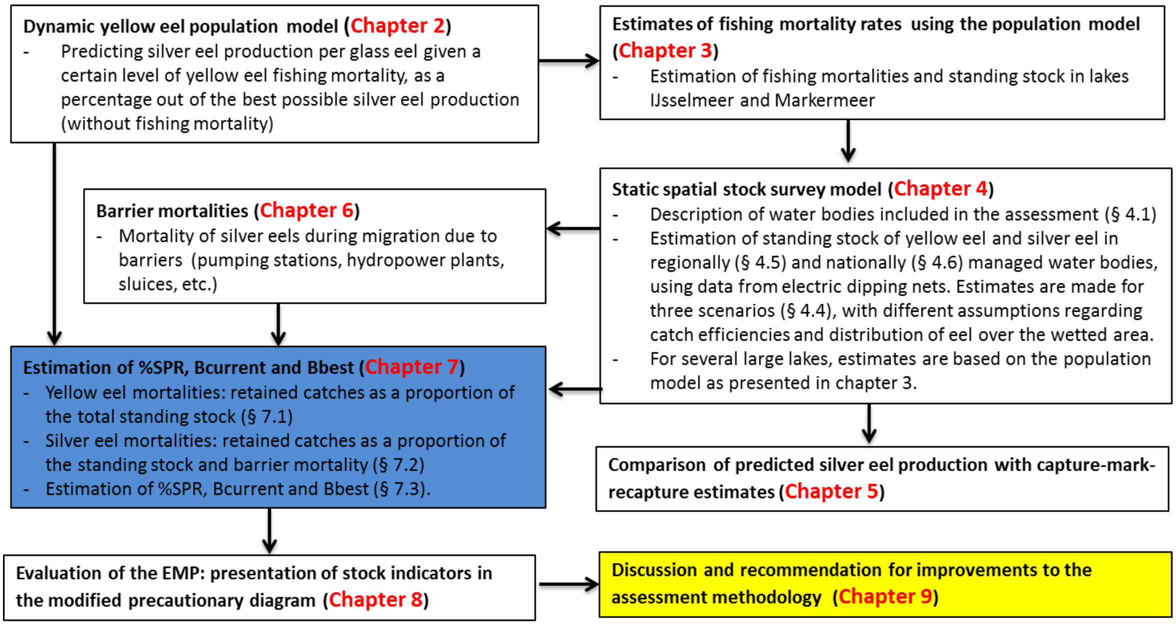

Figure 9.1. A flow diagram representing the key steps in the stock assessment methodology. Chapter 7 is highlighted in blue, because this is where the overall assessment is made for The Netherlands, using the results from the other chapters.

9.1 Dynamic yellow eel population model

The eel population model used in this study has been parameterised using market sampling data on maturity-at-length by sex, weight-at-length by sex, and sex-specific growth curves. This model has played an important role in the final assessment by estimating the contribution of anthropogenic mortality during the yellow eel stage to Lifetime Anthropogenic Mortality (LAM). Also, an attempt was made to estimate mortality rates by comparing ('fitting') model predictions to observed historical stock trends and present-day length-frequency distributions.

The impact of yellow eel mortality on silver eel production was estimated using the population model as presented in Chapter 2. The population model incorporates the key biological processes of eels during the yellow eel stage; sex-specific growth, death and sex-specific maturation. However, this model is necessarily an abstraction of reality. A number of modelling assumptions were made, and varying these modelling assumptions may lead to different estimates of silver eel production. One key assumption to which estimates may be sensitive is that all eels of the same sex are assumed to follow the same average growth curve (paragraph 2.3.2) and have the same average maturation-at-length transition rates (paragraph 2.3.1). Allowing for variation in these curves between individuals may not lead to the same estimates as using an average growth curve.

Uncertainty and possible biases in the model estimates may also arise from the parameter estimates. The following parameters are crucial:

Sex-ratio. The higher the proportion of females, the higher the expected silver eel biomass per glass eel. Estimates of fishing mortalities obtained by fitting model predictions to observed stock trends will increase with increasing proportions of females given that they grow larger and older and (after several years of age) faster than males.

Growth rates. The higher the assumed growth rate the higher, the expected silver eel biomass per glass eel. Estimates of fishing mortalities obtained by fitting model predictions to observed stock trends will increase for increasing growth rates (see also Dekker 2000). In general, the population model could be improved by allowing for variation between individuals in growth rates. Allowing for such variation could lead to different model predictions than just assuming an average growth rate. For example, given high mortalities, eels that grow at average speed will have a low expected spawner production to recruit ratio. If there are some fast growing individuals, these will have a much larger expected spawner production to recruit ratio.

Maturation-at-age. If individuals are assumed to mature at younger ages or lengths, estimates of mortalities will decrease (and vice versa). The proportions of silver eel out of all eels in the retained catches have been used for estimating transition maturation-at-length rates. It is highly recommended that a better basis for estimation of maturation-at-length be made in future, although it is not clear at this point what the best way to do this is.

Different estimates of growth and maturation-at-length were used in the other studies. Witteveen & Bos (2010c), van der Meer (2010) and Dekker (2000) used a constant 3.5 cm per year as a growth curve for both sexes. In contrast, in this report a sex-specific growth curve based on otolith readings is used in which individuals grow faster in the first few years of their lives as yellow eels. With respect to maturation-at-length, eels were assumed to mature at lengths of 45 (males) and 65 cm (females) in van der Meer (2010). The estimates in Dekker (2000) cannot easily be reproduced because they are presented in graphical form. It is impossible to decide at this point which of the models gives the best predictions, given the uncertainty and possible biases in the data used to estimate vital parameters in the models such as sex-specific growth rates, weight-at-length, and maturation-at-length (paragraph 2.3). However, we prefer the model presented in this report on the grounds that is parameterised using extensive recent data of the key biological parameters, collected on eels in The Netherlands.

