Implementation of UK Eel Management Plans (2020–2023)

June 2024

Acknowledgement:

This progress report has been produced jointly for all Administrations by the personnel from the Environment Agency (EA), Natural Resources Wales (NRW), Agri-Food and Biosciences Institute (AFBI) and Scottish Government's Marine Directorate, under coordination of the Centre for Environment, Fisheries and Aquaculture Science (Cefas) at Lowestoft. The monitoring and assessment of eel stocks throughout the UK was funded by the Department for Environment, Food & Rural Affairs (Defra), Department of Agriculture, Environment and Rural Affairs (DAERA), Welsh Government and Scottish Government. All the participating organisations would like to extend their thanks and recognition to the various operational staff who have collected and compiled the data for this report.

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Executive summary

This report outlines the monitoring, effectiveness and outcome of the Eel Management Plans (EMPs) implemented within the 14 UK River Basin Districts (RBDs) for the period 2020-2023 (2020 to 2022 for England, Wales and Scotland; 2021 to 2023 for Northern Ireland). This is in accordance with the Council Regulation (EC) No 1100/2007 as amended by the Common Fisheries Policy (Amendment etc.) (EU Exit) Regulations 2019. The transboundary EMP shared with the Republic of Ireland (North West International) is not reported here because it is included in the report from the Republic to the European Commission.

Fisheries management is a devolved policy area in the UK. EMPs and assessment procedures vary according to different management structures in England, Wales, Scotland and Northern Ireland. Data are tabulated and described for all 14 EMPs together where appropriate, but in some circumstances where methods differ, data are described separately for each nation. Annexes A to C describe these assessment methods.

UK silver eel escapement biomass and mortality rates cannot be measured directly at the RBD scale. Therefore, we present results derived from extrapolations of yellow eel survey data (England and Wales), counts of silver eel in index rivers (Scotland and Northern Ireland) or silver eel mark-release-recapture studies (Neagh Bann – Northern Ireland). These approaches require a number of assumptions about the life history and production of eel, and there is natural variation (spatial and temporal) inherent within the input data used in the analyses. Hence, the outputs are described as ‘best available estimates’ and should be treated as such.

Mean annual silver eel escapement during the reporting period was greater than the long-term objective of 40% of pristine biomass (B_0) only in Scotland RBD (71.8 % of B_0). Despite Neagh Bann meeting the 40% target in the last reporting period (54.1 % of B_0), it has failed to do so in this reporting period (28.7 % of B_0). As a result, DAERA has actioned outputs from the EMP reviews and demands are now with fisheries to bring in restrictions.

There are no recreational landings throughout the UK, so no losses were reported. The losses from commercial fishing decreased in five RBDs, increased in one and did not change in other RBDs during this reporting period. The glass eel fishery in England has been historically restricted to rivers in the Southwest and the Northwest England, and following EU-exit, has only been authorised on the River Parrett (South-West RBD) and the River Severn (Severn RBD), as only these catchments were able to demonstrate non-detriment for eel population as per the UK Non-Detriment Finding (NDF). As a result, mortality rates in the period 2020–2022 were reduced compared to 2008.

Fishing effort for yellow and silver eel is reported more widely across RBDs in England and Wales, and the Neagh Bann in Northern Ireland. Effort fluctuated from year to year but comparing the period 2020–2022 with 2008, yellow and silver eel fishing effort was lower in

all eight RBDs where commercial fisheries operate. Effort within the Neagh Bann dropped substantially during the COVID-19 pandemic, from 87 to 36 vessels and has remained at the lower level of circa 52 vessels throughout the reporting period.

Impacts from other human factors (e.g. turbines and pumps, habitat loss, stocking, etc) decreased in four RBDs during the reporting period, with the exception of South West and Severn, where the proportion of non-fisheries mortalities (ΣH) increased, linked to reduced fisheries mortality (ΣF), but the overall lifetime mortality rate was reduced. The consequent estimated mortality rates are influenced by the underlying changes in eel abundance and therefore it can be difficult to attribute changes to specific events or management actions.

A range of management measures have been implemented to increase and/or protect silver eel production across the UK. In summary, these measures include restrictions on fisheries ranging from changes in quotas and closed seasons to outright bans, the stocking of glass eel, provision of additional eel habitat via removal of barriers to upstream migration or installation of fish passes, and entrainment reduction measures such as screening and “trap and transport”. In addition to the measures previously put in place, new measures have been implemented in 2020–2022. These include closure of eel fisheries in Wales (since 2021), commencement of a managed decline of the yellow and silver eel fisheries in England, increase in available eel habitat (53 new eel passes installed restoring access to over 900 ha of river habitat) and reduction of entrainment impacts by installing 33 new screens and 46 “fish-friendly", or Less Damaging Pumps (LDPs) at pumping stations. In addition, four combustion power stations were decommissioned during the reporting period, further decreasing entrainment impacts.

1. Introduction

This report outlines the monitoring, effectiveness and outcome of the UK Eel Management Plans (EMPs) during the most recent three-year reporting period, as required by the Council Regulation (EC) No 1100/2007, amended by the Common Fisheries Policy (Amendment etc.) (EU Exit) Regulations 2019.

Tables in this report summarise the best available estimates of silver eel escapement biomass (Table 1), mortality rates due to fisheries and other anthropogenic factors (Table 2), quantities of glass eel used for stocking and other purposes and their cost (Tables 3,4 and 5), as well as fishing effort for glass (Table 6), yellow and silver eels (Table 7 and 8) for the 14 River Basin Districts (RBDs) of the UK during the most recent three-year period of the EMPs (2020 to 2022 for England, Wales and Scotland; 2021 to 2023 for Northern Ireland).

1.1 The UK EMP framework

The 14 UK EMPs are set at the RBD (or Eel Management Unit (EMU[1])) level, as defined under the Water Framework Directive (WFD; EC, 2000), covering England, Wales, Scotland, and Northern Ireland (Figure 1). The RBDs in Northern Ireland deviate slightly from those defined for the WFD, owing to their transboundary nature.

Fisheries management is a devolved policy area in the UK and as such EMPs were drawn up by the relevant UK authorities within each of the nations. The implementation of EMPs is managed by different regional agencies: Environment Agency for England; Natural Resources Wales (NRW) for Wales; Marine Directorate for Scotland; and the Department for Agriculture, Environment and Rural Affairs (DAERA) for Northern Ireland. The North Western International EMP is a transboundary plan with the Republic of Ireland (RoI). Its assessment and management are the responsibility of the RoI, and progress with this plan is therefore reported in the Irish Progress Report to the European Commission. The Irish Technical Expert Group on Eel (TEGE) (replaces Standing Scientific Committee for Eel) was established by the respective Ministers from the Department of the Environment, Climate and Communications (RoI) and DAERA (NI). Consultation with the DAERA in Northern Ireland ensures the co-operation with Northern Ireland agencies to cover the specific needs of the trans-boundary North Western International River Basin District EMP. The TEGE comprises scientific advisers drawn from the Marine Institute (MI), Inland Fisheries Ireland (IFI), the Electricity Supply Board (ESB), The Loughs Agency, and the Agriculture, Food and Biosciences Institute for Northern Ireland (AFBINI). Although the scientists are drawn from

[1] In the context of eel management and Eel Management Plans, the terms River Basin District (RBD) and Eel Management Unit (EMU) are interchangeable.

these agencies, the advice from the TEGE is independent of the parent agencies and all data/analyses are jointly agreed before onward submission.

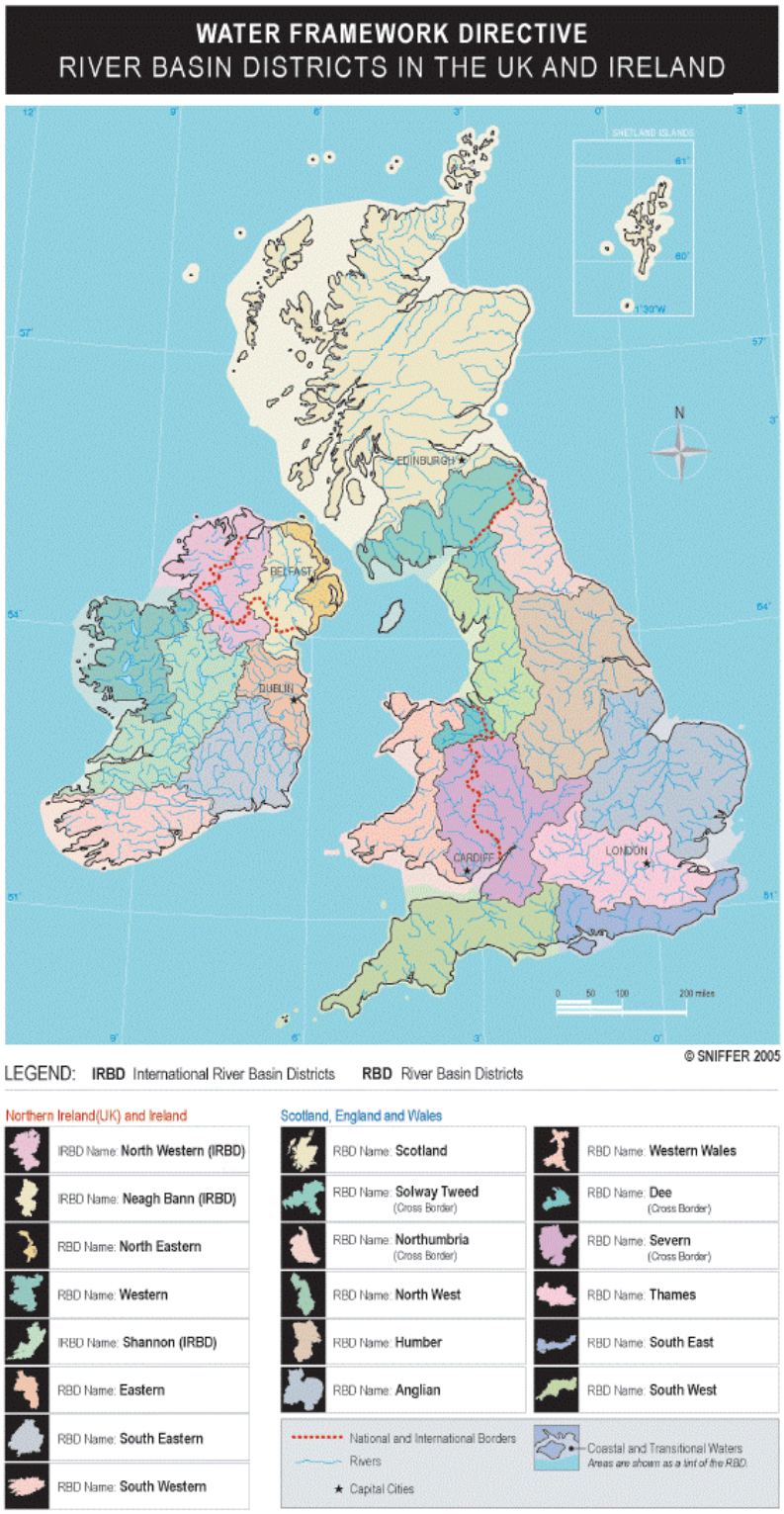

Figure 1. Map of the River Basin District (RBD) layout across the UK, which forms the basis of the associated Eel Management Plans (EMPs).

1.2 The stock indicators

UK is required to report the status of their eel stocks in each EMP in terms of best available estimates of the following stock indicators:

- $B_{current}$: the amount of silver eel biomass that currently escapes to the sea to spawn;
- B_0: the amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the stock;
- B_{best}: the amount of silver eel biomass that would have existed if no anthropogenic influences had impacted the current stock;
- B_x: the amount of silver eel biomass, or equivalent, that is lost to anthropogenic factor 'x' on an annual basis;
- $\sum F$: the fishing mortality rate, summed over the age-groups in the stock;
- $\sum H$: the anthropogenic mortality rate for the combined non-fishery factors impacting on eel.
- $\sum A$: the sum of anthropogenic mortalities, i.e. $\sum A = \sum F + \sum H$;
- $R_{(emu)}$: the catch weight of eel less than 12 cm in length used for stocking, consumption, or aquaculture, within or out of the EMU (or RBD).

1.3 Reporting format for 2024

Following the UK exit from the EU, the European Union (Withdrawal) Act 2018 brought Regulation EC 1100/2007 (EC, 2007) into the UK law, ensuring that its requirements are still met. In line with this, UK is no longer required to report to the EU Commission. However, Northern Ireland remains within the EU under the Windsor Framework agreement with regards to trade, which includes eel trade. The differing management structures within the UK mean that EMPs and assessment procedures vary between England, Wales, Scotland and Northern Ireland (see original EMPs, and information describing updates in assessment methods in Annexes A, B and C of this report). As a consequence, there are some key differences in the manner in which assessments are reported here for the four nations of the UK.