Given that it is difficult to obtain unbiased estimates of these crucial parameters, the estimates of spawner-to-recruit ratios (Chapter 2) and fishing mortalities based on eel population models (Chapter 3) remain uncertain and require careful interpretation.

The usefulness of the eel population model for estimating fishing mortalities using only relative length-frequencies is limited. To interpret present-day data or historical stock trends, a good index of recruitment, trends in sex-ratios, sex specific growth rates, natural mortalities, and migration rates between linked water bodies are required. Because eel recruitment has fallen sharply, it is probably unrealistic to assume that vital parameters have remained constant. In many water bodies in The Netherlands, length-frequency distributions have shifted towards on average large individuals; mean lengths in stock surveys have been increasing in Lakes IJsselmeer, Markermeer and upper reaches of the main rivers. However, a multitude of factors may have caused this, such as decreased natural or anthropogenic yellow eel mortality, a decreased recruitment rate, and an increase in the proportion of females.

The best studied and most data-rich situation is provided by lakes IJsselmeer and Markermeer for which good recruitment indices and long-term length disaggregated stock surveys are available. The main stock trends in lake IJsselmeer; a decreasing recruitment of young eels and an absolute increase in the abundance of larger eels, could be explained reasonably well by the model, mainly by decreasing the estimated fishing mortality over time. However, fishing mortality was the only parameter that was allowed to change over time, and estimates of fishing mortalities were highly sensitive to the assumed sex-ratio. For Lake Markermeer, the trend in increasing mean length could not be captured well by the population model; an initial increase in mean length was followed by a decrease. The interpretation of the stock trend in lake Markermeer was therefore different compared to lake IJsselmeer; no or little estimated decrease in mortalities but a temporary increase in mean length due to reduced recruitment. A final problem in the interpretation of the stock survey data is that no distinction is made between yellow eel and silver eels, whereas the model should only be fitted to yellow eel data which are assumed to be “produced” locally. An alternative interpretation of the observed stock trends is therefore that they were caused by immigration of silver eels from nearby water bodies such as the Veluwerandmeren or the river IJssel. Therefore, estimates of fishing mortalities, even for these “data rich” lakes, using the population model remain highly uncertain, and may be misleading.

Data on relative length-frequencies, from one or a few years, for the reasons outlined in Chapter 3 cannot be used to infer mortalities in a reliable and robust manner. The market sampling data has been useful and necessary to collect the data on biological parameters for the parameterisation of the eel population model. However, it is not possible to underpin national or regional eel fisheries management strategies using relative length frequency distributions as a basis for estimating fishing mortalities.

9.2 Static spatial population model

Anthropogenic mortalities during the yellow eel stage have been estimated using the proportion of the estimated retained catches from commercial and recreational fishers out of the total estimated standing stock of yellow eel with lengths above 30 cm (both types of fisheries are assumed fully selective from 30 cm onwards and fully unselective for eels with lengths below 30 cm). Estimates of the standing stock of yellow eel were in most cases based on fisheries independent survey data, where catches per unit of effort from electric dipping nets (and for some water bodies also other gears) were used to estimate total standing stocks by scaling to total wetted areas of water bodies.

The main advantages for estimating standing stocks 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 and outwith of 1.5 meters of the shore/bank). Experiments can be devised which can test the assumptions and to obtain better estimates.
- The estimates, because they are specific to water bodies, 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.
- This modelling approach can be extended to other species than eel.

The main weakness in the methodology is the uncertainty surrounding estimates of catch efficiencies and scaling of density over available wetted area of water bodies which are crucial parameters in the eel stock assessment. This lack of knowledge results in uncertainty around the estimates of standings stocks as reflected by the range in predictions using the three scenarios with differing catch efficiencies and eel distributions. To improve the quality of the method, we recommend to collect more information on these crucial parameters. A further necessary improvement to the survey model is to obtain better estimates of densities and catch efficiencies of electric dipping nets of eel in ditches as ditches are hardly sampled but do constitute a large area (330 km^2).

9.2.1 Regionally managed waters

In the biomass assessment for the regional managed water bodies choices and assumptions were made based on data availability, time constraints and practicalities. The main problem with WFD fish survey data is the lack of accessibility due to the lack of a central database. For example, not all water bodies were sampled within the WFD sampling program at this point, and not all regional water boards provided sample data, all of which were not available for this analysis. In addition, in the obtained data set data points were present that could not be linked to a water body (nearly 30%), and these were excluded from analysis. Such a mis-match might be due to measurement error in GPS equipment or errors in data entry, such as missing entries or typing errors in water body names. The data eventually used in this assessment were not screened in detail for typing errors and errors in for example identification of water types due to time constraints.

As part of the WFD fish monitoring protocol, data on habitat is collected. However, in the current assessment WFD eel densities in regionally managed waters were not corrected for habitat.

9.2.2 Nationally managed waters

There are some shortcomings and uncertainties in the methodology used for the nationally managed waters. The main rivers in the nationally managed waters have non-connected water bodies that are in the river flood plain, within the winter dykes of the river, and are only connected to the river for a short period of time per year. No eel surveys are conducted in these waters and these water bodies are excluded in the current assessment. However, the surface area of these non-connected water bodies (64 ha) is probably negligible.

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

Different river regions are surveyed in different months. As a result different mean water temperatures, different eel behaviour and different silver eel migration activity, may have influenced the observed densities.

In the current assessment the standing stock of eel in the lakes IJsselmeer, Markermeer, Veluwerandmeren and Grevelingen was determined using the dynamic population model due to the lack of adequate survey data. Capture-mark-recapture studies could provide independent estimates of standing stock in these lakes. A better understanding of the distribution of eel over the surface of these large lakes may allow for the standard up scaling of eel densities from the littoral zone over the whole lake as described in the static spatial model. An alternative could be the development of an electro-beam trawl designed to effectively capture eel >30 cm length in large lakes and wide rivers.

9.3 Silver eel migration model

The model to estimate mortality of silver eels during migration needs to be further evaluated and developed. Whilst several maps and lists of barriers are available (e.g. Kroes et al., 2008; Buijse et al., 2009) formal models to estimate mortalities during passage of fish populations are rare, and the model presented in this report is a start. In particular, the characterisation of water bodies as polder or boezem waters need to 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 model could be based upon the methodology described in Appendix A to estimate migration routes. Whilst this initial attempt to construct the computing routines failed because the computed routes were unrealistic in too many cases, recommendations are made to further improve this spatially explicit route analysis. This spatially explicit route analysis has the potential not only to provide a better estimate of silver eel escapement, but also to identify migration bottlenecks based on biomass of silver eel passing. Moreover, this spatial model is not limited to silver eel migration but can be used for other migratory fish species.

9.4 Estimating lifetime anthropogenic mortalities (LAM)

The mortalities during the yellow eel stage caused by commercial and recreational fishers contribute the most to the estimated LAM. This can be explained by the fact that eels take many years to mature. Also, because escapement is expressed in terms of biomass, females, which take on average longest to mature, contribute most to this. Mortality during the yellow eel stage prevents eels from developing into silver eels. Silver eel mortalities caused by barriers and retained catches can only act on the resulting proportion out of the best possible production of silver eel (if all anthropogenic mortalities during the yellow eel stage were mitigated). This means that the present-day estimated contribution of mortality caused by retained catches of silver eel and barrier mortality contribute relative little to the estimated LAM. The estimated decrease in mortality over the period before and after the EMP attributed foremost to the realised decrease in yellow eel mortality.

Yellow eel and silver eel mortalities have been estimated using the estimated retained catches and barrier mortalities in relation to the estimated standing stock of eel in recent years (2008-2011). The overall estimated LAM is not the same as the LAM that new recruits (glass eels) arriving in 2012 are expected to experience throughout their life span until escapement to the sea as a silver eel. In particular, the effect of the closed areas (main rivers and some large canals) has not yet fully materialised; in the near future the eel stock may increase in the closed area relative to the eel stock in other parts of The Netherlands due to decreased mortality. If landings from other parts of The Netherlands stay the same relative to the stock size, then the mean fishing mortality may be expected to decrease in the near future. However, as discussed above, no empirical evidence has been found as yet of reduced mortalities during silver eel migration on the Meuse river due to the closing of the commercial fisheries (Table 6.7).
We note that the estimated lifetime anthropogenic mortalities in this report are most likely *underestimates* of the true life time anthropogenic mortality as not all sources of mortality have been quantified and accounted for:

- catch-&-release mortality of the recreational fishery
- yellow eel mortality in hydropower plants and pumping stations
- poaching
- the "unknown" mortality observed in the telemetry studies in the Rivier Meuse (Table 6.7)
- impact of human-induced viruses, parasites and pollution

9.5 Evaluation of the EMP

The estimated key stock indicators B_{best}, $B_{current}$, B_0 and ΣA have been evaluated in relation to management targets/limits (40% B_0 and A_{lim} derived from 40% B_0) as formulated in the EC Eel Regulation, using the modified precautionary diagram (Chapter 6). The modified ICES precautionary diagram developed by ICES (2011b) needs to be carefully interpreted. The target biomass (40% B_0) is a **management** target and has **not** been scientifically assessed to determine if it can be used as a true precautionary biological limit reference point. In other words, if all Member States were at 40% B_0 would the eel stock be considered to be "recovered"? Furthermore, as ICES (2011b) derived the A_{lim} from the

Eel regulation's (management target) B_{lim} (40% B_0), with a reducing scale of A below B_{lim}, again there is no guarantee that if mortality is reduced below that level, the eel stock will recover.

Therefor the diagram is acceptable in principle to demonstrate the status of the eel stock with respect to the **management** targets/limits (40% B_0 and A_{lim} derived from 40% B_0) as formulated in the EC Eel Regulation. However, the Advisory Committee (ACOM) of ICES has until now been reluctant to advise on the status of the eel stock without scientifically testing the targets/limits developed by ICES (2011b) 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 of utmost importance 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 ΣA, and
- the unquantified sources of anthropogenic mortality.

9.6 Recommendations

During the development of the current models, the main weaknesses of the current methodology surfaced quickly. Here we list the main recommendations to improve the quality of the assessment before the next evaluation in 2015.

Dynamic Population Model

Key biological parameters: improve to quality of the following key biological parameters
Sex-ratio: Sex ratios could be improved by using eels smaller than 30 cm. These eels could be obtained during the WFD fish sampling.

Growth rate: Growth rates could be improved by including eels smaller than 30 cm. These eels could be obtained during WFD fish sampling.

Maturation-at-age: The silvering ogive for a given area could be improved by using data collected year round. Furthermore, it is recommended to record the stage of the eel (yellow/silver) during research surveys (e.g. IJsselmeer electro-trawl survey)

Anthropogenic mortalities: quantify sources of anthropogenic mortalities that are excluded from the current assessments; 1) catch-&-release mortality of recreational fisheries, 2) yellow eel mortality pumping stations and hydropower plants.

Static Spatial Model

WFD survey data: improve the accessibility of WFD fish survey data of regionally managed waters 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: conduct experiments to determine efficiencies of electrofishing for eel in different WFD water types in both nationally and regionally managed waters.

Spatial distribution: conduct experiments to determine the spatial distribution of eel in wide rivers and lakes in both nationally and regionally managed waters.

Ditches: conduct elecrofishing surveys for eel in ditches to supplement the existing WFD eel survey data in regionally managed waters

Habitat: correct eel densities for habitat in nationally and regionally managed waters

Electro-beam trawl: develop an electro-beam trawl to provide reliable estimates of eel (>30 cm) densities in large lakes and wide rivers.

Silver Eel Migration Model

Migration routes: finalise the GIS model (Appendix A) to improve the estimate of silver eel mortality during migration

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 unclear at the time of the writing of this report whether these mortalities have been included in the LAM of silver eels that were produced in German and/or Belgian waters. It is recommended that come to an agreement on how these mortalities should be accounted for.