Although assessments are updated annually for the Scotland and Northern Ireland EMPs, the input data for the EMPs of England and Wales are derived from a six-year rolling programme of electric fishing surveys and therefore can only be fully updated every six years. For reporting to the three-year cycle of the ICES data call, however, assessments of B_{best} for England and Wales are based on data from surveys that have been conducted

within each reporting period. This means results from a different subset of survey sites are used each year. Differences reported in $B_{current}$ within a reporting period are due to differences between years in the amount of commercial eel catch, or reductions in other anthropogenic mortalities due to improved screening, pump design or eel passage implemented during the reporting period. It is problematic to draw conclusions from apparent trends or otherwise within three-year reporting periods. It may be more informative to compare mean values between three-year reporting periods, but the survey cycle must be taken into consideration in interpreting any apparent trends.

The timeframe of reporting eel population survey data in England and Wales is such that data for 2023 were not available in time to be included in the assessments to meet the original deadline of 30 June 2024 for this report. Therefore, and to retain the three-year reporting schedule followed in previous reports, the biomass and mortality rate estimates for England and Wales are reported for 2020, 2021 and 2022.

The assessment of emigrating biomass and mortality rates for England and Wales since implementation of the EMPs has been estimated for the following time periods:

- 2008–2010
    - Used to assess emigrating biomass and mortality indicators for 2009 and 2010, the first reporting period since the implementation of the EMPs;
- 2011–2013
    - Used to assess emigrating biomass and mortality indicators for 2011, 2012 and 2013.
- 2014–2016
    - Used to assess emigrating biomass and mortality indicators for 2014, 2015 and 2016.
- 2017–2019
    - Used to assess emigrating biomass and mortality indicators for 2017, 2018 and 2019.
- 2020–2022
    - Used to assess emigrating biomass and mortality indicators for 2020, 2021 and 2022.

In contrast, new data supporting the biomass and mortality rate estimates for Scotland and Northern Ireland have been available annually. These are reported here for 2021, 2022 and 2023 for Northern Ireland. However, following the closure of a key monitoring site in Scotland

no assessment is available for 2023 and so the years 2020-2022 are reported here, in line with England and Wales.

1.4 Changes in the assessment method since the 2021 report

1.4.1 The impact of COVID-19 on the stock assessment

The impacts of COVID-19 fell largely into two groups:

- Interruption/cessation of scientific monitoring and national stocking programmes (across different life stages);
- Closure/delay in commercial fishing (glass and yellow eel stages) due to movement restrictions and/or loss of markets as a consequence of social lockdowns.

England and Wales

Across the southern half of England, the collection of glass eel recruitment data from all of the sampling points (including index catchments used in the recruitment analyses) by the Environment Agency were reduced due to movement restrictions and staff availability.

The majority of the electric fishing was not undertaken in 2020 for yellow eel in England. Similarly, in Wales, NRW had been unable to carry out quantitative yellow eel electric fishing surveys on the rivers Teifi (10 sites) and Dee (10 sites) in 2020. Overall COVID-19 impacts mean that for England and Wales, there are very limited yellow eel monitoring data for 2020, which had implications for compliance assessment. For more details, please check section Assessment Methodologies.

For England, the silver eel monitoring at counters was largely undertaken as planned.

Scotland

In Scotland, minor disruptions were reported for recruitment monitoring, with no data collected for one of the three recruitment series. In addition, there were fewer sites monitored for yellow eel abundance, but there were no disruptions to silver eel escapement monitoring. Overall COVID-19 had minimal impact on eel-specific data that are reported from Scotland.

Northern Ireland

In Northern Ireland, COVID-19 impacts were minimal on data series from the North Eastern RBD. However, the effects on the Neagh Bann RBD were larger. The collection of recruitment data had remained unaffected, but the commercial fishing season on Lough

Neagh did not begin in May 2020 as usual, opening on 1st July with a much-reduced fishing fleet than in previous years (36 boats compared to 87), and remaining at the lower level of circa 52 vessels throughout the reporting period. This lower number was influenced by government Furlough scheme payments to self-employed workers (such as fishermen) and the loss in continental markets for yellow eel as a direct result of lockdowns/loss of tourism in Netherlands and Germany.

Silver eel escapement assessment in the North Eastern RBD was impacted by COVID-19 restrictions. In the Neagh Bann RBD, of the four lunar darks normally assessed for silver escapement (September – December inclusive), only the September and November silver eel runs had an associated floy tag escapement estimate in 2020 due to localised COVID-19 positive tests, and the suspension of working on site.

1.4.2 Assessment Methodologies

England and Wales

The assessment method applied across England and Wales is described in Annex A. There was no major change in this assessment method for this latest report. However, the impacts of COVID-19 resulted in minimal yellow eel data for 2020 and reduced data for 2021 available for the Scenario-based Model of Eel Production II (SMEP II). Therefore, we must treat the outputs with caution. In addition, in 2022 a proportion of the catch data was unavailable, which means that the impact of the fisheries during this year may be an underestimate, adding further uncertainty to the biomass and mortality outputs for the reporting period.

Yellow eel data from just 35 sites across six RBDs were available from electric fishing surveys conducted in 2020, with no surveys conducted in 2020 in Humber, North West or Solway Tweed RBDs. Given the reduced survey data, it was agreed that only those rivers with five or more ‘eel present’ sites would be used in the SMEP II modelling. The number of eel index rivers used to estimate B_{best} and $B_{current}$ was reduced from 41 (in 2021) to 24 (in 2024). Survey data from rivers across the whole of the Humber RBD are normally modelled using SMEP II, with the Humber RBD split into 20 reaches. For the 2020-2022 reporting period 20% of reaches were not surveyed at all and 40% were surveyed but no eels were present. Therefore only 40% of reaches provided data for SMEP II. The resultant output was a very low modelled escapement estimate of 0.1 $kg \cdot ha^{-1}$ of silver eel due to the lack of input data. The very low escapement estimate produced a negative $B_{current}$ estimate after taking into account the estimates of anthropogenic impacts. It was hence not possible to produce a mortality estimate ($\sum A$). Consequently, entries for Humber RBD are indicated by “ND”, where relevant in the report tables below.

The effect of new eel passes was estimated as the additional wetted area in hectares (ha) made available to eel by each pass installed, multiplied by the modelled silver eel escapement (from SMEP II) from the relevant RBD for the year of installation and thereafter.

The assessment protocols for estimating impacts of water abstraction, hydropower, pumping stations and cooling water intakes did not change but numbers of installations were updated to reflect installation of new screens, less damaging pumps and decommissioning of facilities.

Emerging data on the Severn RBD and implications for the assessment

In the 2021 we described a study by Aprahamian and Wood (2021), which implied that the relative impact of the glass eel fishery in the Severn may be lower than the estimate used in the assessment (∑F), with other non-fisheries impacts (∑H) likely having a relatively greater contribution to the overall mortality (∑A). Further consideration of this work by UK experts has concluded that this single study is insufficient to result in a change to the assessment methodology for 2024. This is because:

- A comprehensive review of the current assessment methodology is required to enable greater confidence in the relative contribution of ∑F and ∑H to the overall mortality ∑A across all RBDs, but this will require a dedicated project and resources.
- The 2021 study on the River Severn would benefit from being repeated to provide some measure of temporal variability.
- Comparable studies would be required in other RBDs before the same assumptions and methodology could be applied to those other RBDs which support a glass eel fishery.

Scotland

The assessment method applied to the Scotland RBD is described in Annex B. There was no change in the assessment method for this latest report, but following a closure of a key monitoring site, an assessment was not available for 2023. Therefore, biomass and mortality estimates were provided for 2020, 2021 and 2022, as for England and Wales.

Northern Ireland

The assessment methods applied to the Northern Ireland RBDs are described in Annex C.

Since 2018, the calculation for estimated escapement has been changed and further improved by the development of a model combining daily river flow metrics with daily silver eel catch against which daily tag recaptures are assessed. This method has been used to hindcast and revise the calculations for escapement from 2009. Following COVID-19 related impacts and accessibility to relevant detailed data the calculations for the current period have reverted to the original format pre-2018. However, by way of data quality assurance, comparative analyses between the two assessment calculations, found that silver eel

escapement estimates varied little between the two methods, only 3-5%, thus can be considered almost equivalent.

The method for the North Eastern RBD was updated in 2017 to provide new estimates of current ($B_{current}$) and potential (B_{best}) silver eel escapement, by the establishment of a glass eel index site (*in situ*; five years of data) and the direct assessment of silver eel migration in 2017 onwards by netting. However, the direct escapement assessments in this RBD were heavily impacted by flooding (2019 & 2023) and COVID-19 restrictions (2020).

2. Best available estimates of stock indicators and associated information

2.1. Summary results

The best available estimates for the 14 RBDs in the UK during this reporting period are provided for silver eel escapement biomass, mortality rates due to fisheries and other anthropogenic factors, and quantities of glass eel used for restocking (Tables 1, 2 and 3).

Escapement biomass and mortality rates for the UK cannot be measured directly at the RBD scale. Therefore, we present results derived from extrapolations of yellow eel survey data (England and Wales), counts of silver eel in index rivers (Scotland and Northern Ireland) or silver eel mark-release-recapture studies (Neagh Bann – Northern Ireland). These approaches require a number of assumptions about the life history and production of eel, and there is natural variation (spatial and temporal) inherent within the input data used in the analyses. Hence, the outputs are described as 'best available estimates' and should be treated as such.

2.2. Biomass

Mean annual silver eel escapement ($B_{current}$) during the reporting period was greater than the long-term objective of 40% B_0 in one RBD: Scotland, achieving 71.8% of B_0. Despite Neagh Bann meeting the 40% target in the last reporting period (54.1 % of B0), it has failed to do so in this reporting period (28.7 % of B0).

Trends in biomass

For England and Wales, trends in $B_{current}$ are compared with the period estimates, which are expressed as a percentage of B_0, for 2009–2010, 2011–2013, 2014–2016, 2017–2019 and 2020-2022 (Figure 2). Since last reported, escapement decreased in three RBDs, increased

in six RBDs, and remained unchanged in one. No estimate is reported for Humber RBD (as described in Section 1.4). Most changes since the 2021 report have been relatively minor, but substantial reductions have been recorded in Northumbria and Thames RBDs, dropping from 12.6% to 3.8% and 22.4% to 1.8%, respectively. In South East RBD there was a considerable increase from 19.7% to 30.4%. The reductions in modelled silver eel escapement from Northumbria and Thames RBDs could be attributed to lower yellow eel densities recorded in surveys. Densities in the Wear in Northumbria were over 50% lower than in the previous reporting period. For the Thames RBD there were no data from the River Lee, while eel density in the Thames and Medway sites fell by over 70% and 90%, respectively, compared to the previous reporting period. The increase in modelled silver eel escapement from South East RBD is likely because only data from the more productive chalk stream rivers were used in the SMEP II modelling this time, which are not representative of the whole RBD. In addition, there was an increase of over 50% in eel densities recorded in the River Test.

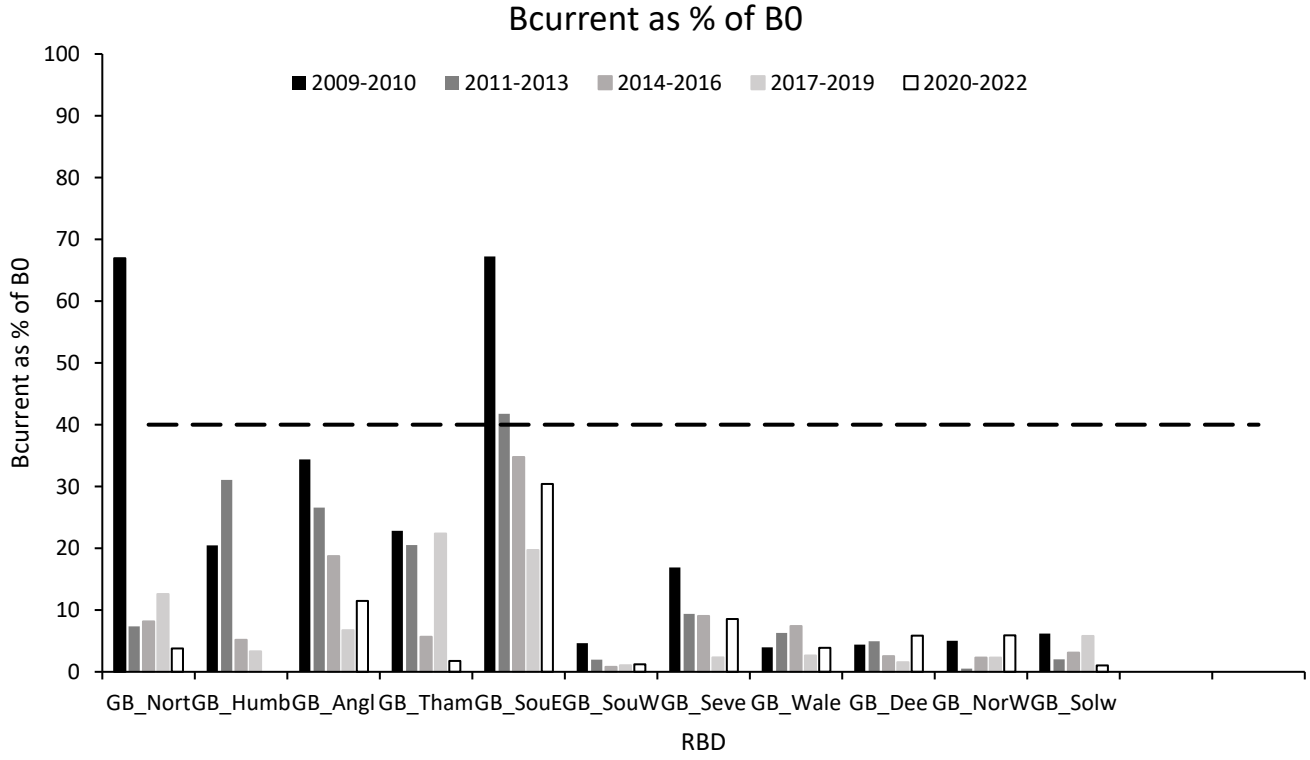

Figure 2. Compliance with the long-term objective of 40% B_0. From left to right, RBD codes correspond to Northumbria, Humber, Anglian, Thames, South East, South West, Severn, Western Wales, Dee, North West and Solway Tweed RBDs. The dashed line represents the long-term objective of 40% B_0.

For the Scotland RBD, as the assessment is conducted separately for each year, a fuller trend analysis is appropriate. $B_{current}$ expressed as a percentage of B_0 declined from 2009 to 2011 but subsequently increased, returning above the long-term objective each year till 2022

(Figure 3). There is however a suggestion of an underlying decreasing trend from 2014 onwards. The years 2014 and 2021 were exceptionally productive for silver eel in Scotland with respective $B_{current}$ estimates at 135% and 111% of B_0. These values are above the B_0 reference value for Scotland RBD, which is based on a mean of several years of historic data.

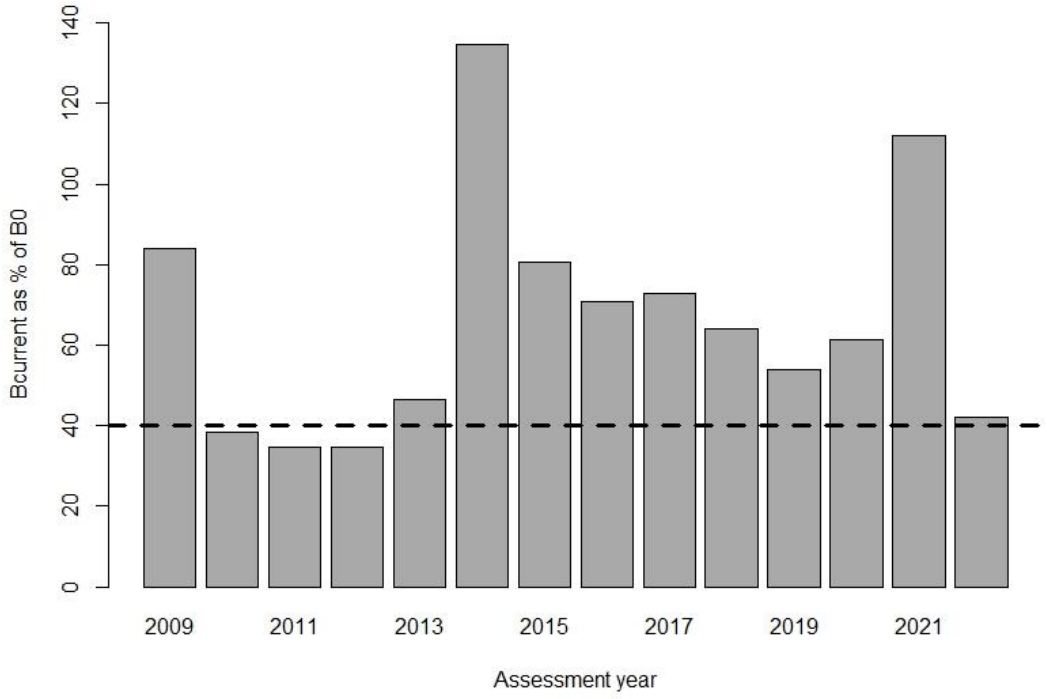

Figure 3. Change in $B_{current}$ as a percentage of B_0 in the Scotland RBD between 2009 and 2022. The dashed line represents the long-term objective of 40% B_0.

For the Neagh Bann RBD in Northern Ireland, $B_{current}$ expressed as a percentage of B_0 increased from 2016, but has been decreasing again since 2020, and has now fallen below the 40% B_0 target (Figure 4). As a result, DAERA has actioned outputs from the EMP reviews and demands are now with fisheries to bring in restrictions. Estimates of $B_{current}$ and B_{best} for the North Eastern RBD are only available for limited years, after a new direct capture method was developed and implemented, but this is often hampered by flood events. Therefore, no analysis of trend is yet possible. However, as this is a natural system with minimal anthropogenic impacts for eel, trends are expected to be reflective of recruitment history.

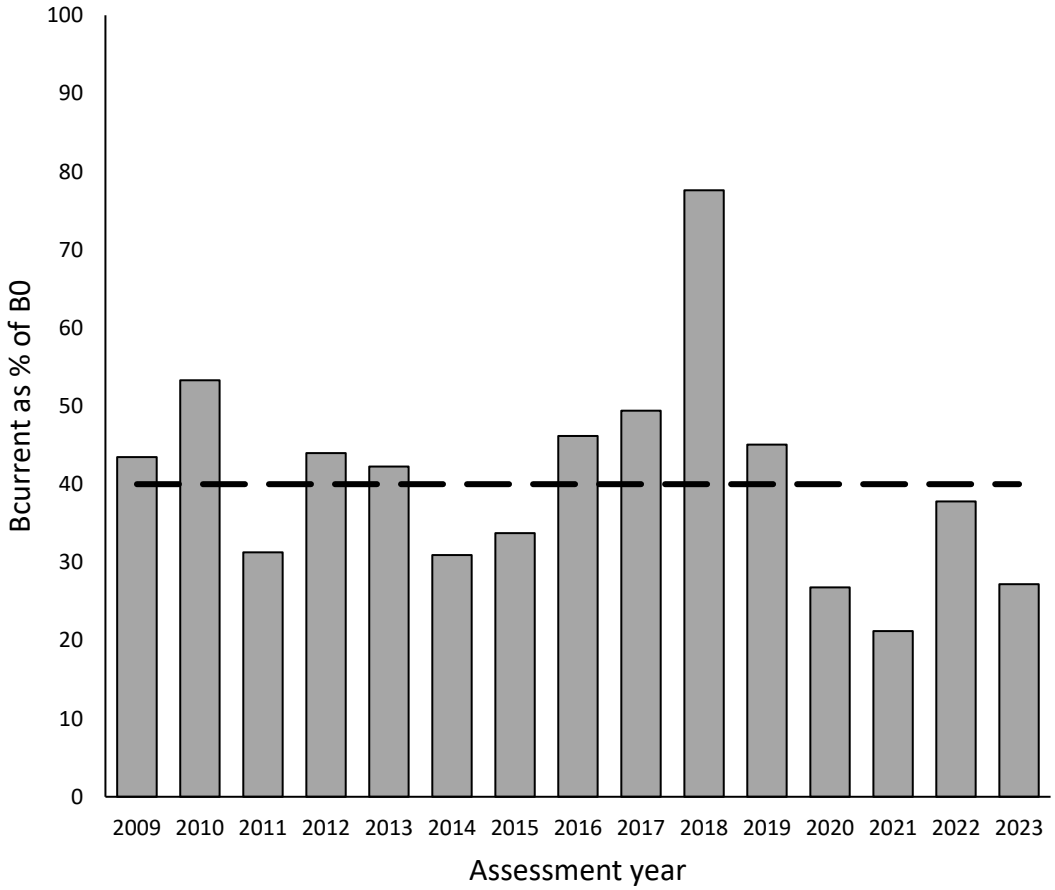

Figure 4. Change in $B_{current}$ as a percentage of B_0 in the Northern Ireland Neagh Bann RBD between 2009 and 2023. The dashed line represents the long-term objective of 40% B_0.

Table 1. Best estimates of silver eel biomass (kg) across England, Wales and Scotland during 2020–2022, and across Northern Ireland during 2020–2023. Note these estimates are based on period means for some data inputs. Mean compliance is based on the data from the most recent three years. Key for terms is provided below.

RBD	B_0	$B_{current}$				B_{best}				Mean compliance ($B_{current}/B_0$ x 100; %)
	Pre-1980	2020	2021	2022	2023	2020	2021	2022	2023	
Northumbria	60876	2274	2323	2292	ND	6412	6412	6412	ND	3.8
Humber	137859	ND	ND	ND	ND	ND	ND	ND	ND	ND
Anglian	341084	44452	36246	36638	ND	80523	80523	80523	ND	11.5
Thames	251699	4403	4482	4491	ND	26776	26776	26776	ND	1.8
South East	121340	36759	36875	37056	ND	53322	53322	53322	ND	30.4
South West	1327684	15423	17179	15415	ND	153098	45401	75173	ND	1.2
Severn	899687	77368	76208	77410	ND	240971	139369	187669	ND	8.6
Western Wales	429944	16712	16712	16712	ND	27082	19679	19679	ND	3.9
Dee	636166	37517	37187	37247	ND	57890	57890	57890	ND	5.9
North West	865449	49713	51526	52093	ND	80229	77368	77368	ND	5.9
Solway Tweed	1473755	15177	15177	15177	ND	19820	19820	19820	ND	1.0
Scotland	267717	164395	299491	112982	ND	201519	368978	140023	ND	71.8
North Eastern	4000	ND*	806	1104	232	ND*	806	1104	232	17.9
Neagh Bann	500000	134000	106000	189000	136000	292600	324000	330000	216300	28.7

Where:

- 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 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;
- ND "No Data", where there are insufficient data to estimate a derived parameter or where data were collected but not available in time to be used in this report;
- ND* "No Data" due to COVID-19 impacts.

2.3. Anthropogenic impacts

Estimates of annual anthropogenic mortality rates for eel attributed to fishing (ΣF), human non-fishing impacts (ΣH) and both combined (ΣA) for each RBD are provided in Table 2. Where impacts have been on glass or yellow eel, these rates are estimated by converting the eel stage affected to silver eel equivalents. All fishing and non-fishing related impacts are then added up to provide total fishing and total non-fishing anthropogenic mortality rates, respectively, expressed as proportions of the overall lifetime mortality rates (see Annexes). In the following discussion, any variation in mortality rate of <0.05 during the reporting period is described as 'no change'.

Although results for England and Wales are derived in part from mean potential eel escapement estimates (B_{best}) for the period 2020–2022, the loss rates from anthropogenic factors are year-specific and therefore it is reasonable to examine trends across the reporting period. Data for Scotland and Northern Ireland are annual and therefore support examination of trends.

Fishing (F)

There are no recreational landings of eel across the UK. Commercial fishing occurred in nine of the eleven RBDs across England and Wales over the reporting period. However, since 2021, adult and glass eel fisheries have been closed in Wales, resulting in zero fishing in Western Wales RBD and the Welsh sections of the Dee RBD, from 2021. Commercial fishing occurs in one RBD in Northern Ireland, but not in Scotland.

Across the reporting period, the relative impact (mortality rate: ΣF) of commercial fishing increased in Anglian RBD, decreased in five RBDs (North West, Severn, South West, Western Wales, and Neagh Bann) and did not change in the others. The increasing rate in Anglian RBD was a result of a low catch in 2020, which may be due to impacts of COVID-19. A decreased rate in Western Wales was due to the closure of the fishery since 2021. Glass eel fishing was not authorised in the North West in 2021 or 2022, resulting in the

decrease in total catch and in ΣF. Since EU-exit and CITES trade restrictions (2021), glass eel fishery has only been authorised on the River Parrett (South-West RBD) and the River Severn (Severn RBD), as only these catchments were able to demonstrate non-detriment for eel population as per the UK Non-Detriment Finding (NDF; Fleming et al., 2023). Consequently, mortality rates in the period 2020–2022 were reduced compared to 2008. After accounting for dealers' notes (see Section A3.1 for more information), glass eel catches in the Severn RBD were 1776 kg in 2020, 66 kg in 2021 and 879 kg in 2022. In the South West RBD, glass eel catches were 1813 kg in 2020, 0 kg in 2021 and 501 kg in 2022. No ΣF rate is reported for Humber RBD (as described in Section 1.4)

Non-fishing (H)

Across the reporting period, the impact (mortality rate: ΣH) of anthropogenic non-fishing factors across most of the RBDs showed very small variations, other than in South West and Severn RBD, where ΣH increased, most likely linked to a reducing ΣF. As mortality rates due to anthropogenic non-fishing factors are estimated as eel losses in proportion to potential eel production, the results are influenced by the underlying changes in eel abundance: a smaller loss can still produce a higher mortality rate if the underlying potential has declined. Therefore, it can be difficult to attribute changes to specific events or management actions.