Furthermore, 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 years to come.

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Quality Assurance

[In this section, the Environmental Group must provide:
- An indication that CRM and/or IRM have been used for quality control. A survey of the results can possibly provide indications on the certified amount of IRM cq CRM standards.
- An indication that is based on ring research. A survey on the Z/scores which have been obtained may also be provided.
- An indication on the validation, detection and reproduction parameters. Deze kunnen uit het validatierapport van de bepaling gehaald worden. Daarbij verdient het aanbeveling om na te gaan of deze gegevens nog relevant zijn binnen het uitgevoerde onderzoek. Het kan bijvoorbeeld voorkomen dat de detectiegrens in het onderzoek anders ligt dan in de validatiestudie doordat er meer of minder monster is ingewogen.
- Indien men geen gedetailleerde informatie verstrekt door validatieparameters kan worden volstaan met een verwijzing naar het kwaliteitshandboek van de afdeling Milieu.
- Decimal characters: Data is in derogation Dutch SI reported a decimal point (.) Instead of a comma (,).
- When work is outsourced this should be stated in the report, including name and address of the company that performed the outsourced work.
- When measurement uncertainties are not listed then they can always be requested by the customer.

Additional research studies can be mentioned]

IMARES utilises an ISO 9001:2008 certified quality management system (certificate number: 57846-2009-AQ-NLD-RvA). This certificate is valid until 15 December 2012. The organisation has been certified since 27 February 2001. The certification was issued by DNV Certification B.V. Furthermore, the chemical laboratory of the Environmental Division has NEN-AND-ISO/IEC 17025:2005 accreditation for test laboratories with number L097. This accreditation is valid until 27 March 2013 and was first issued on 27 March 1997. Accreditation was granted by the Council for Accreditation.

Justification

Report number: C067/12
Project number: 4308501014

The scientific quality of this report has been peer reviewed by the a colleague scientist and the head of the department of IMARES.

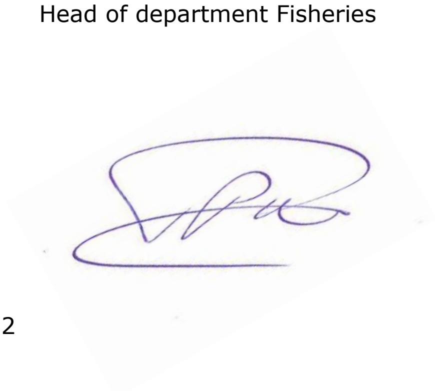
Approved: Dr. J.J. Poos
Researcher

Signature:

Date: 21st June 2012

Approved: Dr. ir. T.P. Bult
Head of department Fisheries

Signature:



Date: 21st June 2012

Appendix A. GIS model to compute possible migration routes of silver eels

For the assessment of silver eel biomass escape from Dutch waters the development of a model was started to include migration routes and barriers along migration routes. Deltares, in the person of Gert-Jan Geerling, was asked to collaborate on this model to provide GIS specialists for the development of routes and maps. The aim was to obtain per WFD water body the distance to the nearest exit point to sea and all barriers along the route. For each barrier an eel mortality estimate was made (see Chapter 6). Silver eel escapement to sea would then follow from the silver eel biomass estimate for each water body and the cumulative barrier mortality for each barrier on the route to sea.
However, due to technical issues and the short time frame this model was not ready for use for this eel assessment. The current version resulted in too many unrealistic migration routes. Nevertheless, recommendations are provided at this stage and the development of the model will be continued.

The following memo is a report on the work done by Gert-Jan Geerling of Deltares and includes a description of data sources, techniques and a discussion on the overall approach.

Memo

A.1 Base data

The base data used are shape files of the Water Framework Directive (WFD) water bodies (WB) in The Netherlands. The water bodies are divided over two shape files, one containing the line WB shapes and the other containing polygon WB shapes. Figure 1 shows the WBs in The Netherlands. All calculations and analyses were done using ArcGIS 10.

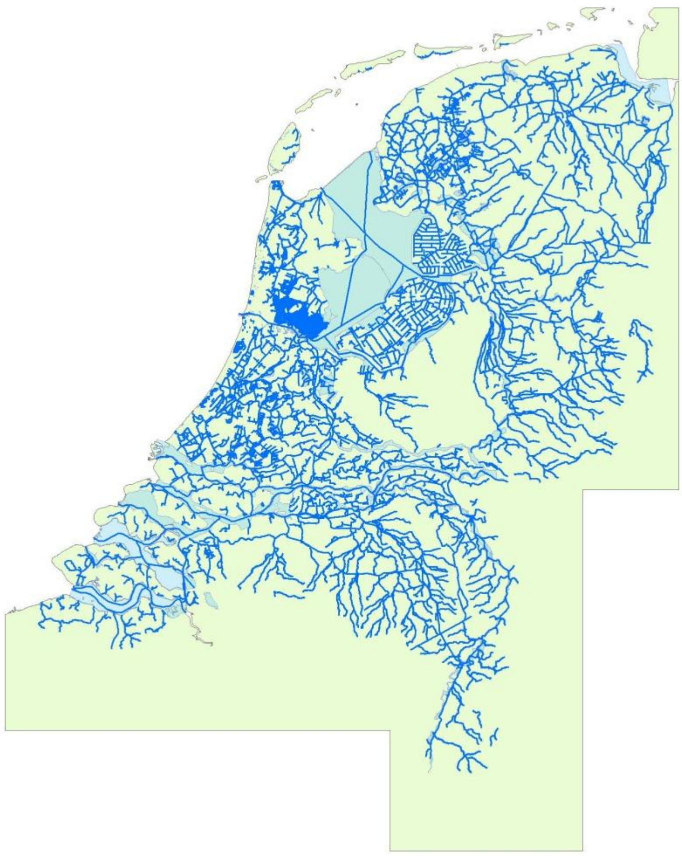


Figure 1. Inland Water Bodies (WB) in The Netherlands.

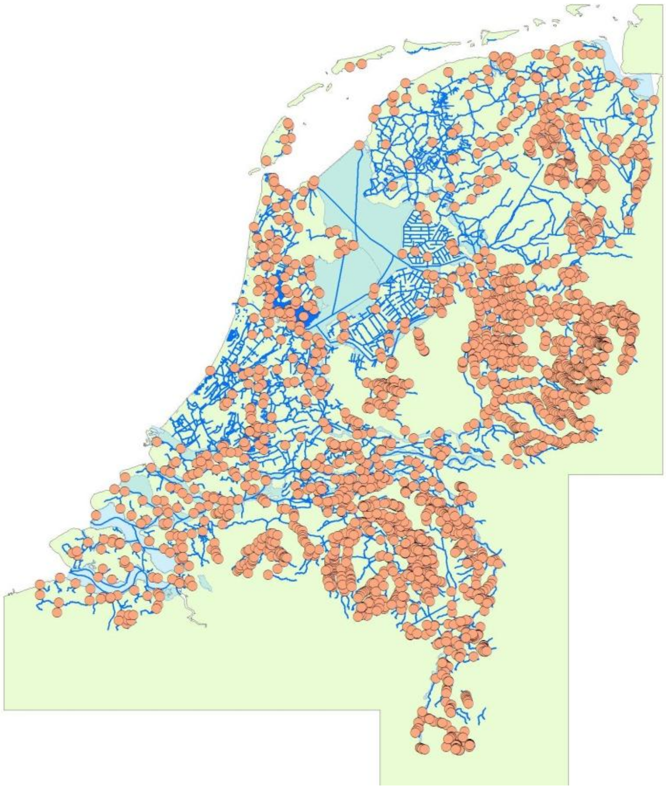


Figure 2. Locations of barriers (source: sportvisserij Nederland).

Base data used: Barrier database vismigratie.nl, Sportvisserij Nederland, 2009. Exit location points for silver eels, Imares, 2012. Ecotope maps, Rijkswaterstaat, 2005. Waterbodies v 2061, Waterportaal, downloaded, 11-11-2011.

A.2 Area, length and typology of water bodies

For the estimate of the amount of potential eel in Dutch WBs the WFD typology was used as a basic classification of the WBs. The WB classification gives information on the stream order, general flow velocity and substrate of the WBs. To obtain the total surface area of WBs, the length of line shapes was calculated and the width was derived from the WB classification. For polygon shapes the surface area and perimeter were calculated.

It was noted that the shapes of large rivers (most notably the types R7, R8, R16) were delineated by their floodplains. The resulting perimeter would not reflect the actual shoreline length but the length of the winter dyke. For the large rivers the ecotope maps (issued by Rijkswaterstaat) were used to calculate actual shoreline lengths of main channel and attached side channels as one group and isolated floodplain waters as another. Of the main channel and side channels the type of shoreline, i.e. artificial or otherwise, was included in the data and reported in meter (m) shoreline.

Base data information: ecotope maps (2005) second edition, source RWS.

A.3 Migration route analysis

A.3.1 Overview

The amount of eel that migrates seawards is based on estimates of the amount of inland eel and the fraction that survives the barriers encountered during seaward migration. To be able to make these estimates the migration route itself has to be known. For this, the shortest route approach is adopted. By calculating the shortest route from a WB to the nearest silver eel exit location both the migration distance and the barriers encountered can be estimated. The distance between barriers and the nearest eel exit locations is used to identify the order of encountered barriers. Basically the following data were created:

- Routing raster of WBs, distance raster, migration direction raster.
- Distance tables of min, max, mean and standard deviation of distance between WB and nearest exit location. The min, max and standard deviation of migration distances are only meaningful for large water bodies.
- Distances of each barrier to nearest exit location.
- Crosstable of barriers encountered on the migration route from each WB to the nearest exit route.

A.3.1 Raster schematisation used

Of the available technical solutions that could be applied for the analysis problem, the raster approach was found to be the most feasible. Upon close inspection, the network analysis (vector) approach needed much manual alteration of the available vector data to ensure a working model while the inclusion of polygon surfaces in such a (line based) network model also posed a challenge.

The water bodies were rasterised in 5m grid cells and to reduce the computation time cell size was enlarged to 125m raster cells what seemed to give enough resolution for most areas. Computation time was especially important as the route for every WB needed to be computed separately to select the barriers on this route, in practice up to 6000 times. The area above Amsterdam was exceptionally detailed in the WFD water body map, see figure below. This amount of detail was deemed too high for a country wide schematization and estimate. Note: in the estimate of amounts of eel based on surface area en lengths of water bodies these highly detailed data were applied (paragraph 2). Only for the route calculation the cruder raster schematization was used.