Overall (A)

The overall impact of anthropogenic mortality factors (ΣA) decreased in four RBDs (Severn, South West, Western Wales and Neagh Bann), increased in one RBD (Anglian), and showed little change in the others. These changes were mainly due to changes in fishing mortality rates. No ΣA rate is reported for Humber RBD (as described in Section 1.4)

Table 2. Best estimates of anthropogenic mortality rates across UK RBDs, during 2020 to 2022 for England, Wales and Scotland, and 2020 to 2023 for Northern Ireland. Note that minor differences in A versus F+H are due to rounding to two decimal places. Note that stocking was not included in the calculation of ΣH or ΣA. Key for terms is provided below.

RBD	ΣF				ΣH				ΣA			
	2020	2021	2022	2023	2020	2021	2022	2023	2020	2021	2022	2023
Northumbria	0.00	0.00	0.00	ND	1.04	1.02	1.03	ND	1.04	1.02	1.03	ND
Humber	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Anglian	0.06	0.22	0.21	ND	0.53	0.58	0.57	ND	0.59	0.80	0.79	ND
Thames	0.00	0.00	0.00	ND	1.80	1.79	1.78	ND	1.81	1.79	1.79	ND
South East	0.01	0.00	0.00	ND	0.37	0.36	0.36	ND	0.37	0.37	0.36	ND
South West	2.03	0.44	1.17	ND	0.26	0.54	0.41	ND	2.30	0.97	1.58	ND
Severn	0.74	0.04	0.42	ND	0.41	0.57	0.48	ND	1.15	0.60	0.90	ND
Western Wales	0.34	0.00	0.00	ND	0.14	0.16	0.16	ND	0.48	0.16	0.16	ND
Dee	0.00	0.01	0.01	ND	0.43	0.44	0.44	ND	0.43	0.44	0.44	ND
North West	0.06	0.01	0.00	ND	0.42	0.40	0.39	ND	0.48	0.41	0.40	ND
Solway Tweed	0.00	0.00	0.00	ND	0.27	0.27	0.27	ND	0.27	0.27	0.27	ND
Scotland	0.00	0.00	0.00	0.00	0.20	0.21	0.21	ND	0.20	0.21	0.21	ND
North Eastern	ND*	0.00	0.00	0.00	ND*	0.00	0.00	0.00	ND*	0.00	0.00	0.00
Neagh Bann	0.78	1.12	0.56	0.46	0.00	0.00	0.00	0.00	0.78	1.12	0.56	0.46

Where:

- ΣF The fishing mortality rate, summed over the age-groups in the stock;
- ΣH The anthropogenic mortality rate outside the fishery, summed over the age-groups in the stock;
- ΣA The sum of anthropogenic mortalities, i.e. ΣA = ΣF + ΣH;
- ND "No Data", where there are insufficient data to estimate a derived parameter or where data were collected but not available in time to be used in this report.
- ND* "No Data" due to COVID-19 impacts.

Stocking

Stocking is considered a practice of taking fish from one or multiple places and moving either up the same catchment or across catchments, which usually involves a temporary holding facility and health checks. It is a relatively minor activity in England and Wales, with only 20 kg stocked in 2020 and 2022 in the Severn RBD (Table 3). Based on the assumptions applied in this report, and assuming access to suitable habitat, these stockings can be calculated as 2376 kg silver eel equivalents in total. Note that potential anthropogenic impacts on these eels have not been taken account in these calculations.

In addition to stocking, there is also a practice of assisted migration in England and Wales, which involves catching fish below barriers and moving over on the same day, with no mixing of source stocks or translocation to other catchments. In the Severn RBD, 379 kg of glass eel caught in 2021 and 200 kg in 2022 were returned to various locations upstream. A further 85 kg of grown-on juveniles caught from Severn RBD in 2021 were returned upstream in 2022. 125 kg glass eel caught from South West RBD in 2021 and 34 kg caught in 2022 were returned as well in assisted migration. Eels that were caught and released under assisted migration were not considered in the assessment.

There has been no eel stocking in Scotland for many years.

Eel stocking is important to the Neagh Bann RBD in Northern Ireland, with 4987 kg of glass eel stocked in total in the Lough Neagh during the reporting period (2020–2023; Table 3). French glass eel comprised 2400 kg (48.1%) of the total biomass stocked, the remainder being sourced from the UK fishery.

Table 3. The amount of glass eel (kg) stocked into UK RBDs, during 2020–2023. Data for 2023 were not available (ND) at the time of the writing for some RBDs. There are no stocking practices (NP) in North Eastern and Scotland RBDs.

RBD	2020	2021	2022	2023
Northumbria	0	0	0	ND
Humber	0	0	0	ND
Anglian	0	0	0	ND
Thames	0	0	0	ND
South East	0	0	0	ND
South West	0	0	0	ND
Severn	20	0	20	ND
Western Wales	0	0	0	ND
Dee	0	0	0	ND
North West	0	0	0	ND
Solway Tweed	0	0	0	ND
Scotland	NP	NP	NP	NP
North Eastern	NP	NP	NP	NP
Neagh/Bann	1714	1033	1442	798

The amount of glass eel caught per annum, as declared to the Environment Agency and Natural Resources Wales, is presented in Table 4 along with the proportions of this catch used for stocking, aquaculture or direct consumption ($R_{(emu)}$), based on declarations at first sale.

Table 4. Weight of glass eel caught in the UK and the percentages sold for stocking, aquaculture or direct consumption ($R_{(emu)}$), according to dealer's reports. Note these percentages may not add up to 100% because of mortality and weight loss after capture. Some data were not available at the time of reporting (ND = no data).

		Percentage used for		
Year	Catch	Stocking	Aquaculture	Direct Consumption
2009	422	100.0	0.0	0.0
2010*	1890	55.4	3.5	0.0
2011+	3641	34.8	63.9	0.0
2012	3819	88.8	11.2	0.0
2013	8659	50.4	49.5	0.0
2014	11600	62.6	30.9	6.8
2015	2800	72.7	27.2	3.6
2016	4279	54.0	45.7	0.3
2017	3530	56.3	43.7	0.0
2018	4660	80.5	19.5	0.0
2019	6950	72.2	27.7	0.0
2020	3566	82.9	17.1	0.0
2021	608	93.2	0.2	6.7
2022	1270	100	0.0	0.0
2023	813	98.0	0.6	1.4

*40.9% of exports were not declared, so could have been either restocking or aquaculture.
+1.22% of exports were not declared, so could have been either restocking or aquaculture.

The evolution of the market price of glass eel, based on the price paid by the Lough Neagh Fishermen's Cooperative Society (LNFCS) for eels for stocking Lough Neagh is presented in Table 5.

Table 5. Cost of purchasing glass eel (€ /kg). ND = no data.

Year	Cost (€) /kg
2009	525
2010	497
2011	353
2012	475
2013	400
2014	225
2015	284
2016	ND
2017	275
2018	250
2019	250
2020	190
2021	210
2022	230
2023	230

2.4. Fishing effort

The time series of fishing effort for commercial eel fisheries in the UK (noting no commercial fishing in Scotland) are presented in Tables 6, 7 and 8 per nation and eel life stage. Given that fishing effort for yellow and silver eel has only recently been disaggregated for England and Wales, it is reported as combined fishing effort for consistency with previous reports (Table 7).

Fishing effort for glass eel is reported for four RBDs during the 2020–2022 period. However, since 2021, the only glass eel fisheries operating were in the Severn and South West RBDs, as glass eel fishing was not authorised elsewhere. Fishing effort has fluctuated in the Severn and South West RBDs over the time series but rates in the 2020–2022 period were much lower than in previous years. Since the UK left the EU, CITES trade restrictions meant glass eel can no longer be exported to EU countries. The restricted market has resulted in reduced fishing effort (Table 6). An internal UK market was re-established post EU-exit in 2021 given the unique circumstances surrounding NI, with a trial road transport shipment of 62 kg to Neagh Bann (it could no longer be flown as it required inspection at NI harbour Inspection

point). The following years of 2022 and 2023 saw stocking transfers from GB to NI of 1123 kg and 297 kg before this import trade was banned by EU CITES.

Table 6. Time series of annual fishing effort for glass eel in RBDs of England and Wales, during 2008–2022. Effort is expressed as dip net nights. There have been no fisheries authorised (NP = not pertinent) in the South East RBD since 2009. Only fisheries in Severn and South West RBDs were authorised from 2021. Data for 2023 were not available at the time of reporting.

Year/RBD	North West	Dee	Western Wales	Severn	South West	South East
2008	194	10	18	4060	2064	0
2009	142	14	16	3020	1344	16
2010	82	14	22	2271	1178	NP
2011	95	23	14	3903	3141	NP
2012	108	32	9	5390	4026	NP
2013	101	12	17	4660	4301	NP
2014	153	0	7	8360	9371	NP
2015	266	39	0	10297	8032	NP
2016	121	8	21	4623	2877	NP
2017	118	17	36	4324	2755	NP
2018	118	27	50	3935	2491	NP
2019	141	27	100	4344	2826	NP
2020	48	0	45	1825	1368	NP
2021	0	NP	NP	514	27	NP
2022	NP	NP	NP	693	314	NP

Fishing effort for yellow and silver eel is reported more widely across RBDs in England and Wales, and the Neagh Bann in Northern Ireland. Effort fluctuates from year to year but comparing rates in the 2020–2022 period with 2008, effort was considerably lower in all six English RBDs (Humber, Anglian, Thames, South East, South West and North West), as well as in Neagh Bann for yellow eel. In 2020 and 2023, fishing effort for yellow eel substantially dropped in the Neagh Bann RBD due to COVID-19 and Cyanobacteria outbreak, respectively. Effort for yellow and silver eel combined was substantially lower in the Dee after the closure of the eel fisheries in the Welsh sections from 2021 (Table 7). No fishing took place in Western Wales RBD during the reporting period, due to COVID then fishery closure (Table 7). Severn and Northumbria RBDs were not included in this comparison given yellow and silver eel fisheries have not been authorised in these RBDs since 2013 and 2010, respectively (Table 8). Fishing effort for silver eel in the Neagh Bann RBD remains the same since 2012 (Table 8).

Table 7. Time series of annual fishing effort for yellow and silver eel (combined) in England and Wales RBDs (trap nights), and yellow eel in Neagh Bann (boat days per season), during 2008 to 2023. ND = data not available to report, NP = not pertinent (no fishery authorised in that year). NOTE: figures for England for 2022 are minimum estimates due to missing catch returns.

Year/RBD	Northumbria	Humber	Anglian	Thames	South East	South West	Severn	Western Wales	Dee	North West	Neagh Bann
Effort					Trap nights						Boat days per season
2008	186	17898	54163	24811	13296	28999	185	186	5102	5909	9650
2009	168	16157	41561	13610	30277	11494	5330	2458	210	548	10860
2010	66	6991	52358	13940	7898	17728	366	331	144	533	10490
2011	NP	19346	99418	18305	6783	17483	1980	557	5184	14604	10440
2012	NP	17380	83572	10267	19315	27885	0	5703	4423	27574	9880
2013	NP	24545	75430	21796	13381	48437	10	302	884	9305	9810
2014	NP	20362	101315	11859	2680	21825	NP	0	5670	251	9590
2015	NP	11510	135164	13656	7987	47654	NP	54	804	397	8850
2016	NP	808	93343	13602	25010	41575	NP	13729	892	1071	8785
2017	NP	5642	89417	11466	6350	45648	NP	2	3426	8727	8698
2018	NP	7883	80440	8561	6847	41019	NP	10	2302	6075	8641
2019	NP	2075	77792	5656	3904	38387	NP	0	3888	9163	8657
2020	NP	93	1592	16	262	678	NP	0	0	252	589
2021	NP	737	2202	3	265	857	NP	NP	74	195	5310
2022	NP	35	1175	27	16	744	NP	NP	60	26	4195
2023	NP	ND	ND	ND	ND	ND	NP	NP	ND	ND	2005

Table 8. Time series of annual fishing effort for silver eel in the Neagh Bann RBD, during 2008–2023. Effort expressed as numbers of fishing weirs.

Year/RBD	Neagh Bann
2008	3
2009	3
2010	3
2011	3
2012	2
2013	2
2014	2
2015	2
2016	2
2017	2
2018	2
2019	2
2020	2
2021	2
2022	2
2023	2

3. Implementation of management measures

3.1. Description of the measures implemented since the adoption of the Eel Management Plans

In England and Wales since the adoption of eel management plans the following actions have delivered:

- 100% mandatory catch and release for eel by angling (introduced 2009);
- Close seasons for net and trap fishing for eel (introduced 2010);
- Limits on the geographical extent of the eel fishery (introduced 2010);
- Restrictions on eel fishing methods and gear (introduced 2010);
- New legislation to require the installation of eel passes and eel screens at structures impacting safe eel passage (introduced 2010);
- Two Fish Recovery and Return/bywash systems at water intakes;
- A reduction in the 2018 fishing season in territorial waters by 40 days (compared to pre-EMP) as a result of EC Regulation 2018/120 (EC, 2018);
- A reduction in the 2019 fishing season in all fisheries by 10 days compared to pre-EMP (fishing season increased by 30 days compared to 2018) as a result of EC Regulation 2019/124 (EC, 2019);
- Closure of the eel fisheries (all life stages) in Wales (introduced 2021);
- Commencement of a managed decline of the yellow and silver eel fisheries in England, through restrictions in eligibility for an eel fishing authorisation, which will lead to a gradual decline in fishing effort (introduced 2022);
- 53 new eel passes in 2020-2022 restoring access to over 900 ha of river habitat (totalling 938 passes restoring access to over 10 200 ha since 2009);
- 33 new screens at water intakes during 2020-2022 (totalling 85 eel screens since 2009, estimated to be protecting over 1600 kg of silver eel equivalents per year);
- 46 “fish-friendly", or Less Damaging Pumps (LDPs) installed at pumping stations in England during 2020-2022 (totalling 155 LDPs since 2011, estimated to be improving access to over 652 ha of upstream habitat);
- Four combustion power stations decommissioned during 2020-2022 (estimated to be saving 2790 kg of silver eel equivalents per year) and one converted to no longer draw cooling water, further decreasing entrainment impacts.

In Scotland, the principal management measure of the EMP was to prohibit fishing for eel, by any method, without a licence, via legislation introduced in 2009. Since then, no licences have been issued to fish for eel in Scotland (with the exception for some small-scale scientific sampling).

In Northern Ireland, the actions described in the 2015, 2018 and 2021 reports continue, and are reported below. No additional management measures have been implemented for the most recent reporting period.

National measures:

- Removal of fyke net as a legal fishing engine in 2010;
- Raising of Minimum Landing Size (MLS) for yellow eel from 300 to 400 mm in 2010;
- Ban on the taking of eel by recreational fishing in 2010, all NI RBDs;
- Establishment of yellow and silver eel commercial traceability system in 2009.

Neagh Bann RBD:

- Closure of one silver eel fishing weir in the River Bann since 2012;
- LNFCS direct funding of PhD project investigating male eels, their silver phase and run timings, differential capture rates and parasite burdens to provide biological information used in the stock assessment method (2014-2021; Dolan C (2021) The biology and ecology of the male European eel (*Anguilla anguilla* L.): considerations for management. PhD Final Thesis, Queens University Belfast);
- Initiatives to reduce capture of undersized eels (<400 mm total length) in long line harvest, by (i) increase in commercial long line hook size (from size 4 to 3) since 2016, and (ii) MRes research project into the development of an alternative eel fishing bait derived from marine discards in 2017 (Wootton J (2017) An assessment of the efficacy of the longline fishing method in Lough Neagh's yellow eel fishery and preference testing of a new bait derived from marine discards. MRes Final Thesis, Chester University);
- LNFCS commissioned an investigation into the prevalence of eel viruses in the Neagh Bann RBD in 2016 (McConville J, Fringuelli E, Evans D, Savage P (2018) First examination of the Lough Neagh European eel (*Anguilla anguilla*) population for eel virus European, eel virus European X and Anguillid Herpesvirus-1 infection by employing novel molecular techniques. Journal of Fish Disease, 1–9);
- Refurbishment of six eel passes within the Neagh Bann RBD since 2016;
- Improvement and modernisation of LNFCS fisheries enforcement vessels since 2017.

North Eastern RBD:

- Creation of glass eel monitoring site since 2012: now established as a new annual index site and reported to ICES since 2017;
- Glass eel stocking of this RBD in 2014 (funded by LNFCS);
- Assessment of recruitment, yellow eel population and migrating silver eel within one region (Killough) of the RBD in 2017 (AFBI (2020) Hillsborough Lake and Hillsborough Castle Ornamental Water Bodies Fish Survey. Report to DAERA).

3.2. Anticipated effect of UK measures on silver eel escapement biomass

With the exception of the large-scale stocking in the Neagh Bann RBD, it is not yet possible to predict when these measures will achieve the required additional silver eel production across the UK. Investment in scientific research continues alongside implementation of management measures to improve our understanding of the situation. However, the timing of the recovery of the eel stock in the UK depends in part on the recovery of the international stock as a whole to provide increased eel recruitment to UK waters, and this trend cannot be predicted at this point in time.

3.3. Planned measures not implemented

There are no specific measures planned in the original EMPs that have not yet been implemented. However, some of the generic measures (e.g. installation of eel passes and screening of water intakes) are ongoing.

3.4. Difficulties encountered in the implementation of the plan

England and Wales

The main difficulties encountered in implementing the measures in England and Wales were those of:

1) Identifying the owner or person responsible for some in-river obstructions. Under these circumstances it is difficult to obtain permission to resolve eel passage at that site, or to apply powers under the Eels (England and Wales) Regulations 2009;

2) Obtaining the necessary resources to improve access to suitable habitat or to prevent entrainment, and to monitor the effectiveness of the measures delivered;

3) The costs of eel screening installations at intake structures can be prohibitive for some operators (up to several million £) and are subject to cost benefit analysis constraints. The Environment Agency spent several years working with operators to agree how to address such affordability issues. A new regulatory approach was implemented in May 2021. Implementation of the Eels (England & Wales) Regulations is largely via scheduled maintenance and/or capital investment programmes. This should mean that delivery of eel measures is more cost effective for operators, but it will mean a slower delivery rate for these improvements.

Scotland

None.

Northern Ireland

The stocking target for the Neagh/Bann RBD was not achieved in any of the reported years because of a disconnect in timing of supply with demand, costs and issues associated with EU exit.

4. References

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Annex A: Methods and data used in the 2020–2022 assessment of England and Wales

A1. Introduction

The assessment approach can be summarised as follows: the best achievable present-day silver eel escapement in the absence of human impacts (B_{best}) is estimated for index rivers within each River Basin District (RBD), using an eel life history model to extrapolate yellow eel density data from surveys across a river basin to whole river estimates of yellow eel numbers at length class, converted to numbers of silver eel at length class and then to silver eel biomass using a length-weight regression, all using the Scenario-based Model of Eel Production II (SMEP II) (Aprahamian *et al.*, 2007; Walker *et al.*, 2013). Silver eel biomass for the river habitat is converted to a mean silver eel production rate across the wetted area of modelled river (i.e. $kg \cdot ha^{-1}$). This production rate is then applied to all wetted area of rivers, lakes, estuaries and lagoons (where present) across the RBD, and finally losses that would have impacted the yellow eel population before they are electro-fished (e.g. glass eel (GE) fishing and barriers) are accounted for to estimate the B_{best} (Equation 1). The losses from various anthropogenic factors are estimated as silver eel equivalent biomass, and these are subtracted from B_{best} to estimate the present-day silver eel escapement to the sea ($B_{current}$) (Equation 2). The silver eel escapement from the historic pre-1980s reference period (B_0) was estimated using the same approach as for B_{best}, using historic survey data. The remainder of this section describes these analyses in greater detail.

$$B_{best} = Silver\ eel\ production\ rate + Barriers\ impact + GE\ catch \quad \text{Equation 1}$$

$$B_{current} = B_{best} - Fishing\ mortality - Non\text{-}fishing\ mortality + Gains\ from\ installed\ passes + Gains\ from\ stocking \quad \text{Equation 2}$$

A2. B_{best}

Estimates of B_{best} were made for the time periods as follows:

- 2005–2007
    - Used to assess emigrating biomass and mortality indicators for “Pre EMP”.
- 2008–2010
    - Used to assess emigrating biomass and mortality indicators for 2009 and 2010.
- 2011–2013

    - Used to assess emigrating biomass and mortality indicators for 2011, 2012 and 2013.
- 2014–2016
    - Used to assess emigrating biomass and mortality indicators for 2014, 2015 and 2016.
- 2017–2019
    - Used to assess emigrating biomass and mortality indicators for 2017, 2018 and 2019.
- 2020–2022
    - Used to assess emigrating biomass and mortality indicators for 2020, 2021 and 2022.

The assessments were based on yellow eel data stored on the National Fish Population Database (NFPD). Only quantitative density and biomass data were included and therefore the following survey types and data were excluded:

- Fishing methods:
    - Fyke netting
    - Fixed traps fishing
    - Portable traps fishing
    - Trapping
    - Dip netting
    - Gill netting
    - Kick sampling
    - Trawl netting
    - Timed surveys
- Where the fished area was less than 10 m^2.
- Where the biomass recorded was greater than 3000 g per 100 m^2.
- Where the length of eel recorded was <50 mm.

B_{best} was estimated for 24 rivers across the eleven RBDs (Table A1), for the 2024 report. Survey data from multiple index rivers were analysed for each RBD and extrapolated to the whole wetted area of the RBD, except for Humber and Dee RBDs where data from all river surveys within each RBD were analysed such that extrapolation was not necessary. The estimate of B_{best} was based on modelled yellow eel data from the rivers sampled in that RBD. Mean values for each reporting period are provided (Table A2).

Table A1. Silver eel production outputs from SMEP II (kg·ha^{-1}) for the rivers analysed for the periods 2005–2007, 2008–2010, 2011–2013, 2014–2016, 2017–2019 and 2020-2022. Please note that for Humber and Dee, only mean values are shown due to a large number of rivers included in the analysis.

RBD	River	2005–2007	2008–2010	2011–2013	2014–2016	2017–2019	2020-2022
Northumbria	Coquet	0.00	0.35	0.38	0.37	0.02	0.06
	Wear	1.41	6.96	0.77	0.86	1.67	0.71
Humber	Humber	0.57	0.79	1.14	0.41	0.36	0.1
Anglian	Great Ouse	1.91	0.43	0.63	0.56	0.49	No estimate
	Suffolk Stour	2.85	2.58	1.27	0.55	0.91	0.77
	Wensum	1.70	1.29	1.30	0.82	0.87	2.00
	Witham	4.27	4.41	2.88	0.89	0.88	No estimate
	Welland	5.37	4.36	5.28	7.18	0.96	No estimate
	Chelmer & Blackwater	11.4	4.26	3.48	1.23	0.85	0.41
	Nene	3.90	1.27	0.97	0.17	0.42	No estimate
Thames	Lee	3.60	1.18	1.94	1.35	1.08	No estimate
	Medway	0.99	1.80	1.21	0.12	2.29	0.16
	Thames	1.95	2.05	1.40	0.22	1.16	0.34
South East	Ouse	2.41	1.03	2.24	2.81	0.65	No estimate
	Itchen	6.17	12.63	7.83	6.36	5.56	4.12
	Test	3.60	12.35	6.14	4.32	1.63	3.50
South West	Dorset Stour	0.14	5.02	0.53	0.16	0.34	No estimate
	Exe	0.04	0.78	0.04	1.27	0.09	1.63
	Fowey	0.67	1.47	0.24	0.24	0.07	0.40
	Frome	4.54	8.46	3.18	1.86	1.1	No estimate
	Hampshire Avon	1.70	3.00	2.76	0.84	2.4	No estimate
	Otter	0.16	0.76	0.63	1.66	0.07	No estimate
	Parrett	0.04	0.21	1.08	0.02	No estimate	No estimate

	Plym	3.34	1.81	2.95	0.62	3.73	No estimate
	Tamar	0.11	0.35	0.06	0.24	0.3	0.2
	Taw	0.00	0.03	0.01	0.21	0.89	1.61
	Teign	0.01	0.05	0.11	0.15	0.07	No estimate
Severn	Severn	1.18	1.48	1.33	0.51	0.42	0.19
	Wye	0.07	0.28	0.58	0.23	0.18	No estimate
	Usk	0.09	4.42	1.49	2.51	0.25	1.87
Western Wales	Clwyd	No estimate	0.05	0.07	0.5	0.2	No estimate
	Teifi	No estimate	0.89	2.44	1.69	0.8	0.64
	Tywi	No estimate	0.97	0.03	1.02	0.05	No estimate
	Wnion	No estimate	1.15	2.04	1.91	0.87	No estimate
Dee	Dee	0.35	2.15	2.44	1.27	0.81	2.72
North West	Bela	No estimate	4.80	0.54	1.1	2.18	1.77
	Derwent	0.05	0.31	0.29	0.23	0.3	0.16
	Ellen	3.40	0.01	0.06	No estimate	0.02	No estimate
	Mersey	0.00	0.11	0.16	0.4	0.04	0.56
	Ribble	0.44	1.33	0.72	0.81	0.45	2.50
	Weaver	1.12	0.03	0.00	0.4	No estimate	No estimate
Solway Tweed	Border Esk	0.13	1.65	0.37	0.59	1.67	0.07
	Eden	0.10	0.20	0.09	0.46	0.29	0.28
	Tweed	No estimate	1.28	0.57	No estimate	No estimate	No estimate

Table A2. Mean silver eel production outputs from SMEP II (kg·ha^{-1}) for the periods 2005–2007, 2008–2010, 2011–2013, 2014–2016, 2017–2019 and 2020-2022 for each River Basin District. Note: with the exception of the Humber and Dee the estimates for each RBD are means of two to eleven index rivers. For Humber and for Dee all available survey data from the RBD were used to produce the mean values.