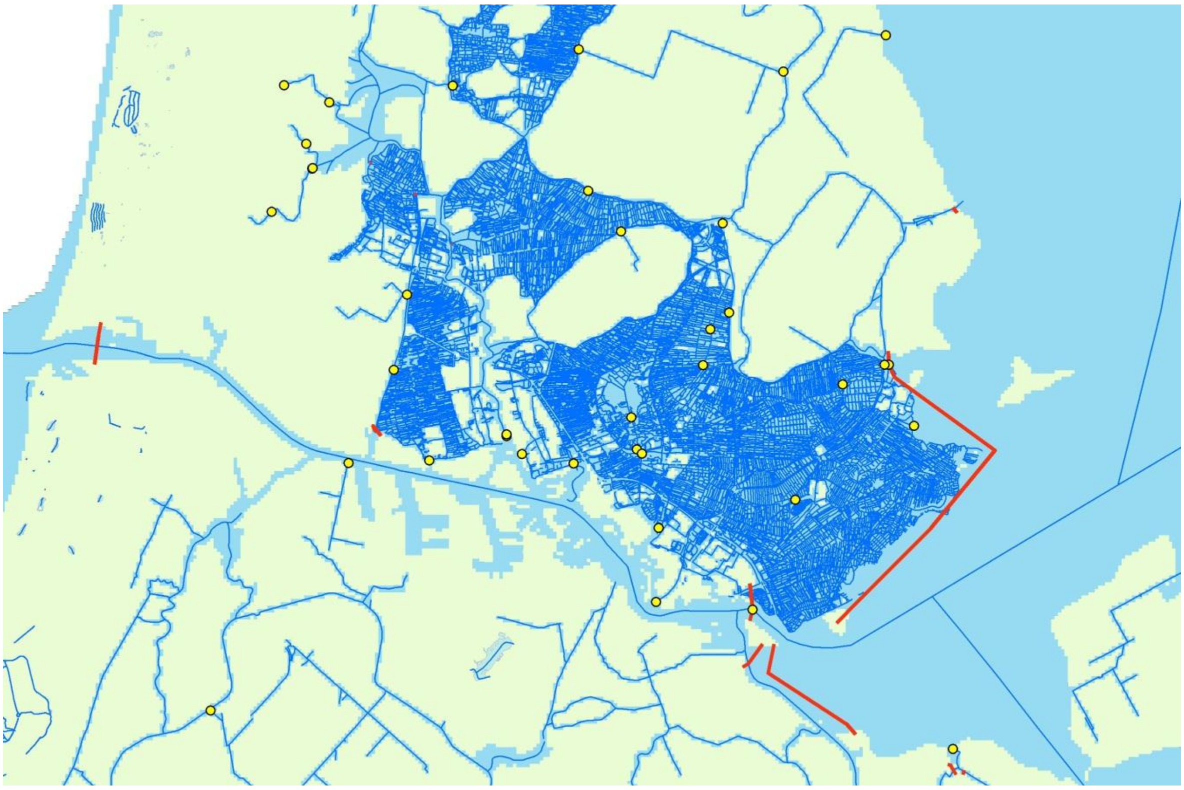


Figure 3. Original Water body lines (dark blue) and rasterised schematic (light blue). The exceptional level of detail of water bodies above Amsterdam is clearly visible, also compare with Figure 1. The yellow dots and red lines depict barriers.

A.3.2 Barrier modeling

Base data for the barrier information were the data collected by the 'sportvisserij Nederland'. These data contain all barriers that have been submitted to Sportvisserij Nederland. The dataset contains 2694 barriers, of these 22 had no coordinates. 1040 barriers are located directly on a line shaped water body; additionally 843 are located within 5m from a WFD-water body and 92 between 5 and 10m. 655 barriers are located more than 10m from a WB, these were omitted as these in randomly checked cases belonged to small ditches that were no part of a WFD water body. In a few cases the barriers consisted of uneven crossings of two water bodies, this could not be incorporated in the raster model.
Most barriers are situated in small streams and are points in the GIS. A selection buffer around a route of 100m was applied to make sure no route passes a barrier without selection. In polygon shaped water bodies more than 100m wide and in areas with a high density of canals and streams, barrier selection might fail. All barriers in polygon shaped water bodies (145) were checked and when necessary replaced by lines, an overview (Figure 4) and detailed examples (Figures 5 and 6) are given in figures. In the figures, the red lines depict barriers that were schematized as lines.

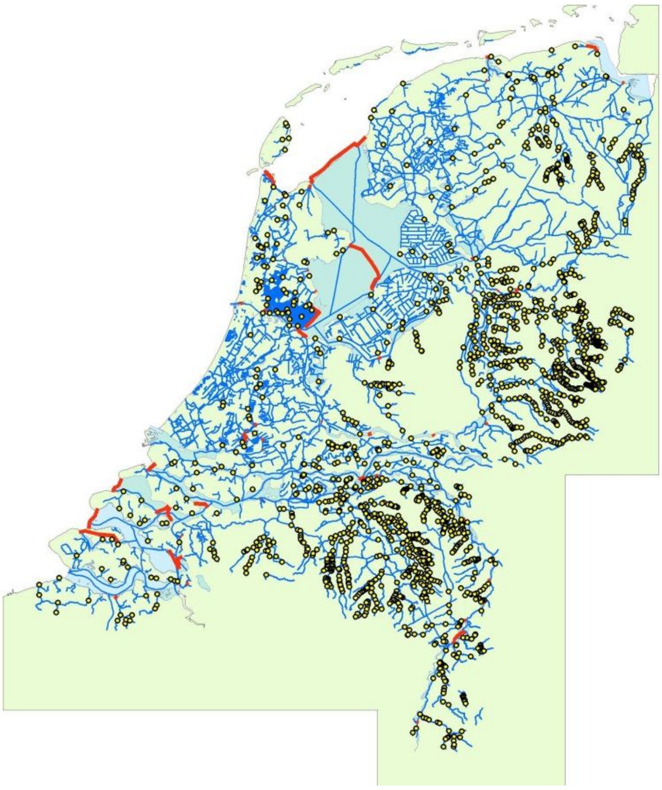


Figure 4. Overview of all barriers used (lines in red, points in yellow).