RBD	2005–2007	2008–2010	2011–2013	2014–2016	2017–2019	2020-2022
Northumbria	0.71	3.66	0.58	0.62	0.85	0.39
Humber	0.57	0.79	1.14	0.41	0.36	0.10
Anglian	4.49	2.66	2.26	1.63	0.77	1.06
Thames	2.18	1.68	1.52	0.56	1.51	0.25
South East	4.06	8.67	5.40	4.50	2.61	3.81
South West	0.98	1.99	1.05	0.66	0.91	0.96
Severn	0.45	2.06	1.13	1.08	0.28	1.03
Western Wales	No estimate	0.77	1.15	1.28	0.48	0.64
Dee	0.35	2.15	2.44	1.27	0.81	2.72
North West	1.00	1.10	0.30	0.59	0.60	1.25
Solway Tweed	0.12	1.04	0.34	0.53	0.98	0.18

A3. Anthropogenic mortality factors and $B_{current}$

The impacts of the anthropogenic (human-induced) mortality factors have been summarised according to four categories as follows:

1. Fishing mortality, relates to the catch of all life stages;
2. Entrainment and mortality at water intakes, includes mortality from pumping stations, critical surface water abstractions, power stations and hydropower facilities;
3. Habitat quantity and quality, relates to the impact of manmade obstructions (including tidal gates); and
4. Stocking, reflects the benefit of stocking and has been reported as a negative impact.

A3.1. Fishing mortality

Recreational catch

It has been illegal to kill eel caught by recreational fishing in England and Wales since 2009. Anyone who does catch an eel on rod-and-line (the only legal recreational instrument) must return it alive to the water from where it was taken. Therefore, it is assumed that there is no retained catch of eel.

Commercial catch

Catch data were available from the glass eel and from the yellow and silver eel (combined) fisheries. In 2009, legislation was introduced to improve the traceability of eel caught, such that there are now three sources of glass eel exploitation data:

1. Catch returns to the Agency – this provides a breakdown of catch by RBD but may underestimate the total catch (C_i);
2. The quantity of glass eel bought by the dealers from the fishermen (consignment notes) – this is the total amount of glass eel caught (C_t);
3. The quantity of glass eel exported from the UK or stocked within the UK – this is the total amount of glass eel caught minus mortality and weight loss post-capture.

For the period 2009 to 2022, the glass eel catch in RBDi was calculated as follows, using the nomenclature 1, 2 and 3 above:

$$G_i = C_t \cdot \left(\frac{C_i}{\sum_{c=0}^{n} C_i}\right)$$

For the years 2006–2008 the estimate of the total glass eel caught in each RBD was:

$$G_i = C_i \cdot \frac{\overline{C}_{t\,(2009-2011)}}{\sum_{c=0}^{n} \overline{C}_{i(2009-2011)}}$$

For 2005 and the early 1980s (Pre 1980 in Table A3) the estimates for both glass and yellow and silver eel (combined) were derived from the import export figures published in the country report to ICES (ICES, 2014: Table 2 [page 830] and Table 26 [page 855], respectively). The partition of the catch to individual RBDs was based on the split in the total catch based on the mean proportions between 2005 and 2013.

The catch for the Solway Tweed RBD prior to the ban on eel fishing in Scotland was assumed to be 10% of the total Scottish catch as the Scottish part of the Solway Tweed represents ≈10% of the freshwater habitat of Scotland.

Yellow and glass eel catches were converted to silver eel equivalents, as follows:

The biomass of yellow eel caught was considered to be the equivalent of the potential silver eel escapement as the instantaneous mortality rate of 0.14 yr^{-1} (Dekker, 2000) approximated to the instantaneous growth rate of 0.2 yr^{-1} (95% CI ± 0.03) (Aprahamian, 1986).

For the glass eel catch, 1 kg of glass eel was considered equivalent to 59.4 kg of silver eel. This was determined assuming:

1. a settlement instantaneous mortality of 0.00915 day^{-1}, (95% CI ± 0.00149 day^{-1}) based on an extrapolation from the study of Bisgaard and Pederson (1991) to a glass eel of 80 mm;
2. a settlement period of 50 days (Briand, 2009) assuming a water temperature of 9 ^{o}C;
3. an annual instantaneous mortality following settlement of 0.14 yr^{-1} (Dekker, 2000);
4. a 50:50 sex ratio; and
5. males maturing at 11.9 (95% CI ± 0.6) (@ 89.9 g [95% CI ± 3.7 g]) and females at 17.8 (95% CI ± 0.8) years (@ 568.9 g [95% CI ± 57.1 g]) (Aprahamian, 1988).

Thus, the losses due to commercial fishing were estimated with the following formula:

$$Banthro_{fi} = \sum_{g=0}^{n} G_i \,.\, 59.4 + \sum_{y=0}^{n} Y_i + \sum_{s=0}^{n} S_i$$

Where:

- $Banthro_{fi}$ is the biomass (kg) of eel in terms of silver eel equivalents that is estimated would be produced in RBD_i if no fishing was present;
- G_i is the biomass (kg) of glass eel caught in fishery g in RBD_i (Table A3);
- Y_i is the biomass (kg) of yellow eel caught in fishery y in RBD_i (Table A4);
- S_i is the biomass (kg) of silver eel caught in fishery s in RBD_i (Table A5).

Table A3. Glass eel catch (kg) by River Basin District (RBD), including the information from the dealers and catch returns to the Environment Agency. NP = not pertinent (no fishery authorised in that year).

RBD	Pre-1980	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Northumbria	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
Humber	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
Anglian	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
Thames	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
South East	0	0	0	0	0	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
South West	17919	3225	722	999	521	282	1079	2033	2161	4536	5624	1365	2069	1721	1898	2881	1813	0	501
Severn	24454	4055	944	1750	554	111	759	1460	1586	3947	6010	1295	2084	1705	2558	3835	1776	66	879
Western Wales	1998	457	55	39	6	0	2	4	0	34	33	0	39	10	27	28	125	NP	NP
Dee	795	202	8	9	3	1	7	21	23	22	0	17	5	10	60	50	0	0	0
North West	4827	860	174	299	137	28	43	123	49	119	133	123	83	84	116	156	48	NP	NP
Solway Tweed	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP

Table A4. Yellow eel catch (kg) by River Basin District (RBD) since 2008. Figures for earlier years can be found in previous EMP reports. Note: due to an apparent anomaly with reported catch figures for Humber in 2018, a three-year mean catch was applied for biomass and mortality calculations (in brackets). NP = not pertinent (no fishery authorised in that year).

RBD	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Northumbria	0	45	60	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
Humber	1429	411	3033	4857	3267	3865	151	1678	155	1542 (2468)	4838 (2468)	1023 (2468)	195	2369	155
Anglian	9903	6616	10708	16478	15335	9351	11000	8082	12273	6129	11796	7432	2270	9099	8546
Thames	5548	4745	5655	6082	1815	3991	3222	2696	2473	2264	1971	1682	29	12	37
South East	602	7029	1432	1879	2116	286	284	12143	825	364	216	200	225	186	0
South West	6626	2546	2722	3792	5966	8688	10117	5642	10261	11168	13347	13014	12410	10611	12543
Severn	27	0	150	350	0	0	NP	NP	NP	NP	NP	NP	NP	NP	NP
Western Wales	118	22	345	252	647	100	0	0	1345	0	0	0	0	NP	NP
Dee	642	70	53	1082	478	152	415	74	73	333	123	608	0	245	200
North West	474	114	150	1477	2972	669	87	93	187	326	154	247	715	424	32
Solway Tweed	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP

Table A5. Silver eel catch (kg) by River Basin District (RBD) since 2008. Figures for earlier years can be found in previous EMP reports. Note: due to an apparent anomaly with reported catch figures for Humber in 2018, a 3-year mean catch was applied for biomass and mortality calculations (in brackets). NP = not pertinent (no fishery authorised in that year).

RBD	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Northumbria	90	10	0	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP
Humber	865	110	199	257	1627	259	81	742	49	22 (393)	1115 (393)	41 (393)	280	876	20
Anglian	1974	592	739	2006	2980	2486	1483	3759	3664	2109	2258	2808	1621	3013	3362
Thames	404	119	67	513	200	308	384	202	152	14	134	4	7	0	9
South East	1650	3198	823	694	650	1991	754	895	252	30	79	60	42	23	28
South West	552	303	172	68	533	950	1167	119	947	1117	1342	1459	1928	2044	1917
Severn	117	1224	100	380	0	0	NP	NP	NP	NP	NP	NP	NP	NP	NP
Western Wales	10	43	9	9	0	0	0	0	150	0	0	0	0	NP	NP
Dee	15	14	15	119	0	31	30	31	24	21	19	173	0	85	70
North West	263	80	72	270	462	105	28	56	33	254	220	277	304	219	30
Solway Tweed	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP	NP

A3.2. Entrainment

Pumping stations

In 2015, 336 of the 946 pumping stations in England and Wales were identified as having the greatest potential to impact eel, based on the distance from head of tide and the predicted prevalence of eel. The predicted prevalence was estimated using a non-parametric geostatistical model (Wyatt, 2005; Wyatt, *et al*., 2007) that related the prevalence of eel to environmental variables (distance from the tidal limit and altitude), and geographic location. The model was used to predict the expected prevalence of eel for a given river type under reference conditions, the pressure variables being set to zero (WFD-UKTAG, 2008a).

To estimate the impact across the RBD, it was assumed that all the area upstream of each of the 336 most impacting pumping stations was lost to eel production (Table A6). The total annual loss in terms of silver eel biomass for RBD (i) was estimated as follows:

$$Banthro_{ji} = \sum_{j=0}^{n} (Bsmep_i \,.\, A_{ji})$$

Where:

- $Banthro_{ji}$ is the biomass (kg) of silver eel that is estimated would be produced in catchment j in RBD$_i$ if no pumping station was present;
- $Bsmep_i$ is the silver eel production from RBD$_i$ (kg·ha^{-1}) estimated by SMEP II;
- A_{ji} is the wetted area (ha) upstream of the pumping station in catchment j in RBD$_i$.

Table A6. Area of habitat lost to eel production (ha) from those pumping stations with the greatest potential to impact eel ("high priority pumping stations")

RBD	Area of habitat lost to eel production (ha) 2017-2019	Area of habitat lost to eel production (ha) 2020-2022
Northumbria	5	5
Humber	3897	3299
Anglian	5234	5194
Thames	28	19
South East	797	797
South West	1621	1621
Severn	119	114
Western Wales	0	0
Dee	0	0
North West	366	366
Solway Tweed	0	0

In the period 2020-2022, eel measures were installed at 46 of the 336 key pumping stations, reducing the estimated area of habitat lost to eel production by 652 ha compared to the 2021 report (Table A6). It should be noted that eel measures were also implemented at a number of pumping stations not used in the original impact calculation.

Surface water abstractions

In 2015, it was determined that surface water was abstracted at 23,106 sites in England and Wales. A total of 530 sites were identified as posing the greatest threat to eel using the following criteria: distance from head of tide, size of the abstraction, predicted presence of eel, and the sensitivity of the water body to abstraction (WFD-UKTAG, 2008b). These identifications were also quality assured by consultation with local experts.

Information on eel entrainment and mortality was available from 10 surface water abstraction sites (APEM, 2007; APEM, 2010; Frear and Axford, 1991). The annual numbers of eel entrained at these 10 sites ranged from zero to 3261 with a mean of 613.8 (95% CI ± 613.8) eel per year. The mean age of those eel was assumed to be two years, which equates to about 150 mm total length. The equivalent in terms of silver eel biomass (calculated as above) was estimated to be 0.03 kg per entrained eel, equating to 19.2 (95% CI ± 19.2) kg·yr^{-1} entrained per abstraction.

The total annual loss in terms of silver eel biomass for RBD_i (Table A7) was therefore estimated as follows:

$$Banthro_{ki} = \sum_{k=0}^{n} K_i \,.\, 19.2$$

Where:

- $Banthro_{ki}$ is the biomass (kg) of silver eel that is estimated that would be produced in RBD_i if no surface water abstraction was present;
- K is the number of surface water abstractions in RBD_i.

In the 2020-2022 period, 33 more eel screens were installed at high priority/critical intakes, resulting in a reduction in the ongoing impact, with 450 critical abstractions remaining (Table A7).

Table A7. Number of critical abstractions and estimated loss of emigrant biomass.

RBD	No. High Priority abstractions still impacting in 2019	Estimated biomass of silver eel entrained ($kg \cdot yr^{-1}$) in 2019	No. High Priority abstractions still impacting in 2022	Estimated biomass of silver eel entrained ($kg \cdot yr^{-1}$) in 2022
Northumbria	11	211	8	154
Humber	15	288	14	269
Anglian	131	2515	120	2304
Thames	39	749	32	614
South East	37	710	32	614
South West	80	1536	75	1440
Severn	16	307	15	288
Western Wales	12	230	12	230
Dee	11	211	11	211
North West	124	2381	124	2381
Solway Tweed	7	134	7	134

Cooling water intakes at Power Stations

Information on eel impingement and/or entrainment at cooling water intakes of power stations was available from five sites. At three sites, only impingement data were available and to account for the quantity of eel that passed through the screens, the catch was raised by x300 for glass eel and x4.3 for yellow eel (APEM, 2012). There was no correction factor applied for silver eel. For the two sites where no size information was available, it was assumed that those eel caught between 1 February and 30 April were glass eel, with yellow eel being caught at all other times. A survival rate of 36% was assumed for glass eel and 75% for yellow eel entrained by the power station (APEM, 2012; Jacobs, 2008). The conversion of glass eel and yellow eel entrainments into silver eel equivalents was as described for the commercial catch (above). The estimated annual biomass of silver eel equivalents entrained by a power station was 697.6 $kg \cdot yr^{-1}$ (95% CI ± 724.2 $kg \cdot yr^{-1}$).

The total annual loss in terms of silver eel biomass for RBD_i (Table A8) was estimated as follows:

$$Banthro_{li} = \sum_{l=0}^{n} L_i \,.\, 697.6$$

Where:

- $Banthro_{li}$ is the biomass (kg) of silver eel that is estimated would be produced in RBD_i if no power station was present;.
- L is the number of power stations in RBD_i.

In 2019, it was estimated that there were 44 power stations across England and Wales where cooling water intakes were impacting eels. By 2022, this number had fallen by five (four stations were decommissioned and one was changed to open cycle gas turbine, no longer abstracting water) that no longer pose an entrainment/mortality risk to eels. The loss of emigrant biomass has therefore reduced accordingly (Table A8).

Table A8. Number of power stations and estimated loss of emigrant biomass.

RBD	Number of Power Stations impacting in 2019	Estimated biomass of silver eel entrained ($kg \cdot yr^{-1}$) in 2019	Number of Power Stations impacting in 2022	Estimated biomass of silver eel entrained ($kg \cdot yr^{-1}$) in 2022
Northumbria	3	2092.8	3	2092.8
Humber	15	10464	13	9068.8
Anglian	2	1395.2	2	1395.2
Thames	8	5580.8	8	5580.8
South East	4	2790.4	4	2790.4
South West	2	1395.2	2	1395.2
Severn	1	697.6	1	697.6
Western Wales	1	697.6	0	0.0
Dee	1	697.6	1	697.6
North West	7	4883.2	5	3488.0.
Solway Tweed	0	0.0	0	0.0

In-river Hydropower facilities (turbines)

The impact of each in-river hydropower facility was estimated according to the B_{best} production ($kg \cdot ha^{-1}$) for the relevant RBD, the area of habitat upstream, the presence or absence of screens (preventing eel entrainment) and the type of turbine.

For those sites with screens (α), the proportion of eel entering the turbine(s) was assumed to be: zero, if the spacing between the bars/mesh was <15 mm; 50%, if the spacing was between 16–29 mm; and 100%, if >30 mm: 27.6% of hydropower schemes (excluding Archimedes screws) are adequately screened to prevent the entrainment of adult eel (i.e., spacing <15 mm).

The estimates of turbine mortality (β) were taken from ICES (2011), and these were: Archimedes screw = 0%, Francis Turbine = 32%, and Kaplan turbine = 38%. All hydropower facilities have some form of bypass channel that provides an alternative route for fish around the turbine. On this basis, it has been assumed that ≈ 50% of the silver eel produced upstream of a turbine will become entrained therein whereas the other 50% use the bypass.

On those river systems where there is more than one hydropower facility, the loss of production from the upstream turbine(s) has been accounted for in estimating the potential impact of turbines further downstream, i.e. the cumulative impact of all turbines has been calculated (Table A9).

$$Banthro_{hi} = \sum_{h=0}^{n} ((Bsmep_i \,.\, A_{hi}) - ((Bsmep_i \,.\, A_{h_u i})\alpha_{h_u}\beta_{h_u}))\alpha_h\beta_h$$

Where:

- $Banthro_{hi}$ is the biomass (kg) of silver eel that is estimated would be produced in RBD_i if no hydropower facilities (h) were present;
- $Bsmep_i$ is the silver eel production from RBD_i ($kg \cdot ha^{-1}$) estimated by SMEP II;
- A_{hi} is the wetted area (ha) upstream of the hydropower station (h) in catchment i in RBD_i;
- $h_u$represents the hydro scheme upstream of hydropower station h.

Table A9. Estimated loss of emigrant biomass due to hydropower stations (kg·yr^{-1}). Note the estimates are based on a list of licensed hydropower stations in existence in 2014.

RBD	Estimated loss of emigrant biomass (kg·yr^{-1}) 2005–2007	Estimated loss of emigrant biomass (kg·yr^{-1}) 2008–2010	Estimated loss of emigrant biomass (kg·yr^{-1}) 2011–2013	Estimated loss of emigrant biomass (kg·yr^{-1}) 2014–2016	Estimated loss of emigrant biomass (kg·yr^{-1}) 2017–2019	Estimated loss of emigrant biomass (kg·yr^{-1}) 2020-2022
Northumbria	10	53	8	9	9	9
Humber	575	592	619	507	507	507
Anglian	0	0	0	0	0	0
Thames	3	2	2	2	2	2
South East	63	135	84	70	70	70
South West	862	867	863	195	195	195
Severn	8	27	17	16	16	16
Western Wales	33	37	56	62	62	62
Dee	2	10	12	6	6	6
North West	79	84	47	54	54	54
Solway Tweed	0	1	0	1	1	1

A3.3. Habitat loss

Barriers

There are about 19,000 potential barriers (partial and complete barriers) to eel migration across England and Wales. The impact of barriers (including tidal gates) was estimated using a general linear model derived from eel data in 27 rivers from 2008 to 2013 (r^2 = 0.196):

$$\gamma_b = e^{(-2.6545-(0.302Log_e(\delta+1))-(0.0401Log_e(\varepsilon+1))-(55.3Log_e(\zeta+1))-(0.2906Log_e(\eta+6))+(1.7152Log_e\theta))-1}$$

Where:

- γ_b is density (# 100 m^{-2}) of eel in the presence of barriers downstream;
- δ is distance (m) upstream of tidal limit;
- ε is the number of barriers downstream of the site to the tidal limit;
- ζ is the gradient (m·m^{-1}) to the site;

- η is the longitude (ºEast) of the site;
- θ is latitude (ºNorth) of the site.

The anthropogenic effect of barriers was estimated by setting ε in the above equation to zero and comparing the ratio of density as estimated from the above equation in the presence and absence of barriers. The mean of all these site ratios was applied to the RBD as a whole, as follows:

$$Banthro_{bi} = \frac{Bcurrent_i}{\overline{\gamma_{bi}/\gamma_i}} - Bcurrent_i$$

Where:

- $Banthro_{bi}$ is the biomass (kg) of eel in terms of silver eel equivalents that is estimated would be produced in RBD_i if no barriers were present;
- $\overline{\gamma_{bi}/\gamma_i}$ is the mean proportion of eel density in the presence of barriers against no barriers present (ε = 0) at sites within RBD_i.

In the 2020-2022 reporting period, a total of 53 eel passes were installed at eel barriers. We can adjust the estimated impact of barriers accordingly by calculating the amount of upstream habitat made available by each eel pass and its corresponding production in terms of silver eel biomass. Hence our estimate of the impact of barriers is reduced.

A3.4. Stocking

Stocking was undertaken using glass eel (Table A10) and converted into silver eel equivalents as described for commercial catch (above). The impact of stocking was considered in the estimation of total anthropogenic mortality when calculating $B_{current}$ but was not included in the estimation of mortality rates (see below A3.5).

Table A10. Amount of glass eel stocked (kg), by RBD since 2012. Figures for earlier years can be found in previous EMP reports. Note that all glass eel all originated in the RBDs of England and Wales.

RBD	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Northumbria	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Humber	10.00	3.00	3.80	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Anglian	1.50	9.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Thames	1.20	2.00	14.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
South East	0.00	7.00	7.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
South West	0.19	12.80	8.70	0.33	0.55	0.00	0.00	0.00	0.00	0.00	0.00
Severn	9.75	21.10	21.50	17.00	17.00	17.00	17.00	0.40	20.00	0.00	20.00
Western Wales	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dee	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
North West	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00
Solway Tweed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

A3.5. Estimation of mortality rates

The sum of lifetime anthropogenic mortalities (∑A) is calculated as:

$$\sum A = -\ln\left(\frac{Bcurrent}{Bbest}\right)$$

Fisheries (∑F) and non-fisheries mortalities (∑H) were assumed to operate during the whole continental life and at the same time, and therefore estimated as:

$$\sum \mathrm{F} = \frac{\mathrm{Ct}}{\mathrm{Ct} + Ht} * \sum \mathrm{A}$$

$$\sum \mathrm{H} = \frac{Ht}{\mathrm{Ct} + Ht} * \sum \mathrm{A}$$

Where:

- C_t is the total fisheries catch in silver eel equivalents;
- H_t is total non-fisheries loss of silver eel equivalents.

Stocking was not considered in mortality calculations to ensure anthropogenic mortalities are not inadvertently masked by large restocking programmes. Therefore, stocked eel (as silver eel equivalents) were subtracted from $B_{current}$ prior to estimating lifetime anthropogenic

mortalities and from total non-fisheries loss prior to calculating fisheries and non-fisheries mortalities.

A4. Estimation of B_0

The target level of escapement (40% of B_0) was determined using two out of three options specified under Article 2(5) of EC Regulation 1100/2007 (EC, 2007). Where available, historic data were used to produce the B_0 target values for those river basins. These were then applied to river systems with limited or missing data, while considering ecology and hydrography of those systems. The approach is described in more detail below.

There are few historic eel surveys available across England and Wales that provide the density, length frequency and sex ratio data necessary to apply the SMEP II approach to estimate RBD-specific B_0. The rivers and survey years available are presented in Table A11.

Complete data are only available from the Severn (1983), Dee (1984) and Thames (1992–1994). These data were applied directly in the SMEP II model to estimate historic potential production (~B_{best}), applying the same approach as described for estimating current B_{best}, above.

As no length data were recorded for the Anglian rivers Stour and Chelmer, the mean eel length for a site was estimated from other rivers as follows:

*Mean total length (mm) = 281.0 (±15.54) + 0.9879 (±0.245) * Distance from tidal limit (km)*
$P<0.001$; $r^2 = 0.23$

The length distribution was estimated using a random number generator based on the mean length (calculated above), a standard deviation (SD) of 102 (the mean SD of all sites where length had been recorded), and assuming a binomial distribution.

As only the mean length and SD were available for the South West rivers (Frome, Fowey Teign, Axe, Otter and Plym), the length distribution was estimated using a random number generator, assuming a binomial distribution.

Table A11. Estimates of silver eel potential escapement (kg·ha^{-1}) for various rivers between 1979 and 1994.