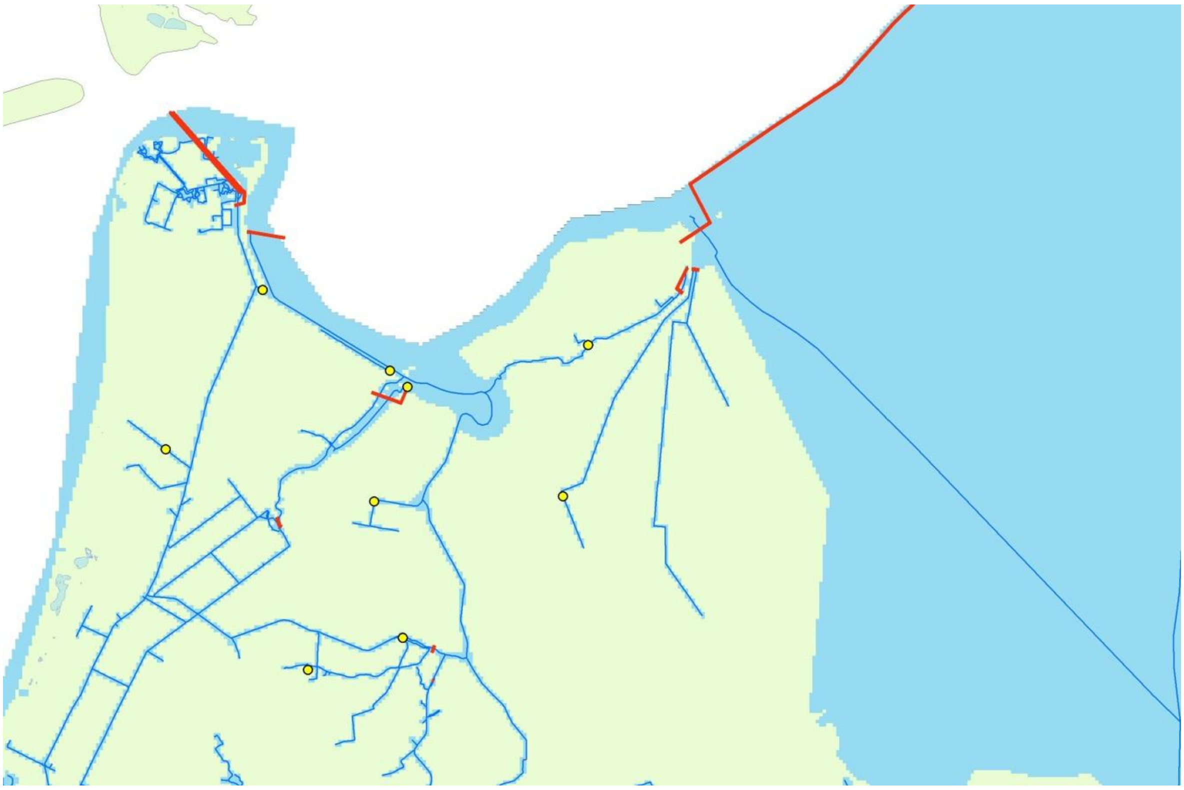


Figure 5. Line barriers (red) and point barriers (yellow) in Noord Holland and Afsluitdijk. Water bodies are shown in blue.

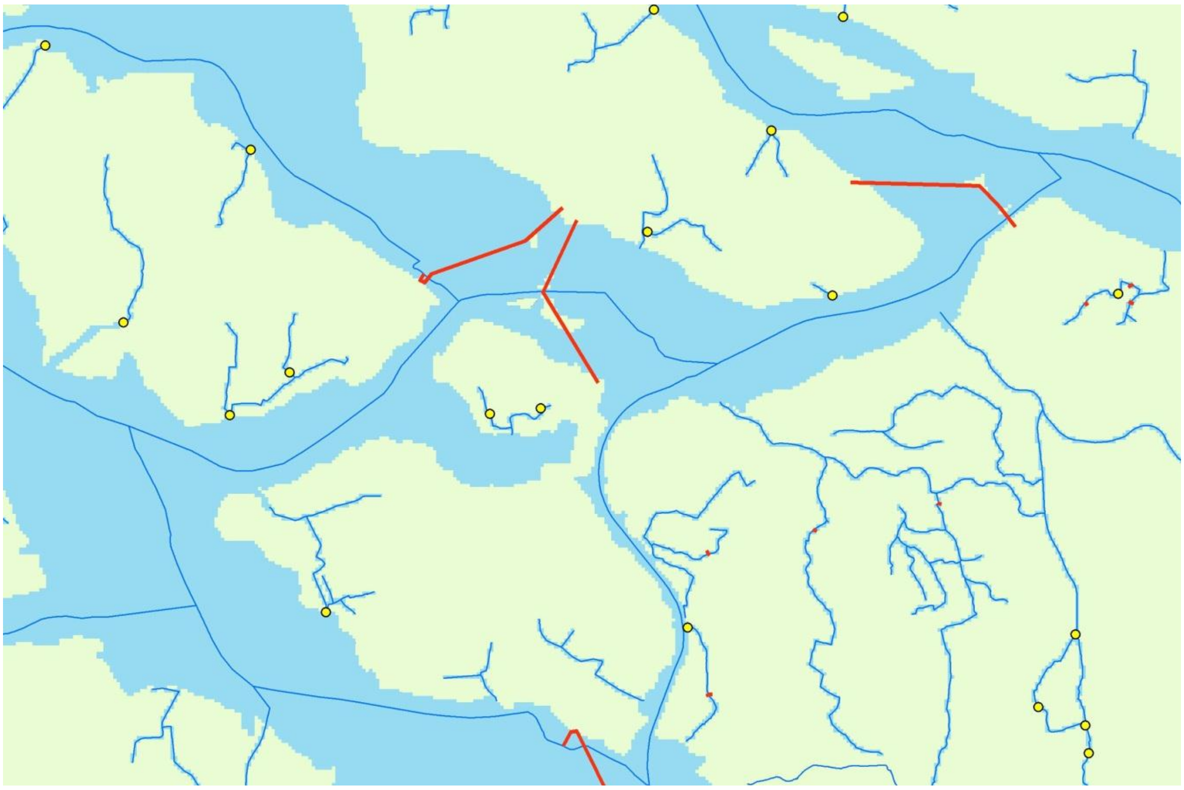


Figure 6. Line barriers (red) and point barriers (yellow) in Zeeland. Water bodies are shown in blue.

A.3.4 Distance maps

Using the cost distance function (ArcGIS) and the rasterised WBs, the distance of each raster cell to the nearest exit location is computed (Figure 7). Based on these data the distance of each WB and barrier is determined. Furthermore the algorithm computes a backlink raster that shows migration direction. This also can be enforced, a feature that can be applied to incorporate for example stream direction in a future iteration of the analysis.