RBD	River (Year)	Potential escapement (kg·ha^{-1})
Anglian	Suffolk Stour (1983)	0.73
	Chelmer (1986)	0.88
Thames	Thames (1992–1994)	2.35
South West	Frome 1990	82.54
	Fowey 1981 &1983	3.06
	Teign 1979	2.20
	Axe 1979	56.78
	Otter 1978	27.24
	Plym 1982	7.17
Severn	Severn (1983)	6.84
Dee	Dee (1984)	29.89

Where potential escapement estimates were available for two or more rivers in the same RBD, the river-specific estimates were combined to provide a mean estimate for the RBD. For the South West RBD, the mean escapement was estimated based on the assumption that 14% of the production is derived from chalk streams (River Frome) and 86% from rain fed rivers (i.e. Fowey Teign, Axe, Otter, Plym) as follows:

*SW RBD (kg·ha^{-1}) = ((Frome*0.138876) + ((Fowey +Teign + Axe + Otter + Plym)/5) * (1–0.138876))*

In the Anglian RBD, the two rivers were given equal weighting because the rivers are similar in character.

Where no historic data were available for any rivers within the RBD, the following assumptions have been made:

- The east coast RBDs (Northumbria, Humber and South East) follow a similar trajectory to that of the Anglian, where current escapement (B_{best}) is greater than "historic" and therefore current production has been taken as B_o.
- The West Wales and North West RBDs were extrapolated from the South West (excluding chalk rivers), Severn and Dee estimates, weighted according to wetted areas.
- The Solway-Tweed estimate was extrapolated based on South West (excluding chalk rivers), Severn and Dee weighted according to area and Tweed production for 2008–2010 was based on the rational that current production on the east coast is higher than historic.
- These potential escapement estimates were then corrected for the impact of barriers (as above) to give an estimate of B_o for each RBD (Table A12).

Table A12. Estimates of B_o ($kg \cdot ha^{-1}$) for River Basin Districts in England and Wales and for the cross border Solway Tweed RBD.

River Basin District	B_o (with barriers; ($kg \cdot ha^{-1}$)	B_o (with no barriers; $kg \cdot ha^{-1}$)	Comment
Northumbria	3.66	5.16	Based on current estimate and the rationale from Anglian that current is higher than historic and as 2008–2010 (3.25 $kg \cdot ha^{-1}$) > 2011–2013 (0.63 $kg \cdot ha^{-1}$)
Humber	1.14	2.38	Based on current estimate and the rationale from Anglian that current is higher than historic and as 2011–2013 (1.14 $kg \cdot ha^{-1}$) > 2008–2010 (0.79 $kg \cdot ha^{-1}$)
Anglian	4.49	6.27	Based on current estimate (2005–2007) being higher than historic (0.81 $kg \cdot ha^{-1}$).
Thames	2.35	5.88	Thames (1992–1994)
South East	8.67	10.60	Based on current estimate and the rationale from Anglian that current is higher than historic and as 2008–2010 (8.67 $kg \cdot ha^{-1}$) > 2011–2013 (5.40 $kg \cdot ha^{-1}$)
South West	28.07	37.03	Pristine production based on 1979–1990 data (28.07 $kg \cdot ha^{-1}$) determined using SMEP II (assumes:14% production from chalk rivers of 82.5 $kg \cdot ha^{-1}$, the remainder from rain fed rivers at 19.3 $kg \cdot ha^{-1}$
Severn	6.84	11.98	Severn 1983
Western Wales	13.98	16.18	Pristine production estimated at 13.98 $kg \cdot ha^{-1}$ based on South West (excluding chalk rivers), Severn and Dee weighted according to area = ((19.29*31050) + (6.84*54542) + (29.89*14129)) / 99721
Dee	29.89	45.02	Dee 1984
North West	13.98	18.50	Pristine production estimated at 13.98 $kg \cdot ha^{-1}$ based on South West (excluding chalk rivers), Severn and Dee weighted according to area = ((19.29*31050) + (6.84*54542) + (29.89*14129)) / 99721
Solway Tweed	13.01	16.84	Based on South West (excluding chalk rivers), Severn and Dee weighted according to area and Tweed production for 2008–2010 based on the rational that current production on the east coast is higher than historic. Assumed 13.98 $kg \cdot ha^{-1}$ for Solway and 1.28 $kg \cdot ha^{-1}$ for Tweed (2008–2010)

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Annex B: Methods and data used in the 2020–2022 assessments of Scotland RBD

B1. Introduction

Eel fisheries in Scotland were banned (without licence) in 2009 as the principal management measure of the EMP for Scotland River Basin District (RBD). Prior to 2009, fisheries were not regulated, so only crude estimates of the scale of the fishery are available (Anon., 2010). Stock assessment methods for the RBD are therefore based on scientific estimates of upstream and downstream counts of eel at traps on three rivers. The estimates of B_0, $B_{current}$ and B_{best} rely on the extrapolation of data from small study areas to the whole RBD, with the inherent possibility of bias. To derive an estimate of current production and anthropogenic mortality for the RBD from the available data has required a number of assumptions; these have tended to be precautionary in nature (i.e. likely to underestimate current production and overestimate current anthropogenic mortality (see Anon., 2010 for details).

From 2013, in keeping with methods used in England and Wales, Scotland adopted the inclusion of estimates of silver eel production for transitional waters based on the simplistic assumption that this is equivalent to silver eel production in the lower lying rivers and lochs of Scotland. Pristine production for transitional waters is assumed to be equivalent to pristine production in Scottish freshwaters during the reference period. For this reason, the inclusion of transitional waters has a relatively small effect (as a percentage of pristine output) on modelled silver eel output. However, because anthropogenic mortality is assumed to be zero in transitional waters, as there are no fisheries, the inclusion of transitional waters leads to a substantial reduction in the estimate of the value of $\sum A$ for the Scotland RBD. All estimates in the current report have been back-calculated to include production from transitional waters, and thus do not match equivalent estimates in the 2012 report to the EU.

B2. B_{best}

Current eel production in Scottish waters is assumed to be limited only by recruitment and barriers to productive habitat. Accordingly, B_{best} is estimated in the same way as $B_{current}$, but including potential production from the habitat area currently assumed to be lost to production due to manmade barriers (including hydropower). This amounts to 42,670 ha of potential eel habitat of which: 31,545 ha are in the lower altitude band, 8,725 ha in the middle band, and 2,400 ha are in the upper band. It is worth noting that the reported area of habitat above manmade barriers is artificially increased, by an unknown extent, above the natural condition, due to the impoundment of waters above dams; accordingly, B_{best} is over-estimated by this method, which in turn leads to an overestimate of mortality due to manmade barriers and hydropower facilities.

B3. Anthropogenic mortality factors and $B_{current}$

The impact of manmade barriers on eel production was estimated in the most conservative way possible: by assuming that all barriers were total and acted to remove all production upstream of the barrier without increasing production downstream (i.e. an assumption that downstream habitat is completely saturated). Hydropower facilities were treated in the same way, even where fish passes allow eel access above the turbines: in this case the conservative assumption is that silver eel mortality moving downstream through the turbines is 100%. Thus, three assumptions are made that overestimate the impact of barriers on eel production: 1) all identified barriers completely exclude eel; 2) all hydropower sites cause 100% mortality of silver eel passing through them; 3) the wetted area of Scotland RBD is 100% saturated with eels. Thus, any wetted areas above hydropower facilities, or other manmade barriers, were removed from the productive area when estimating current production, and the production lost as a consequence was regarded as anthropogenic mortality ($\sum A$), with the separate impacts of a) hydropower facilities and b) other manmade barriers to eel estimated according the area of production lost to each (5 574 ha lost to hydropower, and 37 096 ha lost to other manmade barriers).

Current silver eel output ($B_{current}$) is estimated at three whole-river trap sites, with no known anthropogenic mortality, which measures production across three altitude bands: Shieldaig (0–240 m), Girnock (240–415 m), and Baddoch (> 415 m). The annual production from these three bands is then calculated from the production at the relevant site and the wetted area of habitat in that altitude band in the whole RBD. The total wetted area of freshwater for Scotland RBD, after excluding habitat above manmade barriers, is 111 069 ha of which: 97 684 ha lie in the 0–240 m band, 10 853 ha lie in the 240–415 m band, and 2 532 ha lie in the > 415 m band. Production in transitional waters (60 502 ha) is assumed to be equivalent to the lowest of the three altitudinal bands.

Estimates of silver eel production for pristine conditions (pre-1980) and the most recent years are shown in Table B1.

Table B1. Estimates of silver eel escapement in Scotland RBD in three altitude bands based on whole-river traps at three sites ($kg \cdot ha^{-1}$) pre-1980 and for the most recent years. Estimates for earlier years can be found in previous EMP reports.

Altitude band (m)	Pre-1980	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
0–240	1.18	0.51	0.74	2.24	1.34	1.15	1.12	1.02	0.87	0.99	1.79	0.67
240–415	1.18	0.94	0.54	0.28	0.43	0.59	1.51	0.77	0.61	0.63	1.32	0.54
>415	1.18	0.54	0.42	0.80	0.10	0.55	0.55	0.72	0.20	0.11	0.59	0.53

For the period prior to the introduction of the EMP (and the cessation of the fishery), additional mortality estimates due to the fishery were based on available estimates of the size of the fishery in 2003, yellow eel catches were scaled to silver eel equivalents after Aprahamian (1986).

B4. Estimation of B_0

The pristine production of Scottish waters was estimated in three ways: based on historical silver eel production at a single Scottish site (Girnock Burn) in the period from 1967–1981; by reference to the historical production at a similar site (Burrishoole, Ireland, 1971–79; ICES, 2008) and by reference to an Irish model (ICES, 2008) of five catchments accounting for catchment geology. All three methods gave similar estimates of silver eel production in $kg \cdot ha^{-1}$, and the mean of the three estimates was set as Scotland RBD's pristine production. This estimate of production was then applied to the wetted area of habitat in Scotland, estimated by GIS methods. Areas above natural barriers to eel migration were excluded from the pristine productive wetted area, but areas above manmade barriers (of any era of construction) were included in the pristine productive area. These methods are described in detail in the Scotland RBD EMP (Anon., 2010). Since production of the EMP however, the estimate of pristine production using Girnock Burn data was adjusted to account for a proportion of eel bypassing the trap in spate conditions, as it also was for the 2012 report to the EU. This led to a slight increase in estimated pristine production (averaged from the three methods) to 1.18 $kg \cdot ha^{-1}$.

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Annex C: Methods and data used in the 2021–2023 assessments of Northern Ireland RBDs

Neagh Bann RBD

For the only RBD in Northern Ireland with a fishery, the Neagh Bann RBD, the estimate of pristine escapement (B_0) was determined using historic data including catch and sex ratio, input-output regression analysis and from known productivity of eel growing areas (Section 11.4 of Neagh Bann EMP). Using these three methods pointed to a potential natural output in the range of 400 to perhaps 600 tonnes per annum given historical high natural glass eel supplies. This range would estimate the required 40% level at around 160 t to 240 t, with the midpoint set at 200 t.

In Northern Ireland, the monitoring of silver eel migration and subsequent estimations of silver eel escapement ($B_{current}$) from the Neagh Bann RBD are carried out by direct measurement (section 11.1 of the Neagh Bann EMP). Given the geography of the RBD, in particular the single outflow point of Lough Neagh via the Lower River Bann at Toome, it was possible to initiate an annual mark-recapture programme in 2003, with the objective of estimating escapement of silver eel from Lough Neagh based on the non-recaptured proportion of those tagged silver eel taken back upstream and released. This work was further enhanced and corroborated by implementing a hydro-acoustic tracking study (a not foreseen but implemented measure) in 2011. To date, 14 258 eels have been tagged with Floy™ Tags since 2003 and recaptures recorded at both silver eel sites in the RBD.

In 2018, the calculation for estimated escapement has been amended and further improved by the development of a model combining:

- daily river flow metrics with
- daily silver eel catch,
- against which daily tag recaptures are assessed.

This method has been used to hindcast and revise the calculations for escapement from 2009. Specific details of this mark recapture escapement assessment are outlined in Section 11.2 of the Neagh/Bann EMP (Anon., 2010) and in Aprahamian and Evans *et al.* (2021).

NOTE: See section 1.4 of current EMP Review in relation to revisions in assessment calculations and reversion to pre-2018 method following analyses showing a 3-5% variability between calculations and accessibility issues to refined data.

North Eastern RBD

The estimate of pristine escapement from the North Eastern RBD was calculated with reference to the ecology and hydrology of similar systems (as described in Section 2.4.1 of the North Eastern EMP). Historic escapement was unknown and not monitored because there are no fisheries in this RBD, but all rivers and upland lakes suitable for eel have been assessed as having no or minimal barriers to migration. As such under adequate recruitment levels and an adherence to the management actions laid down in the North Eastern EMP, this RBD should reach or better the 40% target naturally. Data relating to eel population densities and age distribution have been gathered for assessment purposes and are now included within Biomass and Mortality estimates. A glass eel index site has been established and the direct assessment of silver eel migration conducted in 2017 by netting, though this RBD is frequently hampered by high river flows in small river channels affecting surveys.

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