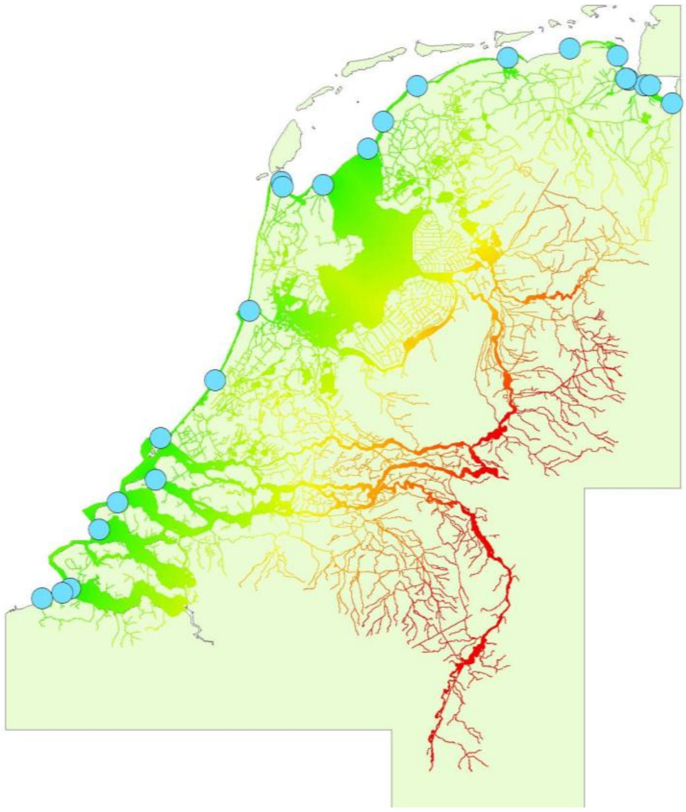


Figure 7. Example result for the distance from each point to the nearest migration exit location (red is the farthest, yellow and green closer to exit locations; exit locations are depicted as blue points).

A.4 Discussion

The shortest route approach did not in all cases give the most logical results (Figure 8). The real migration routes of eels are not known and can only be assumed, but it seems the shortest route approach is in some cases an underestimation of the probable migration distance followed by eels. Causes for this are two-fold. In areas with a high density of isolated water bodies, some water bodies are connected in the raster schematisation that would be isolated in reality. Secondly, the shortest routes use regional waters if this route is indeed shorter.

For the estimation of the eel survival, the barriers encountered along the shortest route between a WB and the nearest exit location are important. The type, amount and order of barriers encountered determine the eel survival. After consideration of the results, it is thought that the shortest route option for WBs, mostly in the eastern and southern Netherlands, gives an inaccurate selection of barriers. The results for the barrier selection were not used in this study to estimate silver eel survival.

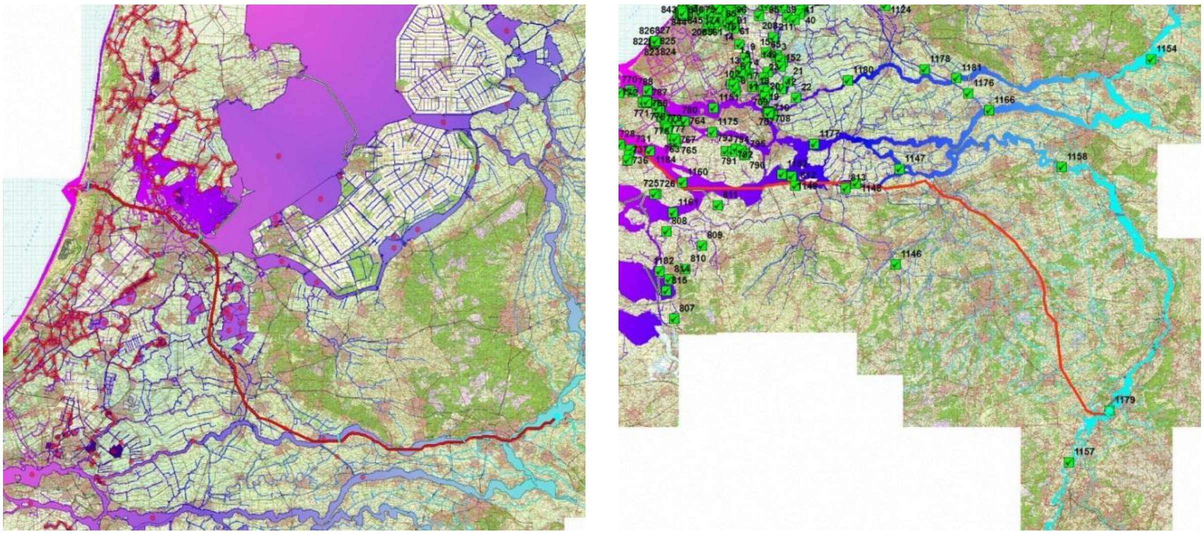


Figure 8. Example routes based on the shortest route approach. A lot of routes follow a "logical" path, however some seem not the most appropriate such as the routes shown in the examples above. The shortest routes here use 'shortcuts' through regional waters.

Possibilities for improvement are:

- To improve the raster model in spatial detail.
- To give larger water bodies that represent (an assumed) preferred route a lower transport friction in the cost-path analysis. This will affect the route chosen by the model.
- To use vector data (network analysis). This is more time consuming and gives the additional challenge of incorporating polygons in a (line based) network. But has the benefit that streams are not spatially simplified.
  - Incorporate 'eel behaviour' in the route modelling. For example divide the migration load over several route possibilities; apply a weighted distribution function at stream intersections based on relative flow capacity; try and incorporate flow direction.
  - Sportvisserij Nederland will also improve the barriers dataset.